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FINAL REPORT

“Study on waste related issues of newly listed POPs and candidate POPs”

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Expert Team to Support Waste Implementation

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1 Executive summary

Based on the background of the amended Stockholm Convention the intention of the study is in particular to provide the European Commission with

- a compilation and evaluation of existing data on the new POPs and candidate POPs
- an assessment of impacts of potential concentration limits under the POP regulation on waste flows and competing legislation
- justified proposal for low (LPCL) and maximum (MPCL) concentration limits for new POPs and candidate POPs.

Substance and waste flows of new and candidate POPs

C-PentaBDE

The most common use of C-PentaBDE in Europe was in flexible polyurethane (PUR) foams, accounting for approximately 95 % of the total use of C-PentaBDE. The treated PUR foams were in turn mainly used for the production of automotive and upholstery applications (e.g. automotive seating, head rests, sofas, mattresses etc.). Other applications and finished articles containing or reported to be possibly contaminated with C-PentaBDE are Polyvinylchloride (PVC), Epoxy Resins, Unsaturated Polyesters (UPE), Rubber, Paint/Lacquers, Textiles, Hydraulic oils and possibly other.

This study exclusively focuses on C-PentaBDE treated PUR foams and associated applications and finished articles (i.e. automotive and upholstery applications).

The automotive sector currently contributes with ~258.3 t/y C-PentaBDE and the upholstery sector with ~96.95 t/y to the total annual amount of C-PentaBDE. Therefore the overall discharge of C-PentaBDE from the investigated sources in Europe accounts to ~355.3 t/y with an overall distribution of the discharge of around 6 % as emissions and around 94 % (~334.6 t/y) to waste. This substance flow corresponds to around 5,872 t/y of C-PentaBDE contaminated waste from automotive and upholstery applications.

C-OctaBDE

The main historic use of C-OctaBDE was in Acrylonitrilebutadiene-styrene (ABS) polymers. Around 95 % of C-OctaBDE supplied in the EU was used in ABS. The ABS in turn was mainly used for housings/casings of Electrical and Electronic Equipment (EEE), typically office equipment and business machines. Other minor uses (5 %) were High Impact Polystyrene (HIPS), Polybutylene Terephthalate (PBT), Polyamide Polymers, with typical concentrations between 12-15 %. Other possible uses found in literature were nylon, low density polyethylene, polycarbonate, phenol-formaldehyde resins, unsaturated polyesters, adhesives and coatings.

The main focus of this study is on the use of C-OctaBDE in EEE, in particular on the ABS

housings/casings of office equipment and business machines.

Electrical and Electronic Equipment (EEE) has been identified as the only relevant source of C-OctaBDE (~131 t/y). The overall discharge of C-OctaBDE from the investigated source in Europe accounts for approximately 128 t/y, with an overall distribution of the discharge of around 2.3 % as emissions and 97.7 % to waste. This substance flow corresponds to approximately 732 t/y of C-OctaBDE contaminated WEEE plastics and is part of the plastic fraction of WEEE from the WEEE categories 3 and 4 amounting to approximately 105 kt/y waste.

PFOS

At present, PFOS is used in the metal plating industry, as content in hydraulic fluids in the aviation industry, in the photographic industry and the semiconductor industry.

Another relevant source of PFOS is sewage sludge. This pathway of PFOS is considered as one of the most important secondary sources of PFOS. As more and more industries stop or reduce the use of PFOS, however, sewage sludge accumulates not only PFOS which is currently used, but also PFOS which has been emitted in the past. Therefore, this sector has been investigated in detail.

The major waste stream from the different sources are Metal plating 102 kt/y (including 2.1 t/y PFOS), fire fighting foams 0.65 kt/y (including 6.5 t/y PFOS), Carpets 1940 kt/y (including 146 t/y PFOS) and Leather 2,380 kt/y (including 5.7 t/y PFOS).

The total waste amount is about 16.2 million t/y (including large amounts of sewage sludge with a low PFOS content) containing about 163 t PFOS.

PeCB

PeCB is not being intentionally produced or used anymore within the UNECE regions, including Europe. The most relevant source of PeCB production and emission is unintentional production and connected releases. As main components incineration and combustion processes of different wastes and materials have been identified, i.e. municipal solid waste incineration, hazardous waste incineration, coal combustion and domestic burning of fuels and mixed waste. Further, also waste water treatment has been considered as relevant.

The total affected waste amount is about 80 million t/y with typical contaminations in the low ppb range. The PeCB amount in waste sums up to about 308 kg/y.

SCCP

At the moment neither natural occurrence nor unintentional production of SCCP are known. All existing SCCP is therefore anthropogenic.

At the moment, the rubber industry is the main application of SCCP (1,254 t/y) followed by the sealants and adhesives sector (459 t/y) and by the paints and varnishes sector (337 t/y). The textile industry covers only a small fraction of the overall used amount of SCCP. Regarding the waste stream

all of the mentioned sections are included in a similar ratio as their current use. Also leathers which have been impregnated with SCCP in the past can enter the waste stream due to their long lifetime. The total SCCP containing waste amount (without sewage sludges) sums up to about 43 kt/y including about 2.1 kt of SCCP. The annual amount of sewage sludge (11.5 million t) includes about 40 t of SCCP.

HBCD

At present, the main uses of HBCD are in the polymer and textile industries. HBCD 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. The focus of this study is on EPS and XPS for the construction sector, EPS/XPS outside the construction sector, HIPS for EEE and on polymer dispersions for textile products.

These sources are currently responsible for approximately 1,196 t/y of HBCD in waste in the EU (EPS for construction ~68.82 t/y, XPS for construction ~75.8 t/y, EPS/XPS other than construction ~3.9 t/y, HIPS ~181 t/y and HBCD from textiles ~866.7 t/y. It is at this point, however, important to notice, that a significant amount of HBCD is each year incorporated into new flame retarded products in the EU (e.g. ~10,431 t/y in 2010), which will become waste in the future. The corresponding HBCD waste amount in 2010 is approximately 32,930 t/y from the investigated sources (~13,495 t/y EPS C&D waste, ~3.790 t/y XPS C&D waste, ~288 t/y of EPS/XPS other than C&D waste, ~4,524 t/y of HIPS waste and ~10,833 t/y of textile waste).

As already mentioned the waste figures are expected to significantly increase, due to huge amounts of HBCD incorporated into new EPS/XPS products for construction purposes and the relatively long life times of the products. The HBCD waste flow is expected to increase from ~32,931 tonnes in 2010 to more than 380,000 tonnes in 2040, reaching its maximum around 2050.

PCP

Pentachlorophenol (PCP) is investigated together with the substances sodium pentachlorophenoxide (NaPCP) and Pentachlorophenyl laureate (PCPL). In the past the PCP was used for many applications however, due to long product life times waste containing PCP can still enter the waste stream. The relevant sources of waste regarding PCP are nowadays only from the wood impregnation industry with 3 million t/y of waste including about 2,000 t of PCP and the textile industry with 9,600 t/y of waste including about 240 t/y of PCP. The waste from the wood industry is typically incinerated.

HCBD

At present there is no intentional use of HCBD in Europe since many years. HCBD is mainly formed as unintentional by-product during several industrial processes, but the available information basis is very scarce. However, production of tri- and tetrachloroethene and tetrachloromethane has been indicated as the most important source. Production of primary magnesium has been identified to belong to the historic uses by now, since its production within EU27 has stopped recently. Unintentional production and releases from the plastic industry may be relevant, but specific data is

not available. The same case has been identified for the sector of hazardous waste incineration and sewage sludge resulting from treatment of industrial waste water. Concluding, more information would be required to make realistic waste flow estimations.

PCN

PCN can be formed unintentionally during various thermal processes. As PCNs exhibit similar formation properties as PCDD/Fs, unintentional production during incineration processes such as municipal solid waste incineration, hospital waste incineration, domestic burning, different metal processing steps such as secondary copper production, secondary aluminium production, magnesium production as well as iron sintering and electrical arc furnace processes for iron production are of relevance. Further, the accumulation in sewage sludge from diffuse sources has been taken into account. The total waste amount is about 27 million t/y including about 3,200 kg of PCN with typical concentration in the low ppb range. Only in exceptional cases the PCN concentration amounts up to 4 ppm (slag from secondary copper production) or 5 ppm (fly ash from hospital waste incineration).

Measurement Methods

The report contains a compilation of analytical measurement methods for the detection of new and candidate POP concentrations in different relevant matrices. The methods are compiled and discussed separately for each substance. Measurement methods are available for all relevant substances that enable the detection of the new and candidate POPs at levels usually well below the proposed limit values. The compilation comprises an evaluation of available screening methods. Details on available measurement methods can be found in the corresponding chapter of the report.

Low (LPCL) and maximum (MPCL) POP concentration limits

The approach used in this report is based on the methodology developed for the determination of limit values for the initial 12 POPs, making use of upper and lower limitation criteria for determination of feasible LPCL.

Based on the evaluation of major substance and waste flows, given the expected trend for remaining stocks of PBDEs and PFOS in used products, and taking into consideration the impacts of potential limit values on the substance flows and potential conflicts with limitation criteria, the project team suggests a preferred option for LPCLs for new POPs and also makes a preferred proposal for candidate POPs ("LPCL 1" in Table 1-1). This option would correspond to a strict application of the precautionary principle in the light of the objective of a fast elimination of the new and candidate POPs. This option aims to eliminate POPs directly upon becoming waste and would prevent that the POPs would be diluted due to recycling in new products. It will be much more difficult to identify and eliminate the concerned substances at later stages when recycled products will become waste. This option in addition takes into consideration the high calorific value of the majority of concerned waste streams and their suitability for thermal treatment – including energy recovery – and adds an additional argument to prefer R1 operations instead of landfilling as predominant treatment option.

Furthermore Table 1-1 lists alternative proposals for a transitional period and/or related to specific

restrictions (“LPCL 2” in Table 1-1). The concentration levels proposed as LPCL 2 could be set for a restricted time frame in order to facilitate the enforcement of the LPCLs once the LPCLs will be put into force. Moreover, stricter limits for the agricultural use of sewage sludge containing relevant POPs could be laid down. This approach takes into account that to date the data background is sometimes limited and the contamination levels in several waste flows are based on modeling and justified assumptions. The corresponding concentration levels could probably be lowered and should be reviewed within an appropriate time frame of e.g. five years. The review should be done in the light of new relevant information. In order to improve the data background, analytical results on concentration levels in relevant waste streams should be generated in due time.

Table 1-1: Overview of preferred proposals (LPCL 1) and alternative transitional or restricted proposals (LPCL 2) for Low POP Concentration Limits (LPCLs) referred to in Article 7(4)(a) and to be specified in Annex IV of the POP Regulation.

Substance	LPCL 1 (ppm)	LPCL 2 (ppm)	Remark
TetraBDE	10	200	LPCL 2 to be reviewed by 2016
PentaBDE	10	200	LPCL 2 to be reviewed by 2016
HexaBDE	10	200	LPCL 2 to be reviewed by 2016
HeptaBDE	10	1000	LPCL 2 to be reviewed by 2016
PFOS	10	50	LPCL 2 Review by 2016; With stricter limitation for sewage sludge at 0.5 or 5 ppm
PeCB	50	50	With potential limitation for sewage sludge at 1 ppm
SCCP	1,000	1,000	
HBCD	100	1,000	LPCL 2 covers separated waste types
PCP	1,000	1,000	Only including Textile waste
HCBD	100	100	To be reviewed by 2016
PCN	10	10	No identified waste type is covered

As regards MPCLs the project team suggests the lower set of MPCL options as preferred solution, if the preferred proposal for LPCLs is selected by the European Commission and the Member States.

Table 1-2: Overview of preferred proposals (MPCL 1) and alternative transitional proposals (MPCL 2) for Maximum POP Concentration Limits (MPCLs) to be specified in Annex V, Para 2 of the POP Regulation

Substance	MPCL 1 (ppm)	MPCL 2* (ppm)
TetraBDE	2,500	25,000
PentaBDE	2,500	25,000
HexaBDE	2,500	25,000
HeptaBDE	2,500	25,000
PFOS	50	500
PeCB	500	5,000
SCCP	5,000	50,000

Substance	MPCL 1 (ppm)	MPCL 2* (ppm)
HBCD	5,000	50,000
PCP	5,000	50,000
HCBD	1,000	10,000
PCN	1,000	10,000

* MCPL 2 to be reviewed by 2016

The preferred proposal (MPCL 1) represents a maximally precautionary approach, taking into consideration concerns that have been raised by certain stakeholders against the MPCL limits for the first 12 POPs. In case the transitional solution is selected for the LPCLs (LPCL 2), the higher set of transitional MPCLs (MPCL 2) should be chosen in order to avoid conflicts between LPCLs and MPCLs.

Proposed LPCLs and related impacts

For all relevant source sectors and each POP substance the impacts of the proposed limit values in have been quantified in terms of POP substance quantities, related waste quantities, costs for changes in waste management, possible effectiveness to eliminate the POP substance and impacts on the elimination capacity. The environmental impact can be quantified in terms of tonnes of eliminated POP substances. The environmental impact is considered to be a function of the quantity of eliminated POP substances. A high quantity of eliminated POP substances is considered to be related to high positive environmental impacts.

For each POP substance the following is briefly discussed for the proposed LPCLs (see chapter 10.3):

- The additional costs for changes in waste management and recommended measures (including an appraisal of the proportionality of the burden)
- Possible impacts on WEEE and ELV recycling targets (only relevant for PBDEs)
- The effectiveness of the limit values concerning the elimination of the POP substance
- The impact in terms of additional elimination capacity
- The impacts on environment and health in terms of a quantification of the POP elimination

Proposed measures and related impacts

The alternative transitional proposals for LPCLs (LPCL 2) take particularly account of the position of some Member States, expressed at or after the TAC Meeting in September 2010, of setting comparatively high (i.e. less strict) concentration limit values initially until more data becomes available.

However, also when the alternative transitional proposals will be set, all appropriate measures to destroy or irreversibly transform the POP substances should be taken as soon as possible in order to

reduce the occurrence and the cycling of these substances and to avoid their carry over and dilution in new diverse products and waste.

Corresponding measures for the individual substances which are particularly considered appropriate are related to specific waste management activities. The relevant impacts related to these measures have been evaluated. If the proposed measures will be efficiently implemented, it is expected that the concentration levels in the relevant waste streams will be reduced below proposed LPCLs (LPCL 1). The most relevant measures are the following:

- Separation and destruction of PUR foams from cars built in 2000 and before
- Separation and destruction of ABS plastics from cathode ray computer and TV monitors produced in 2003 and before
- Separation and destruction of carpets from bulky waste
- Separation and destruction of EPS/XPS insulation boards as far as reasonably possible

Impacts of proposed limit values on current recycling activities

Recycling activities which could be concerned by the proposed limit values are particularly the following:

- ELVs recycling (PBDEs)
- WEEE plastics recycling (PBDEs)
- Recycling of carpets (PFOS)
- Recycling of EPS/XPS foams (HBCD)

The impacts of the proposed LPCLs (LPCL 1) on these activities have been evaluated with the conclusion that they are related to low to moderate economic and waste management impacts. The implementation of the LPCLs is related to specific measures which are generally related to positive environmental and health effects due to the fact that increasing shares of new or candidate POPs will be destroyed or irreversibly transformed.

Possible impacts on product and waste regulation

The proposed LPCLs have been evaluated with respect to their compatibility with European product and waste legislation, in particular possible impacts on the REACH Regulation, the ROHS Directive, the recycling targets of the WEEE Directive and the ELV Directive and finally the Waste Shipment Regulation have been analysed. It can be concluded that proposed LPCLs are compatible with existing EU product and waste legislation. There may be a need to adjust the legislation on shipments of waste.

Possible implications of concentration limits for PBDEs in new articles produced from recycled materials above which the articles cannot be placed on the market

According to Annex I of the POP regulation PBDE containing materials can be used for articles and preparations containing concentrations below 0.1% of the relevant PBDE congeners when produced partly or fully from recycled materials or materials from waste prepared for re-use.

In the opinion of the project team it is not necessary to maintain the 0.1% concentration limit for PBDEs in new articles produced from recycled materials in Annex I of the POP regulation. It could be lowered to the level of the LPCL. Real contamination levels in the relevant waste streams should be evaluated prior to lowering the concentration limits for PBDEs in new articles produced from recycled materials in Annex I.

2 Background and objectives

The European Union is party to two international legally binding instruments related to persistent organic pollutants (POPs):

- The "Protocol to the regional UNECE Convention on Long-Range Transboundary Air Pollution" (CLRTAP)
- The Global "Stockholm Convention" on POPs

Both instruments aim to reduce and eliminate the production, use and releases of POPs in all participating parties. Amongst other issues these regulations contain particular provisions on the environmentally sound management of wastes consisting of, containing or contaminated by POPs (hereafter called "POPs 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 to the environment all over the world and entered the food chain.

Although substantial progress has been achieved in limiting the use and application of these substances and reduce their emissions to environment there are ongoing releases to 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 product and of anthropogenic emission sources have to be addressed. In this framework the waste management is an essential sector for the control of POP releases into the environment. Thus the comprehensive regulation of POP contamination in waste has 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 the so-called POP Regulation. This Regulation besides other foresees an obligation to generally destroy or irreversible transform the POP content of waste above a certain concentration, but also contains the provisions 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 Annex V part 1 of 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 in safe, deep underground, 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 if they do not exceed a maximum POP concentration limit. The maximum POP content limits, however, apply exclusively to landfill sites for hazardous waste and do not apply to permanent underground storage facilities for hazardous wastes, including salt mines.

In 2006 concentration limits were defined for 14 POPs substances and substance classes, on the basis of a scientific study¹, above which the POPs content in waste shall be subject to destruction or irreversible transformation (lower POP concentration limit; LPCL) as well as threshold limits (maximum POP concentration limit; MPCL) above which no derogation from the destruction obligation can be granted.

In 2009 a number of new substances have been added to the annexes of the Stockholm Convention by decision of the fourth conference of the Parties (COP-4). These substances and substance groups² are thus to be subjected to corresponding destruction and elimination obligations or at least to limitations in use. In addition it was decided to undertake a work programme to provide guidance on the best options to restrict and eliminate perfluorooctane sulfonic acid (PFOS), its salts, PFOSF and other chemicals listed in Annex B or A, and Parties were invited to contribute to these tasks.

Because α -, β -, γ -HCH (Hexachlorocyclohexane), Chlordecone and Hexabromobiphenyl (HBB) are already included in the Annexes to the EU POP Regulation since 2006, the remaining substances and substance groups

- tetra-, penta-, hexa-, and hepta- BDE (Polybrominated diphenylethers; PBDEs), Pentachlorobenzene (PeCB), PFOS and its derivatives³

need to be analyzed and subject to POP content thresholds, as so-called “new POPs”.

PBDEs like other POPs have been banned in the EU and may not be placed on the market, but in contrary to the other new POPs they will continue to challenge the waste management sector due to the life-span of major product groups (e.g. vehicles, electronics) containing these substances, and due to the fact that recycling targets are set. Based on this background exemptions allowing continued recycling have been negotiated in the Stockholm Convention.

In addition, 5 further substances and substance groups⁴ are proposed to be added or were recently added to the UNECE POP Protocol. These substances are currently under review procedures and are

¹ Study to facilitate the implementation of certain waste related provisions of the Regulation on persistent Organic Pollutants (POPs)

http://ec.europa.eu/environment/waste/studies/pdf/pops_waste_full_report.pdf

² α -, β -, γ -HCH, chlordecone, HBB, tetra-, penta-, hexa-, and hepta- BDE, PeCB, PFOS (including salts) and PFOSF

³ Perfluorooctane sulfonic acid and its derivatives (PFOS) $C_8F_{17}SO_2X$ ($X=OH$, Metal salt ($O-M^+$), halide, amide, and other derivatives including polymers)

⁴ Short chained chlorinated paraffins, hexabromocyclododecane, pentachlorophenol, hexachlorobutadiene and polychlorinated naphthalenes

likely to be added or proposed to the Stockholm Convention in the course of the next years, so that they are called “candidate POPs”.

Table 2-1 gives an overview of the substances which are relevant for the present project.

Table 2-1: List of new and candidate POPs

Substance name	Substance abbreviation	CAS No(s)	Ban in EU	POP status
Tetrabromodiphenyl ether	TetraBDE	40088-47-9 and other TetraBDE congeners present in commercial pentabromodiphenylether	Since 2004	new POP
Pentabromodiphenyl ether	PentaBDE	32534-81-9 and other PentaBDE congeners present in commercial pentabromodiphenylether	Since 2004	new POP
Hexabromodiphenyl ether	HexaBDE	68631-49-2 207122-15-4 and other HexaBDE congeners present in commercial pentabromodiphenylether	Since 2004	new POP
Heptabromodiphenyl ether	HeptaBDE	446255-22-7 207122-16-5 and other HeptaBDE congeners present in commercial pentabromodiphenylether	Since 2004	new POP
Perfluorooctane sulfonic acid and its derivatives (PFOS) $C_8F_{17}SO_2X$ ($X=OH$, Metal salt ($O-M^+$), halide, amide, and other derivatives including polymers	PFOS	1763-23-1 (PFOS); 307-35-7 (PFOSF) and e.g. (non-exhaustive): 2795-39-3, 29457-72-5, 29081-56-9, 70225-14-8, 56773-42-3, 251099-16-8	Listed in Annex B of the Stockholm Convention as new POP since May 2009; exemptions for specific uses	new POP
Pentachlorobenzene	PeCB	608-93-5	No intentional production or use; listed as new POP in Annex A and C of the Stockholm Convention in May 2009	new POP
Short chained chlorinated paraffins	SCCPs	85535-84-8	Not yet banned in the EU, but restricted use since 2002	candidate POP
Hexabromocyclododecane	HBCD (HBCDD)	25637-99-4 3194-55-6	Not yet banned in the	candidate POP

Substance name	Substance abbreviation	CAS No(s)	Ban in EU	POP status
			EU; still used and produced (Netherlands)	
Pentachlorophenol	PCP	87-86-5	No use and production within EU	candidate POP
Hexachlorobutadien	HCBD	87-68-3	No use and production within EU; unintentional production as by-product	candidate POP
Polychlorinated naphthalenes	PCN	Consist of 75 CAS numbers for 75 PCN congeners	No use and production within EU; unintentional production as by-product	candidate POP

It has to be noted that in addition three pesticides have been proposed to be added to the UNECE POPs Protocol:

- dicofol
- trifluralin
- endosulfan

Since these substances are pesticides and were only used for agricultural purposes, they will not be analysed in the framework of the study. The limit values will be set using an analogous methodology as applied for the other pesticides already covered by the POP Regulation.

Regulations 756/2010 (EU) and 757/2010 (EU) amending Regulation (EC) No 850/2004 include all “New POPs”. However, in Annexes IV and V of the amended Regulation (EC) No 850/2004, concentration limits for the lower and upper content are neither specified for the four relevant PBDEs nor for PFOS and its derivatives. Until such time as concentration limits are established, the competent authorities of Member States may adopt or apply concentration limits or specific technical requirements (according to Article 7.4.(a) and Article 7.5.(a) of Regulation (EC) No 850/2004). Analogue to the other POP pesticides, for PeCB the proposed amendment establishes a limit value of 50 mg/kg in Annex IV and of 5000 mg/kg in Annex V. These proposed limit values for PeCB should be verified.

Although the substances are listed or proposed for inclusion into the Stockholm Convention and/or the UNECE POP protocol, there is lack of knowledge on used quantities, concentrations and sources as well as on aspects related to waste management in terms of disposal and recycling. Consequently further analysis is needed.

Based on this background the main objective of the proposed study is to provide to the Commission necessary scientific information in order to amend the POP Regulation in view of setting limit values for newly listed substances.

In order to meet this objective the following tasks have been performed:

- Compilation and evaluation of existing data on the new POPs and candidate POPs
- Assessment of impacts of potential limit values on waste flows and competing legislation
- Justified proposal for low POP concentration limits (LPCL) and maximum POP concentration limits (MPCL) for new POPs and candidate POPs.
- Preparation of a TAC Meeting to discuss available interim results with relevant experts from Member State authorities
- Conducting a stakeholder consultation to discuss available interim results with relevant stakeholders

In order to facilitate the reading and the understanding of the relation between contamination levels in various compartments, existing legal limit values in various fields, the limit values set under the Basel Convention and the existing and proposed LPCL and MPCL the transcription of units used in the different regulations and Conventions or given for analytical results is shortly presented below:

- ppm = mg/kg = μ g/g
- ppb = ng/g = μ g/kg
- ppt = pg/g = ng/kg.

3 Sources of Information

3.1 Structure of required information

The information needs were structured according to the project objectives and can be categorised for the relevant substances as data on:

- occurrence and levels in different wastes, recycled articles and articles in use
- types of articles recycled or produced from recycled material, extent of recycling, environmentally sound waste management and potentially associated releases
- occurrence and levels in articles that might be recycled in future
- existing concentration limits in waste
- processes using PFOS and its derivatives
- recycling and other management operations for candidate POPs (where appropriate), PBDE, PFOS and its derivatives in articles
- management costs for remediation methods for contaminated sites in relation to PBDE and PFOS and its derivatives in articles if relevant
- PFOS and its derivatives in production processes, environmentally sound management options, recycling operations and potential related releases

Further information needs related to the identification and development of recommendations is:

- information that enables to identify the most appropriate reference measurement methods for the determination of the substances in waste and sampling and analysis methods for PBDE and PFOS and related substances in articles
- information needs such as links to additional relevant information or impact relevant information

3.2 Questionnaire survey

A questionnaire survey was carried out in order to obtain project specific information on the information needs listed in chapter 3.1.

In this context it was important to consider that in November 2009 the Secretariat to the Stockholm convention has launched a request for information on new POPs, including brominated diphenyl ethers and PFOS, in accordance with decision SC-4/19⁵. The information request was related to (1) Brominated

⁵ See <http://chm.pops.int/Programmes/NewPOPs/InformationrequestdecisionSC419/tabid/666/language/en-US/Default.aspx>

diphenyl ethers found in articles, (2) PFOS and its derivatives and (3) other chemicals listed in Annexes A or B of the Convention at the COP4. The information request was sent to National Focal Points and Official Contact Points to the Stockholm Convention including the 27 EU Member States. Due to overlapping objectives with the present project similar information was requested in the corresponding questionnaire of the Stockholm Convention ("Questionnaire SC-4/19"⁵) as is needed for the present project.

Against this background, and in order to reduce the reporting burden of the Member States a questionnaire for the present project was developed in close coordination with the Commission services on the basis of the specific information needs listed above and considering the questions already addressed in the SC Questionnaire SC-4/19.

The questionnaire which is annexed to the present report (see Annex) was sent at the end of June 2010 to approximately 250 stakeholders belonging to the following stakeholder groups:

- Stockholm Convention (Basel Convention) focal points
- Technical Working Group Members Waste Treatment Industries
- Committee for the Adaptation to Scientific and Technical Progress of EC Legislation on Waste (TAC), Regulation (EC) No 850/2004 of the European Parliament and of the Council on persistent organic pollutants
- Committee Waste Framework Directive
- Environment Waste Attachés at the permanent representations to the EU
- Several NGOs
- Industry stakeholders

The list of stakeholders was established in close coordination with the Commission services. The questionnaire was distributed at the end of June together with an accompanying recommendation letter from the Commission services and with a copy of the questionnaire SC-4/19. Stakeholders were invited to provide feedback by 20 of July. Several telephone contacts with stakeholders showed that there is currently not much information available in addition to the information provided to the Secretariat to the Stockholm Convention in accordance with decision SC-4/19. This explains that the feedback to the questionnaire was comparatively moderate.

Both, the information submitted in accordance with SC-4/19 and in response to the information request of the present project were used as important information source for the present report. The information submitted to the Stockholm Convention Secretariat in accordance with decision SC-4/19 is available at the Stockholm Convention website⁶. The Deadline for submission was 1 of July 2010. On 29

⁶ See <http://chm.pops.int/Programmes/NewPOPs/InformationrequestdecisionSC419/NewPOPsInformationdecisionSC419/tabid/712/language/en-US/Default.aspx>

of July 2010 submissions from 28 countries, thereof 9 EU Member States were available. The information related to PBDEs was evaluated in a draft document [Watson et al. 2010]. The draft document and the responses to the questionnaire were used as information source.

In the course of the present project an information exchange was carried out with the Secretariat to the Stockholm Convention and with the authors of the study on the relevant substances (i.e. particularly with the authors of [Watson et al. 2010]). The intention was to exchange results and data in order to use synergies between the work of the Stockholm Convention and the present project at EU level in order to optimise the outcome of both studies.

All responses to the information request are summarized in Annex II. From a total number of approximately 250 contacted stakeholders only 5 Member States provided relevant information on new POPs and the candidate POPs (see Table 3-1; the figure 279 includes some double counted stakeholders).

In addition it should be highlighted that a number of stakeholders attached helpful scientific studies to their submitted responses.

However, based on the low number of received responses it can be assumed that the information on new POPs and POP candidates is rather scarce.

Table 3-1: Overview of contacted stakeholders

	Number of contacted Stakeholders
Industry 1	39
Industry 2	20
NGOs	5
Stockholm Convention Focal Points	26
TWG Members (Waste Treatment Industries)	35
TAC POPs	50
Committee WFD/MS	76
Env. Waste Attachés MS27	27
Other	1
TOTAL	279

As already mentioned additional information was obtained from the SC-4/19 questionnaires. All relevant responses are summarized in Annex III of this report.

Each part of the SC-4/19 Questionnaires is divided in a section A for general information and a section B for more detailed information. All in all it can be summarized that relevant data could be extracted especially from sections A of the SC-4/19 questionnaires. With regard to the more detailed sections (B) it can be concluded that very limited information was provided by only few respondents. This again indicates that the information on new POPs and candidate POPs is quite limited.

3.3 European Commission

The information basis has been complemented with specific information provided by the Commission Services.

The Commission has established a working group at European level for the exchange of data on PBDEs and PFOS. Access to the information exchange is provided via an interest group at the CIRCA website. A dedicated CIRCA interest group "POP waste 2010"⁷ was set up. This CIRCA group serves as a platform, allowing for the exchange of documents, studies and information on the new POPs. Intermediate reports of the present project and other findings will also be posted on the CIRCA group. The project team is integrated in the information exchange in order to have access to relevant information.

3.4 Expert meeting with Member States

An expert meeting of the Committee for the Adaptation to Scientific and Technical Progress of EC Legislation on Waste (TAC), Regulation (EC) No 850/2004 of the European Parliament and of the Council on persistent organic pollutants on 27 September 2010 took place in Brussels on 27 September 2010.

The interim results were presented and discussed. The outcome of the meeting is documented in a summary record of the meeting. The summary record of the meeting, the power point presentation of the interim results and the interim report is available to the above mentioned interest group at the CIRCA website.

According to the available results at the date of the workshop, the meeting concentrated on the new POPs (PBDEs, PFOS and PeCB). Results on the candidate POPs will be elaborated during later stages of the project.

The outcome of the discussions of the meeting is taken into account for the present report.

3.5 Stakeholder consultation on the interim results

At the beginning of November 2011 about 300 stakeholders were invited to comment on the revised interim report and summary report related to the project background, methodology and the interim results for the new POPs (i.e. PBDEs, PFOS and PeCB).

The contacted stakeholders belong to all kinds of relevant groups:

- MS representatives Waste Framework Directive
- MS representatives ELV Directive
- MS representatives POP Regulation
- Environment Attachés of the MS
- NGOs
- Industry stakeholder from different sectors such as recycling, ELV recycling, WEEE recycling and plastics recycling

⁷ Link: http://circa.europa.eu/Members/irc/env/pop_waste_2010/home

The list of stakeholders was established in close coordination with the Commission services. The consultation was distributed at the beginning of November together with accompanying background information. Stakeholders were invited to forward the information to other relevant and interested stakeholders and to provide feedback by 1 of December.

The feedback to the questionnaire was comparatively moderate. Around 10 stakeholders contributed to the commenting process. Among the contributors were two Member States and several industry stakeholders from associations or individual enterprises (EERA, EECA-ESIA, ACEA, FEAD, EUROMETAUX, WVM, FNADE, ARN, ECN). Specific comments and information submitted were taken into account for updating and improving the information for the present report.

3.6 Statistics, data bases and literature

Specific information sources are documented in the references which are listed separately in chapter 6 for the relevant substances and for general references at the end of the report (see chapter 10, "General References")

The following is a list of selected information sources used for the present report:

- Information that is available at international POP Conventions including in particular POP risk profiles and management option dossiers, recent studies conducted for the Secretariat to the Stockholm Convention (e.g. study on PBDE)
- Relevant reports and any other material available at the EU Commission or other European institutions such as the EEA, EUROSTAT or the European Chemicals Bureau and the European Chemicals Agency
- Information accessible via national and international emission inventories (e.g. PRTR) in particular those established as a consequence of CLRTAP
- National and international waste statistics, databases / reports from the Member States, national focal points under the Stockholm Convention and international institutions (such as UNECE, UNEP, FAO, WHO, OECD, IARC, IFCS ...)
- Information from industry associations (CEFIC, Eurochlor etc.)
- Submissions to the information request from the MS in accordance with decision SC-4/19 of COP-4 to the Stockholm Convention
- Submissions to the information request by the project team
- BREF documents for the relevant industry sectors
- EEA databases for emission and waste
- Stockholm BAT-BEP Guidelines (UNEP)
- Basel Technical Guidelines

Each of the subchapters related to the individual substances in chapter 6 contains a list of references cited within the corresponding chapter. In addition a list of general references can be found in chapter 10.

4 Legal Situation

This chapter provides a general overview of legal requirements and provisions related to POPs and new POPs on international, European and national level. Specific limit values and further details are discussed in the corresponding chapters of chapter 5.

4.1 Legal Background at international and at EU level

4.1.1 Specific legal background for POP management

International level

At international level, the legal framework is formed by

- the Stockholm Convention and
- the POPs Protocol under the UNECE CLRTAP Convention.

The **Stockholm Convention** is an international agreement the major objective of the protection of human health and the environment from POPs by means of elimination from use or restriction to use for all POPs and the minimisation of unintentional production. It was opened for signature in 2001 and entered into force 2004. The Convention aims at reducing and eliminating production, use and release of POPs. It exhibits in its Annexes production / marketing bans and restrictions for intentionally produced substances as well as obligations to reduce releases.

The obligations for the parties include measures to reduce releases from stockpiles and wastes. In Article 6 of the Convention, provisions for waste are set out related to environmental sound management (ESM). Besides other wastes shall be disposed of in such a way

- that the persistent organic pollutant content is destroyed or irreversibly transformed so that they do not exhibit the characteristics of persistent organic pollutants or
- otherwise disposed of in an environmentally sound manner when destruction or irreversible transformation does not represent the environmentally preferable option or the persistent organic pollutant content is low.

In the original version, twelve substances/groups of substances were addressed. Four COPs have been held, the last in May 2009 at which the inclusion of nine substances in the Annexes has been concluded (entering into force for the parties in August 2010), so that currently, the following substances are in the scope of the Stockholm Convention:

- Pesticides: aldrin, chlordane, DDT, dieldrin, endrin, heptachlor, hexachlorobenzene, mirex, toxaphene; chlordecone, alpha hexachlorocyclohexane (HCH), beta hexachlorocyclohexane, lindane, pentachlorobenzene;

- Industrial chemicals: hexachlorobenzene (HCB), polychlorinated biphenyls (PCBs), hexabromobiphenyl (HBB), tetra-, penta-, hexa-, and heptabromodiphenyl ether (PBDEs), pentachlorobenzene (PeCB), perfluorooctane sulfonic acid (PFOS), its salts and perfluorooctane sulfonyl fluoride;
- By-products: HCB; PCDD/PCDF, PCBs, alpha- and beta-HCH and PeCB.

The **POPs Protocol** under the UNECE CLRTAP Convention was opened for signature in 1998 and entered into force in 2003. The protocol originally covered 16 substances / groups of substances with POP properties. In December 2009, it was decided to extend the scope of application to seven new substances and to deepen present regulations so that the following 23 substances / groups of substances are currently addressed by the Protocol:

- Aldrin, chlordane, chlordecone, dieldrin, endrin, hexabromobiphenyl, mirex, toxaphene; DDT, heptachlor, hexachlorobenzene, PCBs, HCH including lindane, PCDD/PCDF, PAHs, hexachlorobenzene;
- Hexachlorobutadiene (HCBD), c-octa and c-pentabromodiphenyl ether, pentachlorobenzene, perfluorooctane sulfonates, polychlorinated naphthalenes (PCN), and short-chain chlorinated paraffins (SCCPs)

EU level

The EU is party to both the Stockholm Convention and the POPs Protocol. At Community level, the relevant legal act for implementation of these documents is **POPs Regulation (EC) No 850/2004** (as amended) which, at the same time, sets out further obligations for Member States. The Annexes of POP Regulation are dedicated to prohibitions and restrictions of intentionally produced substances (Annex I and II), release reduction measures (Annex III) and waste management provisions (Annex IV and V).

With regard to the treatment of POP containing wastes, apart from Annexes IV and V, Article 7 is of major relevance. Following Article 7 and in accordance with Article 6 of the Stockholm Convention, wastes containing POPs must be treated in such a way as to ensure that the POP content is destroyed or irreversibly transformed so that the remaining waste and releases do not exhibit the characteristics of POPs. Exemptions of this principle are possible

- if the POP content is below the limit value as of Annex IV „low POP concentration limit (LPCL)“, Article 7(4) lit a.;
- in exceptional cases for specific waste streams from power production, aluminium, lead, copper, zinc and other metallurgical processes, C&D and thermal waste treatment⁸ listed in Annex V for which destruction or irreversible transformation of the POP content does not represent the environmentally preferable option up to maximum POP concentration limits (MPCL).

⁸ 100114*, 100116*, 100207*, 100304*, 100308*, 100309*, 100319*, 100321*, 100329*, 100401*, 100402*, 100404*, 100405*, 100406*, 100503*, 100505*, 100603*, 100606*, 100808*, 100815*, 100909*, 161101*, 161103*, 170106*, 170503*, 170902*, 170903*, 190107*, 190111*, 190113*, 190115*, 190402*, 190403*

POPs Regulation shall be amended in order to comply with recent decisions at Stockholm Convention and POPs Protocol level, amending the Annexes by including the new POPs.

As concerns specific exemption from the ban on marketing and use on intermediate use and other specifications, unintentional trace contamination levels and provisions have been set in Regulation (EU) No 757/2010 amending Regulation (EC) No 850/2004 of the European Parliament and of the Council on persistent organic pollutants as regards Annexes I and III for tetra-, penta-, hexa-, and hepta- PBDE congeners as well as for PFOS and its derivatives.

For PFOS these proposed exemptions foresee to consider a level of contamination of 10 ppm (0.001%) in substances or preparations, a level of 1,000 ppm (0.1%) in semi-finished products or articles or parts thereof, or a PFOS concentration of 1 µg/m² in coated materials as unintentional trace contamination. In addition article in use shall be allowed. Certain applications shall be exempted from the ban by way of derogation until new information and safer alternatives are available, and limited derogations are foreseen for fire-fighting foams (2011) and wetting agents for electroplating systems (2015).

For the relevant PBDE congeners the exemptions foresee a contamination level of 10 ppm (0.001%) as unintentional trace contamination when it occurs in substances, preparations, articles or as constituents of the flame-retarded parts of articles. By way of derogation, the production, placing on the market and use of the following shall be allowed: (a) without prejudice to subparagraph (b), articles and preparations containing concentrations below 0,1 % of heptabromodiphenyl ether by weight when produced partially or fully from recycled materials or materials from waste prepared for re-use; (b) electrical and electronic equipment within the scope of Directive 2002/95/EC. 3. Use of articles already in use in the Union before 25 August 2010 containing Heptabromodiphenyl ether as a constituent of such articles shall be allowed.

With regard to lower/upper POP content limit values, provisional thresholds have been set for PeCB in Regulation (EU) No 756/2010 amending Regulation (EC) No 850/2004 of the European Parliament and of the Council on persistent organic pollutants as regards Annexes IV and V, whereas for PBDEs and PFOS, no limit values have been set yet.

4.1.2 Legal background with regard to waste management

Waste Framework

At EU level, the basic legislation with respect to waste management is **Waste Framework Directive 2008/98/EC** (replacing and repealing former Waste Framework Directive 2006/12/EC with effect of 12 December 2010). The directive contains definitions for waste as well as waste categories and disposal and recovery operations; inter alia it sets for the first time criteria for end-of-waste-status of items and introduces “reuse” as favourable option within the waste hierarchy. Furthermore properties and characteristics rendering waste hazardous as well as specific obligations for hazardous wastes are defined in the new Waste Framework Directive.

Shipment of waste

At international level, transfrontier shipments of waste are regulated by the **Basel Convention** on the control of transboundary movements of hazardous wastes and their disposal, which was opened to signature 1989 and entered into force 1992. The Convention aims, in introducing a system for controlling the export, import and disposal of hazardous wastes and their disposal, to reduce the volume of such exchanges so as to protect human health and the environment. The Basel Convention defines hazardous wastes as being listed in Annex I and exhibiting one of the hazardous characteristics contained in Annex III.

The OEWG under the Basel Convention has worked on the definition of "low POP content" levels for POP substances and substance group subject to the Stockholm Convention and included these values into the "General technical guidelines for the environmentally sound management of wastes consisting of, containing or contaminated with persistent organic pollutants (POPs)" (last issue 2007) but does not yet include limits for new POPs or candidate POPs.

The relevant EU legislation implementing the Basel Convention (as well as OECD Decision C(2001)107/final concerning the control of transboundary movements of wastes destined for recovery operations) is **Waste Shipment Regulation (EC) 1013/2006** which applies to shipments of waste between Member States, within the Community or with transit through third countries, imported into the Community from third countries, exported from the Community to third countries and in transit through the Community, on the way from and to third countries. The Waste Shipment Regulation distinguishes principally two procedures (simplified):

- the "notification" procedure, which consists of prior consent of all involved concerned authorities and which applies to all waste destined for disposal operations and waste destined for recovery operations but listed in Annex IV (wastes of major concern – "amber listed" waste), IVA or not yet any of the lists (Article 3(1) or is restricted for export to third countries (Article 36-38)
- the "Article 18" procedure which does not foresee any prior approval by authorities and which applies to waste of lesser concern (listed in Annex III and IIIB of the Regulation – "green listed" waste,) destined for recovery operations (Article 3(2)).

Few limit values are defined in these lists to determine whether waste being contaminated with hazardous substances is to be listed "amber" or not. The inclusion of the new Pops including the definition of low POP concentration limits (LPCL) in the POPs Regulation is expected to have impact on transports of wastes containing POPs (necessity of notification procedure, ban for export).

Treatment of waste

The Landfill Directive 1999/31/EC sets out requirements for the operation and management of landfills (e.g. technical standards, permit, monitoring and control). Issued by the Council by means of delegation of the Landfill Directive, the **Decision on Waste Acceptance Criteria 2003/33/EC** further specifies requirements for waste acceptance procedures and acceptance criteria, including waste types and limit values for a number of parameters. However, the substances relevant for this project are not addressed

in this document. It is reported that the parameter “AOX” serves as a tool for identifying waste streams contaminated with POPs.

Single waste stream management

WEEE Directive 2002/96/EC aims at preventing WEEE by promoting reuse, recycling and recovery and is of major interest for the project. For the ten WEEE categories defined in Annex IA, the Directive sets targets for recovery and for component, material and substance reuse and recycling. As regards new POPs (PBDEs) the targets set for categories 3 and 4 are of relevance, requesting:

- A recovery rate of a minimum of 75 % by an average weight per appliance, and
- A reuse and recycling rate for component, material and substance of a minimum of 65 % by average weight per appliance

In this context it has to be noted that, the plastic fraction in WEEE range from 10-40%, and that recycling will not be allowed anymore for a WEEE material classified as POP waste as, according to the POP Regulation Article 7(3), disposal or recovery operations that may lead to recovery, recycling, reclamation or re-use of the substances listed in Annex IV to the Regulation shall be prohibited. Therefore, a recycling is only possible if the POP content of the waste is separated before or during recycling and is disposed of appropriately.

According to Annex II “Selective treatment for materials and components of waste electrical and electronic equipment in accordance with Article 6(1)” as a minimum [...] [e.g. plastic containing brominated flame retardants] has to be removed from any separately collected WEEE.

A concentration limit above which the removal has to be carried out is not specified. The obligation can therefore be considered relevant for any detectable PBDE concentration.

The FAQ Document on the RoHS and WEEE Directive states that Article 6 and Annex II to the WEEE Directive lay down the requirements for the treatment of WEEE. To ensure compliance with the waste framework Directive (75/442/EEC), the treatment shall, as a minimum, include the removal of all fluids and a selective treatment for materials and components in accordance with Annex II. Member States authorities that are responsible for the enforcement of the national legislation implementing these provisions have produced guidance document on their national provisions. This document covers (i) an interpretation of the term "has to be removed", (ii) technical guidance regarding treatment of the substances, preparations and components that have to be removed from any separately collected WEEE and (iii) an interpretation of the further provisions of the Annex II. More information can be obtained directly from the Member States applying this guidance document. [EC 2006]

A corresponding guidance document has not been established yet⁹ and it is in the responsibility of the Member States to apply the detection limit or other limit values above which plastic containing brominated flame retardants has to be removed from any separately collected WEEE.

Concerning POPs, the general technical guideline of the Basel Convention requires “Where only part of a product or waste, such as waste equipment, contains or is contaminated with POPs, it should be separated and then disposed of ...” [BC TG 2007]. Concentration limits are not specified. The removal is therefore recommended at any POP concentration.

RoHS Directive 2002/95/EC restricts the use of hazardous substances (among these PBDEs) in new EEE¹⁰. Member States shall ensure that, from 1 July 2006, new electrical and electronic equipment put on the market does not contain [...] PBB or PBDE, above a maximum concentration value of 0.1 % by weight. The limit value of 0.1% is related to the sum concentration of PBDE congeners.

It has to be noted that both Directives (WEEE and RoHS) are currently re-casted. In the current recast text of the ROHS Directive (status 24.11.2010), the concentration value for PBDEs is still 0.1% by weight for PBDE (sum all congeners). DecaBDE is not exempted any more from the ban.

HBDCD was considered as a candidate for inclusion into the planned Annex III of the currently recasted RoHS Directive 2002/95/EC. In the current version of the Directive which was approved by the European Parliament on 24.11.2010, this Annex is deleted. However, the substances which were listed in the planned Annex III were moved to recital 10: “The annexes to this Directive should be reviewed periodically to take into account, inter alia, Annexes XIV and XVII to REACH. In particular, the risks to human health and the environment arising from the use of Hexabromocyclododecane (HBDCD), Bis (2-ethylhexyl) phthalate (DEHP), Butyl benzyl phthalate (BBP) and Dibutylphthalate (DBP) should be considered as a priority. With a view to further restriction of substances, the Commission should re-investigate the substances, which were subject to previous assessments, in accordance with the new criteria of this Directive as part of the first review.” A review and amendment of the list of restricted substances shall be considered by the Commission within 3 years after the entry into force of the Directive, and periodically thereafter on its own initiative or following the submission of a proposal by a Member State (see Article 6 of the current version of the Directive).

ELV Directive 2000/53/EC sets out provisions with respect to end-of-life vehicles aiming at the prevention of waste from vehicles and, in addition, at the reuse, recycling and other forms of recovery of end-of life vehicles and their components.

As regards recycling and recovery the following targets are set pursuant to article 7:

Since 1 January 2006:

- Reuse and recovery of a minimum of 85 % by average weight per vehicle and year.

⁹ Personal communication from DG Environment on 3.12.2010

¹⁰ Excluded only WEEE (Annex I) category 8 “Medical devices” and category 9 Monitoring and control instruments (not prone to contamination with new POPs)

- Reuse and recycling of a minimum of 80 % by average weight per vehicle and year;
- Not lower than 75 % for reuse and recovery and not lower than 70 % for reuse and recycling for vehicles produced before 1 January 1980

From 1 January 2015:

- Reuse and recovery of a minimum of 95 % by an average weight per vehicle and year
- Re-use and recycling of a minimum of 85 % by an average weight per vehicle and year

In this context it is important to note that vehicles in average consist of 75% metal and 25% plastic compounds, but that only part of this is PUR foam prone for C-PentaBDE contamination (for more details see chapter 6.1.9).

4.1.3 *Legal background with regard to chemicals regulation*

Classification and labelling

The new EU system for the classification, labelling and packaging is laid down in **CLP Regulation (EC) No. 1272/2008** implementing the Globally Harmonised System (GHS). The Regulation is related to substances and mixtures (former wording: preparations), describing hazards and classifying chemicals accordingly. Following Article 3(1), a substance or a mixture fulfilling the criteria relating to physical hazards, health hazards or environmental hazards, laid down in Parts 2 to 5 of Annex I is hazardous and shall be classified in relation to the respective hazard classes. The new system will stepwise entirely replace the current system of **Directives 67/548/EEC** (on substances) until December 2010 and **1999/45/EC** (on preparations) until 2015.

Both CLP and the current system introduce an obligation for manufacturers, importers and downstream users to classify substances or mixtures before placing them on the market. In addition, in the Annexes to CLP Regulation and Directive 67/548/EEC, official classifications are provided for a number of chemicals.

Out of the substances relevant for this project, PeCB and PCP are classified as hazardous in both the Annexes to CLP Regulation and Directive 67/548/EEC.

Market and use restrictions for chemical substances

The EU law on chemicals, **REACH Regulation (EC) No 1907/2006**, does not relate to waste but to products placed on the market – however, the scope notably does include recycled products which are submitted to a large extent to the same REACH obligations as newly manufactured items as soon as they leave the waste status. Recycled products are only exempted from registration duties in the case described in Article 2(7) (d)). REACH contains, *inter alia*,

- market and use restrictions of substances (formerly addressed in Directive 76/769/EEC) in Annex XVII;

- criteria for substances to be classified as
- CMR (carcinogenic/mutagenic/reprotoxic)
- PBT (persistent, bioaccumulative, toxic)
- vTvB (very toxic, very bioaccumulative)
- in Annex XIII;
- criteria for substances being classified as of “very high concern” (SVHC) as the basis for submission to authorization regime (see the criteria described in Article 57 and Annex XIII) – note that ECHA regularly updates a candidate list of substances for inclusion into relevant Annex XIV, and that a draft Regulation for completing this Annex for the first time is currently discussed;
- regulation on substances in articles (registration duties for substances in articles according to Article 7(1), enhanced information and notification obligations under certain conditions, Article 7(2)).

Market and use restrictions are laid down in Annex XVII of REACH Regulation notably for

- Polybrominated Diphenylethers;
- PFOS;
- PCP; and
- SCCP.

A SVHC which is included into Annex XIV to the REACH Regulation is subject to the authorisation regime, with Article 56 REACH Regulation setting out the basic principle that an Annex XIV substance must not be placed on the market or used after elapsing of the sunset date, unless an authorisation is granted or one of the other exemptions apply. Regulation (EU) No 143/2011 for the first time has amended Annex XIV, including HBCD. Note that Article 56(6) lit. (b) REACH Regulation REACH foresees that the authorisation regime does not apply for PBT substances such as HBCD, if they are present in a mixture below a concentration limit of 0.1 % w/w.

SCCP are included in ECHA’s current candidate list for inclusion into Annex XIV.

4.1.4 *Legal background in other fields*

Water Framework Directive 2000/60/EC introduces a "Strategy against pollution of water" including as a first step the establishment of a list of priority substances. Priority hazardous substances (PHS) will be subject to cessation or phasing out of discharges, emissions and losses within 20 years after adoption of the Directive.

PHS have been identified by **Directive 2008/105/EC**, providing Environmental Quality Standards (EQS), both for annual average (AA-EQS) and for Maximum Allowable Concentration (MAC-EQS), in [µg/l], for inland surface water and other surface water. The following relevant substances are identified as priority hazardous substances:

- Polybrominated Diphenylethers;
- PeCB;
- HCBd;
- PCP.

PFOS is identified as possible priority substance for review.

Following **PRTR Regulation (EC) No 2006/166**, industrial facilities subject to IPPC regime - apart from their releases of pollutants to air, water, land and wastewater destined for treatment in external wastewater treatment plants - have to report their transfer of waste if exceeding annual threshold levels as laid down in the Regulation. Reporting obligations include:

- Polybrominated Diphenylethers;
- PeCB;
- HCBd;
- PCP.

The following table provides an overview which of the substances relevant for this project is addressed in the mentioned legal documents.

Table 4-1: Overview of relevant legal documents addressing the selected substances

Source	PBDE	PFOS	PeCB	HBCD	HCBd	PCN	PCP	SCCP
International level								
Stockholm Convention	+	+	+		Cand*			Cand*
POPs Protocol	+	+	+	Cand*	+	Cand*	Cand*	+
EU level								
POPs Regulation (EC) No 850/2004	+	+	+					
Waste Framework Directive 2008/98/EC / EWC	+	+	+	+	+	+	+	+
Waste Shipment Regulation (EC) 1013/2006						+		
Landfill Directive 1999/31/EC / WAC								
CLP Regulation (EC) No. 1272/2008			+				+	
Directives 67/548/EEC (on substances)			+				+	
REACH Regulation (EC) No 1907/2006	+	+		+			+	+

Source	PBDE	PFOS	PeCB	HBCD	HCBD	PCN	PCP	SCCP
Water Framework Directive 2000/60/EC	+	+**	+		+		+	
PRTR Regulation (EC) No 2006/166	+		+		+		+	

*proposed as candidate for inclusion into the Convention

**proposed as substance for inclusion

For specific limit values and further details see the corresponding chapters of chapter 5.

4.2 Additional regulation at Member States level

The conditions, under which Member States may take additional action in a field already covered by EU legislation, depend on the legal basis of the relevant act and differ as follows:

If a legal document has its basis in Article 192 of Treaty on the Functioning of the European Union ("TFEU") (Ex-Article 175 of Treaty establishing the European Community), i.e. on the competence of the EU to issue legislation in the area of environment, Member States may lay down more stringent protective measures in this area under the conditions of Article 193 (Ex-Article 176 EC Treaty). Acts in this field comprise notably POPs Regulation and legislation in the field of waste (with the exception of ROHS Directive);

If a legal document has its basis in Article 114 of Treaty on the Functioning of the European Union ("TFEU") (Ex-Article 95 of Treaty establishing the European Community), i.e. on the competence of the EU to issue legislation for the approximation of the provisions laid down by law, regulation or administrative action in Member States which have as their object the establishment and functioning of the internal market, Member States may, under the conditions of Article 114 (4) and (5) respectively, maintain or introduce national provisions on grounds of major needs referred to in Article 36, or relating to the protection of the environment or the working environment. This concerns namely EU legislation on chemicals (REACH and CLP), as well as the ROHS Directive.

4.2.1 Existing concentration limits or other provisions for new POPs and candidate POPs

Any provisions identified on Member State level (or in other developed countries) that are relevant for the determination of low and maximum POP concentration limits (LPCL and MPCL) are compiled in the corresponding chapters of chapter 5.

Investigations included waste management (including recycling and transboundary transport), water, food, and soil and soil protection. Available information and number of provisions (ban, limit or target values) are scarce.

5 Overview of mass flows of new POPs and candidate POPs

5.1 Objectives and methodology

For the evaluation of the available data on the occurrence and levels of new POPs and candidate POPs and the information on relevant processes and waste management options a set of mass flows has been established. Two “types” of mass flows can be differentiated, (a) mass flows of the POP substances (substance flows, such as a C-PentaBDE flow) and (b) mass flows of materials such as flows of articles or wastes. The results of the mass flow analyses should enable a number of conclusions:

- comparison of the path "emission" to the path "waste"
- comparison of the importance of the various POPs
- comparison of the importance of various industrial, public and domestic sectors
- comparison of different management, disposal and recycling options
- relative coverage of wastes and substances by different POP limit values
- provide a basis to establish scenarios on the consequences of different limit values
- provide a basis for a prognosis of future developments.

Furthermore the methodology should be flexible and allow consideration of new data and update the established data base in order to refine/precise conclusions and future prognosis with increased knowledge.

Following these different purposes the mass flows have been calculated by means of a computer based system. As input parameters activity data, waste generation factors and contamination data for articles, wastes and emissions were used as far as accessible for the 27 EU Member States from international data bases (EUROSTAT, IEA, EEA, EMEP, national statistics), literature (BREF documents, UNEP documents on BAT/BEP with respect to POPs) and the results of the questionnaire survey plus unpublished data directly communicated by industry associations, scientific experts or NGOs.

Based on these data, figures on annual amounts of relevant articles and annual discharges of the substances from relevant sectors on EU 27 and Member State level were calculated. Overall mass flows indicating the relative importance of annual discharges to wastes or to recycling in relation to emissions have been established on EU 27 level.

In order to give an overview of the situation in EU 27 the calculation is based on mean values. Activity and contamination data are available in detail in the corresponding subchapters of the interim report. Ranges e.g. for contamination are indicated and reflect the uncertainties which are related to the corresponding results. It is not considered reasonable to calculate the mass and waste flows on the basis of ranges. These are calculated on the basis of “best estimates”. As a consequence precise figures in

waste flows do not represent an accurate result but are related to specific uncertainties which are reflected in the indicated ranges of the background data. An indicative evaluation of the data quality and uncertainties related to the different POP substances and relevant applications is given in chapter 6

As illustrated below the overall structure of the mass flows (macro dimension) follows a material flow station approach, including emissions and discharge to products and waste and its management.

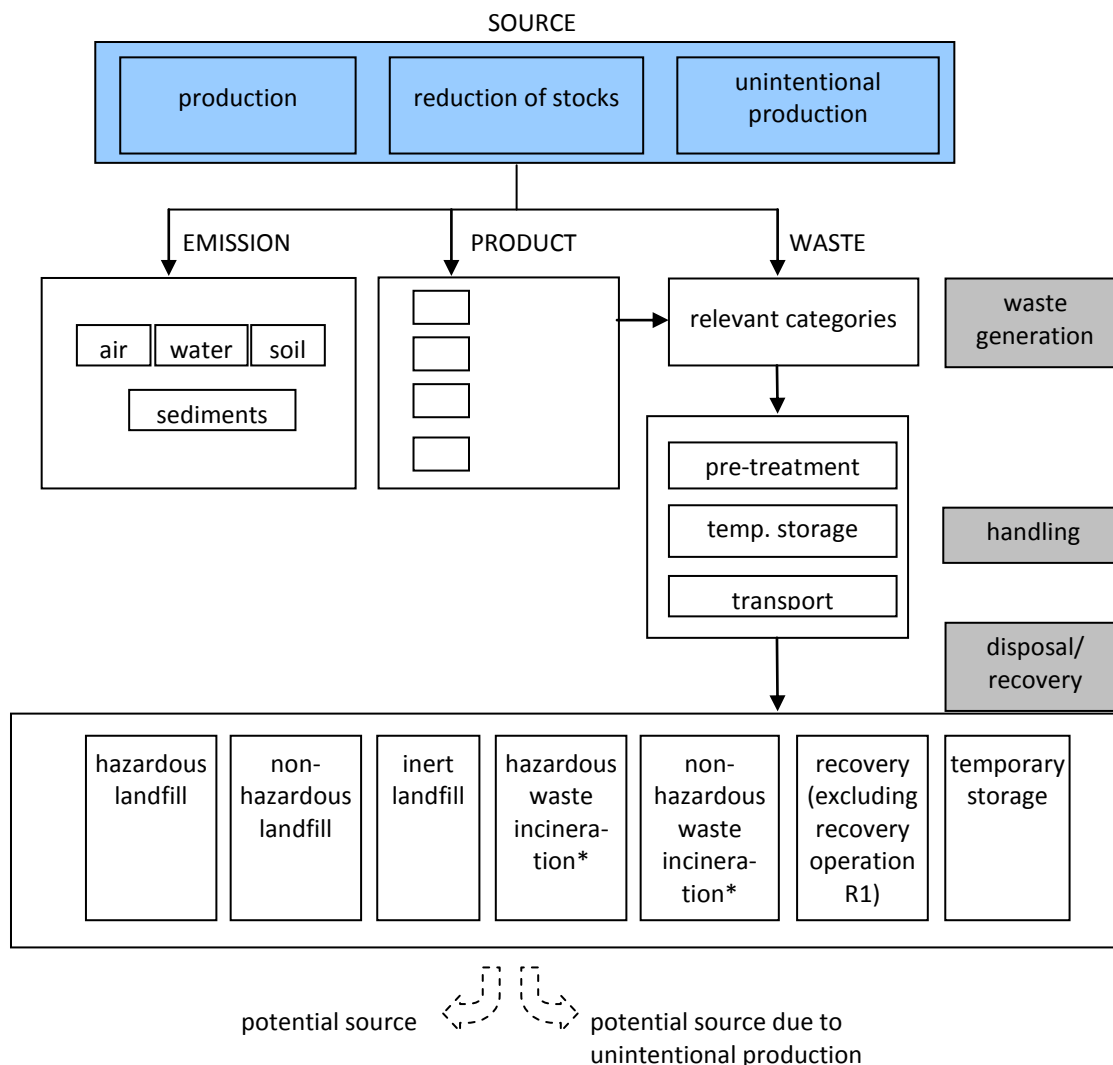


Figure 5-1: General structure of the mass flow (macro dimension)

* including recovery operation R1 (Use principally as a fuel or other means to generate energy)

According to EUROSTAT data from 2010, in 2008 municipal solid waste (MSW) was treated as follows: about 40% was landfilled, about 20% was incinerated, about 23% was recycled and about 17% was composted (E-STAT 2010¹¹). From EUROSTAT data from 2010 data it can furthermore be concluded that

¹¹ http://epp.eurostat.ec.europa.eu/portal/page/portal/waste/data/wastemanagement/waste_treatment

out of the household waste and similar waste which was incinerated in 2008, 39 % was incinerated with energy recovery and 61 % was incinerated without energy recovery¹².

On the basis of this information the significance of the waste treatment options for MSW and similar waste can be calculated.

Table 5-1: Relative importance of waste treatment options for MSW and similar waste (based on [E-STAT 2010] data for the year 2008)

Waste treatment option	MSW treatment	Similar waste, no composting	Similar waste, no composting, no recycling	Similar waste, no landfilling, no composting
Total	100%	100%	100%	100%
Landfilled	40.4%	48.7%	67.1%	n.a.
Incinerated without Energy recovery (D10)	12.1%	14.6%	20.2%	28.5%
Incinerated with Energy recovery (R1)	7.6%	9.2%	12.7%	17.9%
Recycled	22.8%	27.5%	n.a.	53.5%
Composted	17.1%	n.a.	n.a.	n.a.

In order to identify relevant inputs and waste fractions the various stages of the applied process technology have to be taken into consideration for each investigated source sector. To this end processes have been analysed in detail and the relevant process steps with respect to the flow or generation of new or candidate POPs have been identified and relevant input and output streams of the processes have been determined (micro dimension of the mass flows). As far as possible, the often complex and manifold processes have been condensed to simplified and schematic flow charts that represent the relevant input and output streams resulting in flow charts as illustrated in Figure 5-2.

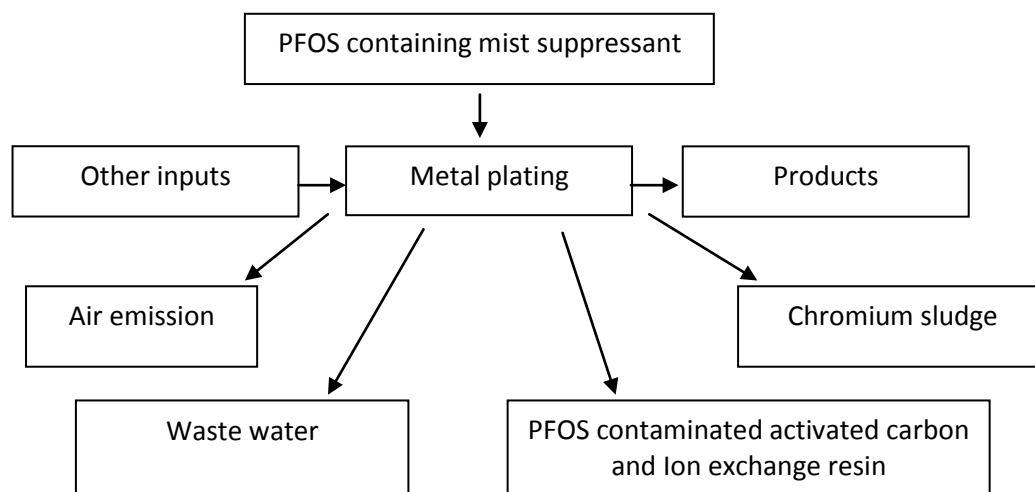
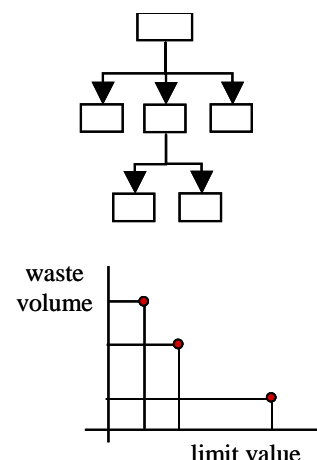


Figure 5-2: Example for process specific differentiation of residues and emissions in the mass flow (micro dimension)

¹² http://epp.eurostat.ec.europa.eu/portal/page/portal/statistics/search_database (Data explorer: treatment of waste)

In order to be able to derive recommendations and conclusions concerning limit values two types of information are important to be taken into consideration in this chapter. This consists of the following:

1. Quantities of new or candidate POPs (substance flows)
2. Volumes and management options for contaminated wastes (material flows e.g. for waste)



The results on mass flows for all new POP substances and all candidate POPs that are in the scope of the present project as well as waste/article flows for the different investigated sectors are drawn up in this chapter. More details are presented in chapter 6.

5.2 Sources of new POPs and candidate POPs entering articles, waste and the environment

5.2.1 C-PentaBDE

Polybrominated diphenyl ethers (PBDEs) are a group of additive brominated flame retardants, which were used in various applications all over the world (e.g. in cars, furniture, textiles, electrical and electronic equipment, etc).

The most common use of C-PentaBDE in Europe was in flexible polyurethane (PUR) foams, accounting for approximately 95 % of the total use of C-PentaBDE. The treated PUR foams were in turn mainly used for the production of automotive and upholstery applications (e.g. automotive seating, head rests, sofas, mattresses etc.).

Other applications and finished articles containing or reported to be possibly contaminated with C-PentaBDE are Polyvinylchloride (PVC), Epoxy Resins, Unsaturated Polyesters (UPE), Rubber, Paint/Lacquers, Textiles, Hydraulic oils and possibly other. Considering the minor amounts of C-PentaBDE used in all these applications (total ~5 %), the diffuse distribution of articles and limited data availability this study will not further elaborate on these uses. When considering imports of articles containing C-PentaBDE into Europe, it has to be acknowledged that particularly the imports of C-PentaBDE in printed circuit boards may be underestimated. However, due to the lack of relevant data it is also in this case not possible to go further into detail.

Therefore, this study exclusively focuses on C-PentaBDE treated PUR foams and associated applications and finished articles (i.e. automotive and upholstery applications).

5.2.2 C-OctaBDE

The main historic use of C-OctaBDE was in Acrylonitrilebutadiene-styrene (ABS) polymers. Around 95 % of C-OctaBDE supplied in the EU was used in ABS (globally ~70 %). Thereby, C-OctaBDE was typically added at concentrations between 10-18 % by weight. The ABS in turn was mainly used for housings/casings of Electrical and Electronic Equipment (EEE), typically office equipment and business machines.

Other minor uses (5 %) were High Impact Polystyrene (HIPS), Polybutylene Terephthalate (PBT), Polyamide Polymers, with typical concentrations between 12-15 %. Other possible uses found in literature were nylon, low density polyethylene, polycarbonate, phenol-formaldehyde resins, unsaturated polyesters, adhesives and coatings.

Therefore, the main focus of this study will be on the use of C-OctaBDE in EEE, in particular on the ABS housings/casings of office equipment and business machines.

5.2.3 PFOS and its derivatives

In the past, perfluorooctane sulfonic acid and its derivatives (PFOS) have been used for a wide range of products and processes. Among these are uses as pesticides, as plumbing fluxing agent, in medical applications and devices, in flame retardants, coatings and coating additives, adhesives as well as uses in rubber and plastics, in upholstery in the leather industry as well as in the carpet industry.

At present, PFOS is used in the metal plating industry, as content in hydraulic fluids in the aviation industry, in the photographic industry and the semiconductor industry.

Another large source of PFOS is sewage sludge. This pathway of PFOS is considered as one of the most important secondary sources of PFOS. As more and more industries stop or reduce the use of PFOS, however, sewage sludge accumulates not only PFOS which is currently used, but also PFOS which has been emitted in the past. Therefore, this sector has been investigated in detail.

PFOS uses in the metal plating, photographic as well as use in hydraulic fluids in the aviation industry have been considered of further relevance. In case of the photographic industry there has been a continuous reduction of the use of PFOS but due to the existing storages of pictures in households or x-ray pictures in hospitals, this industry generates a relevant PFOS containing waste stream from the past.

Of the past uses upholstery and carpets have been further investigated since they play a major role due to their long life time regarding PFOS entering the waste stream. Concerning the use of PFOS in fire fighting foams, it has to be mentioned that by 2011 stocks of fire fighting foams containing PFOS will be eliminated. However, according to the large amounts of PFOS contained in fire fighting foams, this sector has been investigated as well.

Industries with products having a shorter lifetime, such as textiles, paper and cardboards do not influence the current waste stream anymore, even if they still contribute to current and future PFOS emissions in particular as releases from waste disposals. Also industries which used PFOS during their

production processes with end-products not including PFOS, such as the mining industry, do not pose a current input of PFOS to waste streams and therefore have been not considered in the mass and waste flows.

5.2.4 PeCB

PeCB is not being intentionally produced or used anymore within the UNECE regions, including Europe. It has been formally used in small quantities together with PCBs in electrical equipment, as feedstock/intermediate in production of the pesticide quintozene, as flame-retardant, pesticide and in dyestuff carriers. From these historic intentional uses only application of old electrical equipment and use of quintozene might be of relevance within Europe, but have not been further investigated since regulations concerning these uses are already in place: (1) According to Directive 96/59/EC on the disposal of PCBs/PCTs PCB containing equipment and has to be disposed of by the end of 2010. Relevant are all mixtures containing more than 0.005% of PCBs (see articles 3 and 4 Directive 96/59/EC); (2) the quintozene production process has been changed long ago to manufacturing methods not using PeCB; (3) quintozene was banned in Europe in June 2002 and no stockpiles of quintozene except for Lithuania have been reported. For further details on use and production see chapter 6.4.1.

The most relevant source of PeCB production and emission is unintentional production and connected releases. As main components incineration and combustion processes of different wastes and materials have been identified, i.e. municipal solid waste incineration, hazardous waste incineration, coal combustion and domestic burning of fuels and mixed waste. Further, also waste water treatment has been considered as relevant.

A much smaller source of PeCB is represented by industrial processes in the metallurgic, chemical and energy sector. However, it has to be mentioned that most industrial processes use incineration or combustion processes, e.g. industrial production of steel and iron, which according to EUROSTAT data from 2010 approximately uses more than 50% of the coal finally consumed within Europe. For this reason the industrial sectors metallurgy and energy are included in the PeCB mass and waste flow resulting from coal combustion. According to Bailey et al. the share of PeCB unintentionally produced in the sector of chemical industry is very low in comparison to incineration and combustion processes. In a risk management strategy for PeCB, Environment Canada does not mention chemical industry as relevant source [Environment Canada 2005; Bailey et al. 2009]. Accordingly, the present study does not consider chemical industry as a relevant sector for PeCB.

To conclude, the focus of this study is on the relevant processes, i.e. municipal solid waste incineration, hazardous waste incineration, coal combustion, waste water treatment and domestic burning of fuels and mixed waste.

5.2.5 SCCPs

At the moment neither natural occurrence nor unintentional production of SCCP are known. All existing SCCP is therefore anthropogenic. Regarding the UNECE CLRTAP POPs protocol shall be eliminated except for the use as fire retardant in conveyor belts in the mining industry and in dam sealants. At EU level the

SCCP concentration in leather tanning and in the metal industry shall not exceed 1 % of SCCP, which is almost equal to a ban. For all other uses such as rubbers, sealants, adhesives, paints, varnishes and textiles there are no restrictions. However, there has been a dramatically reduction in the total annual sales amounts of SCCP within the EU 27 from about 13,000 t/y in 1994 down to less than 1,000 t/y in 2009. [RPA 2010]

This reduction is mainly based on the restriction regarding the use in the metal plating industry, but also other sections reduced their SCCP consumption dramatically.

At the moment, the rubber industry is the main application of SCCP followed by the sealants and adhesives section and by the paints and varnishes section. The textile industry covers only a small fraction of the overall used amount of SCCP. Regarding the waste stream all of the mentioned sections are included in a similar ratio as their current use. Also leathers which have been impregnated with SCCP in the past can enter the waste stream due to their long lifetime.

5.2.6 HBCD

HBCD is used as an additive flame retardant. At the present, the main downstream uses of HBCD are in the polymer and textile industries. HBCD can be used on its own or in combination with other flame retardants (e.g. antimony trioxide and DecaBDE). HBCD 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.

Examples of EPS end-products containing HBCD are insulation boards for construction in general, for indoor and outdoor building constructions, against frost heaves of road and railway embankments and for transport vehicles as well as packaging material in general excluding food packaging. XPS end-products are used for the same applications. HIPS end-products are applied in electrical and electronic parts, e.g. housings for VCR (video cassette recorder), EEE (distribution boxes for electrical lines) and video cassette housings. Further, HBCD can be used as textile coating agent in polymer dispersion on cotton or cotton/synthetic blends for upholstery fabrics, e.g. residential and commercial furniture and transportation seating, bed mattress ticking, draperies and wall coverings, interior textiles, e.g. roller blinds, and automobile interior textiles. Other minor uses of HBCD are in polypropylene (PP), in styrene-acrylonitrile copolymer resins (SAN resins), in polyvinylchloride (PVC) as well as in unsaturated polyesters, adhesives and coatings, whereas it is unclear whether these uses are relevant within the EU.

The most important use of EPS and XPS is in the construction sector with approximately 90 % of the HBCD consumption followed by the use of EPS and XPS for other uses such as insulation boards for vehicles, packaging material and other minor uses with approximately 6 % of the total consumption. HBCD is used in slightly higher quantities for XPS than for EPS (ratio ~ 52:48). For HIPS and in textiles, HBCD use represent each approximately 2 % of the EU consumption.

In summary, the focus of this study is exclusively on EPS and XPS for the construction sector, EPS/XPS outside the construction sector, HIPS for EEE and polymer dispersions for textile products.

5.2.7 PCP

Pentachlorophenol (PCP) is investigated together with the substances sodium pentachlorophenoxide (NaPCP) and Pentachlorophenyl laureate (PCPL). In the past the PCP was used for many applications as anti-mildew mothproofing and antifungal agent, for wood impregnation and in different industries as leather, textile, mineral oil, glue, paint or pulp industry. Due to national legislation, EU legislations as well as conventions the use of PCP in the EU 27 has stopped. However due to long product life times waste containing PCP can still enter the waste stream. The relevant sources of waste regarding PCP are nowadays only from the wood impregnation industry and the textile industry. PCP and Na PCP have been used in the wood impregnation industry whereas PCPL has been used in the textile industry. About 71 % of all PCP in the waste is from impregnated wood and the rest (29 %) from the textile industry.

5.2.8 HCBd

Hexachlorobutadien (HCBd) has been historically used as solvent for rubber and other polymers, in heat transfer fluids, as transformer liquid, as hydraulic fluid, for the recovery of "snift" (chlorine containing gas in chlorine plants) and washing liquor for removing certain volatile organic compounds (e.g. hydrocarbons) from gas streams. Further, it has been applied as intermediate for the manufacture of flour-containing lubricants and rubbers, as gyroscopic liquid, as biocide as well as fumigant in some EU countries (i.e. France, Italy, Greece and Spain) [UBA 2006]. Other historical uses might include the use in the production of aluminum and graphite rods [UNECE 2007; UBA 2006].

From these historic intentional uses only application of old electrical equipment might be of relevance within Europe, but have not been further investigated for the same reasons as indicated for the substance PeCB.

At present there is no intentional use of HCBd in Europe since many years [UNECE 2007], and also intentional production has stopped [UBA 2006]. HCBd is mainly formed as unintentional by-product during several industrial processes, but the available information basis is very scarce. However, production of tri- and tetrachloroethene and tetrachloromethane has been indicated as the most important source. Production of primary magnesium has been identified to belong to the historic uses by now, since its production within EU27 has stopped recently. Unintentional production and releases from the plastic industry may be relevant, but specific data is not available. The same case has been identified for the sector of hazardous waste incineration and sewage sludge resulting from treatment of industrial waste water. Concluding, more information has to be delivered by the industry.

5.2.9 PCN

Polychlorinated naphthalenes (PCNs) have historically been used until 1989 for several purposes, which are cable insulation, wood preservation, additive in engine oils, electroplating masking compounds, dye production, capacitor and refracting index oils, technical PCB formulations and casting material.

No information is available on production and use of PCNs at the present and therefore it is assumed that no intentional production or uses is occurring within Europe. However, it has been reported that

PCN can be formed unintentionally during various thermal processes [UNECE 2007; Haskoning 2002]. As PCNs exhibit similar formation properties as PCDD/Fs, unintentional production during incineration processes as municipal solid waste incineration, hospital waste incineration, domestic burning, or different metal processing steps such as secondary copper production, secondary aluminium production, magnesium production as well as iron sintering and electrical arc furnace processes for iron production are of relevance. Further, the accumulation in sewage sludge from diffuse sources has been identified and taken into account.

5.3 Investigated sectors

Sectors included so far into the mass flow analyses have been selected depending on their relevance for the mass flows according to available information. The following sectors have been identified as potentially relevant source for new POPs and have been investigated in order to assess the relative importance of the individual sectors in relation to the overall mass flow:

1. C-PentaBDE
 - use in PUR foams for automotive applications
 - use in PUR foams for upholstery applications
2. C-OctaBDE
 - use in housing of Electrical and Electronic Equipment
3. PFOS and its derivatives
 - metal plating
 - photographic industry
 - hydraulic fluids for aviation
 - fire fighting foams
 - past use in leather for upholstery products
 - past use in carpets made of synthetic fibres
 - accumulation in sewage sludge
4. Generation of PeCB during
 - municipal solid waste incineration
 - hazardous waste incineration
 - power production from coal
 - waste water treatment
 - domestic burning
5. SCCPs
 - Rubber industry
 - Sealant and adhesives
 - Paint and varnishes
 - Textile industry
 - Leather industry
 - Sewage sludge
6. HBCD
 - use in Expandable Polystyrene (EPS) for the construction industry
 - use in Extruded Polystyrene (XPS) for the construction industry

- use in EPS/XPS outside the construction sector
 - use in High Impact Polystyrene (HIPS) for EEE
 - use in polymer dispersions for textiles
7. PCP
- Wood impregnation
 - Textile industry
8. HCB
- chemical industry – production of chlorinated solvents
 - plastic industry
 - non-ferrous metal industry-magnesium production
 - hazardous waste incineration
 - sewage sludge
9. PCN
- municipal solid waste incineration
 - hospital waste incineration
 - domestic burning
 - sewage sludge
 - non-ferrous metal industry-secondary copper production
 - non-ferrous metal industry-secondary aluminium production
 - iron and steel production-sinter plants and electric arc furnaces.

Table 5-2 shows which sectors are considered relevant in the scope of the present study so far and which sectors are considered of low relevance due to the low amounts of new or candidate POPs concerned. All mass flows of sectors considered relevant are analysed in detail in the present report and are reflected in the results of the mass flow analysis. All sectors considered not relevant are discussed in chapter 6 but are not analysed in detail and are therefore not reflected in the results of the mass flow analyses.

Table 5-2: Overview of sectors and their relevance in the scope of the present study

Sector	Substance concerned	Relevance of the sector according to the results of the mass flow analysis	
		Considered relevant	Not considered relevant
Use of PUR foams in upholstery and furniture	C-PentaBDE	♦	
Use of PUR foams in automotive applications	C-PentaBDE	♦	
Use of C-PentaBDE in epoxy resins in imported printed circuit boards	C-PentaBDE		♦ (sector may be underestimated)
Use of C-PentaBDE in Polyvinylchloride (PVC), unsaturated polyesters, rubber, paint/lacquers, textiles, hydraulic oils	C-PentaBDE		♦
Use of C-OctaBDE in Acrylonitrilebutadiene-styrene (ABS) polymers for housings/casings of EEE	C-OctaBDE	♦	
Use of C-OctaBDE in High Impact Polystyrene (HIPS), Polybutylene Terephthalate (PBT), Polyamide Polymers and other uses such as nylon and low density polyethylene, polycarbonate, phenol-formaldehyde resins, unsaturated polyesters	C-OctaBDE		♦

Sector	Substance concerned	Relevance of the sector according to the results of the mass flow analysis	
		Considered relevant	Not considered relevant
Metal plating	PFOS	♦	
Photographic industry	PFOS	♦	
Semiconductor industry	PFOS		♦
Hydraulic fluids for aviation	PFOS	♦	
Past use in leather for upholstery products	PFOS	♦	
Past use in carpets made of synthetic fibres	PFOS	♦	
Sewage sludge	PFOS	♦	
Past use in short lifetime products and at low use level (e.g. in medical devices, insect baits, coatings and coating additives, rubber and plastics)	PFOS		♦
Past use in industry processes without relevant contamination of products	PFOS		♦
Municipal solid waste incineration including incineration of biomass	PeCB	♦	
Hazardous waste incineration	PeCB	♦	
Power production (coal) including power production in iron and steel industry	PeCB	♦	
Domestic burning	PeCB	♦	
Sewage sludge (waste water treatment)	PeCB	♦	
Chemical industry	PeCB		♦
Non-ferrous metal industry (aluminum, secondary copper, magnesium)	PeCB		♦
Rubber industry	SCCPs	♦	
Sealant and adhesives	SCCPs	♦	
Paint and varnishes	SCCPs	♦	
Textile industry	SCCPs	♦	
Leather industry	SCCPs	♦	
Sewage sludge	SCCPs	♦	
Use of Expandable Polystyrene (EPS) in construction sector	HBCD	♦	
Use of Extruded Polystyrene (XPS) in construction sector	HBCD	♦	
Use of EPS/XPS outside the construction sector (e.g. packaging)	HBCD	♦	
Use of High Impact Polystyrene (HIPS) in EEE	HBCD	♦	
Use of polymer dispersions for back-coated textiles in various sectors (e.g. upholstery/furniture, transportation etc.)	HBCD	♦	
Wood impregnation	PCP	♦	
Textile industry	PCP	♦	
Chemical industry – production of chlorinated solvents	HCBd	♦ (considered relevant, but only little data available)	
Plastic industry	HCBd	♦ (considered relevant, but no data available)	
Non-ferrous metal industry - magnesium production	HCBd		♦ (production in EU has stopped)
Hazardous waste incineration	HCBd	♦ (considered	

Sector	Substance concerned	Relevance of the sector according to the results of the mass flow analysis	
		Considered relevant	Not considered relevant
		relevant, but no data available)	
Sewage sludge	HCBD	♦ (considered relevant, but no data available)	
Municipal solid waste incineration	PCN	♦	
Hospital waste incineration	PCN	♦	
Domestic burning	PCN	♦	
Sewage sludge	PCN	♦	
Non-ferrous metal industry - secondary copper production	PCN	♦	
Non-ferrous metal industry - secondary aluminium production	PCN	♦	
Non-ferrous metal industry - magnesium production	PCN		♦ (production in EU has stopped)
Iron and steel production - sinterplants and electric arc furnace	PCN	♦ (considered relevant, but no data available)	

5.4 Overview of the substance flows

The overview substance flows for the different new and candidate POPs are based on the results of the detailed mass flows in chapter 6. To give information on the relative importance of different disposal/recovery pathways they provide summarised information from all investigated "sources". Detailed information on the contribution of specific sectors to different disposal/recovery pathways is discussed in the corresponding paragraphs of chapter 6.

5.4.1 C-PentaBDE

5.4.1.1 Substance flow C-PentaBDE

Figure 5-3 shows the overall result of the detailed C-PentaBDE mass flow analyses in chapter 6.1.9 and 6.1.10. The individual streams are listed in Figure 6-11 and Figure 6-15 and are discussed in detail in the corresponding paragraphs of the C-PentaBDE mass flows in chapter 6.

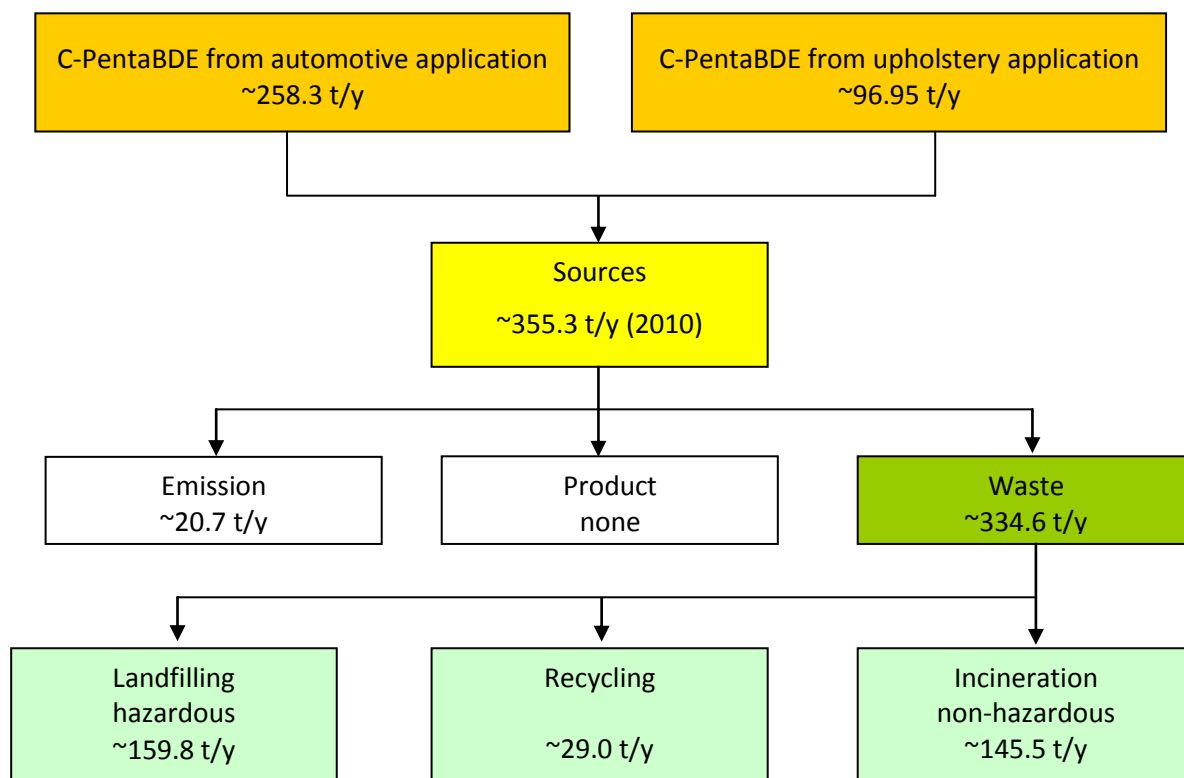


Figure 5-3: Overall mass flow from C-PentaBDE from sources to current disposal/recovery operations in the EU

By taking only the relevant congeners into account, the ~330 t/y of C-PentaBDE correspond to ~105 t/y of TetraBDE, ~200 t/y of PentaBDE, ~20 t/y of HexaBDE and ~5 t/y of HeptaBDE. A small amount is, however, contributed by other congeners of the commercial product. The following table gives an overview of the contribution of the relevant congener groups to the overall C-PentaBDE amount in automotive and upholstery applications.

Table 5-3: Overview of the contribution to the waste flow of the relevant congener groups to the amount of C-PentaBDE in automotive and upholstery applications

Commercial Mixture	Relevant congener groups			
C-PentaBDE	TetraBDE [t/y]	PentaBDE [t/y]	HexaBDE [t/y]	HeptaBDE [t/y]
Automotive 243.3 t/y	75.42	136.25	19.46	2.43
Upholstery 91.3 t/y	28.31	51.13	7.30	0.91
Total	103.73	187.38	26.76	3.34

Due to the limited data availability for a number of sectors, Figure 5-3 is partly based on assumptions and extrapolations. These are explained in the detailed mass flow analyses in chapter 6. Based on the calculations discussed in more detail in the corresponding paragraphs of the detailed mass flow for C-PentaBDE, major sources for the current flow of C-PentaBDE can be pointed out. These are the automotive sector, currently contributing with around 258.3 t/y and the upholstery/furniture sector, currently contributing with about 96.95 t/y to the total annual amount of C-PentaBDE. As shown in Figure 5-3 the overall discharge of C-PentaBDE from the investigated sources in Europe accounts for about 355.3 t/y with an overall distribution of the discharge of around 6 % as emissions (3.9 % volatile and 2 % particulate emissions) and around 94 % to waste.

The following table summarizes the emissions and discharge of C-PentaBDE from investigated sources (i.e. automotive and upholstery applications) in the EU.

Table 5-4: Emissions and discharge of C-PentaBDE from investigated sources in the EU

Sector/Activity	Emission [t/y]		Product [t/y]	Waste [t/y]	Total [t/y]
	volatile	particulate			
Automotive applications	10.10	4.97	0	243.30	258.37
Upholstery applications	3.78	1.86	0	91.31	96.95
Total	13.88	6.83	0	334.61	355.32

Figure 5-4 presents the relative distribution of environmental emissions from investigated sectors in the EU. The most important source of volatile and particulate emissions is the automotive sector, contributing with around 49 % to volatile and 24 % to particulate losses. Upholstery applications are responsible for around 18 % of the volatile emissions and 9 % of particulate losses (see Figure 5-4).

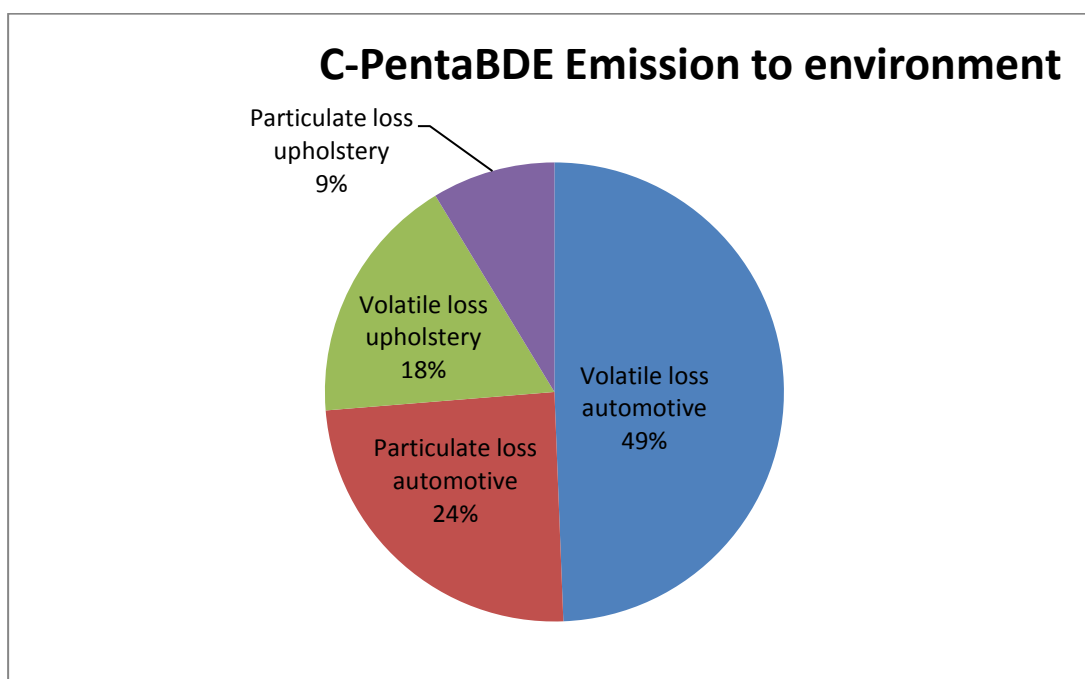


Figure 5-4: Relative distribution of environment emissions from investigated sectors in the EU

On the other hand, Figure 5-5 indicates the relative distribution of discharge of C-PentaBDE to waste from automotive and upholstery applications. It is visible that automotive applications account for the vast majority (~73 %) of the discharge of C-PentaBDE to waste.

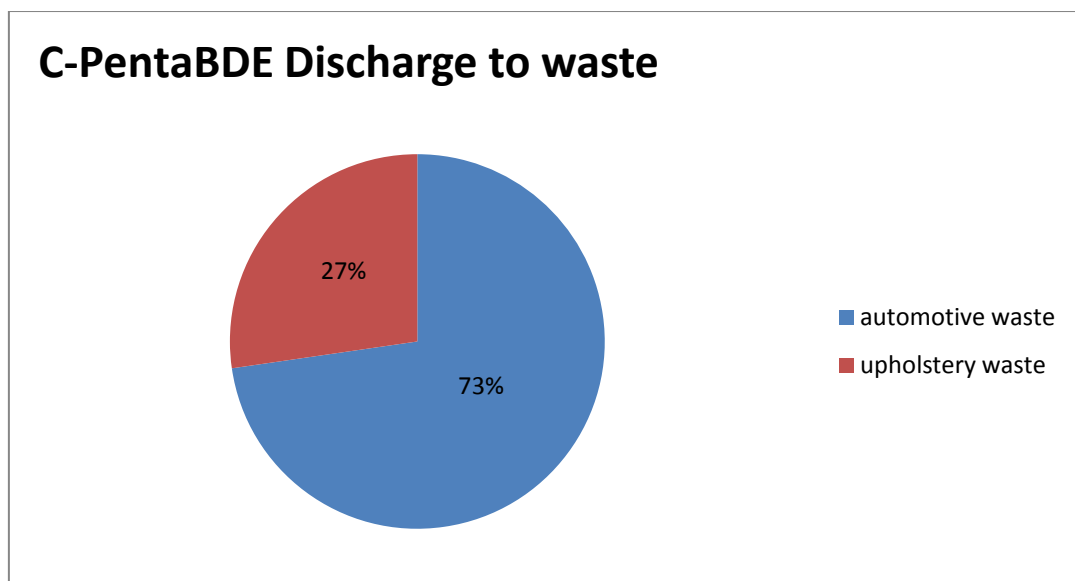


Figure 5-5: Relative distribution of discharge of C-PentaBDE to waste from investigated sectors in the EU

C-PentaBDE discharged to waste is directed to different waste management operations including disposal and recovery operations. As the environmental relevance of the different operations differs, it is of interest to trace the amounts of each “source” sector directed to specific management sectors and to distinguish the relative share of each operation in relation to the overall POP discharge to waste.

The following Table shows how the selected activities contribute to the mass flow to specific disposal/recovery operations.

Table 5-5: Quantitative contribution of C-PentaBDE from different sectors to currently used disposal/recovery operations in the EU

Disposal/Recovery	C-PentaBDE [t/y]
Landfilling	
Automotive applications	98.5
Upholstery applications	61.3
TOTAL	159.8
Incineration	
Automotive applications	98.5
Upholstery applications	18.4
TOTAL	116.9
Energy recovery	
Automotive applications	17.0
Upholstery applications	11.6
TOTAL	28.6
Recycling	
Automotive applications	29.0
Upholstery applications	-

Disposal/Recovery	C-PentaBDE [t/y]
TOTAL	29.0

The following pie chart illustrates the importance of current waste treatment options with respect to the C-PentaBDE flow.

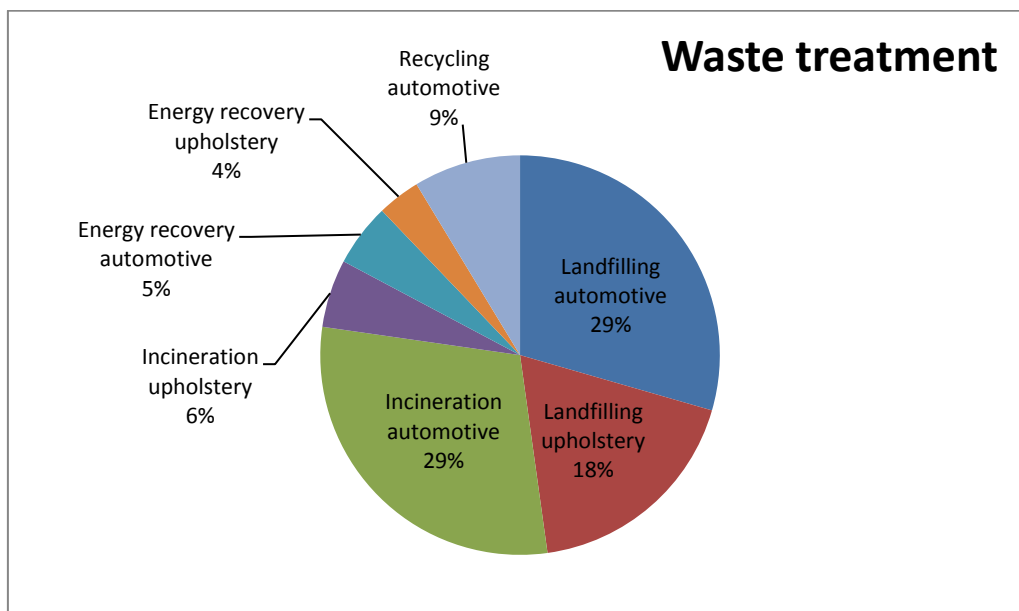


Figure 5-6: Relative importance of different waste treatment options for the C-PentaBDE flow in the EU

As illustrated above, landfilling represents the highest share of approximately 47 %, followed by incineration with around 44 % (D10 and R1) and recycling with around 9 %.

5.4.1.2 Mass flows (articles/waste) related to C-PentaBDE

While the relative contribution of industrial, public and domestic sectors on the overall POP release is interesting for the assessment of potential environmental impact and the assessment of the overall flow and the effectiveness and significance of potential regulation, important information on the potential effects impacts and implications of potential regulation can only be drawn when the amounts of concerned waste streams are known.

Therefore, the relevant waste flows have been established for each investigated sector. Here, the compilation of the related waste streams and corresponding contamination levels on the European scale are presented. The corresponding contribution to the overall discharge of C-PentaBDE is added for better orientation.

Table 5-6: Annual amounts of C-PentaBDE containing residues and the ranges of contamination with C-PentaBDE and corresponding C-PentaBDE amount in investigated sectors in the EU; * ea = estimated average

Sector	Waste	Amount [t]	Contamination [mg/kg]			C-PentaBDE discharge [t/y]	C-PentaBDE to waste [t/y]	Share of total C- PentaBDE to waste [%]
			min	ea* [mg/kg]	max			
Automotive	PUR foam from automotive application	6,082	-	~40,000	-	258.37	243.30	72.71
	Total SLF contaminated with C- PentaBDE	655,873	-	~371	-			
Upholstery	PUR foam from upholstery	2,403	20,000	~38,000	54,500	96.95	91.31	27.29
	Total upholstery waste contaminated with C- PentaBDE	5,215,540	-	~17.5	-			
TOTAL		5,871,413				355.3	334.61	100

5.4.2 C-OctaBDE

5.4.2.1 Substance flow C-OctaBDE

Figure 5-7 shows the overall result of the detailed C-OctaBDE mass flow analyses in chapter 6.2.9. Electrical and Electronic Equipment (EEE) has been identified as the only relevant source of C-OctaBDE (~131 t/y). The overall discharge of C-OctaBDE from the investigated source in Europe accounts for approximately 128 t/y, with an overall distribution of the discharge of around 2.3 % as emissions and 97.7 % to waste.

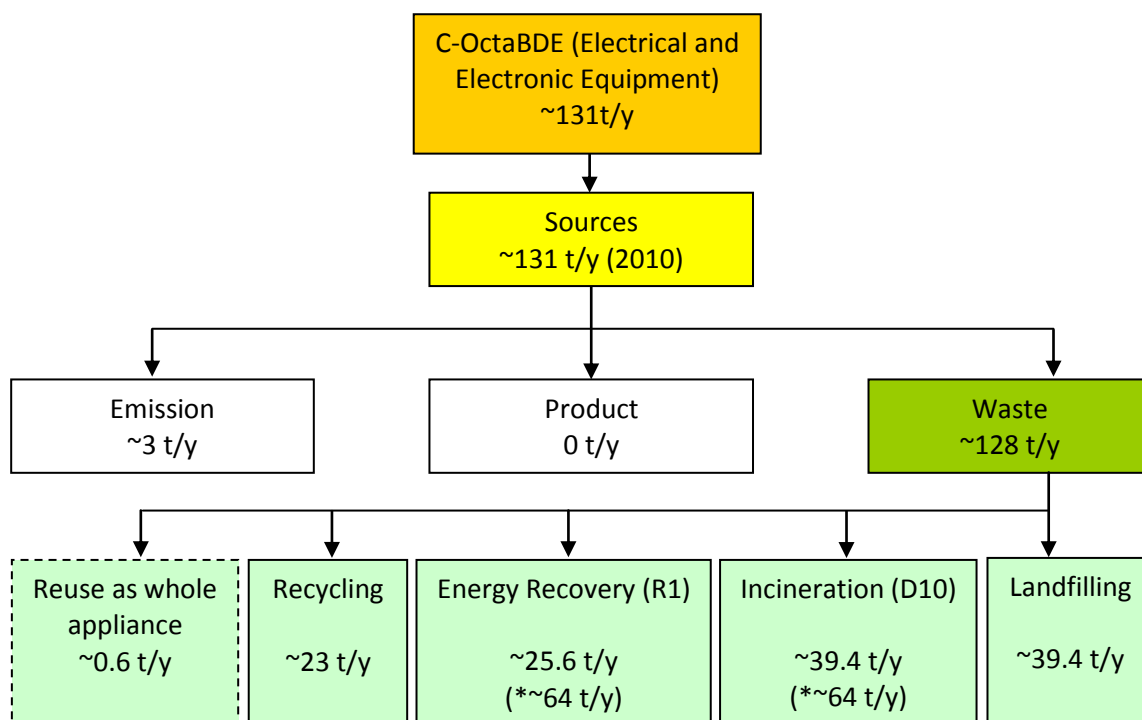


Figure 5-7: Overall mass flow from C-OctaBDE from sources to current disposal/recovery operations in the EU (*Scenario 2: C-OctaBDE containing WEEE plastic successfully separated and incinerated, R1/D10)

Due to the limited data availability, Figure 5-7 is partly based on assumptions and extrapolations. These are explained in the detailed mass flow analyses in chapter 6. For more details please review the corresponding paragraphs of the C-OctaBDE mass flow in chapter 6.

The ~128 t/y of C-OctaBDE correspond to ~8 t/y of PentaBDE, ~8 t/y of HexaBDE and ~45 t/y of HeptaBDE taking the EEE sector into account. Only the relevant congeners have been considered. A significant difference, however, results due to great variations in composition of different C-OctaBDE products. For the calculation an average composition has been applied. Please see Table 6-9 for more detailed information.

Table 5-7: Overview of the contribution of the relevant congener groups to the amount C-OctaBDE for the EEE sector

Commercial Mixture	Relevant congener groups			
C-OctaBDE	TetraBDE [t/y]	PentaBDE [t/y]	HexaBDE [t/y]	HeptaBDE [t/y]
EEE 128 t/y	-	8.00	7.87	45.31

The following table summarizes the emissions and discharge of C-OctaBDE from the investigated source (i.e. EEE) in the EU.

Table 5-8: Emissions and discharge of C-OctaBDE from the investigated source in the EU

Sector/Activity	Emission [t/y]		Product [t/y]	Waste [t/y]	Total [t/y]
	volatile	particulate			
Electrical and Electronic Equipment	0.7	2.6	0	128	131.3
Total	3.3		0	128	131.3

It can be observed that volatile emissions to the environment contribute with 0.7 t/y and the particulate emissions with 2.6 t/y to the overall emissions of 3.3 t/y from EEE.

C-OctaBDE discharged to waste (~128 t/y) is directed to different waste management operations including disposal, energy recovery, recycling and the direct reuse of a whole appliance. As the environmental relevance of the different operations differs, it is of interest to trace the amounts directed to specific management sectors and to distinguish the relative share of each operation in relation to the overall POP discharge to waste.

The following Table shows how the selected activities contribute to the mass flow to specific disposal/recovery operations.

Table 5-9: Quantitative contribution of C-OctaBDE to currently used disposal/recovery operations in the EU

Disposal/Recovery	C-PentaBDE [t/y]	Comment
Landfilling		
WEEE	39.4	
Incineration		
WEEE	39.4/*64.0	*C-OctaBDE containing WEEE successfully separated and incinerated
Energy recovery		
WEEE	25.6/*64.0	* C-OctaBDE containing WEEE successfully separated and incinerated
Recycling		
WEEE	23.0	
Reuse as a whole appliance		
WEEE	0.6	
TOTAL	128/*128	* C-OctaBDE containing WEEE successfully separated and incinerated

The following pie chart illustrates the importance of current waste treatment options with respect to the C-OctaBDE flow.

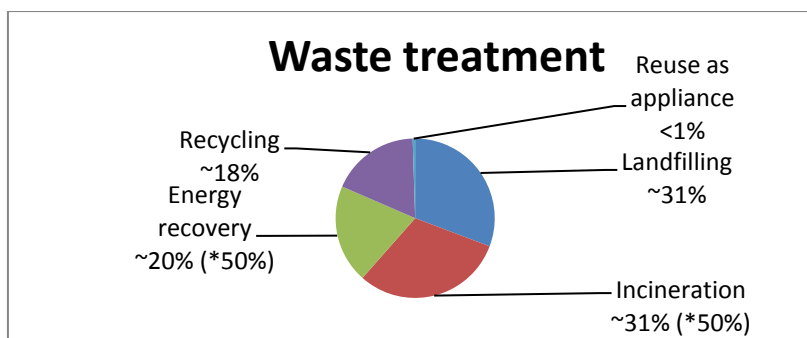


Figure 5-8: Relative importance of different waste treatment options for the C-OctaBDE flow in the EU (*C-OctaBDE containing WEEE successfully separated and incinerated, R1/D10) (*Scenario 2: C-OctaBDE containing WEEE plastic successfully separated and incinerated, R1/D10)

As illustrated above, disposal (landfilling and incineration without energy recovery) represents the highest share of approximately 62 % followed by energy recovery with about 20 %, recycling (~18 %) and the reuse of the appliances itself with less than 1 % of the total share. In case the C-OctaBDE containing WEEE plastic is successfully separated from the non-contaminated fraction it is assumed that the contaminated waste is to equal amounts incinerated with and without energy recovery (scenario 2).

5.4.2.2 Mass flows (articles/waste) related to C-OctaBDE

While the relative contribution of industrial, public and domestic sectors on the overall POP release is interesting for the assessment of potential environmental impact and the assessment of the overall flow and the effectiveness and significance of potential regulation, important information on the potential effects impacts and implications of potential regulation can only be drawn when the amounts of concerned waste streams are known.

Therefore, the relevant waste flow has been established. Besides, the corresponding contamination levels on the European scale are presented. The contribution to the overall discharge of C-OctaBDE is added for better orientation.

Table 5-10: Annual amounts of C-OctaBDE containing residues and the ranges of contamination with C-OctaBDE and corresponding C-OctaBDE amount in the EU; *ea = estimated average

Sector	Waste	Amount [t]	Contamination [%]			C-OctaBDE discharge [t/y]	C-OctaBDE to waste/product [t/y]	Share total C-OctaBDE to waste/products [%]
			min	ea* [mg/kg]	max			
EEE	WEEE contaminated with C-OctaBDE	732	100,000	~175,000	220,000	131.3	128.0	100
	WEEE, category 3 and 4 not contaminated with C-OctaBDE	104,121	-	-	-	-	-	-
TOTAL	WEEE total (incl. category 3 & 4)	104,853		1,230		131.3	128.0	100

5.4.3 Substance flow PFOS

5.4.3.1 Substance flow PFOS

The following figure shows the overall result of the detailed PFOS mass flow analysis. The individual streams from each activity are listed in chapter 6.3.7 to 6.3.13 and are discussed for each activity in the corresponding detailed PFOS mass flow. Sectors not covered with respect to the PFOS mass flow are discussed in chapter 5.2.3. Based on the information presented it can be stated that the most significant processes are covered in the total mass flow shown in following figure:

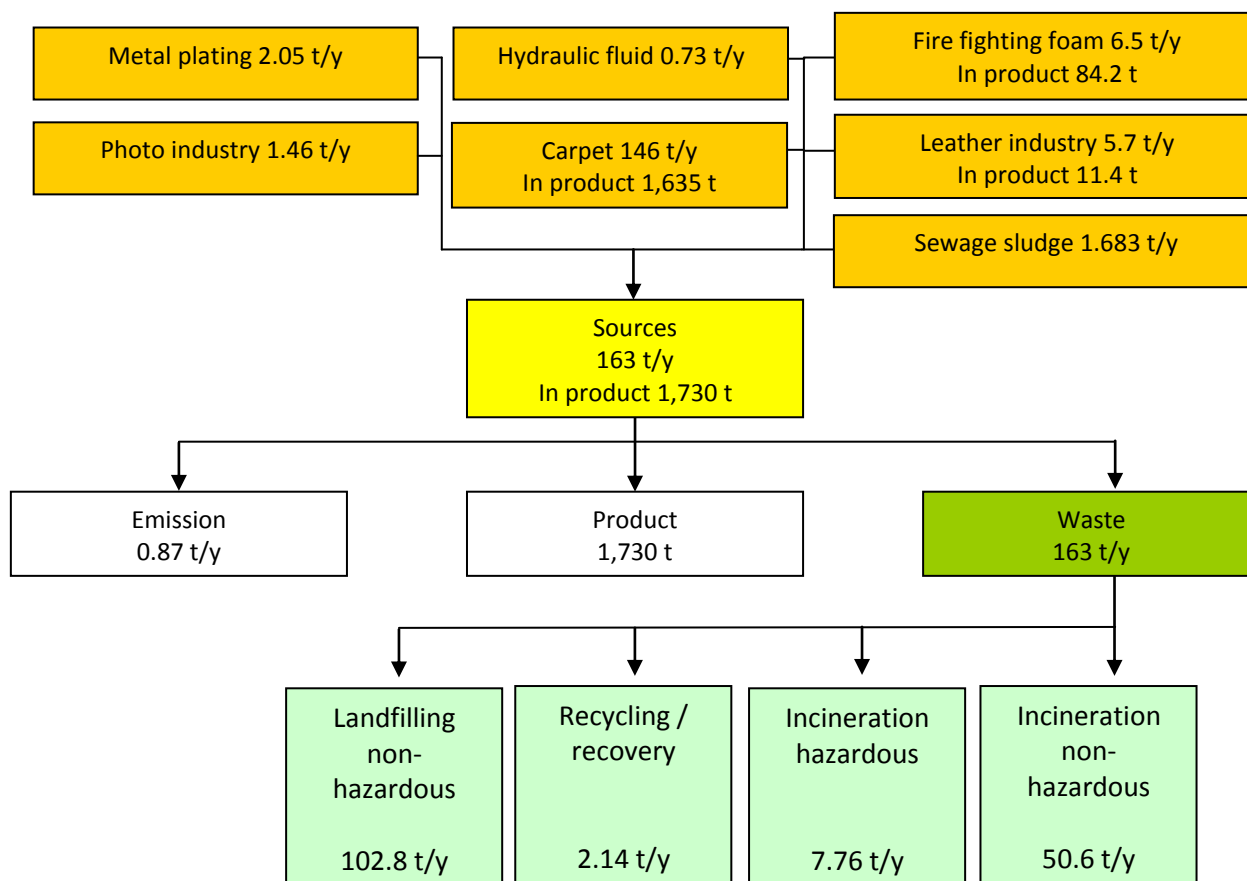


Figure 5-9: Overall mass flow from PFOS from sources to current disposal/recovery operations in the EU

“In Product” represents the existing stock of the substance in product in use. It has to be taken into account that due to limited data for a number of sectors and countries the mass flow is partly based on assumptions and extrapolation. However, the mass flow can give an impression on the dimension of the PFOS flow to the environment and to waste and on the relative contribution of major relevant sectors related to PFOS. The term temporary storage in this context means products which do not enter the waste flow in 2010 but which are expected to become waste in the future (stocks of existing products).

Table 5-11 shows the contribution of the different sectors to the European PFOS flow to emissions, products and waste.

Table 5-11: Contribution of the relevant sectors in the EU 27 to PFOS in emissions, products and waste

Sector/Activity	Emission [t/y]	Product [t/y]	Waste [t/y]	Sum [t/y]
Metal plating	0.371	0	1.68	2.05
Photo industry	0.028	0	1.43	1.46
Hydraulic fluid	0.014	0	0.72	0.73
Fire fighting foams	0.454	84.2	6.05	90.74
Leather industry	0	11.4	5.71	17.12
Carpet industry	0	1,635.2	146.00	1,781.19
Sewage sludge	0	0	1.68	1.68
Total	0.87	1,730.842	163.26	1,894.97

As illustrated the overall discharge of PFOS from the investigated sources in Europe accounts for about 164.5 t/y. Almost all of the PFOS released is discharged to waste (99.95%) whereas only a small fraction (0.05%) reaches the environment. In conclusion it can be stated that the sources of PFOS are dominated by wastes resulting from the sector carpets (88.7%), followed by fire fighting foams (~4.23%) and the leather industry (~3.5%). These sectors are responsible for nearly 100 % of the PFOS emissions and discharges as described in Figure 5-10, whereas contributions of the other sectors are of minor importance.

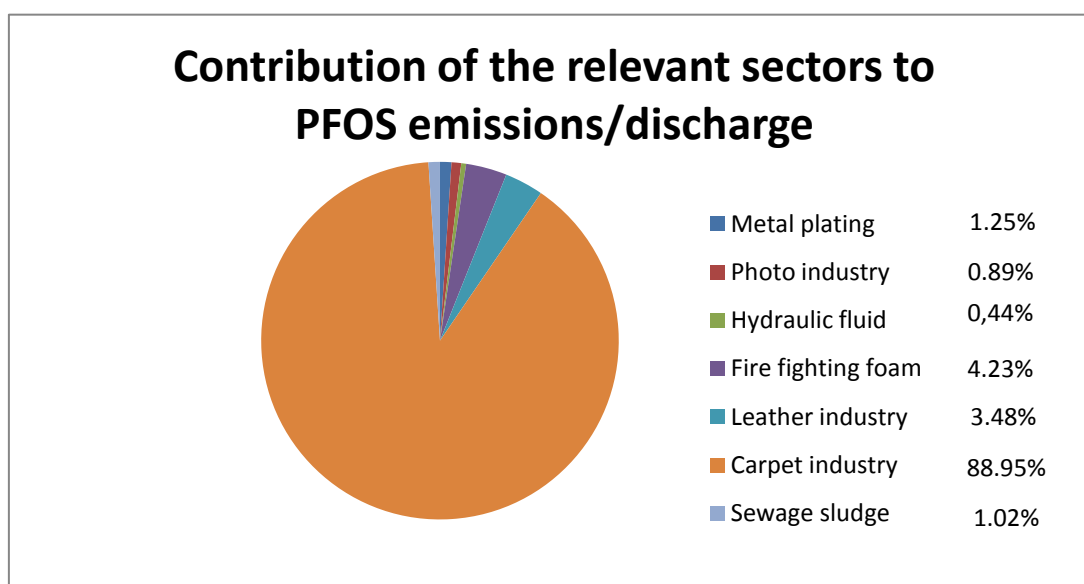


Figure 5-10: Contribution of the relevant sectors to the EU 27 to PFOS emissions to environment and discharge to waste

In order to illustrate the relative importance of different sectors on environmental emissions to air, soil and water as well as discharge to waste, the shares of the investigated sectors are illustrated in the following pie charts.

As illustrated in Figure 5-11 below, the most important source for emission to the environment are fire fighting foams with ~52 % followed by metal plating (~43 %). Also relevant sectors contributing to emissions in the environment are further the photo industry with 3 % and hydraulic fluids with 1.7 %. The leather and carpet sector do not contribute to PFOS emissions to the environment. Losses during lifetime have not been considered due to the lack of data. The given values are therefore representing a

worst case scenario for the waste stream. PFOS is concentrated in sewage sludge during the waste water treatment. Emissions to the environment occur to water (PFOS which is not concentrated in sewage sludge) and if the sewage sludge is used for agricultural purposes. About 5.3 million tonnes of sewage sludge represent approximately 770 kg of PFOS which would be as much as 88 % of the sum of all PFOS emissions from the other industries. These emissions resulting from the waste management options, however, are not illustrated in Figure 5-11 (see also Figure 5-1).

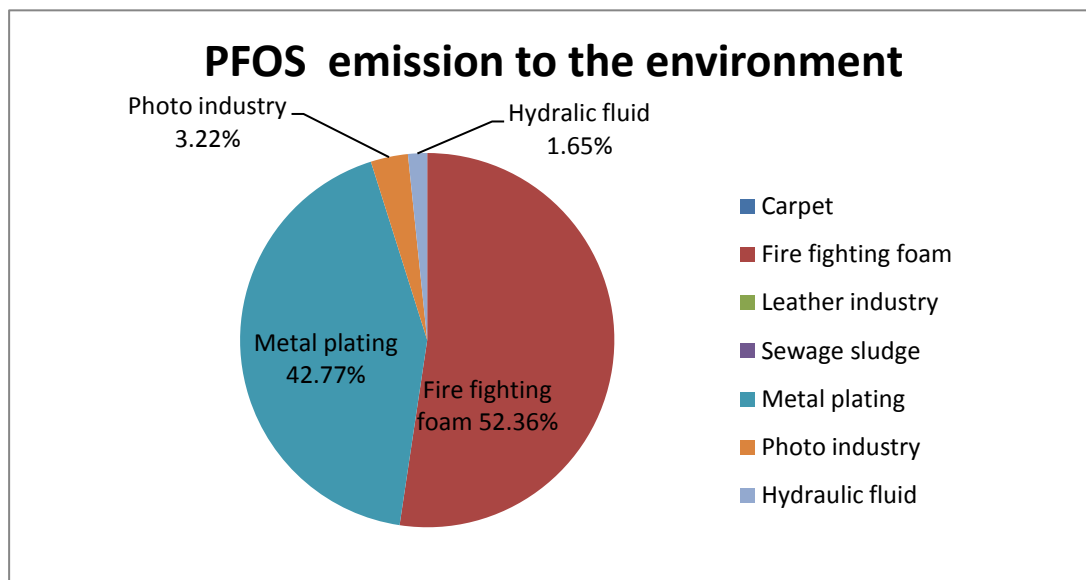


Figure 5-11: Relative distribution of PFOS emissions to the environment from investigated sectors in EU 27

Unlike emissions to the environment discharges to waste are dominated by the carpet sector (94.05 %), followed with great distance by fire fighting foams (4.77 %) and the leather industry (0.90 %). PFOS wastes arising from the other sectors are of minor relevance as shown in Figure 5-12.

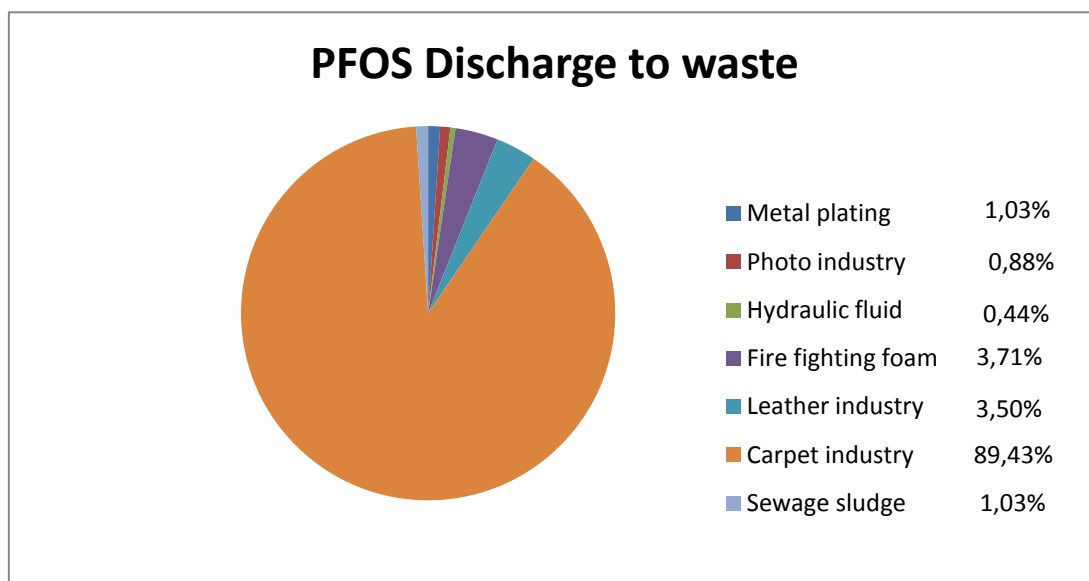


Figure 5-12: Relative distribution to PFOS waste from investigated sectors in EU 27

PFOS discharged to waste is directed to different waste management operations including disposal and recovery operations. As the environmental relevance of the different operations differs, it is of interest to trace the amounts of each “source” sector directed to specific management sectors and to distinguish the relative share of each operation in relation to the overall PFOS discharge to waste. Table 5-12 illustrates how the selected activities contribute to the PFOS flow to specific disposal/recovery operations, whereas Figure 5-13 illustrates the importance of current waste treatment options with respect to the PFOS mass flow.

Table 5-12: Quantitative contribution of PFOS from the different sectors to currently used disposal/recovery operations in EU 27

Treatment / Activity	kg/y	Comment
Non hazardous waste incineration (NHWI)		
Photographic industry	102.6	7,700,000 t x-ray waste
Photographic industry	132.1	30,500t other film
Photographic industry	130.0	30,000 t picture waste
Leather industry	1,152.9	480,000 t upholstery
Leather industry	724.8	300,000 t upholstery
Carpets	29,491.9	390,000 t carpets
Carpets	18,541.9	250,000 t carpets
Sewage sludge	1,906.4	1,900,000 t sludges with low contamination
Sewage sludge	1,003.4	100,000 t sludges with high contamination
Total	50,567.2	
Hazardous waste incineration (HWI)		
Metal plating	336.2	350 t activated carbon filter
Metal plating	336.2	18 t ion exchange resin
Photographic industry	117.0	0.92t solid residue recycling x-ray
Photographic industry	19.5	20 t activated carbon x-ray
Photographic industry	156.0	7,700 t desilvered solution x-ray
Photographic industry	28.1	0.07 t liquid photograph fluids
Aviation industry	715.4	715 t hydraulic fluid
Fire fighting foams	6,049.1	600 t fire fighting foam
Total	7,757.6	
Landfill for non hazardous waste (LNHW)		
Photographic industry	209.4	15,600 t x-ray waste
Photographic industry	269.5	62,200 t other films
Photographic industry	265.0	61,200 t picture waste
Leather industry	3,829.7	1,600,000 t upholstery
Carpets	97,965.6	1,300,000 t carpets
Sewage sludge	171.5	1,700,000 t sludges with low contamination
Sewage sludge	90.3	90,000 t sludges with high contamination
Total	102,800.8	
Recycling / recovery (RR)		
Metal plating	378.1	100,000 t chromium sludge
Metal plating	298.7	1,200 t vacuum distillation concentrate
Metal plating	335.1	1,100 t activated carbon from treatment
Sewage sludge	737.8	7,400,000 t sludges with low contamination
Sewage sludge	388.3	400,000 t sludges with high contamination
Total	2,138.0	
In used products		
Fire fighting foams	84,234.1	8,400 t fire fighting foams stored
Leather industry	11,414.8	4,800,000 t upholstery
Carpets	1,635,192.8	21,700,000 t carpets
Total	1,730,841.6	
Total without PFOS in used products	163,264	
Overall total	1,894,105.3	

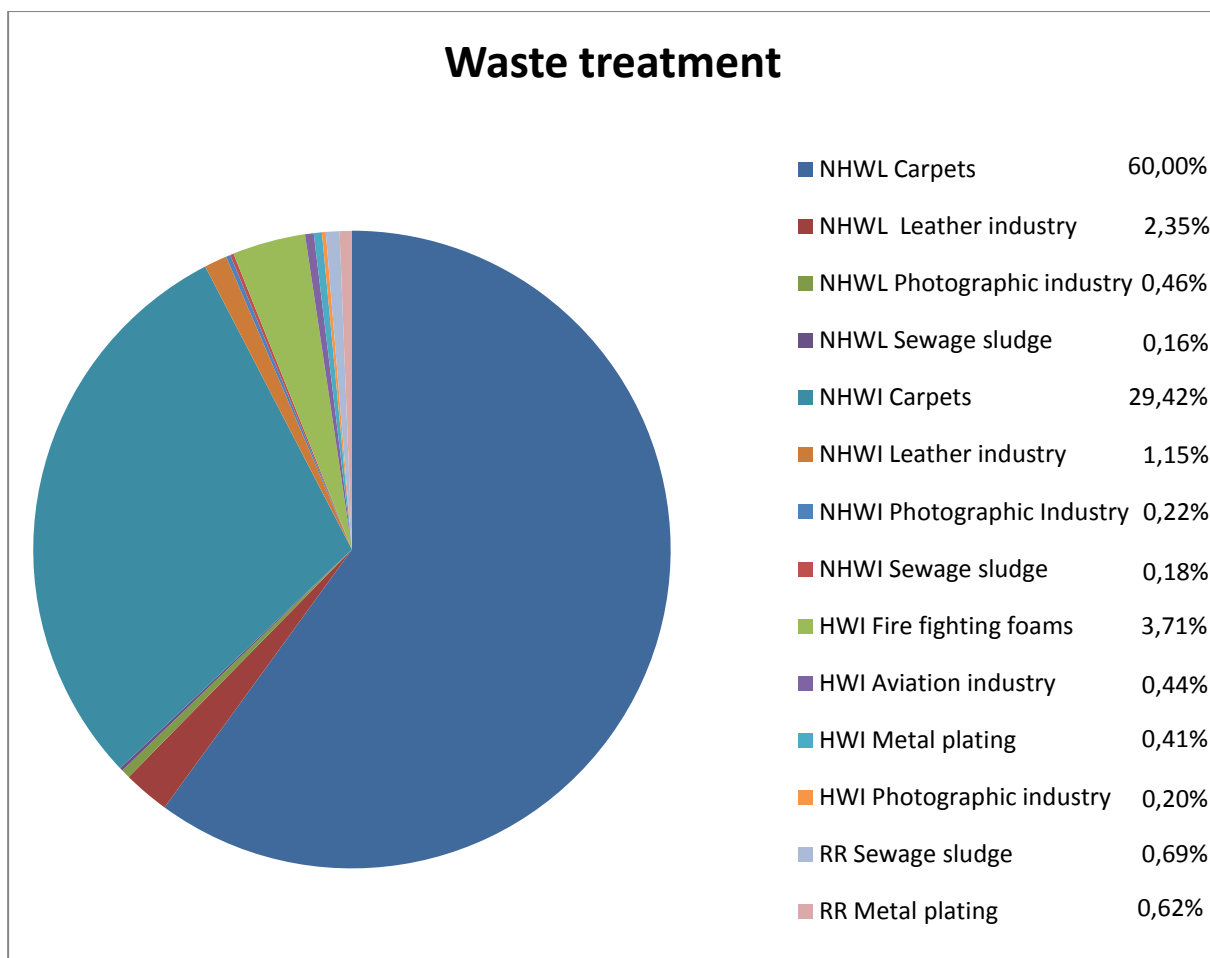


Figure 5-13: Relative importance of different waste treatment options and uses in products with respect to the PFOS mass flow in EU 27

The majority of PFOS waste is currently disposed of at landfills for non-hazardous waste (63%) from within this treatment carpet wastes amounts to about 97%. The second typical treatment of PFOS containing waste is non-hazardous incineration (31%). The major waste is again carpet with a share of about 95 %. 4.75 % of all PFOS containing waste is destroyed in hazardous waste incineration plants with fire fighting foams being the most typical PFOS containing waste types for this treatment with 78%. Only 1.31 % of all PFOS containing waste is currently reused or recycled

5.4.3.2 Mass flows (articles/waste) related to PFOS

While the relative contribution of industrial, public and domestic sectors on the overall POP release is interesting for the assessment of potential environmental impact and the assessment of the overall flow and the effectiveness and significance of potential regulation, important information on the potential effects impacts and implications of potential regulation can only be drawn when the amounts of concerned waste streams are known.

Thus, besides the mass flows on POPs, the waste flows have to be established for each investigated sector as well. As for the POP mass flows, detailed information is contained in the corresponding paragraphs of chapter 6.3. Here a compilation of the related waste streams and corresponding

contamination levels on the EU 27 scale is given. The corresponding contribution to the overall discharge of PFOS is added for better orientation.

Information concerning effects of proposed limit values is compiled and discussed in chapter 9.

Table 5-13: Annual amounts of PFOS containing residues (wastes) and PFOS contamination ranges

Sector/activity	Waste	Amount [t/y]	Contamination data [mg/kg / ppm]			PFOS discharge [kg/y]	PFOS to waste [kg/y]	Share of total PFOS to waste [%]
			mean	min	max			
Metal plating	Chromium sludge	99,506	3.8	-	-	378	378	0.23
	Vacuum distillation concentrate	1,245	240.0	-	-	298	299	0.18
	Activated carbon from treatment	1,130	296.7	-	-	335	335	0.21
	Activated carbon filter	354	950.0	-	-	336	336	0.20
	Ion exchange resin	17.7	19,000.0	-	-	336	336	0.20
	Emission water treated water	3,347,176	0.060			201		
	Emission water treated water	334,718	0.5			169		
	Emission air	0	0.0			0.15		
Photographic industry	Solid residue recycling x-ray	0.91	127,897.68	-	-	117	117	0.072
	Activated carbon x-ray	19.5	1,000.00	-	-	19.50	19.50	0.012
	Untreated waste water x-ray	130,000	0.15	-	-	19.50		
	Desilvered solution x-ray	7,735	20.17	-	-	156.00	156.00	0.096
	x-ray waste NHL	15,571	13.45	-	-	209.35	209.35	0.128
	x-ray waste NHWI	7,634	13.45	-	-	102.65	102.65	0.063
	Other films NHL	62,248	4.33	-	-	269.47	269.47	0.165
	Other films NHWI	30,521	4.33	-	-	132.13	132.13	0.081
	Picture waste NHL	61,225	4.33	-	-	265.05	265.05	0.162
	Picture waste NHWI	30,019	4.33	-	-	129.96	129.96	0.080
	Liquid photographic fluids HWI	0.069	410,000	-	-	28.10	28.10	0.017
	Emission water	55,960	0.15	-	-	8.39		0.000
Aviation industry	Hydraulic fluid HWI	715	1,000			715.40	715.40	0.438
	Emission water	10.0	1,000			10.02		
	Emission air	4.3	1,000			4.29		
Fire fighting foams	Fire fighting foam	605	10,000	-	-	6,049.1	6,049.1	3.705
	Emission water	45	10,000			453.7		
Leather	Upholstery incineration	480,371	2.4	-	-	1,153	1,153	0.71

	Upholstery energy recovery	302,016	2.4	-	-	725	724.8	0.44
	Upholstery landfilled	1,595,689	2.4	-	-	3,830	3,8330	2.35
Carpet	Carpet incineration	391,480	75	-	-	29,492	29,492	18.1
	Carpet energy recovery	246,128	75	-	-	18,542	1,8542	11.4
	Carpet landfilled	1,300,410	75	-	-	97,966	97,965.6	60.0
Sewage sludge	Low contaminated Sewage sludge non-hazardous waste incineration	1,906,384	0.1	-	-	191	191	0.12
	Low contaminated Sewage sludge energy recovery	7,377,749	0.1	-	-	738	738	0.45
	Low contaminated Sewage sludge reuse/recovery	1,714,968	0.1	-	-	172	172	0.11
	High contaminated Sewage sludge non-hazardous waste incineration	100,336	1.0	-	-	100	100	0.061
	High contaminated Sewage sludge energy recovery	388,303	1	-	-	388	388	0.238
	High contaminated Sewage sludge reuse/recovery	90,261	1	-	-	90.3	90.3	0.055
Total		20.09 Mio t				164,130	163,264	100

It has to be mentioned that due to missing availability of data no ranges could be indicated for PFOS concentrations in the above specified contaminated materials. The indicated PFOS concentration value, which has been used for the calculations, is based on calculations derived from available data on PFOS.

Based on the data of the detailed mass flows in chapter 6.3 and the contamination data compiled in Table 5-16 it can be concluded that the PFOS concentration of PFOS contaminated wastes is very different. Contamination of sewage sludge, carpets, upholstery, within the sector photographic industry recycled photo plates, picture wastes, other films, x-ray wastes, desilvered solutions, untreated waste water and chromium sludge from the metal plating sector lie beneath 100 ppm.

In the sector metal plating wastes as concentrates from vacuum distillation, treatment of activated carbon and activated carbon filter, in the photo industry activated carbon wastes as well as contamination of hydraulic fluids in the aviation industry range from 100 to 1000 ppm.

Contamination of some wastes exceeds a contamination value of 1000 ppm. These are ion exchange resin wastes (19,000 ppm) from the metal plating sector, solid residues (127,898 ppm) and liquid fluids (410,000 ppm) from the photographic industry and fire fighting foams (10,000 ppm).

5.4.4 Substance flow PeCB

5.4.4.1 Substance flow PeCB

The following figure shows the overall result of the detailed PeCB mass flow analysis. The individual streams from each activity are listed in chapter 6.4.9 to 6.4.13 and are discussed for each activity in the corresponding detailed PeCB mass flow. Sectors not covered with respect to the PeCB mass flow are discussed in chapter 6.4. Based on the information presented it can be stated that the most significant processes are covered in the total mass flow shown in Figure 5-14.

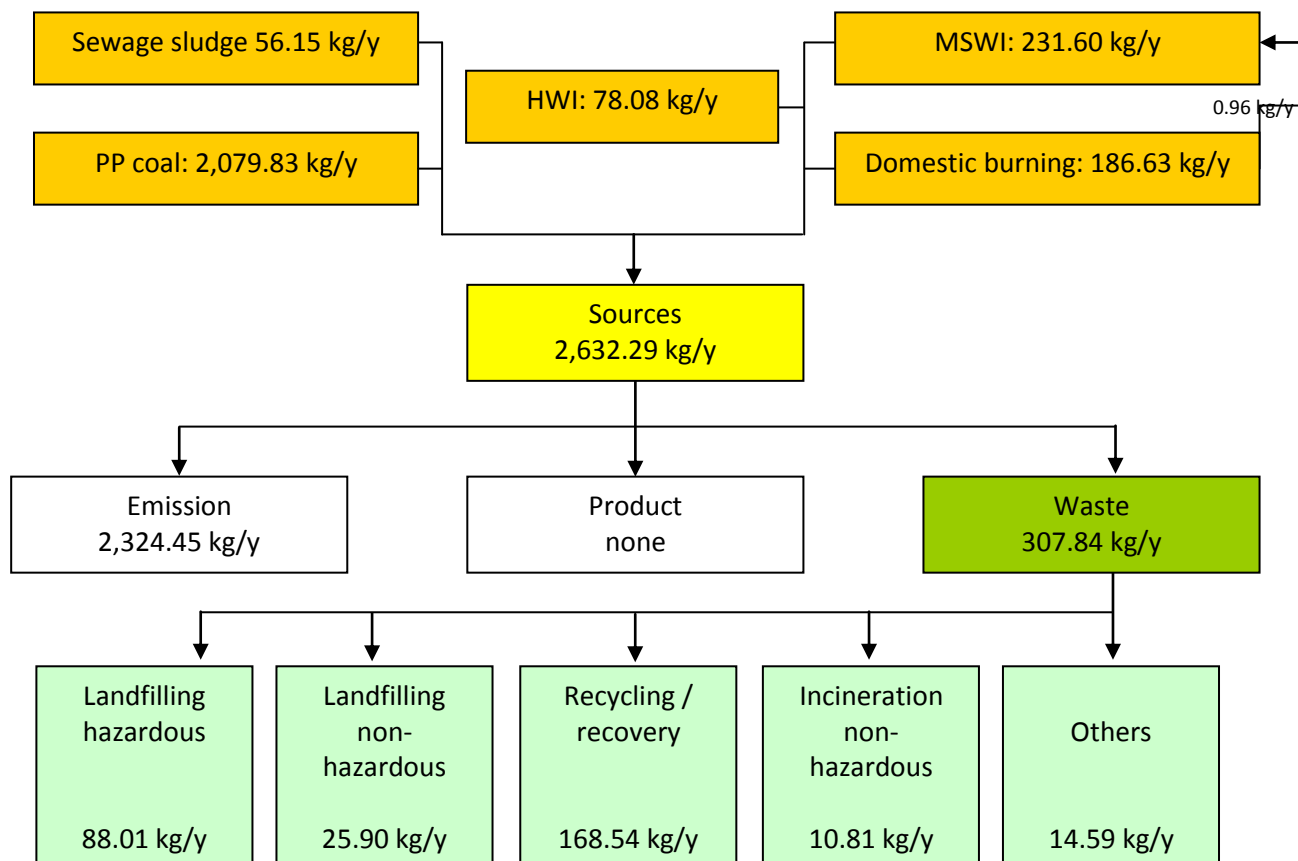


Figure 5-14: Overview of PeCB substance flow

It has to be taken into account that due to limited data for a number of sectors and countries the mass flow is partly based on assumptions and extrapolation. However, the mass flow can give an impression on the dimension of the PeCB flow to the environment and to waste and on the relative contribution of major relevant sectors related to PeCB.

Table 5-14 shows the contribution of the different sectors to the European PeCB flow to emissions, products and waste.

Table 5-14: Contribution of the relevant sectors in the EU 27 to PeCB in emissions, products and waste

Sector/Activity	Emission [kg/y]	Product [kg/y]	Waste [kg/y]	Sum [kg/y]
Municipal solid waste incineration	147.42	0.00	84.17	231.6
Hazardous waste incineration	73.56	0.00	4.53	78.08
Power production (coal)	1,921.62	0.00	158.21	2,079.83
Domestic burning	181.85	0.00	4.78	186.63
Sewage sludge	0.00	0.00	56.15	56.15
Total	2,324.45	0.00	307.84	2632.29

As illustrated the overall discharge of PCB from the investigated sources in Europe accounts for about 2,632 kg/y. Approximately, 88 % of the releases are emitted to air and soil and only approximately 12 % of the releases end up as waste. To conclude it can be stated that the sources of PeCB are dominated by the power production from coal. A total quantity of 79 % originates from this sector as is described in Figure 5-15.

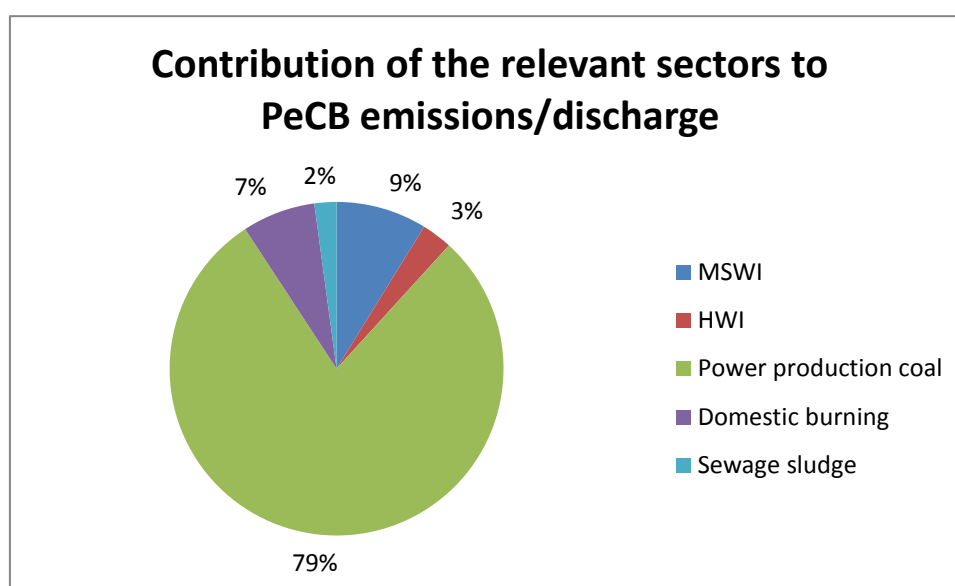


Figure 5-15: Contribution of the relevant sectors to the EU 27 to PeCB emissions to environment and discharge to waste

In order to illustrate the relative importance of different sectors on air emissions and discharge to waste, the shares of the investigated sectors are illustrated in the following pie charts.

As illustrated in Figure 5-16 below, the most important source for emission to the environment is power production of coal with 83 % followed in great distance by domestic burning of solid fuels, wood and mixed wastes (8 %), municipal solid waste incineration (6 %) and hazardous waste incineration (3 %). The share of emissions due to waste water treatment accounts to 0 %. All PeCB emissions are emissions to air, except emissions due to domestic burning. From the 8 % of PeCB released due to domestic

burning <1% (0.88 %) is released to soil. In regard to the total PeCB amount of 2,324.45 kg emitted to the environment the share of PeCB released to soil amounts to 0.07 %.

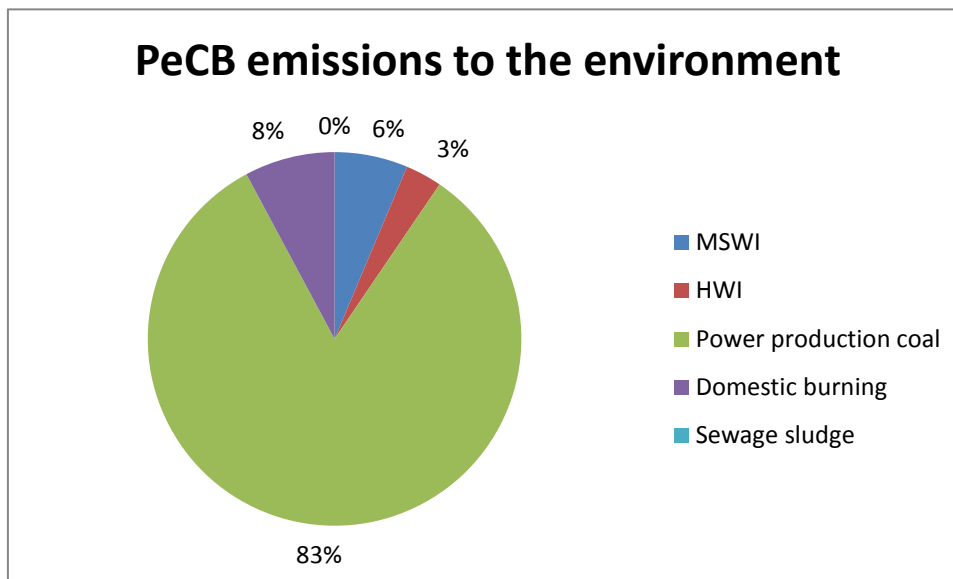


Figure 5-16: Relative distribution of PeCB emissions to the environment from investigated sectors in EU 27

Unlike emissions to air discharges to waste show a more even distribution for the investigated sectors. Based on the results from the mass flows in chapter 6.4 the most important sources for discharge to waste is again power production from coal (51 %), followed by waste resulting from municipal solid waste incineration (27%) and waste water treatment (18 %). PeCB waste arising from domestic burning and hazardous waste incineration accounts for 2 % for each sector. The shares of PeCB discharge to waste from the relevant sectors are shown in Figure 5-17.

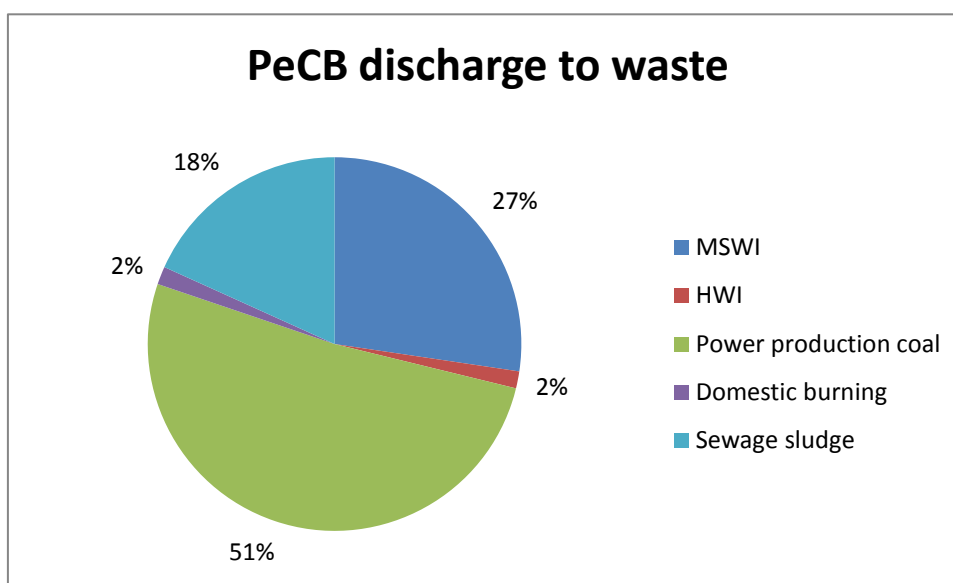


Figure 5-17: Relative PeCB distribution to waste from investigated sectors in EU 27

PeCB discharged to waste is directed to different waste management operations including disposal and recovery operations. As the environmental relevance of the different operations differs, it is of interest to trace the amounts of each “source” sector directed to specific management sectors and to distinguish the relative share of each operation in relation to the overall PeCB discharge to waste. Table 5-15 illustrates how the selected activities contribute to the PeCB flow to specific disposal/recovery operations, whereas Figure 5-18 illustrates the importance of current waste treatment options with respect to the PeCB mass flow.

Table 5-15: Quantitative contribution of PeCB from the different sectors to currently used disposal/recovery operations in EU 27

Treatment / Activity	kg/y	Comment
Non Hazardous waste incineration (NHWI)		
Domestic burning	0.96	50.8 Mio t of wood, solid fuel and MSW are domestically burned, generating about 2.28 Mio t of (fly) ash. 75% of this (1.71 Mio t) enters the waste stream of which 20 % are incinerated
Sewage sludge	9.85	2.03 Mio t sludge (mean 4.85 ppb)
Total	10.81	
Landfill for hazardous waste (including underground storage) (LHW)		
Municipal solid waste incineration (MSWI)	82.17	1.10 Mio t fly ash (mean 74.4 ppb)
Hazardous waste incineration (HWI)	4.40	0.46 Mio t fly ash (mean 9.57 ppb)
Power production from coal (PP coal)	1.44	0.60 Mio t fly ash and FGT residues (mean 2.40 ppb)
Total	86.58	
Landfill for non hazardous waste (LNHW)		
Power production from coal (PP coal)	14.38	5.99 Mio t fly ash and FGT residues (mean 2.40 ppb)
Domestic burning	1.91	75 % of 2.28 Mio t, i.e. 1.71 Mio t, of (fly) ash reaches the MSW stream (mean 1.12 ppb); 40 % is subject to the 103.96 Mio t of MSW landfilled as non-hazardous waste
Sewage sludge	9.60	1.98 Mio t sludge (mean 4.85 ppb)
Total	25.90	
Recycling / recovery (RR)		
Municipal solid waste incineration (MSWI)	2.00	0.03 Mio t fly ash (mean 74.4 ppb)
Hazardous waste incineration (HWI)	0.12	0.013 Mio t fly ash (mean 9.57 ppb)
Power production from coal (PP coal)	139.22	57.98 Mio t fly ashes (mean 2.40 ppb)
Sewage sludge	27.18	5.60 Mio t sludge applied for agricultural use and compost, i.e. application to land (mean 4.85 ppb)
Total	168.54	
Temporary storage (TS)		
Power production from coal (PP coal)	3.16	1.32 Mio t of fly ash (mean 2.40 ppb)
Others (O)		
Domestic burning	1.91	50.8 Mio t of wood, solid fuel and MSW are domestically burned, generating about 2.28 Mio t of (fly) ash. 75% of this (1.71 Mio t) enters the waste stream of which 20 % are incinerated
Sewage sludge	9.51	1.96 Mio t sewage sludge treated by different options, e.g. temporary storage (mean 4.85 ppb)
Total	11.42	
Total	307.84	

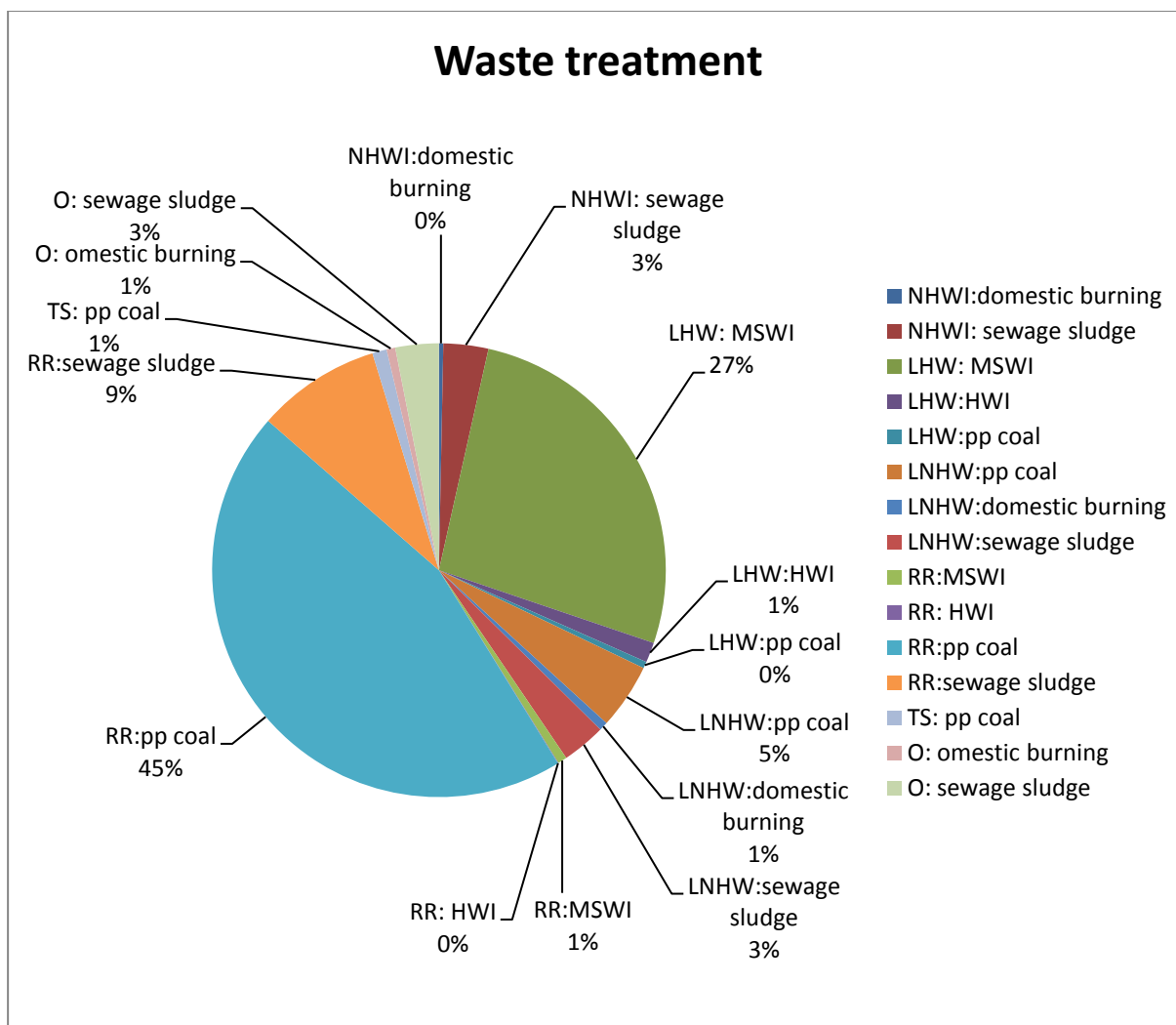


Figure 5-18: Relative importance of different waste treatment options with respect to the PeCB mass flow in EU 27

5.4.4.2 Mass flows (articles/waste) related to PeCB

While the relative contribution of industrial, public and domestic sectors on the overall POP release is interesting for the assessment of potential environmental impact and the assessment of the overall flow and the effectiveness and significance of potential regulation, important information on the potential effects impacts and implications of potential regulation can only be drawn when the amounts of concerned waste streams are known.

Thus, besides the mass flows on POPs, the waste flows have to be established for each investigated sector as well. As for the POP mass flows, detailed information is contained in the corresponding paragraphs of chapter 6.4.9 to 6.4.13. Here a compilation of the related waste streams and corresponding contamination levels on the EU 27 scale is given. The corresponding contribution to the overall discharge of PeCB is added for better orientation.

Information concerning effects of proposed limit values is compiled and discussed in chapter 0.

Table 5-16: Annual amounts of PeCB containing residues (wastes) and PeCB contamination ranges

Sector/activity	Waste	Amount [Mio t/y]	Contamination data [ng/g]			PeCB discharge [kg/y]	PeCB to waste [kg/y]	Share of total PeCB to waste [%]
			mean	min	max			
MSWI	Fly ash	1.13	74.42	11.50	240.00	84.17	84.17	27.34
HWI	Fly ash and residues from FGT	0.47	9.57	0.30	48.00	4.53	4.53	1.47
Power production (coal)	(Fly) ashes	65.88	2.40	0.004	7.00	158.21	158.21	51.39
Waste water treatment	Sewage sludge	11.58	4.85	2.16	37.30	56.15	56.15	18.24
Domestic burning	(Fly) ash from wood	0.55	0.20	-	-	0.11	0.08	0.03
	(Fly) ash from coal	1.70	2.40	0.004	7.00	4.09	3.07	1.00
	(Fly) ash from mixed wastes	0.03	74.4	11.50	240.00	2.17	1.63	0.53
Total		81.35				309.43	307.84	100

Based on the data of the detailed mass flows in chapter 6.4 and the contamination data compiled in Table 5-16 an overview of current levels of contamination as well as their corresponding ranges is illustrated in Figure 5-19 below.

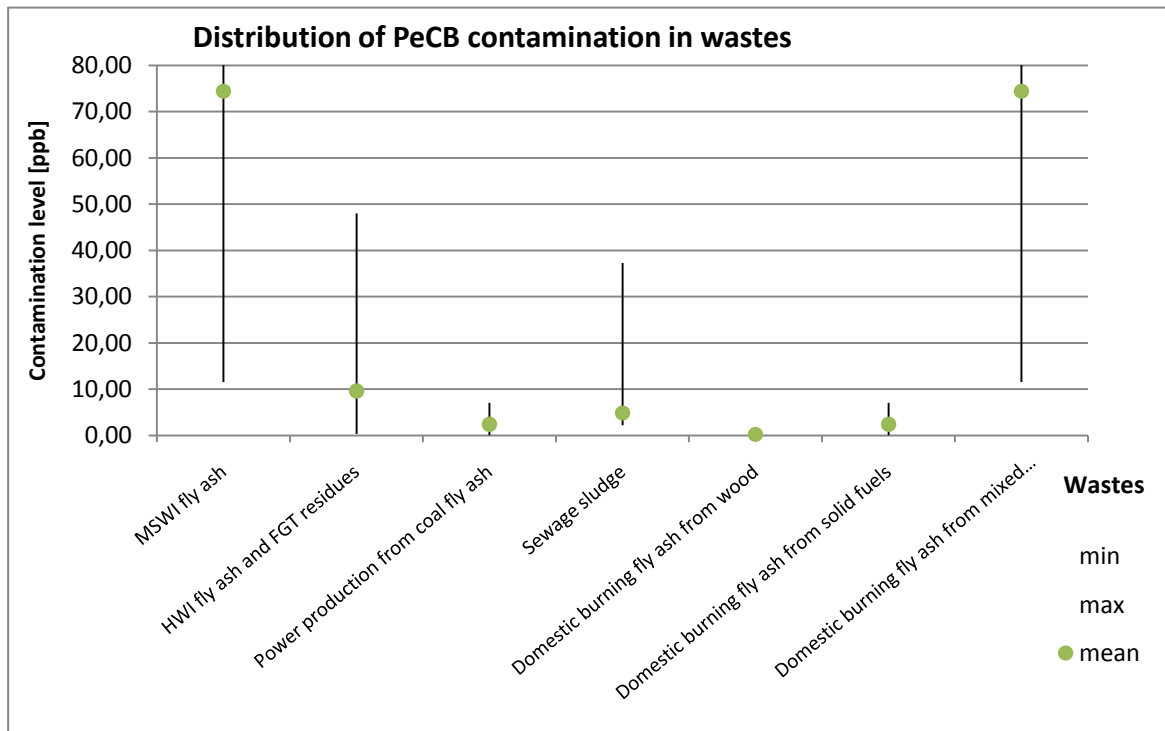


Figure 5-19: Means and ranges of recent PeCB contamination in different waste types in EU 27

As shown in the Figure 5-19 and summarised in Table 5-16 (based on available data) mean concentrations and maximum limits of most high volume waste streams lie below 100 ppb, whereas the

majority of mean PeCB concentrations even lie below 10 ppb. Only a limited number of waste types, i.e. those from fly ash generated from incineration of municipal solid waste in incinerators as well as from domestic burning of mixed household wastes, show concentration ranges with including the mean value lying above 10 ppb. Other wastes containing PeCB concentration ranges above 10 ppb are fly ash from hazardous waste incineration and PeCB concentration values found in sewage sludge, whereas the mean concentration of PeCB lies below 10 ppb. To conclude, PeCB contents in wastes containing the substance are comparatively low.

5.4.5 Substance flow SCCPs

5.4.5.1 Substance flow SCCP

Figure 5-20 shows the overall result of the detailed SCCP mass flow analyses. The individual streams are discussed in detail in the corresponding paragraphs of the SCCP mass flows in chapter 6.

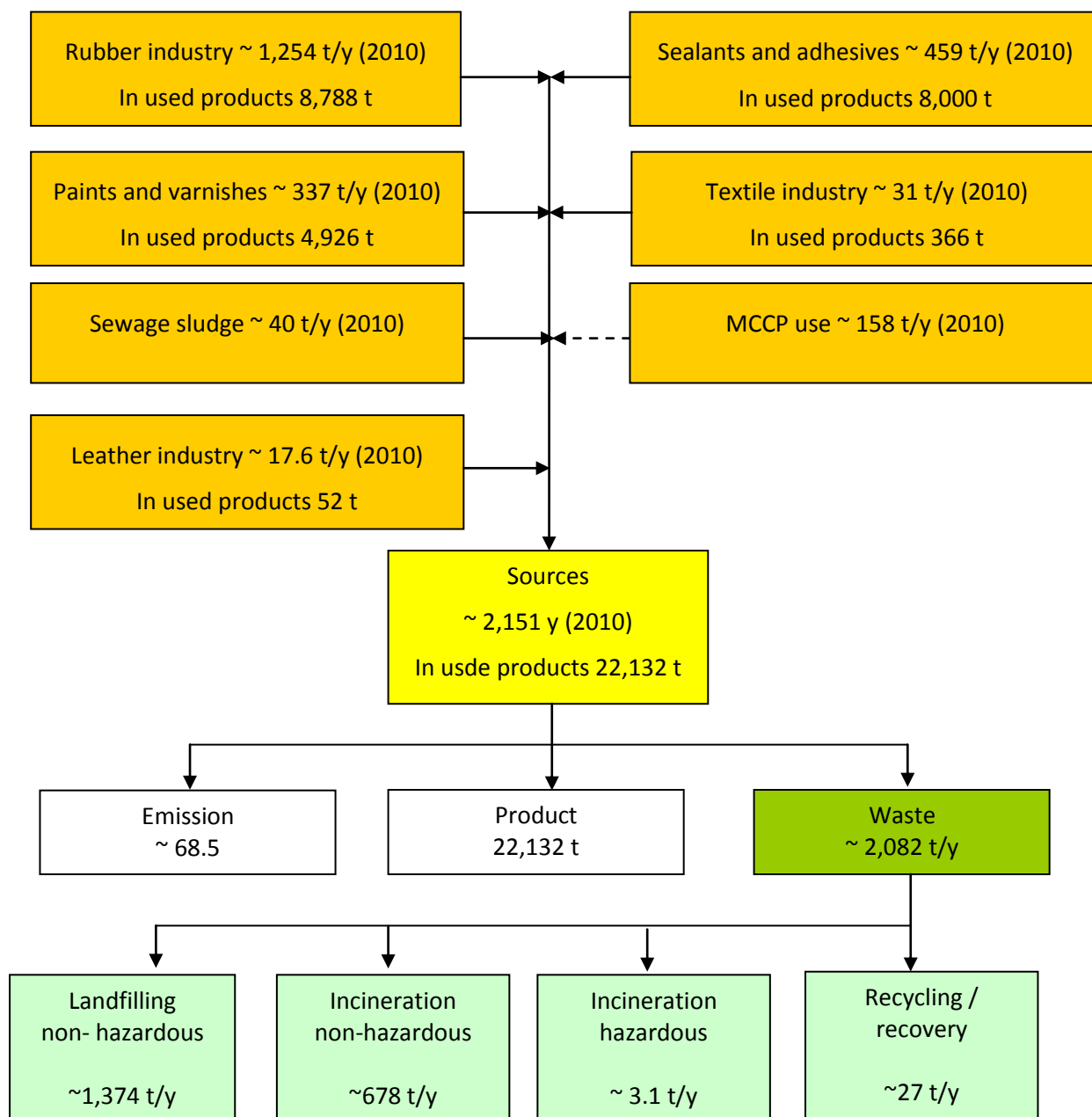


Figure 5-20: Overall mass flow for SCCPs from relevant sources to current disposal/recovery operations in the EU 27.

Due to limited data availability for a number of sectors, Figure 5-20 is partly based on estimations and extrapolations. These are explained in the detailed mass flow analyses in chapter 6. Based on the calculations discussed in more detail in the corresponding paragraphs of the detailed mass flow for

SCCP, major sources for the current waste flow of SCCP can be pointed out. These are the conveyor belts for underground mining, currently contributing with around 1,126 t of SCCP per year. Due to the long lifetime of such conveyor belts the total amount of SCCP still in use is much higher (8,800 t since 1990). As little is known about the distribution of MCCP it is not included in the waste flow diagram and the arrow is dotted.

Figure 5-21 presents the relative distribution of environmental emissions from investigated sectors in the EU. The most important source of emissions are volatile and particulate releases from sealants and adhesives (42 % or 36 t/y) followed by rubber (31 % or 25 t/y) as well as from paints and varnishes (21 % or 17.4 t/y) and textiles (5 % or 4.1 t/y) (see Figure 5-21).

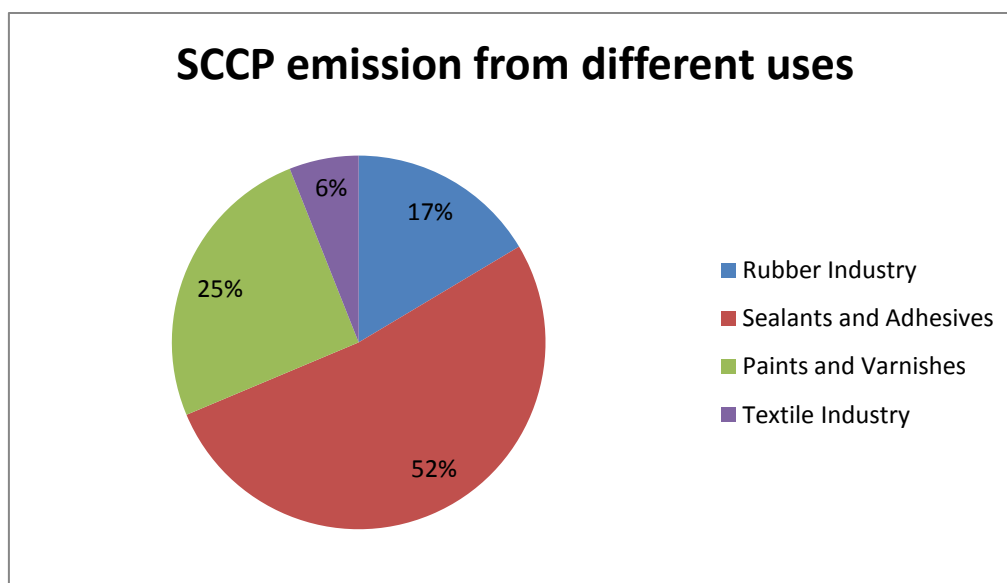


Figure 5-21: Relative distribution of environment emissions from investigated sectors in the EU 27

Figure 5-22 indicates the relative distribution of discharge of SCCP to waste from the different uses. It is visible that the rubber industry has a waste share of about 60 % (40 % + 20 %), the waste from the sealants and adhesives about 21 % (14 % + 7 %) and from paints and varnishes to 15 % (10 % + 5 %). These three uses cover about 96 % of all the SCCP containing waste. In this figure the non-hazardous waste incineration also includes incineration for energy recovery.

Disposal/Recovery	SCCP [t/y]
Paints and Varnishes	41
Textile Industry	3
Leather Industry	2
TOTAL energy recovery	259
Reuse	
Sewage Sludge	27
Overall Total	2,082

The following pie chart illustrates the importance of current waste treatment options with respect to the SCCP flow.

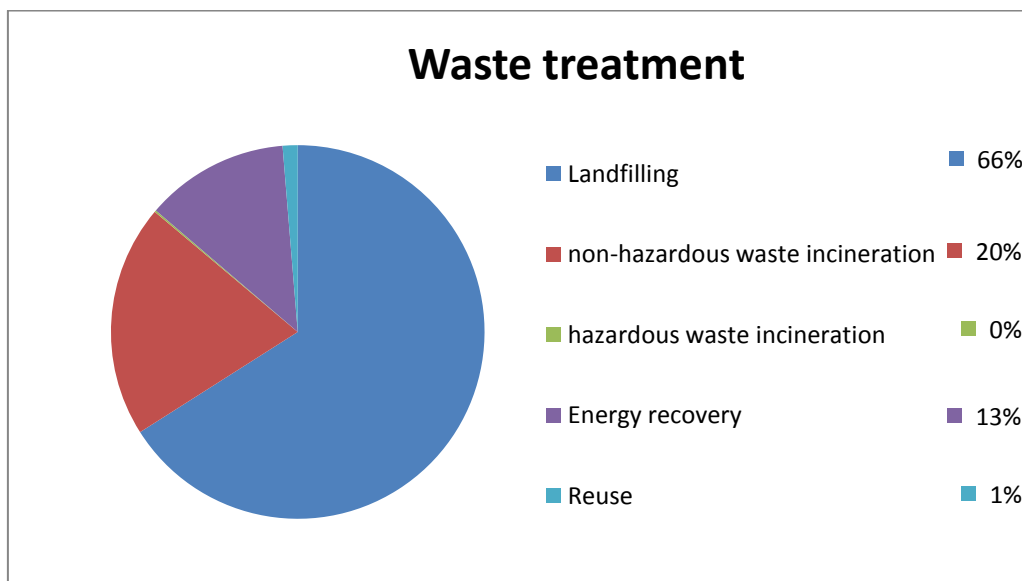


Figure 5-23: Relative importance of different waste treatment options for the SCCP flow in the EU

As illustrated above, landfilling represents the highest share of 66 % followed by non-hazardous waste incineration with about 20%, incineration with energy recovery (~13%) and the reuse with around 1 % of the controlled final disposal of SCCP. The amount of SCCP containing waste which is incinerated in hazardous waste incineration plants is below 0.2 %.

5.4.5.2 Mass flows (articles/waste) related to SCCP

While the relative contribution of industrial, public and domestic sectors on the overall POP release is interesting for the assessment of potential environmental impact and the assessment of the overall flow and the effectiveness and significance of potential regulation, important information on the potential effects impacts and implications of potential regulation can only be drawn when the amounts of concerned waste streams are known.

Therefore, the relevant waste flows have been established for each investigated sector. Here, the compilation of the related waste streams and corresponding contamination levels on the European scale are presented. The corresponding contribution to the overall discharge of SCCP is added for better orientation.

Table 5-18: Annual amounts of SCCP containing residues (waste and products) and the ranges of contamination with SCCP and corresponding SCCP amount in investigated sectors in the EU

Sector /activity	Waste	Amount [t/y]	Contamination data [mg/kg/ppm]			SCCP discharge [t/y]	SCCP to waste [t/y]	Share of total SCCP to waste [%]
			mean	min	max			
Rubber Industry	Production	287	49,053			14	14	0.7%
	Conveyor belts	33,781	33,333			1,137	1,126	54.1%
	Gaskets, hoses,	844	135,000			114	114	5.5%
Sealant and adhesives	Production	59	200,000			12	12	0.6%
	Product	2,058	200,000			447	412	19.8%
Paint and varnishes	Production	233	130,000			30	30	1.5%
	Product	2,229	130,000			307	290	13.9%
Textile Industry	Production	15	200,000			6	3	0.1%
	Product	117	200,000			24	23	1.1%
Leather Industry	Product	4,320	4,000			17	17	0.8%
Sewage sludge	Low contaminated	10,999,100	1			11	11	0.5%
	High contaminated	578,900	50			29	29	1.4%
TOTAL		11,621,944				2,150	2,082	100%

5.4.6 Substance flow HBCD

Figure 5-24 shows the overall result of the detailed HBCD mass flow analyses in Chapter 6. The individual streams are presented in Figure 6-98/Figure 6-99 (EPS), Figure 6-104/Figure 6-105 (XPS), Figure 6-110/Figure 6-111 (EPS/XPS other than C&D), Figure 6-116 (HIPS) and Figure 6-121 (textiles). These are further explained in the corresponding paragraphs of the detailed HBCD substance flows in Chapter 6.

It is important to notice that the major share of HBCD (~10,431 t/y), which is at the present incorporated into new products, will become waste in the future (see Figure 5-24).

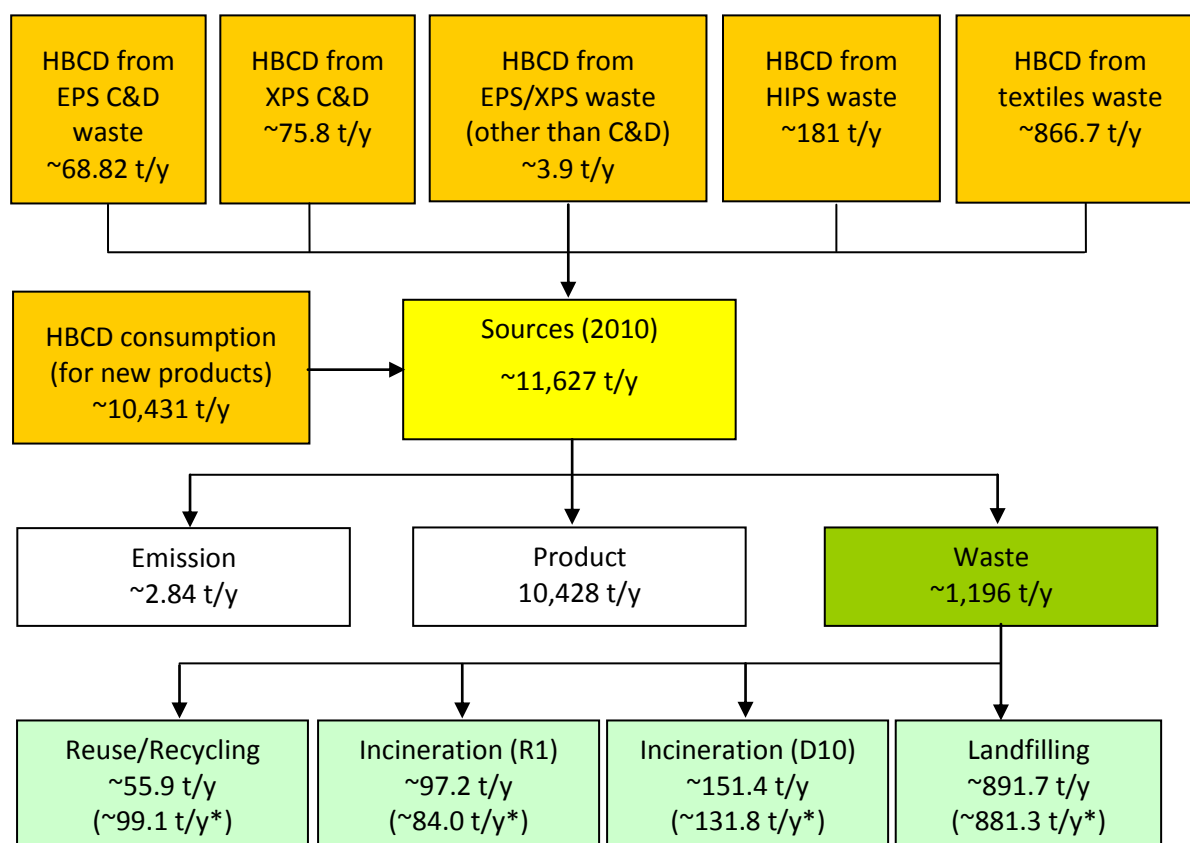


Figure 5-24: Overall substance flow of HBCD from relevant sources to current disposal/recovery operations in the EU (*Scenario 2: HBCD containing waste not separated from other plastic C&D waste or MSW)

The mass flow presented in Figure 5-24 is partly based on assumptions and extrapolations. However, the mass flow gives an impression on the dimension of the HBCD flow to the environment and to waste and on the relative contribution of major sectors in the EU in 2010.

Based on the calculations, discussed in more detail in the corresponding paragraphs of the detailed mass flows for HBCD, major sources for the current flow of HBCD can be pointed out. These are the EPS C&D waste (~68.82 t/y), the XPS C&D waste (~75.8 t/y), EPS/XPS other than C&D waste (~3.9 t/y), HIPS waste (~181 t/y) and HBCD from textile waste (~866.7 t/y). Besides, approximately 10,431 t/y HBCD have been consumed (2010) for the manufacture of new HBCD flame retarded products in the EU. This leads to a total figure of approximately 11,627 t/y (see Figure 5-24).

Table 5-19 summarizes the emissions and discharge of HBCD from investigated sources in the EU. Besides, the amount of HBCD incorporated into new flame retarded products has been summarized.

Table 5-19: Emissions and discharge of HBCD from investigated sources in the EU (2010)

Sector/Activity	Emission [t/y]	Product [t/y]	Waste [t/y]
EPS from C&D waste	0.65	4,471	68.82
XPS from C&D waste	0.71	4,922	75.8
EPS/XPS (other than C&D waste)	0.10	626	3.9
HIPS from WEEE	0.03	208.8	181
Waste Textiles from various products	1.35	207.4	866.7
Total	~2.84	~10,435	~1,196

Figure 5-25 shows the relative distribution of HBCD emissions from the investigated sectors in the EU to the total emission quantity of approximately 2.8 tonnes. The most important sources of emissions are the HBCD treated textiles (~48 %) though the current consumption for textiles contributes only 2 % of the total current consumption. EPS and XPS from C&D contribute together to approximately 48 % of the total HBCD emissions in 2010. EPS/XPS products outside the construction sector are responsible for around 3 % of the total HBCD emissions in the EU. The remaining share of approximately 1 % can be assigned to HBCD treated HIPS (see Figure 5-25).

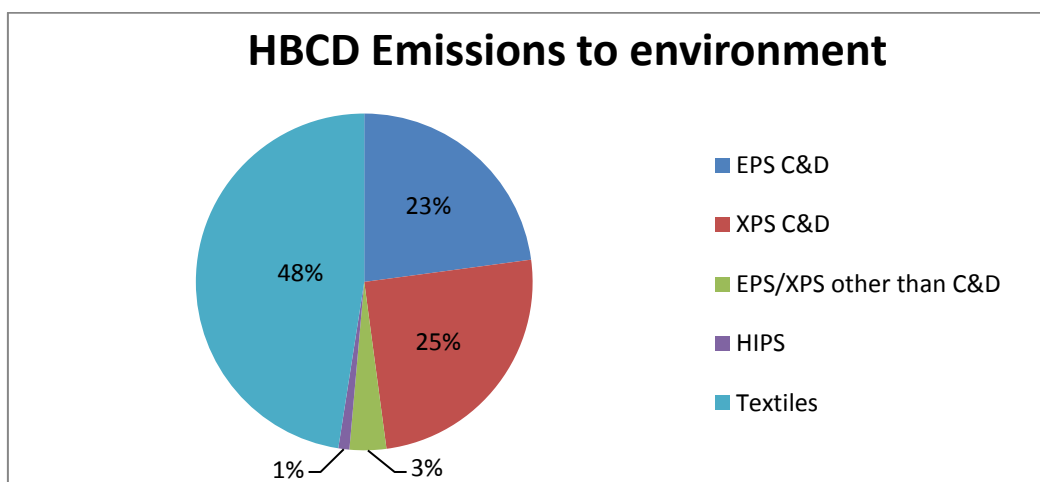


Figure 5-25: Relative distribution of environment emissions from investigated sectors in the EU

The relative distribution of the HBCD emissions to air, waste water and surface water is summarized in Table 5-20.

Table 5-20: Distribution of HBCD emissions to air, waste water and surface water

Emission Sources	to air [%]	to waste water [%]	to surface water [%]
EPS and XPS (construction and non-construction)	41	21	38
HIPS	24	26	50
Textiles	0.1	79.9	20

Figure 5-26 indicates the relative distribution of discharge of HBCD to waste from the investigated sources.

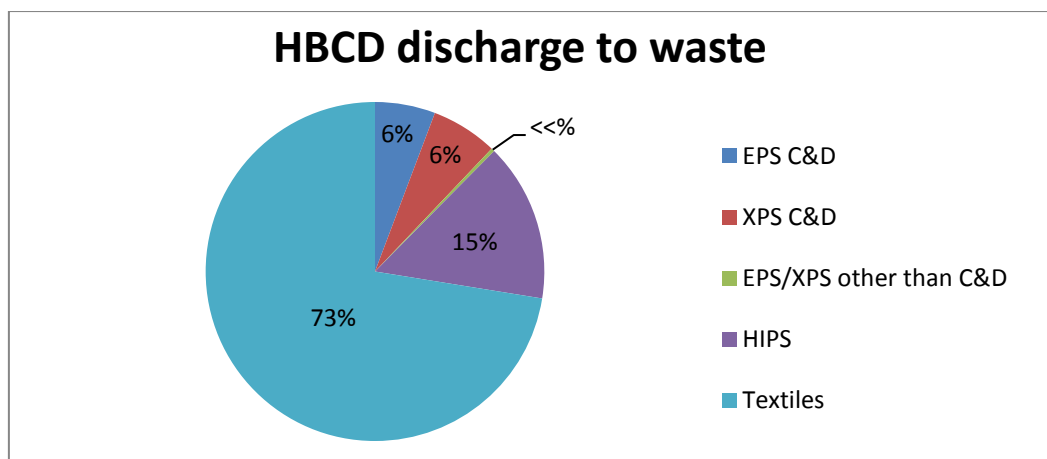


Figure 5-26: Relative distribution of discharge of HBCD to waste from investigated sectors in the EU

Currently, the most important sources, in relation to the discharge of HBCD to waste in the EU are the HBCD treated textiles and HIPS. As shown in Figure 5-26 these sources contribute with approximately 88 % to the overall HBCD discharge to waste (~73 % textiles and ~15 % HIPS). The EPS and XPS from C&D are each responsible for around 6 % (69+76=145 tonnes annual disposal) of the HBCD discharge to waste. The remaining HBCD discharge in the EU can be assigned to EPS/XPS products outside the construction sector (e.g. EPS/XPS packaging materials).

Figure 5-26 illustrates the HBCD discharge situation in 2010. It is, however, important to notice that the discharge situation will dramatically change in the future, when considering the significant amounts of HBCD incorporated into new EPS/XPS products for construction purposes (see Table 5-19) and the comparatively long life times of those products. Therefore, it is expected that in the long terms the C&D sector will be the only relevant sector, contributing to the entire HBCD discharge in the EU which is expected to increase significantly within the coming decades with an expected peak in the 2050ies (see Figure 6-97 and Figure 6-103) when annually more than 5,000 tonnes of HBCD will have to be disposed of from the C&D sector.

Taking as an example the estimated figures for the late 2020ies and the late 2030ies, the following changes, related to the emission and discharge of HBCD in the EU, can be expected (see Figure 5-27).

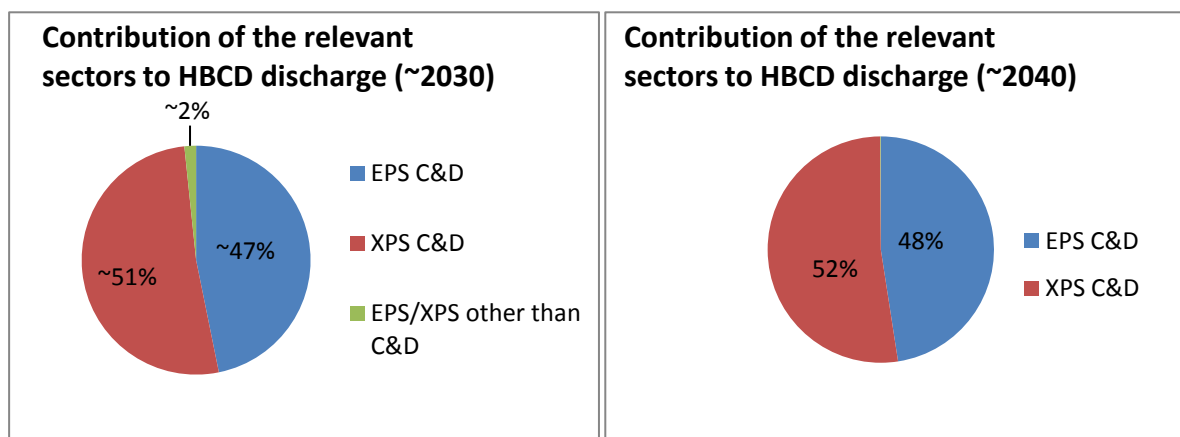


Figure 5-27: Contribution of the relevant sectors to the HBCD discharge to waste in the EU (~2030 and ~2040)

By the end of the 2020ies, the most relevant contribution to the HBCD discharge to waste will be from the C&D sector (~98 %, ~1,300 tonnes annual disposal; EPS and XPS). In addition a minor contribution to the total discharge is expected from EPS/XPS products outside the C&D sector (~2 %). By the end of 2030ies, the C&D sector is expected to be the only relevant sector contributing to the total HBCD discharge (~48 % from EPS and ~52 % from XPS C&D; ~3,200 tonnes annual disposal).

HBCD discharged to waste is directed to different waste management operations including disposal and recovery operations. As the environmental relevance of the different operations differs, it is of interest to trace the amounts of each source/sector directed to specific management operations and to distinguish the relative share of each operation in relation to the overall HBCD discharge to waste.

Table 5-21 shows how the selected activities contribute to the mass flow to specific disposal/recovery operations. The figures in the table are related to certain conditions which are explained in the comments column.

Table 5-21: Quantitative contribution of HBCD from different sectors to currently used disposal/recovery operations in the EU (2010)

Disposal/Recovery	HBCD [t/y]	Comments
Recycling/Recovery	~55.9/*99.1	* HBCD containing waste not separated (Scenario 2)
EPS C&D	5.22/*22.94	
XPS C&D	0/*25.25	No XPS Recycling/Recovery (Scenario 1)
EPS/XPS other than C&D	0.9/1.1*	
HIPS	49.8	
Textiles	0	Recycling/Recovery not relevant for textiles
Incineration (R1)	~97.2/*84.0	* HBCD containing waste not separated (Scenario 2)
EPS C&D	14.25/*8.95	
XPS C&D	17.6/*9.85	
EPS/XPS other than C&D	0.5/*0.4	
HIPS	16.6	
Textiles	48.2	
Incineration (D10)	~151.4/*131.8	* HBCD containing waste not separated (Scenario 2)
EPS C&D	22.29/*14.0	
XPS C&D	26.5/*15.4	
EPS/XPS other than C&D	0.8/*0.6	
HIPS	26.4	
Textiles	75.4	
Landfilling	~891.7/*881.3	* HBCD containing waste not separated (Scenario 2)
EPS C&D	27.05/*22.94	
XPS C&D	31.7/*25.25	
EPS/XPS other than C&D	1.7/*1.9	
HIPS	88.1	
Textiles	743.1	
TOTAL	~1,196 t/y	

It is expected that the majority of HBCD containing waste is currently directed to non-hazardous landfill.

The following pie chart (see Figure 5-28) illustrates the importance of current waste treatment options with respect to the HBCD flow.

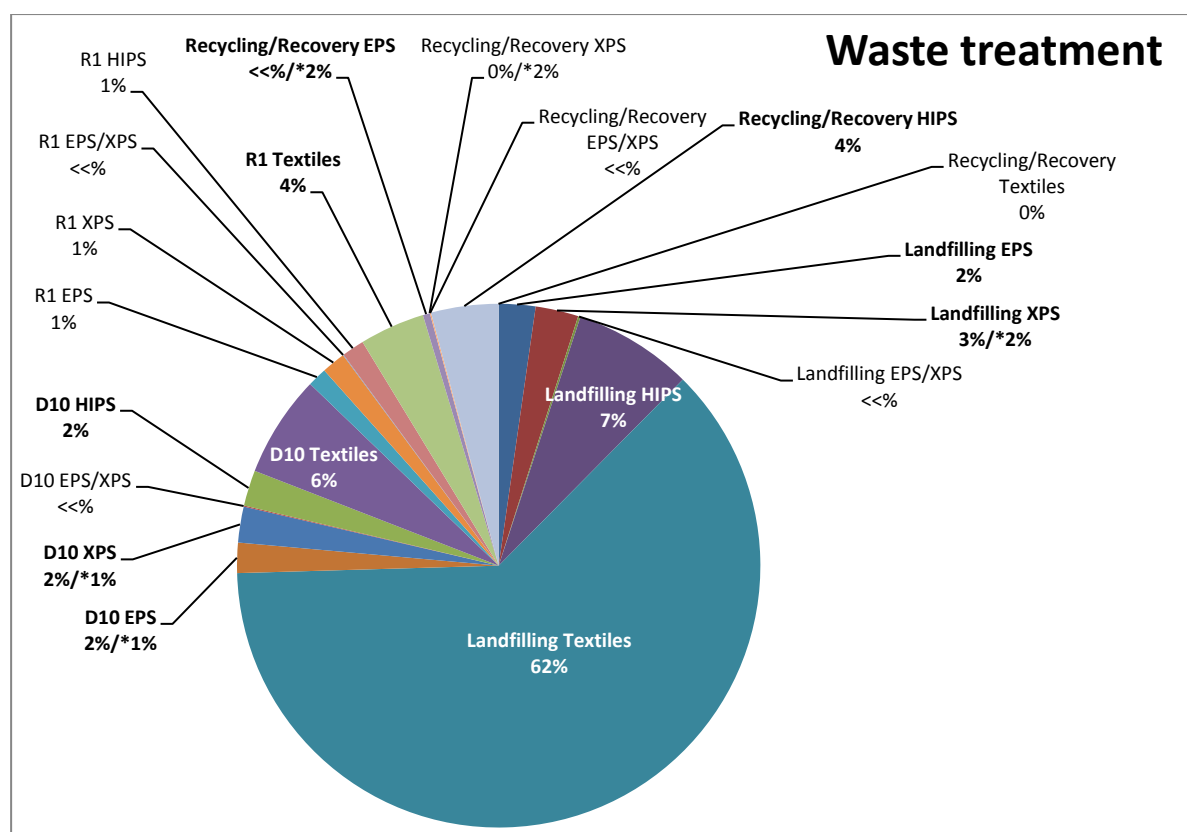


Figure 5-28: Relative importance of different waste treatment options for the HBCD flow in the EU (*In case the HBCD containing waste is not separated)

Currently, the most important waste treatment options, with respect to the HBCD flow in the EU, are landfilling of textiles (~62 %), landfilling of HIPS (~7 %) and incineration without energy recovery of textiles (~6 %). Other important waste treatment options are recycling/recovery of HIPS (~4 %) and incineration with energy recovery of textiles (~4 %). The remaining 17 % are made up of all other relevant waste treatment options (see Figure 5-28).

5.4.6.1 Mass flows (articles/waste) related to HBCD

While the relative contribution of the sectors on the overall HBCD release is interesting for the assessment of potential environmental impact and the assessment of the overall flow and the effectiveness and significance of potential regulation, important information on the potential effects impacts and implications of potential regulation can only be drawn when the amounts of concerned waste streams are known.

The relevant waste flows have been established for each investigated sector. The corresponding contribution to the overall discharge of HBCD is added for better orientation. Besides, the estimated annual figures (for the years 2030 and 2040) have been included in order to demonstrate the significant increase in future waste amounts.

Table 5-22: Annual amounts of HBCD containing residues (waste and products) and the ranges of contamination with HBCD and corresponding HBCD amount in investigated sectors in the EU; *ea = estimated average (used for calculations)

Sector	Waste	Amount [t/y]	Contamination [%]			HBCD discharge [t/y]	HBCD to waste/ products [t/y]	Share of HBCD to waste/products [%]
			min	ea*	max			
In 2010								
C&D	EPS	13,495	0.7	0.51	1.0	69.5	68.82/4,471.4	6/43
	XPS	3,790	1.0	2.0	3.0	76.5	75.8/4,922.3	6/47
Other than C&D (e.g. packaging)	EPS/XPS	288	0.7	1.35	3.0	4.0	3.9/626	<1/6
Electronic	HIPS	4,524	1.0	4.0	7.0	>181	181/208.8	15/2
Various (e.g. automotive)	Textiles	10,834	7.0	8.0	9.0	868	866/207.4	73/2
TOTAL		~32,931				~1,199	~1,196/~10,435	100/100
2030								
C&D	EPS	121,765	0.7	0.51	1.0	>621	621/0	47/0
	XPS	34,200	1.0	2.0	3.0	>684	684/0	51/0
Other than C&D (e.g. packaging)	EPS/XPS	2,207	0.7	1.35	3.0	>29.8	29.8/0	2/0
Electronic	HIPS	0	1.0	4.0	7.0	0	0/0	0/0
Various (e.g. automotive)	Textiles	0	7.0	8.0	9.0	0	0/0	0/0
TOTAL		~158,172				>1,335	~1,335/0	100/0
2040								
C&D	EPS	297,254	0.7	0.51	1.0	>1,516	1,516/0	48/0
	XPS	83,450	1.0	2.0	3.0	>1,669	1,669/0	52/0
Other than C&D (e.g. packaging)	EPS/XPS	0	0.7	1.35	3.0	0	0/0	0/0
Electronic	HIPS	0	1.0	4.0	7.0	0	0/0	0/0
Various (e.g. automotive)	Textiles	0	7.0	8.0	9.0	0	0/0	0/0
TOTAL		~380,704				>3,185	~3,185/0	100/0

Based on the data of the detailed mass flows in Chapter 6 and the contamination data compiled in Table 5-22, an overview of current levels of contamination as well as their corresponding ranges are illustrated in Figure 5-29.

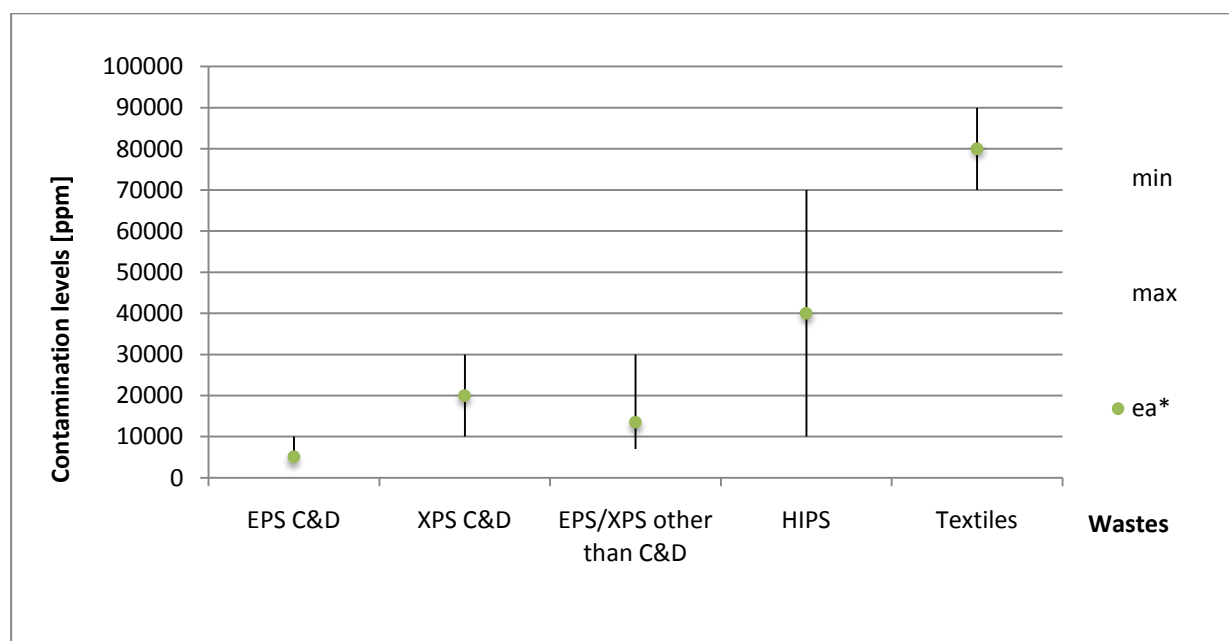


Figure 5-29: Estimated average (ea) and ranges of HBCD contamination in relevant wastes

As shown in the Figure 5-29, the mean concentrations and maximum contamination levels of all relevant waste streams lie below 100,000 ppm (10 %).

5.4.7 Substance flow PCP

5.4.7.1 Substance flow PCP

The following figure shows the overall result of the detailed PCP mass flow analysis. The individual streams from each activity are listed in chapter 6.7.4 to 6.7.5 and are discussed for each activity in the corresponding detailed PCP mass flow. Sectors not covered with respect to the PCP mass flow are discussed in chapter 6.7.3. Based on the information presented it can be stated that the most significant processes are covered in the total mass flow shown in following figure:

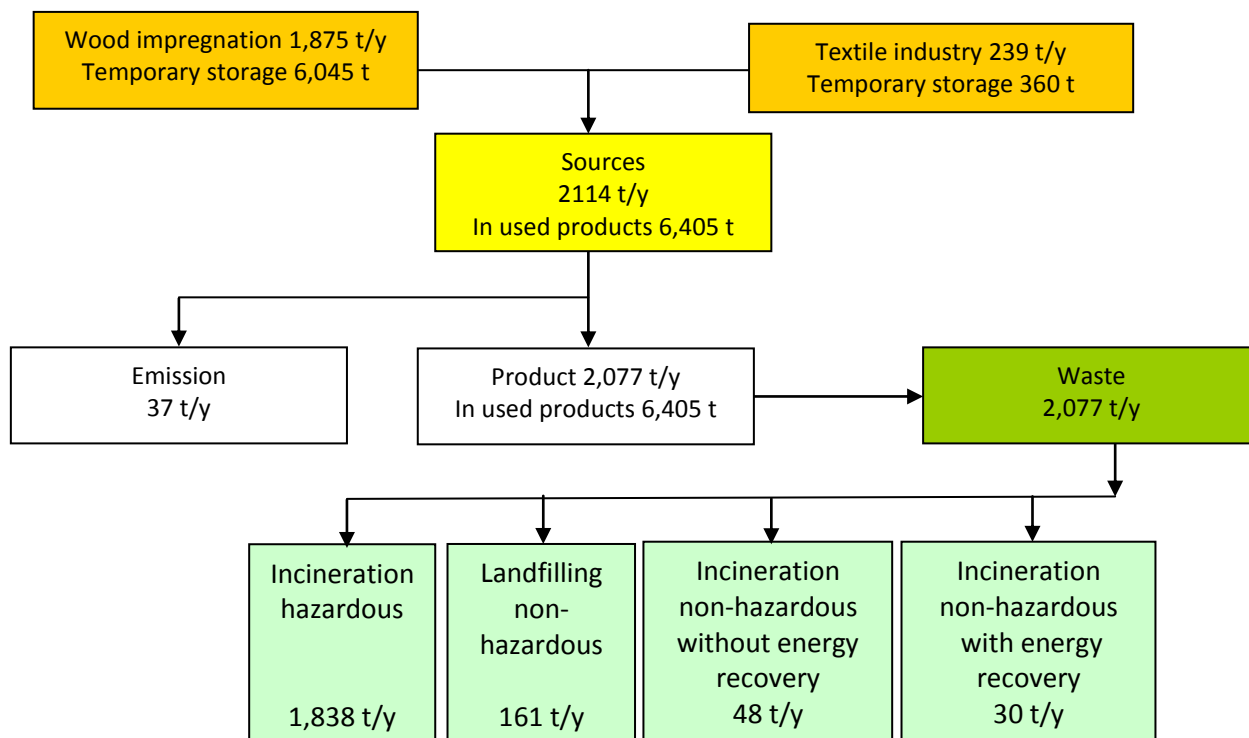


Figure 5-30: Overall mass flow from PCP from sources to current disposal/recovery operations in the EU

Based on the available information, the mass flow gives an impression on the dimension of the PCP flow to the environment and to waste and on the relative contribution of major relevant sectors related to PCP. The term temporary storage in this context means products still in use which do not enter the waste flow in 2010 but which are expected to become waste in the future (stocks of existing products).

Table 5-23 shows the contribution of the different sectors to the European PCP flow to emissions, products and waste.

Table 5-23: Contribution of the relevant sectors in the EU 27 to PFOS in emissions, products and waste

Sector/Activity	Emission [t/y]	Product [t/y]	Waste [t/y]	Sum [t/y]
Wood impregnation	37	6,045	1,838	7,920
Textile industry	0	360	239	599
Total	37	6,405	2,077	8,519

As illustrated the overall discharge of PCP from the investigated sources in Europe accounts to about 2,114 t/y (37 t/y emissions and 2,077 t/y waste). Almost all of the PCP released is discharged to waste (98.2%) whereas only a small fraction (1.8 %) is emitted. In conclusion it can be stated that the sources of PCP are dominated by wastes resulting from wood impregnation (89 %). Figure 5-31 shows the sources contributing to the PCP waste stream.

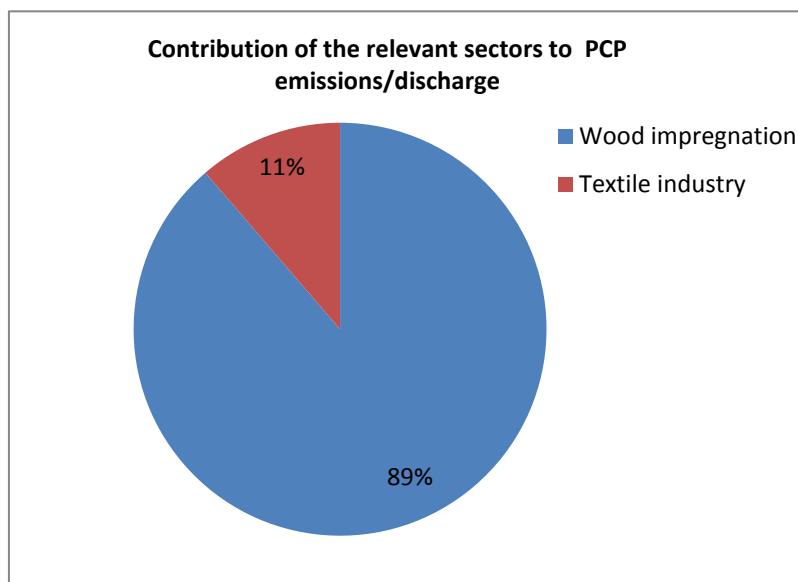


Figure 5-31: Contribution of the relevant sectors to the EU 27 to PCP emissions to environment and discharge to waste

The only current relevant source for emission to the environment is from the wood impregnation industry with 37 t/y.

PCP discharged to waste is directed to different waste management operations including landfilling and incineration. As the environmental relevance of the different operations differs, it is of interest to trace the amounts of each “source” sector directed to specific management sectors and to distinguish the relative share of each operation in relation to the overall PCP discharge to waste. Table 5-24 illustrates how the selected activities contribute to the PCP flow to specific disposal/recovery operations, whereas Figure 5-32 illustrates the importance of current waste treatment options with respect to the PCP mass flow.

Table 5-24: Quantitative contribution of PCP from the different sectors to currently used disposal/recovery operations in EU 27.

Treatment / Activity	t/y	Comment
Non hazardous waste incineration (NHWI)		
Textile industry	48	Without energy recovery
Textile industry	30	With energy recovery
Total	79	
Hazardous waste incineration (HWI)		
Wood impregnation	1,838	Impregnated wood is considered to be hazardous waste
Total	1,838	
Landfill for non hazardous waste (LNHW)		
Textile industry	161	
Total	161	

Treatment / Activity	t/y	Comment
Temporary storage (TS)		
Wood impregnation	6,045	
Textile industry	360	
Total	6,405	
Total without PCP in used products	2,077	
Overall total	8,482	

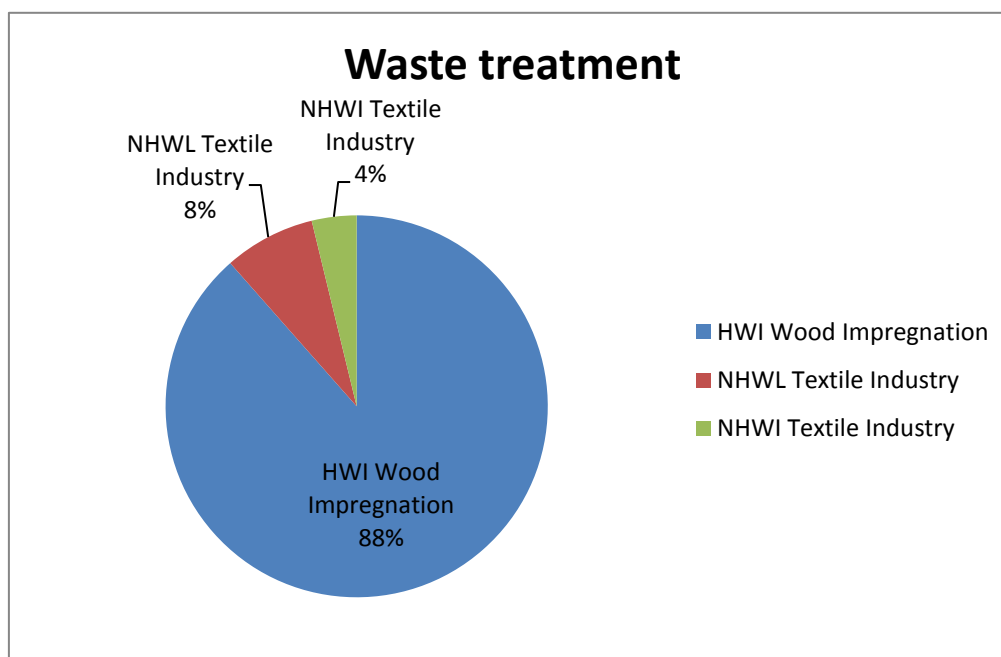


Figure 5-32: Relative importance of different waste treatment options with respect to the PFOS mass flow in EU 27

The most often used treatment option of PCP containing waste is currently hazardous waste incineration with 88%. All of this incinerated waste is impregnated wood. The remainder of PCP containing waste is only coming from the textile industry and is incinerated (either with or without energy recovery) to 67 % and disposed of at landfills for non-hazardous waste to 33 %. From the total PCP containing waste amount this represent 8 and 4 % respectively.

5.4.7.2 Mass flows (articles/waste) related to PCP

While the relative contribution of industrial, public and domestic sectors on the overall POP release is interesting for the assessment of potential environmental impact and the assessment of the overall flow and the effectiveness and significance of potential regulation, important information on the potential effects impacts and implications of potential regulation can only be drawn when the amounts of concerned waste streams are known.

Thus, besides the mass flows on POPs, the waste flows have to be established for each investigated sector as well. As for the POP mass flows, detailed information is contained in the corresponding paragraphs of chapter 6.7. Here a compilation of the related waste streams and corresponding

contamination levels on the EU 27 scale is given. The corresponding contribution to the overall discharge of PCP is added for better orientation.

Information concerning effects of proposed limit values is compiled and discussed in chapter 9.

Table 5-25: Annual amounts of PCP containing residues (wastes) and PFOS contamination ranges

Sector/activity	Waste	Amount[t/y]	Contamination data [mg/kg / ppm]			PCP discharge [kg/y]	PCP to waste [kg/y]	Share of total PCP to waste [%]
			mean	min	max			
Wood impregnation	Impregnated wood	2,941,00	625	150	50,000	1,875	1,838	89 %
Textile industry	Impregnated textile	9,568	25,000			239	239	11 %
Total		2,950,634				2,114	2,077	100 %

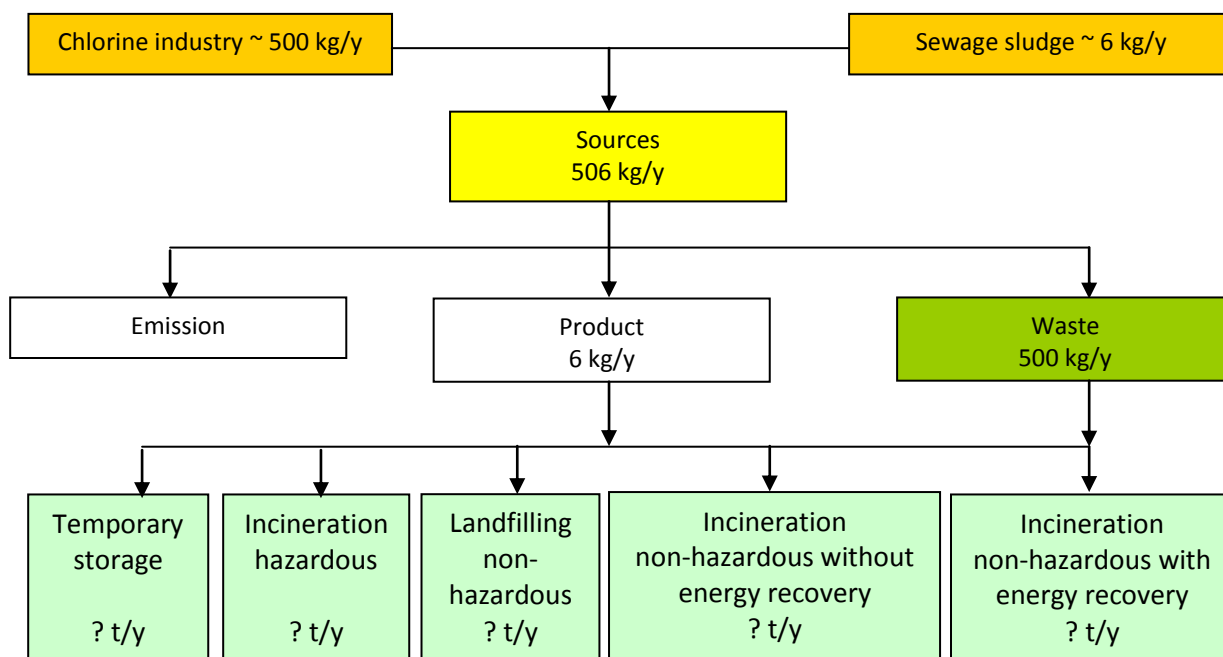
It has to be mentioned that due to missing availability of data no ranges could be indicated for PCP concentrations in the above specified contaminated materials. The indicated PFOS concentration value, which has been used for the calculations, is based on calculations derived from available data on PCP.

Based on the data of the detailed mass flows in chapter 6.7 and the contamination data compiled in Table 5-25 it can be concluded that the PCP concentration of PCP contaminated wastes is very different. Contamination of PCP impregnated wood can range from several 100 ppm to over 50,000 ppm, whereas the PCP concentration of impregnated textiles is in the range of 25,000 ppm.

5.4.8 Substance flow HCBd

5.4.8.1 Substance flow HCBd

Due to the little available data regarding HCBd in the different industry sectors no proper waste flow can be established.



The indicated sectors indicated in the literature are briefly discussed in the following. Details of the different sources are given in section 6.8.

Chemical Industry – Production of chlorinated solvents:

The known HCBd containing outputs can be emission through waste water or wastes. Regarding to the E-PRTR about 84 kg of HCBd have been emitted in 2004 via waste water. The wastes generated at a production site are nowadays considered to be either reused (100 % elimination) or incinerated. In case of incineration destruction over 99.9 % are mentioned in literature. Considering a worst case scenario it is assumed that 50 % is reused with 100 % elimination and that 50 % of the waste is incinerated and that only 99.9 % of the HCBd is destroyed which would leave at the most 500 kg of HCBd in the waste. It is expected that this waste type is typically ash. The concentration and therefore the total amount of effected ash as well as the effected waste type are not known.

Sewage sludge

According to the E-PRTR about 9.82 million tonnes of Sewage sludge contain about 5.89 kg of HCBd which gives an average concentration for 0.6 µg/kg.

Plastic Industry

According to the E- PRTR about 24 kg HCBd have been emitted from the plastic industry via waste water in 2008. This type of emission is referred to the production of basic plastic materials. However the information and data are too vague to be used for proper waste amount estimation and therefore this sector is not analysed. This means that the HCBd amount from this industry is underestimated.

Non-ferrous Metal Industry- Magnesium Production

In EU 27 no production of primary magnesium takes place. Therefore, no unintentional production of HCBd is expected via the relevant process of magnesium electrolysis. Related emissions and waste arising due to production of magnesium do not occur.

Hazardous waste incineration

No information has been found in the literature that HCBd is contained in solid residues resulting from hazardous waste incineration. According to the UNECE 2007 report HCBd emissions to air decreased dramatically in the last years due to the use of incineration. Therefore it is not considered that hazardous waste incineration is a relevant HCBd source.

5.4.9 Substance flow PCN

5.4.9.1 Substance flow PCN

The following figure shows the overall result of the detailed PCN mass flow analysis. The individual streams from each activity are listed in chapter 6.4.9 to 6.4.13 and are discussed for each activity in the corresponding detailed PCN mass flow, as far as available. Sectors not covered with respect to the PCN mass flow are discussed in chapter 6.9.6. Based on the information presented the most significant processes are covered in the total mass flow shown in Figure 5-33.

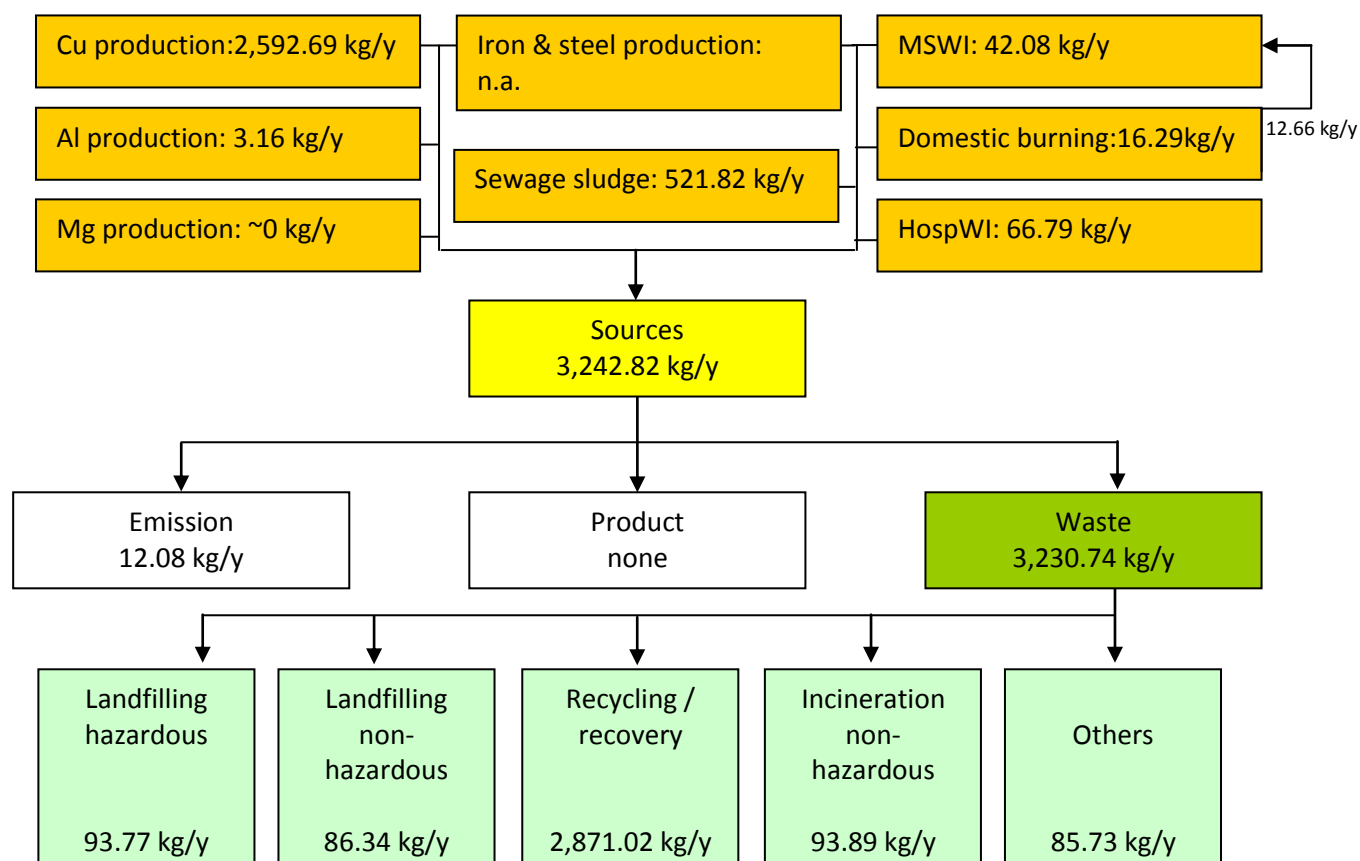


Figure 5-33: Overview of PCN substance flow

It has to be taken into account that due to limited data for most of the relevant sectors the mass flow is mostly based on assumptions and extrapolation. For the sector concerning the iron and steel production only marginal information is available. For this reason this sector could not be included in the following summarising overviews, although it is considered as a relevant contributing sector. Concerning the magnesium production the emissions to environment and waste are zero since the relevant production has stopped in EU27 (in France: 2002, in Norway: 2003) [IMA 2005].

Table 5-26 shows the contribution of the different sectors to the European PCN flow to emissions, products and waste on the basis of the available data.

Table 5-26: Contribution of the relevant sectors in the EU 27 to PCN in emissions, products and waste

Sector/Activity	Emission [kg/y]	Product [kg/y]	Waste [kg/y]	Sum [kg/y]
Municipal solid waste incineration	4.31	0	37.77	42.08
Hospital waste incineration	0.15	0	66.64	66.79
Domestic burning	6.13	0	10.16	16.29
Sewage sludge	0	0	521.82	521.82
Secondary copper production	0	0	2,592.69	2,592.69
Secondary aluminum production	1.49	0	1.67	3.16
Total	12.91	0	3,233.24	3,242.82

As illustrated the overall discharge of PCNs from the investigated sources in Europe accounts for about 3,250 kg/y. Approximately only 0.4 % of the releases are emitted to air and soil and a majority of approximately 99.6 % of the releases end up as waste. To conclude it can be stated that the sources of PCNs are dominated by the secondary copper production sector. A total quantity of ~80 % originates from this sector as is described in Figure 5-34.

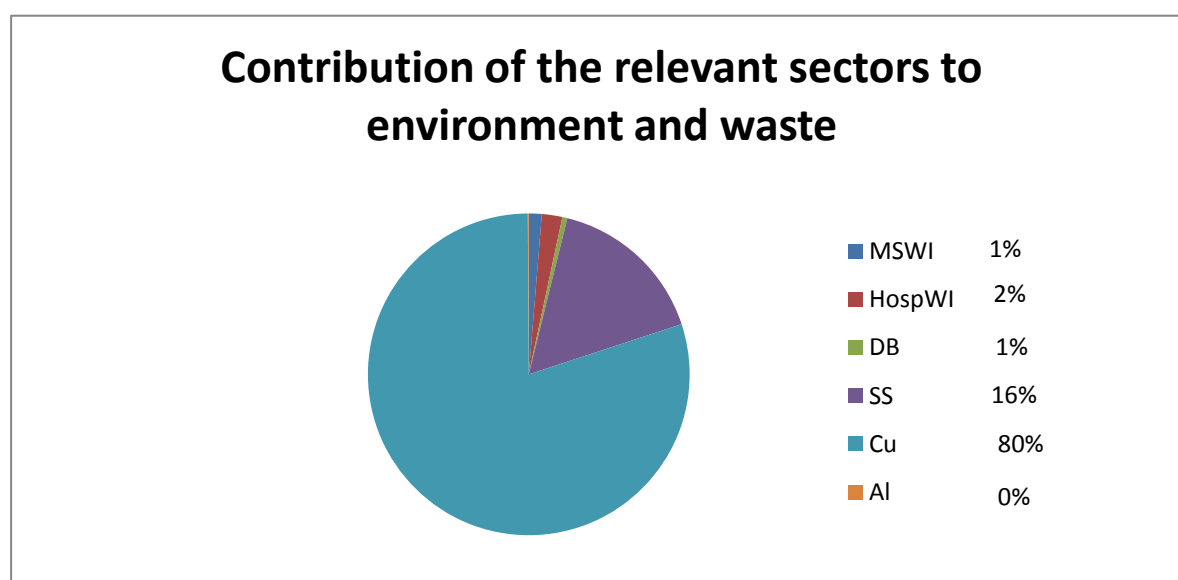


Figure 5-34: Contribution of the relevant sectors to the EU 27 to PCN emissions to environment and discharge to waste

In order to illustrate the relative importance of different sectors on air emissions and discharge to waste, the shares of the investigated sectors are illustrated in the following pie charts.

As illustrated in Figure 5-35 below, the most important source for emission to the environment is domestic burning of solid fuels, wood and mixed wastes with ~51 %, followed by municipal solid waste incineration (~36 %), secondary aluminium production (~12 %) and hospital waste incineration (~1 %). The share of emissions due to waste water treatment and secondary copper production is closed to zero. All PCN emissions are emissions to air, except emissions due to domestic burning. From PCNs released due to domestic burning ~55 % is released to soil. In regard to the total PCN amount of 6.13 kg emitted to the environment from all relevant sectors the share of PCN released to soil from domestic burning amounts to ~51 %.

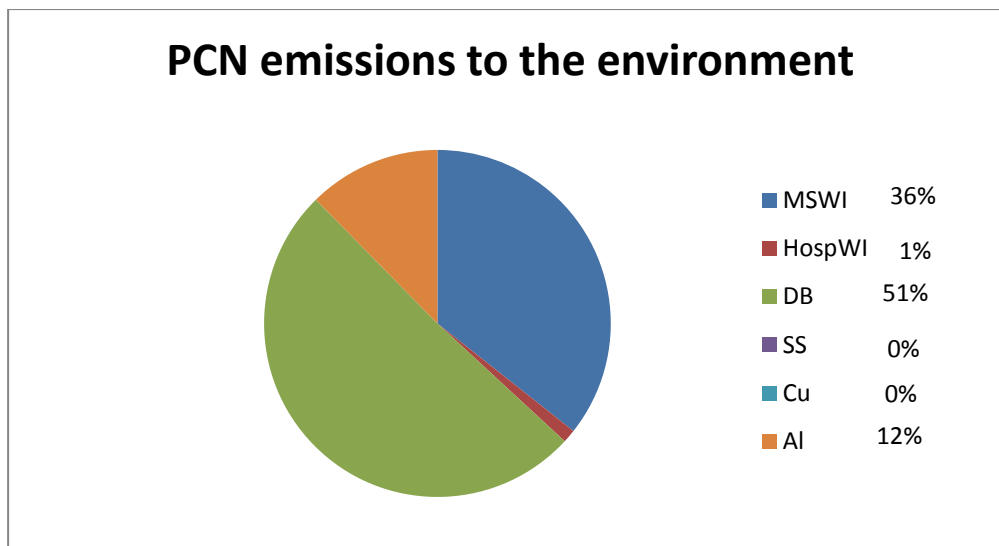


Figure 5-35: Relative distribution of PCN emissions to the environment from investigated sectors in EU 27

Unlike emissions to air discharges to waste show a more uneven distribution for the investigated sectors. Based on the results from the mass flows in chapter 6.9 the most important sources for discharge to waste is the secondary copper production with ~80 %, followed by waste resulting from waste water treatment (~16%). PCN waste arising from the other sectors account for only ~4 % in total, composed of municipal solid waste incineration (1.3 %), hospital waste incineration (2.1 %), domestic burning (0.5 %) and secondary aluminium production(0.1 %). The shares of PCN discharge to waste from the relevant sectors are shown in Figure 5-36.

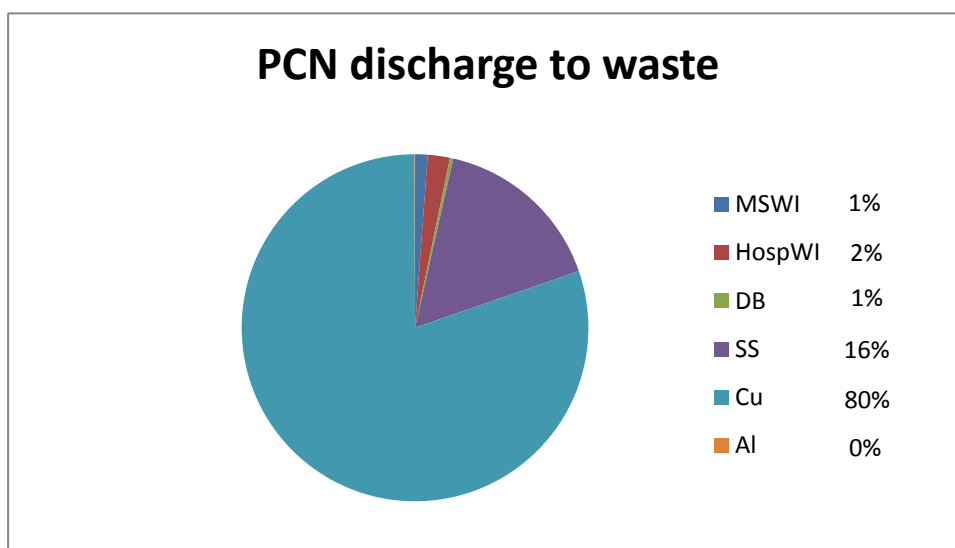


Figure 5-36: Relative PCN distribution to waste from investigated sectors in EU 27

PCNs discharged to waste are directed to different waste management operations including disposal and recovery operations. As the environmental relevance of the different operations differs, it is of interest to trace the amounts of each “source” sector directed to specific management sectors and to distinguish the relative share of each operation in relation to the overall PCN discharge to waste. Table 5-27 illustrates how the selected activities contribute to the PCN flow to specific disposal/recovery

operations, whereas Figure 5-37 illustrates the importance of current waste treatment options with respect to the PCN mass flow.

Table 5-27: Quantitative contribution of PCNs from the different sectors to currently used disposal/recovery operations in EU 27

Treatment / Activity	kg/y	Comment
Non Hazardous waste incineration (NHWI)		
Domestic burning	2.03	75 % of 2.28 Mio t, i.e. 1.71 Mio t, of (fly) ash reaches the MSW stream; thereof 20 % (50.84 Mio t waste with 2.03 kg PCN) is subject to MSWI (mean 1.19 ppb)
Sewage sludge	91.86	2.03 Mio t sludge (mean 45.25 ppb)
Total	93.89	
Landfill for hazardous waste (including underground storage) (LHW)		
Municipal solid waste incineration (MSWI)	5.28	4.11 Mio t bottom ash (mean 1.28 ppb)
Municipal solid waste incineration (MSWI)	22.03	1.10 Mio t fly ash (mean 20.03 ppb)
Hospital waste incineration (HospWI)	0.09	69.17 kt bottom ash (mean 1.28 ppb)
Hospital waste incineration (HospWI)	66.37	12.20 kt fly ash (mean 5.44 ppm)
Total	93.77	
Landfill for non hazardous waste (LNHW)		
Domestic burning	4.06	75 % of 2.28 Mio t, i.e. 1.71 Mio t, of (fly) ash reaches the MSW stream; thereof 40 % (103.96 Mio t waste with 4.06 kg PCN) is landfilled as non-hazardous waste (mean 2.38 ppb)
Sewage sludge	82.27	1.98 Mio t sludge (mean 41.55 ppb)
Total	86.34	
Recycling / recovery (RR)		
Municipal solid waste incineration (MSWI)	9.92	7.73 Mio t bottom ash (mean 1.28 ppb)
Municipal solid waste incineration (MSWI)	0.54	0.03 Mio t fly ash (mean 17.93 ppb)
Hospital waste incineration (HospWI)	0.08	58.81 kt bottom ash (mean 1.28 ppb)
Hospital waste incineration (HospWI)	0.10	0.02 kt fly ash (mean 5.17 ppm)
Sewage sludge	266.03	5.60 Mio t sludge applied for agricultural use and compost, i.e. application to land (mean 47.51 ppb)
Secondary copper production (Cu)	2,592.69	613.8 kt slag (mean 4.22 ppm)
Secondary aluminium production (Al)	1.67	43.35 kt filter dust (mean 38.52 ppb)
Total	2,871.02	
Others (O)		
Domestic burning	4.06	75 % of 2.28 Mio t, i.e. 1.71 Mio t, of (fly) ash reaches the MSW stream ; thereof 40 % (103.96 Mio t waste with 4.06 kg PCN) is destined for MBT, composting, etc. (mean 2.38 ppb)
Sewage sludge	81.66	1.96 Mio t sewage sludge treated by different options, e.g. temporary storage (mean 41.66 ppb)
Total	85.73	
Total	3,230.74	

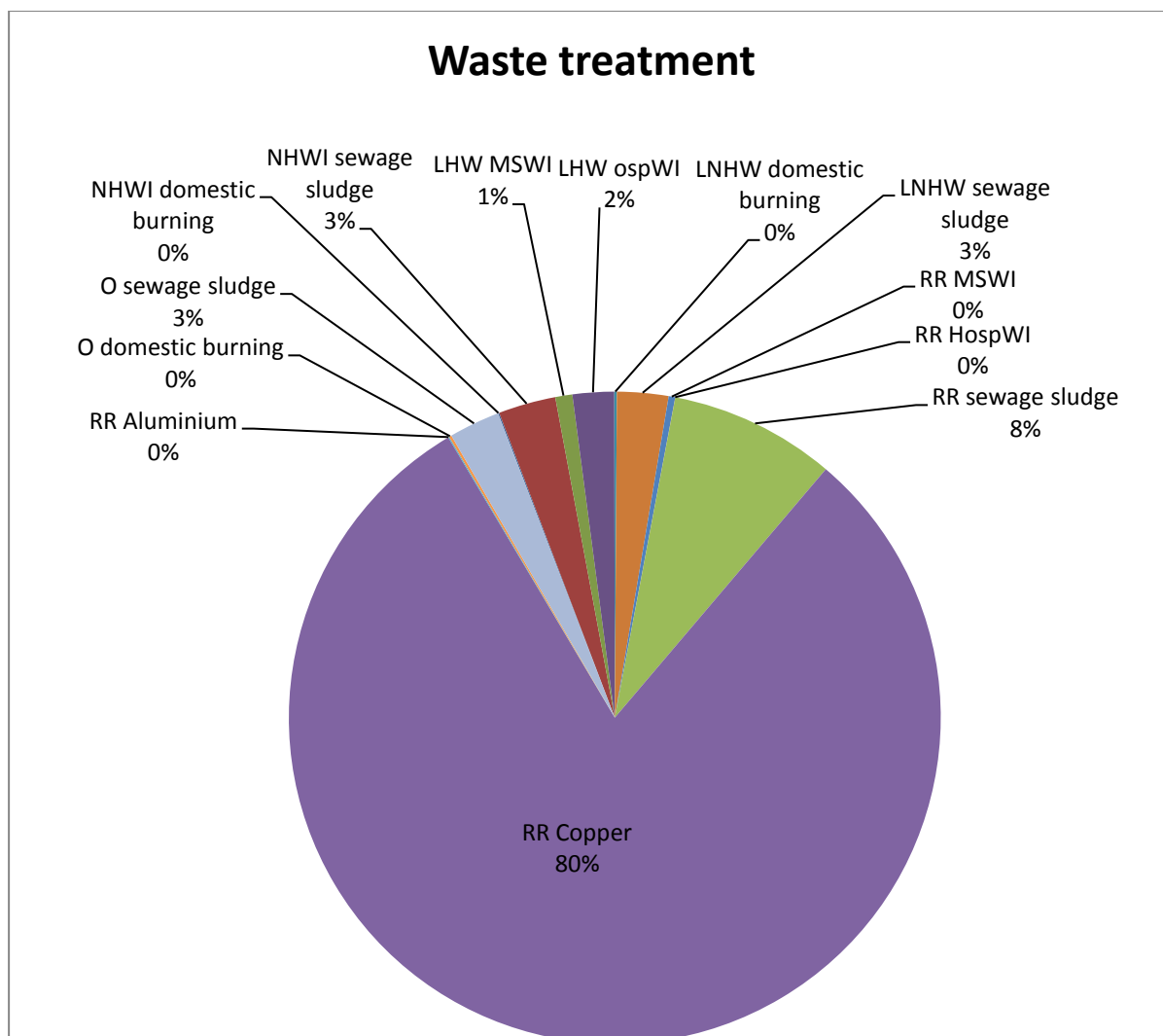


Figure 5-37: Relative importance of different waste treatment options with respect to the PCN mass flow in EU 27

5.4.9.2 Mass flows (articles/waste) related to PCNs

While the relative contribution of industrial, public and domestic sectors on the overall POP release is interesting for the assessment of potential environmental impact and the assessment of the overall flow and the effectiveness and significance of potential regulation, important information on the potential effects impacts and implications of potential regulation can only be drawn when the amounts of concerned waste streams are known.

Thus, besides the mass flows on POPs, the waste flows have to be established for each investigated sector as well. As for the POP mass flows, detailed information is contained in the corresponding paragraphs of chapter 6.4.9 to 6.4.13. Here a compilation of the related waste streams and corresponding contamination levels on the EU 27 scale is given. The corresponding contribution to the overall discharge of HCBd is added for better orientation.

Information concerning effects of proposed limit values is compiled and discussed in chapter 9.

Table 5-28: Annual amounts of PCN containing residues (wastes) and PCN contamination ranges

Sector/activity	Waste	Amount [Mio t/y]	Contamination data [ng/g]			PCN discharge [kg/y]	PCN to waste [kg/y]	Share of total PCN to waste [%]
			mean	min	max			
MSWI	Fly ash	1.13	19.95	0.05	87	22.6	22.6	0.70
	Bottom ash	11.84	1.28	0.95	2.0	15.2	15.2	0.47
HospWI	Fly ash	0.01	5,439	-	-	66.5	66.5	2.06
	Bottom as	0.13	1.28	0.95	2.0	0.16	0.16	0.01
Domestic burning	Ash from wood	0.55	7.60	-	-	4.2	3.1	0.10
	Ash from coal	1.70	5.40	2.5	8.3	9.2	6.9	0.21
	Ash from mixed wastes	0.03	6.00	-	-	0.18	0.13	0.004
Sewage sludge	Sewage sludge	11.58	40.25	3.2	76	522	522	16.15
Secondary copper production	Slag	0.61	4,224	-	-	2,593	2,593	80.25
Secondary aluminium production	Filter dust	0.04	38.43	6.9	92	1.67	1.67	0.05
Total		27.63				3,234	3,231	100

Based on the data of the detailed mass flows in chapter 6.9.9 to 6.9.15 and the contamination data compiled in Table 5-28 an overview of current levels of contamination as well as their corresponding ranges is illustrated in Figure 5-38 below.

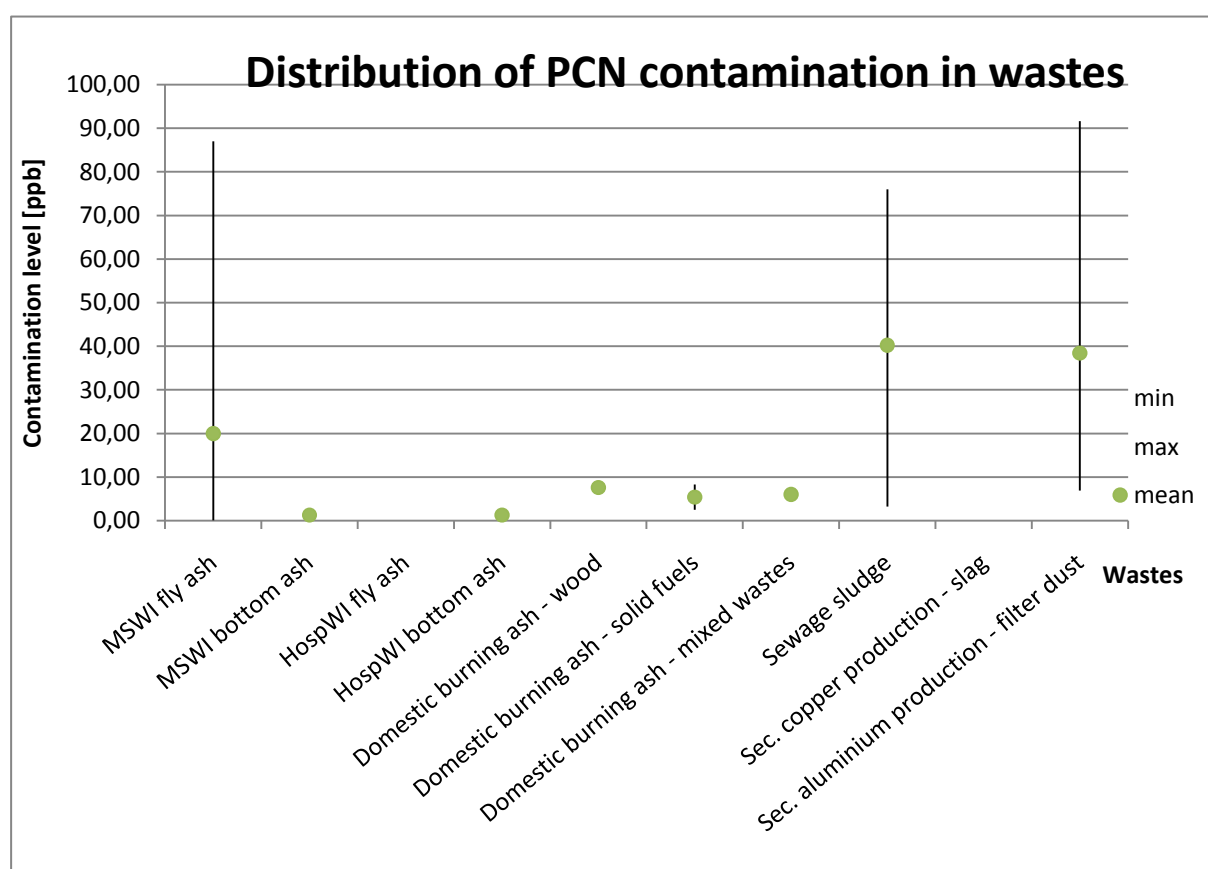


Figure 5-38: Means and ranges of recent H contamination in different waste types in EU 27

As shown in the Figure 5-38 and summarised in Table 5-28 (based on available data) mean concentrations and maximum limits of most waste streams lie below 100 ppb, whereas exceptions are the case for fly ash from hospital waste incineration (5,439 ng/g) and slag generated during secondary copper production (4,224 ng/g). For these two types only one contamination value each, have been found in the literature. Further, investigation in these sectors would be required in order to verify these values or to enable the presentation of a representative range. Approximately half of the PCN concentrations lie below 10 ppb, whereas the same range has been applied to bottom ash from hospital waste incineration as for bottom ash resulting from municipal solid waste incineration in order to enable calculations. High volume waste streams, i.e. sewage sludge and bottom ash from municipal solid waste incineration exhibit medium to low contamination values of 40.25 and 1.28 ng/g, respectively. To conclude, PCN contents in wastes containing the substance are very diverse, whereas important high volume waste streams exhibit relatively low contamination levels.

5.5 Evaluation of the data quality and uncertainties

The mass flows have been based on recent data from various sources. Sources include primary information sources such as measurement data reported from the relevant industry stakeholders and waste management facility operators as well as available statistical and literature sources (e.g. scientific publications, BREF documents, EUROSTAT data or EU reports as well as reports from international bodies such as UNEP or UNECE).

On the basis of these sources, available information is compiled in the corresponding chapters of the present report in particular in chapter 6 on the mass flows related to the individual new and candidate POPs. The information sources are referenced and are listed in the reference lists for each individual POP (see corresponding sub-sections in chapter 6) and in a general reference list (see chapter 10.1).

The establishment of the mass flows for substances and wastes is based on the following 4 categories of basic input data as far as available:

- activity data (e.g. production or use figures for industrial activities, amounts of waste treated, amounts of sewage sludge generated, amounts of waste arising e.g. from ELVs, EEE, construction & demolition etc.)
- factors for the generation of wastes (based on process input and output data)
- average POP contamination levels of input and output flows
- waste treatment data (e.g. amounts of waste landfilled, incinerated and recycled))

It was possible to collect a wide range of reliable basic input data. However due to the fact that the new and candidate POPs are not yet or not yet long in the focus of political interest and sometimes high costs for analysis, the data base is sometimes incomplete and data may not be representative for the typical European situation.

Consequently, the flow assessments are related to specific uncertainties in particular due to:

- incomplete or not reliable activity data for the relevant time frame
- missing or incomplete information on generated waste
- missing or only few data on contamination levels and ranges of relevant materials
- missing or incomplete information on actual waste treatment activities

According to the availability of information, the data background is different for each individual new or candidate POP. The data quality depends on the completeness and reliability of the data for each of the 4 input data categories for each substance and application. The following table gives an overview of how the project team considers the data quality for the individual substances and applications.

Table 5-29: Indicative assessment of the data quality for new and candidate POPs for relevant applications

Substance	Application	Activity data	Generation factors	Contamination levels	Waste treatment
C-PentaBDE	automotive	medium	good	medium	medium
	upholstery	medium	good	medium	medium
C-OctaBDE	EEE	medium	good	good	medium
	Printed circuit boards	low	good	low	medium
PFOS	metal plating	good	medium	medium	low
	photographic	good	medium	medium	low
	hydraulic fluids	good	good	good	medium
	fire fighting foams	medium	good	medium	medium
	past use in leather	medium	medium	low	medium
	past use in carpets	medium	medium	medium	medium
	sewage sludge	good	n.r.	medium	good
PeCB	MSWI	good	medium	medium	medium
	HWI	good	medium	medium	medium
	power from coal	good	medium	medium	medium
	Sewage sludge	good	n.r.	low	good
	domestic burning	low	low	low	medium
SCCPs	rubber	medium	good	medium	low
	sealant and adhesives	medium	good	medium	low
	paint and varnishes	medium	good	medium	low
	textiles	medium	good	medium	medium
	leather	medium	good	medium	medium
	sewage sludge	good	n.r.	medium	medium
HBCD	EPS construction	good	good	good	medium
	XPS construction	good	good	good	medium
	EPS/XPS other	medium	good	good	low
	HIPS	good	good	good	low
	textiles	good	good	good	low
PCP	wood	medium	good	medium	good
	textiles	medium	good	medium	medium
HCB	chemical industry	low	low	low	low
	sewage sludge	good	n.r.	low	good
PCN	MSWI	good	medium	medium	medium
	hospital WI	good	medium	low	medium
	domestic burning	low	low	medium	medium
	sewage sludge	good	n.r.	medium	good
	secondary copper production	medium	medium	low	low
	secondary aluminium production	medium	medium	medium	medium
	iron and steel sinter plants and electric arc furnaces	good	medium	low	medium

The figures and results given in the present report have to be interpreted against the background of the individual data quality.

Calculations are usually based on mean values. Ranges and Member State specific values can be found – if available and considered relevant – in the report. Information on activity data, contamination levels, waste generation and treatment used for mass flow calculations are referenced in detail in the corresponding subchapters.

Contamination levels often vary considerably according to the application of different processes and technologies. In reality contamination levels often show wide ranges. Frequently they are reported in different ways and the way of reporting does not always enable to derive specific mean or median values or a specific range. Ranges e.g. for contamination are indicated in the report and reflect the uncertainties which are related to the corresponding results. The calculated figures shall represent the best estimate based on given or estimated averages. Therefore it has to be kept in mind that the estimated POP mass flows are related to a certain, more or less wide range.

It is not considered reasonable to calculate the mass and waste flows on the basis of ranges. These are therefore calculated on the basis of “best estimates”. As a consequence precise figures in waste flows do not represent an accurate result but are related to specific uncertainties which are reflected in the indicated ranges of the background data.

However, results provide for the first time an overview of the dimension and distribution of contamination of waste with new and candidate POPs and are a basis for the discussion on feasible limits for POP waste in Europe and under the Stockholm Convention and their possible impacts.

The developed methodology is flexible and open to new information, thus providing a valid tool to regularly update the mass flows with upcoming knowledge and to refine impact assessment and scenarios.

6 Detailed mass flow analyses

6.1 C-PentaBDE flows

6.1.1 Use and Production

Polybrominated diphenyl ethers (PBDEs) are a group of 'additive' brominated flame retardants, which have been used in various applications all over the world. PBDEs have been used in cars, furniture, textiles, electrical and electronic equipment, etc. 'Additive' in this connection means that the flame retardants are blended with the product rather than covalently bound into plastics as in the case of 'reactive' flame retardants. Additive flame retardants are incorporated into the polymer prior to, during, or after polymerisation. This makes them, however, more prone to be released into the environment. PBDEs were typically produced at three different degrees of bromination, in particular PentaBDE, OctaBDE and DecaBDE [Alaee et al. 2003; Prevedouros 2004a; SFT 2009].

C-PentaBDE was produced in Israel, Japan, U.S. and the EU. It is possible that China produced for their market as well. Production in the EU ceased in 1997. It is assumed that since the late 1990's C-PentaBDE was mainly produced in the U.S. The sole producer of C-PentaBDE in the U.S (Great Lakes Chemical Corporation; now Chemtura) voluntarily ended the production in 2005. It is assumed that before the phase-out in U.S. more than 97% of C-PentaBDE manufactured worldwide was used in North America [SFT 2009]. In general it can be stated that since 2001 several countries world-wide introduced actions to regulate or voluntarily phase-out C-PentaBDE [Penta_UNEP 2006].

Currently, there is no production of C-PentaBDE in Europe, Japan, Canada, Australia and the U.S. Possibly, production is still located in other parts of the world. However, this is rather unlikely to occur, as no production or use has been reported by any existing bromine industry representations world-wide (e.g. in Asia, South America, etc.). In addition, the Bromine Science Environmental Forum (BSEF) confirmed that C-PentaBDE is no longer produced by at least BSEF member companies since 2004. It was further concluded that there is no production in the Eastern European countries outside the EU, at least by member companies of BSEF [SFT 2009].

6.1.2 Chemical Characteristics of Commercial PentaBDE

The POP regulation focuses on Tetra- and Penta-BDE in the sense of the definition of the Stockholm convention and therefore means 2,2',4,4'-tetrabromodiphenyl ether (BDE-47, CAS No: 40088-47-9) and 2,2',4,4',5-pentabromodiphenyl ether (BDE-99, CAS No: 32534-81-9) and other tetra- and pentabromodiphenyl ethers present in commercial pentabromodiphenyl ether (SC Decision 4/18).

Commercially available PentaBDE (C-PentaBDE) is not a pure substance but is instead a mixture of various congeners as shown in Table 6-1 [Risk 2000]. It refers to a mixture of two major congeners, i.e. tetrabromodiphenyl ether (BDE-47) and pentabromodiphenyl ether (BDE-99; BDE-100). These two congeners have the highest concentration by weight with respect to the other components of the mixture. In addition, trace amounts of tribromodiphenyl ether congeners (BDE-17; BDE-28) and possibly

other congeners may occur, as indicated in Table 6-1. The composition of the mixtures may vary in different regions of the world [SFT 2009; Penta_UNEP 2006].

Table 6-1: Composition of C-PentaBDE [Canada 2006; SFT 2009]

Categories of PBDEs	Tribromodi-phenyl Ethers		Tetrabromodi-phenyl Ethers	Pentabromodi-phenyl Ethers		Hexabromodi-phenyl Ethers	Heptabromodi-phenyl Ethers
CAS No	49690-94-0		40088-47-9	32534-81-9		36483-60-0	68928-80-3
Congeners	BDE-17	BDE-28	BDE-47	BDE-99	BDE-100	BDE-153	BDE-154
Content	Traces		Major	Major	Minor	Minor	Traces
	[~0-1%]		[~24-38%]	[~50-62%]		[4-12%]	[<1%]
Used for calculations			[31%]	[56%]		[8%]	[1%]
			Σ (Tetra-, Penta-, Hexa- and HeptaBDE) ~96%				

At this point it is important to notice that various synonyms and abbreviations are applied within literature. The inconsistency as well as the large variety of synonyms and abbreviations makes it rather difficult to distinguish between individual congeners, congener groups and commercial mixtures. Therefore, for the purpose of this report the abbreviation C-PentaBDE is exclusively applied for the commercial PentaBDE mixture. Congener groups are groups of homologue congeners and are designated according to their degree of bromination e.g. as TetraBDE, PentaBDE, HexaBDE or HeptaBDE. Individual congeners are specifically designated within the report, where appropriate (e.g. BDE-47, BDE-99 etc.).

6.1.3 Environmental Background Levels

C-PentaBDE has spread widely in the global environment. Existing monitoring data show that levels of the substance were detected in atmosphere, biota (marine and terrestrial birds, sea and terrestrial mammals, seafood and fish) as well as in sediments and soil. A comprehensive overview of monitoring data in abiotic media and biota is given in [Penta_UNEP 2006] and [Watson et al. 2010].

Table 6-2 gives an overview of the concentration levels in sediment and soil detected in different parts of the world.

Table 6-2: Levels of PentaBDE (BDE-99) in sediment and soil (DW: dry weight) [Penta_UNEP 2006]

Country/Region	Levels of PentaBDE	References	Comments
Sediment			
Sweden	<0.7-51.4 ng/g DW	[Palm et al. 2002]	Rivers at point source
Western Europe	<0.2-6.9 ng/g DW	[Palm et al. 2002]	Estuaries
Japan, Osaka	9-28 ng/g DW	[Palm et al. 2002]	-
Soil			
United Kingdom	0.078 – 3.2 ng/g DW	[Hassanin et al. 2004]	-

Table 6-3: Hot spot levels of BDEs (total BDEs including BDE-47, BDE-99 and BDE-100) in soil and sediment collected near a polyurethane foam manufacturing facility, levels cited by [Watson et al. 2010]

Country/Region	Congeners	Levels	References	Comments
Sediment				
USA	Total BDEs (sum of BDE-47, -99, -	ND to 132 ng/g dw	[Hale et al. 2002]	Sediment from stream leaving foam plant.

Country/Region	Congeners	Levels	References	Comments
	100)			
USA	Total BDEs (sum of BDE-47, -99, -100)	ND to 0.50 ng/g dw	[Hale et al. 2002]	Pond sediment
Not mentioned.	23 PBDEs (from mono- to hepta-BDEs and Deca-BDE).	4 to 407 ng/g dw (average 125.2 mg/kg) [sic!]	[Wong et al. 2007] as cited by Watson	It is not clearly indicated if µg/kg or mg/kg is correct.
Soil				
USA	Total BDEs (sum of BDE-47, -99, -100)	76.0 ng/g dw	[Hale et al. 2002]	Soil near foam production building
USA	Total BDEs (sum of BDE-47, -99, -100)	ND to 13.6 ng/g dw	[Hale et al. 2002]	Soil downwind of foam plant.
Guiyu area	PBDE (not further specified)	1,169 ng/g dw	[Leung et al. 2006]	Polluted area from e-waste recycling.
Guiyu	Total PBDE (not further specified)	Up to 44,400 ng/g	[Yu et al. 2008]	DecaBDE was the main congener with 31,200 ng/g.
Sewage sludge				
USA	Total BDEs (sum of BDE-47, -99, -100)	1540 ng/g dw	[Hale et al. 2002]	-
USA, Minnesota	Total PBDEs (not further specified)	5,300 ng/g dw	[Oliaei et al. 2002]	Samples were taken from sewage sludge from two waste water treatment plants that received landfill leachate.
Sludge (not further specified)				
USA	BDE-209	1,470 ng/g	[Hale et al. 2002]	BDE-209 was only detected in the sludge samples. Dw not particularly mentioned.

Within the study of Hale et al. (2002), as cited by Watson et al. (2010), BDE-209 was only detected in the sludge samples also collected near a polyurethane (PUR) foam manufacturing facility at relatively high levels up to 1,470 ng/g.

Oliaei et al. (2002) reported PBDEs concentrations up to 5,300 ng/g DW in their study in which samples were taken from sewage sludge from two waste water treatment plants that received landfill leachate. In the study, it was observed that the levels detected inter alia in sediment samples from rivers receiving the related wastewater effluent discharges were up to 1,950 ng/g TOC.

The Environmental Protection Agency in Ireland has recently undertaken sampling of wastes from sewage treatment to determine concentration levels of the new and candidate POPs in those wastes. Samples were obtained from 9 waste water treatment plants located in the Republic of Ireland. According to [EPA IRE 2011a] no Penta-, Hexa- or HeptaBDE were measured above the limit of detection in any of the sewage sludge samples (LOD ranging from <1 µg/kg – 10 µg/kg). Trace contaminations of TetraBDE were detected in some samples, however the concentrations measured are well below the lower limit of the LPCLs proposed in the interim POP study report. For PBDE analysis the extraction method was solvent extraction and analysis standard is apparently comparable to ISO22032:2006.

Leung et al. (2006) observed in their study that Guiyu area has significantly higher contamination levels due to e-waste recycling. Leung et al. detected maximum concentration levels up to 1,169 ng/g DW in

soils near the recycling areas. The determined levels were 10 to 60 times higher than PBDE contamination levels reported for other parts of the world. The continuation of the work of Leung et al. in 2007 [Leung et al. 2007] included surface soils and combusted residue in the Guiyu area. Total PBDE concentrations were highest in the combustion residue of plastic chips and cables which were collected from a residential area (33,000 to 97,400 ng/g DW). In soils from an acid leaching site the detected limits ranged from 2,720 to 4,250 ng/g DW, whereas a printer roller dump site showed levels from 593 to 2,890 ng/g DW. The authors found that DecaBDE was the dominant congener (35-82%) among the study site.

An illustration of congener patterns obtained from the sediment sample collected 5 m downstream of the effluent discharge in the study of [Eljarrat et al. 2007] is given in Figure 6-1. It was reported that BDE-209 is probably a source material for lower brominated products such as HexaBDE 154.

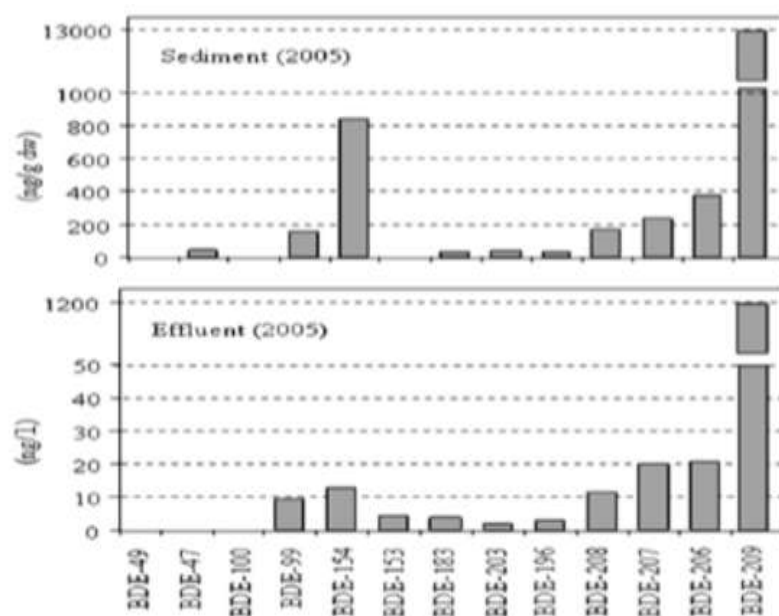


Figure 6-1: Comparison of congener patterns obtained from the sediment sample collected 5m downstream of the effluent discharge, and the effluent sample collected at the discharge to the river [Eljarrat et al. 2007]

As shown in Table 6-2, the detected background contamination levels in sediment and soil ranged from less than 0.2 ng/g in Western Europe to 51.4 ng/g in Sweden. Hot spot levels were detected in sediment, soil, sludge and sewage sludges from waste water treatment plants. The samples for the hot spot levels were taken from areas with high pollution, for instance due to recycling activities of e-waste. The detected levels reached up to maximum levels of 44,400 ng/g in soil. However, it has to be noted that in general total BDE or just PBDE was indicated in the context of hot spot levels. Therefore also other congeners as BDE 209 might be the main congener for the relevant studies. This was at least stated for the study of [Leung et al. 2007].

6.1.4 Legal Situation

6.1.4.1 EU level

POPs related provisions

PentaBDE is listed in Annex I, IV and V to the proposal for an amendment of the POPs Regulation. The Annex lists the individual PBDE congener groups tetraBDE, pentaBDE, hexaBDE and heptaBDE.

Annex I sets out values for unintentional trace contaminants – identically in place for all concerned congener groups – in substances, preparations, articles or as constituents of the flame-retarded parts of articles equal to or below 0.001 % by weight (10 ppm).

Market restrictions are listed in Annex I for all congeners. Production, placing on the market and use shall be allowed

- without prejudice to subparagraph (b), articles and preparations containing concentrations below 0.1% by weight (1,000 ppm) of all congeners by weight when produced partially or fully from recycled materials or materials from waste prepared for re-use;
- electrical and electronic equipment within the scope of Directive 2002/95/EC.

In Annex V, no upper and lower POPs limits have been set for the polyBDE congener groups.

Waste management – WSR

For the purposes of classifying plastic waste under Waste Shipment Regulation, it is common understanding that PBDE containing plastic waste is not to be classified as entry A 3180 (of “amber list”) and the respective threshold for PBBs, as brominated flame retardants PBDEs are not analogues of PBBs. Consequently, currently there is no threshold limit for wastes being listed “amber”, although it is reported by a Competent Authority that based on REACH and RoHS a threshold level of 0.1% by mass (1,000 ppm) is suggested for penta and octa BDEs.

The German competent authorities consider PBDEs as analogues of PBBs in the sense of entry A 3180, as during the evaluation of possible pollutants under health and environmental criteria above all the functional analogy is to be considered apart from structural similarities. Functional analogy means qualitatively comparable physiological effects such as enzyme induction, endocrine effects, immunosuppression etc. In the literature there are many references on the fact that PBDEs show such effects. This led in the long run apart from such characteristics such as bio-accumulation and bio-magnification as well as long range transport to the proposals to list PBDEs in appendix A to the Stockholm conventions. The German authorities are of the opinion, that structural analogy should not be seen to strict. In the German PCB/PCT waste act some selected halogenated monomethyldiphenylmethanes are e.g. expressively listed as “PCB in the sense of this act”.¹³

¹³ This paragraph is based on the German comments to the draft final report; e-mail from 17.03.2011

Waste management – WEEE

WEEE Directive 2002/96/EC aims at preventing WEEE by promoting reuse, recycling and recovery and is of major interest for the project. For the ten WEEE categories defined in Annex IA, the Directive sets targets for recovery and for component, material and substance reuse and recycling. As regards new POPs (PBDEs) the targets set for categories 3 and 4 are of relevance, requesting:

- A recovery rate of a minimum of 75 % by an average weight per appliance, and
- A reuse and recycling rate for component, material and substance of a minimum of 65 % by average weight per appliance

Furthermore according to Annex II “Selective treatment for materials and components of waste electrical and electronic equipment in accordance with Article 6(1)” as a minimum [...] [e.g. printed circuit boards] have to be removed from any separately collected WEEE.

Waste management – ELV

Directive 2000/53/EC sets out provisions with respect to end-of-life vehicles aiming at the prevention of waste from vehicles and, in addition, at the reuse, recycling and other forms of recovery of end-of life vehicles and their components.

As regards recycling and recovery the following targets are set pursuant to article 7:

Since 1 January 2006:

- Reuse and recovery of a minimum of 85 % by average weight per vehicle and year.
- Reuse and recycling of a minimum of 80 % by average weight per vehicle and year;
- Not lower than 75 % for reuse and recovery and not lower than 70 % for reuse and recycling for vehicles produced before 1 January 1980

From 1 January 2015:

- Reuse and recovery of a minimum of 95 % by an average weight per vehicle and year
- Re-use and recycling of a minimum of 85 % by an average weight per vehicle and year

In this context it is important to note that vehicles in average consist of 75% metal and 25% plastic compounds, but that only part of this is PUR foam prone for C-PentaBDE contamination.

Restrictions on marketing and use - REACH

Following Annex XVII, entry No 44 of REACH, “Diphenylether, pentabromo derivative (C₁₂H₅Br₅O)” may not be placed on the EU market or used as a substance or a constituent of substances or of preparations in concentrations higher than 0.1% (1000 ppm) by weight. The same applies for finished articles, which may not be placed on the EU market if they, or flame-retardant parts thereof, contain pentaBDE in concentrations higher than 0.1% (1000 ppm) by weight. This restriction is in force since August 2004.

Note: the PentaBDE congener group or “Diphenylether, pentabromo derivative (C₁₂H₅Br₅O)” as listed under REACH is also listed in the POPs Regulation as the specific PentaBDE congener group meets the POPs criteria on its own. Therefore PentaBDE will be removed from Annex XVII in order to maintain legal certainty. In the future, PentaBDE will be restricted/prohibited exclusively by the POPs Regulation.

Restrictions on marketing and use - RoHS

The RoHS Directive specifically relates to placing on the market of polybrominated diphenyl ethers in EEE (which explicitly is excluded from the restrictions laid down in REACH). Article 4(1) RoHS Directive, together with Annex 1, No. 29 bans the placing on the EU market of new EEE containing more than a maximum PBDE concentration value of 0.1% by weight (1000 ppm) in homogeneous material. This restriction is in force since July 2006 and applies to the sum of all PBDE congeners.

Annex 1 lists “Applications of (...) polybrominated diphenyl ethers (PBDE) which are exempted from the requirements of Article 4(1)” (i.e. from the market restriction). No. 10 of Annex 1 states that “Within the procedure referred to in Article 7(2), the Commission shall evaluate the applications for: - Deca BDE (...)”. By means of Decision 2005/717/EC, Annex 1 of Directive 2002/95/EC was amended by a point 9a., introducing a general exemption of Article 4(1) requirements for DecaBDE. However, the European Court of Justice has annulled the respective part of Decision 2005/717/EC with order of 1 April 2008, declaring that the effects would maintain until 30 June 2008 inclusive.

Note that restrictions apply to recycled products placed on the market which are not subject to waste regime any more as well.

Classification

PentaBDE (substance “diphenyl ether, pentabromo derivative” – CAS No 32534-81-9) is classified

- under CLP Regulation as specific target organ toxic, and acute and chronic aquatic toxic;
- under Directive 67/548/EEC as harmful and dangerous for the environment.

Other congener groups are not classified.

Other fields (water and food)

The Water Framework Directive lists OctaBDE and DecaBDE among relevant substances to be monitored and established Environmental Quality Standards (EQS), both for annual average (AA-EQS) and for Maximum Allowable Concentration (MAC-EQS) for inland surface water for “brominated diphenyl ethers” as follows:

- AA-EQS: 0,0005 [µg/l]
- AA-EQS: 0,0002 [µg/l]

PBDEs are no subject to EU regulations in the field of food and feed safety.

6.1.4.2 Member States level

Waste

Pursuant to the Austrian Waste Management Plan (Chapter 5.3 – version 2009) in cases where the content of 0.1% for the sum of penta-, octa- and decabromodiphenyl ether or the content of 50 ppm (=0,005%) for polybrominated biphenyls in wastes (homogenous material) is exceeded, a notification procedure is required in case of transfrontier shipment, independently from the subsequent recovery operation. If these limits are exceeded the environmentally sound recycling in other fields of application abroad shall be controlled. In case of the presence of higher contents of the above mentioned flame retardants, particularly when the content of octabromodiphenyl ether exceeds 0.5 % by weight (=5,000 ppm), a hazard characteristic (teratogenic) is triggered (ban of export on hazardous wastes to non-OECD Member Countries).

Water

For technical pentaBDE, Austria has laid down an emission limit of 0.5 µg/l for waste water from chemicals industry.

6.1.4.3 Other countries

In Norway, waste containing more than 0.25% (=2,500 ppm) of pentaBDE or octaBDE is defined as hazardous waste and has to be treated according to the regulation on hazardous waste. In Switzerland, Recycling of articles containing PBDEs is not allowed. In Korea, recycling of electronic and electric devices is only allowed if the concentration of penta- and octaBDE in EEE Product is below the threshold of 0.1 % (=1,000 ppm).

6.1.5 Occurrence of C-PentaBDE in Europe

The occurrence of C-PentaBDE in Europe can be considered its consumption. For the purpose of this study the term 'occurrence/consumption' has been defined as: (a) Use of the substance itself in the EU plus (b) Import of the substance in finished articles and materials minus (c) Export of finished articles and materials containing the substance.

- (a) Use

The global market demand for C-PentaBDE decreased from 8,500t in 1999 to 7,500t in 2001. Weber et al. (2010) estimated a higher figure of 8,300t/y for 2001. This figure has been derived from the total estimated distribution figure of ~34 million kg of PBDEs, of which C-PentaBDE made up ~8.3 million kg. The corresponding figures for Europe were indicated as 210t/y in 1999 and 150t/y in 2001 respectively [RPA 2000; Risk 2000; Penta_UNEP 2007; SFT 2009; Weber et al. 2010].

The use in Europe has been declining during the second half of the 1990's and was estimated to be 300t in 1994, used solely for PUR foam production. In addition, it was estimated that approximately 800t entered the EU in finished articles in the same year (see Table 6-4). Further information provided by the

main supplier of C-PentaBDE indicated a figure of 100-125t for use and approximately the same amount for imports in products in 2000. This was also supported by another supplier, presenting the overall figure for use and import of 250t/y [RPA 2000]. Table 6-4 summarizes the main European use and import figures of C-PentaBDE, followed by separate graphical overviews of the amounts used and imported over the time period 1994-2004.

Table 6-4: C-PentaBDE overview of use and imports for the EU

Year	Use [t]	Import in finished articles [t]	Literature source
1994	300	800	[RPA 2000]
1997		500	[Risk 2000]
1998		300	[Risk 2000]
1999	210	---	[Alcock et al. 2003]
2000	100 - 125	100 - 125	[RPA 2000]
2001	150	---	BSEF
2004	0	0	EU Directive 76/769/EC

Figure 6-2 provides a graphical overview of the amounts of C-PentaBDE used (use of the substance itself) in the EU from 1994 to 2004 based on the use figures listed in Table 6-4.

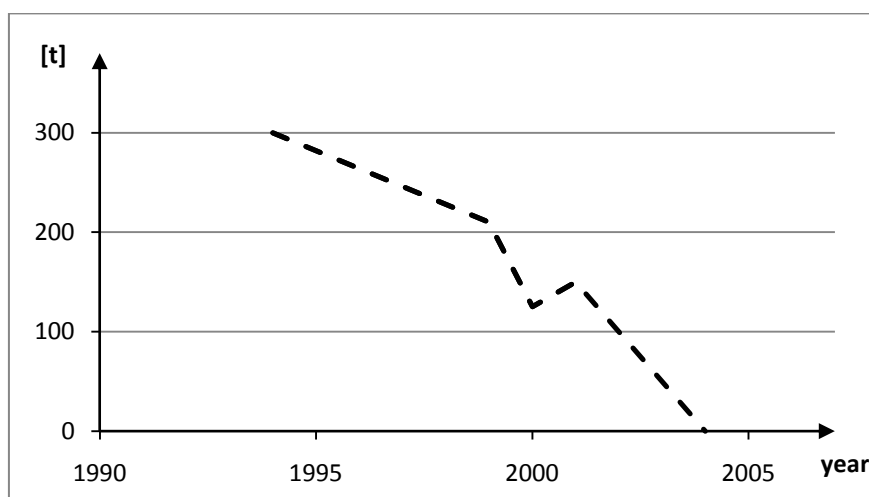


Figure 6-2: Overview of the amounts of C-PentaBDE used in the EU

- (b) Import

In order to be able to quantify the overall occurrence/consumption of C-PentaBDE it is important to assess the import figures. As mentioned previously the UK Environmental Agency estimate that 800t of C-PentaBDE was imported into the EU in finished articles in 1994. The UK Department of Environment, Transport, and the Regions estimated that 100-125t was still imported into Europe by the end of the 90's. Other scientific studies suggested that the import decreased to 500t in 1997, 300t in 1998 and approximately 100t in 2000 (see Table 6-4). However, it cannot be excluded that imported cars (e.g. from Asian countries) contained C-PentaBDE treated PUR foams also after 2000.

For the years before 1986 it was assumed that the imports progressively increased from 100t in 1980 to 430t in 1986. Besides, Prevedouros et al. (2004a) assumed that no imports of C-PentaBDE containing

articles occurred at all during the 70's. Figure 6-3 provides a graphical overview of the amounts of C-PentaBDE imported into the EU in finished articles from 1994 to 2004.

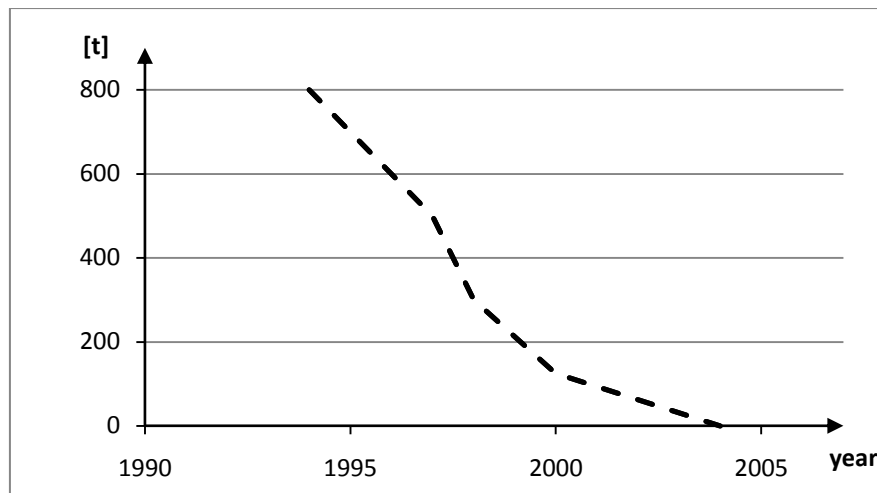


Figure 6-3: Overview of the amounts of C-PentaBDE imported in finished articles

- (c) Export

Prevedouros et al. (2004a) assumed that C-PentaBDE was not exported from Europe during the entire time period of their study (1970-2000). However, they acknowledge that exports, especially UK manufactured automobile applications, probably occurred during that time period. Further discussions on exports of C-PentaBDE containing articles will be presented in Chapter 6.1.9.

- Occurrence/Consumption

By combining the above presented use, import and export figures the occurrence/consumption of C-PentaBDE in Europe can be estimated. Prevedouros et al. (2004a) used seven scenarios to estimate the historic overall occurrence/consumption of C-PentaBDE in Europe (see Figure 6-4). In addition, two annual estimates made by DeWit (2002) and Alcock et al. (2003) for 1999 were included in the graph. The main parameters used to estimate the production figures were:

- reported world bromine production,
- percentage of the world bromine market attributable to BFRs,
- percentage of European consumption,
- fraction of the European BFR market comprised by PBDEs and
- fraction of the European PBDE market comprised by PentaBDE

The authors [Prevedouros et al. 2004a] concluded that the difference in the trend lines is always less than factor 2, in most cases only 1.5 (see Figure 6-4). Besides, most of the scenarios indicate that the peak of C-PentaBDE consumption occurred during the mid-1990s with approximately 1.050-1.250t/y. For the pre-1980 the overall consumption figures are low (see Figure 6-4), as it was assumed that insignificant amounts of C-PentaBDE treated articles were imported and only a small fraction of Bromine was used in the brominated flame retardant industry.

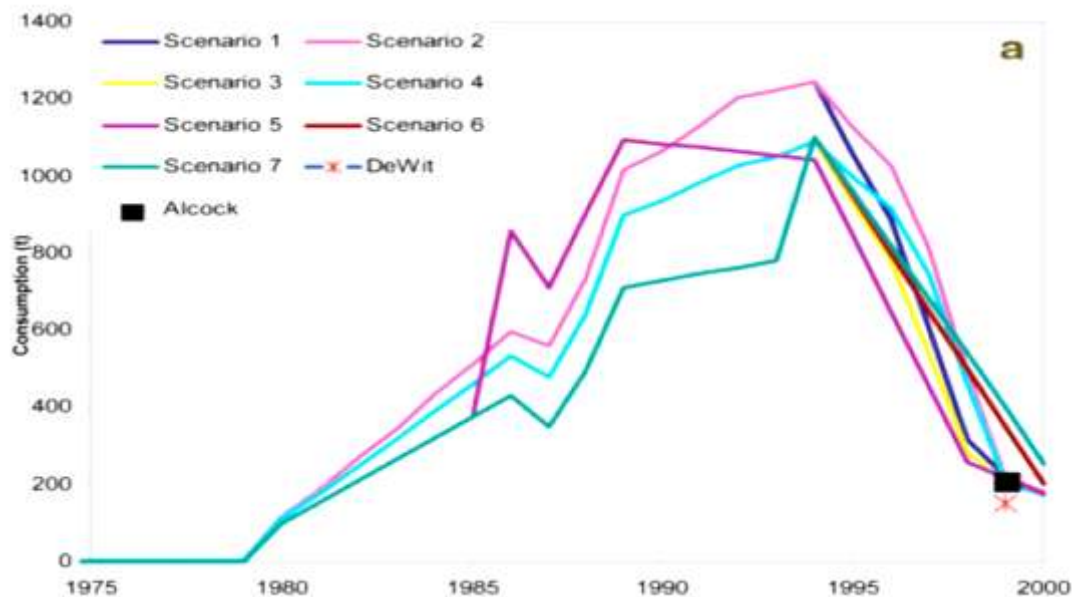


Figure 6-4: Estimated consumption of C-PentaBDE in Europe [Prevedouros et al. 2004a]

For almost the same time period 1970-2001 the Bromine Science and Environmental Forum (BSEF) estimated the global cumulative manufacture/use of C-PentaBDE as 100,000t. This figure is also supported by Watson et al. (2010), estimating the cumulative global production of C-PentaBDE as 91,000-105,000t for the time period 1970-2005.

Alcock et al. (2003) broke this figure further down into 85,000t of use in the US and the remaining 15,000t in Europe. Prevedouros et al. (2004a) estimated for the period from 1970-2000 a total production of 3,000-5,000t of the substance itself in Europe with further 9,000-10,000t of the substance imported in finished articles. This figure quite well corresponds to the cumulative use figure of 15,000t in Europe, as proposed by Alcock et al. (2003).

6.1.6 Relevance of the occurrence of C-PentaBDE in Europe

The historic uses, applications and finished articles containing C-PentaBDE are summarized in Figure 6-5. According to the EU Risk Assessment (2000) the most common use of C-PentaBDE in Europe was in flexible polyurethane foam (PUR foam), accounting for approximately 95% of C-PentaBDE usage. The PUR foam was in turn mainly used for upholstery and automotive applications (see Figure 6-5).

The treated PUR foam usually contained 10-18% of the commercial mixture [Penta_UNEP 2006]. Zweifel (2001) indicated a quite similar concentration range between 6-18% of C-PentaBDE in PUR foams. However, PUR foam manufacturers pointed out significantly lower concentration values, for instance only 2-3% in PUR foams for mattresses and 3-5% for PUR foams in upholstered furniture [VCCEPP Penta 2003]. Cambell & Chemtura (2010) confirmed that the levels of C-PentaBDE used in PUR foam for upholstered furniture ranged approximately between 3-5%, dependant on the PUR foam densities. The producers of automotive applications stated that 4% would be the most realistic concentration value for PUR foams used in the automotive sector [RPA 2000].

In general it can be assumed that exact specifications were frequently corporate secrets and therefore not always available for the general public. The available data, however, indicates that in general polybrominated biphenyl ethers (PBDEs, all congeners) were added at concentrations between ~5-30% by weight, dependant on different factors such as the required flame retardancy, the effectiveness of the flame retardant, synergist within the polymer system applied, physical characteristics of the end product (e.g. colour, density, stability, etc.) and the specific use of the end product [Risk 2000]. In the particular case of C-PentaBDE treated PUR foams it can be summarized that the concentration values found in scientific literature strongly differ from those provided by the PUR foam industry. Taking the whole range of concentration values into account it can be summarised that the concentrations of C-PentaBDE in treated PUR foams range between 2-18%.

Table 6-5 summarizes concentration values of C-PentaBDE in PUR foams (reported to be the most common use of C-PentaBDE).

Table 6-5: C-PentaBDE concentrations in PUR foams

Application/Articles	Concentrations of C-PentaBDE	Literature source
PUR foam	10-18%	[Penta_UNEP 2006]
PUR foam	6-18%	[Zweifel 2001]
PUR foam for mattresses	2-3%	[VCCEPP Penta 2003]
PUR foam for upholstered furniture	3-5%	[VCCEPP Penta 2003]
PUR foam for automotive applications	~4%	[RPA 2000]
PUR foam for upholstered furniture (density: 19kg/m ³)	5.45%	[Cambell & Chemtura 2010]
PUR foam for upholstered furniture (density: 24kg/m ³)	4.30%	[Cambell & Chemtura 2010]
PUR foam for upholstered furniture (density: 29kg/m ³)	2.77%	[Cambell & Chemtura 2010]

Figure 6-5 summarizes the historic uses of C-PentaBDE and indicates their relevance. It serves as a basis for the selection of relevant uses, applications and finished articles in the upcoming chapter.

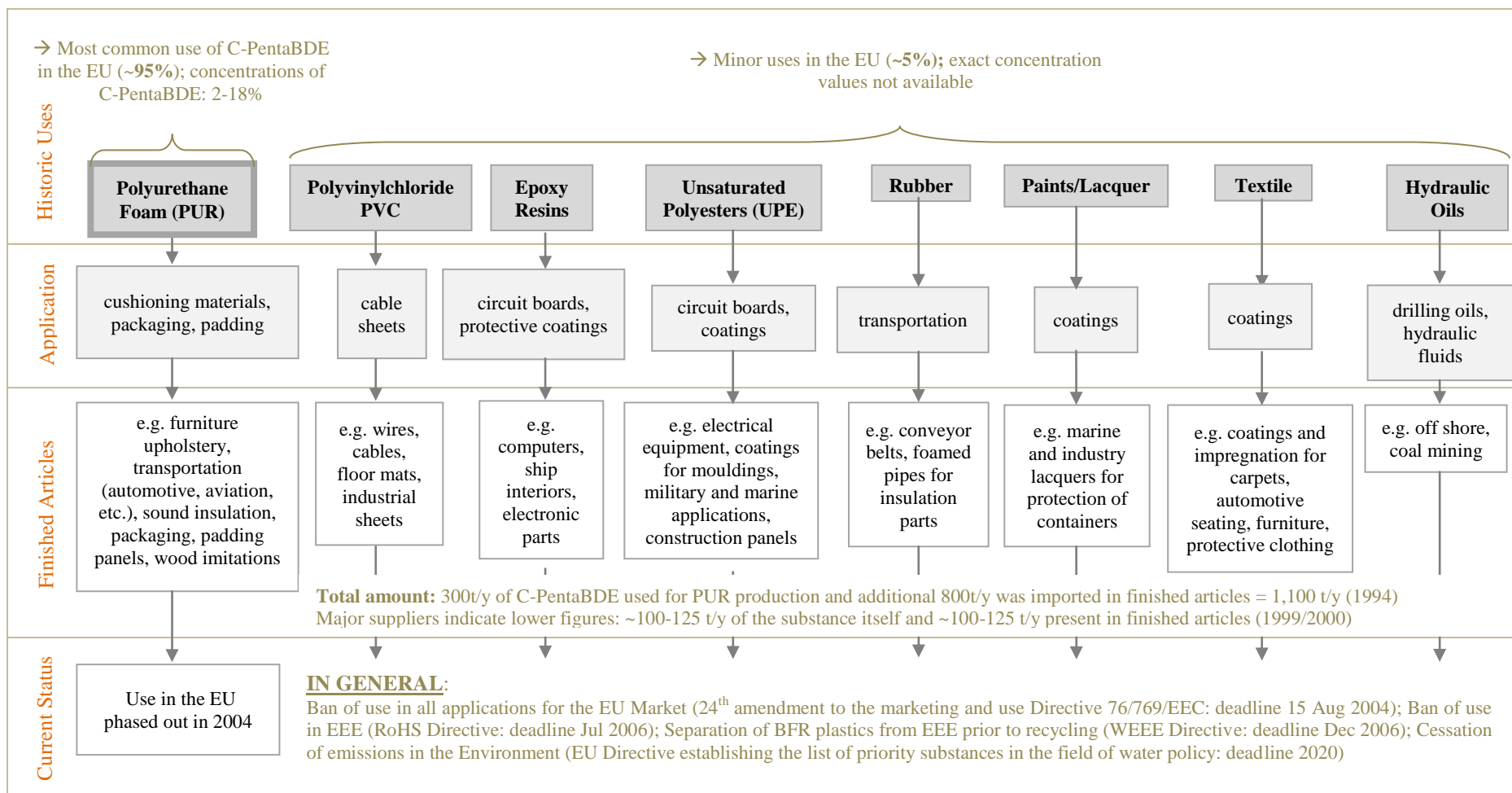


Figure 6-5: Overview of historic uses of C-PentaBDE and their relevance [inspired by SFT 2009]

6.1.7 Selection of Relevant Uses, Applications and Finished Articles

An important precondition for the selection of relevant uses, applications and finished articles is robust quantifiable data. Whilst it is widely accepted that the major use of C-PentaBDE was in flexible PUR foams, significant uncertainties have been identified with regard to the exact distribution of applications and finished articles associated with the uses presented in Figure 6-5.

As shown in Figure 6-5, approximately 95% of C-PentaBDE in Europe was used in flexible PUR foams. These foams were in turn mainly used for the production of automotive and upholstery applications. The use in PUR foam based packaging was estimated to be rather small in Europe, accounting for less than 5t/y [RPA 2000]. Besides, it is assumed that this packaging material has already been landfilled or incinerated. Therefore, this application for C-PentaBDE is not further investigated. Other applications and finished articles containing or reported to be possibly contaminated with C-PentaBDE are listed in Figure 6-5. Among these are Polyvinylchloride (PVC), Epoxy Resins, Unsaturated Polyesters (UPE), Rubber, Paint/Lacquers, Textiles, Hydraulic oils and possibly other [SFT 2009]. Considering the minor amounts of C-PentaBDE used (total ~5%), the diffuse distribution of such articles and the limited data availability, these applications are not further investigated.

Consequently, it would be obvious to focus exclusively on C-PentaBDE treated PUR foams and associated finished products. This decision would also be in line with several scientific studies focusing exclusively on this particular use, assuming that all of the C-PentaBDE in the EU was used in the production of PUR foams (e.g. Risk 2000). This is in fact true for the use of the substance itself, however as this study also aims to consider relevant imports of C-PentaBDE in finished articles and materials additional applications and finished articles could possibly gain in importance.

A Swiss study conducted in 2007 [GEO 2007] indicated that a significant share of C-PentaBDE (~32%) in Switzerland was used in EEE. The relatively high share is justified by the assumption that due to import from Asia the consumption in this area had not been declining as strongly as in other areas. Also Watson et al. (2010) highlighted the use of C-PentaBDE in epoxy resins from printed circuit boards. Leisewitz and Schwarz (2000) reported that PBDEs were used in FR2 laminates which were in turn assembled in household appliances (e.g. televisions, radio, video), vehicle electronics and in white goods (e.g. washing machines, kitchen appliances). According to a Danish EPA study [Danish EPA 1999] the FR2 laminates produced in Europe were treated with TBBPA, those in Asia with C-PentaBDE in the early 1990s. The C-PentaBDE in Asian production was first replaced by DecaBDE and then by TBBPA. The analyses of dismantled printed circuit boards in WEEE performed by Leisewitz and Schwarz (2000) showed that around 25% of FR2 laminates in older applications in Switzerland (produced in 1990) contain C-PentaBDE. Newer products (manufactured in 1998) were assumed to be C-PentaBDE free. More up to date studies have even reported that the use of C-PentaBDE in circuit boards is likely to have continued until recent times in Asia and possibly such circuit boards have been exported to Europe and the US [Watson et al. 2010]. Hester and Harrison (2009) indicated that Pentabromodiphenylether is still used in small percentage of Far East produced FR2 printed circuit laminates. However, these assumptions could not be confirmed.

According to [Swiss 2003] the use of C-PentaBDE in printed circuit boards imported to Switzerland from Asia ended between 1990 and 1998. It is therefore assumed that the import of C-PentaBDE containing EEE into the EU ended around 1994/95. With the average lifetime of household appliances and white goods of 9 and 15 years respectively it can be expected that most of the relevant products containing C-PentaBDE were already treated around 2003 (household appliances) and 2010 (white goods).

All in all, this study will exclusively focus on C-PentaBDE treated PUR foams and associated products. Nevertheless, it cannot be excluded that the imported amounts of C-PentaBDE in printed circuit boards could possibly be underestimated.

6.1.8 Substance Flow of C-PentaBDE

Based on the available data the project team has developed the following substance flow diagram for C-PentaBDE in the EU. In case relevant data was missing it was necessary to make best possible assumptions (i.e. imports and exports of products containing C-PentaBDE). The C-PentaBDE streams which are not in the scope of this study have been highlighted with an X in Figure 6-6.

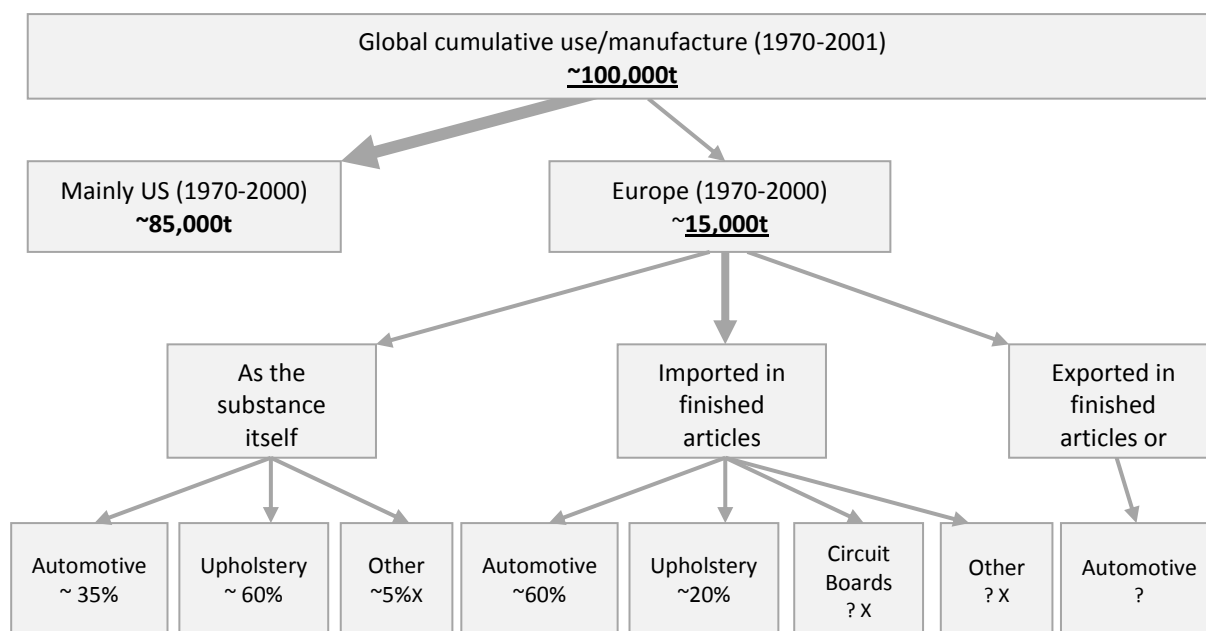


Figure 6-6: Overview of the substance flow contributing to the sources of C-PentaBDE in the EU

It has to be stated that the information provided within literature is limited and differs significantly in order to allow exact distribution estimations. Most of the available distribution data is based on production/consumption of C-PentaBDE in the UK or Switzerland and is therefore difficult to transpose to the EU. Therefore, Figure 6-6 is based on several assumptions, especially in relation to imports and exports of C-PentaBDE treated products. These assumptions should not influence further estimations to a large extent.

In relation to the distribution data for the use of the substance itself the results presented by Watson et al. 2010 are regarded to be most reasonable, especially when considering the fact that around 95% of C-

PentaBDE in Europe was used in flexible PUR foams. This study estimates the distribution to be around 60% in flexible PUR foam in furniture/upholstery, approximately 36% in automotive applications and 4% in all other articles. In addition this study can be seen as the most comprehensive one, assessing several similar studies conducted in this field.

Due to the lack of data, it is assumed that the vast majority of C-PentaBDE was imported in automotive applications (>60%). It is further assumed that the ratio between the imported automotive and upholstery applications is around 3:1. However, no exact distribution data is available. In relation to the exports of C-PentaBDE treated products it has been decided to exclusively focus on PUR foam in automotive applications. For the upholstery applications a conservative estimation is proposed assuming that no exports from Europe occurred at all. The same assumption was also made by Prevedouros et al. (2004a).

Estimations of the current C-PentaBDE stock by Watson et al. (2010) indicate that the peak of C-PentaBDE containing articles being delivered to waste in the EU was during the period 2005-2007. This is mainly due to the comparatively small amounts used in the EU, the early phase out dates and relatively short product life times of articles contaminated with C-PentaBDE. The authors [Watson et al. 2010] therefore conclude that only low levels of C-PentaBDE should currently be present in waste streams being delivered for disposal in the EU. In this regard they particularly highlight older cars and furniture which are kept in use for longer than their average life time expectancy. Nevertheless, they conclude that in case of Europe 80-95% of the C-PentaBDE stocks have already been landfilled or incinerated, leaving the accumulated amount of 750-3,000t that were mainly exported to US for recycling (i.e. carpet padding) together with minor quantities left in long life uses (e.g. rigid foam insulation). According to Prevedouros et al. (2004a), the major stocks of C-PentaBDE are estimated to be in PUR foams with up to 30% in cars, more than 10% in furniture foams and the rest in textiles, building material, packaging and solid applications.

Based on our estimations approximately 92% of the total accumulated C-PentaBDE in finished articles has already been subject to specific waste treatment operations and around 8% of the substance (~737t accumulated in automotive applications plus ~321t accumulated in upholstery applications) remains for treatment in the upcoming years.

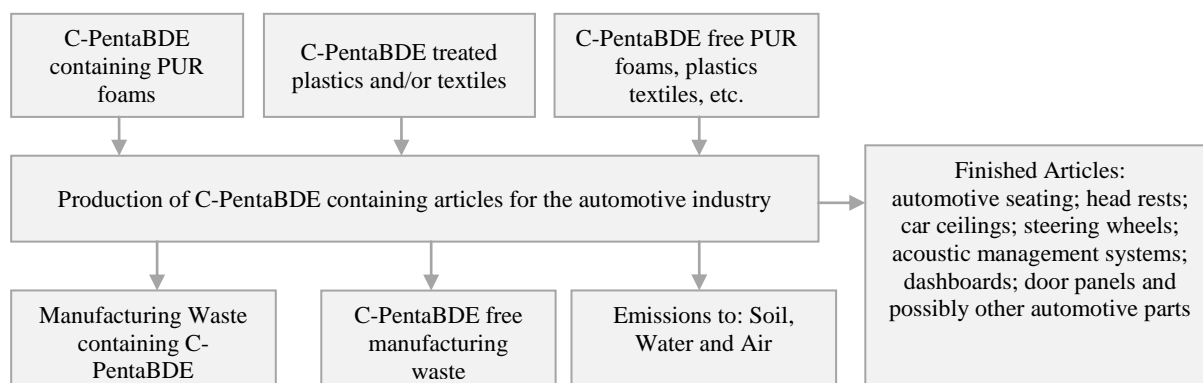
6.1.9 Automotive Applications

A) Background

PUR foam was used in various articles within the automotive sector, including seating, head rests, car ceilings, steering wheels, acoustic management systems, dashboards and door panels (see Figure 6-7). By the end of the 1990's around 100 companies manufactured foam and moulded PUR specifically for the automotive market. At that time, C-PentaBDE was still used in the UK in applications such as car ceilings and headrests and this use was associated with a production of 350t/y of foams for automotive purposes in the UK. For other EU Member States it was reported that the use of C-PentaBDE ceased some time ago (e.g. Italy, Spain, and Germany). This was mainly due to the pressure from car manufacturers. Several manufacturers placed C-PentaBDE on a company's list of 'substances of concern' (e.g. Volvo). The German Automotive Federation (VDA) black-listed C-PentaBDE, so the German car manufacturers were forced to replace it with other substance. In the case of car seating, other flame retardants such as C-DecaBDE, replaced C-PentaBDE [RPA 2000; Prevedouros 2004a]. The Risk Reduction Strategy [Risk 2000] concluded in 2000 that the use of C-PentaBDE in automotive applications already significantly decreased and is expected to completely disappear soon. The European Automobile producers association (ACEA) informed that currently no brominated flame retardants are used for the manufacturing of cars except DecaBDE. According to an internal substance information system of ACEA, C-PentaBDE has not been used since 2000. For the use prior to the year 2000 there is no information available. In addition ACEA has asked suppliers of automobile manufacturers. The suppliers have also no specific information for the time before 2000, however, they assume that the use has already stopped before the year 2000.¹⁴ The information provided by ACEA around the year 2000 describes the European situation. It cannot be excluded that imported cars e.g. from Asian countries contained C-PentaBDE containing PUR foams also after the year 2000, however, specific data are not available.

B) Process input (raw material) and output (waste, air, water, product)

Figure 6-7 indicates the main inputs and outputs associated with the production of C-PentaBDE containing products for the automotive sector.



¹⁴ Personal communication from ACEA, 14.12.2010 and e-mail from ACEA, 20.12.2010

Figure 6-7: Relevant process inputs/outputs and products (automotive industry)

Input:

- Major input of concern is the input of C-PentaBDE treated PUR foam to the production process of C-PentaBDE containing articles for the automotive industry.
- Possibly, C-PentaBDE treated plastic components and/or textiles were also used by some manufacturers. However, as already indicated the main focus of this study is on C-PentaBDE containing PUR foams.
- C-PentaBDE free PUR foams, plastic components, textiles, etc.

Outputs:

- Finished articles: C-PentaBDE was used in various articles within the automotive sector as illustrated in Figure 6-7. Among these articles, especially the car ceilings and headrests can be highlighted as indicated previously.
- Manufacturing Waste (containing C-PentaBDE; C-PentaBDE free): Different wastes were generated during the manufacture of C-PentaBDE containing products. Possibly some PUR foam manufacturing waste was exported from Europe to the US for recycling (i.e. carpet padding).
- Emissions to Soil, Water and Air: Due to the ban, releases from production, handling and processing already ceased.

C) Generation factors for product/waste and other output (air, water)

The generation factor can be assumed as 1, because input and output are considered equal.

D) Contamination data for output (and input)

In general, concentrations of C-PentaBDE in PUR foam products can range from 2 to 18% by weight. The contamination figure of 4% indicated by producers of automotive applications seems to be most realistic and will therefore be applied for further estimations. Consequently, an average car could contain around 250g of C-PentaBDE. However, this figure is rather overestimated, assuming that all PUR foam (~6-7 kg in average car) is treated with C-PentaBDE. For more information regarding contamination values see Table 6-5 and the corresponding explanations.

E) Activity data

In order to estimate the current amount of C-PentaBDE present in automotive applications in the EU (see Table 6-7) it was first of all necessary to derive the export share data of 'new' and 'old' cars from the EU (see Table 6-6). Relevant data on car production and exports has been obtained from the EUROSTAT database [EUROSTAT 2010].

Table 6-6: Share of exported new cars and ELVs [EUROSTAT 2010]

Year	Cars produced in EU	New cars exported from the EU	Old cars exported from the EU	Share (export new cars) [%]	Share (export ELVs) [%]
1999	17,170,429	1,869,687	243,842	10.9	1.4
2000	17,170,429	2,275,356	377,687	13.3	2.2
2001	17,170,429	2,323,473	568,202	13.5	3.3
2002	17,170,429	2,695,094	653,507	15.7	3.8
2003	18,872,764	2,727,853	539,321	14.5	2.9
2004	17,342,130	3,034,409	675,924	17.5	3.9
2005	17,650,184	3,218,498	680,622	18.2	3.9
2006	17,530,617	3,458,756	833,174	19.7	4.8
2007	18,533,681	3,777,105	932,968	20.4	5.0
2008	13,093,197	3,907,316	828,872	29.8	6.3

Table 6-7: Estimated amount of C-PentaBDE in automotive applications in the EU

Year	Total C-PentaBDE consumption (use+import) [t]	Yearly consumption of C-PentaBDE in automotive applications [t]	C-PentaBDE exported in new cars [t]	C-PentaBDE exported in ELVs [t]	Amount of C-PentaBDE in automotive applications [t]
1978	69	37	2,5	0	0
1979	138	75	5,0	0	0
1980	206	112	7,5	0	0
1981	275	149	10,0	0	0
1982	344	186	12,5	0	0
1983	413	224	15,0	0	0
1984	481	261	17,5	0	0
1985	550	298	20,0	0	0
1986	619	336	22,5	0	0
1987	688	373	25,0	0	0
1988	756	410	27,4	0	0
1989	825	448	29,9	0	0
1990	894	485	32,4	1,4	33,4
1991	963	522	35,1	2,8	66,8
1992	1031	559	37,9	4,2	100,2
1993	1100	597	39,4	5,6	133,6
1994	1100	597	39,4	7,0	167,0
1995	970	521	37,2	8,4	200,4
1996	839	446	35,8	9,8	233,8
1997	709	370	33,7	11,2	267,2
1998	578	295	32,5	12,6	300,5
1999	448	219	22,9	14,0	334,0
2000	238	116	15,0	15,4	367,4
2001	244	110	20,3	16,8	400,9
2002	163	73	15,7	18,2	434,2
2003	81	37	7,3	19,6	467,4
2004	0	0	0,0	21,0	500,5
2005	0	0	0,0	22,4	534,9
2006	0	0	0,0	22,4	534,9
2007	0	0	0,0	19,5	464,5
2008	0	0	0	16,7	393,3
2009	0	0	0	13,9	322,8
2010	0	0	0	4,1	258,3
2011	0	0	0	4,8	191,8
2012	0	0	0	3,8	97,4
2013	0	0	0	4,2	85,7

Year	Total C-PentaBDE consumption (use+import) [t]	Yearly consumption of C-PentaBDE in automotive applications [t]	C-PentaBDE exported in new cars [t]	C-PentaBDE exported in ELVs [t]	Amount of C-PentaBDE in automotive applications [t]
2014	0	0	0	2,1	55,6
2015	0	0	0	1,4	28,0
2016	0	0	0	0,0	0,0

With the estimated yearly consumption of C-PentaBDE in automotive applications, the export data (Table 6-6) and the average life time of cars (~12 years) the substance flow of C-PentaBDE in automotive products can be developed.

Table 6-7 shows the estimated amount of C-PentaBDE currently present in automotive applications in the EU (~260t). Besides, it shows that probably the majority of C-PentaBDE in automotive applications will be disposed of by 2015 (considering an average lifetime of 12 years and a complete use phase out in 2004).

According to information from a car recycler, the average age of ELVs in the Netherlands is 16 years. Assuming a corresponding lifetime and a phase out in 2004, the majority of C-PentaBDE in automotive applications would be disposed of around 2020.

The average life time of end-of life vehicles in Germany is about 15 years¹⁵. For the year 2009, statistical data are available of the year of the first registration of cars, which were scrapped within the German scrapping scheme.

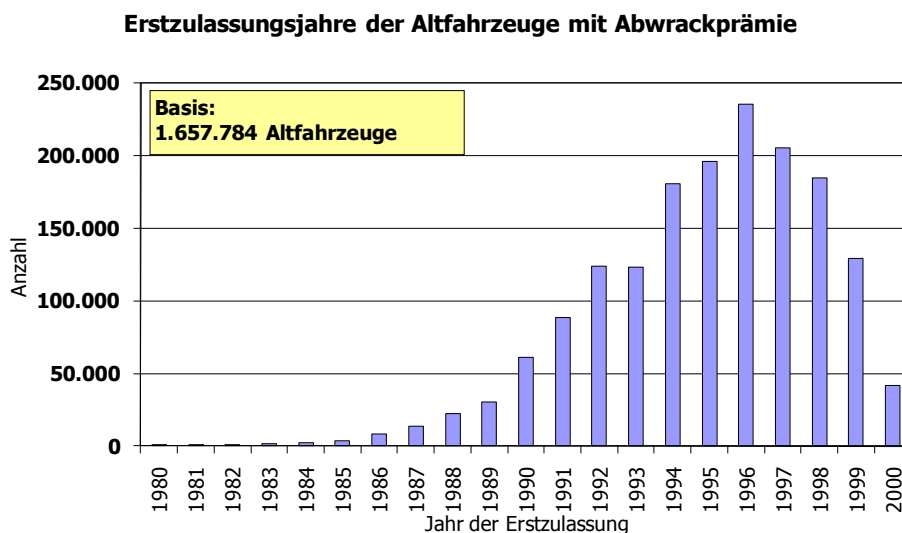


Figure 6-8: Number of cars and year of the first registration of cars, which were scrapped within the German scrapping scheme in 2009

¹⁵ See German report on ELV recovery rates 2008, page 18:

http://www.bmu.de/files/pdfs/allgemein/application/pdf/germany_elv_quota_qualityreport_en_bf.pdf

From the statistical data shown in Figure 6-8 it can be derived for Germany, that about 70% of ELVs are 15 years and younger, that the share of cars which are 16 years and older is approximately 30%, that approximately 5% are older than 20 years and that only an insignificant share of ELVs is older than 25 years (~0.6%). Therefore in 2016, about 30% of ELVs can be expected to be built in 2000 and before, in 2020 the corresponding number is about 5% and in 2023 it will be below 1%.

According to information from the European Automobile Manufacturers Association (ACEA), EU car manufacturers and suppliers did not use C-PentaBDE after 2000. For the time before 2000 there is no information available. However, ACEA assumes that the phase out was already before 2000. Assuming a lifetime of 16 years and a phase out in 2000, the majority of C-PentaBDE in automotive applications would be already disposed of around 2016.

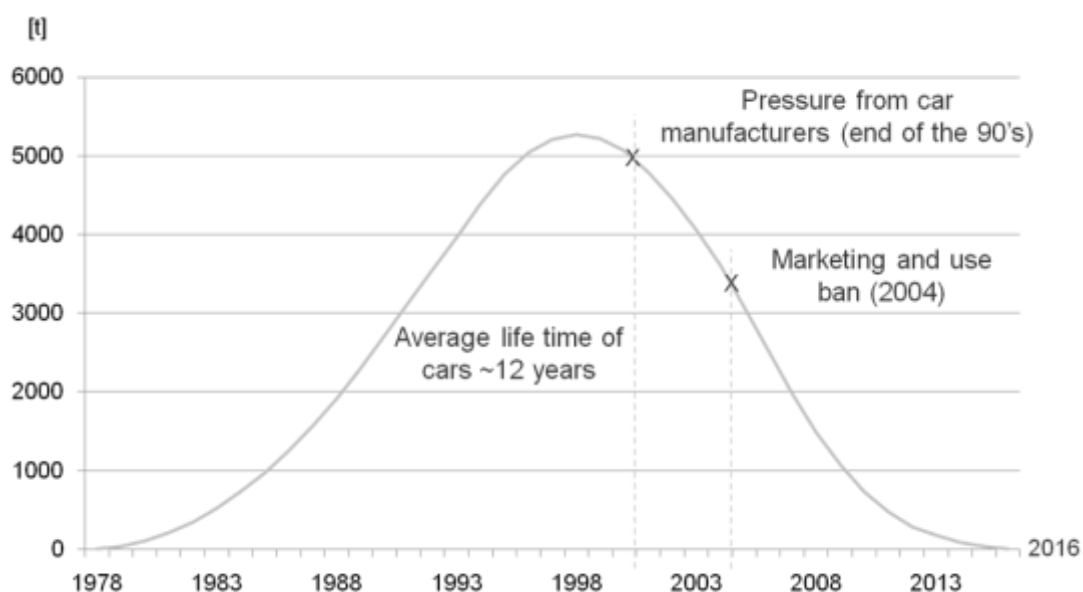


Figure 6-9: Estimated accumulated amount of C-PentaBDE (automotive applications)

F) Waste treatment

Figure 6-9 provides a general overview of how ELVs arise and are treated within the EU. This process might significantly vary among Member States.

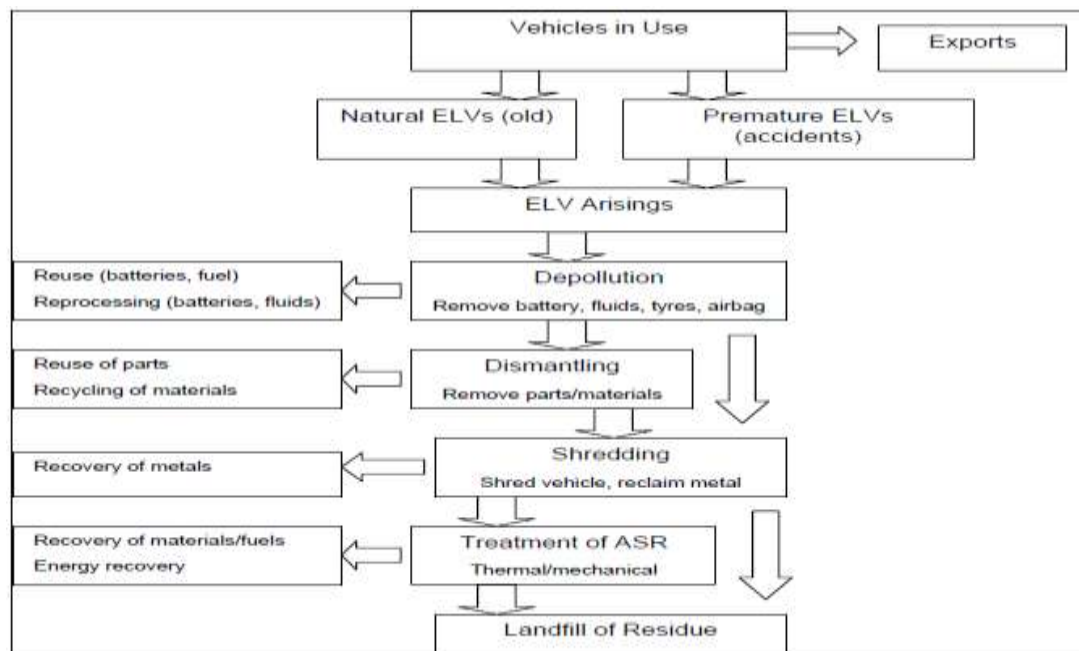


Figure 6-10: ELV arising and treatment in the EU [GHK 2006]

The vehicles that are not exported need to be treated at authorized treatment facilities (ATFs). As shown in

Figure 6-10 the ELVs undergo a process of depollution, involving the removal of fuel, oil and other liquids, as well as the battery, airbags and heavy metals. In the next step the ELVs are either dismantled in order to remove valuable parts and materials, or directly sent to the shredder. In case the valuable parts and materials are removed, these can be reused or recycled. Shredding results in the recovery of metals from ELVs and so called Auto Shredder Residue (ASR). The ASR is typically a combination of materials such as plastics, textiles and glass which has traditionally been landfilled, but is increasingly being treated in order to separate useable fractions and so increase rates of recycling [GHK 2006].

In up to date ELV the ASR is further separated in several fractions. One of these fractions contains textiles, fibers, foams, wood, foil and can be used as secondary fuel (see e.g. Scholz 2009; Process Flow Chart for the processing of Shredder-Light-Density-Fraction, SLF).

The composition of various Shredder-Light-Fractions (SLF) depends on the post shredder techniques which are applied [ARN 2011]. According to [Bartl 2010] the composition of SLF is not constant. One of the reasons is that shredder companies do not exclusively process end-of-life vehicles but also white goods, mixed scrap and other wastes. The portion of end-of-life vehicles therefore ranges between 27-85 %. Besides, [Bartl 2010] highlighted that also the composition of passenger cars has changed. Vehicles constructed in the period 1960 to 1975 contained about 78 % steel and 2.3 % fibres. During the period 1996 to 2000, however, the iron fraction was reduced to about 58 % while the fibre portion

increased to 7.3 %. Finally, it was concluded that SLF consists of the following fractions whose percentages can significantly vary [Bartl 2010]:

- plastics (thermoplastics, duroplastics, elastomers); predominantly in form of bulk but also fibers, films and foams
- other organic compounds (wood, paper, lacquer residue, etc.);
- metals (iron, copper, aluminium);
- minerals (glass, ceramics)
- others (sand, dust, rust, lead, zinc, etc.)

The Environmental Protection Agency (EPA) in Ireland has recently undertaken sampling of shredder residues to determine concentration of the new and candidate POPs in those wastes [EPA IRE 2011a]. The EPA has undertaken sampling and analysis of shredder residues from two facilities in Ireland and of a shred produced by a third facility that does not separate the output from their hammer mill into different streams. According to [EPA IRE 2011a] all Samples (42 in total) were sent to a laboratory in Germany for the determination of individual PBDE congeners including Tetra-, Penta-, Hexa- and HeptaBDE congeners according to ISO22032 (Extraction – GCMS). The following observations from EPA's experiences can be noted:

- The feedstock for these plants typically comprises a combination of materials including end of life vehicles (ELVs), light iron, construction material (e.g. rebar) and other scrap metal. There are also differences in the levels of material separation at each facility.
- The shredder residue is a heterogeneous waste and it is uncertain whether a representative sample for the purposes of PBDE determination could be obtained for regulatory purposes. The preliminary analysis of the 42 samples taken indicated a wide variation in the levels of PBDEs within the waste.
- There is a significant time lag between sampling and the results of analysis being available, making it difficult to see how this analysis could be undertaken in a regulatory context. The EPA is not aware of any laboratory in Ireland, at this point in time, that are providing a commercial service to undertake this type of analysis.
- The majority of the levels of PBDEs from the preliminary analysis tend to be below the low POPs concentration limits proposed in the interim report; however there were some samples that have indicated concentration levels above possible low POP concentration limits, in particular for the Tetra and Penta congeners. It can be postulated that this may be due to the mixed feedstock being processed by these facilities. This however is a tentative observation and will need to be confirmed by the analysis of the second laboratory.

Besides, a total of 10 duplicate samples from the original 42 samples were sent to a UK laboratory for PBDE analysis. The analytical method used is a developed in house method using extraction, clean up and GCMC detection. The methodology is apparently comparable to the methodology highlighted in [Kim et al, 2006 as referred to in Table 7-3]. The following observations can be made:

- The results of the duplicate analysis indicate low levels of POP PBDEs with levels below the low POPs concentration limits proposed in this report

- There are significant differences in the concentration levels when comparing the original sample analysis and the duplicate analysis. This may be attributed to the very heterogeneous nature of the waste samples or the different analytical methods used.
- There was a significant time lag in receiving the results which was mainly due to difficulties encountered by the lab during sample analysis (i.e.: matrix interference). Again it is worth pointing out that the duplicate analysis was carried out in the UK as the Irish EPA is not aware of any laboratory in Ireland at this point in time that are providing a commercial service to undertake this type of analysis [EPA IRE 2011b].

According to the project results, the above indicated concentration levels which were above possible low POP concentration limits, in particular for the Tetra- and PentaBDE congeners, are not to be expected if the contaminated input (i.e. PUR foam from cars built in 2000 and before and eventually ABS plastics from cathode ray computer and TV monitors produced in 2003 and before) is separated prior to the shredding process. The Irish EPA notes that plastics from cathode ray computer and TV monitors are typically not sent to the above mentioned waste streams and do therefore not contribute a relevant input to the above mentioned shredder facilities in Ireland. In other EU Member States the situation is different and shredder fractions may contain relevant PBDE isomers originating from both, i.e. from ELV treatment as well as from WEEE treatment.

Data on the PBDE content of the Fluff — light fraction containing dangerous substances (waste code 19 10 03*) from a data base on waste analyses of the German federal state North Rhine-Westphalia ("ABANDA"¹⁶) gives the following picture on contamination levels of this waste (based on 23 samples): TetraBDE (BDE 47) average 1.8 ppm, maximum 18 ppm, PentaBDE (BDEs 99 and 100) average 2.1 ppm, maxima ~13 ppm and 3.1 ppm, HexaBDE (BDEs 153 and 154) average 0.8 ppm, maxima 6.2 ppm and 1.8 ppm and HeptaBDE(BDE 183) 1.6 ppm, maximum 16 ppm. As both, TetraBDE and HeptaBDE occur in the waste, it is estimated that the input material includes also waste types which contain C-PentaBDE (such as ELVs) as well as C-OctaBDE (such as WEEE). In practice shredder usually receive inputs from several waste types including ELVs and WEEE. The measured average concentrations are all well below the preferred proposed LPCLs.

In specific cases the PUR foam is manually separated prior to the car shredding process and subsequently recycled. According to a car recycler from the NL (ARN), the PUR foam is purified by removing metals, fabric, and plastic parts and then shredded. The shredded foam is mixed with other types of fabrics to be reprocessed into synthetic fleece and lagging for use in new cars. The PU-foam flakes can also be mixed with glue to be pressed into large blocks that can be cut into shape for a variety of applications, including car seat padding, mattresses, sports mats, cow mats and furniture (source: http://www.arn.nl/noezp/video_en/start.html). According to ARN, however, the option to manually dismantle PU foam from cars has a negative residual value and will not be undertaken under market conditions. In the Netherlands it is only undertaken because it is part of the national fund system scheme of 19 materials to be dismantled manually. The Netherlands will be switching over to a post shredder treatment plant in 2011 so that manual dismantling will stop in the Netherlands as of January

¹⁶ <http://www.lanuv.nrw.de/abfall/bewertung/abanda.htm>

2011. Herewith manual dismantling of PU foam from cars would have perished from the EU member states (source: ARN).

The German EPA recommends to remove PBDE containing parts (e.g. seats) also before shredding in order to dispose them of/to decontaminate them separately and to define exemptions for the use as spare parts. According to investigations of the project team the use of car seats (as the only relevant spare part which is expected to contain C-PentaBDE treated PUR foams) is of minor practical relevance.

G) C-PentaBDE flow

According to ACEA, at European level, recycling of PUR foam from cars is not representative. The PUR foam is usually not separated prior to the shredding process. In general, the PUR foam becomes part of the shredder light fraction and is usually subject to R1 (incineration with energy recovery) or D10 (incineration without energy recovery), in Member States where the corresponding capacities are available. In order to achieve the 85% reuse and recycling target of the ELV Directive in 2015 it is intended to increase the material recycling of post shredder fractions such as the shredder light fraction. Corresponding developments and investments are currently under way.¹⁷

According to the recent ESTAT data on materials from shredding of EOL vehicles¹⁸ the shredder light fraction is recycled (11.9%) used for energy recovery (7.0%) and disposed (81%) in EU 27 with significant differences between the single Member States. Disposal comprises the two waste management options “landfilling” and “incineration without energy recovery” (D10). Specific information on the share of these two options is not available. It has therefore been assumed that 40.5% of the shredder light fraction is landfilled and 40.5% is incinerated without energy recovery.

The emissions have been estimated in accordance with the methodology presented in the EU Risk Assessment Report [Risk 2000]. Therefore, 3.9% has been applied for volatile emissions and additional 2% for particulate losses as proposed in the Risk Assessment study. This leads to a total emission of around 15t/y as shown in Figure 6-11 below.

The resulting 243.3 t of C-PentaBDE present in automotive waste is landfilled (~98.5t), incinerated without energy recovery (~98.5t), incinerated with energy recovery (~17.0 t) and recycled (~29.0 t).

¹⁷ Personal communication ACEA, 14.12.2010 and e-mail ACEA, 20.12.2010

¹⁸ Arising in the country and treated within the country; published on 4.10.2010 for the year 2008

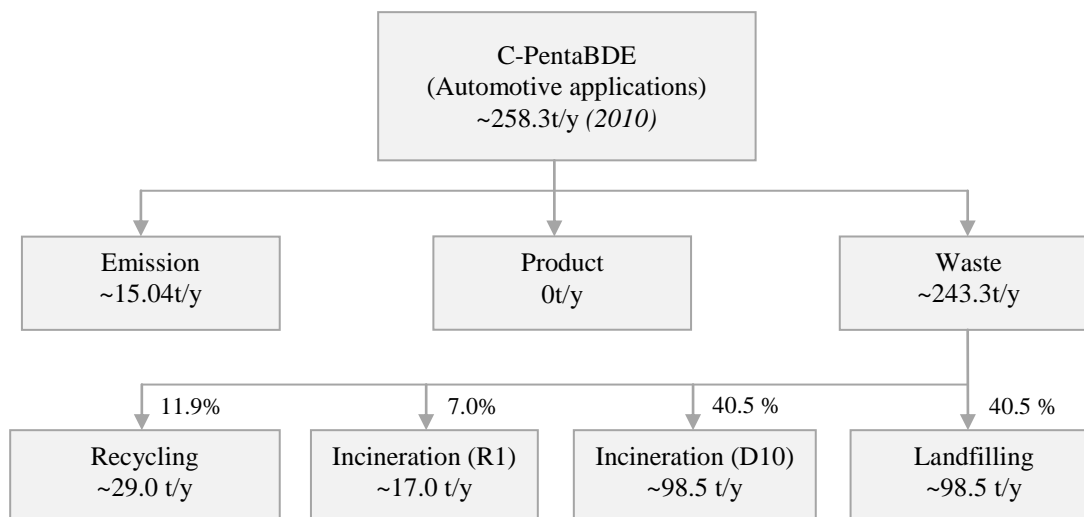


Figure 6-11: C-PentaBDE flow in automotive applications

H) Waste flow for C-PentaBDE

As shown in Figure 6-12 approximately 6,082 t of PUR foam based automotive applications containing C-PentaBDE will have to be treated in the EU in 2010. Figure 6-12 is based on certain assumptions which will be introduced in the following.

The PUR foam containing C-PentaBDE (~6,082 t) with a C-PentaBDE concentration of approximately 4 % is sent to the shredder and is treated together with the SLF. The total SLF (~655,873 t for 2008) has been derived from the data provided in the EUROSTAT database for the EU27 (i.e. Materials from shredding of end-of-life vehicles arising in the country and treated within the country). Besides, the analysis of the EUROSTAT data shows that approximately 15 % of the total end-of-life vehicle's weight can be assigned to SLF. [Bartl 2010] presents higher figures of around 20-25 %. According to ARN, the SLF amounts to about 20% of the ELV weight.

With the above derived EUROSTAT figure for total SLF (~655,873 t), the resulting mixed waste stream will contain around 371 mg/kg C-PentaBDE. This waste will be finally landfilled or incinerated without energy (~40.5 % each), incinerated with energy recovery (7 %) and recycled (~11.9 %), as shown in Figure 6-12 below.

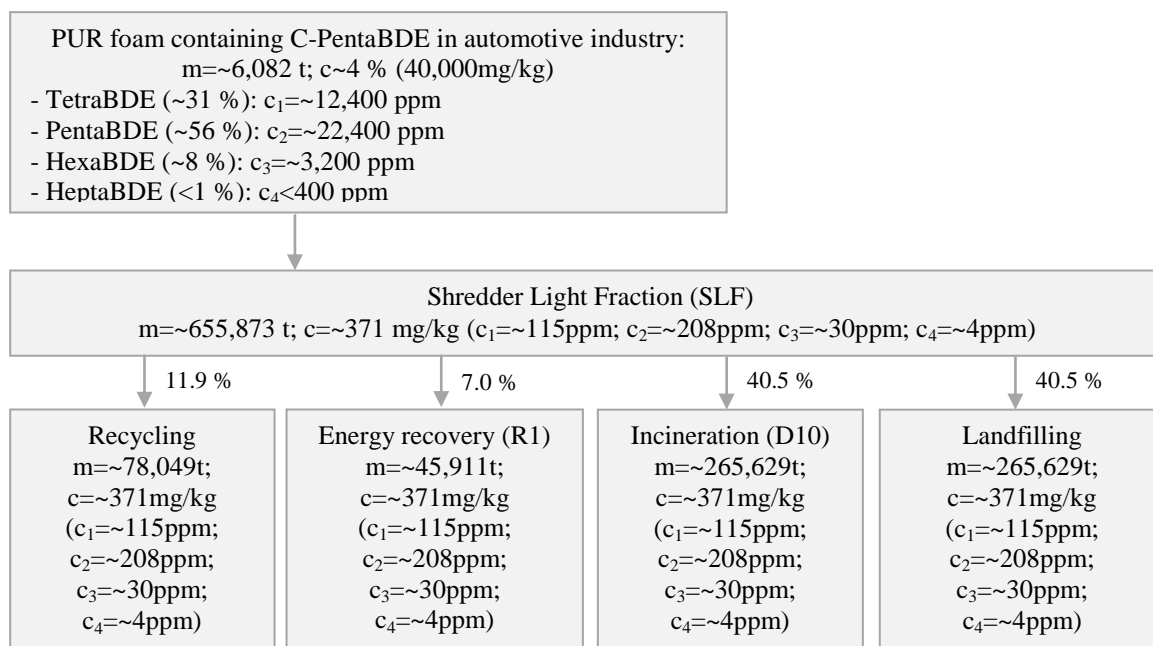


Figure 6-12: Waste flow for C-PentaBDE automotive applications

It is, however, important to notice that the situation displayed in Figure 6-12 might change in the near future (incl. changes of the mixed contamination figure of the waste to be treated).

As mentioned, Directive 2000/53/EC on end-of-life vehicles demands specific quotas for recycling and recovery. Since in 2015 the directive will be further tightened significant improvements to raise the recycling rate will be required. By applying mechanical processes, the SLF can be separated into plastic, fines and fluff. For the plastics several possibilities for recycling already exist. The fines, also called sand or inert fraction, mainly consist of minerals can be landfilled. There is, however, a potential for the improving the recycling rate of the fluff fraction [Bartl 2010].

[Bartl 2010] explains that, even if post shredder technologies are already far developed (i.e. mechanical processes), it seems that there is still a considerable room for improvement. Partially the post shredder technologies produce output streams which are appropriate for further recycling and, thus, commonly an economically marketing is possible. However, there are still a number of fractions that have to be landfilled or otherwise recovered. In order to increase the overall benefit of post shredder technologies it is necessary to change to properties of these fractions and to find attractive possibilities for recycling.

Therefore, if it is assumed that in the near future more focus will be put on the development of post shredder treatment technologies, which will allow the separation of the fibre fraction from the SLF (SLF = 20% of the ELV weight and the fibre fraction of the post shredder treated SLF = 5% of the ELV weight. This fibre fraction contains among other the PU foams.), the C-PentaBDE contamination of the final waste to be treated would significantly increase (approximately by a factor of 4). On the other hand the input of C-PentaBDE containing PU foam in the shredder process will significantly decrease in the coming years due to the phase out of the corresponding use prior or around the year 2000. These aspects should be taken into account when setting the limit values.

6.1.10 Upholstery Applications

A) Background

Flexible PUR foam is widely used as an important component of upholstered furniture and mattresses. This application was one of the two largest uses for C-PentaBDE in the EU. In terms of variations in use among MS, it can be assumed that the most C-PentaBDE was used in the UK, considering the stringent fire legislation for private furniture. UK's PUR foam manufacturers indicated a figure of at least 5t/y of C-PentaBDE for the production of 50t/y of PUR foam [Risk 2000].

B) Process input (raw material) and output (waste, air, water, product)

Figure 6-13 shows the main inputs and outputs associated with the production of C-PentaBDE containing upholstery products.

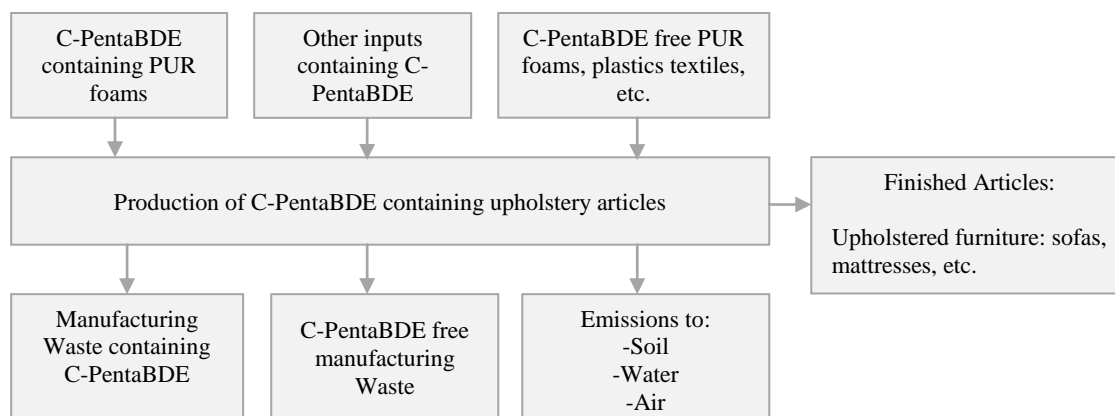


Figure 6-13: Relevant process inputs/outputs and products (upholstery)

Inputs:

- Major input of concern is the input of C-PentaBDE treated PUR foam to the production process of C-PentaBDE containing upholstery articles.
- Possibly, C-PentaBDE treated plastic components and/or textiles were also used by some manufacturers. However, as already indicated the main focus of this study is on C-PentaBDE containing PUR foams.
- C-PentaBDE free PUR foams, plastic components, textiles, etc.

Outputs:

- Finished articles: C-PentaBDE was mainly used in upholstered furniture (e.g. sofas) and mattresses.
- Manufacturing Waste (containing C-PentaBDE; C-PentaBDE free): Different wastes were generated during the manufacture of C-PentaBDE containing products. Possibly some PUR foam manufacturing waste was exported from Europe to the US for recycling (i.e. carpet padding).
- Emissions to Soil, Water and Air. Due to the ban, releases from production, handling and processing already ceased.

C) Generation factors for product/waste and other output (air, water)

The generation factor can be assumed as 1, because input and output are considered equal.

D) Contamination data for output (and input)

Concentrations of C-PentaBDE in PUR foam for upholstery products range from 2 to 5.45 % by weight. Based on information collected in Table 6-5 an average contamination figure for upholstered furniture and mattresses of 3.8 % will be applied for further estimations. For more information regarding contamination values see Table 6-5 and the related explanations.

E) Activity data

The lifetime of upholstery has been considered to be 10 years [GEO 2007]. With the yearly consumption and the average lifetime for upholstery the accumulated amount of C-PentaBDE in upholstery applications can be estimated (see Table 6-8 and the corresponding Figure 6-14).

Table 6-8: Estimated accumulated consumption of C-PentaBDE in upholstery products

Year	Used [t]	Imported [t]	Total consumption [t]	Yearly consumption of C-PentaBDE in upholstery products [t]	Accumulated amount of C-PentaBDE in upholstery applications [t]
1978	19	50	69	21	0
1979	38	100	138	43	21
1980	56	150	206	64	64
1981	75	200	275	86	129
1982	94	250	344	107	215
1983	113	300	413	129	322
1984	131	350	481	150	451
1985	150	400	550	172	602
1986	169	450	619	193	774
1987	188	500	688	215	967
1988	206	550	756	236	1182
1989	225	600	825	258	1397
1990	244	650	894	279	1612
1991	263	700	963	301	1827
1992	281	750	1031	322	2042
1993	300	800	1100	344	2257
1994	300	800	1100	344	2472
1995	282	688	970	331	2665
1996	264	575	839	294	2824
1997	246	463	709	257	2925
1998	228	350	578	219	2967
1999	210	238	448	182	2949
2000	112	125	238	97	2873
2001	150	94	244	112	2691
2002	100	63	163	75	2502
2003	50	31	81	37	2254
2004	0	0	0	0	1948
2005	0	0	0	0	1604
2006	0	0	0	0	1273
2007	0	0	0	0	979
2008	0	0	0	0	722
2009	0	0	0	0	503
2010	0	0	0	0	321

Year	Used [t]	Imported [t]	Total consumption [t]	Yearly consumption of C-PentaBDE in upholstery products [t]	Accumulated amount of C-PentaBDE in upholstery applications [t]
2011	0	0	0	0	224
2012	0	0	0	0	112
2013	0	0	0	0	37
2014	0	0	0	0	0

In contrast to the automotive applications, no data related to import or exports of upholstery articles has been considered. For simplification reasons it is assumed that no imports or exports of upholstery furniture and mattresses occurred at all.

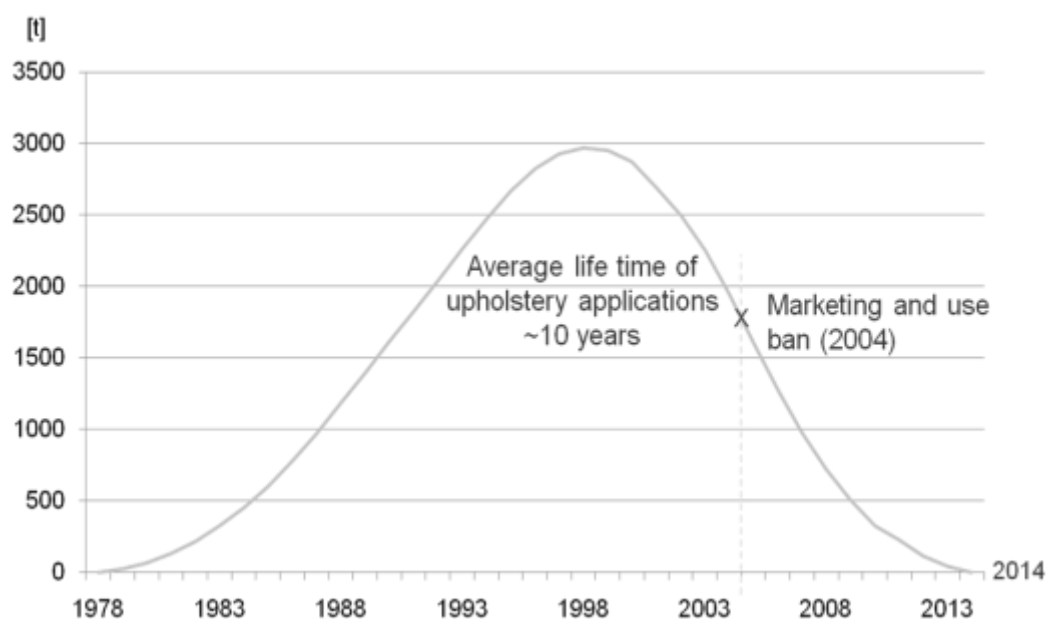


Figure 6-14: Estimated accumulated amount of C-PentaBDE in upholstery applications

F) Waste treatment

According to a study from [EBIA/ISOPA/EUROPUR 2008] discarded upholstery furniture and mattresses are part of the Municipal Solid Waste (MSW) stream and are collected either separately as bulky waste, classified as EWC 200307 – bulky waste or together with the MSW fraction as EWC 200301 – mixed municipal waste. According to the study the following treatment and recovery operations are currently used:

- Incineration
- Landfilling
- Gasification
- Rebonding of foam flocks
- Powder milling

Recycling is technically feasible, by either rebonding foam flocks using the material for new upholstery or by milling the material to PU-powder for reuse in the virgin foam industry. Also gasification of separated PUR foams mixed with organic plastic waste is a recovery possibility, using the waste in refinery processes or as reducing agents in blast furnaces.

However, the recycling processes deliver products with poor quality and limited application outlets or are yet not fully established in the market. Besides, using those processes would require the separate collection and the decontamination of the PUR foam. Building up a separate furniture/mattress collection infrastructure and the separation of PUR foam from other waste materials by the option of manual dismantling is cost intensive and not practiced at the moment. Thus, the recycling processes are currently only used marginally.

Therefore incineration with energy recovery is recommended as the preferred technology from an economically and environmentally view by [EBIA/ISOPA/EUROPUR 2008]. Also landfilling in controlled landfills is still a used treatment option in many EU countries lacking of incineration capacity, but is to be phased out in the coming years. Also Alcock et al. (2003) states that discarded furniture, foam and bedding will be practically disposed of to landfills or be incinerated.

EUROSTAT provides data for the treatment of household and similar wastes. Based on this data and on the assumption that recycling operations are applied only marginally for furniture and mattresses, it is assumed that 67.1 % of the used furniture is landfilled, 20.2 % is incinerated without and 12.7 % is incinerated with energy recovery (see Table 5-1).

G) C-PentaBDE flow

According to our estimations, around 91.31 t/y of C-PentaBDE in upholstery applications have to be treated in the EU in 2010. The C-PentaBDE has been landfilled (~61.3 t), incinerated without energy recovery (~18.4 t) and incinerated with energy recovery (~11.6 t). Besides, with the assumed life time of 10 years all C-PentaBDE will be disposed of by 2014 (see Table 6-8).

With respect to emissions of C-PentaBDE from upholstery applications the same assumptions are applied as in the case of automotive PUR foams (see Chapter 6.1.9, G). This leads to a total emission estimation of around 5.6 t/y as shown in Figure 6-15.

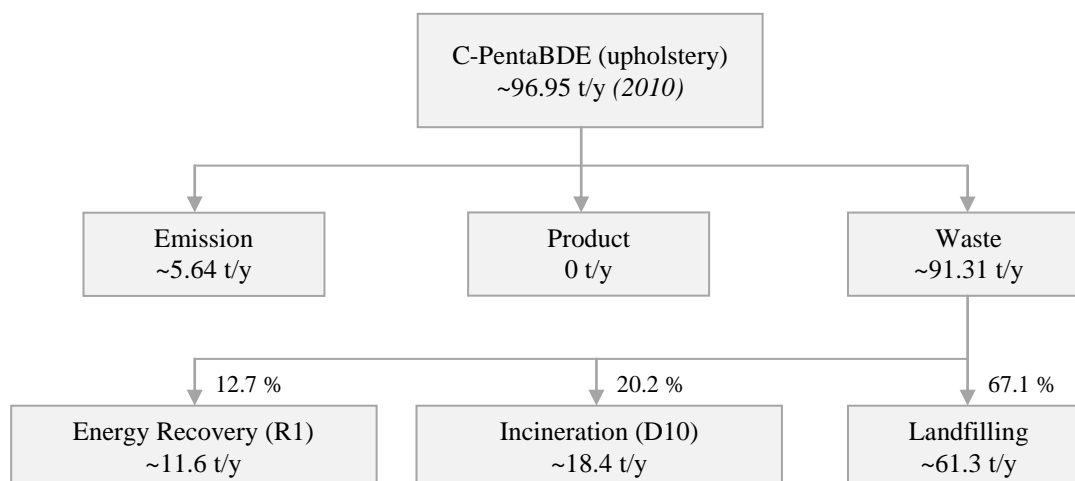


Figure 6-15: C-PentaBDE flow in upholstery applications

H) Waste flow for C-PentaBDE

As shown in Figure 6-16 approximately 2,403 t of PUR foam based upholstery applications containing C-PentaBDE will have to be treated. This figure is based on certain assumptions which will be introduced in the following.

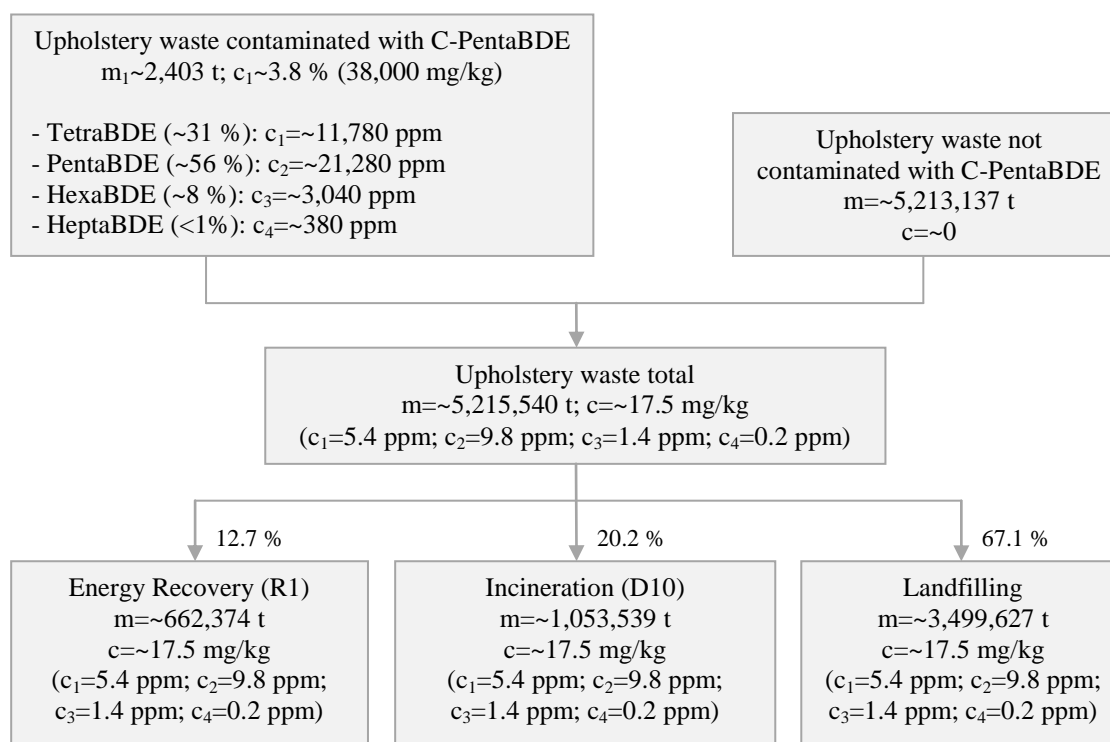


Figure 6-16: Waste flow for C-PentaBDE in upholstery applications

It is first of all assumed that the C-PentaBDE contaminated waste will be treated together with other non contaminated upholstery. From the total Municipal Solid Waste (MSW) generation figure in the EU27 (~260,777,000 t; EUROSTAT 2010) it has been assumed that a share of approximately 5 % can be assigned to bulky waste (~13,038,850 t). The 5 % share has been derived from the German waste generation figures for MSW and bulky waste [UBA 2010]. Furthermore, it has been assumed that in turn around 40 % of this bulky waste is upholstery and mattresses (~5,215,540t). The exact composition of bulky waste in the EU is, however, difficult to descry as it might considerably vary among Member States.

Consequently, due to the increased amount of the total upholstery waste to be treated, the contamination figures decline to approximately 17.5 mg/kg. This waste is finally landfilled (~67.1%), incinerated without energy recovery (~20.2%) and incinerated with energy recovery (~12.7%), as shown in Figure 6-16.

6.1.11 References C-PentaBDE

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6.2 C-OctaBDE flows

6.2.1 Use and Production

Polybrominated diphenyl ethers (PBDEs) are a group of additive brominated flame retardants, which have been used in various applications all over the world. They are flame retardants of the additive type, i.e. they are physically combined with the material being treated rather than chemically as in the case of reactive flame retardants [BiPRO 2007].

The International Uniform Chemical Information Database (IUCLID) reported two producers of Octabromodiphenyl ethers in the EU in 1994. However, both companies stopped production within the EU in 1996/98 [ECB 2003].

The annual world-wide production of all commercial polybrominated diphenyl ethers was estimated to be 40,000t/y in 1994, of which C-OctaBDE accounted for 6,000t/y (i.e. 15%). The production volumes decreased to about 3,800t/y in 2001. According to Watson et al. (2010), the flame retardant Tetrabromobisphenol A (TBBPA) and 1,2-bis (tribromophenoxy)-ethane (TBPE) were increasingly used as a replacement for C-OctaBDE as flame retardant for Acrylonitrilebutadiene-styrene (ABS) polymers through the 90's.

The production of C-OctaBDE was situated in the Netherlands, France, USA, Japan, UK and Israel. According to BSEF, C-OctaBDE is no longer produced in the EU, USA and the Pacific Rim and there is no information that it is produced in developing countries. Therefore, it can be concluded that nowadays it is essentially impossible to buy C-OctaBDE at the global level [BiPRO 2007].

6.2.2 Chemical characteristics of Commercial OctaBDE

The POP regulation focuses on Hexa- and Hepta-BDE in the sense of the definition of the Stockholm convention and therefore means 2,2',4,4',5,5'-hexabromodiphenyl ether (BDE-153, CAS No: 68631-49-2), 2,2',4,4',5,6'-hexabromodiphenyl ether (BDE-154, CAS No: 207122-15-4), 2,2',3,3',4,5',6 heptabromodiphenyl ether (BDE-175, CAS No: 446255-22-7), 2,2',3,4,4',5',6-heptabromodiphenyl ether (BDE-183, CAS No: 207122-16-5) and other hexa- and heptabromodiphenyl ethers present in commercial octabromodiphenyl ether (see SC Decision 4/14).

Commercial OctaBDE (C-OctaBDE) is not a pure substance, but is instead a complex mixture of several congener groups as shown in Table 6-9. The information in Table 6-9 was summarized by UNEP (2008) from various sources (e.g. WHO, OECD, Great Lakes Chemical Corporation, Chemische Fabrik Kalk, etc.).

Table 6-9: Composition of C-OctaBDE [Octa_UNEP 2008]

Main components	Up to 1994 [%]	1997 [%]	2000 [%]	2001 [%]	2006 [%] (USA)	2006 [%] (Germany)	Used for calculations [%]
PentaBDE	10.5-12.0		1.4-12.0	≤0.5			6.25
HexaBDE		5.5		≤12	10.5	0.3	6.15
HeptaBDE	43.7-44.5	42.3	43.0-58.0	≤45	45.5	12.8	35.40
OctaBDE	31.3-35.3	36.1	26.0-35.0	≤33	37.9	21.8	
NonaBDE	9.5-11.3	13.9	8.0-14.0	≤10	13.1	18.9	
DecaBDE	0-0.7	2.1	0-3.0	≤0.7	1.3	49.6	
Σ (Penta-, Hexa- and HeptaBDE)							47.80

It should be noticed that various synonyms and abbreviations are applied within literature. The inconsistency as well as the large variety of synonyms and abbreviations makes it often difficult to distinguish between the individual congeners, congener groups and commercial mixtures. Therefore, for the purpose of this report the abbreviation C-OctaBDE will be applied for the commercial OctaBDE mixture. Congener groups are groups of homologue congeners and are designated according to their degree of bromination e.g. as TetraBDE, PentaBDE, HexaBDE or HeptaBDE. Individual congeners will be specifically designated within the report, where appropriate.

6.2.3 Environmental Background Levels

The commercial product C-OctaBDE has spread widely in the global environment. Existing monitoring data on OctaBDE show that levels of the substance were detected in atmosphere, water, biota and humans as well as in waste effluents and biosolids, sediments and soil. A comprehensive overview of monitoring data in abiotic media and biota is given in [Octa_UNEP 2007].

Table 6-10 gives an overview of the concentration levels in sediment detected in different parts of the world.

Table 6-10: Levels of C-OctaBDE* and its congeners in sediment and soil (DW: dry weight) [Octa_UNEP 2007]

Substance	Country/Region	Levels of PentaBDE	References	Comments
Sediment				
c-OctaBDE	UK	<0.44-3030 µg/kg dw	[Allchin et al. 1999]; [Law et al. 1996]; [Environment Agency UK 1997]	The highest levels were detected in sediments downstream from a warehouse where c-DecaBDE was stored. Therefore, the levels are considered as hot spot concentrations (see table below).
c-OctaBDE	Japan	8 to 21 µg/kg (detection limit 7 µg/kg WW or dw not specified)	[Environment Agency Japan 1991]	The substance was detected in 3 of 51 sediment samples in 1987.
c-OctaBDE	Japan	15 to 22 µg/kg (detection limit 5 µg/kg ww or dw not specified)	[Environment Agency Japan 1991]	The substance was detected in 3 of 135 samples collected in 1988.
PBDEs (Total)	Lake Ontario, area	0.5 to 4.0 µg/kg	[Kolic et al. 2004]	14 tributary sites (only 6 were

Substance	Country/Region	Levels of PentaBDE	References	Comments
Hexa- and HeptaBDE i.e., BDE 138, 153, 154 and 183)	biosolids in Southern Ontario	dw		reported)
PBDEs	Lake of Ellasjøen, Norwegian Arctic	0.73 µg/kg dw	[Evenset et al. 2007]	Historical trends of PBDEs in sediments; contamination is due to both atmospheric and biological transport; maximum value was detected in 2001.
Soil				
Total PBDEs	Latitudinal transect through UK and Norway	0.065 to 12.0 µg/kg dw	[Hassanin et al. 2004]	Determination of the substance in undisturbed surface soils (0-5cm) and subsurface soils from remote/rural woodland and grassland sites.

*In [Octa_UNEP 2007] c-OctaBDE is defined as commercial product with several components in the product, with different properties and potential risks.

An increased trend towards higher concentrations of PBDEs over the period has been observed in Niagara River suspended sediments from 1988 to 2004 [Marvin et al. 2007]. Before 1988, PBDEs (sum of 16 congeners including DecaBDE) were detected at low-ppb concentrations which already showed a slight increasing concentration at low level from 1980 to 1988. However, after 1988, PBDE concentrations in the Niagara River showed a more rapidly increasing trend (maximum of approximately 35 ng/g in 1995). DecaBDE was the predominant congener detected. A similar situation has been reported for Europe by [Eljarrat et al. 2005], and Asia by [Moon et al. 2007]. For sediments at a background location, additional information on detected concentrations of c-OctaBDE components (HexaBDEs 153 and 154) is reported by [Law et al. 2006; Octa_UNEP 2007].

PBDEs have been further detected in different concentrations in soils in the European region as indicated in Table 6-10 [Hassanin et al. 2004] analysed within their study 66 surface soils for 22 Tri- to HeptaBDEs. The concentrations of the total PBDEs were ranging from 0.065 to 12.0 µg/kg DW while the median PBDE concentrations were between 0.61 to 2.5 µg/kg DW, with BDEs 47, 99, 100, 153 and 154 dominating the total concentrations. The median concentration of the sum of these congeners ranged from 0.44 to 1.8 µg/kg DW. According to the study, the congener's patterns in the European background soils relatively closely matched that observed for the C-PentaBDE mixture. An increasing contribution of BDE 47 and other lighter PBDEs – in comparison to the heavier PBDEs which were detected in the samples – was determined northward along the latitudinal transect [Hassanin et al. 2004; Octa_UNEP 2007].

In addition to the above listed concentration limits for soil, hot spot concentration levels of the substance and or its congeners have been detected in waste effluents and sewage sludges. An overview of the determined levels is provided in Table 6-11.

Table 6-11: Levels of C-OctaBDE* and its congeners in sludges and coastal sediments receiving industrial and municipal effluents (DW: dry weight) [Octa_UNEP 2007]

Substance	Country/Region	Compartment	Levels of OctaBDE	References	Comments
C-OctaBDE	UK	Sediment	<0.44-3,030 µg/kg dw	[Allchin et al. 1999];	The highest levels were detected in sediments

Substance	Country/ Region	Compartment	Levels of OctaBDE	References	Comments
				[Law et al. 1996]; [Environment Agency UK 1997]	downstream from a warehouse where c- DecaBDE was stored.
total Hexa- to OctaBDE	Canada, USA	Sewage sludge	40 to 2,080 µg/kg dw	[La Guardia et al. 2001]	11 samples taken; before land application from
PBDE (Hexa- to OctaBDE)	in southern Ontario	Sewage sludge	124 to 705 µg/kg dw	[Kolic et al. 2004]	Samples taken at 12 sites; Hexa- to OctaBDE congeners were not detected in manure samples; detected at very low levels in pulp mill biosolids (up to approximately 3 µg/kg dw)
Total Hexa- and HeptaBDEs (i.e., BDEs 138, 153, 154 and 183)	Tributaries flowing to Lake Ontario; nearby wastewater treatment facilities in southern Ontario	Biosolids (sewage sludge)	111 to 178 µg/kg dw	[Kolic et al. 2004]	Determination of the substance in undisturbed surface soils (0-5cm) and subsurface soils from remote/rural woodland and grassland sites.
sum of Hexa to NonaBDE	Spain	Sludge from municipal wastewater treatment facilities	15.5 to 160 µg/kg dw	[Martínez et al. 2006]	-
		Sludge from industrial facilities	up to 268 µg/kg dw		
PBDEs	Kuwait	Coastal sediments receiving industrial and municipal effluents	0.08 to 3.8 µg/kg dw	[Gevao et al. 2006]	HeptaBDE183 dominated the congener distribution resembling the commercial formulation, Bromkal 79- 8DE. Wastewater discharge from industrial activities appeared to be the primary source of the compounds.

*In [Octa_UNEP 2007] c-OctaBDE is defined as commercial product with several components in the product, with different properties and potential risks.

As shown in Table 6-11, the detected background contamination levels in sediment and soil ranged from 0.065 µg/kg in latitudinal transect through UK and Norway to 22 µg/kg in Japan. Hot spot levels of c-OctaBDE were detected inter alia in UK and Spain in sediment, soil and sewage sludges. The hot spot levels reached up to 3,030 µg/kg in sediment in UK. In the related study, the highest levels were detected in sediments downstream from a warehouse where c-DecaBDE was stored.

Limit values for OctaBDE in waste should not be below the observed maximum values in the environment. As the highest limit value in Japan was 22 µg/kg, it is assumed that 100 µg/kg DW generally will not be exceeded under conditions excluding specific contamination with chemicals. Taking into consideration the relatively large quantity of existing monitoring data, an uncertainty factor of 10 is applied. Therefore, a minimum concentration limit value should not be below 1,000 µg/kg DW or 1ppm DW.

6.2.4 Legal Situation

6.2.4.1 EU level

POPs related provisions

C-OctaBDE is listed in Annex I, IV and V to the proposal for amendment of the POPs Regulation in terms of individual provisions for tetraBDE, pentaBDE, hexaBDE, and heptaBDE congener groups.

Annex I sets out values for unintentional trace contaminants – identically in place for all concerned congener groups – in substances, preparations, articles or as constituents of the flame-retarded parts of articles equal to or below 0.001 % by weight (10 ppm).

Market restrictions are listed in Annex I, again identically for all congener groups. Production, placing on the market and use shall be allowed

- without prejudice to subparagraph (b), articles and preparations containing concentrations below 0.1% by weight (1,000 ppm) of all congener groups by weight when produced partially or fully from recycled materials or materials from waste prepared for re-use;
- electrical and electronic equipment within the scope of Directive 2002/95/EC.

In Annex V, no upper and lower POPs limits have been set for the PBDE congener groups.

Waste management – WSR

For the purposes of classifying plastic waste under Waste Shipment Regulation, it is common understanding that PBDE containing plastic waste is not to be classified as entry A 3180 (of “amber list”) and the respective threshold for PBBs, as brominated flame retardants PBDEs are not analogues of PBBs. Consequently, currently there is no threshold limit for wastes being listed “amber”, although it is reported by a Competent Authority that based on REACH and RoHS a threshold level of 0.1% by mass (1,000 ppm) for octaBDE is suggested.

Waste management – WEEE

WEEE Directive 2002/96/EC aims at preventing WEEE by promoting reuse, recycling and recovery and is of major interest for the project. For the ten WEEE categories defined in Annex IA, the Directive sets targets for recovery and for component, material and substance reuse and recycling. As regards new POPs (PBDEs) the targets set for categories 3 and 4 are of relevance, requesting:

- A recovery rate of a minimum of 75 % by an average weight per appliance, and
- A reuse and recycling rate for component, material and substance of a minimum of 65 % by average weight per appliance

Furthermore according to Annex II “Selective treatment for materials and components of waste electrical and electronic equipment in accordance with Article 6(1)” as a minimum [...] [e.g. plastic

containing brominated flame retardants] have to be removed from any separately collected WEEE. This means that plastics containing brominated flame retardants have to be removed from any separately collected WEEE. Thereby, MS shall ensure that manufacturers or third parties acting on their behalf set up WEEE treatment systems which may be done on the individual basis and/or collectively. Best available treatment, recovery and recycling techniques should be applied. Besides, MS may set up minimum quality standards for the treatment of collected WEEE.

Restrictions on marketing and use - REACH

Following Annex XVII, entry No 45 of REACH, “Diphenylether, octabromo derivative (C₁₂H₅Br₅O)” may not be placed on the EU market or used as a substance or a constituent of substances or of preparations in concentrations higher than 0.1 % (1,000 ppm) by weight. The same applies for finished articles, which may not be placed on the EU market if they, or flame-retardant parts thereof, contain octaBDE in concentrations higher than 0.1 % (1,000 ppm) by weight. This restriction is in force since August 2004.

Note: the OctaBDE congener group or “Diphenylether, octabromo derivative (C₁₂H₅Br₈O)” as listed under REACH is not listed in the POPs Regulation as the specific OctaBDE congener group does not meet the POPs criteria in its own. Therefore OctaBDE will continue to stay in REACH Annex XVII as it meets the PBT criteria.

Restrictions on marketing and use - RoHS Directive

The RoHS Directive relates to use of polybrominated diphenyl ethers – not further specified – in EEE. Article 4(1), together with Annex 1, No. 29 prohibits the use of PBDE in new EEE in concentrations above 0.1 % by weight (1,000 ppm) in homogeneous material. This restriction is in force since July 2006 and applies to the sum of PBDE congeners.

Annex 1 lists “Applications of (...) polybrominated diphenyl ethers (PBDE) which are exempted from the requirements of Article 4(1)” (i.e. from the market restriction). No. 10 of Annex 1 states that “Within the procedure referred to in Article 7(2), the Commission shall evaluate the applications for: - Deca BDE (...)”. By means of Decision 2005/717/EC, Annex 1 of Directive 2002/95/EC was amended by a point 9a., introducing a general exemption of Article 4(1) requirements for DecaBDE. However, the European Court of Justice has annulled the respective part of Decision 2005/717/EC with order of 1 April 2008, declaring that the effects would maintained until 30 June 2008 inclusive.

Note that restrictions apply to recycled products placed on the market which are not subject to waste regime any more as well.

Classification

OctaBDE is not classified under the CLP Regulation or under Directive 67/548/EEC.

Other fields (water)

The Water Framework Directive lists OctaBDE and DecaBDE among relevant substances to be monitored and establishes Environmental Quality Standards (EQS), both for annual average (AA-EQS) and for Maximum Allowable Concentration (MAC-EQS) for inland surface water for tetra-, penta and hexaBDE¹⁹ as follows:

- Inland AA-EQS: 0,0005 [µg/l]
- Other surface waters AA-EQS: 0,0002 [µg/l]

6.2.4.2 Member States level

Waste

Pursuant to the Austrian Waste Management Plan (Chapter 5.3 – version 2009) in cases where the content of 0,1 % for the sum of penta-, octa- and decabromodiphenyl ether in wastes (homogenous material) is exceeded, a notification procedure is required in case of transfrontier shipment, independently from the subsequent recovery operation. If these limits are exceeded the environmentally sound recycling in other fields of application abroad shall be controlled (precautionary principle). In case of the presence of higher contents of the above mentioned flame retardants, a hazard characteristic (teratogenic) is triggered (ban of export on hazardous wastes to non-OECD Member Countries).

6.2.4.3 Other countries

In Norway, waste containing more than 0.25 % of octaBDE is defined as hazardous waste and has to be treated according to the regulation on hazardous waste. In Switzerland, Recycling of articles containing PBDEs is not allowed. In Korea, recycling of electronic and electric devices is only allowed if the concentration of octaBDE in EEE Product is below the threshold of 0.1 %.

6.2.5 Occurrence of C-OctaBDE in Europe

The occurrence of C-OctaBDE in Europe can be considered its consumption. For the purpose of this study the term ‘occurrence/consumption’ has been defined as: (a) Use of the substance itself in the EU plus (b) Import of the substance in finished articles and materials minus (c) Export of finished articles and materials containing the substance.

- (a) Use

The total global market demand for C-OctaBDE decreased from 3,825 t in 1999 to 3,790 t (~610 t in EU) in 2001. Manufacturers estimated that approximately 1,350 t/y of C-OctaBDE was imported into the EU in finished articles (or master batch) and ~450 t/y used as the substance itself in 1999. For the purposes of the EU Risk Assessment (2003) a higher figure of 2,550 t/y for the total EU use was applied, as the maximum amount used in the past (1994), [BiPRO 2007; IPEN 2004].

¹⁹ For the group of priority substances covered by brominated diphenylethers (No 5) listed in Decision No 2455/2001/EC, an EQS is established only for congener numbers 28, 47 (tetra), 99 and 100 (penta), 153 and 154 (hexa)

All in all it can be summarized that the use of C-OctaBDE is phased out in the EU (including Norway and Switzerland). The placing on the market and use of C-OctaBDE was totally banned in 2003 (Directive 2003/11/EC). According to Bromine Science Environmental Forum (BSEF) the use of C-OctaBDE as flame retardant in polymer pellets in the USA stopped in 2004 and there are no more stockpiles left [BiPRO 2007].

- (b) Import

C-OctaBDE was imported into the EU as a component of finished articles or master batch (polymer pellets containing additives). Reliable figures for likely quantities involved are not available. Manufacturers estimate that a figure of approximately 1,350 t/y was realistic for the imports of C-OctaBDE into the EU in finished articles or master batch in 1999. Since the ban of C-OctaBDE in 2004 the import of articles containing C-OctaBDE into the EU is prohibited.

- (c) Export

For the purpose of this study it is assumed that no exports of products containing C-OctaBDE occurred at all. However, in case of E-waste which could possibly be contaminated with C-OctaBDE the situation is different. E-waste is currently being transported in huge amounts around the globe as highlighted by Watson et al. (2010). The authors further estimated that more than 80 % of computer E-waste is being exported to Asia, in particular to China. For instance, a recent study published by the German UBA and German Federal Ministry for Environment, Nature Conservation and Nuclear Safety (BMU) indicates that more than 155,000 t of waste is exported annually to non-European countries. This waste stream includes about 50,000 t of PC and television monitors, which could possibly also contain Polybrominated flame retardants. It was even highlighted that sometimes this appliances are re-classified as “functional” in order to be shipped to Asia and Africa. A quite similar export figure (160,000 t/y) was estimated for the UK [Watson et al. 2010].

- Occurrence/Consumption

The total production of all PBDEs from 1970 to 2005 was estimated to be 1,300,000-1,500,000 t [Watson et al. 2010]. This figure is further broken down to:

Commercial Mixture Total	Accumulated production (1970-2005) 1,300,000-1,500,000 [t]
C-DecaBDE	1,100,000-1,250,000 [t]
C-OctaBDE	102,700-118,500 [t]
C-PentaBDE	91,000-105,000 [t]

According to Watson et al. (2010), most of the C-OctaBDE containing wastes in the EU have already been treated through the existing waste management infrastructure. The authors [Watson et al. 2010] assess the situation to be similar to C-PentaBDE in Europe. In case of Acrylonitrilebutadiene-styrene (ABS), [Watson et al. 2010] indicate that most of the plastics have been landfilled, but with substantial amounts exported for recycling.

6.2.6 *Relevance of the occurrence of C-OctaBDE in Europe*

The main historic uses, applications and finished articles containing C-OctaBDE are summarized in Figure 6-17. It can be observed that the most common historic use of C-OctaBDE was in ABS polymers. Around 95 % of C-OctaBDE supplied in the EU was used in ABS (globally ~70 %). Thereby, C-OctaBDE was typically added at concentrations between 10-18 % by weight. The ABS in turn was mainly used for housing of Electrical and Electronic Equipment (EEE), typically office equipment and business machines. Other minor uses (~5 %) were High Impact Polystyrene (HIPS), Polybutylene Terephthalate (PBT), Polyamide Polymers, with typical concentrations between 12-15 % as indicated in Figure 6-17. Other possible uses found in literature were nylon, low density polyethylene, polycarbonate, phenol-formaldehyde resins, unsaturated polyesters and adhesives and coatings [BiPRO 2007].

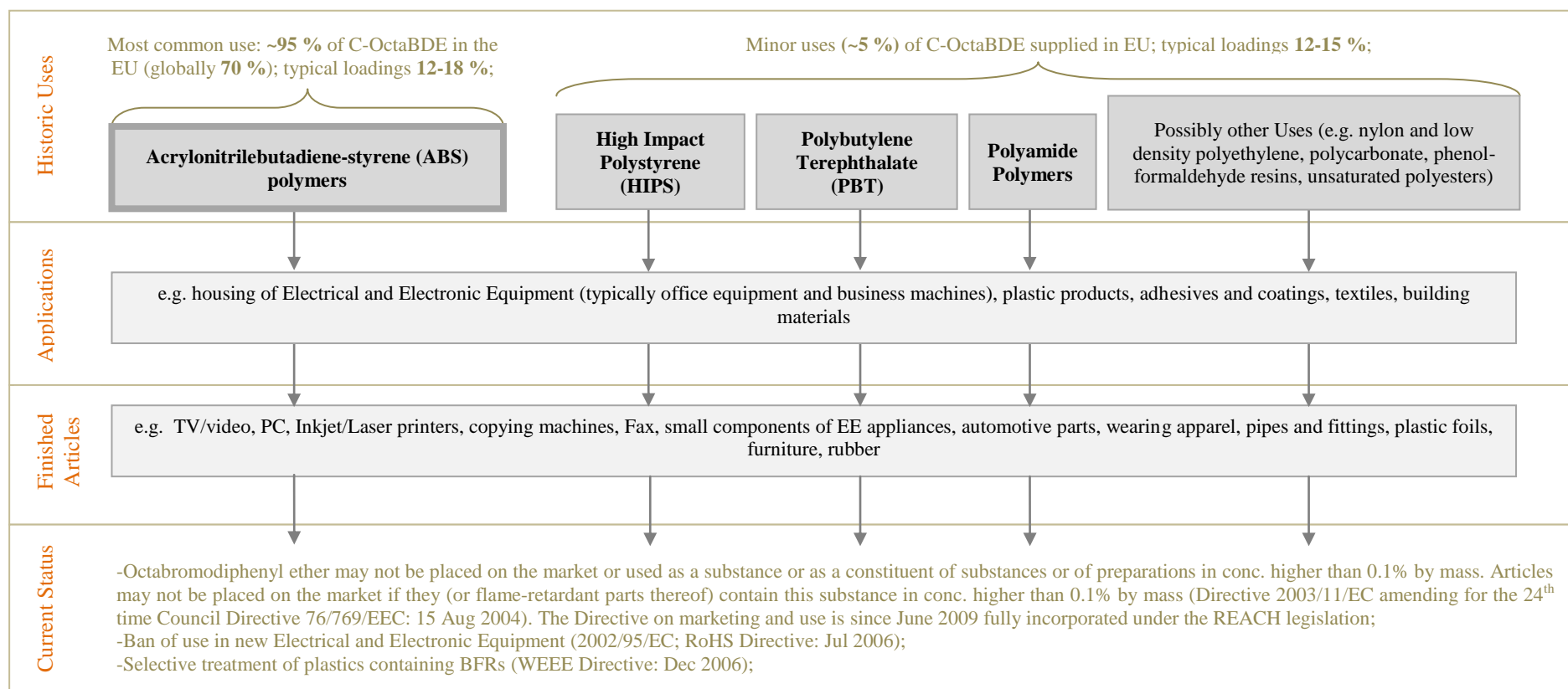


Figure 6-17: Summary of main historic uses of C-OctaBDE

6.2.7 Selection of relevant Uses, Applications and Finished Articles

As indicated previously, the most common historic use of C-OctaBDE was in ABS polymers. It is estimated that around 95 % of C-OctaBDE supplied in the EU was used in ABS. On the global scale this use was reported to account for around 70 %.

The ABS in turn was mainly used for housing of Electrical and Electronic Equipment, typically for the housing of office equipment and business machines [ECB 2003]. This is also in line with the assumptions made by Watson et al. (2010), indicating in their study that the major use of OctaBDE was in casings for business machines, computers and monitors. Besides, as identified during the data review process, this use is by far the most discussed use of C-OctaBDE within scientific literature.

Therefore, the main focus of this study will be on the use of C-OctaBDE in EEE, in particular on the ABS housings/casings of office equipment and business machines.

6.2.8 Substance Flow of C-OctaBDE

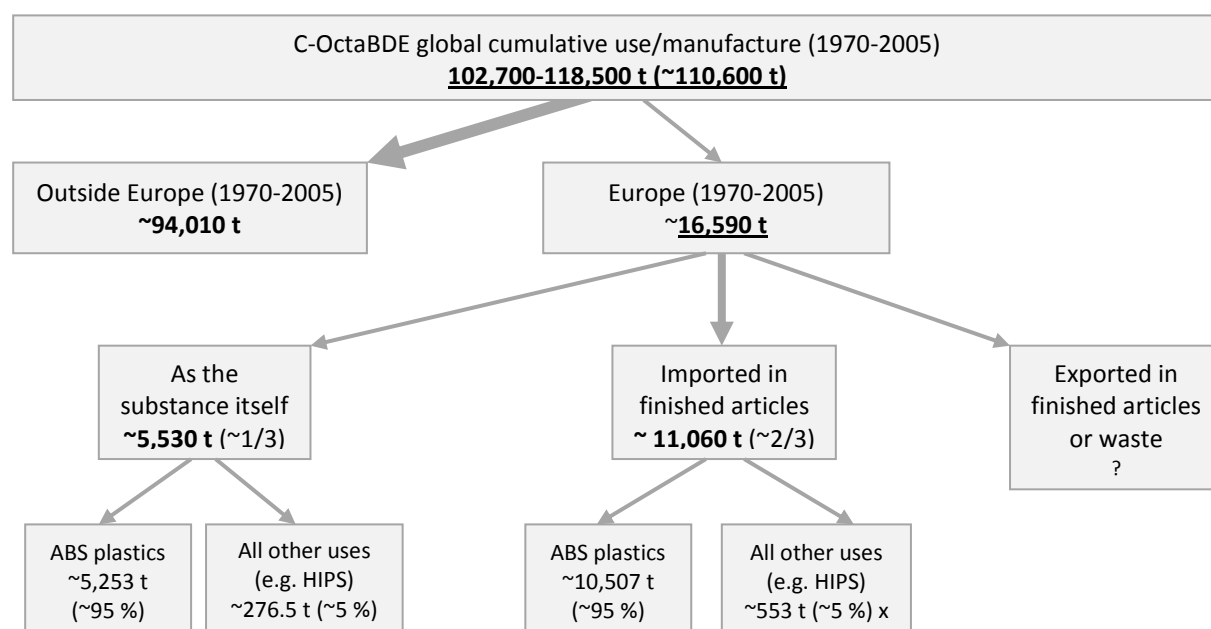


Figure 6-18: Overview of the substance flow contributing to the sources of C-OctaBDE in the EU

As indicated previously the global cumulative use/manufacture of C-OctaBDE was around 102,700-118,500 t for the time period 1970-2005. Due to the lack of exact distribution data it has been assumed that the most of C-OctaBDE was used outside Europe (~85 %), similar to the distribution of C-PentaBDE. The remaining 15 % were either used as the substance itself or imported into Europe in finished articles (or master batch). This is also in line with the estimations made by BSEF, stating that the total market demand for Octabromodiphenyl ether was split with around 40 % each being used in America and Asia, around 15 % in Europe and approximately 5 % in the rest of the world [Watson et al. 2010].

It has been further assumed that more than 2/3 of C-OctaBDE was imported into Europe in finished articles or master batch and the remaining 1/3 was used as the substance itself. This quite well corresponds to the previously indicated EU use figures of 1,350 t/y (imported in finished articles) and 450 t/y (used as substance itself) in 1999.

Finally, it has been assumed that 95 % of the substance itself was incorporated into ABS plastics in Europe, as well as imported into Europe in finished articles. A similar assumption was made in the EU Risk Assessment Report [ECB 2003] indicating that all of the Octabromodiphenyl ether used in any one country is in polymer applications (e.g. ABS).

Material streams with a total share less than 5 % are on the one hand widely distributed and not well characterised within literature. Therefore, considering the minor amounts of C-OctaBDE used, the diffuse distribution of such article and the limited data availability, these applications will not be further investigated and have been highlighted with an X in Figure 6-18.

Recent estimations of the current stocks of C-PentaBDE made by Watson et al. (2010) indicate a similar situation to existing stocks of C-PentaBDE, with the vast majority of stocks having reached the end of their useful lives. These wastes have been treated through the existing waste management infrastructure. In case of the ABS plastics the authors [Watson et al. 2010] conclude that most have been consigned to landfill but with substantial exports for recovery.

6.2.9 Housing of Electrical and Electronic Equipment

A) Background

As already indicated the main use of C-OctaBDE was in ABS casings/housings of office equipment and business machines (i.e. casing of computers, monitors). It was reported that C-OctaBDE was used in concentrations between 12-18 % by weight. However, the use of C-OctaBDE in ABS was declining throughout the 90's as it was increasingly replaced by alternative flame retardants such as Tetrabromobisphenol A [Watson et al. 2010]. Besides, there was a shift from the ABS (90's) towards PC/ABS and HIPS for outer casings. For monitors and personal computers PC/ABS dominates at present, while HIPS is mainly used for TV casings [Swiss 2003].

B) Process input (raw material) and output (waste, air, water, product)

Figure 6-19 shows the main inputs and outputs associated with the production of C-OctaBDE containing Electrical and Electronic Equipment.

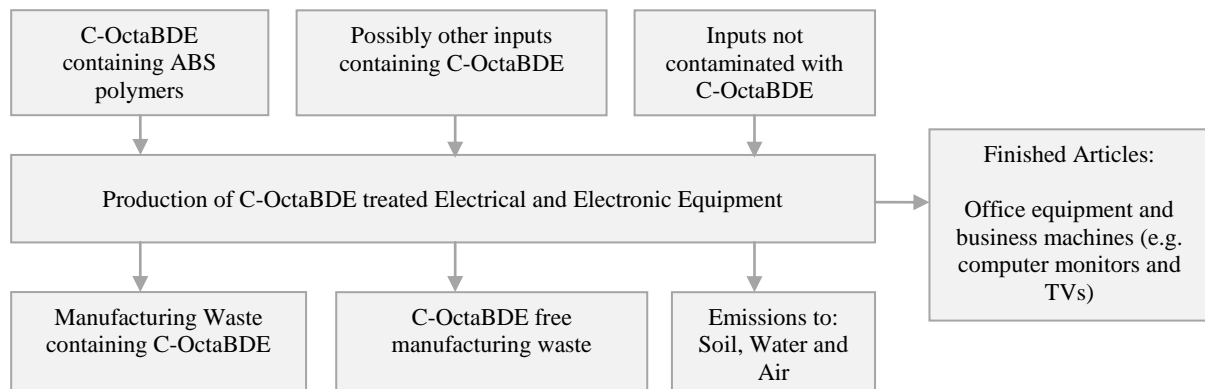


Figure 6-19: Relevant process inputs/outputs and products (EEE)

Inputs:

- The main focus of this study is on C-OctaBDE containing ABS polymers. This is regarded as the major input to the production process of C-OctaBDE containing EEE.
- Possibly other C-OctaBDE treated plastic components. However, as already indicated the main focus is on C-OctaBDE containing ABS polymers.
- C-OctaBDE free inputs

Outputs:

- Finished articles: C-OctaBDE was mainly used in housings/casings of various office equipment and business machines.
- Manufacturing Waste (containing C-OctaBDE; C-OctaBDE free): Different wastes were generated during the manufacture of C-OctaBDE containing products.
- Emissions to Soil, Water and Air: due to the ban, releases from production, handling and processing already ceased.

C) Generation factors for product/waste and other output (air, water)

The generation factor can be assumed as 1, because input and output are considered equal.

D) Contamination data for output (and input)

Several literature sources [e.g. ECB 2003; Octa_UNEP 2008] indicate a contamination figure of 12-18 % of C-PentaBDE in ABS polymers. Other sources [e.g. Zweifel 2001] indicate even higher C-OctaBDE contamination values (i.e. 18-22 %). Therefore, the average contamination figure of 17.5 % will be applied for further estimations.

Table 6-12 summarizes average concentrations of OctaBDE in relevant WEEE. It is, however, not always clear whether the studies refer to congeners, congener groups or commercial mixtures of OctaBDE. In few cases the studies refer explicitly to particular congeners (e.g. BDE-153, BDE-183). All data was obtained through sampling at WEEE recycling facilities in Switzerland, the Czech Republic and Germany.

Table 6-12: Concentrations of OctaBDE in WEEE [Swiss 2003; CreaSolv 2004]

Products	Concentrations /Loadings [mg/kg]	Data available for/(Location)	Comments
EEE (not further specified)	530 ± 30	sampled in 2003 (CH)	(Bulk WEEE input) average concentrations in small size WEEE (e.g. small household appliances, office and communication appliances, entertainment electronics) sampled in a CH Recycling Plant
housing of EEE and business machines	2,000	sampled in 2003 (CH)	average concentration values for octaBDE in the plastic fraction only
TV/PC housings (plastics)	7,500 ± 600	sampled in 2003 (CH)	mean concentrations in typical WEEE fraction
TV housings rear covers	7,700 ± 3600	sampled in 2003 (CH)	mean concentrations in typical WEEE fraction
monitor housings, television sets and video devices	>1,000	2009 (CH)	max. concentration value defined in RoHS directive is expected to be exceeded in ABS (in particular: monitor housing, television sets and video devices)
Casings (Monitor-, Computer-, Inkjet/Laser printers, Copying machines, Fax)	154,000 (200,000)	CH study, but var. sources e.g. form GER and DK (1998/90)	Concentration of OctaBDE in the plastic casings for older products (1990) and new products (1998).
Small components of EE appliances	150,000	CH study, but var. sources e.g. form GER and DK (1998)	concentration of OctaBDE with respect to the weight of small plastic components
Plastic Panels (solid matrix)	BDE-153,BDE-154:<0.01	CZ (same for 1995,2001 and 2003)	Plastic panels from Pioneer brand stationary radio cassette players with amplifier produced in 1995 (ABS), 2001 (PS) and 2003 (PS)
	BDE-183: 0.029; 222.2; 955.1	CZ (diff for 1995,2001 and 2005)	
	BDE-203,BDE-205: <0.18	CZ (same for 1995,2001 and 2003)	

Products	Concentrations /Loadings [mg/kg]	Data available for/(Location)	Comments
	BDE-209: 1.17; 1193; 1070	CZ (diff for 1995,2001 and 2005)	

Considering that most of the contamination data in Table 6-12 is based on samples from 2003 or even from the 90ies, the indicated contamination levels s lower nowadays. In a more recent study [EMPA 2010], C-OctaBDE was detected at concentrations above the ROHS-MCV of 1 g/kg in plastics from Information and Communication Technology (ICT) and consumer equipment (average 1 g/kg; maximum 1.6 g/kg), in Cathode Ray Tube (CRT) monitors (average 3.0 g/kg; maximum 10.6 g/kg) and in CRT TVs (average 1.2 g/kg; maximum 3.5 g/kg). Therefore, the study concludes that OctaBDE concentrations have to be expected in plastics from ICT (with the highest levels in monitors), as well as plastics from CRT TVs. This is seen as a consequence from the application of OctaBDE in ABS plastics in the past.

According to information provided by industry, analytical results from spot tests from 2010 of printed circuit boards used as input material in a secondary thermal metal reclamation process showed levels ranging between 3 and 300 ppm for the sum of PBDEs (Sum Tri- to DecaBDE with the relevant congener groups TetraBDEs < 100 ppm, PentaBDEs < 100 ppm, HexaBDEs < 50 ppm and HeptaBDEs < 10 ppm). Another spot test of printed circuit boards showed levels of nearly 1000 ppm for the sum of Tri- to DecaBDE (with the relevant congener groups TetraBDEs < 100 ppm, PentaBDEs < 100 ppm, HexaBDEs < 50 ppm and HeptaBDEs ~10 ppm). These spot measurement results cannot be considered representative. It can, however, be concluded that the PBDE content in the input fraction (which consists of printed circuit boards and the WEEE shredder heavy fraction) is subject to high variation. Both, the WEEE shredder heavy fraction as well as printed circuit boards may be contaminated with PBDEs in a range which is relevant in the context of the discussion on low POP content levels. However it is assumed that the PBDE content of the shredder heavy fraction has significantly lower levels of the relevant PBDE congener groups (analytical results are not available).

The Environmental Protection Agency in Ireland has recently undertaken sampling of WEEE plastics of monitors and of wastes from sewage treatment to determine concentration levels of the new and candidate POPs in those wastes. According to [EPA IRE 2011a] no Penta-, Hexa- or HeptaBDE were measured above the limit of detection in any of the sewage sludge samples (LOD ranging from <1 µg/kg – 10 µg/kg). Samples were obtained from 9 waste water treatment plants located n the Republic of Ireland. Samples of plastics from computer monitors were taken from a WEEE processing facility for the purpose of PBDE analysis.

A total of 15 samples of plastics from mainly computer monitors were taken from a WEEE processing facility for the purpose of PBDE analysis. The year of manufacture of the monitors that were samples ranged from 1991-2003. The samples were sent to a laboratory in Germany for PBDE analysis. The analytical method used was based on EN 62321:2009, apparently with slight modifications. The following observations can be made [EPA IRE 2011b]:

- 11 samples (including from monitors with year of manufactured date ranging from: 1991-2002) indicated levels ranging from non-detectable concentrations to low levels of POPs PBDEs which are all below the low POPs concentration limits proposed in this study.
- 4 samples indicated varying degrees of notable concentrations levels of POP PBDEs:
 - 1 sample exceeded the Low POP Concentration Level (Option 1 i.e. 10 ppm) for PentaBDE (year of manufacture: 1992)
 - 2 samples exceeded the Low POP Concentration Level (Option 1 i.e. 10 ppm) for HexaBDE (year of manufacture: 1992, 1995)
 - 4 samples exceeded the Low POP Concentration Level (Option 1 i.e. 10 ppm) for HeptaBDE (year of manufacture: 1992, 1995, 1998 and 2003). It is notable that a 2003 computer casing exceeded the LPCL 1 for HeptaBDE.
 - 2 samples exceeded the Low POP Concentration Level (Option 2 i.e. 200 ppm) for HexaBDE (year for manufacture: 1992, 1995)
 - 2 samples exceeded the Low POP Concentration Level (Option 2 i.e. 1,000 ppm) for HeptaBDE (year for manufacture: 1992, 1995)

Whilst there was evidence of the year of manufacture on the computer monitors it was very difficult to ascertain the year of manufacture of the TV monitors during sampling. This was one of the reasons why most of the samples were taken from the computer monitors. Therefore, it may be difficult to identify TV monitors that were manufactured prior to 2003 for the purpose of separation of potential POP containing waste plastics [EPA IRE 2011b].

France has provided information on levels of PBDEs in WEEE together with their comments on the draft final report. Computer casing have been shredded and sorted in order to obtain a polymeric fraction containing mainly ABS material. The samples were taken from two plants which are dedicated to the WEEE recycling. Test portions of three different samples of ABS have been analysed with the procedure described in the annex A of the standard EN 62321²⁰ (GC-MS).

The analysis performed to determine the brominated flame retardants concentration in the three samples shows the presence of PBDEs at levels between 600 and 740 mg/kg for the sum of PBDEs (including DecaBDE). The concentrations for the relevant congeners were about 150 ppm for HexaBDE and between 60 and 100 ppm for HeptaBDE.

E) Activity data

In order to get an indication on export of WEEE, European Environmental Agency (EEA), made assumption on the example of exported colour television sets. Following this data about 3.6 million colour television were exported outside the EU, of which 15.000 tonnes of colour television sets are exported from the EU to African countries (in 2005). Whether these televisions were functioning TV sets or were destined for scrapping can only be assumed. The total export of used computers, mobile

²⁰ EN 62321: Electrotechnical products - Determination of levels of six regulated substances (lead, mercury, cadmium, hexavalent chromium, polybrominated biphenyls, polybrominated diphenyl ethers)

phones, printers, CD players etc. — of which an unknown quantity may be waste — to these regions is expected to be significantly higher [EEA 2009].

The European Topic Centre on Resource and Waste Management (ETC/RWM) states that illegal shipments of WEEE is often used by the different interpretation of waste and second hand goods by the exporting and importing countries, even by the authorities and enforcement agencies of the countries [ETC/RWM 2008].

Following the analysis of the treatment situation for waste electrical and electronic equipment and electronic scrap of [UBA 2010], infrastructure in the countries of destination is mostly not fulfilling the minimum level of protection for the environment and human health. Dismantling takes often place essentially through the informal sector with no control or monitoring in place. As problematic issues the recycling of equipment containing large quantities of raw material which cannot be recycled or marketed and the recycling is problematic with the treatment infrastructure set up in the countries of destinations. The risks for human health and the environment can be differentiated into four areas [UBA 2010]:

- Treatment steps in which a mechanical disassembling involves destruction of the material structures (e.g. the smashing of cathode ray tubes);
- Hazardous substances which are contained in the products and which might be emitted e. g. by thermal processes. The quantity and number of these substances are reduced through the ecological optimisation of the products within the scope of manufacturers' activities and the limitation through the RoHS-Directive from 2007. The use of such equipment as used equipment in the destination countries takes place, however, time-delayed (commensurate with the service life in Germany);
- Auxiliary products from separation and/or recycling processes (e.g. liquids and sludge from leaching);
- Substances which result from the handling or recycling (e.g. PCDD/F with thermal processes).

Data on exports of WEEE has been obtained from the EUROSTAT database 'WEEE collected and treated in 2006'. Only the categories 3 (IT & Telecommunication) and 4 (Consumer Equipment) have been considered. The figures for the EU27 indicate that around 432,676 t of WEEE (category 3 and 4) were collected in 2006 in the EU, of which around 97 % was treated within the EU and only around 3 % was exported for treatment. These figures of course do not consider illegal shipments.

Therefore, for the purpose of this study 3 % for WEEE exports will be applied for further estimations, however, keeping in mind that this figure could possibly be significantly higher due to illegal shipment of WEEE.

According to GEO (2007) the estimated lifetime of office and consumer electronics is approximately 9 years and is even expected to decrease to 7 years in the future. The Environmental Protection Agency in Ireland has recently undertaken sampling of shredder residues to determine the concentration levels of the new and candidate POPs. While the results of this analysis will not be available until March, it was observed that the age of the monitors being processed ranged from 1991 – 2003. While a full age

profile of the monitors was not undertaken it appears from the limited sampling undertaken that the time line for relevant WEEE plastics to be contained in the waste stream may be greater than suggested above [EPA IRE 2011a]. Accordingly the lifetime of 7 to 9 years may be underestimated.

TV and computer monitors provide two of the richest sources of plastics. The plastic content in these articles can vary between 10 and 40 % by weight [Retegan et al. 2010]. Therefore, the average of 25 % will be applied for further estimations.

According to our estimations, the 128 t/y of C-OctaBDE in EEE has to be treated in the EU in 2010. Besides, it can be summarized that with the assumed lifetime all C-OctaBDE will be disposed of by approximately 2012.

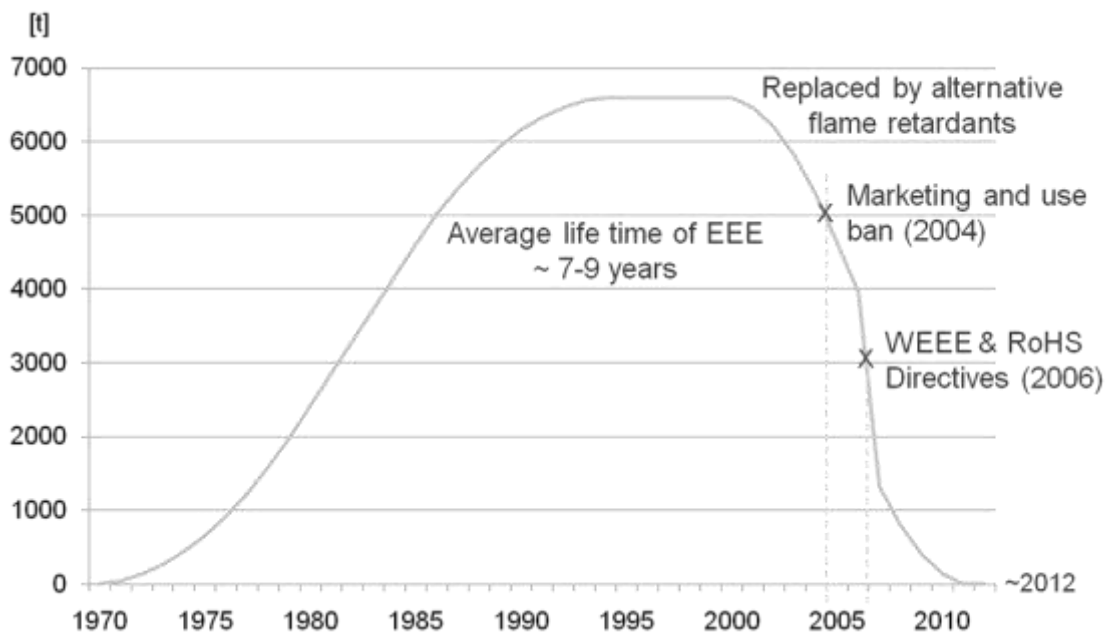


Figure 6-20: Estimated accumulated amount of C-OctaBDE in EEE

F) Waste treatment

The current waste management practices for waste containing C-OctaBDE are summarized in Table 6-13. The information has been obtained from responses submitted to the Questionnaire on 'new POPs' in accordance with SC-4/19. Until now, the below listed EU countries provided relevant information in relation to waste management options for waste containing C-OctaBDE. Most of the waste management options also consider the treatment of articles contaminated with C-PentaBDE.

Table 6-13: Summary of Waste Management Options [SC-4/19: BG, DE, SE, AT and LT 2010]

Country	Current Waste Management Options
Austria [AT]	Recycling Operations: -Plastic recovery: density separation and IR technology- accumulation of penta-and octaBDE in separated plastic fractions, which are not destined for recycling but for incineration Disposal: - Incineration in line with the Austrian Waste Incineration Ordinance; In Austria a ban on landfilling of plastic waste and other wastes showing a high total organic carbon content (national TOC-limit: 5% or calorific value at max. 6000kJ/kg) has been introduced in 2004. Export permits for landfilling of such waste abroad would not be granted in accordance with the EC Waste Shipment Regulation ; → End-of Life vehicles: Shredded fluff (containing PBDE) has to be incinerate as landfilling is not allowed; → WEEE: Plastics containing PBDE has to be incinerated as landfilling is not allowed
Bulgaria [BG]	Permanent storage shall be allowed only when all of the following conditions are met: 1. The storage takes place in one of the following locations: safe, deep, under-ground, hard rock formation; salt mines; a landfill site for hazardous waste, provided that the waste is solidified or partly stabilized where technically feasible as required for classification of the waste in subchapter 1903 of decision 2000/532/EC. 2. The provisions of Council Directive 1999/31/EC and Council Decision 2003/33/EC were respected. 3. It has been demonstrated that the selected operation is environmentally preferable.
Czech Republic [CZ]	Recycling Operations: - Separate collection of different waste streams of EEE, i.e. large household appliances, small household appliances, IT and communication equipment, consumer equipment, etc. Disposal: - Specially engineered landfill, hazardous waste incineration, environmentally sound remediation, waste incineration, base-catalysed decomposition (BCD), thermal desorption
Germany [DE]	Recycling Operations: - Dismantling (separation of contaminated from halogen-free plastic fractions) - Sorting (NIR and X-ray transmission) - Solvent based operations (separation of polymer and contaminants through solvent based operation) - Feedstock recycling (depolymerisation of plastic waste) Disposal: -Incineration and thermal energy recovery
Poland [PL]	Recycling Operations: - Use as a fuel, recycling/reclamation of metals and metal compounds, other activities including the use of waste in whole or in part Disposal: - main method is incineration
Sweden [SE]	Recycling Operations: - Recycling of metals from printed circuit boards; - WEEE plastics and plastics parts are sorted out at pre-treatment plants and afterwards incinerated - Printed circuit boards are extracted and sent for metal recovery; - Energy recovery from incineration of WEEE plastics; - Handling of shredded light fractions (SLF) and fines from scrapped cars. Disposal: - Incineration with energy recovery is the only method in use at large scale for the handling of articles

Country	Current Waste Management Options
Latvia [LT]	Disposal: The decision whether collected EEE waste for disposal could be treated as dangerous waste shall be made in accordance with the List of Waste types (Annex 2 of the Rules of Waste Management). This list is used for the classification of dangerous waste in line with Decision of the European Commission 2000/532/EC. The composition of hazardous waste collected is not chemically analysed. The EEE waste which is defined as dangerous is being transferred to the company UAB "Tiksika" conducting the long storage and treatment of dangerous waste. This company stores or transports the dangerous waste to other countries for final disposal.

According to Watson et al. (2010), there is only a limited number of full scale operating plants for the recycling of PBDE containing articles. Some companies perform full scale recycling of plastics from WEEE (e.g. Axion Recycling, MBA Polymers, RUAG, and WERSAG) however most do not indicate the details related to the separation techniques applied.

G) C-OctaBDE flow

Based on the information and assumptions presented in this chapter as well as on the data obtained and evaluated from the EUROSTAT database, the following C-OctaBDE flows have been developed.

In the first scenario, data obtained from the EUROSTAT database has been taken into consideration. The evaluated EUROSTAT data indicates that approximately 20 % of the WEEE (category 3 and 4) is incinerated with energy recovery, around 18 % is recycled and only a small fraction (~0.5 %) is directly reused as a whole appliance. Consequently, around 61 % of the WEEE is disposed of to landfills or possibly incinerated without energy recovery. The waste management options evaluation has been only performed for category 3 (IT and communication) and category 4 (consumer equipment), considering the relevance of these two categories in relation to C-OctaBDE. As the EUROSTAT database does not distinguish between disposal to landfills and incineration without energy recovery (D10) it is assumed that equal waste shares will be incinerated without energy recovery and landfilled (see Figure 6-21).

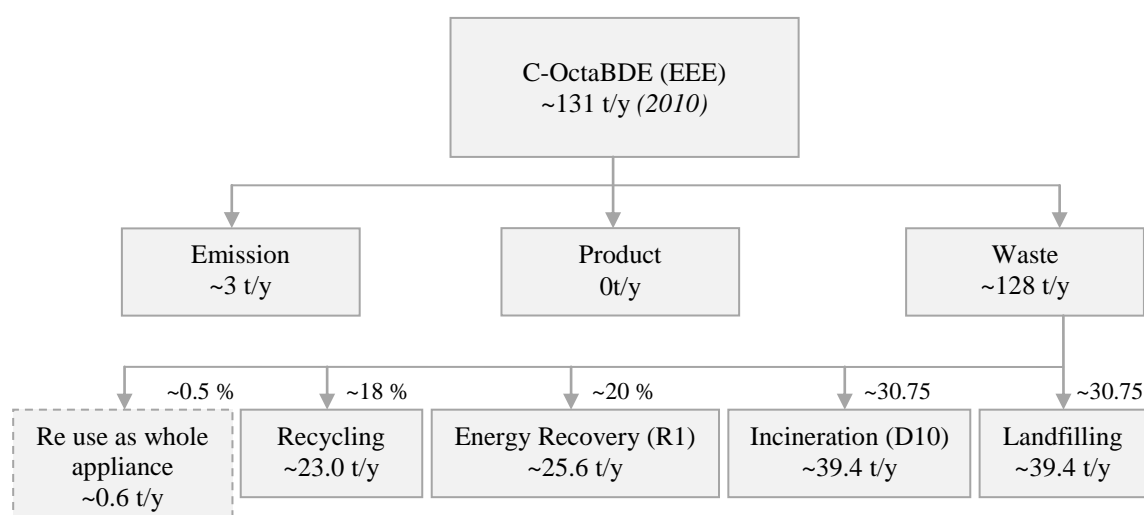


Figure 6-21: C-OctaBDE flow (Scenario 1: C-OctaBDE containing WEEE treated together with WEEE categories 3 and 4)

In the second scenario it is assumed that the C-OctaBDE contaminated plastics can be successfully separated from the non contaminated WEEE fraction, as requested by the WEEE Directive. The contaminated plastic fraction will then, to equal shares, be incinerated with and without energy recovery as shown in Figure 5-21.

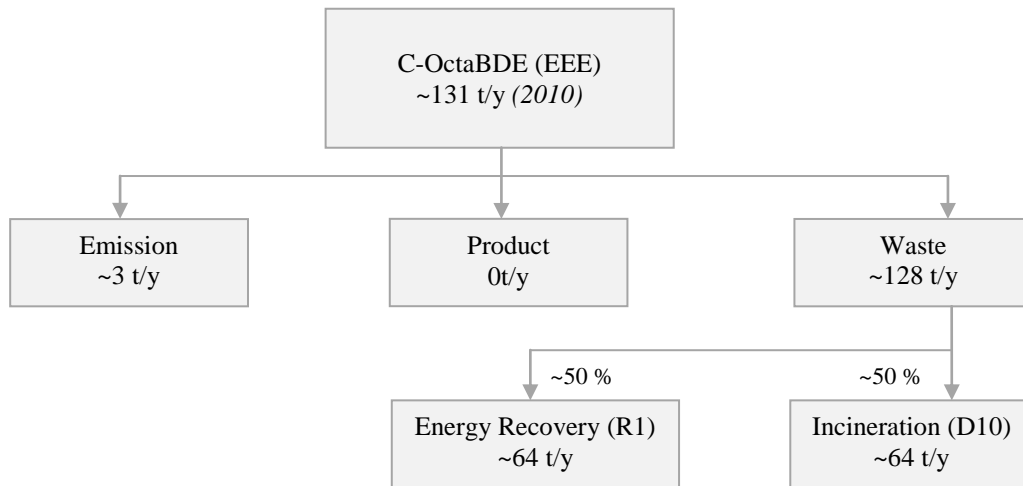


Figure 6-22: C-OctaBDE flow (*Scenario 2: C-OctaBDE containing WEEE successfully separated and incinerated, R1/D10)

The emissions have been estimated in accordance with the methodology presented in the EU Risk Assessment Report [ECB 2003]. Therefore, 0.54 % has been applied for volatile emissions and additional 2 % for particulate losses as proposed in the Risk Assessment study. This leads to a total emission of around 3 t/y.

H) Waste flow

As shown in Figure 6-23 approximately 732 t of WEEE containing C-OctaBDE will have to be treated in the EU in 2010.

As already mentioned, in the first scenario (see Figure 6-23) the C-OctaBDE contaminated waste is treated together with other non contaminated WEEE of the category 3 and 4. The total amount of WEEE of the categories 3 and 4 to be treated in the EU27 has been obtained from the EUROSTAT database (~419,414 t/y). Only around 25 % of the WEEE categories 3 and 4 can be assigned to plastics (~104,853 t/y).²¹

Consequently, due to the increased amount of the WEEE plastics to be treated, the contamination figures decline to approximately 1,230 mg/kg. This waste will be finally disposed (~61%), incinerated with energy recovery (~20%), reused/recycled (~18%) and possibly reused as a whole appliance (~0.5%), as indicated in Figure 6-23. Besides, it is assumed that equal waste amounts will be disposed of by incineration without energy recovery and landfilling.

²¹ French authorities noted that in France in 2009 more than 20,000 tonnes of WEEE plastics have been treated. It has to be noted that the 104,853 tonnes mentioned here are only the plastic fraction of WEEE categories 3 and 4. The total amount of WEEE plastics at EU 27 level is considerably higher.

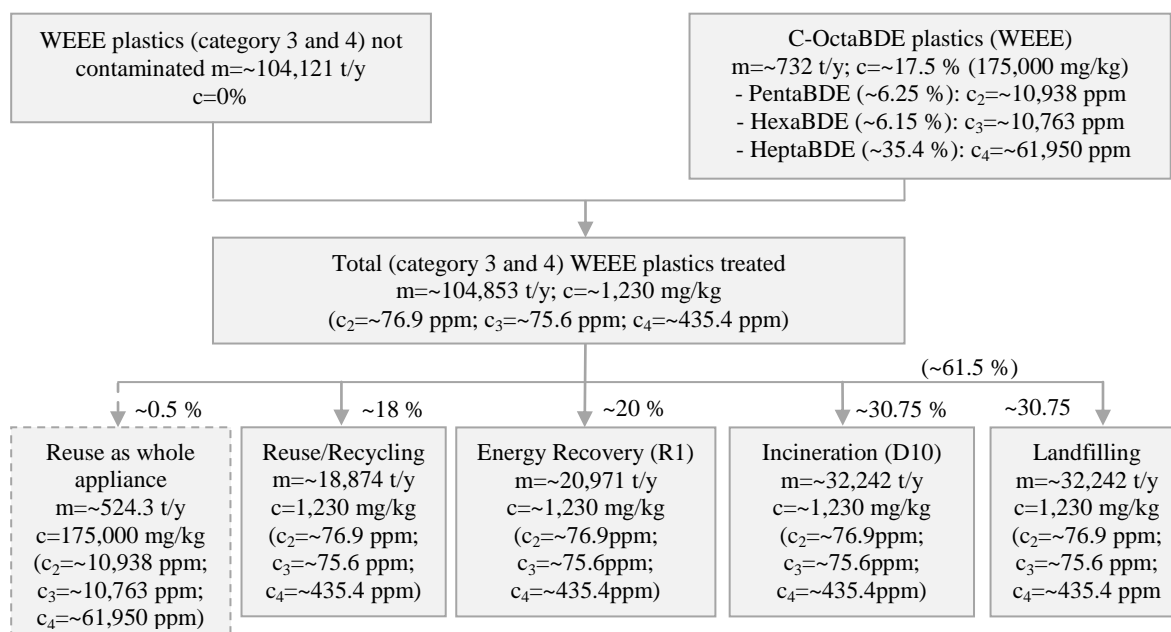


Figure 6-23: Waste flow for C-OctaBDE and relevant congener groups in WEEE plastics (Scenario 1: C-OctaBDE containing WEEE plastics treated together with WEEE plastics categories 3 and 4)

The second scenario (see Figure 6-24) is based on the assumption that the 732 t/y of C-OctaBDE contaminated plastics ($\sim 175,000$ ppm) can be successfully separated from the non contaminated WEEE fraction, as requested by the WEEE Directive. The contaminated plastic fraction will then, to equal amounts, be incinerated with and without energy recovery as shown in Figure 6-24.

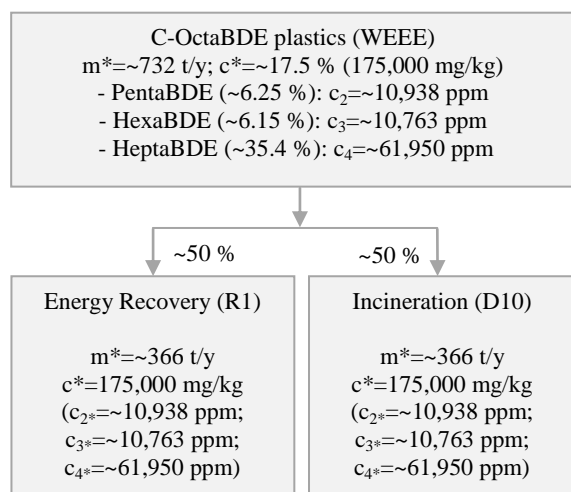


Figure 6-24: Waste flow for C-OctaBDE and relevant congener groups in WEEE plastics (*Scenario 2: C-OctaBDE containing WEEE plastics successfully separated and incinerated, R1/D10)

6.2.10 References C-OctaBDE

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6.3 PFOS flows

6.3.1 Use and Production

Perfluorooctane sulfonic acid and its derivatives (PFOS) are synthetically produced fully fluorinated organic molecules. PFOS is persistent in the environment and has to be destroyed by incineration. It is estimated that the global production of PFOSF, which is the basic chemical for the production of PFOS, has been 96,000 tons between 1970 and 2002. On 16 May 2000, 3M, the major global producer of PFOS, announced its phase-out of the production of PFOS.

Other companies which produced PFOS have been among others Du Pont, Bayer, Clariant Ciba Speciality Chemicals, Daikin, Arkema, AGC Chemicals/Ashi Glass or Solvay Solexis. The increased awareness about concerns for PFOS led to a strong decrease in the use of PFOS between the years 2000 and 2004. Relevant alternatives for the use of PFOS are perfluorinated telomers with shorter chain length.

6.3.2 Chemical Characteristics of PFOS and its derivatives

Perfluorooctane sulfonic acid and its derivatives (PFOS) are synthetically produced fully fluorinated organic molecules. PFOS is a fully fluorinated anion, which is commonly used as a salt or incorporated into larger polymers. PFOS and its closely related compounds are members of the large family of perfluoroalkyl sulfonate substances [UNEP_2006a].

Chemical name: perfluorooctane sulfonate and derivatives (PFOS)

Molecular formula: $C_8F_{17}SO_3^-$

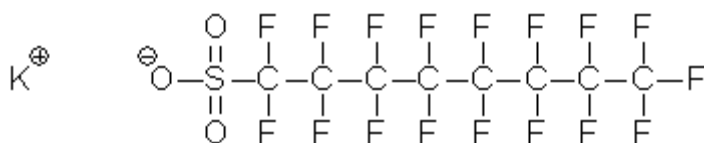
PFOS as an anion does not have a specific CAS number. The parent sulfonic acid and some of its commercially important salts are listed below:

CAS Registry Number: perfluorooctane sulfonic acid (CAS No. 1763-23-1)
potassium salt (CAS No. 2795-39-3)
diethanolamine salt (CAS No. 70225-14-8)
ammonium salt (CAS No. 29081-56-9)
lithium salt (CAS No. 29457-72-5) [UNEP_2006a]

Another derivative considered as important is the fluoride salt, where PFOS is the cation:

heptadecafluorooctanesulfonyl fluoride (CAS No. 307-35-7) [ESIS 2010]

Below structure, molar mass and physico-chemical properties of the potassium salt are indicated.

Structure:

Molar Mass: 538.21 g/mol [GESTIS 2010]

Physico-chemical properties [UNEP_2006a]

Melting point	>400 °C
Boiling point	Not measurable
Vapour pressure (at 100 °C)	3.31 x 10 ⁻⁴ Pa
Partition coefficient Log Kow	Not measurable
Water solubility	519 mg/L (20±0.5 °C) 680 mg/L (24-25 °C)
Air-water partition coefficient	<2 x 10 ⁻⁶
Henry's Law Constant	3.09 x 10 ⁻⁹ atm m ³ /mol pure water

The wide use of PFOS is due to its surfactant properties with extremely low surface tension and refractive index. It is oleophobic as well as hydrophobic and can therefore be used as a repellent for water, soil and dirt. The PFOS salt is more hydrophilic and disperses more easily in the water environment than the non-dissociated acid and sulfonamides, which are less hydrophilic and are more volatile and will be able to be transported long-range by air.

The stable and non-reactive properties of PFOS result from the strong carbon-fluoride bond. PFOS can even resist strong acids and high temperatures and is not degradable in the environment. More complex PFOS related chemicals can degrade to the basic PFOS molecule during its use or in the environment. The majority of PFOS related substances are polymers of high molecular weights, of which PFOS is only about 1 % of the whole molecule. Also these polymers have been considered in the mass flow analyses and other evaluations of the present report.

6.3.3 Environmental Background Levels

PFOS has spread widely in the global environment. Existing monitoring data on PFOS show that concentration levels of the substance were detected in different environmental matrices including sediments and soil as well as in biota [UNEP 2006b].

Table 6-14 and Table 6-15 give an overview of the concentration levels in sediment and sewage sludge detected in the USA and Canada.

Table 6-14: Levels of PFOS in sediment and soil (DW: dry weight; ww: wet weight; ND: not detected) [UNEP 2006b]; [HelsinkiComission_2009]

Country/Region	Compartment	Levels of PFOS	References	Comments
Gulf of Finland	Sediment	0.22-1.0 µg/kg ww	[Kallenborn et al. 2004]	2 Finnish coastal sediments sampled in 2003, note that concentrations are expressed as wet weight.
Canada, Niagara-on-the-Lake in the Niagara River	Sediment	0.005 – 1.1 µg/kg	[Furdui et al. 2005]	Not indicated whether dw or ww.
Kattegat, Great Belt and Sound, Denmark	Sediment	<1.0 µg/kg dw	[Strand et al. 2007]	6 Danish coastal sediments sampled in 2003-2004.
Eckernförde Bight, Germany	Sediment	0.58 µg/kg dw	[Theobald et al. 2007]	1 German coastal sediment sampled in June 2005.
Kiel (inner) Bight, Germany	Sediment	0.32 µg/kg dw		2 German coastal sediments sampled in June 2005.
Mecklenburg Bight, Germany	Sediment	0.67 µg/kg dw		1 German coastal sediment sampled in June 2005.
Arkona Sea, Germany	Sediment	0.03 µg/kg dw		1 German open sea sediment sampled in June 2005.
Ruden, Germany	Sediment	0.13 µg/kg dw		1 German open sea sediment sampled in June 2005.
Stettin, Germany ²²	Sediment	0.03 µg/kg dw		1 German open sea sediment sampled in June 2005.

Table 6-15: Levels of PFOS in sewage sludge including hot spot levels (DW: dry weight); [NRW_2009]; [BLU_2010]; [Weber et al. 2010]; [OECD_2002]

Country/Region	Compartment	Levels of PFOS	References	Comments
USA	Municipal waste water treatment plant sludge	0.2 – 3.120 µg/kg dw	[OECD_2002]	Sampled in 6 Urban Centres in the USA.
Hot spot levels (sewage sludge and contaminated soil)				
USA	Sediment	ND – 53.1 µg/kg dw	[OECD 2002]	Sampled in 6 Urban Centres in the USA.
Nordrhein-Westphalen, Germany	Sewage sludge	6,300 µg/kg	[NRW_2009]	The maximum concentration level detected within the samples in different waste water treatment facilities is indicated.
Bayern, Germany	Sewage sludge	1,900 µg/kg dw	[BLU_2010]	Dry product was examined by 30°C.

²² Remark of the authors: Stettin is not in Germany. It is assumed that the sample was taken in the Bay of Szczecin.

Country/ Region	Compartment	Levels of PFOS	References	Comments
Düsseldorf, Germany	Soil	Up to 6,410 µg/kg	[Weber et al. 2010]	The maximum value is indicated detected by examination of PFC contaminated soils after using fire fighting foams. In the study it is assumed that at least in soil PFOS has the highest share in the detected concentrations which is estimated to be 92-100%.

The concentration levels of PFOS indicated in the Table 6-15 by [OECD_2002] have been detected within studies carried out in the US. The hot spot levels have been determined in surface water and sediment downstream of a production facility as well as in water treatment plant effluents, sewage sludge and landfill leachate at a number of urban centres. Different city types were involved in the studies. The cities Decatur (AL), Mobile, Columbus (GA) and Pensacola have been manufacturing or industrially using fluorochemicals whereas Cleveland (TN) and Port St. Lucie are two cities that do not have significant fluorochemical activities. In general, the cities without such relevant industry activities inhabited the lower end of the indicated ranges in all taken water, sewage sludge and sediment samples. However, in addition to exemption for water samples in Port St. Lucie, the findings in the municipal wastewater treatment plant effluent and sludge in Cleveland were intermediate ranges [UNEP_2006b].

In the study of Furdui et al. published in 2005, it was stated that preliminary findings suggest that the concentration levels of PFOS increased during the period from <400 pg g⁻¹ in the early 1980 to 1000 pg g⁻¹ in 2002 [Furdui et al. 2005].

[Weber et al. 2010] reported that analytical studies in a garden plot showed that PFOS contaminated groundwater used for irrigation lead to translocation of the contaminants to the topsoil and up to 18 µg/kg of PFOS have been identified. Cultivated plants grown in the same garden plot showed residues of up to 32 µg/kg of which PFOS was one of the main components next to PFH_xS and PFDoA. It was not specifically mentioned that the plants have absorbed the substance by soil and not only by the water itself.

As shown in Table 6-14, in the environment the detected background contamination levels of PFOS in sediment ranged from less than 0.005 µg/kg to a maximum concentration level of 1.1 µg/kg dw in sediment determined in Niagara-on-the-Lake in the Niagara River (Canada). In general, the determined concentration levels in sediment were below 1 µg/kg DW. The highest hot spot levels were detected in sewage sludges (maximum levels of 1,900 µg/kg and 6,300 µg/kg) and in PFOS contaminated soils after using fire fighting foams in Düsseldorf (Germany) reaching 6,410 µg/kg.

6.3.4 Legal Situation

6.3.4.1 EU level

POPs related provisions

PFOS is listed in Annex I, IV and V to the proposal for an amended POPs Regulation. The substance is identified as Perfluorooctane sulfonic acid and its derivatives (PFOS) $C_8F_{17}SO_2X$ ($X=OH$, metal salt ($O-M+$), halide, amide, and other derivatives including polymers).

Annex I sets out values for unintentional trace contaminant in substances, preparations or articles as mentioned in Article 4(1) (b) to which the control of production, placing on the market and use as of Article 3 shall not apply:

- concentrations of PFOS equal to or below 10 mg/kg (0.001 % by weight, 10 ppm) when it occurs in substances or in preparations;
- concentrations of PFOS in semi-finished products or articles, or parts thereof, if the concentration of PFOS is lower than 0.1 % by weight (1,000 ppm) calculated with reference to the mass of structurally or micro-structurally distinct parts that contain PFOS;
- amount of PFOS is lower than 1 $\mu\text{g}/\text{m}^2$ of the coated material for textiles or other coated materials.

For diverse uses, limited exemptions or restricted uses are listed in Annex I for PFOS, subject to review under the conditions indicated:

- until June 2011 for firefighting foams,
- until 26 August 2015 for wetting agents for use in controlled electroplating systems,
- photo resists or anti-reflective coatings for photolithography processes,
- photographic coatings applied to films, papers, or printing plates,
- mist suppressants for non-decorative hard chromium (VI) plating in closed loop systems,
- hydraulic fluids for aviation.

In Annex V, no upper and lower POPs limits have been set for PFOS.

Waste management

Wastes containing PFOS are subject to the general rules of EU waste legislation (see chapter 4.1.2). Provisions specifically related to PFOS containing wastes have not been identified.

Restrictions on marketing and use

Following Annex XVII, entry No 53 of REACH (Regulation (EC) No 1907/2006), PFOS (similarly identified as within the POPs Regulation)

- shall not be placed on the market, or used, as substances or in mixtures in concentrations equal to or greater than 50 mg/kg (0,005 % by weight),
- shall not be placed on the EU market in semi-finished products or articles, or parts thereof, if the concentration of PFOS is equal to or higher than 0.1 % by mass calculated with reference to the mass of structurally or microstructurally distinct parts that contain PFOS or, for textiles or other coated materials, if the amount of PFOS is equal to or higher than 1 µg/m² of the coated material.

These measures have to be applied by the Member States since June 2008. Some exceptions have been included (corresponding to the envisaged exemptions in the POP Regulation).

- Metal Plating
- Hydraulic fluids for the aviation industry
- Photo industry
- Semiconductor industry
- Fire fighting foam (storage until 27 June 2011).

Classification

There is no official classification of PFOS or its salts under CLP Regulation or Directive 67/548/EEC as such. The various salts may be classified by the manufacturer.

Other fields

The Water Framework Directive establishes a list of priority substances in the field of water policy. PFOS is identified as a possible priority substance for review, see Annex III Dir 2008/105.

PFOS is not subject to EU regulations in the field of food and feed safety.

6.3.4.2 Member States level

Waste management

No additional limit value has been identified for the management of PFOS containing waste at Member State level.

Waste water treatment

Limit values of 100 µg PFOS+PFOA/kg (0.1 ppm) have been reported from Austria and as an orientation values, from North Rhine-Westphalia. According to [NRW_2009] sewage sludges with maximum concentration levels up to 200 µg/kg of PFOA+PFOS do not represent a risk for the examined subject of protection in the environment. North Rhine-Westphalia therefore envisaged introducing a

concentration limit value of 100 µg/kg (0.1 ppm) for sewage sludge that is used on agricultural soils [NRW_2009].

6.3.4.3 Other countries

PFOS related regulations have been reported from Korea:

- Fire fighting foam waste containing 0.005 % by weight or more of PFOS and PFOS related compounds is defined as hazardous waste and has to be treated according to the regulation on hazardous waste. Permissions to export hazardous waste for recycling are not granted.
- It is prohibited to produce, import, export and market impregnated articles containing more than 0.005% of PFOS and PFOS related compounds. It is prohibited to produce, import, export, market and use fire fighting foam containing more than 0.005 percent PFOS and PFOS related compounds. From 1 July 2007 it is prohibited to produce, import, export and market textiles if the amount of PFOS and PFOS related compounds are similar to or higher than 1 microgram per square meter. Prohibition of production, import, export and use of PFOS and PFOS related compounds for all uses, except for aviation hydraulic fluids, certain photo purposes and hard metal plating in closed-loop systems, will be fully implemented in 2010.

6.3.5 Occurrence and Relevance of PFOS and its derivatives in Europe

Since about 2002 the use of PFOS in the different industries has significantly dropped [DEFRA_2004b]. Especially in industries where PFOS free alternatives are available, a shift to PFOS free products and processes have occurred.

Regarding exceptional uses of PFOS, fire fighting foams have to be highlighted as they have to be destroyed by 27 June 2011. It is expected that in the following year, large stocks of fire fighting foams will be thermally destroyed and that from 27 June 2011 on this source can be considered as negligible.

All the other applications have to be below the PFOS limit values since 2008 and, in general, these have often already been PFOS free in 2002. However, depending on the lifetime of the products, it is possible that they still enter the waste stream and can lead to PFOS emissions. This is especially true for products with comparatively long lifetimes such as upholstery (leather industry) or carpets for which lifetimes of ten and 14 years, respectively, are stated. Therefore, the corresponding waste streams will still contain PFOS until approximately 2015. Other industries with products having a shorter lifetime than upholstery and carpets such as textiles, paper and cardboards do not have an influence on the PFOS emissions from products during their service life at the present time.

Also industries which used PFOS during their production processes, but where PFOS was not included in the final product, and who had to cease the use of PFOS, such as the mining industry, do not pose a current source in regard to PFOS emissions.

In the past, PFOS has been used for a wide range of products and processes. Among these industries and products are pesticides and insecticides, the rubber industry, plumbing fluxing agents, medical applications, flame retardants or adhesives [POPRC_2010]. PFOS might have been used in several individual applications, but the used amounts have been relatively small and unimportant. There is no information available which justifies that one of these applications should be further investigated. The use of PFOS for pesticides in Europe for example is only known from one Italian producer who manufactured about 500 kg annually before 1999. In the meanwhile, the production has stopped and the pesticides have already been totally used.

In some areas in Europe, elevated concentrations of PFOS in soil or water have been indicated [NRW_2007c]. Former PFOS production sites as well as the use of fire fighting foams can be regarded as responsible for these elevated PFOS concentrations among others reason. Another source might be an upstream located industry, which used PFOS in its production process. Also the use of contaminated sewage sludge for agricultural purposes can be a source of PFOS. Sewages sludges from waste water treatment plants which are located downstream from chromium plating plants, paper and cardboard facilities or textile plants are prone to contain elevated PFOS concentrations. Even in case the industry does not use PFOS any more at their site, the persistent nature of PFOS in combination with its surface properties, a higher PFOS concentration can even be found years after the use of a PFOS free alternative. PFOS contaminated soils are a continuous source of PFOS leachate if not properly managed.

6.3.5.1 Current uses

The major current use of PFOS in Europe is considered to take place in the metal plating industry with a yearly consumption of about 6,500 kg of PFOS [ZVO_2010b]. According to conversation with industry representatives, the use of PFOS in this sector could have dropped further in the last years, as there has been a common change to PFOS free products as H4PFOS, which is only a partly fluorinated octane sulphate.

The use in the photographic industry is about 1,000 kg [DEFRA_2004b] and the amount of PFOS for hydraulic fluids for the aviation industry is about 730 kg [DEFRA_2004b]. Regarding the hydraulic fluids for the aviation industry it has to be mentioned that the needed PFOS concentration can be very low (according to 1907/2006/EG PFOS concentration below 0.1 % do not have to be stated in Material Safety Data Sheet). Thus, it might be that the producer of the hydraulic fluid is not obliged to mention PFOS in the corresponding Material Safety Data Sheet of the product. The costumers might therefore consider that their substance is PFOS free, which is not necessarily true.

PFOS/PFAS based chemicals are used in the semiconductor industry for the fabrication of imaging devices such as digital cameras, cell phones, printers, scanners etc [DEFRA_2004b]. PFOS is used in different fluids in this industry. These fluids are photoresists, top anti-reflecting coating (TARC), bottom anti-reflecting coating (BARC), edge bead removers (EBRs) and developer. The different uses can be differentiated in critical applications (where no PFOS-free alternatives are available) including photoresists, TARC and BARC and non-critical applications (where PFOS-free alternatives are available) including developer and EBRs. It was stated that the Semiconductor Industry Association in Europe (SEMI) eliminated the use of PFOS in non-critical applications. It was further stated that non-critical uses

of PFOS have been phased out in Europe with a timeline of 2007 due to the efforts which the industry companies have invested in and undertaken in the past decade to move away from PFOS [Personal Communication from Industry 10, 2010] and has in the meanwhile significantly decreased. The current total consumption of PFOS used by the European semiconductors in 2009 was considered to be about 9.3 kg/year (5.4 kg via resists, 1.04 kg via BARC and 2.9 kg via TARC). About 3.8% of the PFOS used in resists, 0.5 % used in BARC and 7 % used in TARC end up in waste water. Together about 0.4 kg/y are emitted as trace concentration via waste water [ESIA 2010].

The remainder of the PFOS is either destroyed during the process or becomes part of the production waste which is usually incinerated resulting in PFOS getting destroyed [ESIA-SEMICON_2006]. Figure 6-25 shows the drop of used PFOS in the semiconductor industry.

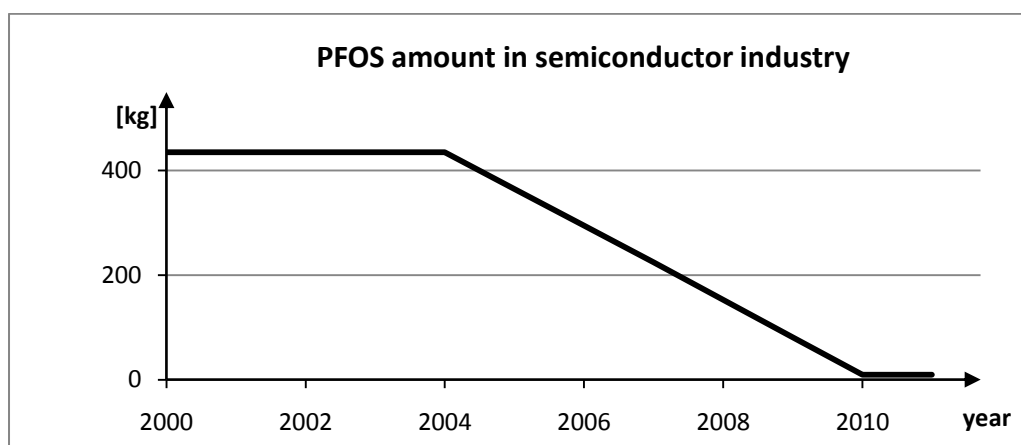


Figure 6-25: Change in the use of PFOS in the semiconductor industry in the EU (annual consumption of PFOS for the use in different applications from 2000 to 2010)

The use of PFOS contained in fire fighting foam is a decreasing sector as all stored foams have to be destroyed by 27 June 2011. It is expected that the remaining stock of approximately 6,377 tons of foam which contain about 64 t PFOS (see section fire fighting foams) will be thermally destroyed by that time.

6.3.5.2 Past uses

There is a long list of different past applications of PFOS. Many of the uses have stopped between 2000 and 2004 due to the voluntary production phase out of 3M. Especially the products with a short life time can be considered as historical uses without an influence on the current PFOS emissions. However, two products could be identified with a lifetime which is long enough to have an impact on the PFOS amount in the waste stream. These two products are upholstery made of leather and carpets made of synthetic fibres.

6.3.5.3 Unintentional production

There is no information on the occurrence of PFOS and its derivatives due to unintentional production.

6.3.6 Selection of Relevant Uses, Applications and Finished Articles for PFOS

6.3.6.1 Current uses

The current uses are nearly all uses, which are allowed according to Regulation 2006/122/EC. This includes the metal plating industry, use in hydraulic fluids in the aviation industry, photographic industry and semiconductor industry. Table 6-16 shows the different used amounts of PFOS in the different industries.

Table 6-16: PFOS amounts currently used in the relevant industry sectors

Industry	PFOS amount
Metal plating industry	6,500 kg/y
Photographic industry	562 kg/y used + ~1,280 kg from historical storage
Semiconductor industry	9.3 kg
Hydraulic fluids in aviation industry	730 kg
Fire fighting foams	~ 90 t in stocks which have to be destroyed by 27 June 2011.

The use of PFOS in fire fighting foams is not considered to be a relevant use nowadays as the active production of PFOS containing fire fighting foams has ceased in Europe at least before 2006 and the remaining stocks have to be destroyed by 27 June 2011.

In case of the photographic industry there has been a continuous reduction of the use of PFOS but due to the existing storages of pictures in households or x-ray pictures in hospitals this industry generates a relevant PFOS containing waste stream from the past.

Considering the gathered information regarding the used amount of PFOS in the semiconductor industry this section is not considered relevant for further investigation.

Another large source of PFOS is sewage sludge. Especially when located at contaminated sites or downstream from industries, which use or have been using PFOS, high levels of PFOS can be found in the sewage sludges. History has proven that improper use of contaminated sewage sludges can lead to local contaminations [NRW_2007c]. This pathway of PFOS is considered as one of the most important secondary sources of PFOS, as more and more industries stop or reduce the use of PFOS, however,

sewage sludge accumulates not only PFOS which is currently used but also PFOS which has been emitted in the past.

The use of PFOS in the semiconductor industry has significantly reduced in the last years. It was estimated by the European Semiconductor Industry Association (ESIA) that the total used amount of PFOS from critical application is about 9.34 kg/y of which about 0.4 kg/y are emitted as trace amount through waste water. This industry is therefore not further investigated.

6.3.6.2 Past uses

Especially industries, which used PFOS only in their processing steps, can be considered as an irrelevant historical use. This regards for example the industry of chemical driven oil production [POPRC_2010]. The increasing negative image of PFOS, the raised awareness and the voluntary phase out of PFOS production by 3M lead to a significant drop of use or phase out in many PFOS consuming industries.

There are industries that used PFOS that ended up in the product [DEFRA_2004b], but were not further investigated. This was particularly due to expected small amounts of PFOS used, short life time and stop of production but also due to the lack of information. The corresponding industries are:

- Medical devices (e.g. ethylene tetrafluoroethylene copolymer (ETFE) layers and radio-opaque ETFE production, in-vitro diagnostic medical devices, and CCD colour filters)
- Insect baits (use e.g. against leave cutter ants)
- Coatings and coating additives (e.g. varnishes, inks)
- Rubber and plastics (e.g. mouldings)

Other industries which applied PFOS in the final product have been investigated further. Their typical stock and life times have been considered to evaluate their potential input in the waste streams. It has been identified, that paper and cardboard (product life time less than 1 year) industry as well as the textile industry (product life time about 4 years) can be considered to be irrelevant. Whereas the continuing use of carpets and leather products especially from the upholstery industry still leads to a PFOS content in the waste stream due their long average life times of 14 and 10 years, respectively.

In case of the textile industry it is difficult to indicate a general life time of the relevant products. It is stated that PFOS in the textile industry has been used in a magnitude of applications as all-weather clothing, umbrellas, bags, sails, tents, parasols, sunshades, footwear, rugs, mats, fast food restaurant wear, industrial apparel, helmets, mechanics' overall clothing and head covers for workers in clean rooms etc [POPRC_2010], [DEFRA_2004b]. Some of the use as work clothes will have a short life time of maybe one year, whereas other applications as tents might have a much longer life time (> 10 years). Especially the professional uses which might have to be collected are considered to have a short lifetime and it is therefore assumed that they are not relevant at present. Products with longer life times are considered to be used by non commercial end users. These products are assumed to end up as part of municipal solid waste (MSW) in non-hazardous landfills or non-hazardous incineration plants sooner or later. However, this fraction is considered to be very small compared to the overall MSW and a separation does not take place. It is stated that about 197 t/y of PFOS have been used for carpet, leather

and textile together [DEFRA_2004b]. Due to the calculations presented at a later stage of this report it is estimated that about 40-45 t/y have been used in the textile industry. Due to the lack of data and the diffuse distribution and final disposal this section is not further investigated.

6.3.7 Metal plating

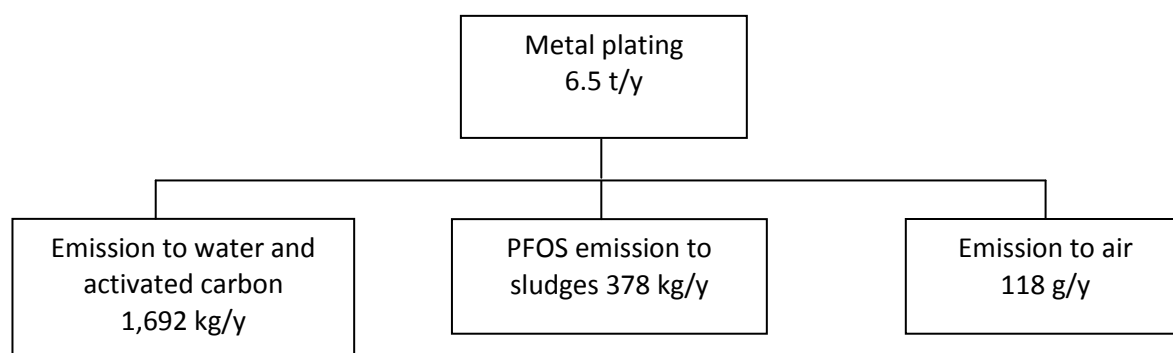
A) Background

PFOS is used as a mist suppressant in different processes of the metal plating industry [Personal Communication from Industry 1, 2010]. PFOS lowers the surface tension of the chromium bath and therefore reduces the size of soaring bubbles in the bath. The surface of the bath is therefore calmer and emissions to air are reduced [OSPAR_2006]. After it became evident that PFOS is a substance of concern, most of the industries switched to other alternatives [Personal Communication from Industry 2, 2010]. However, in some branches of the industry a shift to other chemicals was difficult. Especially, in hard chrome plating, plastic plating as well as decorative chromium plating, which uses Cr (VI) baths, the industry continued to use PFOS to limit Cr (VI) emissions to air. In the last years investigations for alternatives have been ongoing. The industry continuously uses more H₄PFOS as an alternative for PFOS [Hansgrohe_2008], [Personal Communication from Industry 1, 2010].

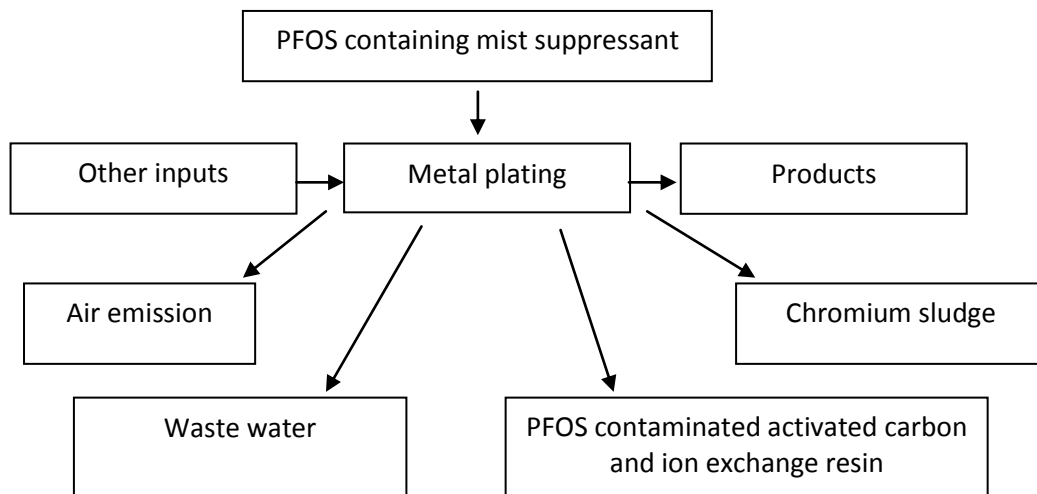
The relevant PFOS derivatives in these industries are the potassium, lithium, diethanolamine, and ammonium salts of perfluorooctane sulfonic acid, as well as quaternary ammonium salts and amines. According to the UK survey the quaternary ammonium salt of PFOSA (CAS No. 56773-42-3) seems to be used most [DEFRA_2004b].

In the metal plating industry, the main emissions of PFOS are found in the wastewater, whereas the amounts of emissions to air are considered to be very low. Sewage sludge is another relevant pathway for PFOS through which it can leave the facility as waste [Hansgrohe_2008]. In many cases activated carbon is used to capture PFOS in the waste water [PFSO_ZVO_2009].

Based on different reports and information from industry the following average flow has been calculated



B) Input/Output



Input:

The sold mist suppressant solutions can be directly poured into the chromium bath, which can exhibit a PFOS concentration of 5-50%. Typically, 10% solutions are used [DEFRA_2004b], [RIVM_2009]. The concentration in the bath is adjusted to be about 2.5 [Personal Communication from Industry 2, 2010] to 10 mg PFOS/l chromium bath (2.5-10 ppm) [UK-EA_2004].

Output:

Products: the chromate articles from this industry are used in aircraft, medical industries, vehicles and general engineering. The products from this industry should be PFOS free due to the rinsing in the metal plating process.

Air: the emission to air from this industry can be considered to be very small as PFOS is used as a mist suppressant and should therefore avoid the production of haze over the chromium bath.

Waste water: according to the BAT BREF document "Surface treatment of metals and plastics" emissions from these industries are primarily to water, and about 300,000 tonnes of hazardous waste is produced per year (an average of 16 tonnes per installation), mainly as sludge from waste water treatment or spent process solutions [BAT-BREF Surface treatment_2006]. Due to the long plating times and high process temperatures (50-60 °C) there is usually sufficient evaporation to return used rinse-water from the subsequent rinsing station. This may enable closed loop operation, with or without an evaporator. Alternatively, there may be high water consumption [BAT-BREF Surface treatment_2006]. According to [Hansgrohe_2008] not all the process water is run in a closed circle and some of the water, which is - among other uses - used to rinse the final product, has to be drained. It is considered that the majority of the waste water within Europe is sent to a proper treatment plant, which is able to hold back

the majority of the PFOS in the wastewater. According to several discussions with the industry, a lot of efforts have been done in the last 3-5 years to reduce the PFOS emissions into water. Waste water which is not treated in the facility will lead to increased PFOS concentration in the sewage sludge of downstream waste water treatment plants. Municipal sewage plants usually only have a limited PFOS reduction potential, since only treatment plants specifically designed for the handling of PFOS, are capable to perform PFOS reduction.

PFOS contaminated **activated carbon** or **ion exchange resins**: as the majority of waste water is treated within the chromium facility, activated carbon or ion exchange resins contaminated with PFOS is generated.

Chromium sludge: during the chromium process chromium sludge is generated which consist among others of different chromates. The chromium sludge is filtered and removed from the chromium bath. Filter press treatment of metal hydroxide sludge at pressures above 15 bar can give a final cake with 15-35 % solids, which can contain high concentration of PFOS [BAT-BREF Surface treatment_2006]. In some cases the chromium sludge might also contain some activated carbon, in case an activated carbon treatment is used, which shall reduce the PFOS content in the waste water.

C) Generation Factors

No information could be found which could support the evaluation of PFOS amount or concentration in different waste streams depending on area of plated surface or energy used. For emissions to air the calculation of the Technical Guidance Document has been performed by a report from the UK [UK-EA_2006]. Air emission for one site in Europe has been calculated to be 79 mg. It was considered that there are about 300 sites in UK and that the UK represents about 20 % of Europe. The total European emission therefore sums up to 118 g/y. It is stated in the report that this value seems to be very small and that it might be that some PFOS is destroyed during the electrochemical process during metal plating. Reports from the industry [Hansgrohe_2008] show that there is a drastical drop of PFOS concentration during the process when electricity is applied. Investigations have been conducted to use this degradation effect as a wastewater treatment possibility. As a consequence there is a difference of PFOS input and output which can be explained with the effect of electrochemical degradation in the metal plating process.

D) Contamination data

The main information of this chapter is based on a recent report from the company Hansgrohe and communication with the corresponding author [Hansgrohe_2008].

Untreated Waste water	The concentration of PFOS in the waste water can vary in a wide range as this depends very much on the installation and process of the facility. Values found were in the range of ~ 0.3 and ~ 1 ppm.
Chromium sludge	The only chromium sludge analyses found has been in the range of 3.1 and 3.8 ppm.

Activated carbon treatment	In some cases activated carbon is added to the wastewater before the filter press. In this process, the activated carbon becomes part of the chromium sludge. Regarding the measurements of Hansgrohe (PFOS concentration, water throughput, activated carbon amount) it was calculated that the PFOS content of the activated carbon in the chromium sludge is about 300 ppm. Due to the combination of the activated carbon from the activated carbon treatment with the chromium sludge, the PFOS concentration of the sludge is increased to about 7 ppm.
Activated carbon filter	Depending on activated carbon type the filter quality for different substances and the loading capacity until the substance breaks through the filter can be very different. Regarding to a report from Hansgrohe a PFOS loading of up to 950 ppm was measured before the activated carbon had to be exchanged.
Ion exchange resin	It was stated by the industry that ion exchange resins have a higher loading capacity than activated carbon and that they can contain up to 19,000 ppm of PFOS.

E) Activity data

Reports from 2000 to 2004 and in some cases even later estimate that the yearly consumption of PFOS in the metal plating industry was about 10 t [DEFRA_2004b]. More recent estimations consider that the amount has dropped to 6.5 t [ZVO_2010b]. The consumption of 6,500 kg PFOS in 2010 results in emissions of 169 kg to water and 0.308 kg to air. In a report from the UK the emissions to air and waste have been calculated [PFOS UK-EA 2004]. From the industry it is expected that about 5 % of The PFOS ends up in waste water, which is about 325 kg/y.

The annual consumption of PFOS in the EU was about 10 t in 2002 (no changes in consumption have been observed between 1999 and 2004, neither volume or market size; [DEFRA_2004b]). It was stated that the use of 10 t represents the use of PFOS in North America (including Mexico) and Europe [Personal Communication from Industry 3, 2010]. Data from Asia are not provided. It is estimated that this amount has dropped to 6.5 t in 2010 [ZVO_2010b]. About 50 % is used in Europe. Within Europe FR, ES, IT and DE consume about 70 % [ZVO_2010b]. According to [CARCAL_2009] about 4,195 kg of PFOS have been used in the metal plating industry in 2008 within nine countries from Europe. Industry representatives considered the use of 6.5 t of PFOS in the metal plating industry very high as it is nowadays state of the art to have shifted to alternative substances as H4PFOS ([Personal Communication from Industry 1, 2010], [Personal Communication from Industry 2, 2010], [Personal Communication from Industry 4, 2010], [Personal Communication from Industry 5, 2010]).

A consumption of 6.5 t PFOS in 2010 is assumed for further analyses in the present report.

In the metal plating industry PFOS is used as a mist suppressant and the concentration of the PFOS in the processing bath has to be regularly adjusted. During the last years the input of PFOS in the industry has continually decreased as illustrated Figure 6-26.

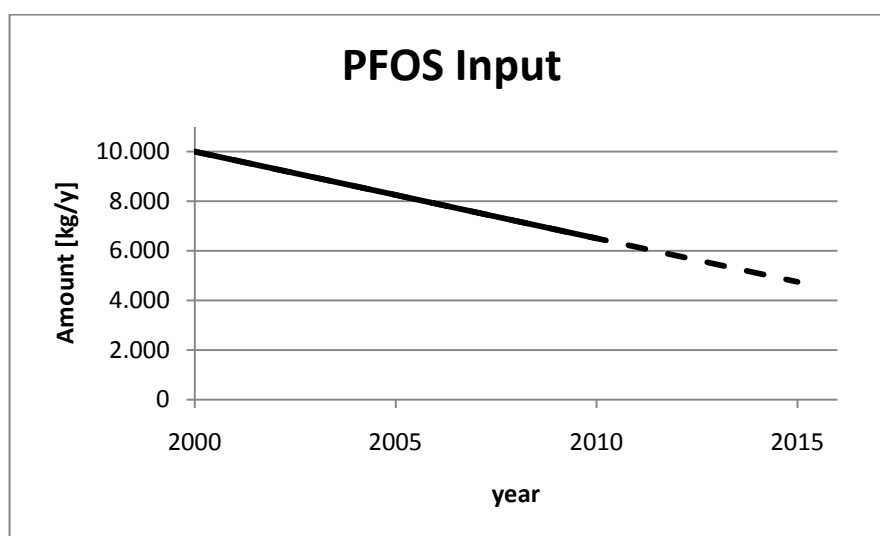


Figure 6-26: Use of PFOS in the metal plating industry in Europe (estimation based on several sources)

F) Waste Treatment

Possibilities to concentrate and treat PFOS in chromium bath sludges are under investigation. No waste stream containing PFOS from the metal plating industry could be detected except sludge from the chromium bath (sometime containing activated carbon), the vacuum distillate or contaminated filters from the waste water treatment. Sludges from waste water treatment plants treating the PFOS contaminated water from the chromium plating facility are dealt with in section 6.3.13. The main information of this chapter is based on a report from the company Hansgrohe and communication with the corresponding author [Hansgrohe_2008].

Chromium sludge:

Drying of the sludge may be required prior to stabilisation. The volume of waste solidified with hydraulic binders' increases in volume 1.5 to 2 fold. Waste treated with hydraulic binders (such as lime) may have a pozzolanic action and develop mechanical properties that increase stability with time. However, the surface active agents present may interfere with this reaction [BAT-BREF Surface treatment_2006].

Chromium sludge is either used for recovery of chromium or stabilised. This is usually carried out off-site by specialist waste management contractors [Personal Communication from Industry 4, 2010].

There is no information on the share of chromium sludge which is stabilised and disposed in hazardous landfills or recovered in construction. For the mass flow it is assumed that 50% are disposed in hazardous landfills and 50% are recovered in metal reclamation in a thermal process.

Treatment of waste water:

According to the report of Hansgrohe [Hansgrohe_2008] PFOS concentration in waste water that is released without any treatment ranges between 0.3 and 1 ppm, which represents about 17 to 49 % of the input. In general, it can be assumed that the majority of the facilities are nowadays equipped with a waste water treatment plant and that maybe 10% of the waste water leaves the facility untreated.

For the other 90 % of waste water treated different PFOS reduction technologies could be identified, which are:

- Vacuum distillation (captures ~88% of PFOS from the waste water stream)
- Activated carbon treatment (captures ~98.8 % of PFOS from the waste water stream)
- Activated carbon filter (captures ~99.1 % of PFOS from the waste water stream)
- Ion exchange resin (captures ~99.1 % of PFOS from the waste water stream)

No information could be found regarding the share of each technology in the industry. Therefore, it is assumed that each of the technologies has an equal share of the treated waste water of 22.5 %. The four water PFOS reduction technologies (each 22.5%) and the untreated waste water (10%) sum up to 100 %.

Vacuum distillation

In case of vacuum distillation a PFOS rich (240 ppm) distillate is generated which is considered as hazardous waste and is reused for metal recovery [Hansgrohe_2008]. It is assumed that the sludge is recovered for metal reclamation in a thermal process.

Activated carbon treatment

Activated carbon from the activated carbon treatment is part of the chromium sludge and has a PFOS content of approximately 300 ppm. The activated carbon from the activated carbon treatment is part of the chromium sludge and therefore reused, either in a metal recovery or a stabilisation process [Personal Communication from Industry 4, 2010]. For the mass flow it is assumed that all the chromium sludge is recovered in metal reclamation in a thermal process.

Activated carbon filter

Activated carbon from the filter is regularly exchanged and incinerated in a hazardous waste incineration plant. It was stated by the industry that the activated carbon can bear a load of less than ~950 ppm, before a PFOS breakthrough can be detected.

Ion exchange resin

Ion exchange resin from the filter is regularly exchanged and incinerated in a hazardous waste incineration plant. It was stated by the industry that the activated carbon can bear a load of 19,000 ppm before a PFOS breakthrough can be detected.

G) Substance Flow

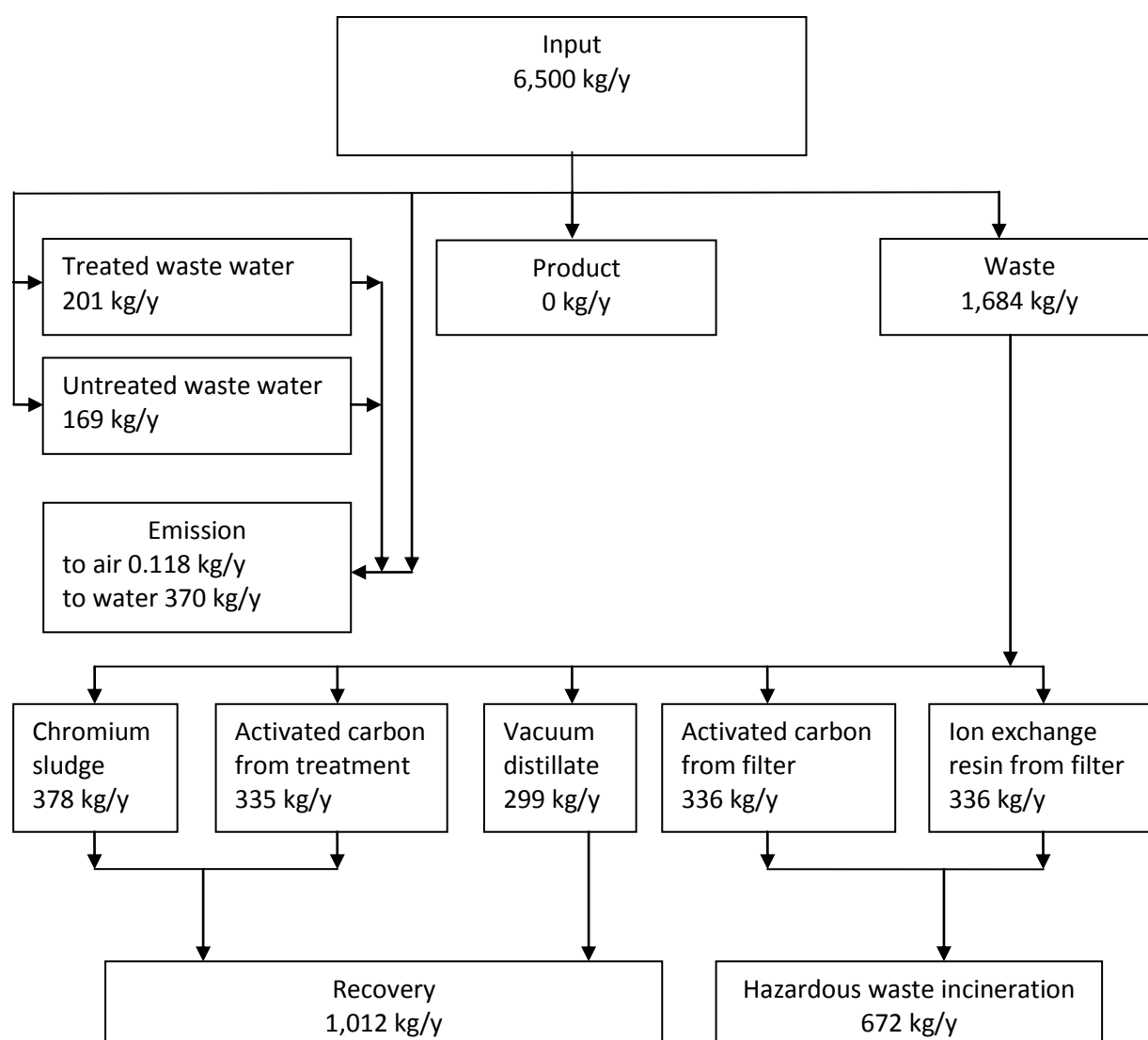


Figure 6-27: Substance flow PFOS from the sector of metal plating

At a European level it is estimated, that the yearly use of PFOS is about 6,500 kg.

The chromium sludge, which is normally removed from the waste water flow by a filter press, includes approximately 378 kg PFOS/y and is reused for metal. In case an activated carbon treatment technology is used to remove PFOS from the waste water, the activated carbon can be found in the chromium sludge, which increases the PFOS amount to about 717 kg/y. There is the possibility to stabilise the chromium sludge and reuse it as construction material or to landfill it. As no numbers are available and the sewage sludge contains a high value for metal recovery it was assumed that all chromium sludge is recovered.

The remainder of PFOS which is not degraded in the electrochemical process follows the waste water stream. This covers about 26 % of the input which represents about 1692 kg/y. As little is known about the waste water treatment, it was estimated that 10 % of the waste water leaves the chromium facilities untreated, representing about 169 kg of PFOS per year as emission to water. 90 % of the waste water is

treated with a vacuum distillate, an activated carbon filter, an activated carbon treatment or an ion exchange resin filter technology. In absence of concrete numbers it was estimated that each of the technology has the same share of use. This implements that through each technology between 298 and 336 kg PFOS/y is extracted from the waste water. The waste water technology is capable to capture about 88 to 99 % of the PFOS.

The waste water output consists of two streams. One is the treated waste water stream, which covers about 90 % (3,347,176,080 kg water with 201 kg PFOS/y) of the water output and the second is the untreated waste water stream covering 10 % (334,717,608 kg water with 169 kg PFOS/y) of the water output. Together the total emissions into water are summed up to 370 kg PFOS emissions per year.

H) Material Flow

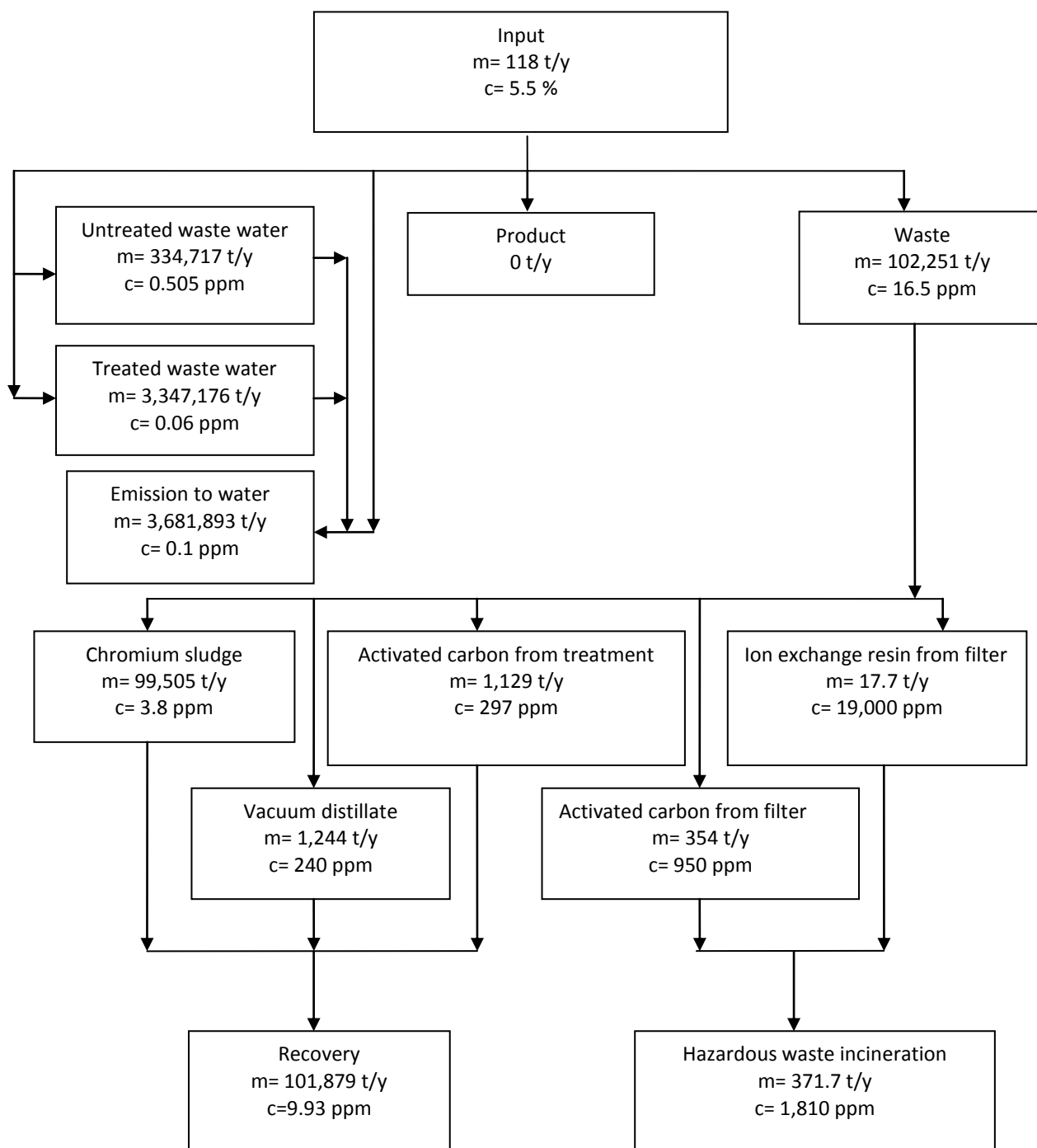


Figure 6-28: PFOS related waste flow related to the metal plating industry

6.3.8 Photo Industry

A) Background

PFOS derivatives are used in the photographic industry as ingredient in photographic material (film, paper, plate). The processed photo area in the old EU member states before 2001 had a total size of about 640 Mio m² per year of which about 78 Mio m² have been in the X ray sector [UBA_2001].

Table 6-17 gives an overview of production amounts for 2000 of relevant products containing PFOS:

Table 6-17: Production amounts of relevant products containing PFOS in 2000 [UBA_2001]

Type of photographic process	Produced area [million m ² /y]	Paper products [million m ² /y]	Film products [million m ² /y]
b/w negative film	1.5		1.5
colour negative film	34		34
colour reversal paper	2	2	
colour positive paper	383	383	
b/w positive paper	10	10	
colour reversal film	2		2
b/w X-ray	78		78
cine and television film	12.7		12.7
b/w reprographic	115		
TOTAL	638.2	395	128.2

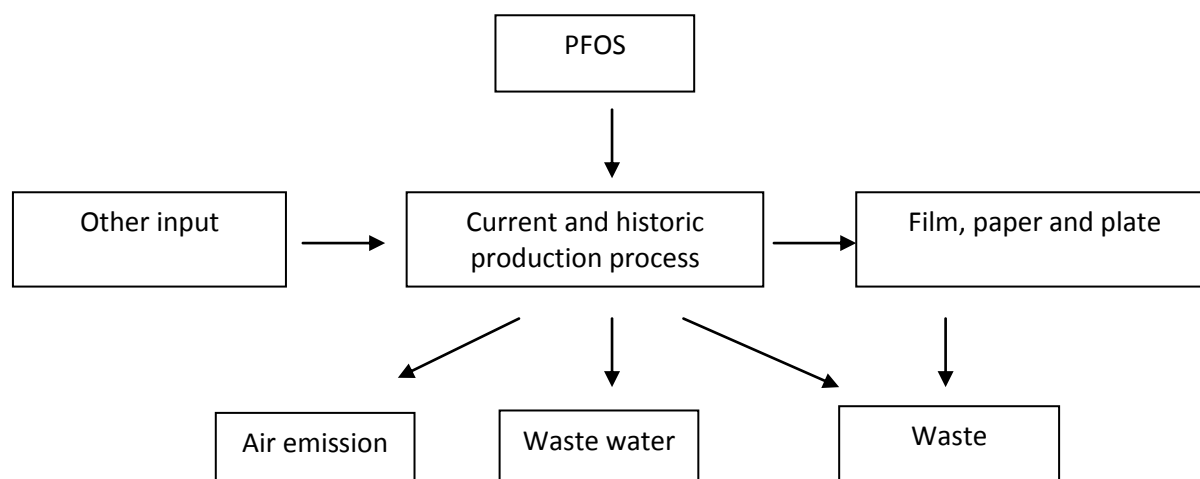
In one report [UK-EA_2004] the European Photographical Industry (EPCI) indicates that the main area of use for PFOS is in the medical area (x-rays), with 85% of EU use in this sector. Six substances which are used in this area were identified to include PFOS salts and PFOS polymers. It is assumed that for the production of films PFOS salts were used followed by use of PFOS-polymers in subsequent steps. The polymers are assumed to contain 1% of PFOS substances as residuals. The EPCI has provided data on the use of PFOS in the EU. The amount of PFOS used in the production of films in the EU was estimated to be 850 kg per year. Additionally, finished articles containing PFOS imported into the EU add a further 150 kg per year. Exports of articles containing PFOS account for 250 kg per year. Hence, the overall amount of PFOS which stayed in the EU amounted to 750 kg in the year 2004. Information from the EPCI indicates a concentration of PFOS in films of 0.1- 0.8 µg/cm². According to the emission scenario document on the photographic industry in the Technical Guidance Document, x-ray films have a double coating of photographic material. As this is the main use area of PFOS according to the information from the EPCI, the higher contamination value of 0.8 µg/cm² was used for the calculations. The surface area of film containing 750 kg PFOS amounts to 9.4x10¹¹ cm². For comparison, the amount of x-ray film used in the EU according to the EPCI amounts to 7.9x10¹¹ cm² [UK-EA_2004] and 7.8x10¹¹ cm² in report [UBA_2001], which is in good agreement to each other.

In the report [DEFRA_2004b] it is also estimated that at least 850 kg of PFOS have been used for the production of film. Less than 50 kg PFOS/y have been used for paper products and <100 kg PFOS/y for plates. It is also stated that between 2000 and 2004 there has been a PFOS reduction of 83% in the processing of film, paper and plates due to the use of alternatives in different processing steps in all three sections. It is therefore assumed that the same relative reduction took place in case of the x-ray

and plate industry. In case of the paper products a stronger decline was suggested due to the ongoing digitalisation.

For the calculation of the PFOS use in the paper industry the 395 million m² and a PFOS use of 0.1 µg/cm² has been considered.

B) Input/Output



Input:

PFOS and its derivatives have been historically used in the form of coating solutions for the production of film, paper and plate, which is still the current method at the moment. The current input is the present use of PFOS in different coating solutions. In general, the photographic industry can be split into paper, film and plate production.

Output:

Products: one output is the stock of historically used PFOS in different film, paper and plate products, which have been produced over 10 years ago and are now either landfilled or reused. To calculate the corresponding stocks, the historical production data have to be considered (see section (E) activity data). A second output consists of the products from current production.

Emission to air: air emissions do not contain relevant quantities of PFOS [DEFRA_2004b] and are therefore not relevant for further consideration.

Emission to water: waste water from production facilities may contain considerable quantities of PFOS.

Waste: waste consists of used films, papers and plates after their typical lifetime. The waste arises in a longer time span after the production of the individual products. Unused coating solutions are recycled and do not enter the waste stream.

C) Generation Factors

Products: in 2000, the use of 5,882 kg PFOS resulted in the production of 78 million m² x-ray film (use 2,891 kg), 50 million m² other film (use 1,861 kg), 395 m² photographic paper (use 50 kg) and photographic plates (use <100 kg). The development of the use figures from 2000 to 2010 is shown in Table 6-19.

Waste water: about 13.6 % of the total used PFOS amount is not contained in the products (film, paper and plate). It is estimated that currently in the western European countries, which use about 95% of all coating solutions [Personal Communication from Industry 6, 2010] the recovery rate is about 80 % and in the eastern European countries, holding about 5 % of all coating solutions [Personal Communication from Industry 6, 2010], the recovery rate is about 20%. Therefore, it is estimated that about 77 % of the waste is recovered whereas about 23 % of the 13.6 % used ends up in waste water (i.e. 3.13 % of the total PFOS amount used).

Air emissions: not considered relevant.

Waste: waste consists particularly of the products that are discarded after their product lifetime.

D) Contamination data

Table 6-18 shows an estimation of the PFOS absolute amounts contained in the relevant products from the photographic industry. Due to changes in the industry sector (see section (E) activity data), the amounts have considerably decreased from 2000 to 2010.

Table 6-18: Overview of PFOS quantities contained in products from the photographic industry

Year	Film			In Paper [kg/y]	In Plate [kg/y]	In Total [kg/y]
	In X-ray film [kg/y]	In Other film [kg/y]	In total film [kg/y]			
2000	624	402	1026	395	86	1507
2001	624	329	953	307	86	1346
2002	624	256	880	219	86	1185
2003	624	183	807	131	86	1024
2004	624	110	734	43	86	864
2005	624	110	734	0	72	806
2006	624	110	734	0	58	792
2007	624	110	734	0	43	777
2008	478	88	566	0	29	595
2009	333	66	399	0	14	413
2010	187	44	231	0	0	231

It is estimated that x-ray films contain 0.8 µg/cm² or 13.45 ppm of PFOS. Average content of PFOS in other films is about 0.1 µg/cm² or 4.3 ppm. It is assumed that photographic paper contains about 0.1 µg/cm². Assuming an average of about 225 g/m², this would lead to an average contamination level of 4.44 ppm. Due to the drastic reduction of films during the last couple of years, the amount of PFOS used has been reduced, accordingly.

E) Activity data

Regarding two reports, there has been a reduction of 83% in the total amount of PFOS used in imaging products between 2000 and 2004 [DEPA_2005], [DEFRA_2004b].

Table 6-19 shows the estimated annual PFOS input in the photographic industry for the time from 2000 to 2010.

Table 6-19: PFOS used in the different sections of the photographic industry between 2000 and 2010.

Year	Film			Use Paper [kg/y]	Use Plate [kg/y]	Use Total [kg/y]
	Use X-ray film [kg/y]	Use Other film [kg/y]	Use total film [kg/y]			
2000	2891	1861	4752	730	400	5882
2001	2349	1428	3777	560	325	4662
2002	1807	994	2801	390	250	3441
2003	1265	561	1826	220	175	2221
2004	723	128	850	50	100	1000
2005	723	128	850	0	83	933
2006	723	128	850	0	67	917
2007	723	128	850	0	50	900
2008	554	102	656	0	33	689
2009	385	76	462	0	17	478
2010	217	51	268	0	0	268

For the calculation of the above table the following data has been considered:

Calculation for 2004: in 2001 the amount of x-ray picture has been indicated to amount to 78 million m² [UBA_2001]. This number can be used for the calculation for the year 2004 because the digital technology for x-ray pictures did not start before 2008. The concentration of PFOS on x-ray films is considered to be about 0.8 µg/cm² [UK-EA_2004]. This would mean that 624 kg of PFOS are on the film surface. According to [PFOS-UK-EA_2004] the x-ray films cover a PFOS share in the film section of 85 %. The total amount of PFOS on other film surfaces is therefore 110 kg/y. This sums up to 734 kg/y. This fits quite well with the estimation of [DEFRA_2004b] that about 850 kg/y of PFOS related substances have been used to produce films while the amount in film used in the EU is 750 kg/y. The difference of 850 kg/y and 734 kg/y can be explained as the difference between PFOS used for the production and PFOS left on the surface of the product. With this ratio the needed amount of PFOS for x-ray films and other films in 2004 has been calculated and is 723 kg/y and 128 kg/y, respectively.

It is further estimated that the amount of PFOS used in the paper and plate industry has been about 50 and 100 kg/y, respectively [DEFRA_2004b], which means that 43 kg and 86 kg/y were applied on the surface of the materials.

Calculation for 2000: for the calculation of the PFOS consumption in the photographic industry it has been stated in [DEFRA_2004b] that between 2000 and 2004 there has been a decrease of 83 % on the PFOS use. Therefore, the total PFOS consumption in 2000 has been 5,882 kg/y. This is due to two

effects. One effect has been a general elimination of PFOS in several applications²³. The second reason for the drop of used PFOS is the increase of digital pictures at that time. As no more detailed information is given it is estimated that the decrease of PFOS use due to the elimination of PFOS substances has been similar in all three areas, film, paper and plates and that in addition to this other films than x-ray films and the paper section had an additional drop of used PFOS due to the increase of digital pictures.

As shown in Table 6-19 the yearly production of films has been about 127 million m²/y, whereas the x-ray pictures had a share of 61 %. Until 2004 the x-ray share increased to 85 % and it is considered that the total amount was 78 million m²/y. Therefore, the other film section has dropped from about 50 million m²/y to 13.8 million m²/y, which represents a reduction of about 73 %. This also means that the film production was higher by 365% in 2000 than in 2004. To calculate the relative general reduction of all three sections, it is assumed that a reduction due to digital pictures did not take place. Then the yearly use of PFOS in 2004 would have been for other films 447 kg/y (365 % of 128 kg/y) and for the paper section 182 kg/y (365% of 50 kg/y). In this case the total yearly amount would have been 1,470 kg/y instead of 1,000 kg/y. Considering the used amount of 5,882 kg in 2000, the reduction would have been 75 %. This means that due to the elimination of PFOS in the production, the used PFOS amount was reduced by 75 %. By the additional decrease of 73% of the production of other films and paper (which represent a 100% PFOS reduction), due to the increase of digital pictures, a total reduction of 83 % between 2000 and 2004 is reached.

Calculation for 2010: regarding a conversation with the industry [Personal Communication from Industry 6, 2010] it was stated that since 2008 the quality of digital pictures is sufficient for x-ray and other pictures and that there has been a drop of coating solutions in this section of about 70%. Therefore we can assume that the amount of PFOS used in the total film application reduces accordingly.

On the other hand, there has been an increase in market share in the section of plate and paper. According to a personal contact [I&P 2010], the three industries have replaced PFOS. In paper and plate application almost completely, due to the fact that the usage of PFOS is less critical in these two areas.

²³ PFOS derivatives have e.g. been eliminated in defoamers used in the production of processing chemical for films, papers and printing plates, photoacid generators in photolithographic processing, solutions used in the manufacturing of printing plates, processing of films and papers, use as surfactants in photolithographic processing solutions used in the manufacture of printing plates [DEFRA_2004b].

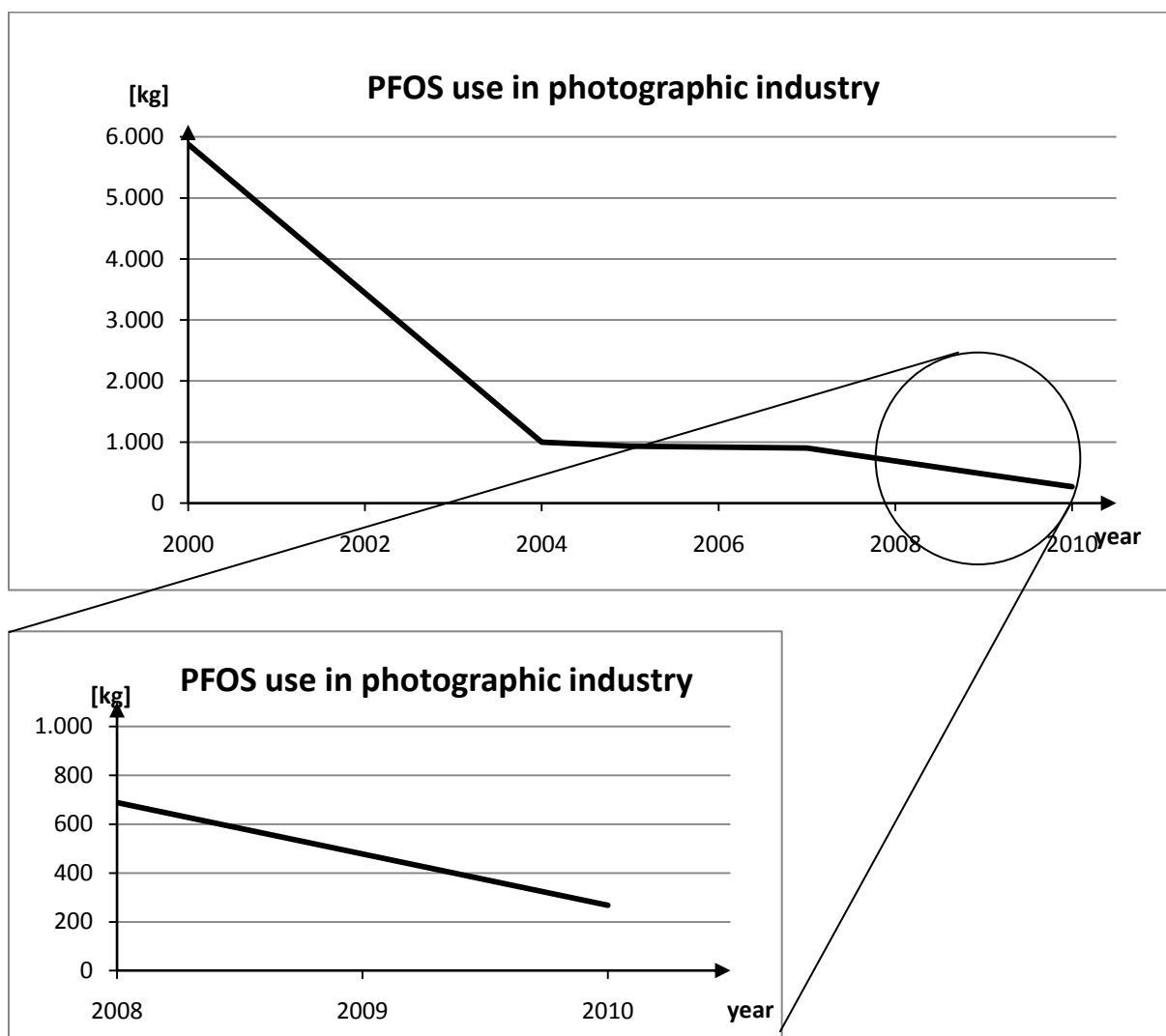


Figure 6-29: Illustration of the estimated use of PFOS in the photographic industry from 2000 to 2010

F) Waste Treatment

Products from current production:

It is estimated that x-ray pictures which are kept at hospitals or similar institutions (considered to be about 50 % of all x-ray pictures) will be treated by recycling companies after 10 years. Therefore, in 2010 the x-ray pictures made in 2000 are given to the recycling companies. For the recycling of films the surface of the films is cleaned and the carrier material which consists of PET is recovered. The collected chemical from the surface of the film contains, among other, PFOS and silver. There are different recycling possibilities to recover the silver. PFOS either ends up in a thermal destruction or is emitted to waste water. The remaining waste from the recycling process which includes the PFOS is incinerated.

It is estimated that the majority of the home made pictures (film and paper) becomes part of the municipal solid waste (MSW) stream and is therefore landfilled or incinerated. It is also assumed, that X-ray films which are not kept at hospitals end up in the MSW stream.

In 2010 a negligible amount of PFOS has been applied on the surface of aluminium plates from the 'computer to plate' (CTP) process and is therefore not further investigated.

Products from historical production (< 2000): about 50 % of the produced x-ray films will end up as MSW. With a density of about 1.4 g/cm³, a thickness of about 0.5 mm and a total area of 78 million m² (production of 2000 which becomes waste in 2010), this sums up to about 23,205 t of x-ray film waste with a PFOS contamination of 13.45 ppm, i.e. 312 kg PFOS. From this part, 50 % of this will end up in incineration plants and 50 % will be landfilled Europe wide.

About 50 % of the produced x-ray films will end up in recycling plants. This corresponds to 23,205 t of PET and 312 kg PFOS. There are two main processes with a similar share in the industry. One process is to dissolve the x-ray surface and desilver the received liquid electrochemically. No waste water is generated and the liquid is forwarded to cement plants. Due to the high nitrogen content in the liquid it can be used to reduce the NO_x emissions of the cement plant. For each 3,000 t of recycled x-ray films about 2 million litres are sold to the cement plant. Considering that in this process about 11,602.5 t/y of x-ray pictures are treated, about 7,735,000 litres are sold to the cement plants. Following the estimation that the density accounts to about 1 g/cm³ and contains about 156 kg PFOS, then the PFOS content is about 20ppm [Personal Communication from Industry 8, 2010]. It is considered that the PFOS content is destroyed in this process.

In the second process, the surface is removed from the x-ray surface and sludge is generated from centrifugation. It is not known how much of the PFOS will be separated with the sludge (assumption: 75% is contained in the sludge).

For the calculation of the PFOS content in the sludge the following assumption have been made:

Thickness of the surface layer of the film:	45 µg [DEFRA_2004b]
Density of the surface layer:	1.4 g/cm ³ [DEFRA_2004b]
PFOS content of the surface layer	0.8 µg/cm ² [DEFRA_2004b]

This results in a surface layer weight of 6,255 µg/cm² and a PFOS content of 12.79 %. It was estimated that 75 % of the 156 kg of PFOS will end up in the sludge, which is about 117 kg. Considering that the sludge consists purely of the surface layer, the sludge amount would therefore be about 914.8 kg/y. The remainder (25%) ends up in waste water.

The final 39 kg will end up in the waste water. Before a water treatment was installed at one location, the wastewater had a PFOS content of 150,000 ng/l. After the installation of a wastewater treatment, the PFOS concentration was below the limit of detection of 100 ng/l. The waste water amount is

260,000,000 kg, containing 39 kg of PFOS. Europe wide it is assumed that about 50 % of the recycling facilities are equipped with a waste water treatment plant. This would mean that only 130,000,000 kg of waste water with a PFOS content of 150,000 ng/l are emitted and that about 19,500 kg of activated carbon with a PFOS content of 1,000 ppm are incinerated from this process every year [Personal Communication from Industry 9, 2010]. The photos from private use which are part of the household waste are either disposed of on a landfill site or in case sieves are used, in a mechanical biological treatment (MBT) plant. In the MBT plant the majority will end in the coarse fraction and will be incinerated (Europe wide it can be estimated that the ratio is about 1:1). The produced amount of paper pictures of 395 million m² in 2000 is estimated to become waste in 2010. With a PFOS concentration of 0.1 µg/cm² this represents about 359 kg of PFOS. The photographic paper is estimated to weight about 225 g/m². Then the PFOS content is 4.44 ppm and the total weight is about 88,894,750 tons.

G) PFOS flow

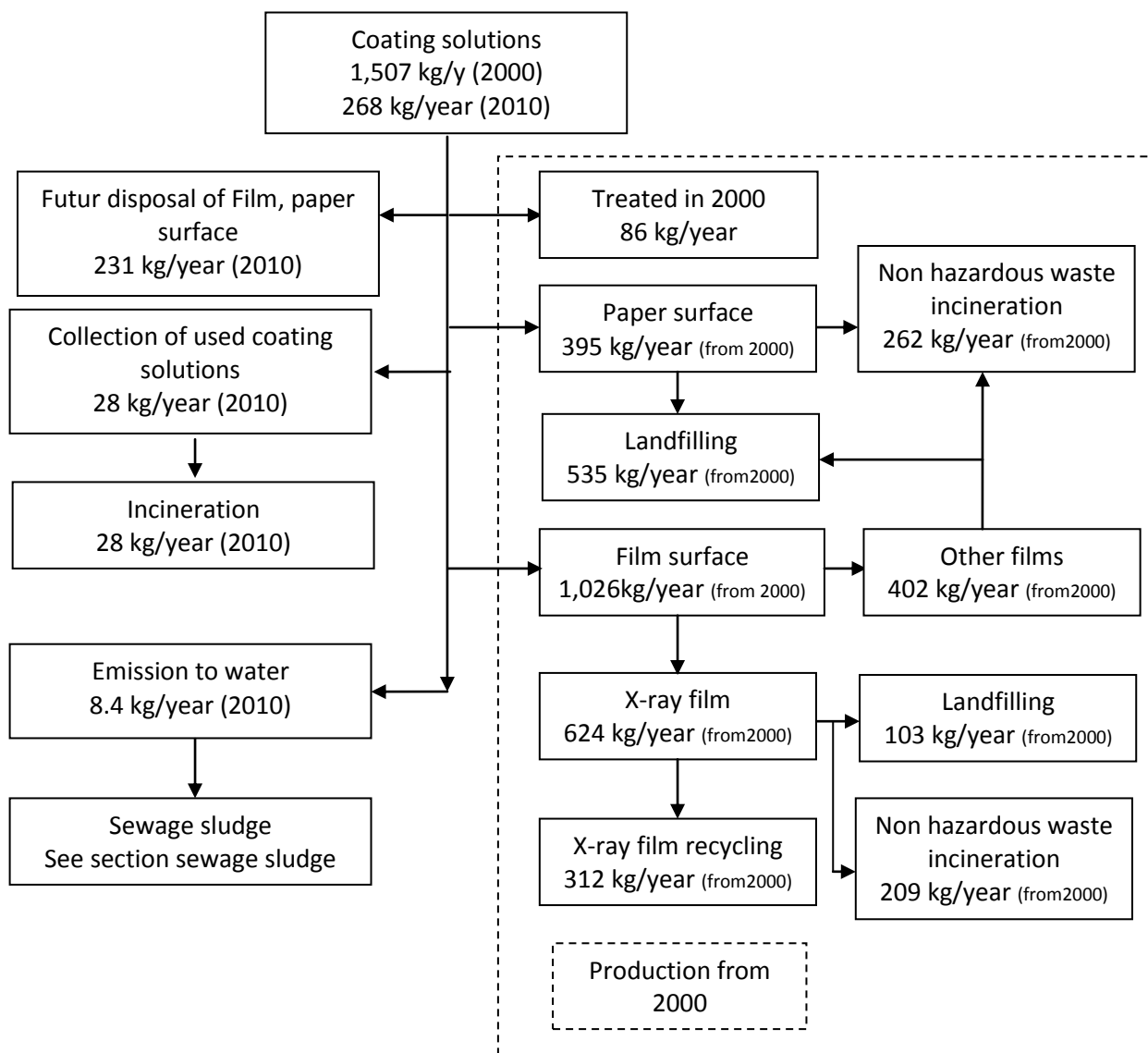


Figure 6-30: Illustration of the PFOS flows from the photographic industry

The left side of Figure 6-30 represents the PFOS path of 2010. From the used 268 kg (217 kg x-ray films + 51 kg other films), 231 kg end up on the surface of film and paper and will be disposed of in the future. About 36 kg of the coating solutions are not contained in products and thereof about 77 % or 28 kg are given to recycling companies, where they will end up in incineration plants. A PFOS amount of 8.4 kg are emitted to water of which the majority will end up in the sewage sludge.

The dotted lined box on the right side in Figure 6-30 represents the PFOS amount which has been applied onto film paper and plate surfaces during production in 2000. An amount of 86 kg PFOS applied onto plates have been already treated in 2000. The remaining 1,421 kg PFOS are on the surface of film and paper until 2010. In case of the paper section, PFOS ends up in MSW and is disposed of or incinerated to 60 and 40 %, respectively. An amount of 1,026 kg PFOS can be found on the surface of photographic films, whereas 624 kg is applied on the surface of x-ray films and 402 kg on the surface of other films. The other films have the same fate as the photographic paper (MSW), while the x-ray films streams split to storage in households and storage in hospitals in an equal amount. The household fraction becomes MSW and ends up as the photographic paper and the other films. 312 kg of PFOS which are on the surface of x-ray films and stored in hospitals will be subject to recovery.

The following diagram shows the resulting PFOS substance flow. Flows in cursive letters are only relevant for the year 2000 and have in this industry no influence for the year 2010.

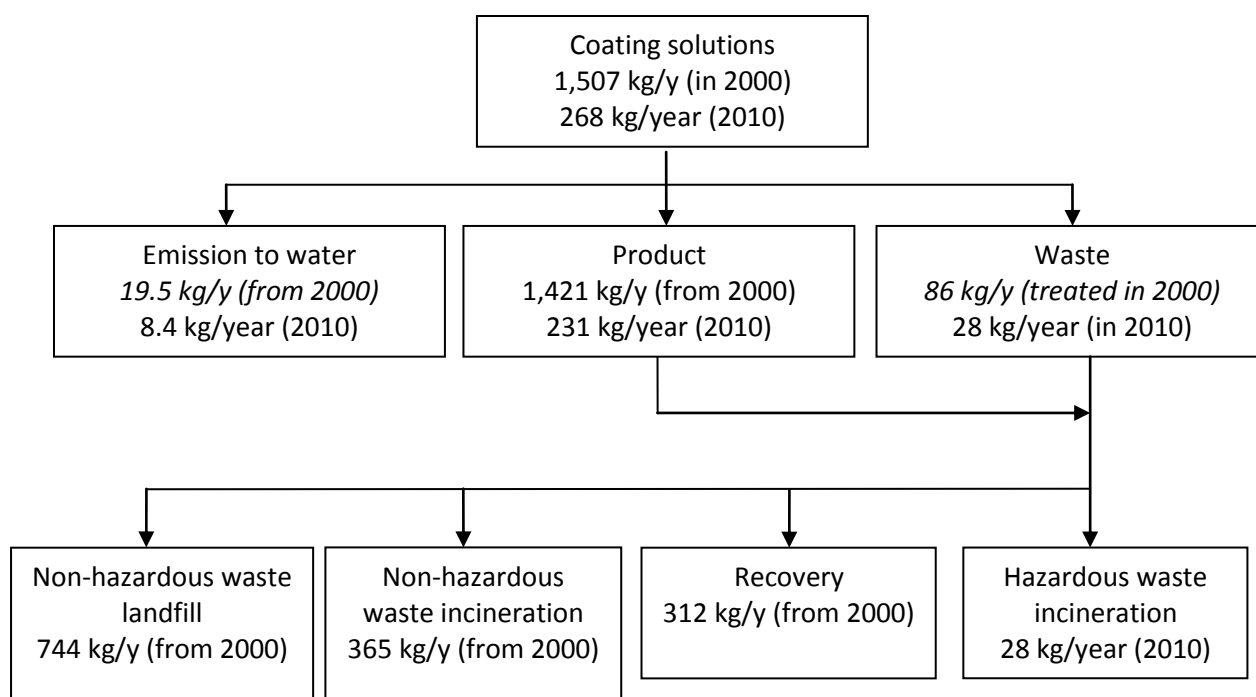


Figure 6-31: Substance flow PFOS from the photographic industry

H) Waste flow

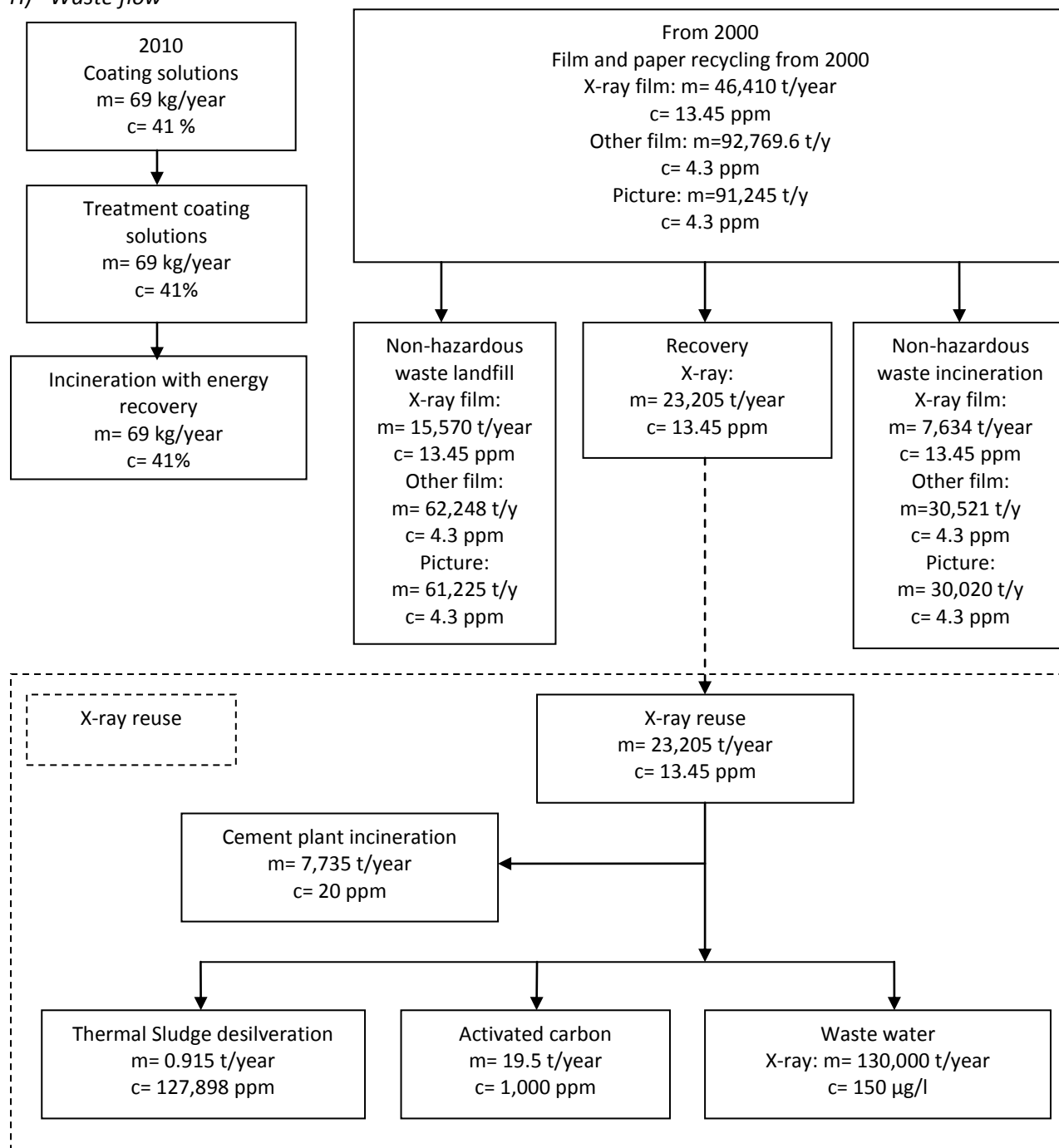


Figure 6-32: PFOS related waste flow from the photographic industry

Figure 6-32 has two relevant input flows. One is from the current production in 2010 and the other is the stock of PFOS containing materials from 2000 which became waste in 2010.

From the 268 kg PFOS which are used in coating solutions in the photographic industry in 2010, 36 kg (28 kg of collected surface solution and 8.4 kg emissions) do not end up on the surface of film and paper. From this 36 kg unused PFOS from the production of 2010, 28 kg are considered as waste. This material is handled by waste management companies. It is used in incineration processes (NO_x reduction) in cement plants.

The second waste stream is from the storage of film and paper from 2000. PFOS on the surface of paper, 50 % of x-ray films and other films end up in non hazardous landfills or non hazardous waste incineration to 60 and 40%, respectively. The remaining 50 % of x-ray films are reused. The reuse processes of x-ray films are explained in more detail in section waste treatment.

6.3.9 Hydraulic fluids in aviation industry

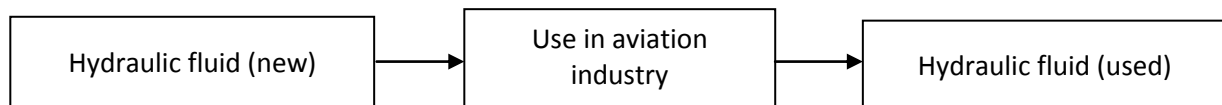
A) Background

In the aviation industry hydraulic fluids are necessary to transfer the break pressure to the breaking system of the tyres. PFOS is added to inhibit erosion (and to control damages) of mechanical parts of hydraulic systems such as servo valves that are used in aircrafts. The lower corrosion effect appears by altering the electrical potential at the metal surface and prevents its electrochemical oxidation [DEFRA_2004b] [OSPAR_2006] [DEPA_2005]. The PFOS containing hydraulic oils are used in all commercially available aircrafts, many military and general aviation aircrafts' throughout the world [DEFRA_2004b].

In up to date reports it is indicated that there are no PFOS free alternatives for the use in hydraulic fluids [POPRC_2010]. However, based on a literature search, one company could be found who claims to have a PFOS free product [Personal Communication from Industry 11, 2010].

PFOS derivatives used in hydraulic fluids are the potassium perfluoro-ethylcyclohexyl-sulfonates [DEFRA_2004b]. The global market for this substance in aircraft hydraulic fluids is about 2.2 tons per year [DEPA_2005]. It can be assumed that one third (0.73 tons) is used in the EU each year.

B) Input/Output



Input: hydraulic fluid (new)

Output: hydraulic fluid (used) connected with releases to the environment during service life

C) Generation Factors

The generation factor from hydraulic fluids is 1. All the used hydraulic fluid is either emitted or ends up as waste without a relevant change in concentration or mass.

D) Contamination data

The provided concentrations of PFOS in hydraulic fluids range from 0.05% to 0.1 % [DEFRA_2004a; PFOS_DEFRA_2004b; PFOS_POPRC_2010; PFOS_SC_2009]. For the calculation in this report a concentration of 0.1 % has been used.

E) Activity data

From the worldwide use of 2.2 tons of PFOS used for hydraulic fluids in the aviation industry it is expected that about 730 kg are used in Europe. Since the total global market for fluorinated compounds in aircraft hydraulic fluids is about 2.2 tons per year, it can be assumed that one third (0.73 tons) is applied in the EU each year for this purpose [DEPA_2005]. It is not clear if this consumption refers to the EU 15 or EU 25. In case the consumption refers to the EU 15 the amount should be increased by about 20 % for the EU 27. If the amount refers to the EU 25 about the same amount can be assumed for the EU 27. For this report it is expected that the consumption of 730 kg PFOS per year covers the EU 25 (the report [DEPA_2005] was finished three months after the EU enlargement) and is therefore also reasonable for the EU 27 consumption.

F) Waste Treatment

It is estimated that 2 % of the hydraulic fluids are emitted to the environment during their service life, whereas 1.4 % are emitted to soil and 0.6 % to water [DEFRA_2004b]. The remaining 98 % of the hydraulic fluids are substituted by new hydraulic fluids and are handled by specialised waste management companies. There are two possible treatment options. One option is a physical chemical treatment to generate a new product and the other option is incineration. Two oil recycling companies [Personal Communication from Industry 12, 2010] [Personal Communication from Industry 10, 2013] have been contacted, who stated that they either do not accept halogenated oils or, in case the national limit values are not exceeded, they incinerate the oils in their own energy recovery incineration facility. Furthermore, two airline companies have been contacted (Air Berlin, Lufthansa). Both companies indicated that the hydraulic fluids were subject to incineration only. The amount of recycled hydraulic fluids is considered negligible and therefore not further considered.

Therefore, it has been assumed that all the collected hydraulic fluids are incinerated in hazardous waste incineration facilities.

G) Substance Flow

For the development of the substance waste flow a PFOS concentration of 0.1 % has been used.

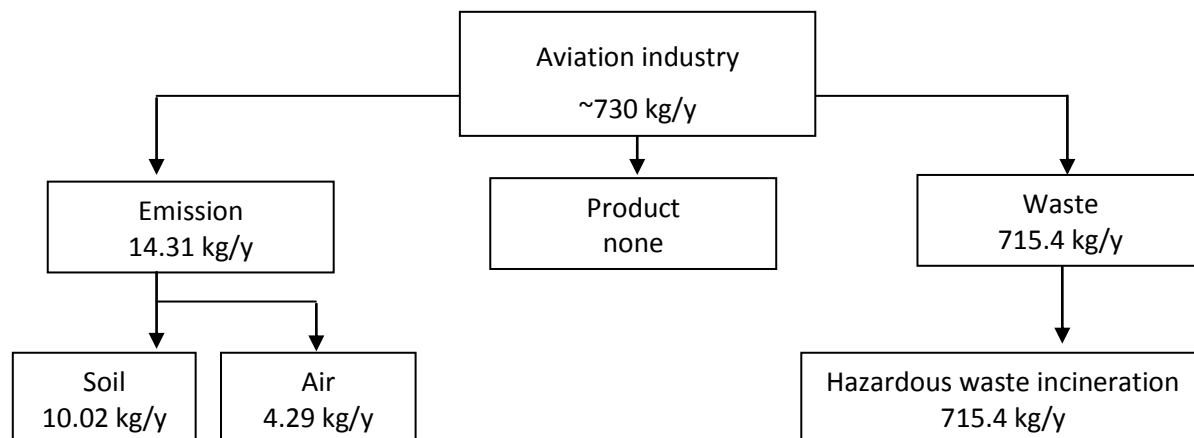


Figure 6-33: PFOS substance flow related to use in hydraulic fluid in the aviation industry

H) Material Flow

Figure 6-34 represents the material flow of PFOS containing hydraulic fluids in the aviation industry

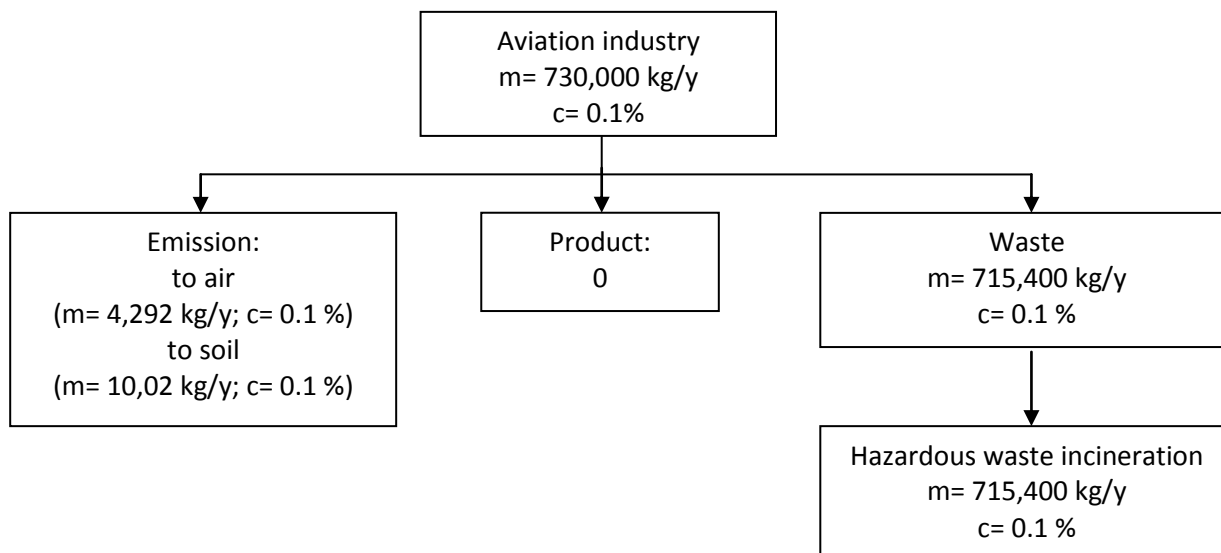


Figure 6-34: Waste flow for hydraulic fluids for use in the aviation industry in EU 27

6.3.10 Fire fighting foams

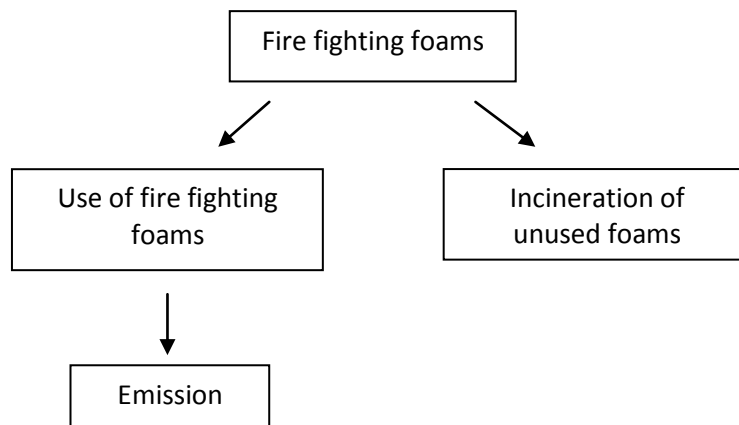
A) Background

PFOS containing fire fighting foams are used for fires from flammable liquid. The low density foam forms a blanket which extinguishes the fire upon application. Fire fighting foams generally consist of a mixture of foam concentrate and water, which forms foam when it gets in contact with air. There exist different types of foams classified as follows:

- FP foams (fluoroprotein foams) used for hydrocarbon storage tank protection and marine applications
- AFFF (aqueous film forming foams) used for aviation, marine and shallow spill fires
- FFFP foams (film forming fluoroprotein foams) used for aviation and shallow spill fires
- AR-AFFF (alcohol resistant aqueous film forming foams), which are multi-purpose foams
- AR-FFFP (alcohol resistant film forming fluoroprotein foams), which also are multi-purpose foams

According to the UK risk reduction strategy on PFOS, any of the above listed types of fire-fighting foams may contain PFOS or surfactants. However, based on a UK survey, only the AFFF and the AR-AFFF foams manufactured before 2002 did contain PFOS at that time [DEFRA_2004b]. The following flow chart gives an overview of PFOS use quantity in fire fighting foams in the EU as well as its fate.

B) Input/Output



Input:

Existing stocks of fire fighting foam

Output:

Fire fighting foam which is not used anymore and not temporarily stored; or which is released from use for fire fighting or training.

C) Generation Factors

The generation factor of PFOS containing fire fighting foams is 1. The foam is either used during an emergency or training, which most likely results in emission to the environment, or it is treated as waste due to exceedance of the expiration date and is subject to incineration. Remaining stocks of fire fighting foams have to be destroyed by 27 June 2011.

D) Contamination data

The concentration of fire fighting foams is reported to lie between 0.5-1.5 % [DEFRA_2004b], but also concentration values of 6, 7 or even 10 % can be found [SwissConfederation_2009; PFOS_EDG_2010; PFOS_MSDS_Fumetroltm 140; PFSO_CARCAL_2009]. For the calculation of the mass flows a value of 1% is applied.

E) Activity data

In the UK it was estimated for 2004 that 76,187 litres of PFOS based foams were held by fire authorities and 2,367,233 litres by emergency stores at industrial complexes making a total of 2,443,420 litres. For the calculation it was assumed that the relative density is 1 and that the PFOS content is 1 %. It was further estimated that an average of 15 % is annually used [DEFRA_2004b]. The report states that storage of UK represents about 20 % of the European market (EU25).

For the selection of the time frame the legal situation stipulated in 1907/2005/EC has been considered. The directive stipulates that only fire fighting foams which have been placed on the market before 27 June 2006 can be used until 27 June 2011. Consequently, it has been assumed that until 2006 the amount of PFOS containing fire fighting foams stored remained the same as in 2004. PFOS containing foams which have been bought after 27 June 2006 had to be substituted by new fire fighting foams latest on 27 June 2008. This results in the same amount as considering a complete purchase stop in 2006. There might have been a purchase reduction of PFOS containing foam before 2006 but as the Directive was published in December 2006 no urgent need was given.

From 2006 on an annually usage of 15 % is considered until 2010. In 2011 all the remaining PFOS containing foam has to be removed. Table 6-20 and Figure 6-35 indicate estimated quantities of fire fighting foam stocks and uses assumed for 2004 to 2012 within the EU.

Table 6-20: Estimated quantities of PFOS containing fire fighting foams stored in EU and estimated use quantities from 2004 to 2012

Year	Stored PFOS containing foam in UK [t]	Stored PFOS containing foam in the EU 27 [t]	Stored PFOS amount in foams in the EU 27 [t]	Annual use of PFOS in foams in the EU 27 [t/y]
2004	2,443	12,217	122	18.3
2005	2,443	12,217	122	18.3
2006	2,443	12,217	122	18.3
2007	2,077	10,385	104	15.6
2008	1,765	8,827	88	13.2
2009	1,501	7,503	75	11.3
2010	1,275	6,377	64	9.6
2011	1,084	5,421	54	8.1
2012	0	0	0	0.0

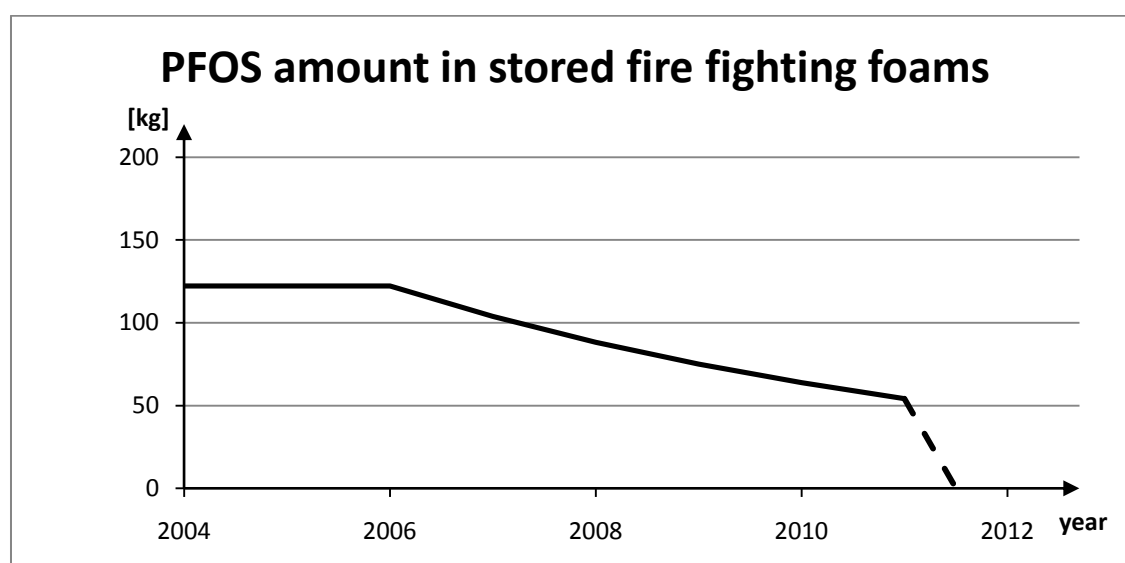


Figure 6-35: Time trend of reduction of PFOS quantities stored in fire fighting foams from 2004 until 2012 in the EU

In different reports it is stated that the yearly consumption of 15 % is only true for the average use of all fire fighting foams and that for PFOS containing foams a yearly use only accounts to 0.5 % [DEFRA_2004b], [UK-EA_2004]. In case a yearly consumption of 0.5 % is used, the exchange of expired foam has to be considered, which is about 1/15 (6.67 %) of the stock due to the lifetime of fire fighting foams of 15 years. Together the reductions sum up to about 7.17 %. Using this lower consumption the following result is received.

Table 6-21: Corrected estimated quantities of PFOS containing fire fighting foams stored in EU and estimated use quantities from 2004 to 2012

Year	Stored PFOS containing foam in UK [t]	Stored PFOS containing foam in the EU [t]	Stored PFOS amount in foams in the EU [t]	Annual removal of PFOS in foams in the EU [t/y]
2004	2,443	12,217	122	8.76
2005	2,443	12,217	122	8.76
2006	2,443	12,217	122	8.76
2007	2,268	11,342	113	8.13
2008	2,106	10,529	105	7.55
2009	1,955	9,774	98	7.00
2010	1,815	9,074	91	6.50
2011	1,685	8,423	84	6.04
2012	0	0	0	0.00

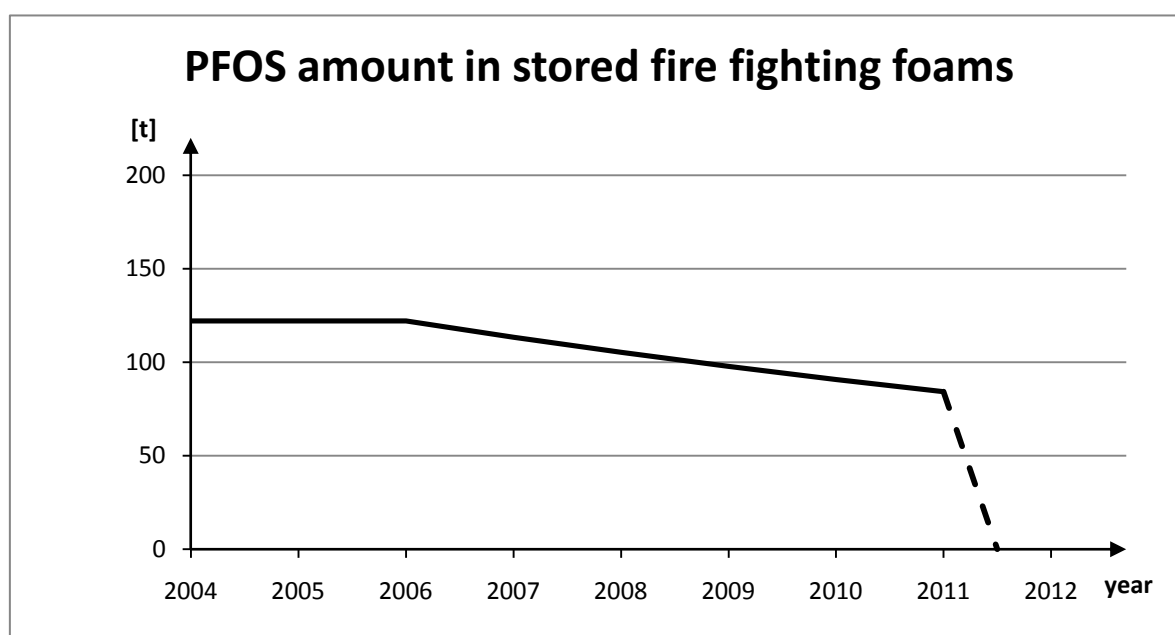


Figure 6-36: Corrected time trend of reduction of PFOS quantities stored in fire fighting foams from 2004 until 2012 in the EU

It can therefore be estimated that between 54 t and 87 t of PFOS contained in stored fire fighting foams have to be destroyed by 27 June 2011.

According to [CARACAL_2008] about 16.4 t PFOS is stored in fire fighting foams in 9 EU Member States. The details are described in the following table.

Table 6-22: Overview of fire fighting foam quantities, related PFOS concentrations and resulting PFOS amount in some Member States [CARACAL_2008]

Member state	Foam amount [t]	PFOS concentration [%]	PFOS amount [t]
BE	663	0.5-6	~4.5
DE	916	0.75	6.9
ES			2.0
FR	54.9	0.5-1	~0.4
HU	40.2	3-6	~1.8
NL	18,540		
AT	53.3	1.5	0.8
PL	30		
FI	1,224		
TOTAL	20,299	2.15	16.4

In Table 6-22 PFOS amounts in specific Member States were calculated from the information available [CARACAL_2008]. Netherlands reported to have stored 18,549 m³ of fire fighting foams. This would represent about 185 t of PFOS, which is nine times more than for the rest of Europe. According to [DEFRA_2004b] the market of PFOS in fire fighting foams in the Netherlands has been between 1.13 and 3.81 t per year in 2002.

In the following the calculation of the Netherlands is explained as described in the RIVM report [RIVM_2009]. The calculation is based on sales volumes of one producer in the Netherlands, which are listed in Table 6-23.

Table 6-23: Sales volume of one fire fighting foam producer in the Netherlands [RIVM_2009]

Year	Sales Volume [m ³]
1997	246
1998	212
1999	295
2000	212

It is also indicated that from 2001 on the sales volumes decreased due to legal requirements. From the life time of 15 to 20 years. For the calculation 20 years are used. A lognormal distribution has been used for the sales, of which the 95th percentile has been taken and used as the average yearly sales amount for the last 20 years. The market share of the known producer has been provided to be between 25–50 %. From this range a 25 % share has been used for the calculation of the total amount of PFOS containing fire fighting foams. The resulting 24,720 m³ have been reduced by 25 % due to usage. A final amount of 18,540 m³ is received which includes ~185 t PFOS. The calculation in [RIVM_2009] represents a worst case scenario which seems to overestimate the current amount.

In case mean values are used and a linear sales volume between 2001 and 2006 are considered (see Table 6-24), a total volume of 5,062 t of fire fighting foams, i.e. 51 t PFOS, results.

Table 6-24: Estimated sales volume between 1991 and 2006 using mean values.

Year	Sales volume [m³]	Year	Sales volume [m³]
1991	241	1999	295
1992	241	2000	212
1993	241	2001	201
1994	241	2002	161
1995	241	2003	121
1996	241	2004	80
1997	124	2005	40
1998	212	2006	0

In Figure 6-37 the estimated sales volumes of PFOS contaminated foam in the Netherlands from one producer is shown.

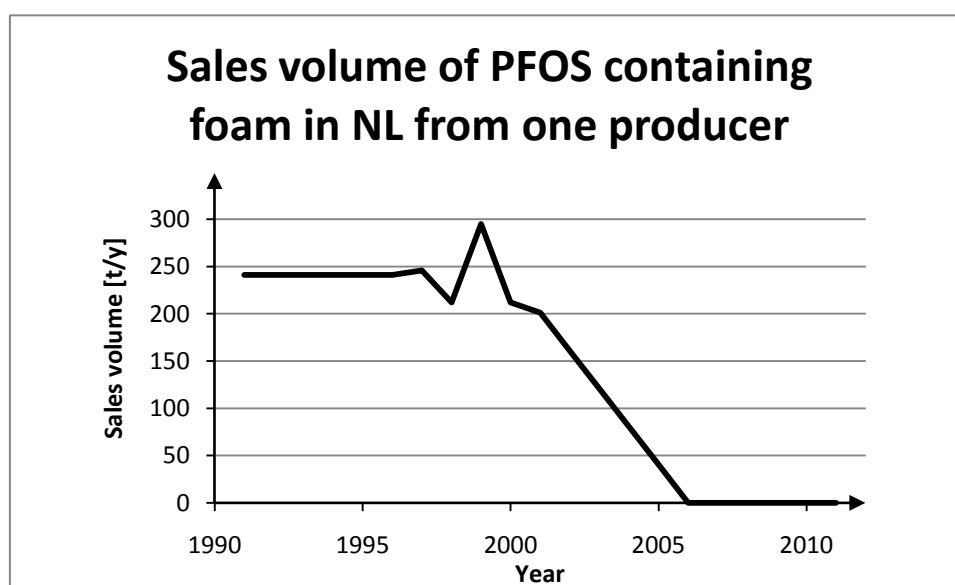


Figure 6-37: Sales volume of PFOS containing fire fighting foams in the Netherlands from one company

Table 6-25 summarises the different used values and the corresponding results.

Table 6-25: Comparison of worst case scenario mean value and best case.

Parameter	Range	Worst case	Mean values	Best case
Life time [years]	15-20	20	17.5	15
Sales volume [t/y]	See Table 6-24	95th percentile of lognormal distribution = 309 m³	Mean value 241m³ with linear drop after 2001 see Table 6-24	Mean value 241m³ with sales stop in 2001
Market share of know producer [%]	25-50	25	37.5	50
Total Stock reduction [%]	25	25	25	25
Current stock [m³]		18,540 (2008)	5,062 (2010)	1,809 (2010)

In order to come to a best estimate on the basis of this data background, the mean values derived from [RIVM_2009] have been applied. Considering furthermore that the stock in the Netherlands corresponds to about 5 % of the total amount of PFOS containing fire fighting foam in the EU 27, the total amount in the EU would be 101,234 t which include about 1,012 t of PFOS. This represents about 10 times the amount which was received from the UK study [PFOS DEFRA 2004b].

Information from the UNEP questionnaire was scarce and has been evaluated according to the following table.

Table 6-26: Information on fire fighting foam quantities, related PFOS concentrations and resulting PFOS amounts for some Member States according to the UNEP questionnaire

Member state	Foam amount [t]	PFOS concentration [%]	PFOS amount [t]
AT	0.8	1-2%	0.012
BG	11.0	3%	0.33
DE	1,050	3-10%	68.25
SE	0.25	1%	0.025
TOTAL	1,062.05	3%	68.617

In the questionnaire [UNEP DE_2010] Germany stated that they still store about 1,000 tonnes of PFOS containing foams with a concentration of 3-10 % considering an average concentration of about 6.5 %. This would result in approximately 68 t of stored PFOS in foams in Germany, which would be slightly more than the estimated Europe wide amount.

Considering all received data from the different literatures [DEFRA 2004], [CARACAL 2008], [RIVM 2009], [UNEP DE 2010] The range of PFOS containing fire fighting foam stocks for the EU27 is therefore between 54 and 1,012 t, with a best estimate of about 94 t.

It has to be mentioned that these estimates are related to considerable uncertainties. In case the legislation is fulfilled, all of this foam should be removed anyway by 27th June 2011 from fire fighting grids, with a possible other storage of maximum 1 year. The amounts of fire fighting foams do therefore not have a consequence onto the elaboration of the limit values in section 8 and 9.

F) Waste Treatment

PFOS containing fire fighting foams which have not been used and which have to be destroyed are incinerated as hazardous waste. For fire fighting foams which have been used, it is assumed that approximately 70% are directed to waste water treatment and the remaining 30 % are emitted directly to water and soil [SwissConfederation_2009].

G) Substance Flow

According to Table 6-21 it is estimated that only 0.5 % of the available stocks in 2010 (92.71 t) will be used in 2010. It is estimated that, from the amount used, 30 % is emitted to the environment (water and soil) and 70 % is subject to waste water treatment. In case of the waste water treatment path some of the PFOS will end up in the sewage sludge. For further information see section 6.3.13 on sewage sludge.

Every year about 6.6 % of the foams exceed their expiry date and have to be exchanged. The corresponding 6.05 kg (6.6 % of ~91 kg stored in 2010) of PFOS is considered as hazardous waste and is incinerated at hazardous waste incineration plants. In the year 2011 all the remaining stocks of fire fighting foams containing a PFOS amount of 84.2 t will have to be exchanged and therefore will have to be destroyed by hazardous waste incineration.

The following PFOS mass flow results:

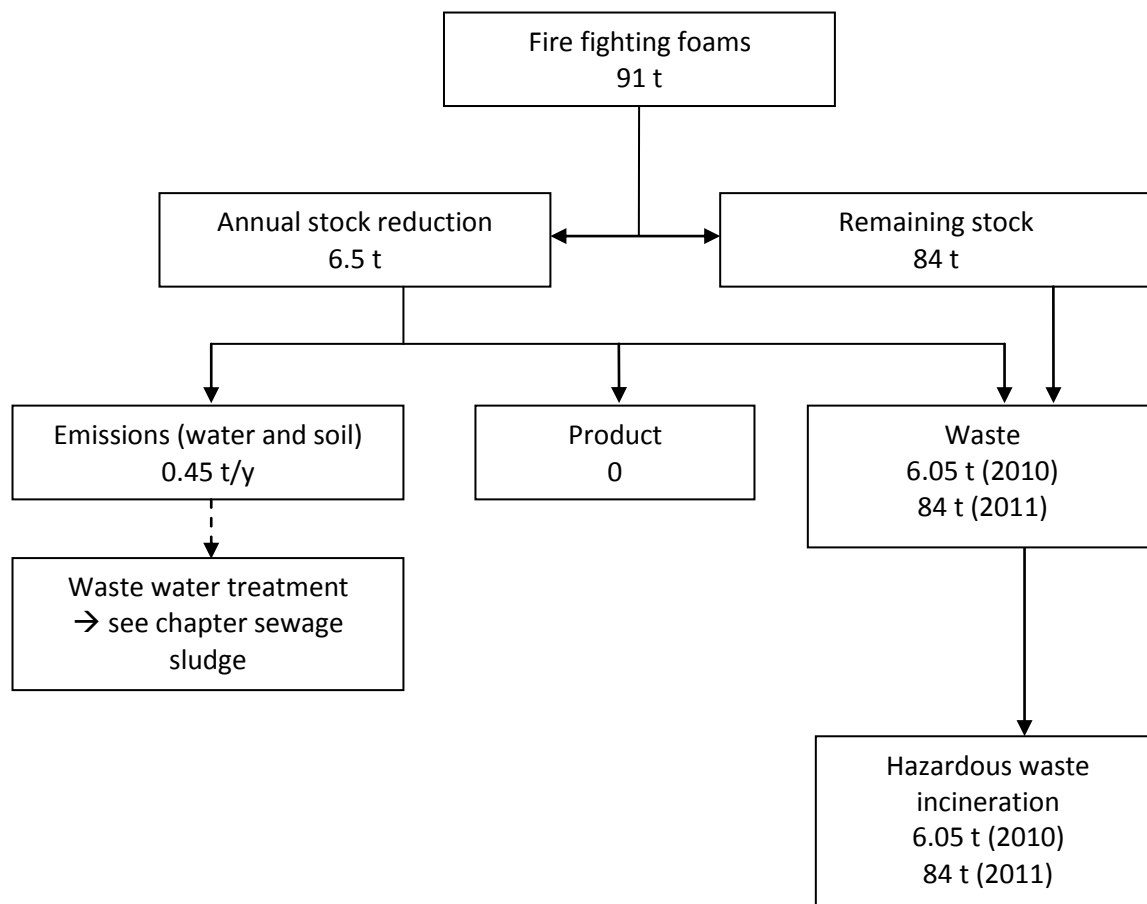


Figure 6-38: Assessment of the PFOS flow related to use in fire fighting foams estimated for 2010 in EU 27

H) Material Flow

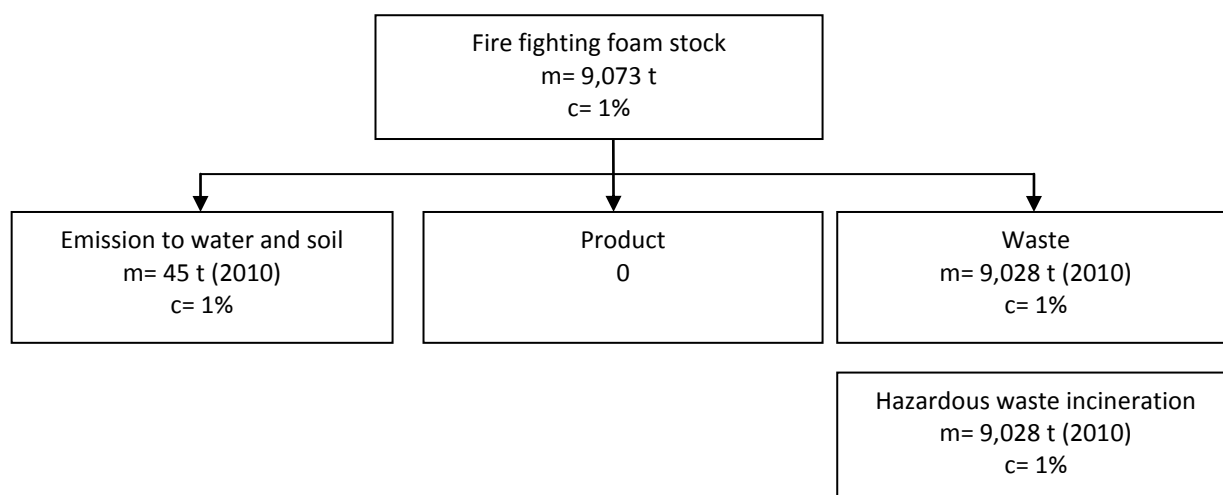


Figure 6-39: Waste flow for fire fighting foams in EU 27

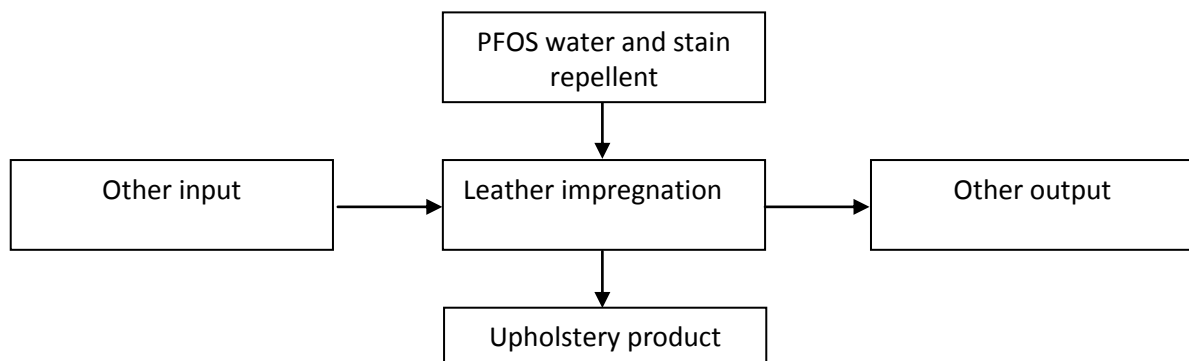
The concentration of PFOS within the waste is always 1 %. Once it is used in case of an emergency, the foam is emitted to the environment. The corresponding concentrations in soil and water, i.e. sewage sludges, are discussed in the corresponding section.

6.3.11 Leather industry

A) Background

The impregnation of leather with PFOS is a historical process. The type of leathers which has been impregnated with PFOS are leathers without finishes. These leathers have often been used for upholstery and belong to the higher price segment. The aim was to receive a highest possible natural look of the surface. For this purpose PFOS has only been sprayed onto the surface of such leathers to receive water and stain repellent properties. However, such leather products have been very sensitive and have only been a niche product [Personal Communication from Industry 14, 2010; PFOS_RIKZ_2002]. With the voluntary phase out from 3M from the PFOS production, it is expected that also the production of PFOS treated leathers has stopped.

B) Input/Output



Input:

PFOS has been used for treating the surface of natural leather used for upholstery. The stain and water repellence was received by spraying a PFOS solution onto the surface.

Output:

The only relevant output from the former process of leather impregnation with PFOS solutions are the upholstery products containing PFOS. As PFOS is not used anymore in this industry Emissions from the process are not considered and are not relevant today.

C) Generation Factors

Losses during the product lifetime due to wear out and emissions in the impregnation process have not been considered. The generation factor in this industry is 1. The use of 1 kg PFOS results in 1kg PFOS in the treated upholstery products.

The average lifetime of upholstery is estimated to be about 10 years which would mean that upholstery which is disposed of in the year 2010 has been bought approximately in 2000. In 2000 the industry still used about 5.7 t of PFOS per year for the treatment of leather upholstery which is in 2010 the output as bulky waste. It has to be expected that a dramatic drop of PFOS containing bulky waste will take place by 2012 and that by 2015 almost all PFOS containing upholstery should have been treated as waste.

It is estimated that about 20 % of upholstery consists of leather which again contains about 0.04 % of PFOS. The PFOS concentration of the whole upholstery product is therefore about 80 ppm. In correlation with the used 5.7 t of PFOS the bulky waste should have a yearly amount of about 71,342 tons of which about 14,268 tons are leather.

D) Contamination data

Concentrations of fluorochemicals in leather industry products are very low (0.025-0.05% on leather weight) [RIKZ_2002] and an average concentration of 0.04 % PFOS treated leather is applied. The average mass of leather is approximately 6 kg/m². Therefore the PFOS load is about 2.4 g/m².

No emission estimates have been made [RIKZ_2002].

E) Activity data

The PFOS impregnated leathers are considered to be at the upper price level and used only for leathers without finishing. This means that only a small fraction of the consumed leather should contain PFOS. It has therefore been assumed that at 30 % of the leathers produced before 2002 contained PFOS. After 2002 the amount of PFOS treated leather sold in Europe can be considered negligible [Personal Communication from Industry 14, 2010]. According to the VDL (Verein Deutscher Lederindustrie) 20 % of the leather is used for shoes, 20 % for other leather applications and 60 % for the automotive industry and upholstery [VDL 2010]. The share of automotive industry is considered to be similar as for the upholstery industry. It is therefore estimated, that 30 % of the total amount of leather is used for each segment.

The following table summarizes the annual European leather production before 2002 [Euroleather_2010].

Table 6-27: Annual European leather production from 1997 to 2002 [Euroleather_2010]

Member state	1997 [1000 m ² /y]	1998 [1000 m ² /y]	1999 [1000 m ² /y]	2000 [1000 m ² /y]	2001 [1000 m ² /y]	2002 [1000 m ² /y]
Austria	5,836					
Belgium	0,399			0,349	1,094	0,415
Denmark	6,1			0,725	0,78	0,8
France	7,1	11	10	10	30,766	8,7
Germany	17,5	16,5	16,4	14,968	15,5	16,5
Greece	41,5	5		0,1		
Hungary						
Italy	147,78		160	198,3	199,337	181,612
Ireland	4			4,08		
Luxembourg						
Netherlands*	7			3,9		4
Spain	48,65	44,9	44,22	45,32	49,05	42,906
Portugal	10,68	12,085	10,881	8,809	9,58	8,606
UK	13	11	15,3	14,5	13,4	12,15
Sweden	2,6	2,4		2,6	2,9	2,5
Finland	1,841		0,905	0,16	13,4	0
Total	306,986	102,885	257,706	303,811	335,807	278,189

The following figure visualises the shares of the leather production in the different European countries.

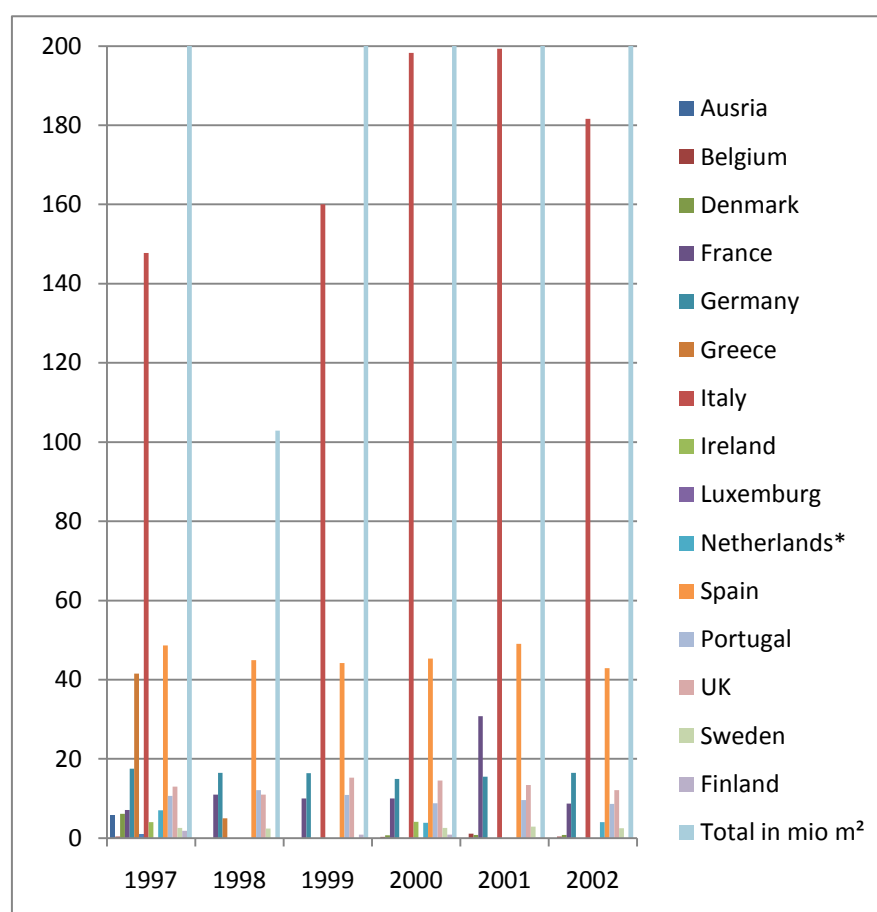


Figure 6-40: Overview of the leather production of different EU Member states [Euroleather_2010]

This results in an annual average production of 264,230,667 m²/year. To calculate the average PFOS consumption in this industry the following factors have been used:

Upholstery share of leather industry:	30%	[VDL_2010]
Average leather weight:	6 kg/year	[RIKZ_2002]
PFOS concentration of leather w/w:	0.04% (0.025-0.05%)	[RIKZ_2002]
Share of PFOS treated leather:	3%	[Personal Communication from Industry 15, 2010]
Lifetime of upholstery:	10 years	[GEO 2007]

With this data an average annual consumption of 5.7 t of PFOS before 2002 has been calculated.

The annual consumption of PFOS for the textile, carpet and leather industry before 2002 has been estimated to be 197 t/y in Europe [DEFRA_2004b]. The leather industry therefore only had a small share in comparison with the other two industries.

Using the yearly production before 2002 and the life time of 10 years the following table has been prepared which demonstrates the drop of left PFOS amount stored in European households due to upholstery.

Table 6-28: Reduction of PFOS contained in upholstery in households

Year	PFOS amount in upholstery in households [t]
1999	57
2000	57
2001	57
2002	57
2003	51
2004	46
2005	40
2006	34
2007	29
2008	23
2009	17
2010	11
2011	6
2012	0

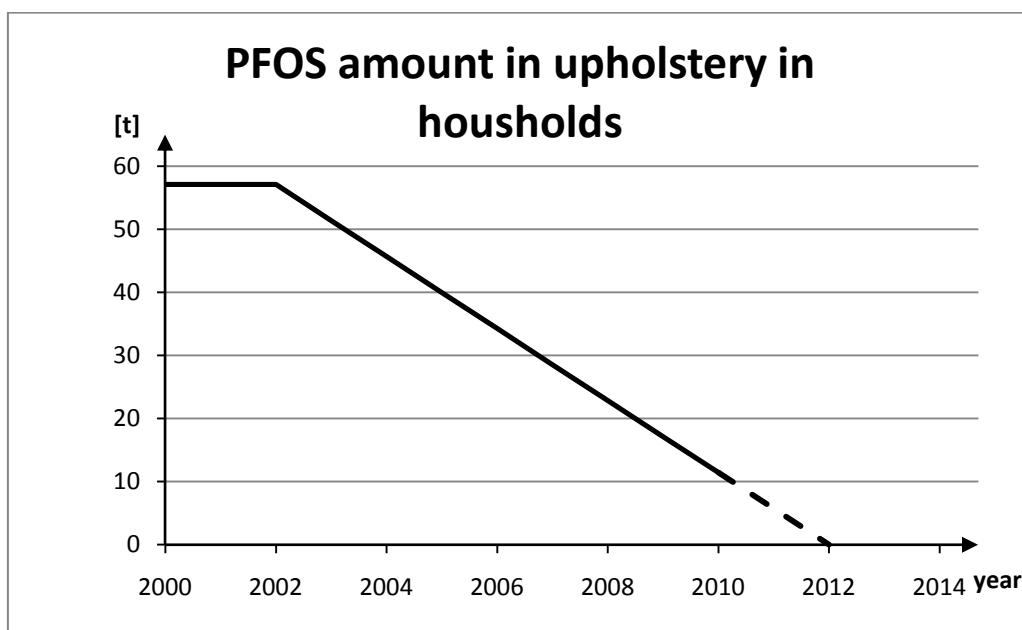


Figure 6-41: Reduction of PFOS contained in upholstery in households

F) Waste Treatment

One recycling possibility of leather is done by Salamander Industry Product “SIP” which processes off cuts from the leather industry. This leather has to be from a vegetable tanning process without finishing. The so called bonded leather is produced from chrome shavings as well as vegetable-tanned leather scraps, which are milled. Oil and natural latex as a binder, are added. In this way, bonded leather is almost completely produced from renewable sources. They are the world market leader and to their knowledge this is the only recycling possibility of leather [Personal Communication from Industry 16, 2010].

A recycling except for energy recovery of used leather is not known except cutting and sewing of the old leather to produce a new product (upholstery leather becomes a bag). This is only done by small companies in hand work with low production quantities.

Considering that only 30% of the leather is used for upholstery and that only 3 % of this leather contains PFOS which has a concentration of 0.04 % the PFOS concentration in mixed leathers would be 3.6 ppm.

According to a study from [EBIA/ISOPA/EUROPUR 2008] discarded upholstery furniture and mattresses are part of the Municipal Solid Waste (MSW) stream and are collected either separately as bulky waste, classified as EWC 200307 – bulky waste or together with the MSW fraction as EWC 200301 – mixed municipal waste. According to the study the following treatment and recovery operations are currently used:

- Incineration
- Landfilling
- Gasification

- Rebonding of foam flocks
- Powder milling

Incineration with energy recovery is recommended as the preferred technology from an economically and environmentally view by [EBIA/ISOPA/EUROPUR 2008]. Also landfilling in controlled landfills is still a used treatment option in many EU countries lacking of incineration capacity, but is to be phased out in the coming years. Also [Alcock et al. 2003] states that discarded furniture, foam and bedding will be practically disposed of to landfills or be incinerated.

EUROSTAT provides data for the treatment of household and similar wastes. Based on those data and on the assumption that recycling operations are applied only marginally for furniture and mattresses, it is assumed that 67.1 % of the used furniture is landfilled, 20.2 % is incinerated without and 12.7 % is incinerated with energy recovery (see Table 5-1).

G) Substance Flow

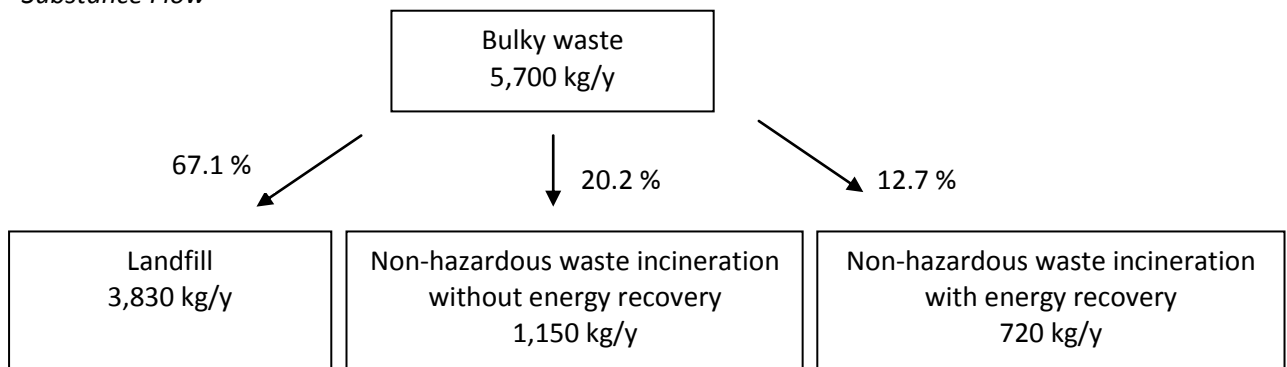


Figure 6-42: Substance flow of PFOS in the leather industry.

H) Material Flow

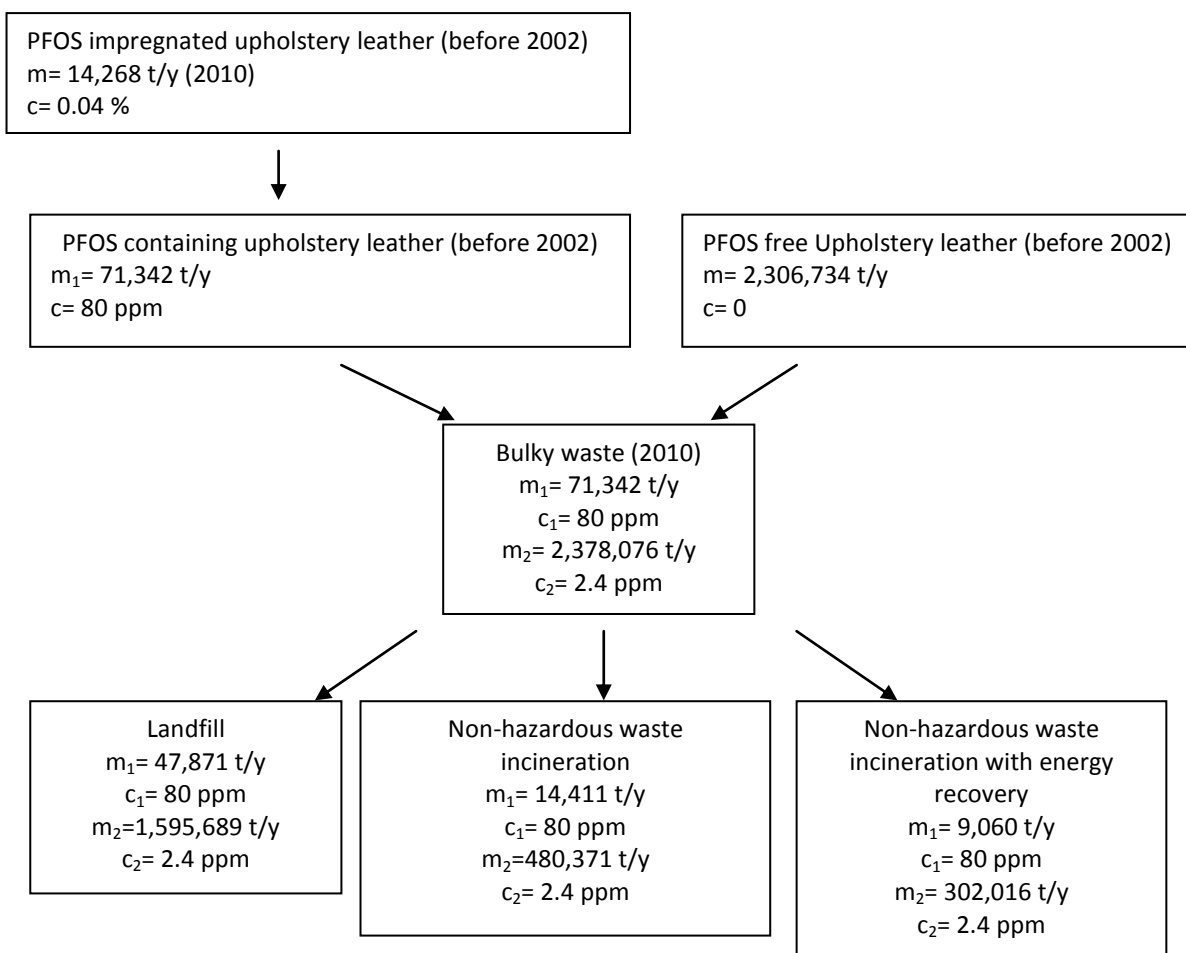


Figure 6-43: Material flow of PFOS in the leather industry

It is estimated that about 20 % of upholstery consists of leather which again contains about 0.04 % of PFOS. The total PFOS concentration of the upholstery is therefore about 80 ppm. In correlation with the used 5.7 t of PFOS, the bulky waste should have a yearly amount of about 71,342 tons of which about 14,268 tons are leather.

Taking further into account that only 3 % of the leather upholstery is treated with PFOS, the average concentration of PFOS in all leather upholstery would be only 2.4 ppm.

Before 2002 about 5.7 tons of PFOS have been used for the production of 71,342 tons of leather upholstery, representing about 3 % of the total leather upholstery market which therefore accounts to 2,378,067 tons.

6.3.12 Carpet

A) Background

The treatment of carpets is a historical use of PFOS. It is expected, that the use stopped approximately in the year 2002 with the voluntary phase out of 3M from the PFOS production. Taking into account that the life time of carpets is about 14 years, it is very likely that PFOS containing carpets can still be found in the bulky waste of households at least until 2016.

PFOS has only been used for the impregnation of carpets made of synthetic fibres, which have a market share of approximately 86%. Wool made carpets cover the remaining 14% [GUT_2009].

Life time of a carpet is considered to be about 10-18 years [GUT_2009] and an average of 14 years has been used for the calculations.

B) Input/Output

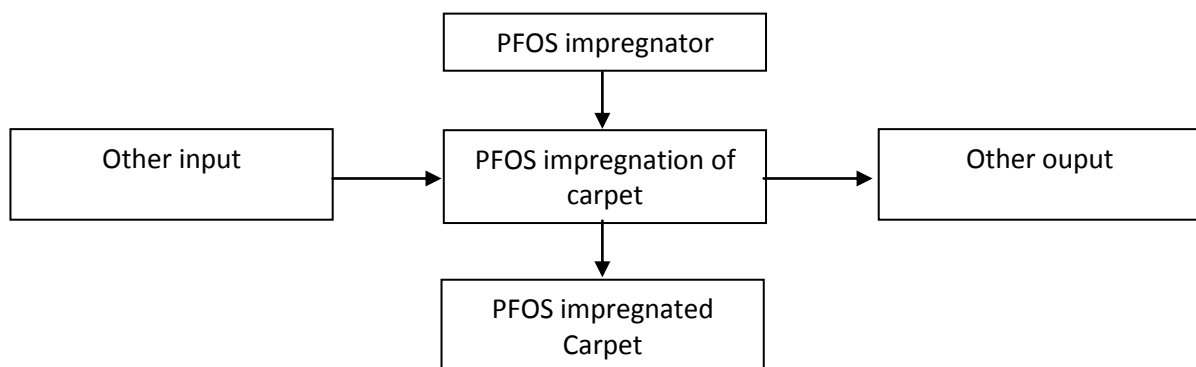


Figure 6-44: Input/Output overview of PFOS use for carpets

Input:

The only relevant output from the former process of carpet impregnation with PFOS solutions are the carpets containing PFOS. Emissions from the process are not considered and are not relevant today.

Output

The annual input before 2002 is the output of 2010. This means that 1,938,018 t of carpets with an average PFOS content of 75 ppm (see contamination data) ends up as bulky waste in 2010. It is expected that this will hardly change before 2016. Losses due to wear out have not been considered.

C) Generation Factors

Losses during the product lifetime due to wear out and emissions in the impregnation process have not been considered. The generation factor in this industry is 1. The use of 1 kg PFOS results in 1kg PFOS in the impregnated carpets.

The use of PFOS in the carpet industry is a historical use. However due to its long life time of 14 years [GEO_2007] the production of 1996 ends nowadays up as bulky waste, which contains PFOS. For this purpose it is necessary to estimate the annual production amount of carpets and their PFOS concentration before the phase out of PFOS in the carpet industry which was presumably around the year 2002.

D) Contamination data

For the given amount of synthetic fibres 1,666,695 t/y, about 146 t of PFOS have been used before 2002 for the production of carpets. For PFOS impregnated carpets the PFOS content would therefore be 88ppm.

Considering that the carpet waste is a mixture of all carpets with an annual production of 1,938,018 t/y and the use of 146 t PFOS per year, the total average concentration of PFOS in carpets is about 75 ppm.

E) Activity data

In the report [GUT_2009] the production of tuft carpets, which have a market share of about 58 %, are provided for Western and Eastern Europe and are summarised in the following table.

Table 6-29: Overview of tuft carpets

Countries	2005 [10^6 m ²]	2006 [10^6 m ²]	2007 [10^6 m ²]	2008 [10^6 m ²]	2009 [10^6 m ²]	Average
Western Europe	587	588	584	522	430	542
Eastern Europe	40	44	48	53	45	46
TOTAL	627	632	632	575	475	588

For the annual production of carpets before 2002 the average production of the years 2005 to 2009 of 588 million m² has been assumed. Considering the above mentioned market share of tufted carpet of 58 %, a total annual carpet production of 1.014 billion m² for Europe results.

Report [GUT_2009] includes a table providing the composition of the average Dutch made tufted carpet, which sum up to a total weight of 1.911 kg/m² of which the pile is about 558 g/m². With the total weight, the total annual carpet production of Europe is about 1,938,018 t/y. This means that the amount of synthetic fibre made carpets, which have a market share of 86 % is about 1,666,695 t/year. Considering that only the pile is relevant for the synthetic fibre the amount of synthetic fibres used is 486,665 t/y. These fibres are impregnated with about 3% of a PFOS polymer which again consists of 1 % of PFOS. Therefore the content of the impregnated synthetic fibre is about 0.03 %.

According to [POPRC_2010] a PFOS concentration of 3 % was considered for textiles. As both materials are synthetic fibres and both are treated for stain and water repellence, the same concentration has

been used for carpets. The same report also states that the PFOS content of carpets is 15 %. However, this value seems to be very high, especially as this ratio refers to the total carpet weight. It is assumed that this value is misinterpreted from the Report [RIKZ_2002] which states:

“These soil repellent products for carpets are generally referred to as Scotchgard products, which is the brand name of the 3M product for this application. The commercial products for carpet protection contain approximately 15% fluoroalkyl acrylic polymers (Tomasino, 1992, 3M, 2000b, 3M, 2000c).”

Therefore it can be stated, that the 15 % concentration levels represent the concentration of PFOS in the carpet protection product.

F) Waste Treatment

Several attempts have been done in Europe for recycling of carpets, but have not been successful. One intensive project has been the “Recycling of Carpet Material”-Project [RECAM_1997].

There are ongoing recycling projects in the US. However, due to the different composition of the carpets as well as the difference concerning leasing and buying of carpets, this system cannot easily be adopted in Europe.

In Europe each year 1.6 million t of carpet end up in landfills (70%) or municipal waste incineration (30%) [RECAM_1995]. The incineration is considered to be 15 % energy recovery and 15% non hazardous waste incineration.

These figures are from the mid 1990ies and it should be considered that the share of incineration has meanwhile significantly increased. Carpets usually become part of bulky waste. Therefore, analogue to considerations made for upholstery, and considering that recycling operations are applied only marginally for carpets, it is assumed that 60 % of the used carpets are landfilled, 22 % are incinerated and 18 % are incinerated with energy recovery.

G) Substance Flow

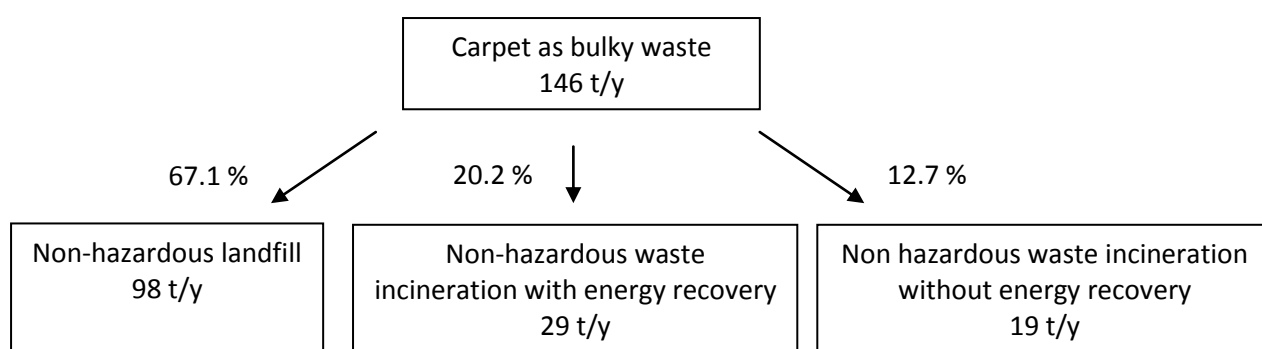


Figure 6-45: Substance flow of PFOS in the leather industry

H) Material Flow

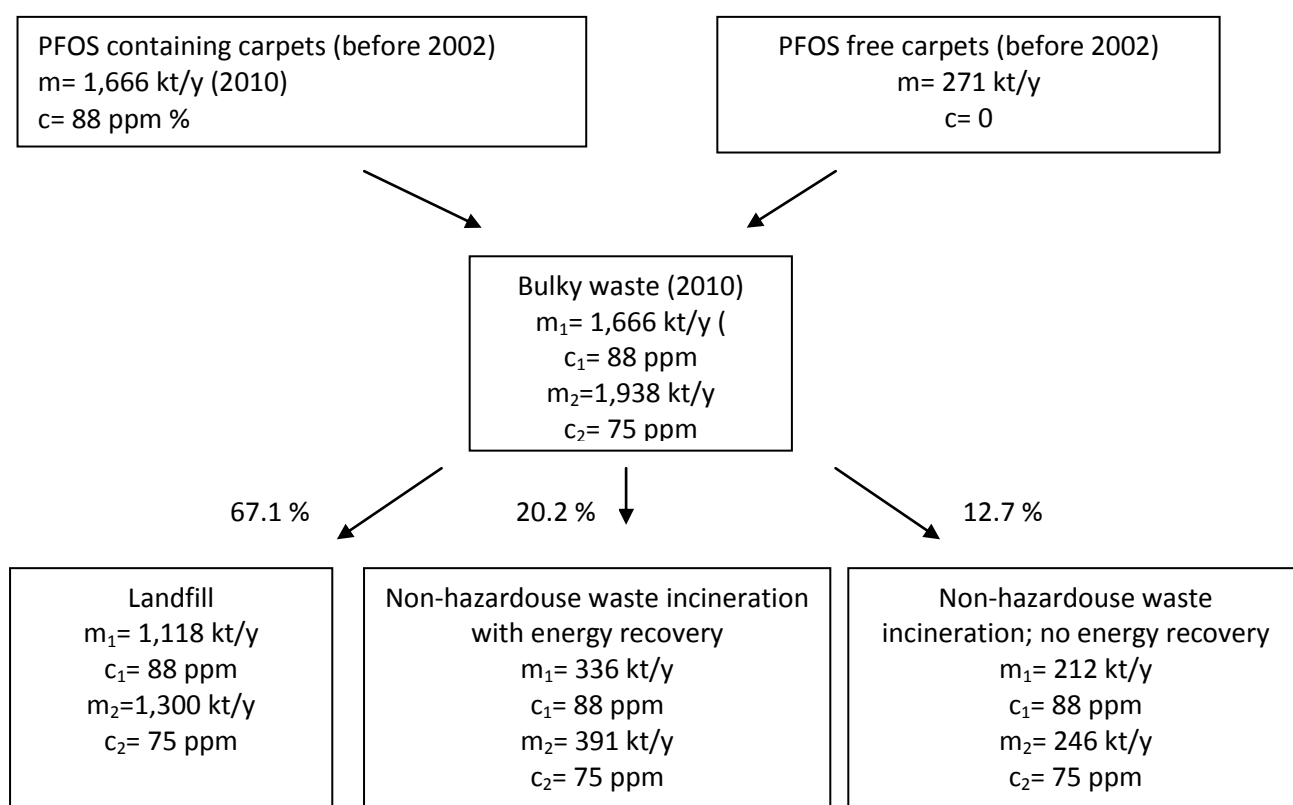


Figure 6-46: Material flow of PFOS in the carpet industry

M_1 represents the amount and concentration if only carpets made of synthetic fibres are considered. If all carpets are mixed, the corresponding amounts and concentrations are provided as m_2 and c_2 respectively.

This estimation of 1.6 million tonnes of carpet [RECAM_1995] fits well with the 1.67 million tonnes of carpet made of synthetic fibres or the total amount of 1.94 million tons (including wool carpets).

6.3.13 Sewage sludge

A) Background

One route for PFOS to the environment may be through sewage treatment plants (STPs) and landfills, where elevated concentrations have been observed compared to background concentrations. Once released from STPs, PFOS will partially adsorb to sediment and organic matter. A substantial amount of PFOS may also end up in agricultural soil, due to the application of sewage sludge. The primary compartments for PFOS are therefore believed to be water, sediment and soil [RIKZ_2002] [UK-EA_2004].

Based on information from the Report [UN-HABITAT_2008], calculating the average sewage sludge production (6 million metric tonnes) per person from about 260 million inhabitants, it is estimated that the yearly production of sewage sludge in EU 27 (500 millions) is about 11.000.000 metric t per year dry mass. Regarding the data from a report for the EU [EU_2008], the EU 27 wide production of sewage sludge is about 11.3 million t of sewage sludge, of which ~40% are used in agriculture.

B) Input/Output

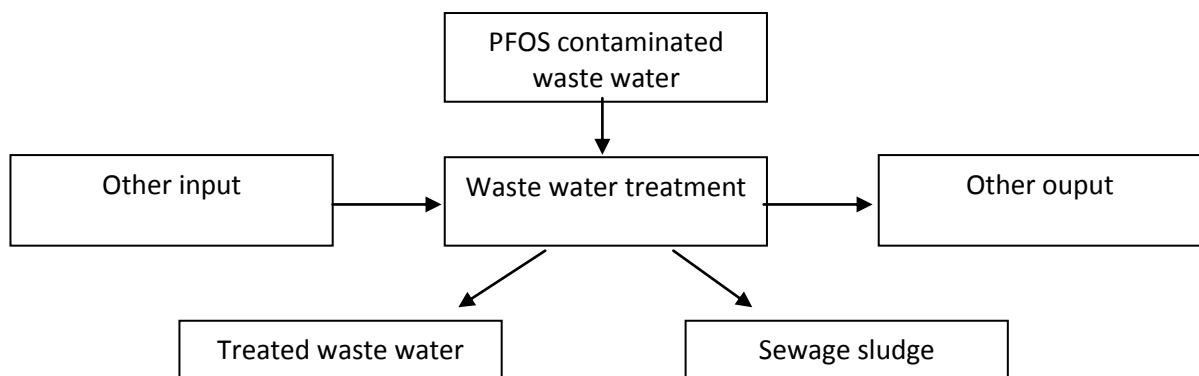


Figure 6-47: PFOS relevant Input/Output overview of sewage sludge

Input:

Waste water from different sources, including PFOS related sources such as chromium plating facilities, photographic industry, semiconductor industry or other relevant companies e.g. textile or film recycling facilities.

Output:

Sewage sludge from waste water treatment

C) Generation Factors

It is estimated that about 5% of the sewage sludges is contaminated with 1 ppm PFOS and 95% of the sewage sludges are contaminated with 0.1 ppm PFOS.

D) Contamination data

As described in the background section two different sewage sludge contamination values regarding PFOS have been considered.

Due to a contaminated site in North Rhine Westphalia there have been several measurements and studies about PFOS content in sewage sludges.

In Germany sewage sludges, which contain more than 100 µg/kg PFOS, may not be used for agricultural uses. About 7% of the Bavarian sewage sludges exceed this limit value. [BLU_2010]. In North Rhine Westphalia about 9% of the sewage sludges from waste water treatment plants exceed the value of 100 µg/kg. Sources with higher sewage sludge contaminations arise from the chromium plating industries, contaminated soils, or relevant companies (textile or film recycling companies). The contaminated sewage sludges contained up to 6,000 µg PFOS/kg close to galvanic industries [NRW_2007c].

Considering that Germany has a high concentration of industry, it is assumed that the content of the Europe wide sewages sludges with a PFOS concentration exceeding 100 µg is less than the assessed 7 or 9% of Germany. Therefore, it is estimated that in average 5% of the European sewage sludge is contaminated with more than 100 µg PFOS/kg.

The sewage sludges with PFOS contents above 100 µg/kg in North Rhine Westphalia are listed in the following table. The values represent the highest concentration at each location [NRW_2007c].

Table 6-30: measured PFOS concentration of sewage sludges exceeding 100 µg/kg [NRW_2007c]

Location	PFOS concentration [µg/kg]	Location	PFOS concentration [µg/kg]	Location	PFOS concentration [µg/kg]
1	6300	14	960	27	240
2	6180	15	940	28	200
3	5300	16	910	29	200
4	3000	17	890	30	160
5	2980	18	520	31	150
6	2100	19	510	32	150
7	1500	20	420	33	150
8	1400	21	410	34	130
9	1300	22	340	35	120
10	1100	23	325	36	110
11	1080	24	330	37	132
12	1030	25	320	Average	1167
13	1000	26	290		

As there are also three very high hot spots included it was decided that for high PFOS contaminated sludges a value of 1000 µg/kg or 1 ppm will be applied.

The PFOS contamination of sewage sludges from municipal sewage plants mainly treating water from households is clearly below 100 µg/kg [NRW_2007c]. For calculation of the mass flow an average concentration of 100 µg/kg or 0.1 ppm was applied. The low contaminated sewage sludge has a share of 95 % of the total sewage sludge amount and the high contaminated sewage sludge has a share of 5 %.

The Environmental Protection Agency (EPA) in Ireland has recently undertaken sampling of wastes from sewage treatment to determine concentration levels of the new and candidate POPs in those wastes. According to [EPA IRE 2011a] trace contaminations of PFOS were detected in 2 samples, however the concentrations measured are well below the lower limit of the LPCLs proposed in this study. For PFOS analysis, the extraction method was direct injection and analysis standard based on Environmental Agency's method.

E) Activity data

The European yearly production of sewage sludge is about 11.578×10^6 t/y.

The data from the following table has been taken from EUROSTAT. For Portugal no sewage sludge amount was available and has therefore been taken from the report [UN-Habitat_2008]. For the distribution of the use of sewage sludge in Portugal the average usages from Europe have been adopted. From the total amount of 11,578 million kg/y only the data of the usage of 10,653 million kg/y are known. The missing 8% have therefore been given to "other uses". The received total amount of sewage sludge from EUROSTAT fit very well with the amount given by [UN-Habitat_2008] of 11,370 million kg/y.

Table 6-31: Annual sewage sludge production in Europe [EUROSTAT 2010]

Use of sewage sludge of Europe						
Member state	Total amount [10 ⁶ t/y]	Landfill [10 ⁶ t/y]	Incineration [10 ⁶ t/y]	Agriculture and compost [10 ⁶ t/y]	Others uses [10 ⁶ t/y]	Total treatment [10 ⁶ t/y]
AT	255	25	98	113	18	254
BE	112	9	36	21	45	111
BG	40	21	0	6	0	27
CY	9	1	0	5	0	6
CZ	172	9	0	135	28	172
DE	2,049	5	965	1,079	67	2,116
DK	0	0	0	0	0	0
EE	31	5	0	5	19	29
ES	1,065	168	41	687	169	1,065
FI	160	10	0	147	3	160
FR	1,060	222	178	632	27	1,059
GR	126	123	0	0	3	126
HU	286	137	1	131	16	285
IE	88	5	0	63	22	90
IT	1,056	440	31	466	116	1,053
LT	76	9	0	32	0	41
LU	12	4	0	8	0	12
LV	23	0	0	10	9	19
MT	0	0	0	0	0	0
NL	541	36	287	83	116	522
PL	1,088	298	34	163	595	1,090
PT	237	40	45	118	34	237
RO	758	100	4	20	37	161
SE	210	24	0	104	22	150
SI	21	9	5	4	4	22
SK	332	9	0	39	9	57
UK	1,771	96	282	1,235	158	1,771
TOTAL	11,578	1,805	2,007	5,306	1,517	10,635
Share	100%	16%	17%	46%	13%	92%

Use of sewage sludge of Europe						
Member state	Total amount [10 ⁶ t/y]	Landfill [10 ⁶ t/y]	Incineration [10 ⁶ t/y]	Agriculture and compost [10 ⁶ t/y]	Others uses [10 ⁶ t/y]	Total treatment [10 ⁶ t/y]
Unknown						8%
TOTAL share	100%	16%	17%	46%	21%	100%
Total amount	11,578	1,805	2,007	5,306	2,460	

F) Waste Treatment

According to EUROSTAT data, at European level sewage sludge is treated with the following shares:

- 16% are landfilled
- 17% are incinerated
- 46% are used in agriculture and compost
- 21 % are applied for others uses

It has been proven that the PFOS content of contaminated sludge which is incinerated at a temperature of at least 850 °C is completely destroyed [NRW_2007a].

G) Substance Flow

Figure 6-48 shows the substance flow of PFOS in sewage sludge. Currently low and high contaminated sewage sludges are used in the same manner.

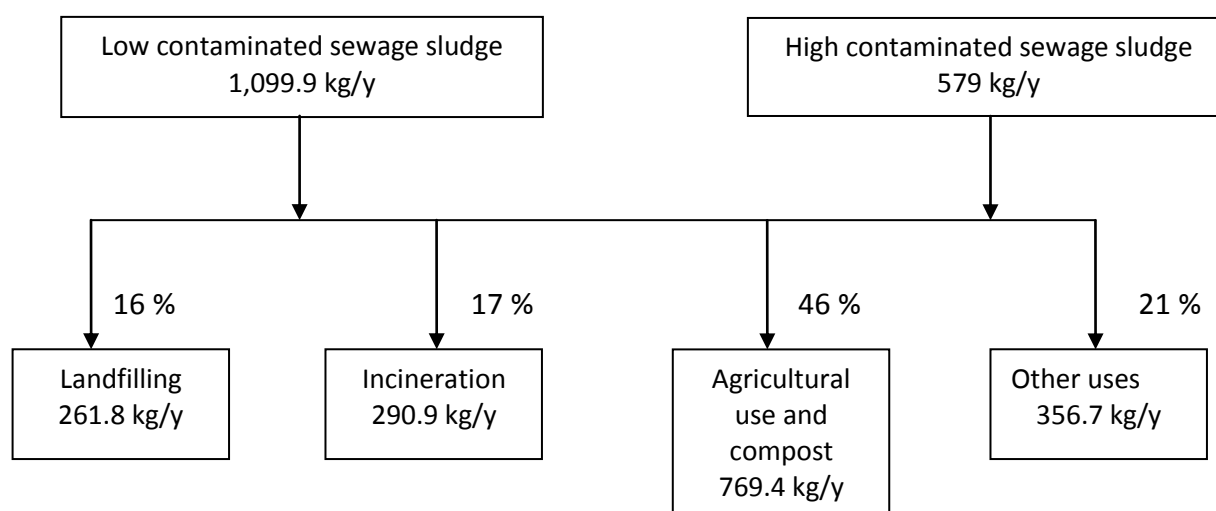


Figure 6-48: Substance flow of PFOS in sewage sludge

H) Material Flow

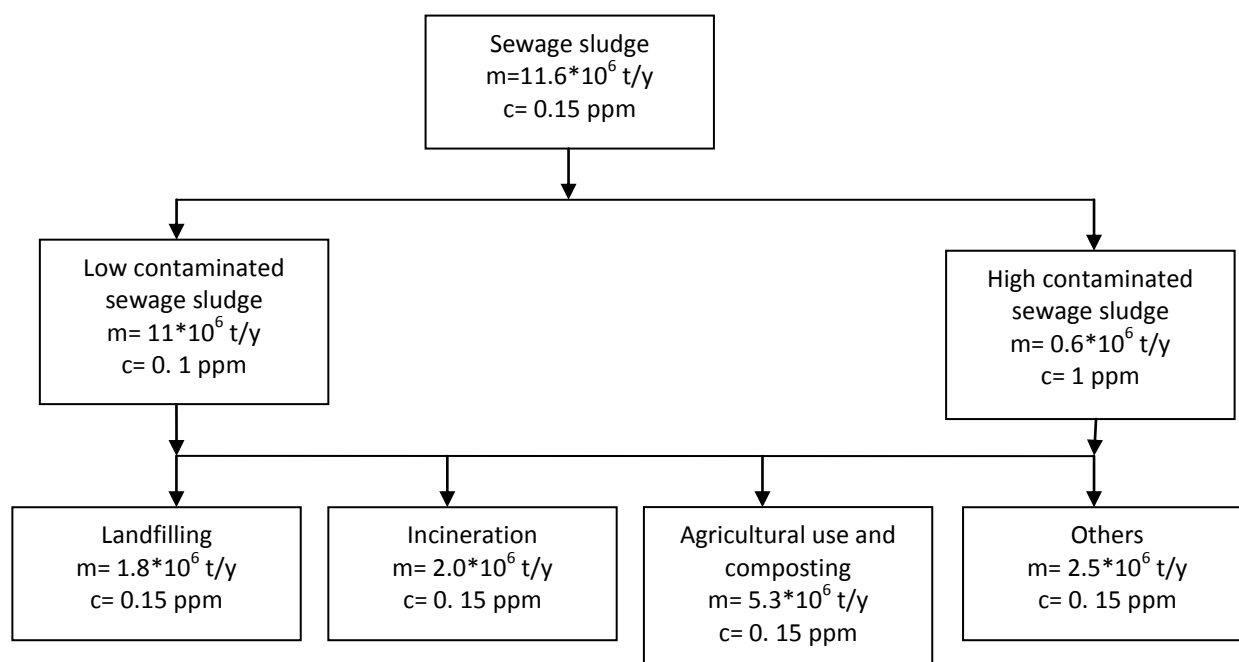


Figure 6-49: Material flow of PFOS in sewage sludge

At the current state the PFOS content is only seldom measured in sewage sludge. Therefore, it is not expected that any differentiation of the use of sewage sludge concerning its PFOS content is done Europe wide in the moment. The awareness of contaminated sewage sludges is given in some countries or local areas, but too little is known to make a more distinguished Europe wide estimation. It has to be considered that the agricultural use represents an emission to the environment. From the 5.3 million t of sewage sludge containing about 770 kg of PFOS which are used for agricultural use about 0.265 million tonnes are highly contaminated (1%) which contain about 265 kg of PFOS. 5 % of the total sewage sludge amount contains about 34 % of the total PFOS amount in sewage sludges.

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6.4 PeCB flows

6.4.1 Use and Production

Pentachlorobenzene (PeCB) belongs to the chemical group of chlorobenzenes and has been used in the past as pesticide as well as chemical intermediate for the production process of the fungicide quintozone (pentachloronitrobenzene), as flame retardant, in dyestuff carriers and as a component in polychlorinated biphenyl (PCB) products. There is no quantitative information available on historic use and production [UNEP/POPS/POPRC.4/15/Add.2].

Except its use and production in small quantities for laboratory purposes, there is no intentional use reported at present in the UNECE region [Euro Chlor 2007, UNEP/POPS/POPRC.3/20/Add.7, UNEP/POPS/POPRC.4/15/Add.2] and therefore it is unlikely that it is still used as pesticide or as flame retardant on its own in Europe. There is no information on its current use in dyestuff carriers. It can be assumed that this application has ended [Environment Canada 2005]. PeCB is registered neither in any of the products of the Swedish products register nor as pollutant in other chlorinated products [Haskoning 2001]. Its use as fungicide and flame retardant outside the UNECE region in the rest of the world is unclear [UNEP/POPS/POPRC.2/13, Euro Chlor 2007, Haskoning 2001].

However, PeCB still can be found in old electrical equipment in use in North America and Europe, in which it has been applied as dielectric fluid in combination with PCBs [UNEP/POPS/POPRC.3/20/Add.7, Environment Canada 2005, Haskoning 2001, Euro Chlor 2007]. It is a component of chlorobenzene mixtures to reduce viscosity of PCB products for heat transfer and of chlorobenzene mixtures combined with PCBs for use in electrical equipment, e.g. transformers [Euro Chlor 2007, Environment Canada 2005]. After all chemical companies (members of ICCA/WCC/EuroChlor) stopped the production and marketing of PCBs in the 1980s, also the amount of PeCB used for this purpose declined considerably after 1980 [UNECE 2007, Haskoning 2001, UNEP/POPS/POPRC.3/20/Add.7]. Since PCBs are being taken out of service in many countries of the world, it is expected that related PeCB emissions will further decrease with time [UNEP/POPS/POPRC.3/20/Add.7].

In the past, PeCB has further been used as feedstock/intermediate in the production of the fungicide quintozone resulting in impurities of PeCB in the end product and therefore unintentional releases to the environment. Major U.S. and European manufacturers of quintozone changed their manufacturing process to eliminate this use of PeCB [UNEP/POPS/POPRC.4/15/Add.2, Haskoning 2001]. Since PeCB is not a metabolite of quintozone, quintozone produced according to the new process is no source of PeCB to the environment [Haskoning 2001]. Quintozone use stopped in the EU in 2002 [UNEP/POPS/POPRC.4/15/Add.2]; according to the International Pesticide Elimination Network on POPs (IPEN) quintozone was banned in the EU already in 1991 [UNEP/POPS/POPRC.4/15/Add.2]. However, PeCB still might be present as an impurity in quintozone stockpiles. Globally, quintozone still might be used, although it is unclear if it is manufactured using PeCB [UNECE 2007].

It was reported that pentachlorobenzene can be produced by denitrification of pentachloronitrobenzene as well as the reductive dechlorination of hexachlorobenzene [WHO IPCS 1991]. About 50 % of the world production of all chlorobenzenes has been manufactured in the USA

with the remainder being produced in Western Europe and Japan, whereas monochlorobenzene was approximately 70 % of the total chlorobenzene world production [WHO IPCS 1991]. No recent information is available except that PeCB is not produced anymore in North America and Europe since there is no commercial demand for it anymore. No recent data are available on production outside the UNECE region [UNECE 2007]. It has been reported that the Czech Republic may be an exception to this, since uncertainty on the production of PeCB existed [UNECE 2006]. The Czech Republic reported no current production of PeCB in their country (besides Italy, Cyprus, Denmark, France, UK, Netherlands, Germany, Lithuania, Mauritius, Turkey, Canada, USA, Republic of Korea) [UNECE 2007, UNEP/POPS/POPRC.3/20/Add.7, UNEP/POPS/POPRC.3/20/Add.2]. PeCB has not been reported by EU industry as an HPVC or LPVC (high/low production volume chemical). No trade or stockpiles have been reported [UNEP/POPS/POPRC.3/20/Add.7].

There are no relevant indicators that intentional use and production of PeCB are of relevance in Europe. In fact, the most relevant source of PeCB emitted to the environment is caused by unintentional production due to different processes (i.e. power production from coal, MSWI, HWI, domestic burning and waste water treatment which leads to the generation of sewage sludge containing PeCB). These processes will be explained in more detail in the following chapters.

6.4.2 Chemical Characteristics of PeCB

PeCB belongs to the group of chlorobenzenes, which are characterised by a benzene ring in which the hydrogen atoms are substituted by one or more chlorines. The chlorobenzenes are neutral, thermally stable compounds with increasing stability and higher melting and boiling points with increasing chlorine substitution [UNEP/POPS/POPRC.4/15/Add.2]. In PeCB five hydrogen atoms are substituted by chlorine. In its pure form it is a crystalline, colourless solid, which is highly inflammable and nearly insoluble in water [GESTIS 2010].

IUPAC Name: pentachlorobenzene

CAS Chemical Name: benzene, pentachloro-

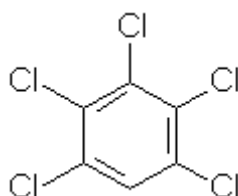
Synonyms: 1,2,3,4,5-pentachlorobenzene; pentachlorobenzene; PCB; PeCB; QCB; quintochlorobenzene

CAS Registry Number: 608-93-5

EINECS Number: 210-172-0

Trade Names: none [UNEP/POPS/POPRC.4/15/Add.2]

Structure:



Formula: C₆H₁Cl₅

Molar Mass: 250.34 g/mol [GESTIS 2010]

Physico-chemical properties

Melting point	82-85 °C	[GESTIS 2010]
Boiling point	275-277 °C	[GESTIS 2010]
Density (at 17 °C)	1.834 g/cm ³	[GESTIS 2010]
Vapour pressure (at 100 °C)	0.16 mbar	[GESTIS 2010]
Partition coefficient Log Kow	Recommended: 5.17 4.88-6.12 Recommended: 5.17-5.18	[GESTIS 2010] [UNEP/POPS/POPRC.3/20/Add.7]
Water solubility (at 25 °C)	0.135-3.46 mg/L Recommended: 0.55 mg/L	[UNEP/POPS/POPRC.3/20/Add.7]

6.4.3 Environmental background levels

PeCB has spread widely in the global environment. Different concentration levels have been detected in abiotic environmental matrices of remote regions such as the (ant)arctic region. Monitoring data on PeCB are also available for temperate zones where the substance was found in abiotic and biotic media. Existing monitoring data on PeCB show that levels of the substance were detected in atmosphere, biota and humans as well as in sediment and soil. A comprehensive overview of monitoring data in abiotic media and biota is given in [UNEP/POP/POPRC.3/INF/21].

Table 6-32 gives an overview of the concentration levels in sediment and soil detected in different parts of the world.

Table 6-32: Levels of PeCB in sediment and soil including hot spots (DW: dry weight)
([UNEP/POP/POPRC.3/20/Add.7]; [UNEP/POP/POPRC.3/INF/21])

Country/Region	Compartment	Levels of PeCB	References	Comments
Sediment				
Remote lakes in northern Canada	Sediment surface layers	<0.01 to 0.73 µg/kg sediment	[Muir et al. 1995]*	Concentrations were measured in a period of time estimated between 1979-1988. Not mentioned if these concentrations are based on wet or dry weight basis.
Alaskan arctic lakes	Sediment	averaged 0.10 ±0.10 µg/kg DW	[Allen-Gil et al. 1997]*	Sampled in four lakes from 1991 to 1993.
Elbe river, Germany	Sediment	<1 - 71 µg/kg	[Witter et al. 1998]*	Not mentioned if these concentrations are based on wet or dry weight basis.
Yangtse river, China	Sediment	3 µg/kg	[Jiang et al. 2000]*	Samples were taken in 1998. Not mentioned if these concentrations are based on wet or dry weight basis.
Taiwan	Sediment	Up to 15.7 µg/kg	[Lee et al. 2000]*	Samples were taken in 1996. Not mentioned if these concentrations are based on wet or dry weight basis.
Northern Norway and	Sediment	2 to 5 µg/kg DW	[Strachan et al. 2001]*	Bottom sediment samples taken from harbours

Country/Region	Compartment	Levels of PeCB	References	Comments
the Kola Peninsula in the arctic				
Spree river; Havel river, Germany	Sediment	<10 - 17; <10 - 76 µg/kg	[Schwarzbauer et al. 2001]*	Samples were taken in the period from 1993 to 1994. Not mentioned if these concentrations are based on wet or dry weight basis.
Mulde reservoir, Germany	Sediment	0.4 µg/kg	[Zoumis et al. 2001]*	Not mentioned if these concentrations are based on wet or dry weight basis.
Sweden	Sediment (lakes)	<6 µg/kg DW	[Sternbeck et al. 2003]	Samples were taken in 2002.
	Sediment	<4 µg/kg DW		
Masan bay, Korea	Sediment	Up to 0.28 µg/kg	[Hong et al. 2003]*	Samples were taken in 1997. Not mentioned if these concentrations are based on wet or dry weight basis.
Sweden	Sediment	1 µg/kg	[Kaj and Dusan 2004]	Samples were taken in 2004. Not mentioned if these concentrations are based on wet or dry weight basis.
Lippe river, Germany	Sediment	1 to 6 µg/kg	[Kronimus et al. 2004]*	Samples were taken in the period from 1999 to 2001. Not mentioned if these concentrations are based on wet or dry weight basis.
Ebro river, Spain	Sediment	1.17 µg/kg	[Lacorte et al. 2006]*	Samples were taken in 2003. Not mentioned if these concentrations are based on wet or dry weight basis.
Kishon river, Israel	Sediment	0.01 to 0.06 µg/kg	[Oren et al. 2006]*	Not mentioned if these concentrations are based on wet or dry weight basis.
Soil				
Coastal areas of Victoria Land (Antarctica)	Soil	0.4 and 1.3 µg/kg DW	[Borghini et al., 2005]	In these soil samples PeCB was the dominant organic compound.
Hot Spot levels (sediment and soil)				
USA	Sediment	147 to 5100 mg/kg	[ATSDR 2007]	The ATSDR database from the US Government contains 41 records of polluted sites with PeCB. The maximum concentrations are indicated. Not mentioned if these concentrations are based on wet or dry weight basis.
	Soil	0.43 to 2040 mg/kg		
Danube Region	Sediment	0.001 to 3500 µg/kg	[Slobodník and Dogterom 2003]	Danube Regional Project for the European Water Framework Directive; samples for sediment were taken in 2002. Not mentioned if these concentrations are based on wet or dry weight basis.

*data cited by [ICCA/WCC 2007]

A large quantity of monitoring data on PeCB in temperate zone exists as well as for remote areas. The substance was detected in abiotic media and biota and mainly originates from developed countries. It appears that the general concentration levels of PeCB in these zones are decreasing. This is a representative pattern for most of the POPs [UNEP/POP/POPRC.3/20/Add.7].

Nevertheless, in temperate zones a clear trend of PeCB presence due to its concentration levels in sediment cores is possible to be derived [UNEP/POP/POPRC.3/20/Add.7]. The decrease of PeCB concentration, for example, was observed in sediment cores from the industrial region from Lake Ontario, which is located close to the Niagara River in Canada. An increase of the concentration was reported for the time period from the early 1900 to 1960-1970 by [Durham and Oliver 1983]* and [NYDEC 1998]* as cited by [ICCA/WCC 2007]. The measured peak concentration was over 100 µg/kg. After the period 1960-1970 the concentrations decreased to about 10% of the peak concentration by 1980 ([Durham and Oliver 1983]*; [NYDEC 1998]* as cited by [ICCA/WCC 2007]; [UNEP/POP/POPRC.3/20/Add.7]). Also the concentrations of PeCB in sediment of the Ketelmeer in The Netherlands decreased by 37% between 1972 and 1988 [Beurskens et al. 1993].

As shown in Table 6-32, in the environment, the detected background contamination levels in sediment and soil ranged from less than 0.01 µg/kg in remote arctic regions to 76 µg/kg in Europe (Germany). Hot spot levels were detected inter alia in the Danube Region in sediment and ranged up to 3500 µg/kg. From records of PeCB polluted sites in the US the maximum concentration levels reach up to 5100 mg/kg in sediment and up to 2040 mg/kg in soil.

6.4.4 Legal Situation

6.4.4.1 EU level

POPs related provisions

PeCB is listed in Annex I, III, IV and V to the proposal for the amendment of the POPs Regulation. Annex I does not foresee any exemption from the use and production ban. In Annex V, a provisional low (LPCL) and maximum (MPCL) POP concentration limits have been set. The limits are subject to review

- LPCL: 50 mg/kg;
- MPCL: 5000 mg/kg.

Waste management

Wastes containing PeCB are subject to the general rules of EU waste legislation (see chapter 4.1.2). Provisions specifically related to PeCB containing wastes have not been identified.

Classification/chemicals legislation

PeCB is classified

- under CLP Regulation as flammable (sol 1), acute toxic, and acute and chronic aquatic toxic;
- under Directive 67/548/EEC as harmful and dangerous for the environment.

As regards potential risks from PeCB as impurity in quintozone, it should be noted that quintozone is not included as active substance in Annex I to Directive 91/414/EEC (repealed by 1107/2009). That means that authorisations for plant protection products containing quintozone may not be granted in the EU.

Other fields (water, PRTR, food)

PeCB is identified as priority hazardous substance in the Water Framework Directive. Environmental Quality Standards (EQS), both for annual average (AA-EQS) and for Maximum Allowable Concentration (MAC-EQS) for inland surface water for PeCB as follows:

- Inland AA-EQS (i): 0.007 [µg/l]
- Other surface waters AA-EQS (o): 0.0007 [µg/l].

In the PRTR Regulation, it is laid down that PeCB releases of >1 kg/y to air, water, or land have to be reported by the concerned facilities.

PeCB is no subject to EU regulations in the field of food and feed safety.

6.4.4.2 Member States level

Waste water management

Under the regulations of the Department of Environment and Transport and the Regions (DETR) UK, Scotland and Northern Ireland, it is prohibited to discharge PeCB directly or indirectly to groundwater [Environment Canada 2005].

In the Czech Republic PeCB is part of an integrated monitoring program on POPs, which provides information on the Central European levels of POPs, the long-term trends in those levels and the impact of various sources and the effectiveness of measures applied to reduce the impact [UNEP/POPS/POPRC.5/15/Add.2]

6.4.4.3 Other countries

In the USA PeCB is listed under the U.S. Clean Water Act as toxic and is classified as non-priority substances [Environment Canada 2005; Haskoning 2001]. PeCB is also regulated under the U.S. Safe Drinking Water Act [Haskoning 2001]. Further, PeCB is listed in the U.S. EPA's Final Rule on the Water Quality Guidance for the Great Lakes System (2000) [Environment Canada 2005].

In Canada PeCB was added to the List of Toxic Substances of the Canadian Environmental Protection Act/ Prohibition of Certain Toxic Substances Regulations in 2005. Canadian regulations allow the use of PeCB in chlorobiphenyl liquid such as liquid used for servicing equipment containing chlorobiphenyls. PCBs are regulated under the Chlorobiphenyls Regulations and Storage of PCB Material Regulations [UNECE 2007; UNEP/POPS/POPRC.5/15/Add.2; UNEP Guidance Material Draft 2009]. The Canadian Council of Ministers of the Environment established an interim chronic exposure water quality guideline for the protection of freshwater aquatic life of 0.006 mg/L for PeCB [Environment Canada 2005]. Various other initiatives also contribute to reductions in PeCB emissions in Canada such as the development of Canada-wide Standards for the reduction of dioxins and furans. They have entered into force in 2001 for municipal solid waste, hazardous waste, medical waste and sewage sludge incineration and 2004 for conical municipal waste combustion (phased out in Newfoundland by 2008; new conical waste combustors prohibited in Canada in all jurisdictions).

6.4.5 Occurrence of PeCB in Europe

6.4.5.1 Past intentional uses

PeCB has been intentionally produced for several uses in Europe in the past. Past uses include uses as pesticide, as flame retardant, in dyestuff carriers, as dielectric fluid in combination with PCBs in electrical equipment and as feedstock in the production of the pesticide quintozone. Within Europe the intentional use of PeCB for these applications has stopped long ago since no indicator on current use and production of PeCB exists in Europe.

Pesticide

According to various sources, PeCB has been used as pesticide (fungicide) in the past, whereas it is unknown whether it is still used in other parts of the world for this application [Haskoning 2001; UNECE 2006; Euro Chlor 2007; UNEP/POPS/POPRC.2/13; UNEP/POPS/POPRC.3/20/Add.7]. In Norway PeCB is not registered in the Norwegian Product Register (for all chemical products imported or manufactured in quantities more than 100 kg/y), but it has been used as pesticide before 1974 [UNECE 2006]. It was reported by one source that PeCB was applied in a pesticide to combat oyster drills [WHO IPCS 1991]. Another source mentions that PeCB has been used as fungicide for wood and textiles, but this application is unlikely to take place in Europe nowadays [INERIS 2005].

Flame retardant

It was reported that PeCB has been used as flame retardant in the past [Haskoning 2001; INERIS 2005; UNECE 2006; Euro Chlor 2007; UNEP/POPS/POPRC.2/13; UNEP/POPS/POPRC.3/20/Add.7]. It has been mentioned that PeCB for this use has been applied in Italy until the 1980s [UNECE 2007].

Dyestuff carriers

Very little information exists on the use of PeCB in dyestuff carriers. It was reported that PeCB and TeCB have been used for this application, whereas it is unknown whether they have been used together or separately [UNEP/POPS/POPRC.3/20/Add.7]. However, use of PeCB in dyestuff carriers has been discontinued [Environment Canada 2005; UNEP/POPS/POPRC.3/20/Add.7].

Past use in electrical equipment

PeCB still can be found in old electrical equipment in use in North America and Europe, in which it has been applied in combination with PCBs as dielectric fluid [UNEP/POPS/POPRC.3/20/Add.7; Haskoning 2001; Environment Canada 2005; Euro Chlor 2007]. It is a component of chlorobenzene mixtures to reduce viscosity of PCB products for heat transfer and of chlorobenzene mixtures combined with PCBs for use in electrical equipment, e.g. transformers [Euro Chlor 2007, Environment Canada 2005].

After all chemical companies (members of ICCA/WCC/EuroChlor) stopped the production and marketing of PCBs in the 1980s, also the amount of PeCB used for this purpose declined considerably after 1980 [UNECE 2007, Haskoning 2001, UNEP/POPS/POPRC.3/20/Add.7].

In Canada emissions of PeCB due to PCB use losses were assumed to amount to 5.63 kg/y with releases to air (0.004 kg/y), water (4.171 kg/y) and soil (1.451 kg/y) [Environment Canada 2005]. Global emissions of PeCB have been estimated to amount to 373 kg/y based on the Canadian value for PeCB release and estimations on PCB releases [Euro Chlor 2007]. No specific information is available on quantities of PeCB releases for Europe for this use.

In the EU, the use of PCB was banned long ago (Directive 76/768/EEC) and regulations have been set for the elimination of existing equipment. The equipment has been subject to the PCB disposal directive (1996/59/EC) requiring final phasing out of equipment containing liquids with PCB >500 mg/kg until 2010. Equipment between 50 and 500 ppm may stay in use until its life-time expires but shall be included in the inventories. Equipment with amounts of PCB <50 mg/kg are subject to the ELV (2000/53/EC) and the WEEE directive (2002/69/EC) [BiPRO 2005].

Quintozene production

In the past PeCB has been used as feedstock/intermediate in the production of the fungicide quintozene (pentachloronitrobenzene) resulting in impurities of PeCB in the end product and therefore unintentional releases to the environment. Major U.S. and European manufacturers of quintozene changed their manufacturing process to eliminate this use of PeCB by producing quintozene through chlorination of nitrobenzene [UNEP/POPS/POPRC.4/15/Add.2, Haskoning 2001]. Comments from the World Chlorine Council (WCC) indicate that some production of quintozene from smaller producers anyhow may have resulted in releases of PeCB as impurity to the environment. According to larger producers such as Amvac Chemical and Uniroyal Chemical Corporations, PeCB impurities have no longer been present in their production processes after changing it [UNECE 2006].

Since PeCB is not a metabolite of quintozene, quintozene produced according to the new process via chlorination of nitrobenzene can be excluded as source for PeCB emissions to the environment [Haskoning 2001]. No information exists on when exactly the changes in production have been implemented (it can be assumed prior to 2001).

Quintozene was banned in the EU in 2002 [UNEP/POPS/POPRC.4/15/Add.2]; according to the International Pesticide Elimination Network on POPs (IPEN) quintozene was banned in the EU already in 1991 [UNEP/POPS/POPRC.4/15/Add.2]. However, PeCB still might be present as an impurity in quintozene stockpiles as far as existing within Europe [UNECE 2007]. This case was indicated for Lithuania [Haskoning 2001]. Table 6-33 gives an overview of the situation in different Member States of the EU before the ban of quintozene in 2002 as far as reported [Haskoning 2001].

Table 6-33: Overview of quintozene production and use in the EU before its ban in 2002

MS/country	Status	Production of quintozene	Use of quintozene	Source
Czech Rep.	Info available 2002	Info available 2002	Info available 2002	[Haskoning 2001]
Denmark	Pesticide approval expired 1985; last sale in 1984		No use	[Haskoning 2001]
Estonia:	Banned in 1999		No use	[Haskoning 2001]
Finland	Marketing and use banned in 1996;	Banned	Banned	[Haskoning 2001]

MS/country	Status	Production of quintozone	Use of quintozone	Source
	registration voluntarily withdrawn in 1991 (before: used as fungicide); no import, no production, (probably) no stockpiles left			
France	Still marketed		Use	[Haskoning 2001]
GB	Council Directive 90/533/EEC bans the use with more than 10g/kg PeCB since 1991 [UNECE 2007]	No production; production until 1991 [UNECE 2007]	Use	[Haskoning 2001], [UNECE 2007]
Germany	Completely banned as pesticide	Banned	Banned	[Haskoning 2001]
Greece	Still marketed		Use	[Haskoning 2001]
Ireland	Still marketed		Use	[Haskoning 2001]
Latvia	Banned for import, marketing, use as pesticide (exceptions), not banned for production; no use; no stockpiling	Not banned for production	No use	[Haskoning 2001]
Lithuania	No registration as pesticide, restricted to professional users, existing stockpiles: 479 kg quintozone, 40177 kg Pentatium (20% quintozone) in farm storages		Restricted to professional quintozone users	[Haskoning 2001]
Netherlands	Banned	No production	No use	[Haskoning 2001]
Poland	Banned for marketing and use in 1996; import not possible (except scientific purposes); presumably no production and export	No production	No use	[Haskoning 2001]
Spain	Still marketed		Use	[Haskoning 2001]
Sweden	Banned in 1985	No production	No use	[Haskoning 2001]

In most EU countries it has been indicated that no production and use of quintozone takes place (DK, EE, FI, DE, NL, PO, SE). In Latvia quintozone was banned for import, marketing and use as pesticide with few exceptions, but it has not been banned for production. Anyhow, they reported no use and no stockpiles. No information was available on the current situation in the Czech Republic. FR, GB, GR, IE, LI and ES indicated that quintozone still was used until its ban in 2002, whereas in UK production took place until its ban due to Council Directive 90/533/EEC in 1991. No information on stockpiles of quintozone in these countries exists. Lithuania reported restriction of quintozone use to professional users and mentioned existing stockpiles of quintozone (479 kg) and Pentatium (40,177 kg), a formulation containing 20 % quintozone (i.e. 8,035.40 kg quintozone).

Since other existing stockpiles have not been reported for Europe, it is assumed that no significant amounts of PeCB in quintozone stockpiles exist in the EU. If the stockpiles in Lithuania still exist, there is no information whether these stockpiles contain PeCB impurities. However, it is assumed that these stockpiles do not exist anymore.

6.4.5.2 Current unintentional production & releases

At present the most relevant source of PeCB emitted to the environment is caused by unintentional production due to different processes. PeCB can be unintentionally formed whenever organic compounds are burned or exposed to high energy in presence of a chlorine source [Environment Canada 2005].

In an estimate from Bailey from 2007 the annual world wide release of PeCB amounted to 85,000 kg/y, whereas unintentional releases due to combustion of biomass, solid waste and coal are responsible for 43,900 kg/y, 32,740 kg/y and 6,113 kg/y, respectively, followed by industrial processes as aluminum production (1100 kg/y) and other chemical manufacturing processes and waste disposal (431 kg/y). Unintentional releases from pesticide use (342 kg/y), of old electrical equipment (373 kg/y) and of chlorinated solvents (<2 kg/y) play a minor role in comparison to the releases due to incineration/combustion and industrial processes [Euro Chlor 2007]. An update of these estimates resulted in total annual emissions of ca. 120,902 kg/y based on values indicated in Table 6-34 [Bailey et al. 2009], giving more importance to pesticide use and degradation of chemicals.

The table also indicates Canadian values reported by Environment Canada. For Canada similar sources have been identified accounting for 41.8 kg/y PeCB releases [Environment Canada 2005]. These releases were distributed between open burning of backyard trash (21.93 kg/y), incineration/combustion of municipal solid waste (2.36 kg/y) and hazardous solid waste (1.84 kg/y), magnesium production in the industry (1.53 kg/y), pesticide use (6.2 kg/y), pentachlorophenol (PCP) treated wood (2.34 kg/y), dielectric fluid (5.6 kg/y) and solvent use (0.04 kg/y) [Environment Canada 2005].

Table 6-34: Overview of estimated annual global and Canadian emissions of PeCB [Bailey et al. 2009; Environment Canada 2005]

Emission source	Worldwide amount [kg/y]	Canada [kg/y]
PCB use losses	400	5.60
Chlorinated solvents	<2	0.04
Pesticide use	5,400	8.54
Chemical manufacturing and waste disposal	400	-
Aluminum production	1,100	Magnesium production: 1.53
Incineration of solid waste	3,600	4.20
Uncontrolled combustion of solid waste	28,000	21.93
Combustion of coal	11,000	-
Combustion of biomass	45,000	-
Degradation of other chemicals	26,000	-
Total annual emission	120,902	41.8

For Europe similar sources have been assumed since no other data is available except for PeCB releases reported by European facilities to the European Pollutant Release Transfer Register (PRTR). According to the PRTR, PeCB releases amounted to 1,879.2 kg in 2008 (with 1,850 kg to air and 29.2 kg to water). The industrial sectors responsible for these releases are facilities in the metallurgic sector, facilities in the field of waste management as well as mineral oil and gas refineries in the energy sector [PRTR 2007; PRTR 2008]. An overview of the distribution is shown in Figure 6-50.

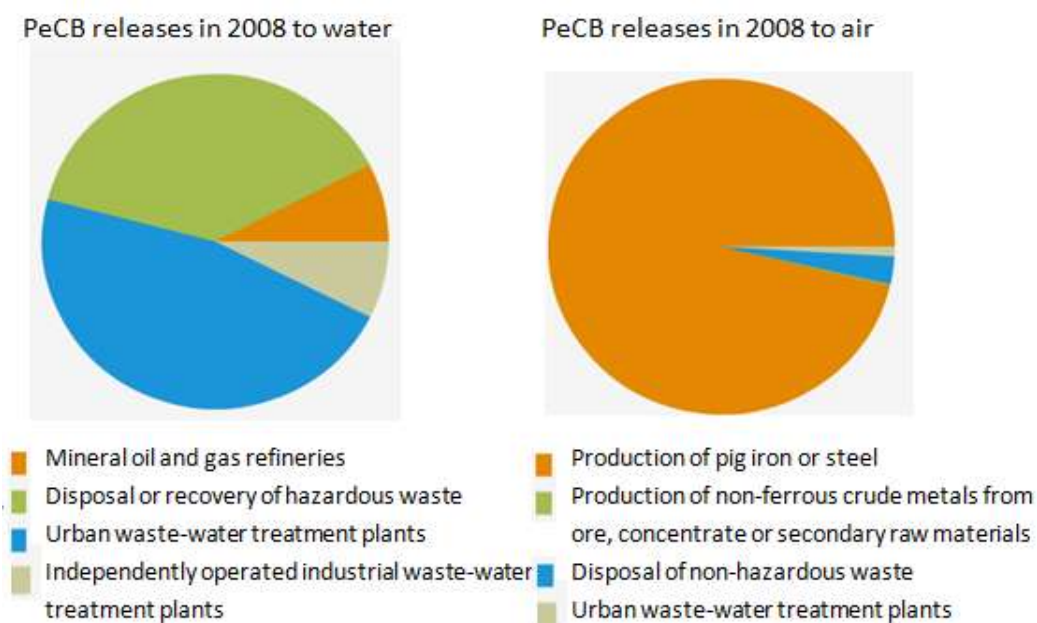


Figure 6-50: Overview of PeCB releases to water and soil from facilities in Europe in 2008 [PRTR 2008]

According to the information available an overview of possible unintentional PeCB production and emission sources has been allocated, which is schematically described in Figure 6-51.

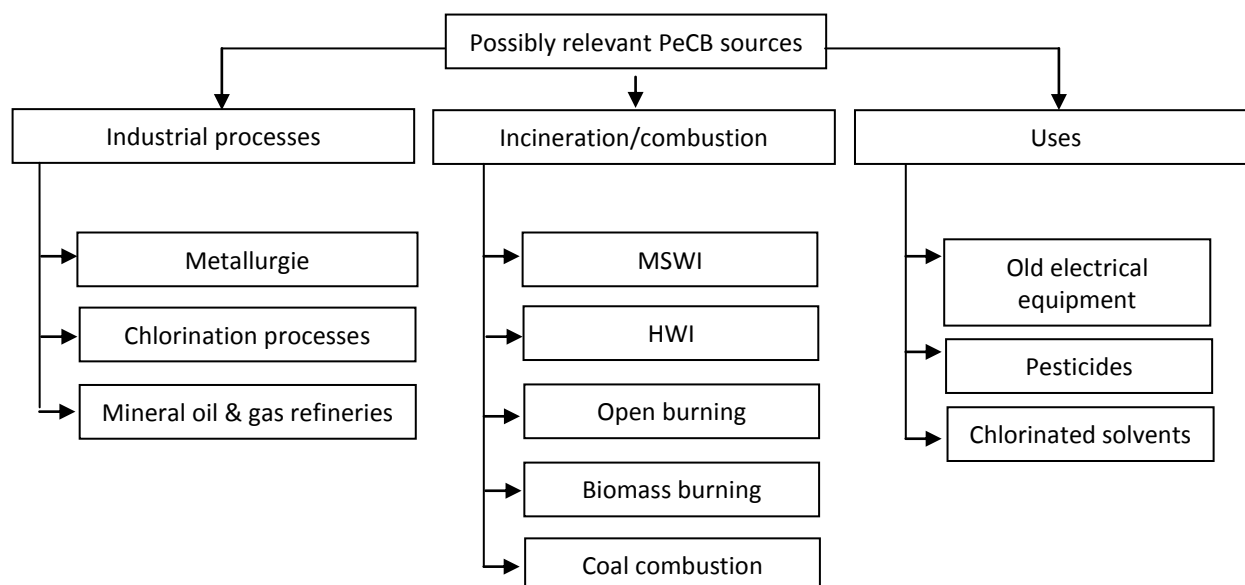


Figure 6-51: Overview of possibly relevant sources of PeCB in Europe

Incineration and combustion processes

The main source of unintentional PeCB production is incineration/combustion of wastes and materials. The main sources identified are incineration of wastes, whereas uncontrolled incineration due to open

burning of household waste, e.g. barrel burning and backyard trash burning, account for high amounts [Bailey et al. 2009; Environment Canada 2005]. Municipal solid waste incineration (MSWI) and hazardous waste incineration (HWI) are further sources of unintentional PeCB production [Environment Canada 2005]. Bailey et al. further stated that combustion of biomass is a significant source of PeCB releases worldwide; also combustion of coal has been indicated as relevant source [Euro Chlor 2007].

Industrial processes

It has been reported that PeCB can be formed unintentionally as by-product in the industrial sectors.

In the metallurgic sector it has been reported that PeCB has been produced in aluminum production during the use of hexachloroethane (HCE) to remove dissolved hydrogen from molten aluminum in foundries, secondary copper production and magnesium production [Euro Chlor 2007]. Further sources might be production of niobium, tantalum and the pigment titanium dioxide [Euro Chlor 2007]. Releases of PeCB have been also reported by Environment Canada for production of Magnesium [Environment Canada 2005]. Data reported under the European PRTR indicate PeCB releases to air from production of iron and steel as well as non-ferrous crude metals from ore, concentrates or secondary raw material [PRTR 2008].

Further, unintentional production of PeCB can occur during various chlorination processes in the chemical industry as reported by several sources [Euro Chlor 2007, UNECE 2007, Environment Canada 2005, Environment Canada 1994]. PeCB has been reported as by-product or contaminant during the production of other chlorinated organic substances such as tetrachlorobenzene (TeCB), hexachlorobenzene (HCB), quintozone, 1,1,1-trichloroethane, carbontetrachloride, trichloroethylene and ethylenedichlorid [Environment Canada 1994].

Current uses and related unintentional releases

Besides releases from old electrical equipment, PeCB can be contained as impurity in different products from the chemical industry. This includes pesticides, as already described for quintozone, and chlorinated solvents.

Small amounts of PeCB have been found in perchloroethylene and carbon tetrachloride [Environment Canada 2005; INERIS 2005]. Annual releases in Canada from these solvents have been reported to amount to only ~37g/y for each substance [Environment Canada 2005].

Besides impurities in quintozone, it has been reported that PeCB impurities are contained in the fungicide pentachlorophenol (PCP), which is used for wood preservation especially in Canada [UNECE 2007]. Within most EU countries PCP use has stopped and will be banned latest in 2010 [UNECE 2007].

Other pesticides might containing PeCB impurities besides quintozone and PCP are clopyralid, atrazine, chlorothalonil, dacthal, lindane, picloram and simazine [UNEP/POPS/POPRC.2/13]. Environment Canada reported that HCB is an impurity of above listed pesticides and in this way PeCB might be an impurity as well according to assumptions of the US EPA [Environment Canada 2005]. Technical grade HCB has been indicated to contain about 98% HCB, 1.8% PeCB and 0.2%TeCB. HCB has already been listed in Annex A

and C of the Stockholm Convention and it may thus be expected that HCB is of minor importance as a source for PeCB [UNEP/POPS/POPRC.3/20/Add.7]. According to an US EPA study PeCB was identified in quintozone, endosulfan, chlorpyrifos-methyl, atrazine and clopyralid, but not in simazine, chlorothalonil, picloram and dacthal [UNEP/POPS/POPRC.3/20/Add.7].

From the above listed pesticides quintozone, PCP, atrazine, lindane and endosulfan are banned within Europe, only clopyralid and chlorpyrifos-methyl are currently not [EU PD 2010]. PeCB is not contained in chlorothalonil, dacthal, picloram and simazine [UNEP/POPS/POPRC.3/20/Add.7].

6.4.6 *Relevance of the occurrence of PeCB in Europe*

According to the information available, it can be assumed that past uses of PeCB as pesticide, flame retardant and in dyestuff carriers are of no relevance for the occurrence of PeCB in Europe and therefore can be disregarded when allocating mass and waste flow.

From the past uses only the use of PeCB in old electrical equipment and existing stockpiles of quintozone potentially containing PeCB impurities may be of relevance, whereas both are considered of only low importance.

The use of PCB was banned long ago (Directive 76/768/EEC) in the EU while existing equipment was allowed to be kept in use. However, these equipment has been subject to the PCB disposal directive (1996/59/EC) requiring final phasing out of equipment containing liquids with PCB >500 mg/kg until 2010. Equipment between 50 and 500 ppm may stay in use until its life-time expires but shall be included in the inventories. Equipment with amounts of PCB <50 mg/kg are subject to the ELV (2000/53/EC) and the WEEE directive (2002/69/EC) [BiPRO 2005].

In the EU quintozone was banned in June 2002 [UNEP/POPS/POPRC.4/15/Add.2]. Therefore, only existing stockpiles within Europe which might contain PeCB impurities may be of relevance, e.g. as reported for Lithuania [Haskoning 2001]. Since quintozone stockpiles have not been reported within Europe except for Lithuania in 2001, it is unlikely that significant amounts still exist. There is no information whether these quintozone stockpiles in Lithuania still exist and whether they contain PeCB impurities. The latter is considered unlikely since the production process of quintozone has been prior to 2001 using manufacturing methods without PeCB via nitrobenzene. Remaining quintozone stockpiles manufactured via nitrobenzene, i.e. not including PeCB impurities, are of no relevance since PeCB has been reported by van de Plassche et al. to be no degradation product of quintozone [Haskoning 2001]. Bailey et al. reported that PeCB is a degradation product of HCB and quintozone [Bailey et al. 2009]. However, both substances are banned within Europe. HCB is listed as POP. Other pesticides which might contain PeCB as impurity are already banned in Europe except clopyralid and chlorpyrifos-methyl [EU PD 2010]. Both pesticides might contain small amounts of PeCB in form of an impurity according to a study from the US EPA [UNEP/POPS/POPRC.3-/20/Add.7]. However, it is assumed that these impurities only play a minor role since no further information exist.

Relevant quantities of PeCB still exist in Europe in contaminated sites due to the historic disposal of so called "HCB waste". For example at a factory in the Czech Republic 80,000 drums of HCB waste was

deposited in a landfill adjacent to the river Elbe (Ministry of Czech Republic 1991). Subsequent leaching of the waste into the river resulted in high concentrations of HCB accumulating in Elbe sediments downstream of the production and landfill area (Heinisch et al 2006a, Heinisch et al 2006b; quoted from [UNEP/POPS/POPRC.6/INF/21]). According to the existing legal situation, the corresponding quantities are considered outside the scope of current waste flows. However, they may become waste in the future.

To conclude, at present the most relevant source of PeCB emitted to the environment is caused by unintentional production due to different processes connected with releases to air, water and soil. These production processes are incineration/combustion processes of different kinds of wastes or other materials and PeCB by-product formations due to different industrial processes and waste water treatment.

6.4.7 Selection of Relevant Processes

According to various sources, it is obvious that intentional production and use of PeCB is not relevant anymore within Europe. From the former intentional uses of PeCB only uses in electrical equipment together with PCBs and uses in quintozone stockpiles might be of importance, but have been assumed to be insignificant, since regulations leading to their decrease are already in place.

The focus in this study for PeCB has been set on processes leading to unintentional production of PeCB. Most important are incineration and combustion processes since they account for the highest unintentional production and emission of PeCB. Municipal solid waste incineration (MSWI), hazardous waste incineration (HWI), coal combustion, domestic burning and waste water treatment (generation of sewage sludge) are investigated in detail. Combustion of Biomass in Europe significantly varies between Member States. At European level bio waste is usually collected with municipal solid wastes and is contained in shares between 30%-40% in MSW (even up to 60%), [Green Paper 2008]. Therefore, no extra mass and waste flow for PeCB from biomass incineration has been established. The corresponding amounts are contained in the MSWI mass flow.

Compared to incineration and combustion processes, industrial processes, where PeCB can be produced as by-product are of comparatively low relevance. Nevertheless, it has to be mentioned that most industrial processes use incineration or combustion processes, e.g. industrial production of steel and iron, which according to EUROSTAT data approximately uses more than 50% of the coal finally consumed within Europe. For this reason the industrial sectors metallurgy and energy have been set in close relation to coal combustion processes and are therefore included in the PeCB mass and waste flow resulting from coal combustion. An exception to this is the chemical industry, in which PeCB may be produced as by-product in several chlorination processes, e.g. in the production of solvents. However, according to Bailey et al. the share of PeCB unintentionally produced is very low in comparison to incineration and combustion processes [Bailey et al. 2009]. Environment Canada does not mention chemical manufacturing as source of PeCB releases [Environment Canada 2005]. Releases of PeCB from application of chlorinated solvents are of minor importance [EuroChlor 2007]. Mass flows resulting from the chemical industry are not specifically investigated.

6.4.8 Substance Flow of PeCB

Based on the available data the project team has developed the following initial substance flow diagram for PeCB. Since there is no intentional use or production of PeCB within the EU, the substance flow diagram is related to unintentional production and related releases of PeCB within the EU.

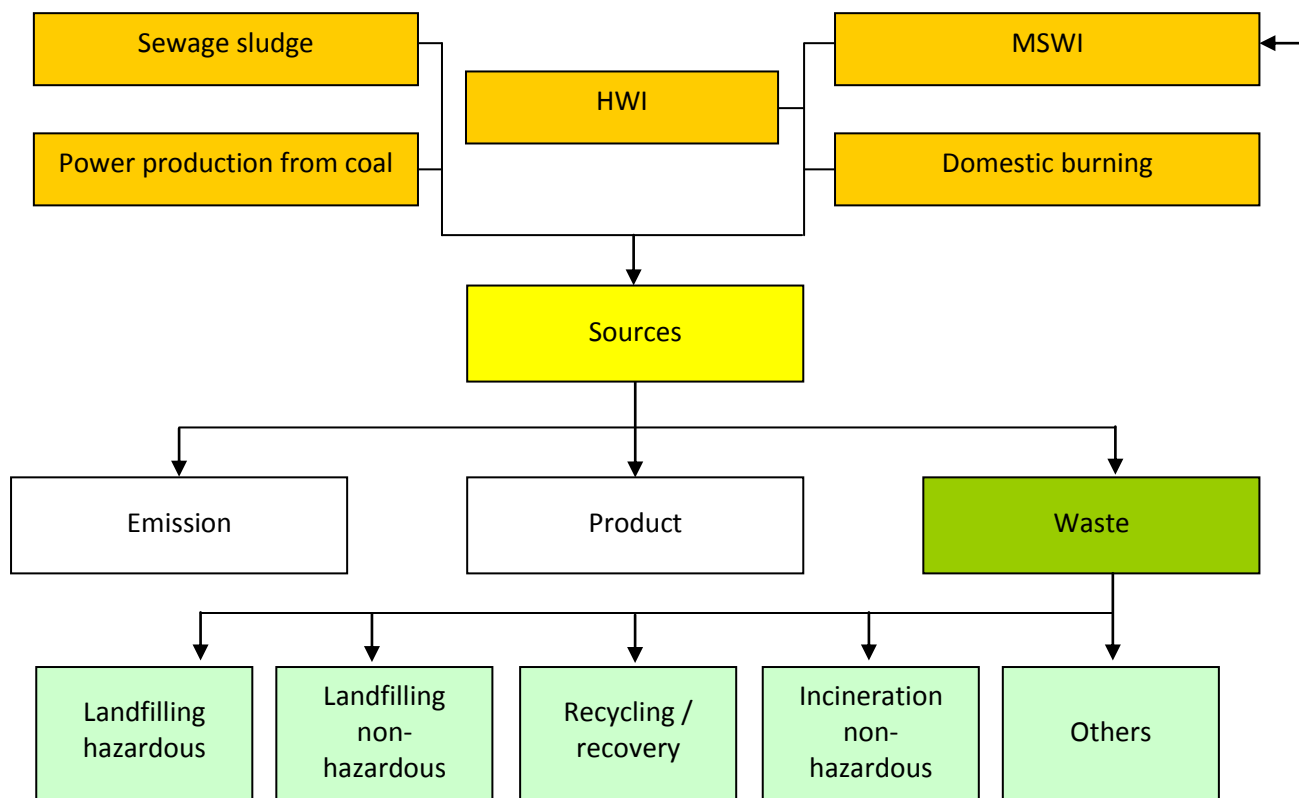


Figure 6-52: Overview of initial PeCB substance flow

Figure 6-52 gives an initial overview of the major sources for PeCB in the EU. At this stage of the project the main purpose of this figure is to identify and summarise relevant unintentional production processes and related releases of PeCB in order to enable an assessment of the relevance of the mass flows of materials containing PeCB.

6.4.9 Municipal solid waste incineration (MSWI)

A) Background

The objective of waste incineration is to generate energy and to treat wastes so as to reduce their volume and hazard, whilst capturing (and thus concentrating) or destroying potentially harmful substances that may be released from incineration. It is unlikely that municipal solid waste as input contains significant amounts of PeCB, but it has been shown that residues resulting from the incineration process contain relevant amounts of PeCB.

Operating conditions are laid down in Article 6 of the Waste Incineration Directive²⁴. The relevant processes and technical aspects in waste incineration are described in detail in the corresponding BAT reference document [BREF WI 2006].

Basically, incineration is the chemical reaction (oxidation) of a combustible material with oxygen. Waste is generally a highly heterogeneous material, consisting essentially of organic substances, minerals, metals and water. During incineration, flue-gases are created that will contain the majority of the available fuel energy as heat. The organic fuel substances in the waste will burn when they have reached the necessary ignition temperature and come into contact with oxygen. The actual combustion process takes place in the gas phase within fractions of seconds and simultaneously releases energy. If the calorific value of the waste and oxygen supply is sufficient, it can lead to a thermal chain reaction and self-supporting combustion, i.e. there is no need for the addition of other fuels.

The main stages of incineration are:

1. drying and degassing – here, volatile content is evolved (e.g. hydrocarbons and water) at temperatures generally between 100 and 300 °C. The drying and degassing process does not require any oxidising agent and only depend on the supplied heat.
2. pyrolysis and gasification – pyrolysis is the further decomposition of organic substances in the absence of an oxidising agent at approx. 400 – 700 °C. Gasification of the carbonaceous residues is the reaction of the residues with water vapour and CO₂ at temperatures, typically between 700 and 1000 °C, but can occur at even higher temperatures. Thus, solid organic matter is transferred to the gaseous phase. In addition to the temperature, water and steam support this reaction.
3. oxidation - the combustible gases created in the previous stages are oxidised, depending on the selected incineration method, at flue-gas temperatures generally between 850 and 1150 °C according to the Waste Incineration Directive.

These individual stages generally overlap, meaning that spatial and temporal separation of these stages during waste incineration may only be possible to a limited extent. Indeed the processes partly occur in

²⁴ DIRECTIVE 2000/76/EC OF THE EUROPEAN PARLIAMENT AND OF THE COUNCIL of 4 December 2000 on the incineration of waste

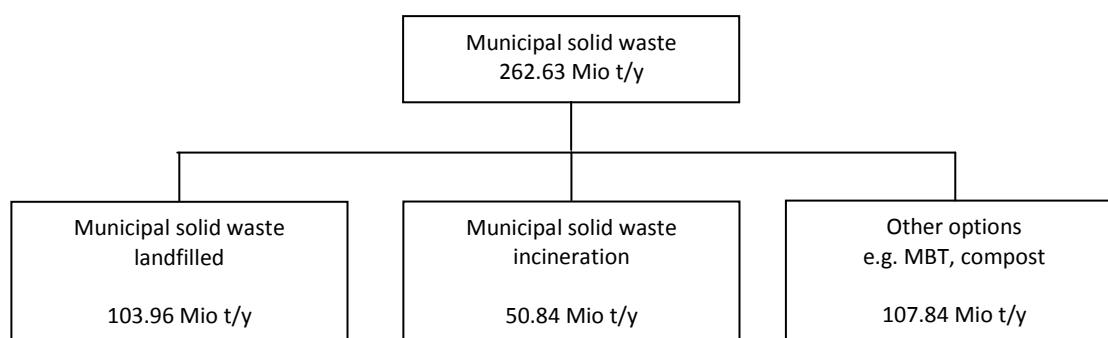
parallel and influence each other. It is possible, using in-furnace technical measures, to influence these processes so as to minimise polluting emissions.

Such measures include furnace design, air distribution and process control. With municipal solid waste incineration the main constituents of the flue-gas are: water vapour, nitrogen, carbon dioxide and oxygen. Depending on the composition of the material incinerated and on the operating conditions, small amounts of heavy metal compounds remain (among others) and CO, HCl, HF, NO_x, SO₂, VOCs, PCDD/PCDF, PCB and chlorinated benzenes such as PeCB are formed in off gases.

With appropriate combustion conditions organic compounds are almost completely destroyed during incineration, volatile heavy metals and inorganic compounds (e.g. salts) are totally or partly evaporated. These substances are transferred from the input waste to the flue-gas and are adsorbed to the fly ash [BiPRO 2005].

It has to be noted that municipal solid waste in Europe is either treated by municipal solid waste incineration or sent to other management options such as non-hazardous waste landfills, mechanical biological treatment (MBT) or composting. Given that the contamination level of MSW is close to zero (no production and use of PeCB within the EU), the PeCB input via handling, landfilling, composting and MBT treatment of MSW can be regarded as negligible. Only emissions due to incineration are to be taken into account

Based on the average of waste incineration in EU 27 the following average flow for municipal solid waste has been calculated for EU 27:



Approximately 20 % of municipal waste is incinerated, 40 % is landfilled and 40 % is treated with other options [EUROSTAT 2010].

In the following, only the path of municipal solid waste incineration is investigated.

B) Process input (raw material) and output (waste, air, water, product)

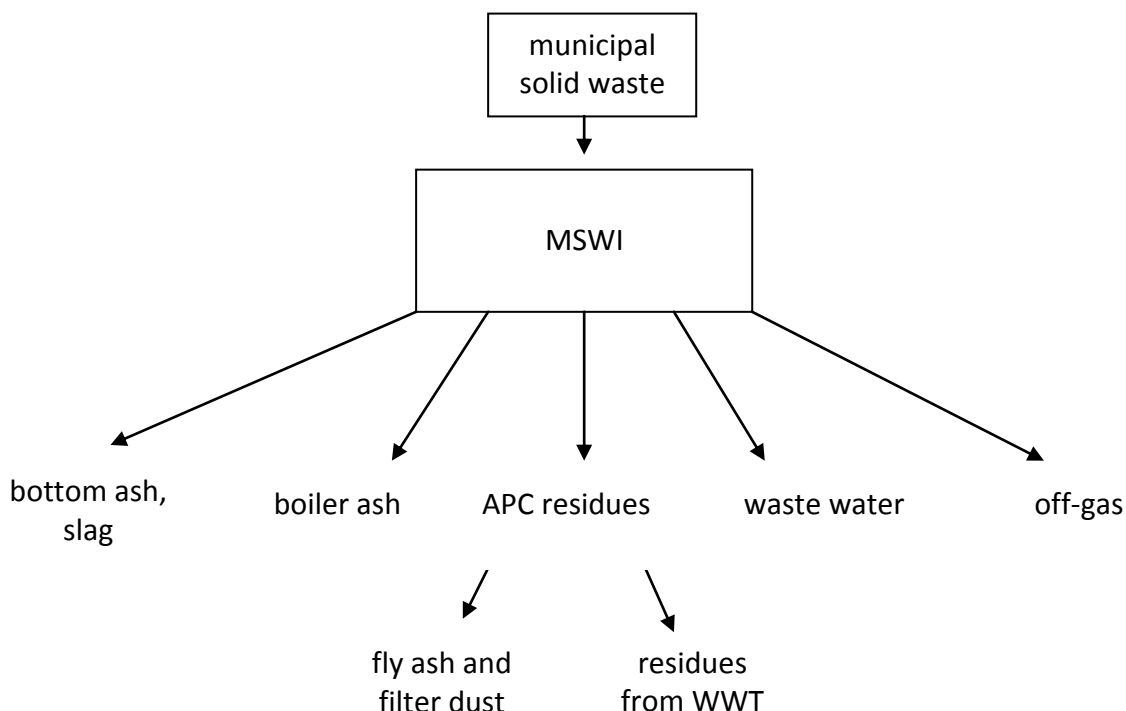


Figure 6-53: Relevant input and output from municipal solid waste incineration

Input:

- Municipal solid waste. The precise determination of the composition of MSW for incineration is difficult. The composition of municipal waste is treated in sections 1.5 and 2.2.1 of the BAT reference document [BREF WI 2006].

Output:

- Bottom ash or slag (EWC 190111*, 190112) is a solid mineral residue that remains from the incineration process. It results from grate incineration of waste. Because of its large volume, this is an important type of residue.
- Boiler ash (EWC 190115*, 190116) is collected in the boiler of waste incineration plants.
- Fly ash (EWC 190113*, 190114) or dust and additives is a fraction of solid mineral residue (APC residues; EWC 190107*) that remains from the incineration and that ends up in solid residues from dust removal including solid additives such as lime and activated-carbon.
- Residues from flue gas treatment (FGT) also contain fly ash. Residues from FGT are a mixture of calcium and/or sodium salts, mainly chlorides and sulphites/sulphates. There are also some fluorides and non-reacted reagent chemicals (e.g. lime or sodium carbonate). Residues from FGT

also include filter cakes²⁵ (EWC 190105*) from the physico-chemical treatment of waste water from wet flue-gas treatment. This material is characterised by very high heavy metals content, but can also include insoluble salts and gypsum. These residues may contain PeCB and are therefore sometimes pre-treated before landfilling.

- Treated waste water is the remaining fluid residue, which is treated in waste water treatment plants.
- Off-gas is the remaining gaseous end-product, which is released to the environment (also stack-gas).

In practice boiler ash and APC residues from FGT are often mixed together and disposed of as one waste stream. In the PeCB flow the fly ash and residues from FGT are assessed as one material flow because in literature, residues from dust removal steps and specific fractions of residues from FGT cannot always be differentiated unambiguously. At least fly ash contains relevant amounts of PeCB.

C) Generation factors for product/waste and other output (air, water)

The proportions of solid residue vary greatly with the waste type and detailed process design. The following figures indicate typical amounts reported for outputs related to the incineration of 1 tonne of municipal solid waste [BiPRO 2005]:

Bottom ash	233 kg/t MW (range from 185 to 275)
Boiler ash	3 kg/t MW (range from 2 to 4)
APC residues:	37.75 kg/t MW (range from 10 to 80), thereof 22.25 kg/t MW fly ash and filter dust and 15.50 kg/t MW remaining APC residues

D) Contamination data for output (and input)

Bottom ash	It has not been reported in literature that PeCB is contained in bottom ash.
Boiler ash	It has not been reported in literature that PeCB is contained in boiler ash.
APC residues:	The most significant amounts of PeCB have been found in fly-ash samples. Korenkova et al. reported that PeCB concentration in a fly-ash sample taken from an Italian municipal solid waste incinerator amounted to 31 ng/g (31 ppb) fly-ash [IPEN 2007]; in Norway fly-ash samples from an municipal solid waste incinerator contained 50 ng/g (50 ppb) [Viau et al. 1983], in UK 11.5 ng/g (11.5 ppb) [Balampanis et al.

²⁵ In literature the term “filter cake” is used as well for filter cakes from dust removal via fabric filter as well for filter cakes from wet FGT systems.

2010], in Sweden 240 ng/g (240 ppb) [Öberg et al. 2007] and in Germany 39.6 ng/g (39.6 ppb) [Schreiner et al. 1986]. An average contamination factor of **74.4 ng** (74.4 ppb) **PeCB/g fly ash** has been used for the calculation of the mass flows.

Off gas

Jay et al. reported that off-gas released by a municipal solid waste incinerator still included PeCB in an amount of **2.90 mg PeCB/t MSW** burned [Jay et al. 1995].

E) Activity data

Data on municipal solid waste are mainly derived from European statistics [EUROSTAT 2010]. The following table shows amounts of municipal solid waste incinerated in Europe according to the EUROSTAT database:

Table 6-35: Quantities of MSW generated in EU 27 and amounts of MSW incinerated in EU 27 in 2008

Country	MSW generated [mio t/y]	MSW incinerated [mio t]	Reference year	Reference for amount of MSW incinerated
AT	5.03	1.37	2008	[EUROSTAT 2010]
BE	5.34	1.79	2008	[EUROSTAT 2010]
BG	3.53	0.00	2008	[EUROSTAT 2010]
CY	0.61	0.00	2008	[EUROSTAT 2010]
CZ	3.22	0.36	2008	[EUROSTAT 2010]
DE	47.53	15.79	2008	[EUROSTAT 2010]
DK	4.44	2.40	2008	[EUROSTAT 2010]
EE	0.69	0.00	2008	[EUROSTAT 2010]
ES	26.44	2.44	2008	[EUROSTAT 2010]
FI	2.79	0.48	2008	[EUROSTAT 2010]
FR	35.14	11.13	2008	[EUROSTAT 2010]
GB	35.03	3.41	2008	[EUROSTAT 2010]
GR	5.12	0.00	2008	[EUROSTAT 2010]
HU	4.54	0.39	2008	[EUROSTAT 2010]
IE	3.27	0.08	2008	[EUROSTAT 2010]
IT	33.85	4.16	2008	[EUROSTAT 2010]
LT	1.35	0.00	2008	[EUROSTAT 2010]
LU	0.35	0.12	2008	[EUROSTAT 2010]
LV	0.74	0.00	2008	[EUROSTAT 2010]
MT	0.29	0.00	2008	[EUROSTAT 2010]
NL	10.31	3.37	2008	[EUROSTAT 2010]
PL	12.21	0.08	2008	[EUROSTAT 2010]
PT	5.07	0.97	2008	[EUROSTAT 2010]
RO	8.20	0.00	2008	[EUROSTAT 2010]
SE	4.81	2.34	2008	[EUROSTAT 2010]
SI	0.94	0.01	2008	[EUROSTAT 2010]
SK	1.78	0.16	2008	[EUROSTAT 2010]
EU-27	262.63	50.84		

F) Waste treatment

The high mineral content of incineration ash residues can make them suitable for use as road or other construction material. Use is possible if the material complies with a set of environmental and technical criteria. This requires an optimisation of the ash quality through primary (e.g. optimised combustion process) or secondary measures. Due to its large production volume, treatment for recycling is mainly applied to MSW bottom ash. Bottom ash use is promoted in the Netherlands (>90% used), Denmark (90%), Germany (80%), France (>70%). Recovery of bottom ash in Belgium and the UK is 21% [BiPRO 2005]. Based on these figures it is assumed that more than 50% of bottom ash is used for construction. The remainder is usually directly land filled or land filled after certain treatment. Bottom ash is not mixed with APC residues and is used as secondary raw material in a number of processes. Filter and boiler ash treatment is performed in only a few installations in Europe.

According to the BAT reference document on waste incineration [BREF WI 2006], the following measures are available for pre-treatment improving waste characteristics prior to disposal or recovery:

- solidification
- thermal treatment (vitrification, melting, sintering)
- extraction and separation
- chemical stabilisation
- other methods

The treatment shall improve the leaching properties or reduce the contamination level in the residues. After pre-treatment the residues are normally disposed of at landfills for hazardous waste or are stored underground.

The most important pre-treatment method for wastes intended for landfilling is solidification. Solidification can principally be achieved by addition of several, mostly inorganic, binder reagents (cement, lime and other pozzolanic materials such as fly ash from coal fired power plants, blast furnace bottom ash or cement kiln dust) or organic binders such as bitumen/asphalt, paraffin and polyethylene. However, immobilisation in cement blocks is the prevalent technique.

No information is available concerning the relevance of the other different treatment options. It can be expected that the amount of PeCB contained in the FGT residues is finally landfilled as hazardous waste or stored underground.

In the Netherlands about 36% of fly ash is recovered in asphalt production [BiPRO 2005].

G) PeCB flow

The available data and assumptions allow an estimation of PeCB emissions to air and discharge to waste via solid residues resulting from MSWI. The following table shows a country specific estimation on emitted amounts:

Table 6-36: PeCB mass flow for MSWI in EU 27

Country	MSW incinerated [mio t]	Air [kg/y]	Fly ash and filter dust [kg/y]
AT	1.37	3.96	2.26
BE	1.79	5.18	2.96
BG	0.00	0.00	0.00
CY	0.00	0.00	0.00
CZ	0.36	1.04	0.59
DE	15.79	45.78	26.14
DK	2.40	6.95	3.97
EE	0.00	0.00	0.00
ES	2.44	7.07	4.04
FI	0.48	1.40	0.80
FR	11.13	32.28	18.43
GB	3.41	9.89	5.65
GR	0.00	0.00	0.00
HU	0.39	1.13	0.65
IE	0.08	0.25	0.14
IT	4.16	12.07	6.89
LT	0.00	0.00	0.00
LU	0.12	0.36	0.21
LV	0.00	0.01	0.00
MT	0.00	0.00	0.00
NL	3.37	9.76	5.57
PL	0.08	0.22	0.13
PT	0.97	2.81	1.60
RO	0.00	0.00	0.00
SE	2.34	6.77	3.87
SI	0.01	0.04	0.02
SK	0.16	0.46	0.26
EU-27	50.84	147.42	84.17

At a European scale the estimated emissions of PeCB due to MSWI amount to 231.60 kg/y. Thereof 147.42 kg/y are emitted to air and 84.17 kg/y are emitted as fly ash to waste. Thereof approximately 36% have been assumed to be recovered (e.g. construction) in the Netherlands, i.e. 2.00 kg/y, and the rest (82.17 kg/y) disposed off in landfills as hazardous waste. The following PeCB flow results:

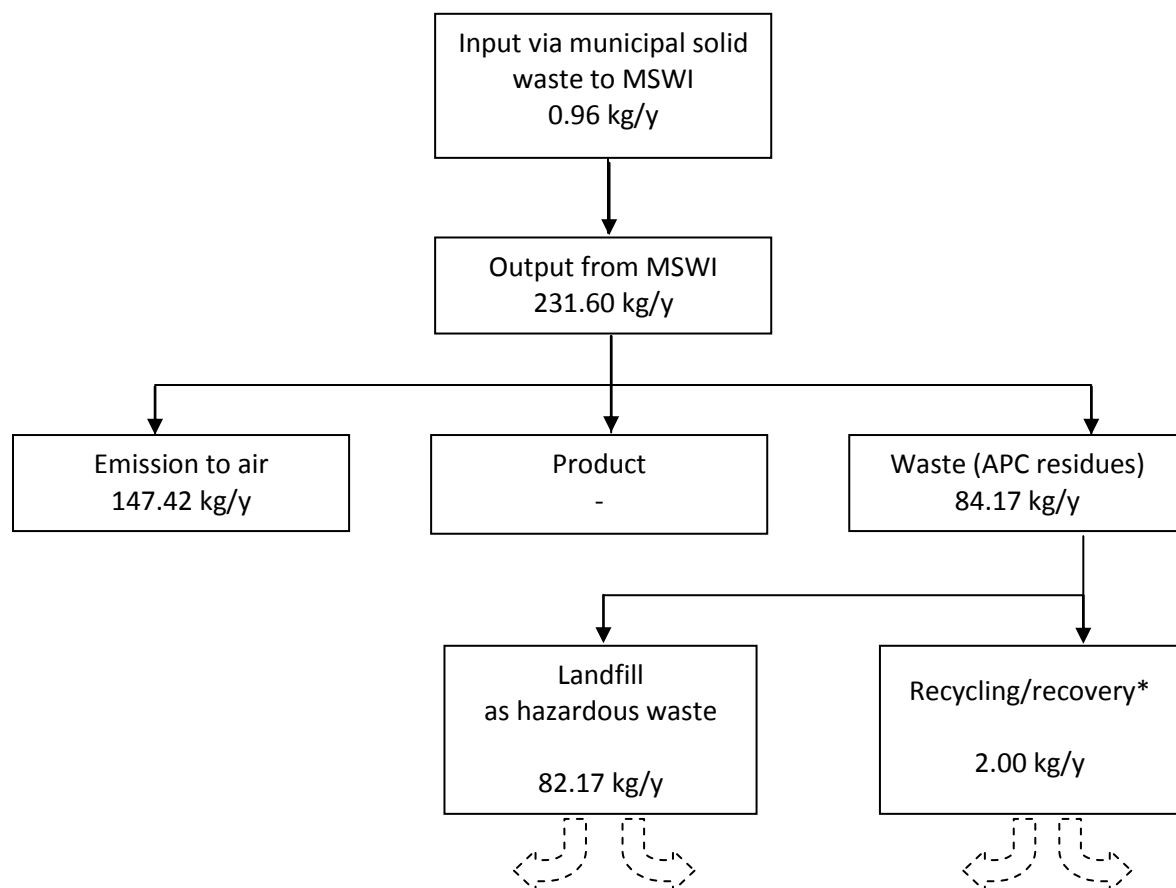


Figure 6-54: Assessment of PeCB flow related to MSWI

H) MSWI waste flow for PeCB

In order to illustrate the amounts of residues generated in EU 27 and the importance of currently used disposal/recovery operations which are of high relevance in the evaluation of impacts of proposed limit values, a waste flow has been drawn up in the following figure. Mean contaminations and concentration ranges for each of the residues have been included for better interpretation as far as available in literature.

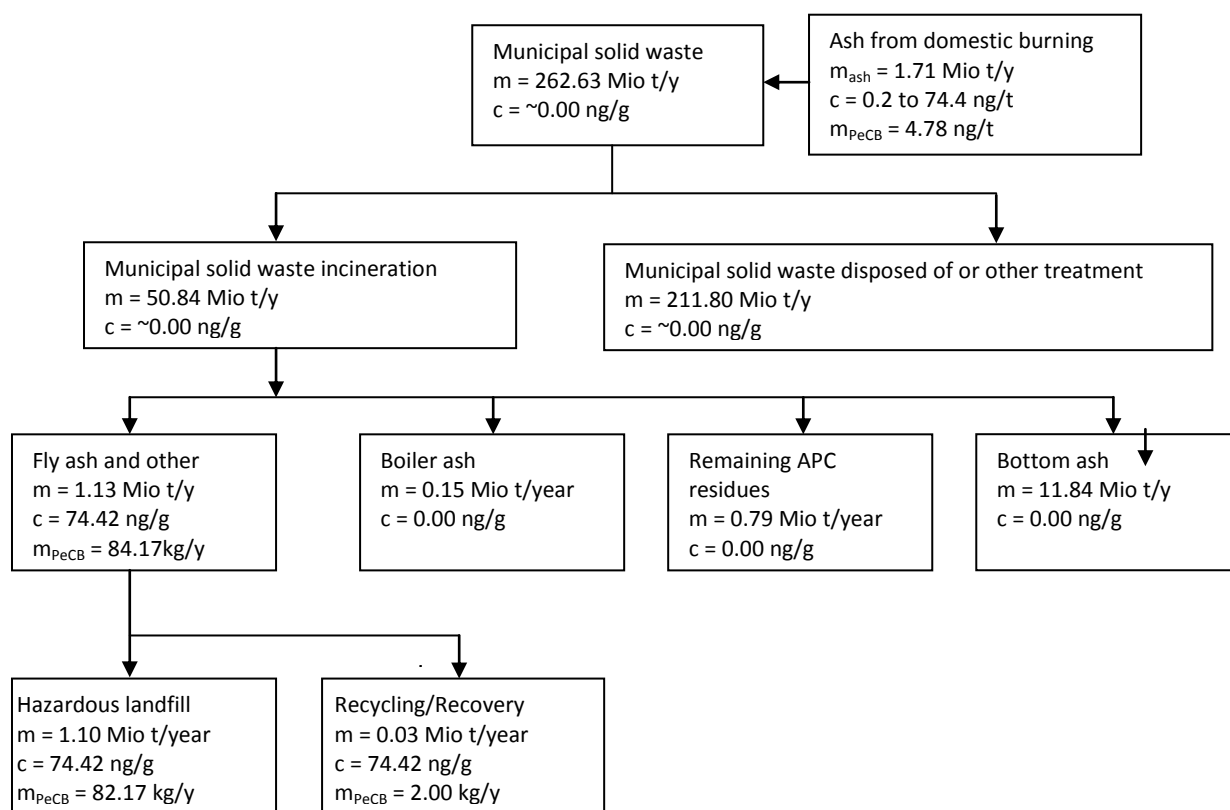


Figure 6-55: Detailed waste flow for MSWI in EU 27

6.4.10 Hazardous waste incineration (HWI)

A) Background

In contrast to municipal solid waste incineration, hazardous waste incineration is usually carried out in rotary kiln incinerators or in static furnaces. The relevant processes and aspects of waste incineration are described in detail in the corresponding BAT reference document [BREF WI 2006].

B) (B) Process input and output

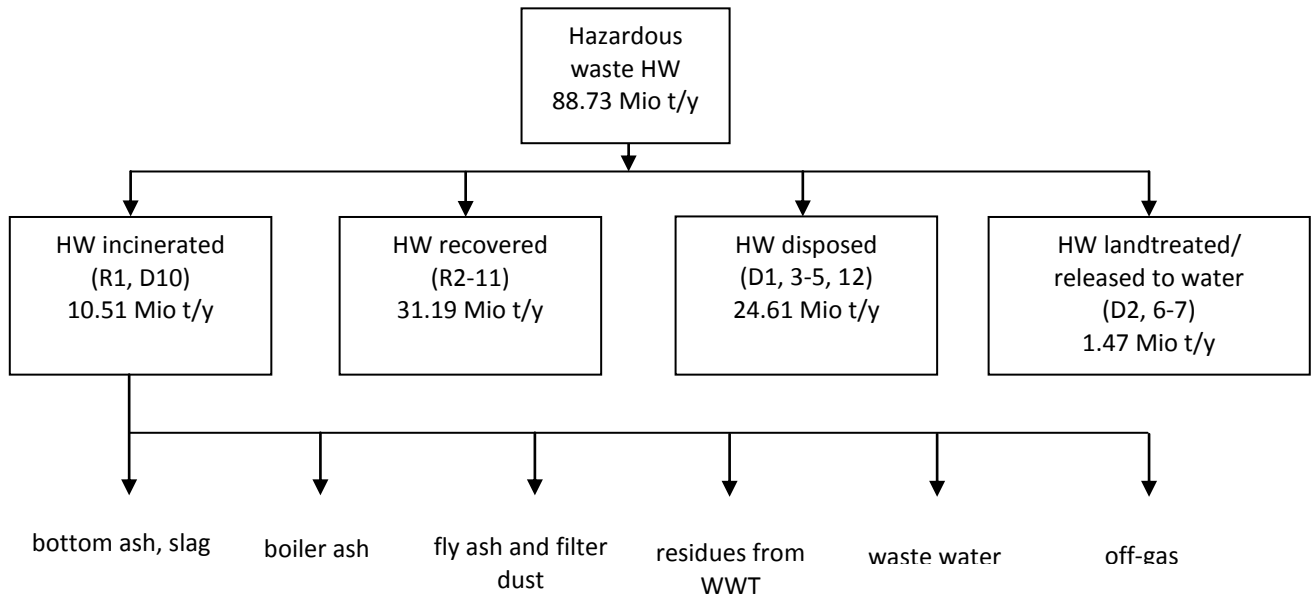


Figure 6-56: Relevant input and output of hazardous waste incineration and relevance of other hazardous waste management options (values [EUROSTAT 2010])

Input:

- Hazardous waste

Output:

The output from hazardous waste incineration is not fundamentally different from that from municipal solid waste incineration with generally higher content of heavy metals. The outputs are

- Bottom ash and slag : 190111*, 190112
- Boiler ash: 190115*, 190116
- Fly ash or dust and other residues from flue gas treatment (FGT): 190113*, 190114, 190107*
- Treated waste water

- Treated off-gas

C) Generation factors

The proportions of solid residue vary greatly according to the waste type and detailed process design. According to information compiled in the BREF document on waste incineration or provided from EURITS the following figures indicate typical amounts reported for outputs related to the incineration of 1 tonne of hazardous waste [BiPRO 2005]:

Bottom ash ~ 140 kg/t HW (range from 83 to 246)

Boiler ash ~ 29 kg/t HW (range from 12 to 68)

Fly ash and residues from FGT: ~45 kg/t HW (range from 20 to 109)

D) Contamination data

Bottom ash No information has been reported that PeCB is contained in bottom ash.

Boiler ash No information has been reported that PeCB is contained in boiler ash.

Fly ash and residues from FGT: Reported values in fly ash amount to an average of 9.57 ng PeCB/g (9.57 ppb) fly ash [Schreiner et al. 1986].

Off gas It has been reported that on average 7.00 mg PeCB are found in off-gas per tonne HW incinerated [Kaune et al. 1994].

E) Activity data

The following table shows amounts of hazardous waste incinerated in Europe according to EUROSTAT statistics:

Table 6-37: Amount of hazardous waste incinerated in EU 27 in 2006

Country	HW generated [mio t/y]	HW incinerated [mio t] (R1+D10)	Reference year	Reference for amount of MSW incinerated
AT	0.96	0.15	2006	[EUROSTAT 2010]
BE	4.04	0.60	2006	[EUROSTAT 2010]
BG	0.79	0.08	2006	[EUROSTAT 2010]
CY	0.05	0.00	2006	[EUROSTAT 2010]
CZ	1.31	0.10	2006	[EUROSTAT 2010]
DE	21.71	3.17	2006	[EUROSTAT 2010]
DK	0.49	0.08	2006	[EUROSTAT 2010]
EE	6.62	0.03	2006	[EUROSTAT 2010]
ES	4.03	0.41	2006	[EUROSTAT 2010]
FI	2.71	0.17	2006	[EUROSTAT 2010]
FR	9.62	2.85	2006	[EUROSTAT 2010]

Country	HW generated [mio t/y]	HW incinerated [mio t] (R1+D10)	Reference year	Reference for amount of MSW incinerated
GB	8.45	0.45	2006	[EUROSTAT 2010]
GR	0.27	0.02	2006	[EUROSTAT 2010]
HU	1.30	0.06	2006	[EUROSTAT 2010]
IE	0.71	0.04	2006	[EUROSTAT 2010]
IT	7.46	0.64	2006	[EUROSTAT 2010]
LT	0.13	0.01	2006	[EUROSTAT 2010]
LU	0.23	0.03	2006	[EUROSTAT 2010]
LV	0.07	0.01	2006	[EUROSTAT 2010]
MT	0.05	0.00	2006	[EUROSTAT 2010]
NL	4.95	0.81	2006	[EUROSTAT 2010]
PL	2.38	0.16	2006	[EUROSTAT 2010]
PT	6.06	0.13	2006	[EUROSTAT 2010]
RO	1.04	0.10	2006	[EUROSTAT 2010]
SE	2.65	0.31	2006	[EUROSTAT 2010]
SI	0.12	0.03	2006	[EUROSTAT 2010]
SK	0.53	0.06	2006	[EUROSTAT 2010]
EU-27	88.73	10.51		

F) Waste treatment

Treatment and recycling of solid residues from HWI follows the same principles as treatment of residues from MSWI. More than 50% of bottom ash is used for construction purposes. Boiler ash and fly ash and other reaction products are usually mixed together and the bulk of these FGT residues is either directly landfilled or landfilled after further treatment. Slag is usually disposed of in landfills without further treatment or it is recycled.

G) PeCB flow

The available data and assumptions allow an estimation of PeCB emissions to air and to waste via exhaust air and solid residues resulting from HWI. The following table shows a country specific estimation on emitted amounts:

Table 6-38: PeCB mass flow from hazardous waste incineration (HWI) in EU 27 in 2006

Country	HW incinerated [mio t]	Air [kg/y]	Fly ash and filter dust [kg/y]
AT	0.15	1.07	0.07
BE	0.60	4.22	0.26
BG	0.08	0.58	0.04
CY	0.00	0.01	0.00
CZ	0.10	0.73	0.05
DE	3.17	22.16	1.36
DK	0.08	0.59	0.04
EE	0.03	0.23	0.01
ES	0.41	2.85	0.18
FI	0.17	1.20	0.07
FR	2.85	19.96	1.23
GB	0.45	3.18	0.20
GR	0.02	0.15	0.01

Country	HW incinerated [mio t]	Air [kg/y]	Fly ash and filter dust [kg/y]
HU	0.06	0.44	0.03
IE	0.04	0.31	0.02
IT	0.64	4.48	0.28
LT	0.01	0.06	0.00
LU	0.03	0.18	0.01
LV	0.01	0.05	0.00
MT	0.00	0.00	0.00
NL	0.81	5.64	0.35
PL	0.16	1.12	0.07
PT	0.13	0.91	0.06
RO	0.10	0.67	0.04
SE	0.31	2.19	0.13
SI	0.03	0.20	0.01
SK	0.06	0.40	0.02
EU-27	10.51	73.56	4.53

At a European scale the estimated emissions of PeCB due to HWI in 2006 amount to 78.08 kg/y. Thereof 73.56 kg/y (7 ppb of the incinerated hazardous waste) are emitted to air and 4.53 kg/y (The ash generation factor is 4.5% and the ash is contamination with 9.57 ppb) are emitted as fly ash to waste. Thereof approximately 36% have been assumed to be recovered (e.g. construction) in the Netherlands, i.e. 0.12 kg/y, and the rest (4.40 kg/y) disposed off in landfills as hazardous waste with no consequence or low relevance for the ecosystem. The following PeCB flow results:

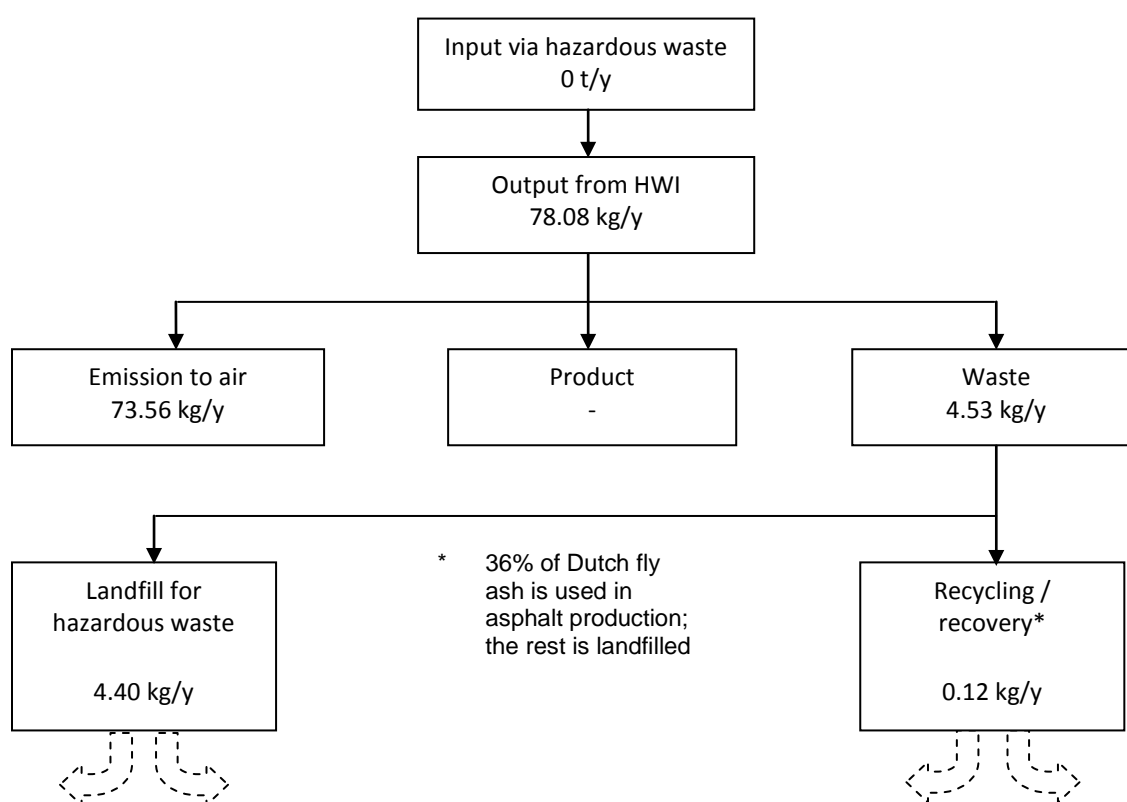


Figure 6-57: Assessment of the PeCB flow related to hazardous waste incineration

H) HWI waste flow

In order to illustrate the amounts of residues generated in EU 27 and the importance of currently used disposal/recovery operations which are of high relevance in the evaluation of impacts of proposed limit values, a waste flow has been drawn up in the following figure. Mean contaminations and concentration ranges for each of the residues have been included for better interpretation.

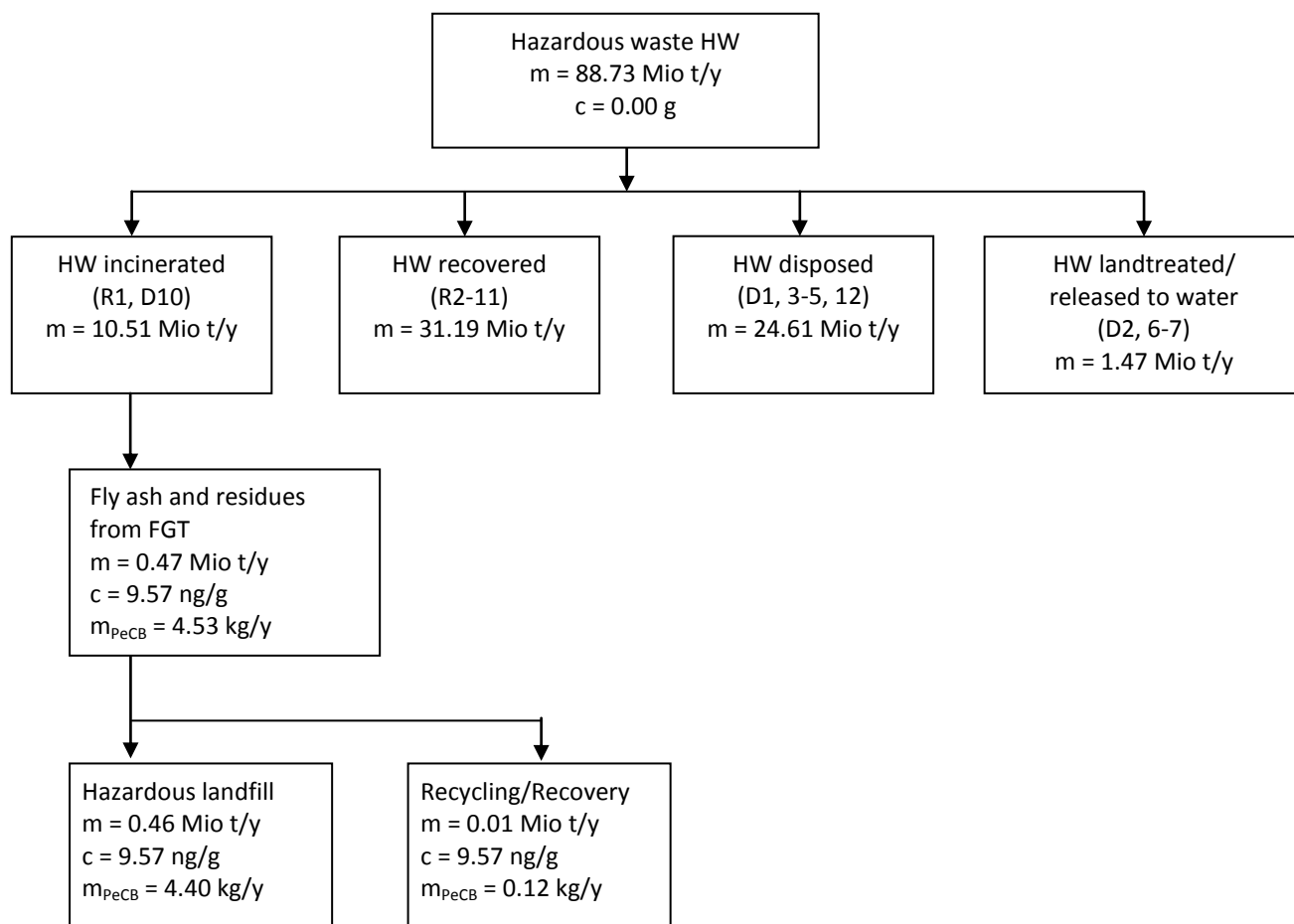


Figure 6-58: Detailed waste flow for HWI in EU 27

6.4.11 Coal combustion

A) Background

The relevant processes and aspects related to the combustion of conventional fuels (such as coal, lignite, peat, liquid and gaseous fuels) including co-combustion of waste and recovered fuel are described in detail in the corresponding BAT reference document [BREF LCP 2004].

B) Process input and output

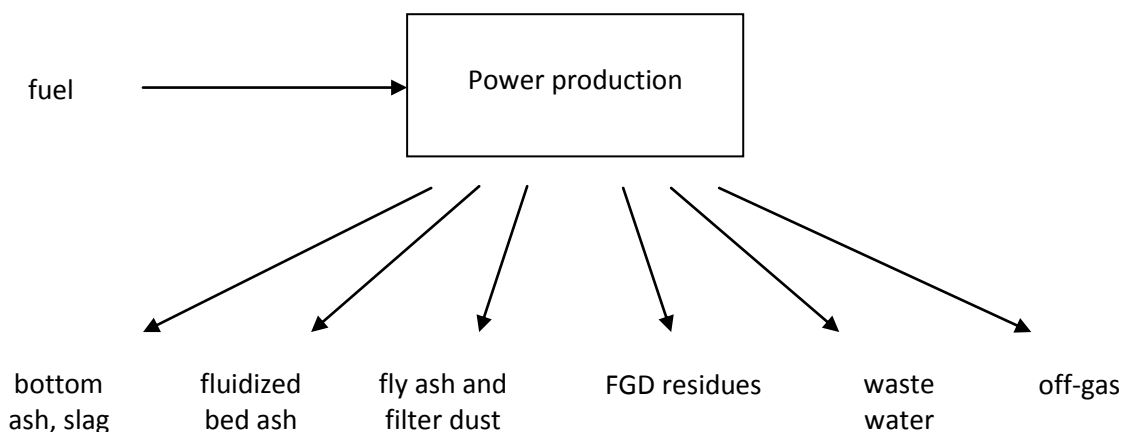


Figure 6-59: Relevant input and output of power production

Input:

- Fuel

Output:

Residues directly related to the process of combustion of fossil fuels are ashes (fly and bottom ash) and residues that are generated by the desulphurisation of flue-gases

- Bottom ash and/or boiler slag (EWC 100101, 100115): Bottom ash is non-combustible material that settles to the bottom of the boiler and remains in the form of unconsolidated ash. If combustion temperatures exceed the ash fusion temperature, the ash remains as slag in a molten state until it is drained from the bottom of the boiler as boiler slag
- Fluidised bed ash: The operation of a fluidised bed combustion installation for a solid fuel such as coal but also for burning biomass and peat is related to the generation of ash, which is a composition of spent bed material and fuel ash. Ash is removed from the bottom of the fluidised bed combustion chamber

- Fly ash (EWC 100102, 100116*, 100117): Fly ash represents the part of the non-combustible material that is carried out of the boiler along with the flue-gas. Fly ash is removed from the flue gas stream.
- Flue-gas desulphurisation (FGD) residues (EWC 100105, 100107) and by-products: Fossil fuels such as coal, peat and oil contain varying amounts of sulphur. To avoid high emissions of sulphur dioxide to the atmosphere, large combustion plants (in particular plants above 100 MW_{th}²⁶ are usually equipped with flue-gas desulphurisation plants (FGD). The different desulphurisation techniques currently in use result in the generation of a number of residues and by-products. Wet lime/limestone scrubbers, for instance, produce gypsum as a by-product, whereas dry scrubber systems produce a mixture of non-reacted sorbent (e.g. lime, limestone, sodium carbonates, calcium carbonates), sulphur salts and fly ash as residue.
- Treated off-gas

C) Generation factors

The proportions of solid residue vary greatly in particular according to the fuel type used and the process design. The largest amount of ash is generated by the combustion of coal and lignite, followed by the combustion of peat and biomass, whereas gas-fired facilities generate very low quantities of ash. The amount of ash generated from a liquid fuel fired facility is higher than from a gas-fired boiler, but compared to the amount of ash from coal combustion, the quantities are rather small. The PeCB flow assessments are restricted to emissions from combustion of coal, lignite and derivatives.

The following figures indicate typical amounts reported for outputs related to the incineration of 1 tonne of coal [BiPRO 2005]:

Off gas	~ 14,000 Nm ³ /t coal
Bottom ash	16 kg/t coal
Fly ash and residues from FGT:	84 kg/t coal
Ash from coal incineration	60 kg/t coal

An average of 80 kg ash (no differentiation of ashes) per tonne solid fuel has been taken as the basis for the calculation of the PeCB flow into waste.

D) Contamination data

Ash from coal incineration	contamination values for fly ashes from coal incineration range from 0.0004 to 7.0 ng/g. The PeCB flow is calculated on the basis of an estimated average of 2.40 ng/g (2.4 ppb) [Schreiner et al. 1986].
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²⁶ megawatts thermal energy

Flue gas

An average emission level of PeCB in flue gas from combustion of different fuels have been averaged by Bailey et al. to amount to 2.45 mg/t coal combusted [Bailey et al 2009].

E) Activity data

The following table shows the gross inland consumption of solid fuels consisting of coal, lignite and their derivatives according to EUROSTAT statistics for EU 27 in 2008. Domestically consumed solid fuels have been abstracted from these values and have been investigated in the chapter concerning domestic burning.

Table 6-39: Annual consumption of solid fuels for power production in the EU in 2008

Country	Gross inland consumption [Mio t/y]	Consumption of households [Mio t/y]	Final consumption of solid fuels [Mio t/y]
AT	5.32	0.13	5.19
BE	6.82	0.26	6.56
BG	33.20	0.40	32.80
CY	0.04	0.00	0.04
CZ	54.53	1.30	53.23
DE	242.03	1.75	240.28
DK	6.87	0.00	6.87
EE	16.05	0.02	16.02
ES	25.90	0.30	25.60
FI	13.45	0.05	13.40
FR	19.55	0.55	19.01
GB	58.67	0.99	57.68
GR	65.20	0.04	65.16
HU	11.52	0.41	11.11
IE	6.41	1.13	5.29
IT	24.68	0.01	24.67
LT	0.39	0.09	0.29
LU	0.11	0.00	0.11
LV	0.18	0.03	0.15
MT	0.00	0.00	0.00
NL	12.90	0.01	12.88
PL	135.48	9.29	126.19
PT	4.16	0.00	4.16
RO	40.39	0.08	40.31
SE	4.20	0.00	4.20
SI	5.24	0.00	5.24
SK	8.10	0.20	7.90
EU 27	801.37	17.03	784.34

F) Waste treatment

Ash and FGT residues are by far the largest quantities of waste from power production. These residues are partly discharged to a landfill or can be used for different purposes such as an additive in cement

and concrete production; an aggregate in concrete, asphalt, mine reclamation or waste stabilisation; and as an ingredient in many other products. Gypsum, a by-product from the desulphurisation plant, is largely used in the production of gypsum board [BREF LPC 2004].

Solid residues from power production are used in construction industry and underground (~56%), restoration of open cast mines, quarries and pits (~33%), landfill (~9%) and temporary stockpiles (~2%), [BREF LPC 2004]. The fate of the different fractions in the year 1999 for EU 15 is shown in the following table:

Table 6-40: Treatment of solid residues from power production in EU 15 in 1999 [BREF LPC 2004]

Residue	Construction, underground mining	Restoration of mines, quarries, pits	Land filling	Temporary stockpile
bottom ash	~42	~38	~19	~1
boiler slag	100			
fluidised bed ash	~43	~40	~17	
fly ash and APC residues	~45	~43	~10	~2
gypsum and SDA product	~90	5-10	1-2	

Being well aware of the fact that construction is a recycling/recovery treatment but underground mining has to be considered a disposal treatment, these activities should be separated. As this is not possible on the basis of the available data background, construction and underground mining as well as restoration of mines, quarries and pits is allocated to the treatment category “recycling, recovery”. Residues that are land-filled are either disposed as hazardous waste or as non-hazardous waste (assumption: relation hazardous: non-hazardous ~ 1: 10).

G) PeCB flow for power production from coal and lignite

The available data and assumptions allow an estimation of PeCB emissions to solid residues (general estimation for ashes: bottom ash, fluidised bed ash, fly ash including FGT residues) resulting from power production from coal and lignite. The following table shows a country specific estimation on emitted amounts in the year 2008:

Table 6-41: PeCB mass flow from power production via coal, lignite and derivatives in EU 27 in 2008

Country	Total solid fuels [Mio t/y]	PeCB emission to air [kg/y]	PeCB discharge to fly ashes [kg/y]
AT	5.19	12.72	1.05
BE	6.56	16.08	1.32
BG	32.80	80.35	6.62
CY	0.04	0.10	0.01
CZ	53.23	130.42	10.74
DE	240.28	588.68	48.47
DK	6.87	16.84	1.39
EE	16.02	39.25	3.23
ES	25.60	62.73	5.16
FI	13.40	32.83	2.70
FR	19.01	46.56	3.83
GB	57.68	141.31	11.63
GR	65.16	159.64	13.14

Country	Total solid fuels [Mio t/y]	PeCB emission to air [kg/y]	PeCB discharge to fly ashes [kg/y]
HU	11.11	27.23	2.24
IE	5.29	12.95	1.07
IT	24.67	60.45	4.98
LT	0.29	0.72	0.06
LU	0.11	0.26	0.02
LV	0.15	0.36	0.03
MT	0.00	0.00	0.00
NL	12.88	31.56	2.60
PL	126.19	309.17	25.45
PT	4.16	10.19	0.84
RO	40.31	98.75	8.13
SE	4.20	10.30	0.85
SI	5.24	12.83	1.06
SK	7.90	19.36	1.59
EU 27	784.34	1921.62	158.21

At a European scale the estimated emissions amount to 2,079.83 kg/y. Thereof approximately 1,921.62 kg (2.45 mg/t incinerated coal) are emitted to air and 158.21 kg (84 kg of fly ash per tonne incinerated coal contaminated with 2.4 ppb) are part of the fly ashes treated as waste.

The following PeCB flow results:

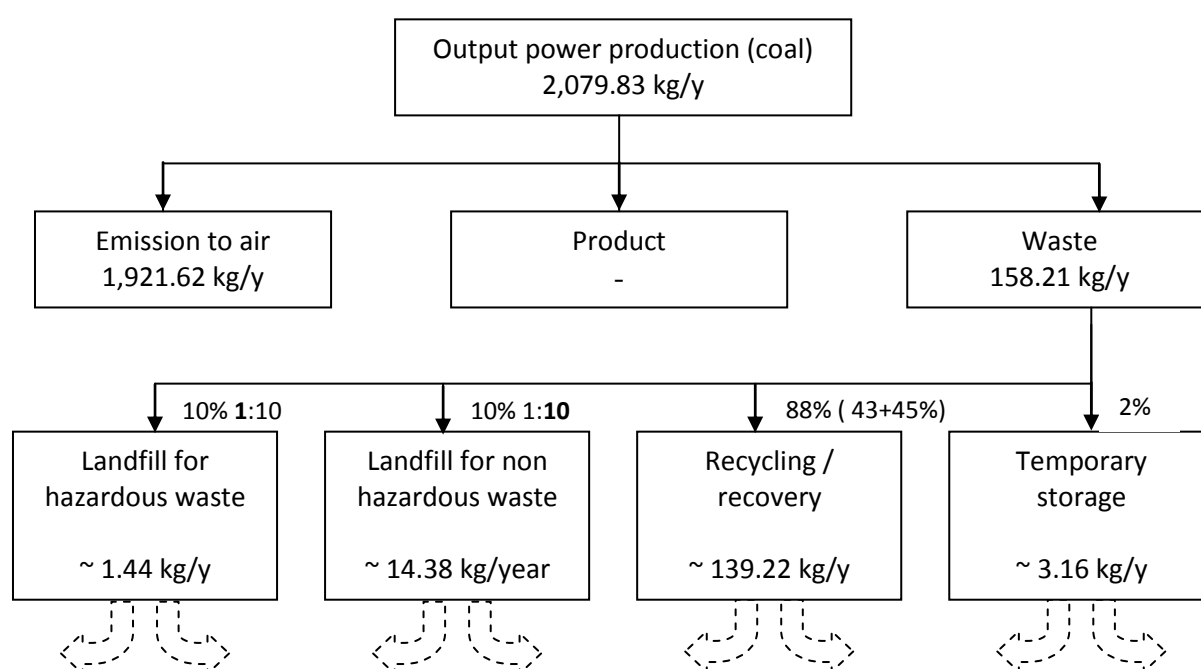


Figure 6-60: Assessment of the PeCB flow related to power production from coal and lignite

H) Waste flow for power production from coal

In order to illustrate the amounts of residues generated in EU 27 and the importance of currently used disposal/recovery operations which are of high relevance in the evaluation of impacts of proposed limit

values, a waste flow has been drawn up in the following figure. Mean contaminations and concentration ranges for each of the residues have been included for better interpretation.

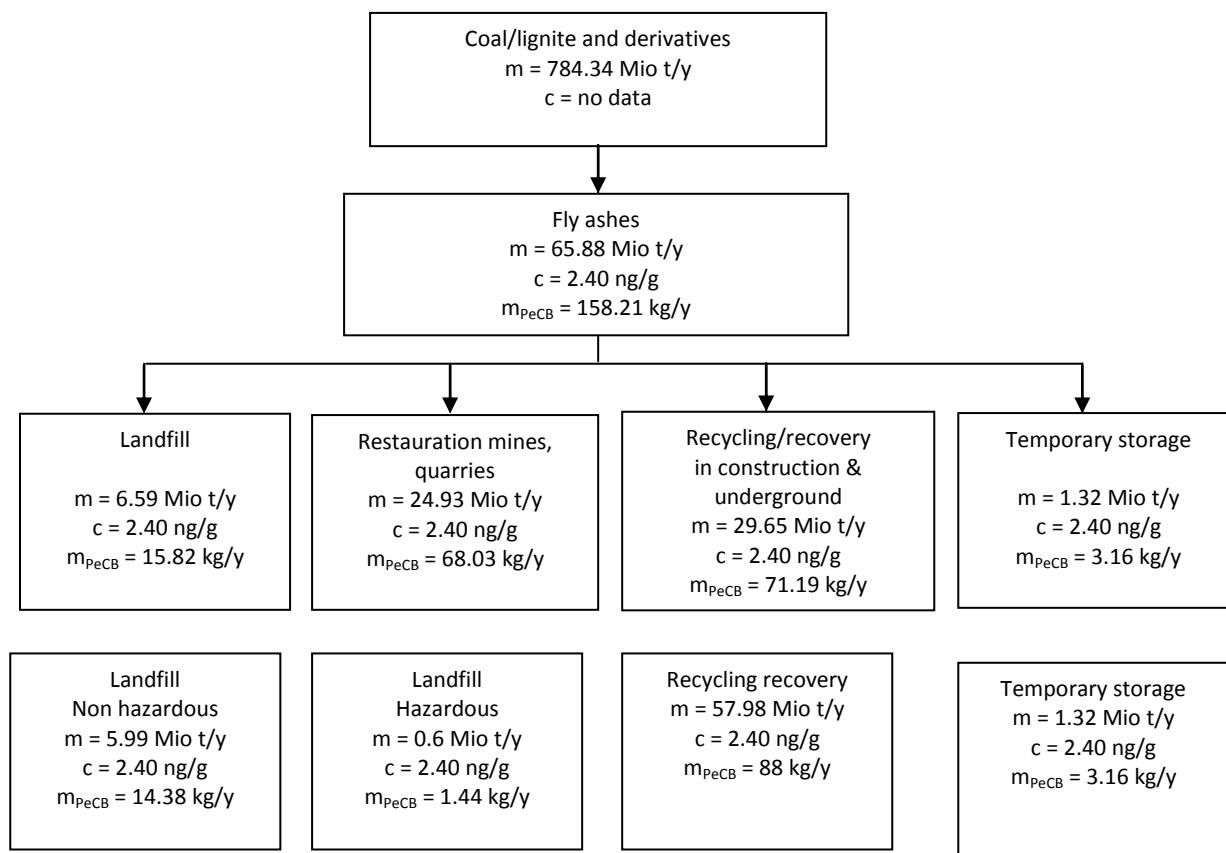


Figure 6-61: Detailed waste flow for coal fired power plants in EU 27 in 2008

6.4.12 Sewage sludge

A) Background

As PeCB tends to adsorb on particles, a relevant share of the PeCB content of waste water accumulates in sewage sludge during waste water treatment. The sewage sludge mentioned in this report is urban sewage sludge as well as sewage sludge resulting from independent and other waste water treatment facilities.

B) Process input (raw material) and output (waste, air, water, product)

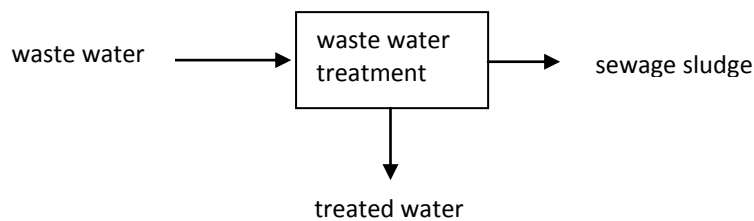


Figure 6-62: Relevant input and output of waste water treatment

Input: waste water

Output: sewage sludge and treated water

C) Generation factors for waste and other output (air, water)

No generation factor is used because the amount of sewage sludge produced from the wastewater depends on the composition of the wastewater and the treatment processes. Following the low solubility of PeCB in water the resulting treated water is not taken into further account.

D) Contamination data for output (and input)

Sewage sludge: As contamination data a median of 4.85 mg PeCB/t sewage sludge (4.85 ppb) has been used according to a study of Wang et al. for sewage sludge samples analysed in the UK [Wang et al. 1994]. The Environmental Protection Agency (EPA) in Ireland has recently undertaken sampling of wastes from sewage treatment to determine concentration levels of the new and candidate POPs in those wastes. According to [EPA IRE 2011a] no PeCB was measured above the limit of detection in any of the samples (LOD ranging from <100 µg/kg – 300 µg/kg). For PeCB analysis, the extraction method was solvent extraction and analysis standard based on US EPA Method 625.

E) Activity data

In all 27 Member States a total amount of approximately 11.5 Mio t of sewage sludge from waste water treatment has been calculated from EUROSTAT data (values derived from amount of sewage sludge

from urban waste water treatment, from other waste water treatment and independently operated waste water treatment). The Portuguese value originates from another source [UN-Habitat 2008]. For Denmark no value has been available. The amounts for the countries are given in Table 6-42.

Table 6-42: Annual amount of sewage sludge in the EU 27

Country	Sewage sludge produced [Mio t/y]			Total amount of sewage sludge [Mio t]	Reference
	Urban waste water treatment	Other waste water treatment	Independent waste water treatment		
AT	0.26	0.00	0.00	0.26	[EUROSTAT 2010]
BE	0.11	0.00	0.00	0.11	[EUROSTAT 2010]
BG	0.04	0.00	0.00	0.04	[EUROSTAT 2010]
CY	0.01	0.00	0.00	0.01	[EUROSTAT 2010]
CZ	0.17	0.00	0.00	0.17	[EUROSTAT 2010]
DE	2.05	0.00	0.00	2.05	[EUROSTAT 2010]
EE	0.03	0.00	0.00	0.03	[EUROSTAT 2010]
ES	1.07	0.00	0.00	1.07	[EUROSTAT 2010]
FI	0.16	0.00	0.00	0.16	[EUROSTAT 2010]
FR	1.06	0.00	0.00	1.06	[EUROSTAT 2010]
GR	0.13	0.00	0.00	0.13	[EUROSTAT 2010]
HU	0.18	0.10	0.00	0.29	[EUROSTAT 2010]
IE	0.09	0.00	0.00	0.09	[EUROSTAT 2010]
IT	1.06	0.00	0.00	1.06	[EUROSTAT 2010]
LT	0.08	0.00	0.00	0.08	[EUROSTAT 2010]
LU	0.01	0.00	0.00	0.01	[EUROSTAT 2010]
LV	0.02	0.00	0.00	0.02	[EUROSTAT 2010]
MT	0.00	0.00	0.00	0.00	[EUROSTAT 2010]
NL	0.37	0.17	0.00	0.54	[EUROSTAT 2010]
PL	0.53	0.56	0.00	1.09	[EUROSTAT 2010]
PT	0.24	0.00	0.00	0.24	[EUROSTAT 2010]
RO	0.10	0.66	0.002	0.76	[EUROSTAT 2010]
SE	0.21	0.00	0.00	0.21	[EUROSTAT 2010]
SI	0.02	0.00	0.00	0.02	[EUROSTAT 2010]
SK	0.06	0.28	0.00	0.33	[EUROSTAT 2010]
UK	1.77	0.00	0.00	1.77	[EUROSTAT 2010]
EU 27	9.82	1.76	0.002	11.58	[EUROSTAT 2010]

F) Waste treatment

The relevant treatment of sewage sludge includes land filling, incineration, application on land (agricultural use and compost) and other options. The amount of sewage sludge undergoing the different treatments is specific for each country and for each type of sewage sludge. In Table 6-43 values in % for each Member State are shown for the treatment of urban sewage sludge, which have been derived from EUROSTAT data for each country as far as available (not available for PT and DK).

Table 6-43: Treatment of urban sewage sludge in Member States (PT, DK: no data; MT: no production)

Country	Landfill [%]	Incineration [%]	Application to land [%]	Others [%]	Reference
AT	10	38	44	7	[EUROSTAT 2010]
BE	8	32	19	41	[EUROSTAT 2010]
BG	53	0	15	33	[EUROSTAT 2010]
CY	0	0	63	38	[EUROSTAT 2010]
CZ	5	0	78	16	[EUROSTAT 2010]
DE	0	47	53	0	[EUROSTAT 2010]
EE	17	0	14	69	[EUROSTAT 2010]
ES	16	4	65	16	[EUROSTAT 2010]
FI	6	0	92	2	[EUROSTAT 2010]
FR	21	17	60	3	[EUROSTAT 2010]
GR	98	0	0	2	[EUROSTAT 2010]
HU	29	0	68	3	[EUROSTAT 2010]
IE	6	0	72	23	[EUROSTAT 2010]
IT	42	3	44	11	[EUROSTAT 2010]
LT	12	0	42	46	[EUROSTAT 2010]
LU	33	0	67	0	[EUROSTAT 2010]
LV	0	0	43	57	[EUROSTAT 2010]
NL	4	68	13	15	[EUROSTAT 2010]
PL	23	0	23	53	[EUROSTAT 2010]
RO	44	0	0	56	[EUROSTAT 2010]
SE	11	0	50	39	[EUROSTAT 2010]
SI	43	24	19	14	[EUROSTAT 2010]
SK	16	0	70	14	[EUROSTAT 2010]
UK	5	16	70	9	[EUROSTAT 2010]
EU 27 (average)	21	10	45	24	[EUROSTAT 2010]

G) PeCB flow

Member State specific percent values for urban sewage sludge have been used for the estimation of PeCB transferred to the different treatment stations from the total amount of sewage sludge produced in each country. For Portugal the European average value for the sludge treatment has been applied. The values are given in Table 6-44.

Table 6-44: PeCB mass flow for sewage sludge in EU 27

Country	Amount sewage sludge [Mio t/y]	Amount PeCB [kg/y]	Landfilled [kg/y]	Incineration [kg/y]	Application to land [kg/y]	Others [kg/y]
AT	0.26	1.24	0.12	0.48	0.55	0.09
BE	0.11	0.54	0.04	0.17	0.10	0.22
BG	0.04	0.19	0.10	0.00	0.03	0.06
CY	0.01	0.04	0.00	0.00	0.03	0.02
CZ	0.17	0.83	0.04	0.00	0.65	0.14
DE	2.05	9.94	0.02	4.68	5.23	0.00
DK	0.00	0.00	0.00	0.00	0.00	0.00
EE	0.03	0.15	0.03	0.00	0.02	0.10

Country	Amount sewage sludge [Mio t/y]	Amount PeCB [kg/y]	Landfilled [kg/y]	Incineration [kg/y]	Application to land [kg/y]	Others [kg/y]
ES	1.07	5.17	0.81	0.20	3.33	0.82
FI	0.16	0.78	0.05	0.00	0.71	0.01
FR	1.06	5.14	1.08	0.86	3.07	0.14
GR	0.13	0.61	0.60	0.00	0.00	0.01
HU	0.29	1.39	0.40	0.00	0.94	0.05
IE	0.09	0.43	0.02	0.00	0.31	0.10
IT	1.06	5.12	2.13	0.15	2.26	0.58
LT	0.08	0.37	0.04	0.00	0.16	0.17
LU	0.01	0.06	0.02	0.00	0.04	0.00
LV	0.02	0.11	0.00	0.00	0.05	0.06
MT	0.00	0.00	0.00	0.00	0.00	0.00
NL	0.54	2.62	0.11	1.78	0.34	0.40
PL	1.09	5.28	1.24	0.02	1.22	2.80
PT	0.24	1.15	0.24	0.12	0.52	0.27
RO	0.76	3.68	1.62	0.00	0.00	2.06
SE	0.21	1.02	0.12	0.00	0.50	0.40
SI	0.02	0.10	0.04	0.02	0.02	0.01
SK	0.33	1.61	0.26	0.00	1.12	0.23
UK	1.77	8.59	0.47	1.37	5.99	0.77
EU 27	11.58	56.15	9.60	9.85	27.18	9.51

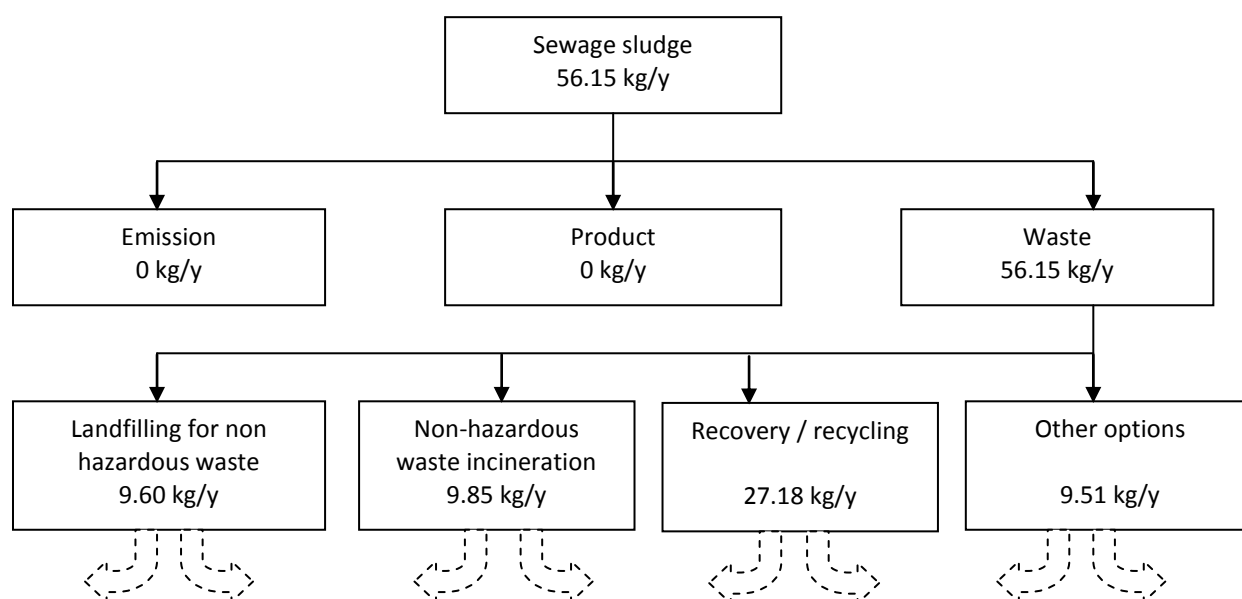


Figure 6-63: Assessment of the PeCB flow related to sewage sludge

H) Waste flow for sewage sludge

In order to illustrate the amounts of residues generated in EU 27 and the importance of currently used disposal/recovery operations which are of high relevance in the evaluation of impacts of proposed limit values, a waste flow has been drawn up in the following figure. Mean contaminations and concentration ranges for each of the residues have been included for better interpretation.

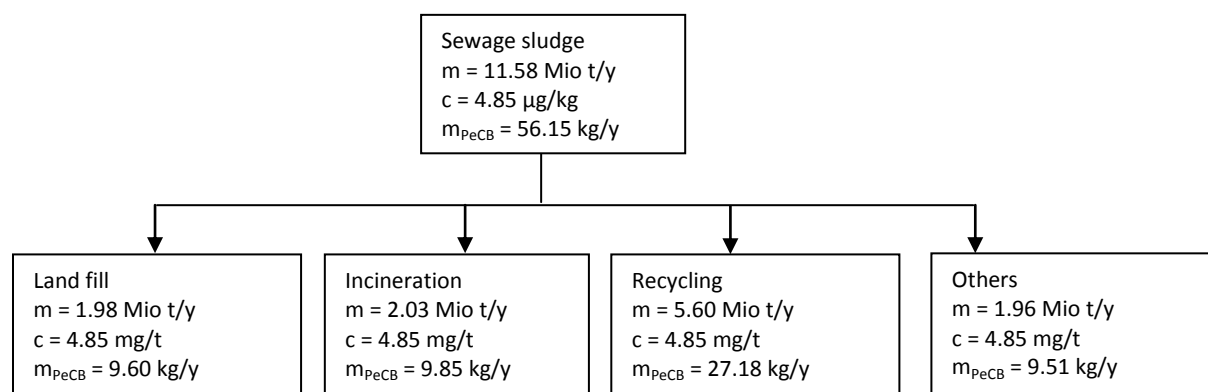


Figure 6-64: Waste flow for sewage sludge in EU 27

6.4.13 Domestic Burning

A) Background

Domestic burning of wood, fossil fuels and mixed wastes accounts to private burning in single stoves or open burning places. Domestic burning is associated with high air emissions as flue gas treatment is not performed. In addition process conditions can strongly vary due to specific oven characteristics and the properties of the used fuel. As a consequence measured contamination and emission data and derived emission factor are highly inhomogeneous to a large range of possible results. In general it can be assumed that emissions exceed the discharge via residues by far and that burning of treated wood or co-combustion of waste significantly increases formation and discharge of PeCB.

B) Process input (raw material) and output (waste, air, water, product)

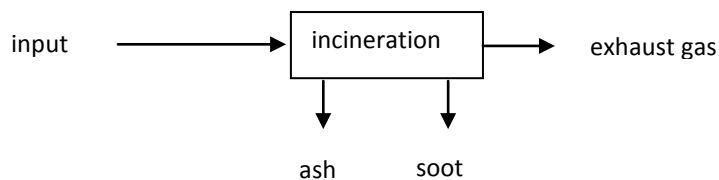


Figure 6-65: Relevant input and output of domestic burning

Input	Wood and wood waste, solid fuels (coal, lignite and their derivatives) and mixed fuels or wastes
Output:	exhaust gas, ashes and soot

C) Generation factors

Ash:	For coal, the waste generation factor is 0.1 t ash/t coal, for wood it amounts to 0.017 t ash/t wood [BiPRO 2005].
Soot:	The generation factor for wood is 0.0003 t soot/t wood and for coal 0.0006 t soot/t coal [BiPRO 2005].

D) Contamination data for output (and input)

Exhaust gas:	Domestic burning of coal was estimated to result in 2.45 mg PeCB/ton coal combusted by Bailey et al. [Euro Chlor 2007]. Combustion of wood has been assumed by Bailey et al. to amount to 1.2 mg/t wood burned according to a study from Zimmerman et al. in 2001 [Euro Chlor 2007]. According to Lemieux et al. open burning of household waste led to resulting
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emissions of PeCB of 76 mg/t mass burned [Euro Chlor 2007].

Ash: For burning of coal, the contamination factor of the resulting fly ash ranged from 0.004-7 ng PeCB/g ash [Schreiner et al. 1986]. An average value of 2.40 ng/g (2.4 ppb) has been applied for burning of solid fuels.

For wood, a contamination factor of 0.2 ng/g (0.2 ppb) for the resulting ash has been used according to Schreiner et al.

For municipal solid waste, the same contamination factor of 74.4 ng/g (74.4 ppb) as in the chapter for MSWI has been applied.

Soot: No information has been available of the contamination of PeCB in soot. Since the generation factor of soot for wood and coal are relatively small, they are considered as insignificant.

E) Activity data

The final energy consumption in EU 27 of solid fuels in the domestic sector in 2008 amounts to 17.03 Mio t and for wood and wood waste 32.26 Mio t [EUROSTAT 2010]. For the amount of mixed solid fuels burned domestically, a share of 0.5% from a total of 262.63 Mio t MSW generated has been assumed.

Table 6-45: Annual consumption of solid fuels, wood and mixed solid fuels for domestic burning in EU 27 in 2008

Country	Solid fuels [Mio t/y]	Wood & wood waste [Mio t/y]	Reference	Mixed solid fuels (assumption) [Mio t/y]
AT	0.13	1.63	[EUROSTAT 2010]	0.03
BE	0.26	0.22	[EUROSTAT 2010]	0.03
BG	0.40	0.64	[EUROSTAT 2010]	0.02
CY	0.00	0.01	[EUROSTAT 2010]	0.00
CZ	1.30	1.06	[EUROSTAT 2010]	0.02
DE	1.75	5.35	[EUROSTAT 2010]	0.24
DK	0.00	0.92	[EUROSTAT 2010]	0.02
EE	0.02	0.39	[EUROSTAT 2010]	0.00
ES	0.30	2.06	[EUROSTAT 2010]	0.13
FI	0.05	1.13	[EUROSTAT 2010]	0.01
FR	0.55	6.67	[EUROSTAT 2010]	0.18
GR	0.04	0.60	[EUROSTAT 2010]	0.18
HU	0.41	0.47	[EUROSTAT 2010]	0.03
IE	1.13	0.02	[EUROSTAT 2010]	0.02
IT	0.01	1.42	[EUROSTAT 2010]	0.02
LT	0.09	0.40	[EUROSTAT 2010]	0.17
LU	0.00	0.02	[EUROSTAT 2010]	0.01
LV	0.03	0.73	[EUROSTAT 2010]	0.00
MT	0.00	0.00	[EUROSTAT 2010]	0.00
NL	0.01	0.23	[EUROSTAT 2010]	0.00
PL	9.29	2.45	[EUROSTAT 2010]	0.05
PT	0.00	1.16	[EUROSTAT 2010]	0.06
RO	0.08	3.42	[EUROSTAT 2010]	0.03

Country	Solid fuels [Mio t/y]	Wood & wood waste [Mio t/y]	Reference	Mixed solid fuels (assumption) [Mio t/y]
SE	0.00	0.62	[EUROSTAT 2010]	0.04
SI	0.00	0.32	[EUROSTAT 2010]	0.02
SK	0.20	0.04	[EUROSTAT 2010]	0.00
UK	0.99	0.30	[EUROSTAT 2010]	0.01
EU 27	17.03	32.26		1.31

F) Waste treatment

There is no official information available on the further treatment of the solid residues from domestic burning. However, according to experience and expert information it can be assumed that the residues are either added to the municipal solid waste or applied to land. As a first approach a share of application to land versus discharge to municipal solid waste of 1:3 has been assumed. According to chimney sweeper information removed soot is completely disposed of with municipal solid waste. Based on the data from the mass flow on MSW a distribution of 20% directed to MSWI, 40% disposed of at non-hazardous waste landfills and 40% directed to other treatment options as MBT can be used as European average for the calculation of the mass flow [BiPRO 2005].

G) PeCB mass flow

The available data and assumptions allow an estimation of PeCB emissions to air and discharge to ashes. The following table shows a country specific estimation on emitted amounts:

Table 6-46: PeCB mass flow for domestic burning of wood and solid fuels in EU 27

Country	Fuel consumption (Mio t/y)			Emission to air [kg/y]			Total emission [kg/y]	Discharge to ash [kg/y]			Total to waste [kg/y]
	Solid fuel	Wood	MSW	Solid fuel	Wood	MSW		Solid fuel	Wood	MSW	
AT	0.13	1.63	0.03	0.32	1.96	1.91	4.19	0.03	0.006	0.04	0.08
BE	0.26	0.22	0.03	0.64	0.26	2.03	2.93	0.06	0.001	0.04	0.11
BG	0.40	0.64	0.02	0.98	0.77	1.34	3.09	0.10	0.002	0.03	0.13
CY	0.00	0.01	0.00	0.00	0.01	0.23	0.24	0.00	0.000	0.01	0.01
CZ	1.30	1.06	0.02	3.19	1.27	1.22	5.67	0.31	0.004	0.03	0.34
DE	1.75	5.35	0.24	4.29	6.42	18.06	28.77	0.42	0.018	0.39	0.83
DK	0.00	0.92	0.02	0.00	1.10	1.69	2.79	0.00	0.003	0.04	0.04
EE	0.02	0.39	0.00	0.05	0.47	0.26	0.78	0.00	0.001	0.01	0.01
ES	0.30	2.06	0.13	0.74	2.47	10.05	13.25	0.07	0.007	0.22	0.30
FI	0.05	1.13	0.01	0.12	1.35	1.06	2.54	0.01	0.004	0.02	0.04
FR	0.55	6.67	0.18	1.35	8.01	13.35	22.71	0.13	0.023	0.29	0.45
GR	0.04	0.60	0.18	0.10	0.72	13.31	14.13	0.01	0.002	0.29	0.30
HU	0.41	0.47	0.03	1.00	0.56	1.94	3.51	0.10	0.002	0.04	0.14
IE	1.13	0.02	0.02	2.77	0.03	1.72	4.52	0.27	0.000	0.04	0.31
IT	0.01	1.42	0.02	0.02	1.71	1.24	2.97	0.00	0.005	0.03	0.03
LT	0.09	0.40	0.17	0.22	0.48	12.86	13.57	0.02	0.001	0.28	0.30
LU	0.00	0.02	0.01	0.00	0.02	0.51	0.53	0.00	0.000	0.01	0.01
LV	0.03	0.73	0.00	0.07	0.88	0.13	1.09	0.01	0.002	0.00	0.01
MT	0.00	0.00	0.00	0.00	0.00	0.28	0.28	0.00	0.000	0.01	0.01
NL	0.01	0.23	0.00	0.02	0.27	0.11	0.41	0.00	0.001	0.00	0.01

Country	Fuel consumption (Mio t/y)			Emission to air [kg/y]			Total emission [kg/y]	Discharge to ash [kg/y]			Total to waste [kg/y]
	Solid fuel	Wood	MSW	Solid fuel	Wood	MSW		Solid fuel	Wood	MSW	
PL	9.29	2.45	0.05	22.76	2.94	3.92	29.62	2.23	0.008	0.09	2.32
PT	0.00	1.16	0.06	0.00	1.39	4.64	6.03	0.00	0.004	0.10	0.11
RO	0.08	3.42	0.03	0.20	4.11	1.93	6.23	0.02	0.012	0.04	0.07
SE	0.00	0.62	0.04	0.00	0.74	3.12	3.86	0.00	0.002	0.07	0.07
SI	0.00	0.32	0.02	0.00	0.39	1.83	2.22	0.00	0.001	0.04	0.04
SK	0.20	0.04	0.00	0.49	0.04	0.36	0.89	0.05	0.000	0.01	0.06
UK	0.99	0.30	0.01	2.43	0.36	0.68	3.46	0.24	0.001	0.01	0.25
EU 27	17.03	32.26	1.31	41.75	38.71	99.80	180.26	4.09	0.11	2.17	6.37

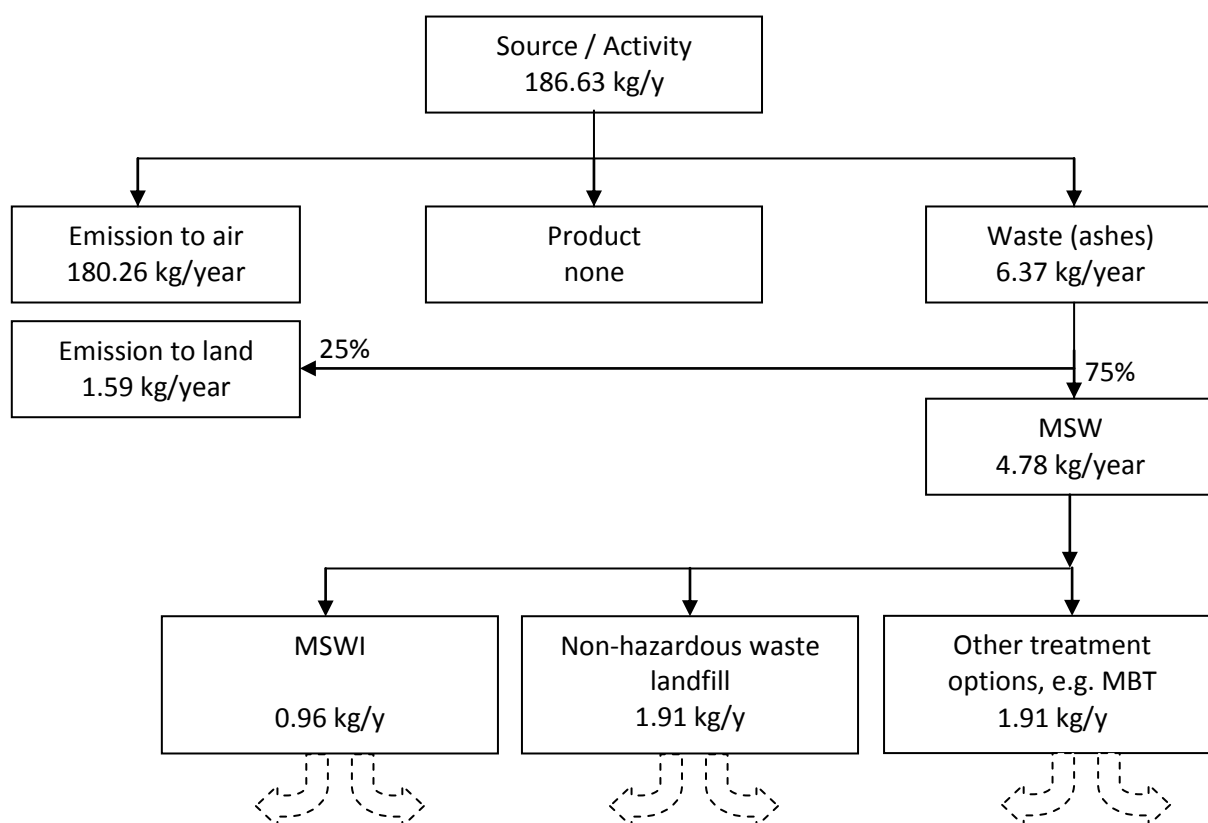


Figure 6-66: Assessment of the PeCB flow related to domestic burning

H) Waste flow for domestic burning of wood, fossil fuels and MSW

In order to illustrate the amounts of residues generated in EU 27 and the importance of currently used disposal/recovery operations which are of high relevance in the evaluation of impacts of proposed limit values, a waste flow has been drawn up in the following figure. Mean contaminations and concentration ranges for each of the residues have been included for better interpretation.

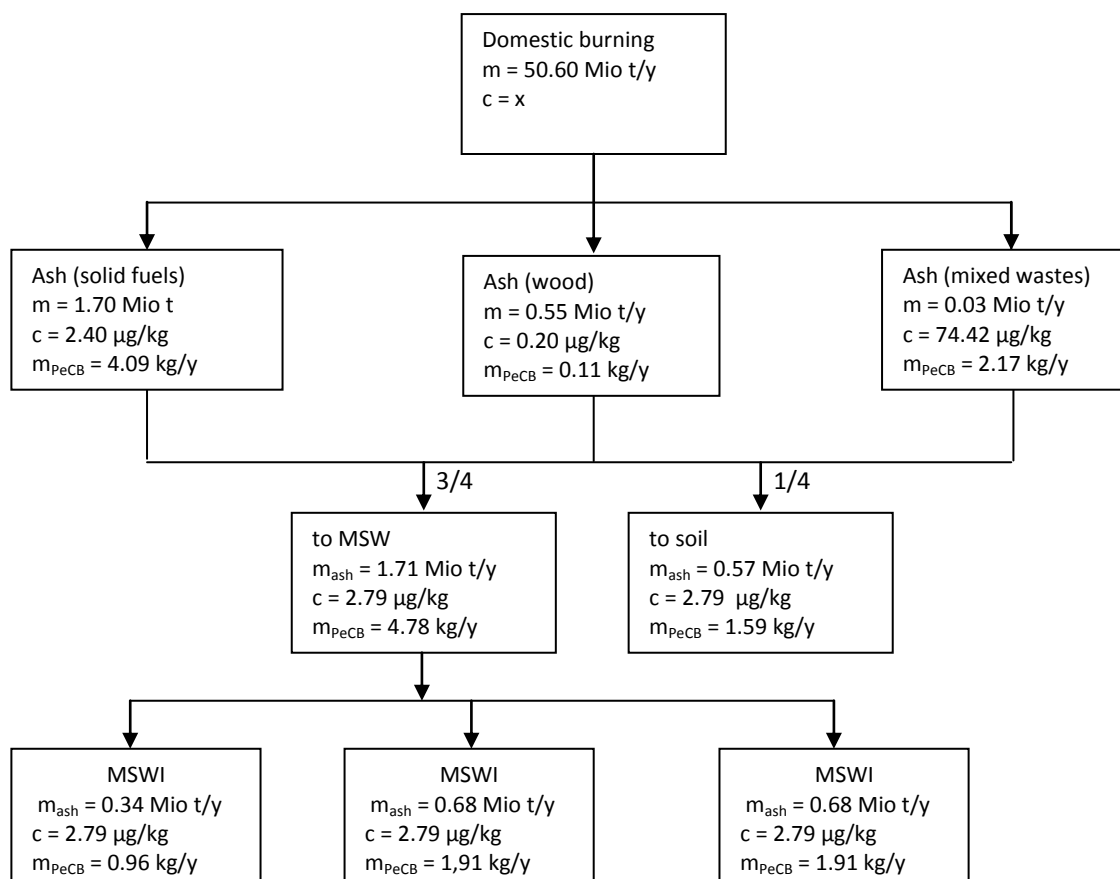


Figure 6-67: Detailed waste flow for domestic burning of wood, solid fossil fuels (coal and derivatives) and mixed wastes in EU 27

Since 25% of the ashes are emitted to land the remaining 75% are subject to MSW, which in return is incinerated to 20%, landfilled to 40% and treated with other options as MBT to 40%. The waste flow for MSWI is described in detail in the chapter MSWI.

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6.5 SCCP flows

6.5.1 Use and Production

Short chain chlorinated paraffins (SCCPs) are still in use within Europe. Uses include application in paints, adhesives and sealants, plastics and rubber, flame retardants as well as textiles and polymeric materials (e.g. PCB substitute in gaskets). Since 2002 the use of SCCPs in the EU in metal working and leather fat liquoring industry has been subject to restrictions under the EU Directive 2002/45/EC, which presumably has led to a dramatic decrease of SCCP use quantities after 2002. In Germany, for instance, the most important uses of SCCPs were banned by the above mentioned Directive (74 % of the total uses), [POPRC 6/INF/15/]. Recent information on use quantities indicate that SCCPs are still used in the EU 27. [SCCP RPA 2010]

Chlorinated paraffins (CPs) of various chain lengths are currently produced in the USA, EU, Russia, India, China, Japan and Brazil [POPRC 6/11]. Within the EU, Slovakia reported on production quantities of 560, 354, 480 and 410 t for the years 2004 to 2007, respectively. In Germany the production of SCCPs by the companies Clariant, Hoechst and Huels stopped in 1995. Hoechst reported on production quantities between 9,300 to 19,300 t between 1993 and 1995, respectively [POPRC 6/11]. Sales of SCCPs in the EU constantly declined partly due to the phasing out of production and use in Germany as well as the EU marketing and use Directive; they amounted to 13,200 t in 1994, 7,370 t in 1997 and 4,080 t in 1998. The amount sold in 1998 represented 6.4 % of the worldwide CPs sale [POPRC 6/11], (i.e. total sold in 1998: 63,750 t).

In summary, SCCPs are still used and produced within Europe. However, use and production have significantly declined since 1998 and especially after 2002 due to the introduction of EU regulations. The current use of SCCP in the EU 27 is considered to be about 530 t [SCCP RPA 2010].

6.5.2 Chemical Characteristics of SCCP and its derivatives

Chemically, SCCPs may be defined as straight-chain chlorinated alkanes with a carbon chain length between 10 and 13. More restrictive definitions can specify a certain degree of chlorination (e.g. > 48%). Additionally, a definition could include the individual chain-length compounds covered by the definition, for instance a single chain of 12 carbons, chlorinated, or could include a specific chemical definition of a class, e.g. $C_xH_{(2x-y+2)}Cl_y$ where $x = 10-13$ and $y = 1-13$ [POPRC 6/INF/15]. The term SCCPs as listed in Annex I and II of the CLRTAP POP protocol means “chlorinated alkanes with a carbon chain length of 10 to 13 carbon atoms and the degree of chlorination of more than 48% by weight.”²⁷

For the chemical industry, the term “chlorinated paraffins” refers to a mixture of chlorinated alkanes with distributions in both carbon chain length and chlorine content. The commercial product, consisting mainly of the chemically defined SCCPs, would also contain impurities such as chlorinated paraffins outside of the defined range for SCCPs, and chlorinated alkenes, branched alkanes, and aromatics, as part of the SCCPs mixture [POPRC 6/INF/15].

²⁷ According to Decision 2009/2, adopted on 18 December 2009 at the twenty-seventh session of the Executive Body, held in Geneva, Switzerland from 14 to 18 December 2009

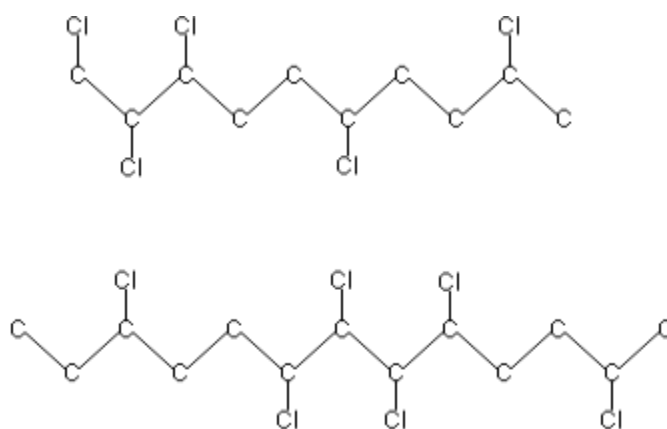
Chemical name:

Alkanes, C10-13, chloro; alkanes, chlorinated; alkanes (C10-13), chloro-(50-70%); alkanes (C10-13), chloro-(60%); chlorinated alkanes, chlorinated paraffins; chloroalkanes; chlorocarbons; polychlorinated alkanes; and paraffins chlorinated. The synonyms are general in nature, and encompass much more than the substance represented by either the CAS number given or C10-13 chlorinated alkanes in general.

Molecular formula: $C_{10-13}H_{10-25}Cl_{3-12}$ or $C_xH_{(2x-y+2)}Cl_y$ ($x=10-13$; $y=3-12$)

CAS Registry Number: Short -chain Chlorinated Paraffins (CAS No.85535-84-8)

Structure: Two SCCP Compounds ($C_{10}H_{17}Cl_5$ and $C_{13}H_{22}Cl_6$) [POPRC 6/11]



Molar Mass: 320-500 g/mol [SCCP ERA 2000]

Physico-chemical properties [SCCP POPRC 6/11] [SCCP BUWAL 2003], [SCCP ERA 2000], [SCCP ERA 2008]

Aggregate state	Clear or yellowish viscose fluid/white solid	[SCCP BUWAL 2003]
Pour Point	-30.5-20 °C	[SCCP ERA 2000]
Density	1.18-1.59 g/cm ³ (20°C)	[SCCP BUWAL 2003]
Boiling point	> 200°C (decomposition above 200°C)	[SCCP BUWAL 2003]
Vapour pressure (at 100 °C)	0.028-2.8 x 10 ⁻⁷ Pa, 0.0213Pa	[SCCP POPRC 6/11] [SCCP ERA 2008]
Partition coefficient Log K _{ow}	4.48-8.69	[SCCP POPRC 2008]
Water solubility	150-470 µg/l	[SCCP ERA 2008]
Henry's Law Constant	0.7-18 Pa·m ³ /mol	[SCCP POPRC 6/11]

6.5.3 Environmental Background Levels

SCCP has spread widely in the global environment. Existing monitoring data on SCCP shows that concentration levels of the substance were detected in Arctic marine mammals in concentrations up to 426-526 µg/kg [SCCP OSPAR 2001; SCCP OSPAR 2006].

Table 6-47 and Table 6-48 give an overview of the concentration levels detected in sediments, soils and sewage sludge.

Table 6-47: Levels of SCCP in sediment and soil (DW: dry weight; ww: wet weight; ND: not detected) [SCCP OSPAR 2001], [SCCP POPRC 6/11]

Country/Region	Compartment	Levels of SCCP	References	Comments
Germany (1994) downstream from a chlorinated paraffins production plant	Sediment	17-83 µg/kg dw	[SCCP OSPAR 2001]	From the EU Risk Assessment Report (1999),
Three lakes in Canada	Surface sediment	18-275 µg/kg dw	[SCCP OSPAR 2001]	From the EU Risk Assessment Report (1999),
Harbour areas along Lake Ontario	Surface sediment	7.3-290 µg/kg	[SCCP OSPAR 2001]	From the EU Risk Assessment Report (1999), dw or ww is not known
Detroit River at Lake Erie in Canada	Sediment	1.8 µg/kg	[SCCP OSPAR 2001]	From the EU Risk Assessment Report (1999), dw or ww is not known
Lake Hazen on Ellesmere Island in the Arctic	Sediment	4.5 µg/kg dw	[SCCP OSPAR 2001]	From the EU Risk Assessment Report (1999),
Iqualit, Nunavut, in Canadian Arctic	Soil next to a landfill	60.4±54.9 µg/kg dw	[POPRC 6/11] (Dick et.al. 2010)	
Detroit river at Lake Erie and Middle Sister Island in western Lake Erie	Sediment	245 µg/kg dw	[POPRC 6/11] (Tomy et al. 1997)	
Harbour area along Lake Ontario	Surface Sediment	5.9-290 µg/kg dw	[POPRC 6/11] (Muir et al. 2001)	Highest level measured was 410 µg/kg dw from the Niagara basin
Iqualit, Nunavut, in Canadian Arctic	Sediment	5.2±138.1 µg/kg dw	[POPRC 6/11] (Dick et.al. 2010)	
United Kingdom	Soil	600 – 10,300 µg/kg dw	[POPRC 6/11] (Nicholls et al. 2001)	
11 Czech rivers and five drainage vents near industrial areas	Sediment	LOD-347 µg/kg dw	[POPRC 6/11] (Příbylová et al. 2006)	LOD is not provided
Czech, Košetice, Zlín and Beroun area	Sediment	24-46 µg/kg dw, 16-181 µg/kg dw, 4.6-22 µg/kg dw	[POPRC 6/11] (Štejnarová et al. 2005)	Košetice (reference area), Zlín (rubber, tanning, textile), Beroun (ciments, machinery)
Japan, two rivers	Sediment	5-484 µg/kg dw	[POPRC 6/11] (Iino et al. 2005)	Five of six samples had concentrations < 196 µg/kg

Table 6-48: Levels of SCCP in sewage sludge including hot spot levels (DW: dry weight) [SCCP OSPAR 2001] [SCCP POPRC 6/11], [SCCP UBA 2006], [SCCP BUWAL 2003]

Country/ Region	Compartment	Levels of SCCP	References	Comments
Germany (1991-1993) near a metal working plant	Sewage sludge	47 – 65 µg/g	[SCCP OSPAR 2001]	From the EU Risk Assessment Report (1999),
United Kingdom	Sewage sludge	6,900-200,000 µg/kg dw	[POPRC 6/11] (Steven et.al. 2002)	From 14 WWT
Germany Hessen (2003)	Sewage Sludge	Average: 382 µg/kg dw; maximum 537 µg/kg dw	[SCCP UBA 2006]	9 samples
Germany	Sewage sludge	47,000 (1993)-65,000 (1991) µg/kg ds avg. 56,000 µg/kg ds	[SCCP BUWAL 2003]	
Swiss	Sewage sludge	500-80,000 µg/kg avg. 34,100 µg/kg ds	[SCCP BUWAL 2003]	Samples were taken between 1983 and 1985,
Swiss	Sewage sludge	5.5 t in sewage sludge	[SCCP BUWAL 2003]	

As shown in Table 6-47, the detected background contamination levels of SCCP in sediment ranged from the LOD to a maximum concentration level of 484 µg/kg (0.484 ppm) DW in sediments determined in Japan. In general the determined concentration levels in sediment were below 300 µg/kg (0.3 ppm) DW. The highest hot spot levels were detected in sewage sludge (maximum levels of 200,000 µg/kg (200 ppm) and in SCCP contaminated soils in the UK reaching to 10,300 µg/kg (10.3 ppm), [POPRC 6/11]. In comparison to river and lake sediment concentrations, SCCP contents in sewage sludge are much higher, especially from WWTPs serving industrial areas. It is reported that the high differences of the values between 500 and 80,000 µg/kg (0.5 and 80 ppm) from Switzerland seems to be due to different shares of waste water from industries as metal working industry. It is assumed that especially the improper disposal of lube oil might be a reason for the higher concentrations. It should be considered that these measurements took place in the 80's, at the time where the use of SCCP was much higher [SCCP BUWAL 2003].

To conclude, background levels in soil and sediment are expected to be usually below 1 ppm and are not expected to exceed levels of 10 ppm.

6.5.4 Legal Situation

In December 2009 it was decided to list SCCPs in Annex I and II of the UNECE CLRTAP POPs protocol (EB Decision 2009/2). Accordingly, production and use of SCCPs are scheduled for elimination except for the restricted uses as fire retardants in (a) conveyor belts in mining industry and (b) in dam sealants. These uses should be eliminated once suitable alternatives are available.

SCCPs are proposed to be listed in Annexes A, B or C of the Stockholm Convention (UNEP/POPS/POPRC.2/INF/6). At the sixth meeting of the POPRC in October 2010, the committee reviewed the revised risk profile and decided to postpone any decision-making. In the meantime, it will gather additional information on its environmental and health effects and trends in the levels in the environment to assist decision-making at its eighth meeting.

EU level

The placing on the market of SCPP's 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 was banned by Directive 2002/45/EC of the European Parliament and of the Council of 25 June 2002 amending for the twentieth time Council Directive 76/769/EEC relating to restrictions on the marketing and use of certain dangerous substances and preparations (short-chain chlorinated paraffins). The Member States are obliged to apply the relevant measures as from 6 January 2004 at the latest.

All of the parties stated that they have implemented Directive 2002/45/EC, which includes the following provisions:

Alkanes, C₁₀-C₁₃ chloro (short-chained chlorinated paraffins)

- 1, May 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;
 - for fat liquoring of leather,

SCCPs are restricted accordingly under Annex XVII of REACH as it currently stands. The existing restrictions only apply for two specific applications of SCCPs: as an additive in metal working fluids and in leather fat liquoring. Other known uses as flame retardants in rubber formulations and as additives in sealants, paints and textile coating systems are not covered at EU-level through REACH Annex XVII.

In October 28, 2008, SCCPs were included in the Candidate list for eventual inclusion in Annex XIV of REACH as PBT/vPvB substance. SCCPs were included in the First Recommendation for inclusion of substances in Annex XIV by ECHA. On 20-21 September 2010, the REACH Committee decided on the first substances to be included in Annex XIV; SCCPs are not listed in the Commission Proposal that was sent out for agreement in this Committee meeting.

On June 4, 2009, the Netherlands requested for an Article 69(5) REACH decision with the proposal to re-examine the SCCP restriction in Annex XVII. The letter containing this request may be seen as a preliminary RMO analysis. By the time the letter was sent, the UN-ECE decision on banning SCCPs was not taken yet. On December 14, 2009, the REACH Committee discussed the request of the Netherlands to re-examine the listing of SCCPs as entry 42 of Annex XVII, with a positive outcome. In the

COMMISSION DECISION 226/2010 of 20 April 2010 (notified under document C (2010) 1942) this request was formally granted.

PARCOM decision 95/1 is more stringent than current EU legislation and aims to phase out the use of SCCPs by restricting also the other uses of SCCPs. PARCOM Decision 95/1 requires Contracting Parties to phase-out the use of SCCPs as plasticiser in paints, coatings and sealants; as flame retardants in rubber, plastics and textiles; and their use in metal working fluids. The phase-out for these uses should be achieved by 31 December 1999, except for uses as plasticizer in sealants in dams and as flame retardant in conveyor belts for the exclusive use in underground mining which should be phased-out by 31 December 2004 [SCCP OSPAR 2006]. Already by 2006 several Member States have implemented this decision (Belgium, Denmark, Finland, France, Germany, Netherlands, Norway, Sweden, Switzerland and the United Kingdom) and approved by the European Community and Spain (see Table 6-49).

Table 6-49: Implementation of PARCOM Decision 95/1 (t/y) [SCCP OSPAR 2006]

Contracting Party	Reservation	Report submitted 2005/2006	Measure applicable	Means of implementation by		
				Legislation	Administrative action	Voluntary agreement
Belgium	NO	YES	YES	YES		
Denmark	NO	NO	YES	YES		
France	NO	NO	YES	YES		
Finland	NO	YES	YES	YES		
Germany	NO	YES	YES	YES		
Iceland	NO	NO		No evidence supplied that measure is implemented		
Ireland	NO	NO		No evidence supplied that measure is implemented		
Luxembourg	NO	NO		No evidence supplied that measure is implemented		
Netherlands	NO	YES	YES	YES		
Norway	NO	YES	YES	YES		
Portugal	NO	NO		No evidence supplied that measure is implemented		
Spain	NO	YES	YES	YES		
Sweden	NO	YES	YES	YES		
Switzerland	NO	YES	YES	YES		
United Kingdom	YES	(YES)	NO	YES		

As the UK has a reservation on PARCOM Decision 95/1, they are not subject to implementation reporting. The UK still submitted an implementation report on a voluntary basis.

6.5.5 Occurrence and Relevance of SCCP and its derivatives in Europe

SCCP has been used in many applications. These applications are in metal fluids, in leather tanning, coatings as paintings and varnishes, sealant and adhesives, rubber and plastic textiles as well as lava lamps or lube oil. SCCP is also existent in applications which include MCCP to an impurity level of 1 % [POPRC 6/INF/15]. MCCP has been used as an alternative material of SCCP and with the constant decrease of SCCP in the last years, MCCP has increased.

6.5.5.1 Current uses

For most applications SCCP could still be used. However, due to different voluntary approaches as PARCOM Decision 95/1 the used SCCP amount has been decreasing in all industries in the last years.

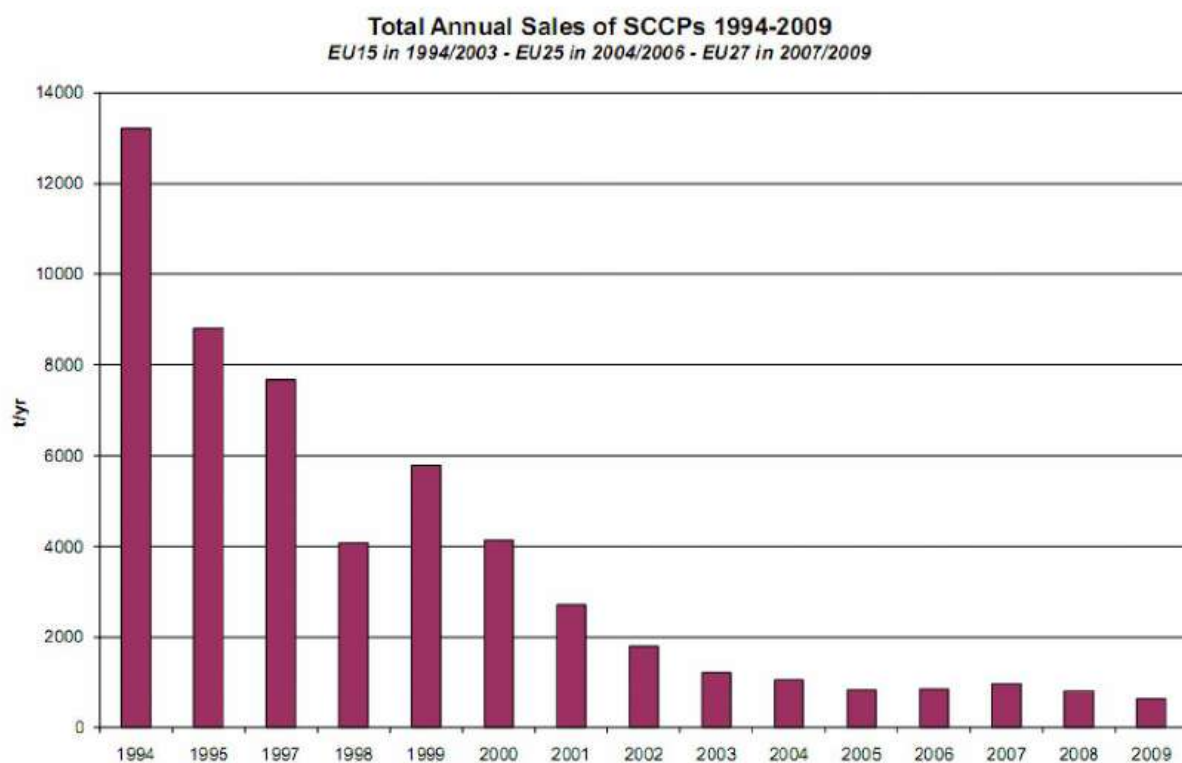


Figure 6-68: Total annual Sales of SCCP in the EU 1994-2009 [SCCP RPA 2010] (provided by Eurochlor)

The calculation figures which are provided by the evaluation report of possible restriction for SCCP [SCCP RPA 2010] are used for this report.

6.5.5.2 Past uses

After the Directive 2002/45/EC has come into force in January 6 2004, the marketing and use of SCCPs for metal working fluids and fat liquoring as substances or as constituents of other substances or preparations in concentrations higher than 1% has been restricted. Consequently the use of SCCP in the metal and leather industry as metal working fluids or fat liquoring has ceased. This is based on the fact that SCCP is only effective in concentrations of typically 5-10 % in oil-based metal cutting fluids and typically 20 % in the leather fat liquoring mix. In this context one possible exception has to be named, which is in emulsion-based metal working fluids, where the final chlorinated paraffin concentration in the final emulsified fluid can be <1 %. However, the supplied lubricants typically have chlorinated paraffin contents of 5 % and up to around 8 % and are then diluted before use. Therefore, it is unlikely that lubricants supplied with <1 % SCCP contents would be effective after dilution. [SCCP ECHA 2009]

The amount of SCCP used in the EU in the metal and leather industry is subjected to a continuous tendency to fall. In 1994, 9,380 t/y of SCCPs were consumed in the metal working industry, whereas in

1998 the amount fell to 2,018 t/y. A similar development can be found in the leather industry, where the usage of SCCP fell from 390 t/y in 1994 to 45 t/y in 1998. Due to the Directive 2002/45/EC these numbers are expected to be zero nowadays [POPRC 6/11].

Even though SCCP is not used in the production of leather anymore, there are still remainder of SCCP containing leathers that end up as waste every year. Therefore, this sector is investigated in the report.

In different literature (especially when citing older literature sources), PVC has been considered as one of the possible uses of SCCP ([SCCP BUWAL 2003]; [SCCP UBA 2006]). In other reports it was stated that SCCP has not been used in PVC in the EU at least since 1994 ([SCCP DG Env 2007]; [SCCP ERA 2008]; [POPRC 6/INF/15]). Especially in the Technical guidance document [SCCP ERA 2008] it is explained in detail that this was an error in the report figures. SCCP in PVC was only found in PVC mats which have been 30 years old and not in nowadays PVC products.

According to this information this probable historical use of SCCP in the EU is not further investigated.

6.5.5.3 Unintentional production

It is known that chlorinated hydrocarbons can occur in nature, but up to now no natural emergence of SCCP could be identified. It can therefore be stipulated that all existing SCCP is anthropogenic. One secondary source of SCCP is sewage sludge. The low water solubility as well as the high K_{ow} value lead to an enrichment of SCCP in the sludge. It is therefore expected that about 90 - 93 % of SCCP is gathered in the sewage sludge and that only 7-10 % pass through a sewage sludge treatment plant ([SCCP DETR 1997]; [SCCP BUWAL 2003]).

6.5.6 Selection of Relevant Uses, Applications and Finished Articles for SCCP

6.5.6.1 Current uses

All relevant current uses of SCCP, which are in (1) **rubber**, (2) **sealants and adhesives**, (3) **paints and varnishes** and (4) **textiles** and which can/will end up in possible waste streams are considered in this report.

The use of SCCPs in lava lamps is not considered. A single lava lamp might contain a high amount of CPs but the total amount from this use is little [SCCP BUWAL 2003]. A problem is that lava lamps are often disposed of as municipal solid waste. It has to be assumed that during the waste processing lamps will break and the SCCP will leak out and is mixed with the rest of the waste. Therefore, it has to be considered that these lamps will be land filled and incinerated in the same ratio as other MSW. The use of SCCP in Lava lamps will not be considered as the lamps become part of the household waste. Only a marking of the lava lamps with collecting points could help, but such a system is to the current knowledge not in place.

6.5.6.2 Past uses

For the use of SCCP as fat liquoring in the **leather** industry the SCCP waste did not vanish immediately with the ban of SCCP, because 95-99 % of the SCCP is embedded in the leather end-products [SCCP BUWAL 2003]. After the product life time these materials become waste. Therefore, the corresponding

waste flow is analysed in the present report.

Metal working fluids as well as fat liquoring in the leather industry are considered to be SCCP free since 2004. As metal working fluids are not parts of the final product, the ceasing of SCCP in this industry implements, that also SCCP containing waste ceased. Therefore, these industries are not further investigated in this report.

While literature (especially older literature sources) indicate that significant amounts of short-chain chlorinated paraffins were used in PVC in the late 1990s ([SCCP BUWAL 2003]; [SCCP UBA 2006]), recent reports state that this was an error. Moreover, industry has indicated that this was an error in the reported figures and that short-chain chlorinated paraffins were not used in this application over the period for which data has been collected (1994 to the present), ([SCCP ECHA 2008]; [SCCP DG Env 2007]; [SCCP POPRC 6/11]). However, in 30 year old PVC mats, which have been analysed in 2002, SCCP could be detected, [SCCP ECHA 2008]. It is however not known whether this mat has been produced in Europe, as the country of production was not mentioned. However, the amount of SCCPs which is possibly still present in old PVC products is considered to be low. The flow of SCCPs from PVC to waste is therefore not analysed in the present report.

6.5.7 Rubber

A) Background

One of the four main applications of SCCPs is the use in the rubber industry for underground mining. Other rubber applications are shoe soles, industrial sheeting, gaskets and hoses ([SCCP RPA 2010]; [SCCP ECHA 2008]).

The key application for rubber conveyor belts, which are fire retarded with SCCP, is in underground coal mining. Different rubbers can be used for this purpose as natural rubber, styrene and butadiene rubber, polybutadiene rubber, acrylonitril, EDPM elastomer (terpolymer of ethylene, propylene and di- or polyene), chlorbutadiene (e.g. chloroprene) and PVC. Each of these rubbers requires different SCCP concentrations [SCCP RPA 2010]. The rubber may be used as a skim coat and in form of a cover. The conveyor belt itself can be made with steel or with textile carcasses. Especially the steel carcasses are equipped with rubber covers whereas conveyor belts with textile carcasses tend to have either a rubber or PVC cover [SCCP RPA 2010]. The amount of SCCP used for conveyor belts within the rubber industry is estimated to be about 80 %. The remaining SCCP in the rubber industry is used for shoe soles, gaskets and hoses, which is explained later in this report.

It is stated that some 30 years ago flame retarded conveyor belts based on styrene and butadiene rubber had to be replaced by self-extinguishing conveyor belts based on chloroprene rubber in European coal mining (this might not be true across the EU due to different safety requirements) [SCCP RPA 2010]. Since chloroprene is highly fire resistant by its nature, only a little amount or no flame retardants are necessary. PVC shows a similar behaviour. [SCCP RPA 2010]

Assuming an SCCP consumption of 162 t/y [SCCP RPA 2010], (see Table 6-50) and that conveyor belts account for 75 – 90 % [SCCP RPA 2010] of the SCCPs consumption, with an average SCCP concentration of 10% the amount of rubber that may contain SCCPs is about 1,215 - 1,460 t/y [SCCP RPA 2010] of rubber for conveyor belts.

The remaining 10 - 25 % [own calculation for 2010] for the other rubber applications correspond to 16.2 to 40.5 t/y [own calculation 2010]. With a concentration range of 10 – 17 % [SCCP ECHA 2008], this represents rubber amounts of 95 to 405 t/y [SCCP RPA 2010], with an average value of 250 t. Therefore, SCCPs are of limited importance to the wider rubber industry or indeed the conveyor belt industry, although it is correct that SCCP-containing rubber has been used in some niche markets with particular safety requirements, such as the underground coal mining sector [SCCP RPA 2010]. It has been estimated that about 33,800 t/y of conveyor belts containing SCCP and about 204,200 t/y of SCCP free conveyor belts have been disposed of in 2010.

The annual estimated tonnage used in the EU is summarised in Table 6-50 and a more detailed distribution and extrapolation for the EU 27 is provided in Table 6-62.

Table 6-50: Estimated tonnage of SCCPs used in the EU (t/y) ([SCCP ECHA 2008]; [SCCP RPA 2010])

Use area	1994 (EU 15)	1995 (EU 15)	1998 (EU 15)	2001 (EU 15)	2003 (EU 15)	2004 (EU 15)	2007 (EU 27)	2009 (EU 27)
Rubber [t/y]	1,310	Na	638			< 600	Major use	162

Table 6-51: Overview of the extrapolated SCCP consumption for EU 27 in the rubber industry in the last decades.

Year	SCCP consumption in the rubber industry [t/y]	SCCP for conveyor belts [t/y]	SCCP for other products [t/y]
1979	1638	1310	328
1980	1638	1310	328
1981	1638	1310	328
1982	1638	1310	328
1983	1638	1310	328
1984	1638	1310	328
1985	1638	1310	328
1986	1638	1310	328
1987	1638	1310	328
1988	1638	1310	328
1989	1638	1310	328
1990	1638	1310	328
1991	1638	1310	328
1992	1638	1310	328
1993	1638	1310	328
1994	1638	1310	328
1995	1428	1142	286
1996	1218	974	244
1997	1008	806	202
1998	798	638	160
1999	765	612	153
2000	732	585	146
2001	699	559	140
2002	666	533	133
2003	633	506	127
2004	600	480	120
2005	527	422	105
2006	454	363	91
2007	381	305	76
2008	308	246	62
2009	235	188	47
2010	162	130	32

For the used amount in 1994 and 1998, which was only available for the EU 15, 25 % are added, which represents the ratio of inhabitants to receive the used amount for the EU 27. The resulting amounts are 1,638 t/y and 798 t/y respectively. Figure 6-69 illustrates the estimated annual usage of SCCP in the rubber industry.

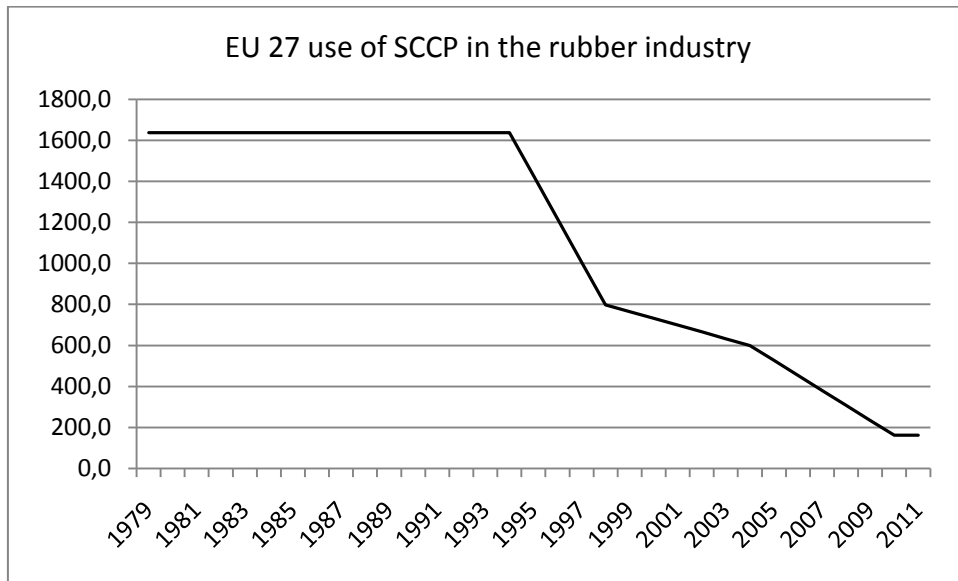


Figure 6-69: Trend of the use of SCCP in the rubber industry within the EU 27

B) Input/Output

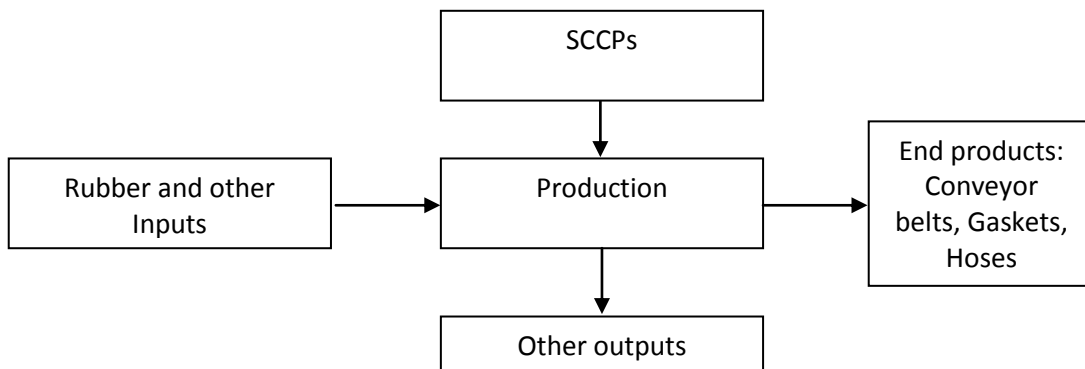


Figure 6-70: Input/Output of SCCP in the rubber industry

Input:

The relevant input of concern is the SCCPs. In the rubber industry it is mainly used for high density conveyor belts for the underground mining industry. SCCPs are additive flame retardants and so physically incorporated into the rubber matrix [SCCP ECHA 2008].

Output:

Relevant outputs of the production process are flame retarded end products containing SCCPs (currently conveyor belts and in the past gaskets and hoses).

A further source is the release from recycled SCCP-containing conveyor belts. These belts are increasingly being reused or recycled by reduction to powder and subsequent manufacture of new belts, curtains, mats and building materials [Semperit 2010]. Information on releases from recycling of the belts containing SCCP or during the service life of articles produced from the recycled rubber is not available. Especially the uses of such reused articles are likely to be wide dispersive and a potential source of SCCP releases. Other uses of rubber containing SCCPs could be in the production of technical products such as gaskets and hoses [SCCP ECHA 2008]. According to different literature sources, SCCP is mainly applied in conveyor belts for underground mining, for which it is still used in the EU 27. The use of SCCP for gaskets and hoses is considered to be smaller compared to the conveyor belts. Therefore, it is estimated that about 80 % of the SCCP is used for conveyor belts and the remaining 20 % to equal shares for gaskets and hoses. According to literature the used amount of SCCP for conveyor belts within the rubber industry is about 75-90 % [SCCP RPA 2010].

C) Generation factors

Conveyor belts:

It is assumed that from the 100 % of sold SCCP about 5% becomes waste during production as e.g. cut-offs. Therefore, the waste generation factor at production is 5 %. Consequently, 95 % of the input is contained in products placed on the market.

From the remaining 95 % of SCCP in products about 2.3 % will be emitted during the life time. The emission factor during the life time is therefore 2.185 % from the total amount of SCCP which was used for the production. Throughout the lifetime the following emissions are used: volatile loss 0.05 %; leaching loss 0.25 %; erosion/particulate loss 2 % [SCCP ECHA 2008]. This leads to an emission of about 2.3 % of the 0.95 kg in the SCCP or 2.185 % of the total input.

About 95 % of the SCCP used for conveyor belts becomes part of the sold product. After its lifetime about 92.815 % of the SCCP consumed for conveyor belts will become waste.

An average content of 10 % SCCPs in rubber for conveyor belts (major use) means that a consumption of 1 tonne of SCCPs would be related to the production of approximately 10 tonnes of flame retarded rubber. An average content of 13.5 % SCCPs in rubber for other (minor uses) means that a consumption of 1 tonne of SCCPs would be related to the production of approximately 7.4 tonnes of flame retarded rubber.

Other rubber products (Gaskets, shoe soles, hoses):

From the total SCCP sold about 20 % is used for other rubber products. Like in case of conveyor belts it is assumed that about 5 % will be lost during production.

Releases of SCCPs during service life of other rubber products such as gaskets, hoses, shoe soles are not taken into account, as the relevant data is missing.

After the product life time about 95 % of the SCCP consumed for other rubber products will become

waste.

D) Contamination Data

SCCPs are generally used at an application rate of 10 - 17 % [SCCP ECHA 2008]. In case of high density conveyor belts, used in the mining industry, the SCCP content is expected to be about 10 % in the rubber, which again has a total share of 33 % of the total conveyor belt [SCCP RPA 2010]. The SCCP concentration of the total conveyor belt is therefore expected to be about 3.3 % (33,000 ppm).

According to the Prodcom database, the total volume of conveyor belts sold in the EU was 237,880 t in 2007 [SCCP RPA 2010]. Conveyor belts account for 80 % of the total EU consumption of SCCPs in the rubber sector. With a SCCP concentration of 3.3 % the estimated consumption of 163 t of SCCPs would correspond to a conveyor belt tonnage of 3,900 t/y. This represents about 1.6 % of the total conveyor belt production. Hence, if all the conveyor belts would be mixed together, an SCCP concentration of 545 ppm would result. The share of 1.6 % of SCCP containing conveyor belts represents the current production quantities. In the past the use of SCCP was much higher and therefore a higher share of the conveyor belts containing SCCPs, currently entering the waste stream, has to be assumed.

In the following, a current annual disposal of about 1,126 t of SCCP contained in conveyor belts is deduced. (For this calculation a lognormal distribution has been used for each production year. The set parameters have been an average lifetime of 12.5 years and a standard deviation of 2.5 years. This means that about 98% of all the conveyor belts are disposed of between a service life of 2 -30 years.) With the concentration of 3.3 %, this represents about 33,800 t of SCCP containing conveyor belts, which in turn represents about 14 % of the total conveyor belt production. If all waste conveyor belts are mixed, the SCCP concentration is about 4,700 ppm. This is much higher than the current average concentration of 545 ppm due to higher historical inputs.

However, [SCCP RPA 2010] states that it is not sure whether the numbers collected from Prodcom refer only to the rubber element or the entirety of conveyor belts that may contain rubber components. For this calculation it is assumed that the weight of the entire conveyor belt has been given.

20% of the annual SCCP consumption of 163 t/y for rubbers are used for other rubber products, with a concentration of 10-17 % [SCCP ECHA 2008] and an average value of 13.5 % (135,000 ppm). About 190-330 t of other rubber therefore contain SCCP. This is an evidently insignificant amount in the context of the wider EU rubber industry [SCCP RPA 2010].

E) Activity data

About 75-90% [SCCP RPA 2010] of SCCP in the rubber industry is used for conveyor belts for underground mining. For the calculation a best estimate of 80 % has been applied.

Different information exists regarding the average lifetime of conveyor belts. One literature source estimates the average life time to be about 10 years [SCCP ECHA 2008], whereas [SCCP RPA 2010] indicates an average lifetime of 15 years. From different personal contacts it was stated that conveyer belt can have a life time between 2 and 30 years ([SCCP Sempertrans 2010]; [SCCP TiS 2010]). For the

purpose of this study a lognormal distribution was used with an average life time of 12.5 years and a standard deviation of 2.5 years. The amount of SCCP used for the production of conveyor belts, the remaining amounts in the years 2010 and 2011 from each production year and the expected disposed amounts from each year of the production are provided in Table 6-52. The values of SCCP consumption for conveyor belts from Table 6-62 and the values of SCCP in conveyor belts from Table 6-63 differ due to production losses of 5 %.

Table 6-52: Estimated tonnage of SCCP used for the production of conveyor belts in the EU (t/y)

Year	SCCP in conveyor belts [t/y]	Still in use 2009 [t/y]	Still in use 2010 [t/y]	Disposed of 2010 [t/y]
1984	1,245	0.2	0.0	0.16
1985	1,245	0.9	0.2	0.66
1986	1,245	3.2	0.9	2.32
1987	1,245	10.2	3.2	7.02
1988	1,245	28.3	10.2	18.10
1989	1,245	68.2	28.3	39.86
1990	1,245	143.1	68.2	74.95
1991	1,245	263.4	143.1	120.34
1992	1,245	428.4	263.4	164.99
1993	1,245	621.6	428.4	193.18
1994	1,245	814.7	621.6	193.14
1995	1,085	854.0	710.3	143.74
1996	925	817.7	728.4	89.37
1997	766	722.7	676.7	46.04
1998	606	591.5	572.1	19.37
1999	581	575.5	567.0	8.43
2000	556	553.8	550.7	3.14
2001	531	529.9	528.9	0.99
2002	506	505.2	504.9	0.27
2003	481	480.3	480.2	0.06
2004	456	455.3	455.3	0.01
2005	401	400.3	400.3	0.00
2006	345	344.5	344.5	0.00
2007	290	289.1	289.1	0.00
2008	234	233.7	233.7	0.00
2009	179	178.3	178.3	0.00
2010	123	0.0	(122.9)	
Total		9,914	8,788 (8,911)	1,126

It is therefore expected that about 1,100 t of SCCP bond in conveyor belts will be disposed of in 2010. Considering that the SCCP concentration is 3.3 % the total amount of conveyor belt which becomes waste is about 33,300 tonnes in 2010. The table already includes life time losses which are at the end of the life time in average about 2.3 %. From the table it can be seen that the majority of the conveyor belts which are becoming waste in 2010 have been produces between 1988 and 1998.

It is assumed that the other applications of SCCP in the rubber industry (hoses, gaskets, industrial

sheets, shoe soles) have approximately a life time of 5 to 10 years with a mean value of 7.5 years. As it can be seen in

Table 6-51, the amount of SCCP used in this relevant time range of 2001 till 2005 for other products have been 625 t or ~120 t/y. With a production loss of 5% approximately 114 t/y are included in products. Considering a concentration of about 13.5 %, the total produced rubber amount has been ~844 t/y. As a rough estimate it is considered that the annual product input between 2001 and 2005 represents the waste output in 2010 (i.e. approximately 800 t of waste containing about 114 t of SCCP).

As the life time of rubber products other than conveyor belts is about 7.5 years, the amounts of SCCP still in use of these products is 7.5 times higher than the annual consumption of 114 t/y. This results in 855 t of stored SCCP in other rubber products that are still in use.

F) Waste Treatment

Industry and literature provide information that conveyor belts can be collected and recycled. Possible recycling options after powdering are playground safety mats or floor coverings for children playgrounds and animal mats and flooring. Also some reuses as carpets have been mentioned (see [SCCP Sempertrans 2010]; [SCCP TiS 2010]).

Most conveyor belts or hoses contain a lot of reinforcing fibres or steel, much more than for instance used in a tyre. Although fibres and steel can be separated during the recycling process, the high amount present in conveyor belts could cause problems. [SCCP RPA 2010]

Intensive investigations have shown that even though some conveyor belts might be recycled, the amount of recycled SCCP containing conveyor belts should be negligible. [SCCP RPA 2010]

It is also mentioned that one rubber company has tested its reclaimed material for SCCPs, Polybutadien rubber (BR) reclaim and natural rubber (NR) reclaim. Both had a SCCP content of <10 mg/kg (<10 ppm) [SCCP RPA 2010].

In case of shoe soles, gaskets and hoses waste it is considered that the major part is non-hazardous waste and is disposed of as such. Due to their nature, these products are widely distributed in the countries and have to be considered as diffuse sources of SCCP containing waste and will therefore end up similar as non-hazardous waste.

Throughout the life time the following emissions are used: volatile loss 0.05 %; leaching loss 0.25%; erosion/particulate loss 2% [SCCP ECHA 2008]. This sums up to about 2.3 %. Using the average life time from the lognormal distribution an annual loss has been calculated of 0.15 %. With the remaining conveyor belts, still used from the different production years, the emission in 2010 has been calculated to be about 11.3 t.

No data is available regarding SCCP containing waste from the production of SCCP containing rubber materials. It can be assumed that a typical loss is about 5 %. Considering the use of about 162 t of SCCP in 2010 the amount of production waste which is disposed of is about 8 t/y. This amount represents the

production of conveyor belts as well as other rubber products including SCCP. This amount is very low compared to the amount of SCCP containing conveyor belts.

G) *Substance flow*

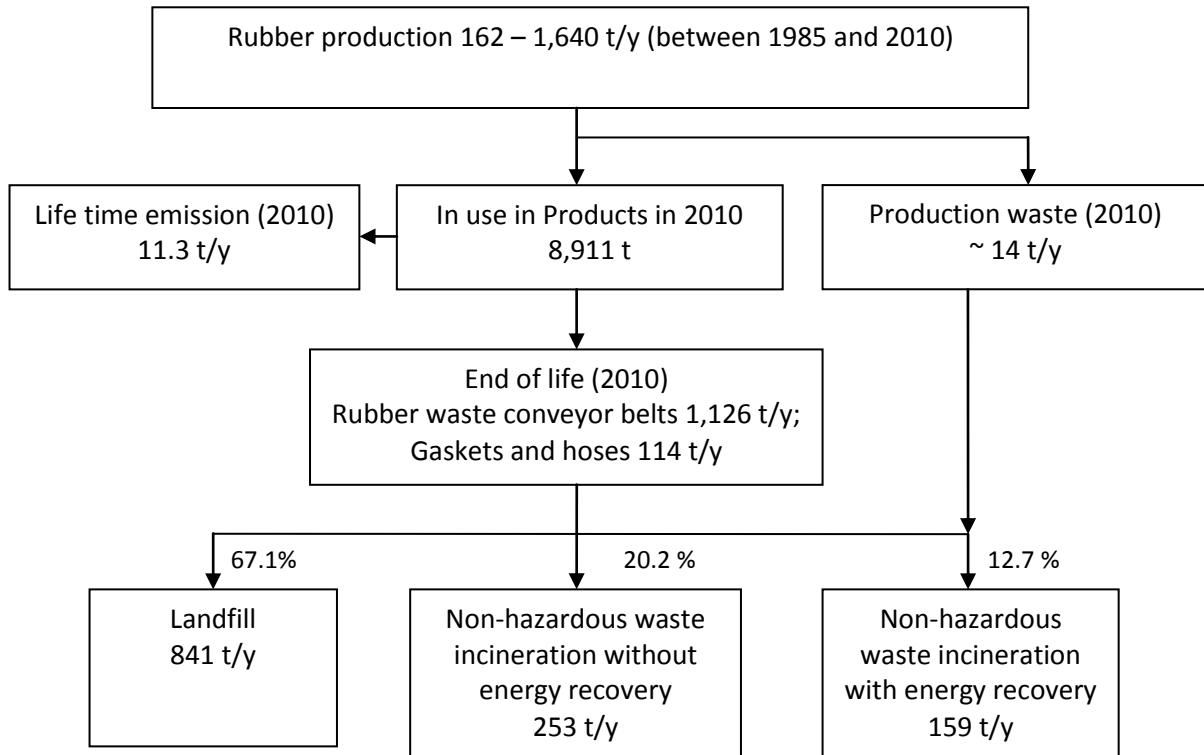


Figure 6-71: Substance flow of SCCP in the rubber industry

It is considered that the rubber waste is treated similar to MSW but it is neither composted nor recycled. It is therefore assumed that about 67.1 % of the conveyor belts are landfilled, 20.2 % are incinerated without energy recovery and 12.7 % are incinerated with energy recovery. As conveyor belts have a high caloric value they could easily be used as solid fuels e.g. for cement kilns and therefore the estimated share for incineration with energy recovery might be considered too little.

H) Material flow

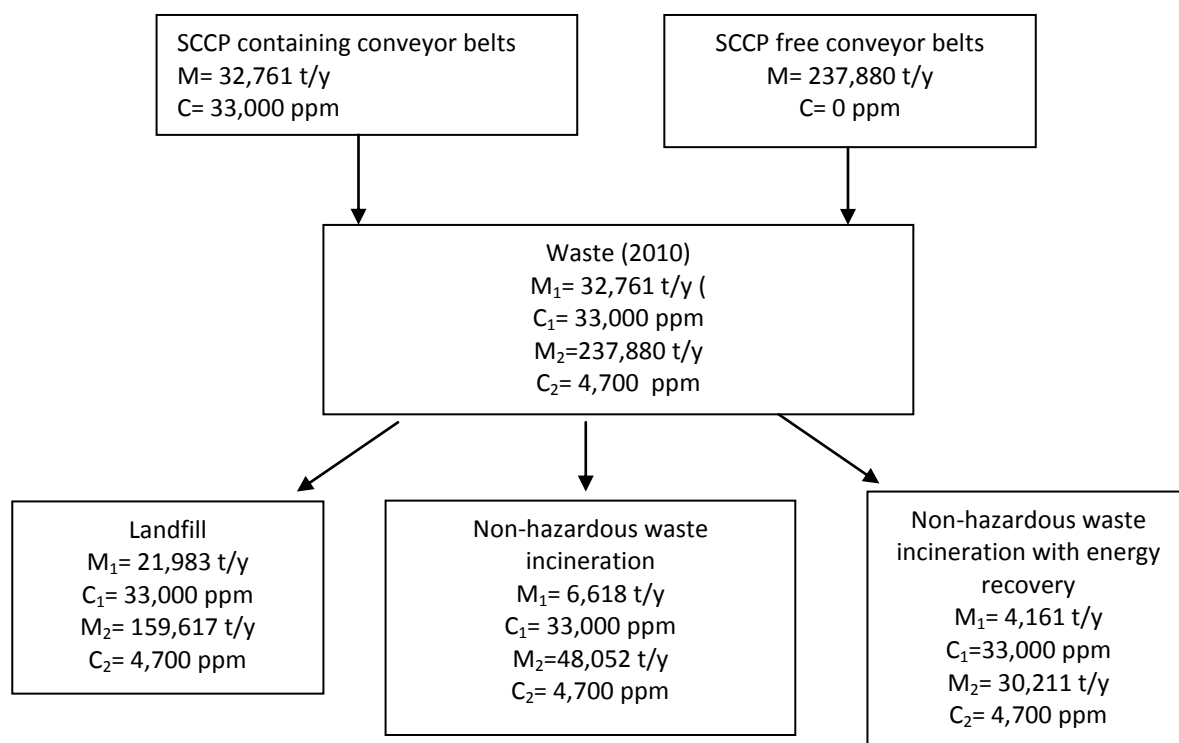


Figure 6-72: Material flow of SCCP in the rubber industry

It is considered that the annual production of conveyor belts has been constant in the last decades with about 238,000 tonnes per year, which therefore have to be disposed of annually. About 32,800 tonnes of SCCP containing conveyor belts have been disposed of in 2010. This means that about 204,200 tonnes of SCCP free conveyor belts have been disposed of in 2010.

M₁ and C₁ represent the numbers if the SCCP contaminated conveyor belts are collected separately. M₂ and C₂ represent the numbers if the SCCP contaminated conveyor belts are mixed with SCCP free conveyor belts.

Waste streams from gaskets, hoses and shoes are not described as they only represent about 2.5% of the waste stream.

6.5.8 Sealants and adhesives

A) Background

SCCPs are used in sealants and adhesives. However the distinction of sealant and adhesive is difficult, as some sealants can be used as adhesive and vice versa [SCCP ECHA 2008]. The term 'sealants' is therefore used in this report for both – sealants and adhesives.

The SCCP is physically incorporated (mixed) into the sealant where it acts as fire-retardant and/or plasticiser. It is contained in Polymers as acrylic and butyl sealants, polyurethane, polysulfide but it is not included in modern silicone rubber. [SCCP ECHA 2008]

SCCP containing sealants have a wide range of application such as the filling of expansion and movement joints, filling of gaps around doors, windows, dorm windows and arches, for sealants for water storage applications (reservoirs) but also protecting areas from oil and fuel spillages, areas around petrol stations, sewage treatment works, sealants for underground facilities such as basements and subways but also for the water proofing of constructions where water flow occurs such as bridges and culverts. Sealants are also used for automotive windows and sealants that may act as intumescent or as sealant for water proof roof coating [SCCP ECHA 2008]. Chlorinated paraffins with high chlorine contents are also used in sealants for double and triple-glazed windows [SCCP BUWAL 2003]. 5-20% of the tonnage sold to the sealants industry is associated with dam sealants (spillways and sea defence) [SCCP RPA 2010].

Figures on the EU consumption of SCCPs for sealants are provided by [SCCP ECHA 2008] and [SCCP RPA 2010]. For the year 1994 the SCCP amount was increase by 25 % to account for the new EU 12 Member states on the basis of a per capita approach. The amount in 2004, representing the EU 25 has not been adapted as it should represent the EU 27 well enough. In Figure 6-73 the line represents the extrapolated amount of SCCP for the EU 27, whereas the dots show the information from the literature and Table 6-53.

Table 6-53: Estimated tonnage of SCCP used in the EU (t/y) [SCCP ECHA 2008]

Use area	1994 (EU 15)	1995 (EU 15)	1998 (EU 15)	2001 (EU 15)	2003 (EU 15)	2004 (EU 25)	2007 (EU 27)	2010 (EU 27)
Rubber [t/y]	695	n.a.				< 300	Major use	237

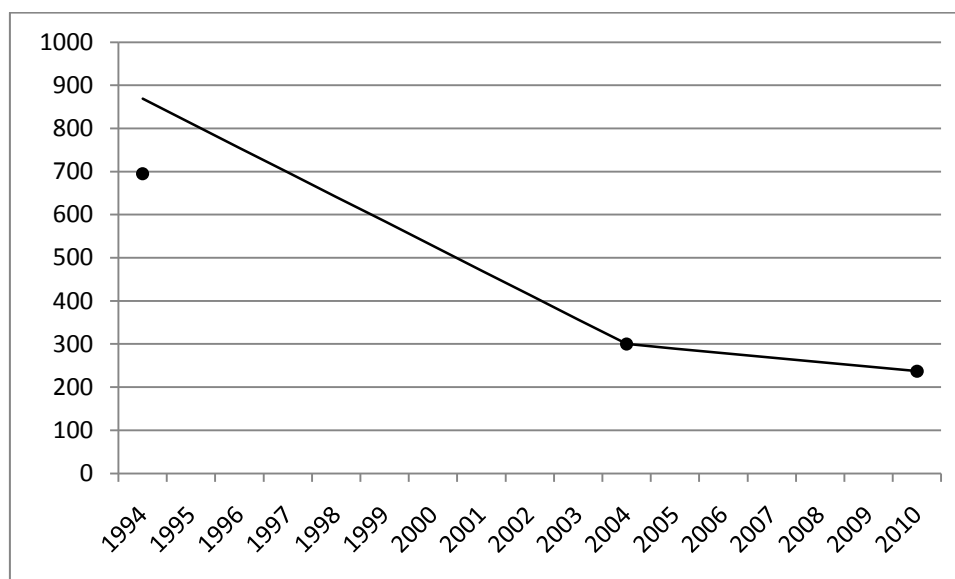


Figure 6-73: Annual consumption of SCCP for sealant and adhesives from 1994 to 2010.

Table 6-54 gives an overview of the different amounts of SCCP which have been consumed in the sealant industry in the last 17 years with detailed information about losses from processing as well as life time emissions and the corresponding waste amounts.

Table 6-54: Overview of SCCP and waste amounts in the EU 27 within the last 17 years

Year	SCCP consumption [t/y]	SCCP waste from processing [t/y]	Applied SCCP amount [t/y]	SCCP lifetime emissions [t/y]	SCCP in product [t/y]	Waste amount [t/y]
1994	869	43	825	66	759	3,796
1995	812	41	771	62	710	3,547
1996	755	38	717	57	660	3,299
1997	698	35	663	53	610	3,050
1998	641	32	609	49	560	2,802
1999	584	29	555	44	511	2,553
2000	528	26	501	40	461	2,305
2001	471	24	447	36	411	2,057
2002	414	21	393	31	362	1,808
2003	357	18	339	27	312	1,560
2004	300	15	285	23	262	1,311
2005	290	14	275	22	253	1,265
2006	279	14	265	21	244	1,219
2007	269	13	255	20	235	1,173
2008	258	13	245	20	225	1,128
2009	248	12	235	19	216	1,082
2010	237	12	225	18	207	1,036
Total	8,008	400	7,607	609	6,999	34,993
Average	471	24	447	36	412	2,058

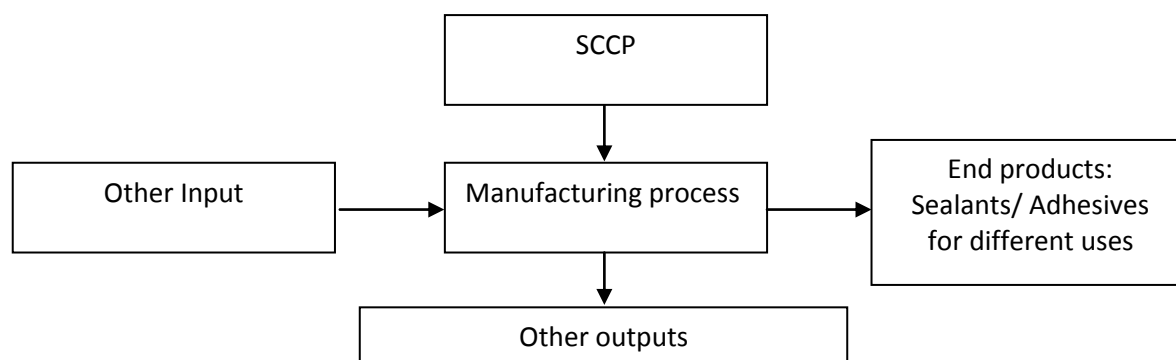
B) Input/Output

Figure 6-74: Input/Output of SCCP in the rubber industry

Input:

The relevant inputs of concern are the SCCPs. It is thought that SCCP is more often used in sealants than in adhesives. However the distinction of sealant and adhesive is difficult, as some sealants can be used as adhesive and vice versa [SCCP ECHA 2008].

Output:

The relevant outputs of concern are SCCP containing sealants. These are used in many applications as packaging, automotive or construction and building.

C) Generation factors

From the 100 % of SCCP used in the sealant industry it is assumed that about 5 % are becoming waste during the processing step. 95 % of the SCCP is placed as sealant on the market.

From the 95 % of the SCCP sold in sealants it is assumed that about 8 % are emitted during lifetime. The life time generation factor in this industry is therefore 7.6 % of the total amount of SCCP used from the sealant industry. Throughout lifetime the following emissions are used: volatile loss over life time 0.25 %; leaching loss over life time 0.75%; Erosion/particulate loss over life time and disposal each 2-5% using 3.5 % for the calculation. [SCCP ECHA 2008]

From the 95 % of the SCCP sold in sealants about 92 % will become waste. The end of life generation factor is therefore 87.4 % of the consumed SCCP. The consumed SCCP amounts also include the 5% losses during sealant production.

An average content of 20 % SCCPs in sealants means that a consumption of 1 tonne of SCCPs would be related to the production of approximately 5 tonnes of flame sealants.

D) Contamination Data

The chlorinated paraffin is typically added at a concentration of 5-14 % by weight, but concentrations up to 20 % (200,000 ppm) by weight can also be used in exceptional cases [SCCP ECHA 2008]. Other

literature [SCCP RPA 2010], state that the concentration of SCCPs may vary widely and can be between 2 (in the adhesive for road marking strips) and 95% (in accelerators for two-part sealants). The most common concentrations shown in Safety Data Sheets are generally 10-30% (but note that the concentrations shown may (but not always) only reflect the presence of SCCPs in one of the two parts of the sealant in case of a two-part sealant. For the calculation a concentration of 20 % has been used as a best estimate. With the annual amount of 237 tonnes of waste a total amount of approximately 1,185 tonnes results. This tonnage is a very small fraction of the EU consumption of sealants and adhesives of 3 million tonnes in 2009 [SCCP RPA 2010]. If all sealants would be mixed together an average concentration of about 80 ppm would result. [SCCP RPA 2010]

It has to be considered that sealants or adhesives are tightly bond on the surface they have been applied on. A correctly sorted waste stream, mainly consisting of sealants and adhesives is therefore nearly impossible.

E) Activity data

About three million tonnes of product from the sealant and adhesive industry have been consumed in 2009. [SCCP RPA 2010] Compared to this the 237 t of SCCP or the amount of approximately 1,200 tonnes of SCCP containing sealants and adhesives is only small.

The share of different industries using sealants and adhesives are shown in the following table:

Table 6-55: End-use market share of sealants and adhesive in the EU 27 in 2009 [SCCP RPA 2010]

End-use market	Market share [% of tonnage]
Packaging	36
Wood working	10
Transportation	8
Consumer/DIY	9
Other	36
Total	100

Given that sealants and adhesives are used for a very broad range of product with very different life times ranging from several months (packaging) up to several decades (building and construction). It is impossible to give an accurate estimation about the current accumulated amount as historical data are missing and a more detailed identification of current and past uses in this industry as well as shares would be necessary.

F) Waste Treatment

Due to the nature of sealants and adhesives it has to be assumed that when the different application enter the waste stream the sealant and adhesives will be bond on the surface of the material where it was applied and will be part in the waste treatment steps of different waste flows.

Remainder of Sealants and adhesives in the package are considered to be disposed of shortly after application. It is assumed that the majority of the material will end up as MSW and will neither be recycled nor composted. The waste is therefore landfilled to 67.1 %, incinerated without energy

recovery to 20.2 % and incinerated with energy recovery 12.7 % see Table 6-67.

G) *Substance flow*

About 5% of the SCCP becomes waste during its processing step because it will not be completely emptied. It will stay in the tube or container and will be disposed of short after application.

Throughout lifetime the following emissions are used: volatile loss over life time 0.25 %; leaching loss over life time 0.75%; Erosion/particulate loss over life time and disposal each 2-5% using 3.5 % for the calculation. [SCCP ECHA 2008] This sums up to emission of about 8 %. This emission of 8 % is only relevant for the 95% of SCCP which are applied on surfaces and represent 7.6 % of the total amount of SCCP in the Sealant industry. For the annual emissions 8% the annual application of SCCP in sealants (447 t/y) have been used for calculation, which are about 36 t/y.

It is assumed that about 8,000 t of SCCP has been used for sealant production since 1994. The average use has therefore been 470 t/y. There are no information available about the amount which is still in use or which amount become waste every year from the applied sealants. Due to this lack of knowledge it has been considered that the annual input from the last 17 years represents the annual waste stream occurring nowadays. From the 470 t/y only 447 t/y (or 95 %) are applied and 8% of lifetime emissions are considered and therefore about 412 t/y of SCCP are assumed to be disposed of in 2010. Due to the wide use of Sealants, automotive, construction or DIY the typical disposal of 67.1, 20.2 and 12.7 % for landfilling, incineration without energy and incineration with energy recovery respectively have been considered.

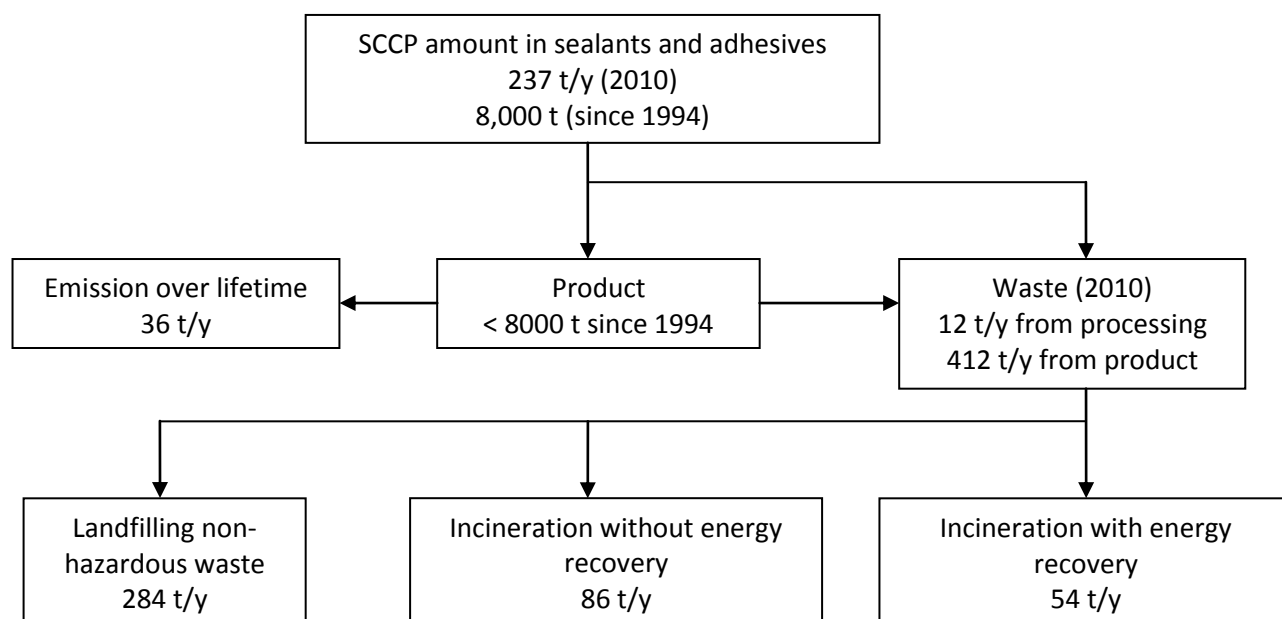


Figure 6-75: Substance flow of SCCP in sealants and adhesives

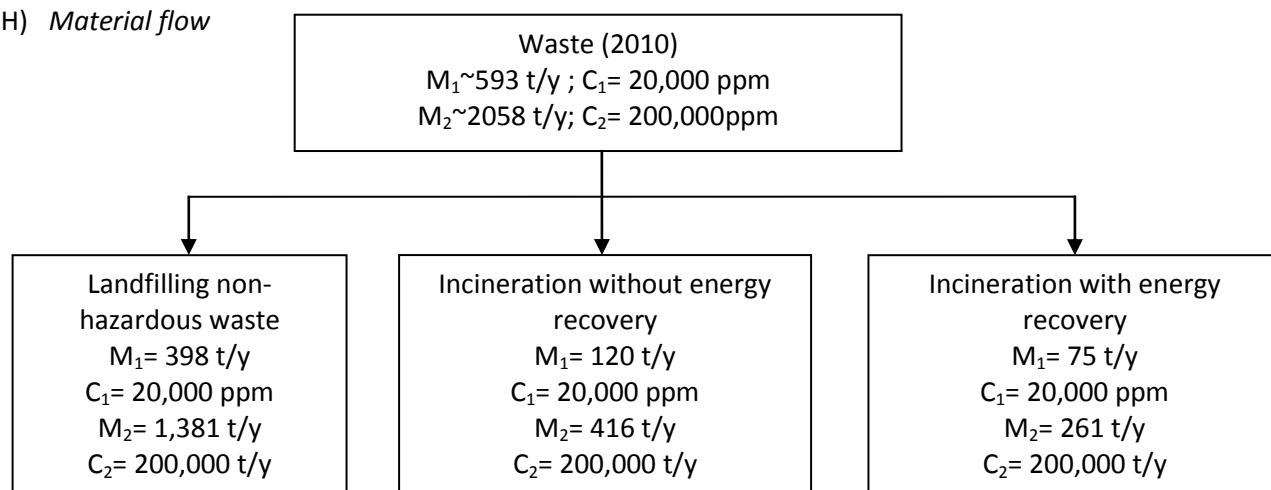
H) *Material flow*

Figure 6-76: Material flow of SCCP in sealants and adhesives

It is assumed that an average 500 ml container for sealants and adhesives weighs about 750 grams (250 g container material and 500 g sealant or adhesive), 5 % of the sealant is considered to be left in the container representing 25 g or 10 % of the total empty container. The 25 g of sealant include about 20 % or 5 g of SCCP. The total SCCP concentration is therefore about 2% or 20,000 ppm.

M_1 and C_1 represents the amount and concentration of the waste stream from the processing step. M_2 and C_2 represent the amount of sealant and the SCCP concentration at the end of life with the simplification that no mixing takes place.

6.5.9 Paints and varnishes

A) Background

The main function of chlorinated paraffin in paints is as a plasticiser but they can also be used to improve water resistance, chemical resistance, corrosion resistance [SCCP RPA 2010] and the non-flammability of paints [SCCP ECHA 2008].

In a study of the British coating Federation it was tried to identify the number of sites where coatings containing chlorinated paraffins might be used in the United Kingdom, but this did not prove to be possible. The major users of the paints are professional painters and specialist applicators, but some DIY paints containing chlorinated paraffins may be used by the general public. In the United Kingdom, it was estimated that there would be around 40,000 users of coatings containing chlorinated paraffins for water proofing of walls, with around 1,000-1,500 users of paints and coatings for other uses. [SCCP ERA 2008]

Table 6-56: Estimated tonnage of SCCP used in the EU (t/y) [SCCP ECHA 2008]

Use area	1994 (EU 15)	1995 (EU 15)	1998 (EU 15)	2001 (EU 15)	2003 (EU 15)	2004 (EU 25)	2007 (EU 27)	2010 (EU 27)
Paint [t/y]	1,150	n.a.	726			< 100	Minor use	101

In the year 1998 the data were only available for the EU 15 to estimate the use of SCCP in the paint and varnishing industry for the EU 27 the use would be 1437 tonnes in 1994 and 907 tonnes in 1998. The line in Figure 6-77 represents the figures extrapolated on the basis of a per capita approach for the EU 27 whereas the dots represent the amounts given by [SCCP ECHA 2008].

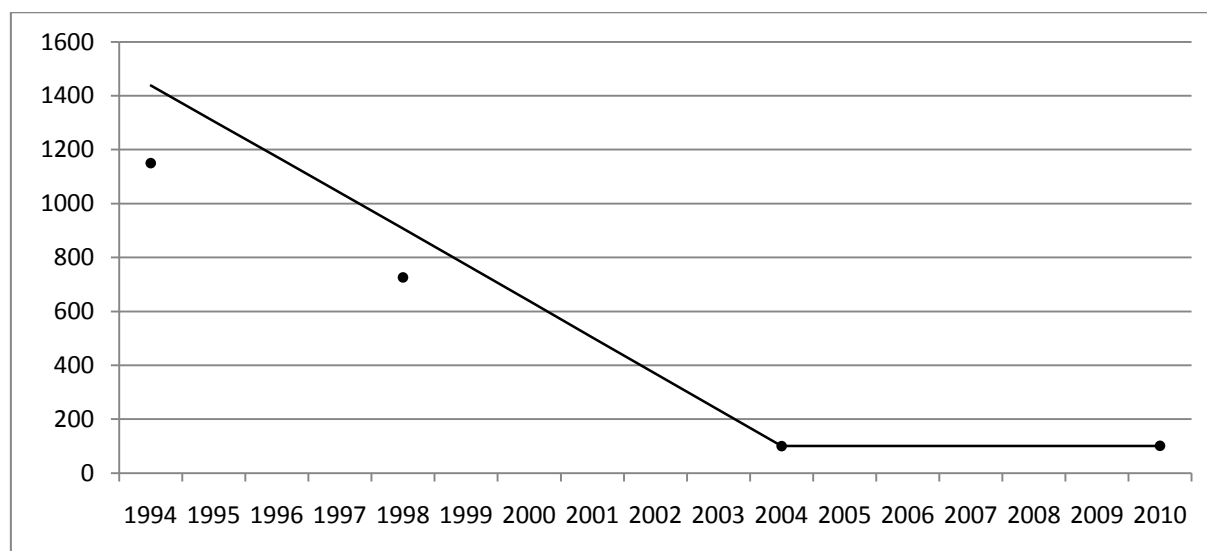


Figure 6-77: Annual consumption of SCCP for paint and varnishes.

Table 6-57 gives an overview of the different amounts of SCCP which have been consumed in the paint industry within the last 17 years with detailed information regarding processing as well as life time emissions and waste amounts.

Table 6-57: Overview of SCCP and waste amounts in the EU 27 within the last 17 years

Year	SCCP consumption [t/y]	SCCP waste from processing [t/y]	Applied SCCP amount [t/y]	SCCP lifetime emissions [t/y]	SCCP in product [t/y]	Waste amount [t/y]
1994	1,150	345	805	45.4	760	5,842
1995	1,044	313	731	41.2	690	5,304
1996	938	281	657	37.1	620	4,765
1997	832	250	582	32.9	549	4,227
1998	726	218	508	28.7	479	3,688
1999	622	187	435	24.5	411	3,158
2000	517	155	362	20.4	342	2,628
2001	413	124	289	16.3	273	2,098
2002	309	93	216	12.2	204	1,568
2003	204	61	143	8.08	135	1,038
2004	100	30	70	3.96	66	508
2005	100	30	70	3.96	66	509
2006	100	30	70	3.97	66	510
2007	101	30	70	3.97	66	511
2008	101	30	70	3.98	66	511
2009	101	30	71	3.99	67	512
2010	101	30	71	3.99	67	513
Total	7459	2,238	5,221	295	4,926	37,892
Average	438.74	132	307	17.35	290	2,229

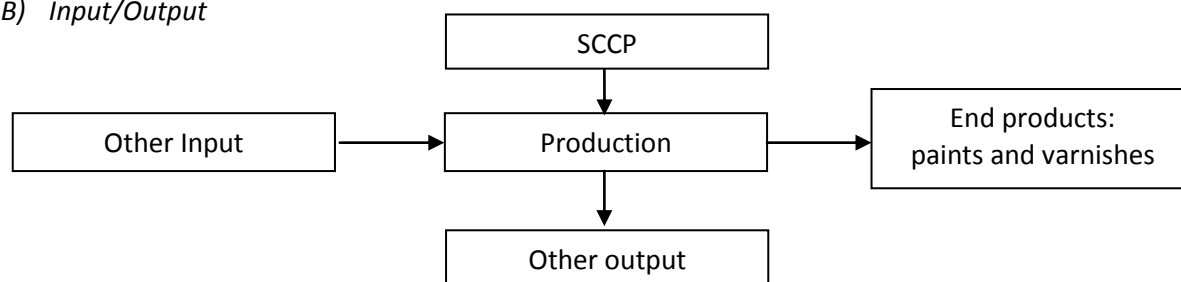
B) Input/Output

Figure 6-78: Input/Output of SCCP in the rubber industry

Input: The relevant inputs of concern are SCCPs. In [SCCP RPA 2010] it is indicated that the predominant types of chlorinated paraffins used in paints are long-chain grade length. The main applications of SCCP for paints are considered to be in chlor-rubber and acrylic protective coatings (e.g. road marking paints) as well as intumescent paints [SCCP RPA 2010]. According to a RPA report [SCCP RPA 2010] road marking paints appear to be a key application.

Output: The paints are used mainly in industrial/specialist applications such as marine primer paints and fire retardant paints. The SCCP is mixed into the paint during the formulation step and becomes physically entrained in the coating once applied [SCCP ECHA 2008].

Other applications include road marking paints, anti-corrosive coatings for metal surfaces, swimming pool coatings, decorative paints for internal and external surfaces, and primers for polysulfide expansion joint sealants, SCCPs may also be used in cross-linkable polyester systems with peroxides for the production of long-term road markings and it may be found in unsaturated polyester resin which is used in the production of fibre reinforced composites. [SCCP RPA 2010]

C) Generation factors

Considering an average content of around 10% in SCCP containing paints and varnishes a consumption of 1 t of SCCPs would be related to the production of approximately 10 t of end products (see Table 6-58).

It is assumed that in the processing step only 70% of the paint is applied whereas the other 30 % become waste during application.

According to ECHA the following emission during life time are given: volatile loss over life time 0.4 %; leaching loss over life time 1 %; Erosion/particulate loss over life time and disposal 2 - 6.5 % using 4.25 % for calculation [SCCP ECHA 2008]. This sums up to emissions of about 5.65 %. From the 70 % of the applied SCCP 5.65 % will be emitted during lifetime and the amount of SCCP ending up in waste is therefore only 66 %.

Due to the drying of the paint the SCCP concentration increases by about 25 to 33 % [SCCP ECHA 2010]. In the calculation of the waste flow it is therefore considered, that the SCCP concentration increased from 10 % in wet paint (original product) to about 13 % in dry paint (state of the product when it

becomes waste).

D) Contamination Data

Table 6-58: Overview of the content of SCCPs in SCCP containing paints (t/y) [SCCP ECHA 2008]

Paint Type	Use	SCCP content [%]
Organic solvent borne chlorinated rubber primers and topcoats	Used among others as top cover for zinc rich primers	1-5
Organic solvent borne chlorinated rubber systems	Swimming pools/fishponds	5-20
Organic solvent borne zinc rich (epoxy) primers	Metal: 80 -90 % zinc dust by weight. Used as primers for steel in extremely corrosive conditions such as maritime. Alkyd resin can be top-coated with chlorinated rubber or acrylic resin (bridges, ships)	2-5
Organic solvent borne acrylic container coatings	Metal: any kind of container.	2-10
Organic solvent borne chemical and water resistant coatings	Metal	5-20
Organic solvent borne vacuum metallising lacquers	Plastic: The vast majority of the paint used for plastic coating is applied by the automotive industry (~75 %). The other 25 % are mainly used for coatings in the electronic industry, e.g. for TV's, Hi-Fi's and computer housings. [SCCP VOC 2009]	1-5
Organic solvent borne flame retardant coating for wood	Wood	1-5
Organic solvent borne intumescent coating for structural steel	Steel	20-30
Organic solvent borne floor paints	Concrete	5-10
Organic solvent borne water-proofing coatings for walls	Concrete	5

The total European market for paints and varnishes was estimated to be about 6.67 to 9.1 million metric tons in 2006. The current market in the EU is estimated to be about 7.6 million tonnes [SCCP RPA 2010]

In intumescent coatings the SCCP concentration may range from 2.5 to 10 % ([SCCP ECHA 2008], [SCCP RPA 2010]), but there are products with very low concentration for SCCPs (< 0.025%) which is apparently the result of using MCCP [SCCP RPA 2010]. In road marking paints it can be fairly low from < 1% to 10 % but typically towards the lower end of this scale. In anti-corrosive and protective coatings the SCCPs concentrations could be 10-15%. [SCCP RPA 2010]

According to [SCCP ECHA 2008] a SCCP concentration of 10 % is considered typical for most paint types and is therefore used as a best estimate for the calculation. Additional it is stated in [SCCP RPA 2010] that Eurochlor suggests a SCCP concentration of 4-15% of the paint and 5-20% after drying.

E) Activity data

Recent estimates for use of SCCPs in paints and coatings in EU 15 indicate that the use of SCCP for paint and varnishes is only a minor amount with < 100 t/y (2004). However, it is considered that there is now little or no use of SCCPs in paints and coatings in the EU, because SCCPs are proposed as a PBT substance [SCCP ECHA 2008]. Another report [SCCP RPA 2010] indicates that the annual consumption of SCCP is currently about 101 t/y. Considering a loss during application of 30 % (30 t/y) and life time emissions of another 5.65 % (4 t) the amount of SCCP in paints which will become waste from the production from 2010 is about 67 t. Since 1994 a total amount of about 4,900 t of SCCP have been used in the EU 27 in the paint and varnish industry, with an average annual input of 290 t/y.

The annual product amounts of SCCP containing waste vary between 500 to 5,800 kg/y with an annual average of 2,200 kg/y within the last 17 years. This tonnage is a very small fraction of the estimated EU market for paints of 7.6 million metric tons per year [SCCP RPA 2010].

SCCP are of limited importance to the wider paints and coatings industry. However, this may not be true for all stakeholders (for instance road marking paint manufacturers). [SCCP RPA 2010]

The two main materials onto which SCCP containing paints and varnishes are used on are: metal and concrete. Other materials as plastic and wood are less often coated with SCCP containing paints.

For most application SCCP free alternatives are already available (e.g. SCCP free water based road marking paints.) and the share of SCCP containing waste is therefore very little. If all paints are mixed together an average SCCP content of about 13 ppm (101/7,600,000) would result.

F) Waste Treatment

It is assumed that for the majority of the painted waste containing SCCP it is not possible to be collected and treated in an appropriate manner. Due to the wide range of application a standard disposal to landfill, incineration without energy recovery and incineration with energy recovery with the percental ratio of 67.1, 20.2 and 12.7 % respectively (see Table 6-57 and Table 6-59).

G) Substance flow

As described above it is assumed that about 30 % of the used SCCP in paints becomes waste during application and the annual consumption of SCCP in paints and varnishes in 2010 has been about 101 t. This results in about 30 t/y of SCCP in Paints which becomes waste due to application of the product. Additional to this the waste arising from the applied paint also has to be taken into consideration. The annual EU consumption of SCCPs for paints and varnishes in the last 17 years has been about 7,500 t of which 5,220 t have been applied. The annual average is therefore about 307 t/y. This average annual input in the past is considered to be equal to the emissions and annual disposal amounts nowadays. Considering the average life time emissions of 5.65% (17.4 t/y) the remainder which will be disposed of is 290 t/y.

The accumulated amount of SCCP in paints, which after consideration of the life time emission, will

become waste is about 4,926 t since 1994 or in average over 17 years 290 t/y.

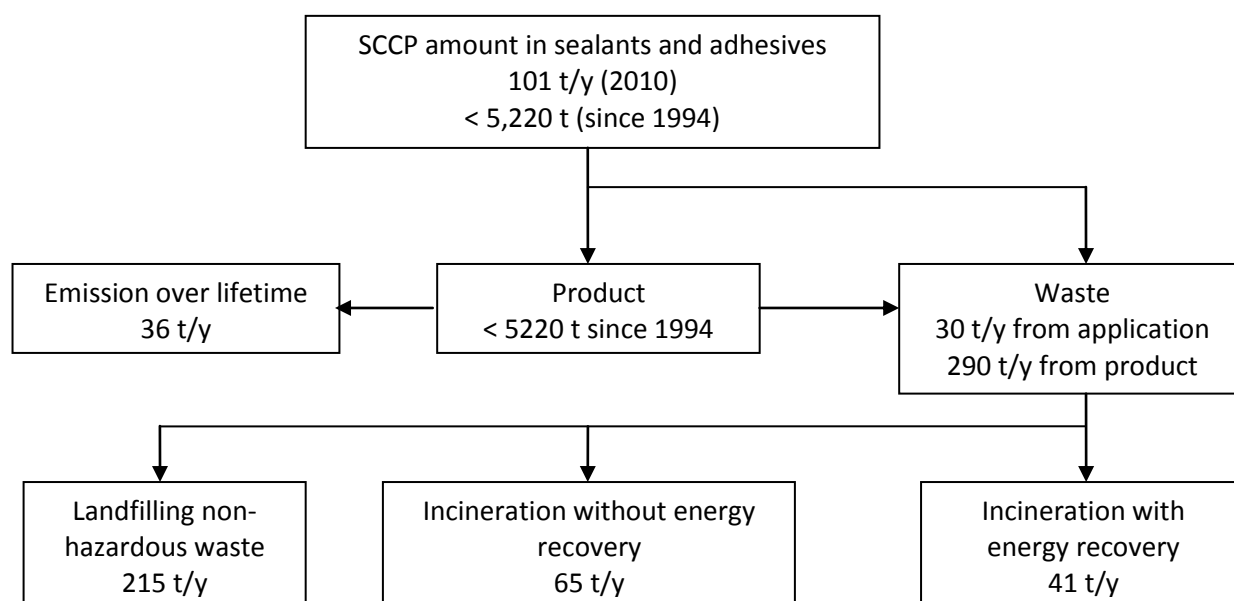


Figure 6-79: Substance flow of SCCP paint

H) Material flow

The following table shows the estimated used amounts of SCCP contained paints. For the corresponding paint amount the dry paint has been considered and therefore the SCCP concentration has been increased from 10 % in wet paint to 13 % in dry paint.

Table 6-59: Annually used amount of SCCP containing paint in the EU 27²⁸

Year	SCCP in applied paint [t/y]	Corresponding paint amount [t/y]
1994	760	5,842
1995	690	5,304
1996	620	4,765
1997	549	4,227
1998	479	3,688
1999	411	3,158
2000	342	2,628
2001	273	2,098
2002	204	1,568
2003	135	1,038
2004	66	508
2005	66	509
2006	66	510
2007	66	511
2008	66	511
2009	67	512
2010	67	513
TOTAL	4,926	37,892

Table 6-59 shows the annual input of SCCP and SCCP containing paint from the last 17 years. The values already include the life time emissions and therefore represent the amounts which will become waste at the end of lifetime.

The accumulated amount of used SCCP containing paint since 1994 is about < 38,000 tonnes. This is an average annual input of 2,229 t/y. The amount is considered to be the annually disposed of waste of SCCP containing dry paint with a concentration of 13 % in 2010.

In the year 2010, 101 t of SCCP have been used for paints. This represents about 777 t of dried paint with a SCCP content of 13 %. From this amount about 30 % or 233 t have become waste in 2010. The rest of 544 t will ended up in the product, of which 31 t will be emitted during life time.

The total amount of SCCP containing paint becoming waste is therefore estimated to be about 2,460 t/y (2,229 t/y from disposed of products from the past + 233 t/y processing waste) in 2010.

28 Lifetime emissions considered

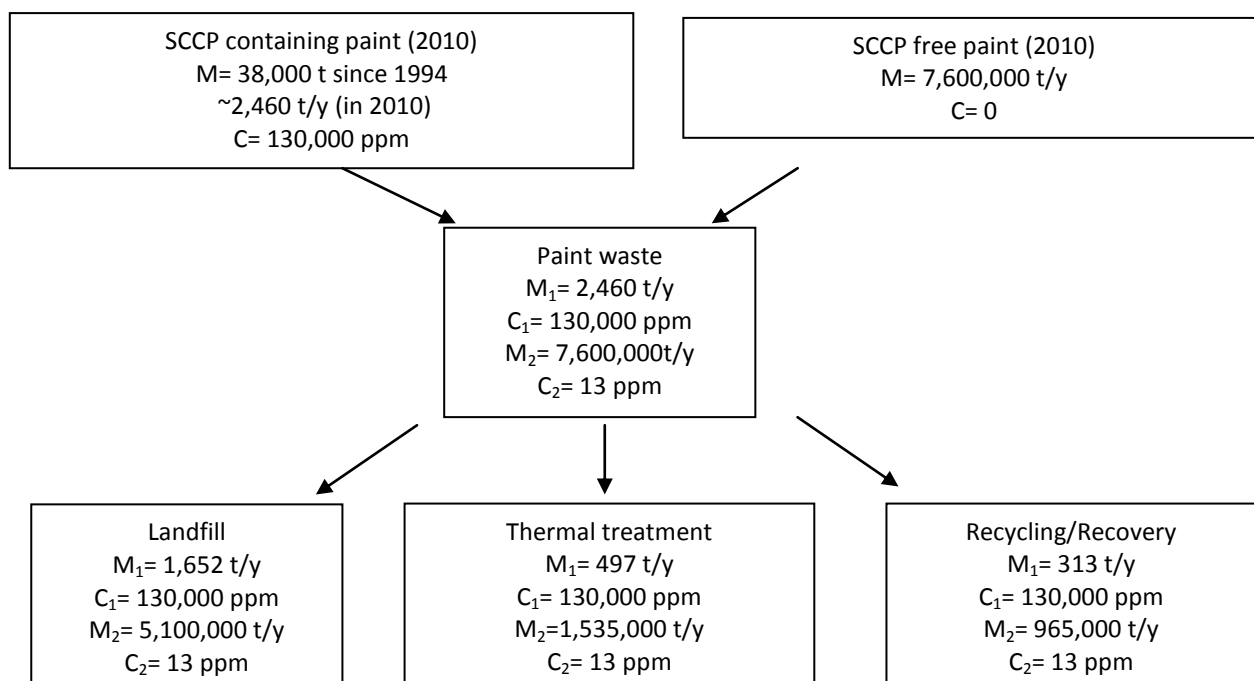


Figure 6-80: Material flow of SCCP in paints and varnishes

6.5.10 Textile

A) Background

The current uses of SCCPs in textiles are as a flame retardant for the back coating of textiles with a small amount also being used in other textile treatments such as waterproofing. In the back-coating process, the SCCP is applied to the back of the textile in a viscous polymer latex. Once cured the SCCP is effectively incorporated in a polymer matrix. The actual types of textiles for which SCCPs are used as a flame retardant are unclear. However, based on a comparison with the known usage of other flame retardants that are used in the back-coating of textiles (for example decabromodiphenyl ether and hexabromocyclododecane) it is assumed that typical applications have been in textiles in furniture upholstery, seating upholstery in transport applications and interior textiles such as blinds and curtains [SCCP RPA 2010]. It was stated, [SCCP TEGEWA 2010] that after an investigation some years ago, the phase out of SCCP in the leather industry, the textile industry joined this phase out and that since several years there are as far as known no SCCP containing Textiles manufactured within the EU 27 anymore.

At present, it is suggested that tents are the only application of relevance to SCCPs. SCCPs are used for rot-flame-water proofing of heavy textiles ($> 350 \text{ g/m}^2$ up to 750 g/m^2) [SCCP RPA 2010] with an average weight of 550 g/m^2) which require specialist equipment for their processing. The types of fibres involved may be polyester-cotton, cotton or linen-flax, it is also fair to assume that this hot-wax application of SCCP-containing emulsion has historically been prominent in the UK but not really elsewhere in the EU [SCCP RPA 2010].

The following table gives an overview of the used amounts of SCCP in the textile industry in the last years.

Table 6-60: Estimated tonnage of SCCP used in the EU (t/y) for textiles ([SCCP ECHA 2008], [SCCP RPA 2010])

Use area	1994 (EU 15)	1995 (EU 15)	1998 (EU 15)	2001 (EU 15)	2003 (EU 15)	2004 (EU 25)	2007 (EU 27)	2010 (EU 27)
Textile back-coating [t/y]	163	32				< 100	Minor use	29
Textile waterproofing [t/y]	20							

The numbers of 1994 and 1995 are only from the EU 15. However due to the reported use concentrated in the UK no adjustment has been performed for EU 27.

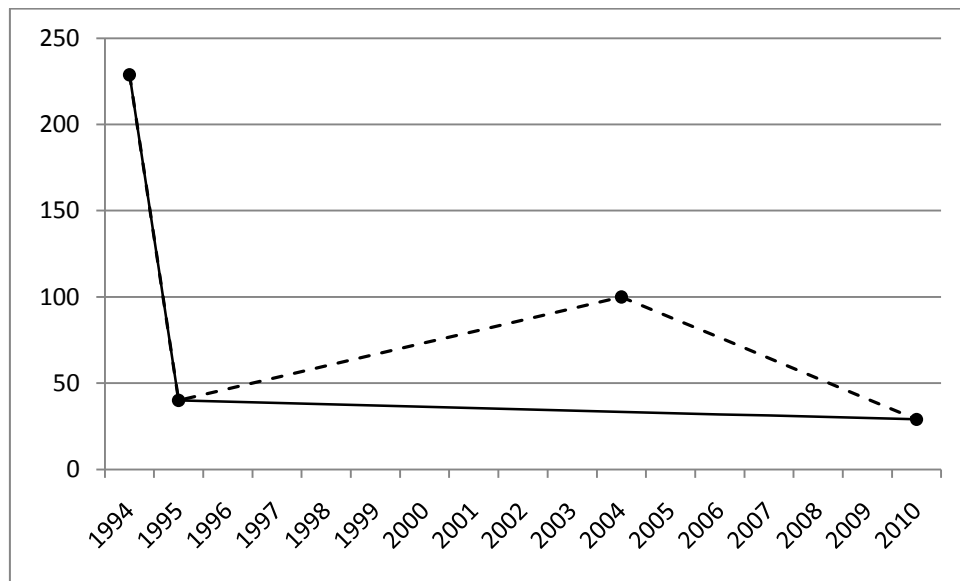


Figure 6-81: Used amount of SCCP in the textile industry

It is stated in the report [SCCP ECHA 2008] that the amount of SCCP used in 2004 can only be provided as < 100 tonnes due to confidential reasons. However it is also stated, that between 2001 and 2003 the use in textile and rubber have decreased by a factor of three and that until 2004 a further decrease, particular in the use for textile, paints and sealants and adhesives took place [SCCP ECHA 2008]. Together with the provided data with a SCCP use in textile of 29 tonne in 2010 [SCCP RPA 2010] the continuous drawn line in Figure 6-81 seems to be more reliable and the underlying figures are used for the calculation of the mass flows.

Due to the cost intensive treatment it is expected that only small amounts of textiles are imported which have been treated with chlorinated paraffins and if they have been treated they should be labelled as flame proved textiles [SCCP BfR 2002].

B) Input/Output

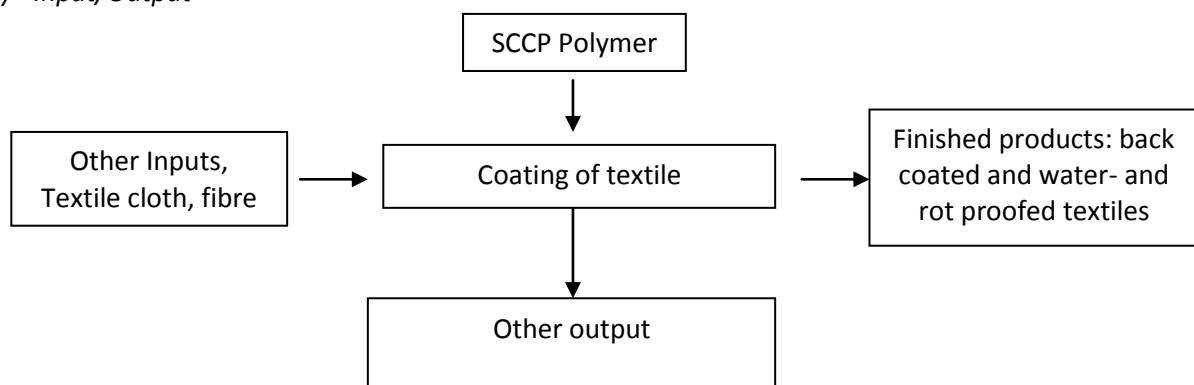


Figure 6-82: Material flow of SCCP in the plastic industry

Input: The textile clothes and fibres and the SCCP Polymers are combined. Relevant input is the SCCP in

the polymer latex.

Output:

Relevant outputs are the coated textiles. The resulting coated material has a SCCP concentration of about 4 to 50 % of SCCP [SCCP RPA 2010], [SCCP Seibersdorf 1998]. 20 % have been used for calculation. The use of SCCP in the Textile industry is not very well known and includes many different products. It is likely that SCCPs may be found in application in furniture upholstery, seating upholstery, transport applications, interior textiles such as blinds and curtains, industrial protective clothing, [SCCP ECHA 2008] [SCCP UBA 2006], carpets [SCCP TEGEWA], isolation textiles and decoration textiles in public areas [SCCB BfR 2002].

C) Generation factors

About 80 % of the SCCP (similar to chapter PCPL) will become part of the textile, whereas the rest will end up as waste or will be emitted [PCP IEP 2002]. The production losses are considered to be to equal shares emissions and waste.

Throughout lifetime the following emissions are used: volatile loss over life time 0.125 %; leaching loss over life time 0.25%; Erosion/particulate loss over life time and disposal each 2%. [SCCP ECHA 2008] The life time emissions sum up to about 4.4 %. Emissions from handling and the processing step have been stated by ECHA to be far below 1% [SCCP ECHA 2008], which is considered to be too little.

Considering an SCCP content of 20 % in the coated textile, 1 tonne SCCP is related to the production of 5 tonnes coated textiles.

As a consequence, from the use of 1 kg of SCCP 0.8 kg is part of the coated products and 0.765 kg of SCCPs is in products at the end of their lifetime, equal to 3.824 kg of textile waste.

D) Contamination Data

The estimated concentration of SCCP in treated textiles is assumed to be between 4 and 15 % [SCCP RPA 2010]. Regarding information of [TEGEWA 2010] a range of 10-20% was considered. According to [SCCP Seibersdorf 1998] it is assumed that the average weight of textile which is treated with flame retardants is about 400 g/m² and the typical concentration is between 3 to 30 % but can be as high as 50 %. Three examples with 10, 20 and 30 % of flame retardants are calculated in [Seibersdorf 1998].

From the above information a best estimate is made and a concentration of 20 % SCCP for textiles is used for the calculation of the mass flows.

E) Activity data

Within the last 16 years a stable use of ~30.5 t/y (29-32 t/y) from 1995 to 2010 has been considered. It is assumed that the average life time of the typical SCCP containing textiles is < 15 year, which is used for the calculation as a worst case scenario (1995-2010). In average about 24.4 t/y of SCCP become part of

the final product of which 23.3 t/y will be disposed of and 1.1 t/y is emitted during lifetime. An overview is given in Table 6-64.

Table 6-61: Overview of SCCP consumption and losses during production and life time within the last 17 years.

Year	SCCP consumption [t/y]	SCCP losses from processing [t/y]	Applied SCCP amount [t/y]	SCCP lifetime emissions [t/y]	SCCP in product [t/y]
1994	183	36.60	146	6.44	140
1995	32	6.40	26	1.13	24
1996	32	6.36	25	1.12	24
1997	32	6.32	25	1.11	24
1998	31	6.28	25	1.11	24
1999	31	6.24	25	1.10	24
2000	31	6.20	25	1.09	24
2001	31	6.16	25	1.08	24
2002	31	6.12	24	1.08	23
2003	30	6.08	24	1.07	23
2004	30	6.04	24	1.06	23
2005	30	6.00	24	1.06	23
2006	30	5.96	24	1.05	23
2007	30	5.92	24	1.04	23
2008	29	5.88	24	1.03	22
2009	29	5.84	23	1.03	22
2010	29	5.80	23	1.02	22
Average (1995-2010)	30.5	6.1	24.4	1.1	23.3

Considering the last 15 years as relevant the average amounts are multiplied with 15 resulting in 366 t of SCCP which have been sold in textiles of which 350 t will end up as waste (16 tonnes will be emitted). In general the mean value which is still in use is therefore 358 t of SCCP in textiles.

F) Waste Treatment

As mentioned above about 50 % of the losses during processing is expected to be emitted whereas the other 50 % will end up as waste. Waste from the production is considered to be treated as hazardous waste and will mainly be incinerated. Therefore, the average annual SCCP amount in waste as well as emissions from the processing step are considered to be 3.05 kg.

It is expected that polymer coated textiles are not recycled, as the fibres cannot be regained. Therefore the majority of SCCP containing textiles will end up as MSW or bulky waste. It is assumed that 67.1 % of the end of live textile is landfilled, 20.2 % is incinerated and 12.7 % is incinerated with energy recovery.

G) Substance flow

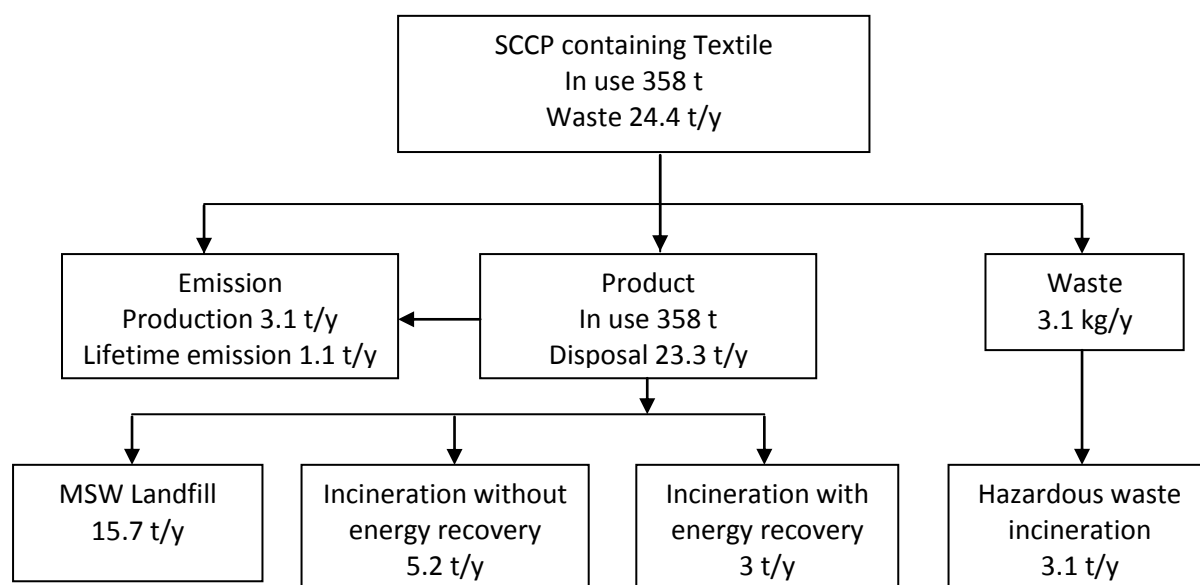


Figure 6-83: Material flow of SCCP in the plastic industry

From the 32-29 tonnes (mean value 30.5 t/y) of SCCP used per year only 26 and 23 t/y (mean value 24.4t/y) of SCCP becomes part of the textile product due to process losses of 6.4-5.8 t/y (mean value 6.1 t/y) (50% emission, 50 % waste). After additional emission during life time of 1.02-1.13 t/y (mean value 1.07 t/y) only 24.5 to 22.2 tonnes (mean value 23.3 t/y) of SCCP contained in textiles will become waste per year.

The emissions from production and life time losses are combined to about 4.12 t/y.

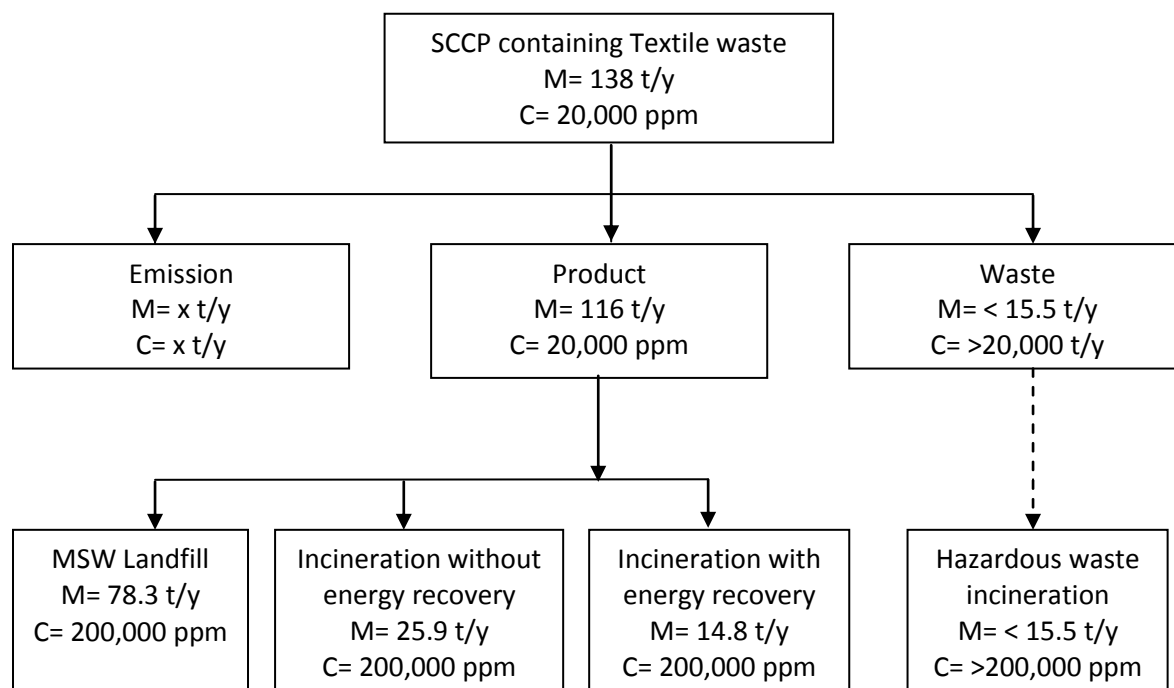
H) *Material flow*

Figure 6-84: Material flow of SCCP in the plastic industry

The SCCP concentration as well as the total amount of waste from the production of SCCP containing textiles is not known. However due to the final SCCP concentration of 20 % of the textile the collected waste as the remainder of containers should have at least this concentration. It is assumed that by far the majority of the collected waste from production will be incinerated as hazardous waste. The waste from end of life textiles are expected to become MSW and will be landfilled to 67.1 %, incinerated without energy recovery to 20.2% and incinerated with energy recovery to 12.7 %.

6.5.11 Sewage sludge

A) Background

As SCCP tends to adsorb on particles, a relevant share of the SCCP content of waste water accumulates in sewage sludge during waste water treatment. The sewage sludge mentioned in this report is urban sewage sludge as well as sewage sludge resulting from independent and other waste water treatment facilities.

It is expected that about 90% to 93% of the SCCP is adsorb on the sewage sludge and the rest stays in the water phase ([SCCP BUWAL 2003], [SCCP DETR 1997], [SCCP ERA 2000]).

Table 6-62: Levels of SCCP in sewage sludge including hot spot levels (DW: dry weight)

Country/ Region	Compartment	Levels of SCCP	References	Comments
Liverpool, UK (1980)	Sewage sludge	4-10 µg/g SCCP and MCCP	[SCCP ERA 2000]	(Campbell and McConnell, 1980)
Manchester, UK (1980)	Sewage sludge	< LOD (0.05 µg/g) SCCP and MCCP	[SCCP ERA 2000]	(Campbell and McConnell, 1980)
UK (1998)	Digested sewage sludge	2.9;12.1; 11.8; 17.1; 3.4; 1.8; 6.7; 93.1 µg/g=> 18.6 µg/g CP	[SCCP ERA 2008]	(CEFAS 1999)
United Kingdom (~2002)	Sewage sludge	6.9-200 µg/g dw (SCCP); 42 µg/g dw mean value (14 WWTP)	[SCCP POPRC 2010b] [SCCP POPRC 2007] [SCCP ERA 2008]	(Steven et.al. 2002)
Germany (1991-1993) near a metal plant	Sewage sludge	47 (1993)-65 (1991)µg/g dw 56 µg SCCP/g dw SCCP	[SCCP BUWAL 2003] [SCCP OSPAR 2001]	(Rieger and Balschmiter 1995) (EU Risk Assessment Report (1999))
Germany 1999-2000	Sewage sludge	51 WWTP: all < LOD (< 1 µg/g) CP	[SCCP UBA 2006]	(Hartmann et al. 2004)
Germany Hessen (2003)	Sewage Sludge	Average: 0.382 µg/g dw; maximum 0.537 µg/g dw SCCP	[SCCP UBA 2006]	(ISA RWTH 2005)
Swiss	Sewage Sludge	200,000 t sewage sludge including 2 t SCCP => 10 ppm; SCCP	[SCCP BUWAL 2003]	(BUWAL 1999), (Müller et.at. 1985)
Swiss	Sewage sludge	0.5-80 µg/g => 34.1 µg/g dw CP	[SCCP BUWAL 2003]	(Müller et.at. 1985)
Swiss	Sewage sludge	5.5 t in sewage sludge (CH) SCCP	[SCCP BUWAL 2003]	(ERA 2000)
Japan (1979)	Sewage sludge	24 out of 51 0.6- 10 µg/g CP	[SCCP ERA 2000]	Environment Agency Japan 1991

As can be seen in the table above the highest level of SCCP which has been measured has been in the United Kingdom with 200 µg/g DW (=200 ppm) with an average value of 42 ppm. This is similar to one German study, with values ranging from 47 to 65 ppm. It is reported that in the UK the highest

concentrations of SCCP are in sludge from industrial catchments [SCCP POPRC 6/11]. Also in case of Germany it is reported that this high values have been measured near a metal working plant [SCCP OSPAR 2001]. There have been measurements from several WWTP in the UK (Manchester) as well as in Germany (51 WWTP) when the LOD (0.05 and 1 ppm respectively) of SCCP was not reached in the sewage sludge. Also due to the strong reduction of SCCP in industry the amount of high contaminated SCCP in sewage sludge is much lower than suggested by the measurements which are mostly over 10 years old.

The levels found in digested sewage sludge prior to application onto soil were in the range 1.8-93 µg/g dry weight and the SCCP levels in soil where the sludge was applied were generally not detected

B) Input/Output

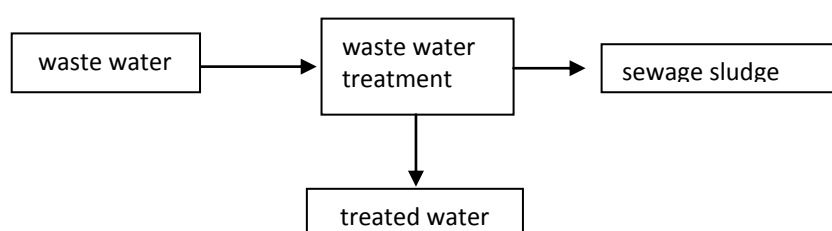


Figure 6-85: SCCP relevant Input/Output overview of sewage sludge

Input: waste water containing low contents of SCCPs

Output: sewage sludge containing elevated contents of SCCPs and treated water containing lowered contents of SCCPs

C) Generation factors

No generation factor is used because the amount of sewage sludge produced from the wastewater depends on the composition of the wastewater and the treatment processes. Following the low solubility of SCCP in water the resulting treated water is not taken into further account.

D) Contamination Data

Sewage sludge: As contamination data a range of 0-200 ppm SCCP in sewage sludge is given. It seems that especially in sewage sludge in industrial catchments SCCP above 10 ppm can be found, whereas other sewage sludge are often below 1ppm. Similar to the PFOS containing sewage sludge it is considered that about 95 % of the sewage sludge is only contaminated little, whereas 5 % are higher contaminated.

Reported SCCP contents in sewage sludge from relevant industrial catchments in the UK and Germany range from 42 to 65 ppm. As a best estimate a content of 50 ppm is used for high contaminated sludges. As a worst case estimate, a content of 1 ppm can be used for low contaminated sludges.

The Environmental Protection Agency (EPA) in Ireland has recently undertaken sampling of wastes from sewage treatment to determine concentration levels of the new and candidate POPs in those wastes.

According to [EPA IRE 2011a] no SCCPs (C10, C11, C12, C13) were measured above the limit of detection (<100 µg/kg) in any of the samples. For SCCP analysis the extraction method was solvent extraction and analysis standard is apparently comparable to ISO22032:2006.

E) Activity data

The European annual production of sewage sludge is about 11,578 * 10⁶ t/y. The data from Table 6-63 have been taken from EUROSTAT. For Portugal no sewage sludge amount was available and has therefore been taken from the report [PFOS_UN-Habitat_2008]. For the distribution of the use of sewage sludge in Portugal the average usages from Europe have been adopted. From the total amount of 11,578 million kg/y only the data of the usage of 10,653 million kg/y are known. The missing 8 % have therefore been given to “other uses”. The received total amount of sewage sludge from EUROSTAT fit very well with the amount given by [PFOS_UN-Habitat_2008] which is 11,370 million kg/y.

Table 6-63: Annual sewage sludge production in EU 27 [EUROSTAT 2010]

Use of sewage sludge of Europe						
Member state	Total amount [10 ⁶ t/y]	Landfill [10 ⁶ t/y]	Incineration [10 ⁶ t/y]	Agriculture and compost [10 ⁶ t/y]	Others uses [10 ⁶ t/y]	Total treatment [10 ⁶ t/y]
AT	255	25	98	113	18	254
BE	112	9	36	21	45	111
BG	40	21	0	6	0	27
CY	9	1	0	5	0	6
CZ	172	9	0	135	28	172
DE	2,049	5	965	1,079	67	2,116
DK	0	0	0	0	0	0
EE	31	5	0	5	19	29
ES	1,065	168	41	687	169	1,065
FI	160	10	0	147	3	160
FR	1,060	222	178	632	27	1,059
GR	126	123	0	0	3	126
HU	286	137	1	131	16	285
IE	88	5	0	63	22	90
IT	1,056	440	31	466	116	1,053
LT	76	9	0	32	0	41
LU	12	4	0	8	0	12
LV	23	0	0	10	9	19
MT	0	0	0	0	0	0
NL	541	36	287	83	116	522
PL	1,088	298	34	163	595	1,090
PT	237	40	45	118	34	237
RO	758	100	4	20	37	161
SE	210	24	0	104	22	150
SI	21	9	5	4	4	22
SK	332	9	0	39	9	57
UK	1,771	96	282	1,235	158	1,771
TOTAL	11,578	1,805	2,007	5,306	1,517	10,635
Share	100%	16%	17%	46%	13%	92%
Unknown						8%
TOTAL share	100%	16%	17%	46%	21%	100%
Total amount	11,578	1,805	2,007	5,306	2,460	

F) Waste Treatment

According to EUROSTAT data, at European level sewage sludge is treated with the following shares:

- 16% are landfilled
- 17% are incinerated
- 46% are used in agriculture and compost
- 21 % are applied for others uses

It is considered that SCCP is decomposed at about 200 °C [SCCP BUWAL 2003] and that at an incineration temperature of at least 850 °C a complete destruction of SCCPs is expected.

G) Substance flow

Figure 6-86 shows the substance flow of PFOS in sewage sludge. Currently low and high contaminated sewage sludges are used in the same manner.

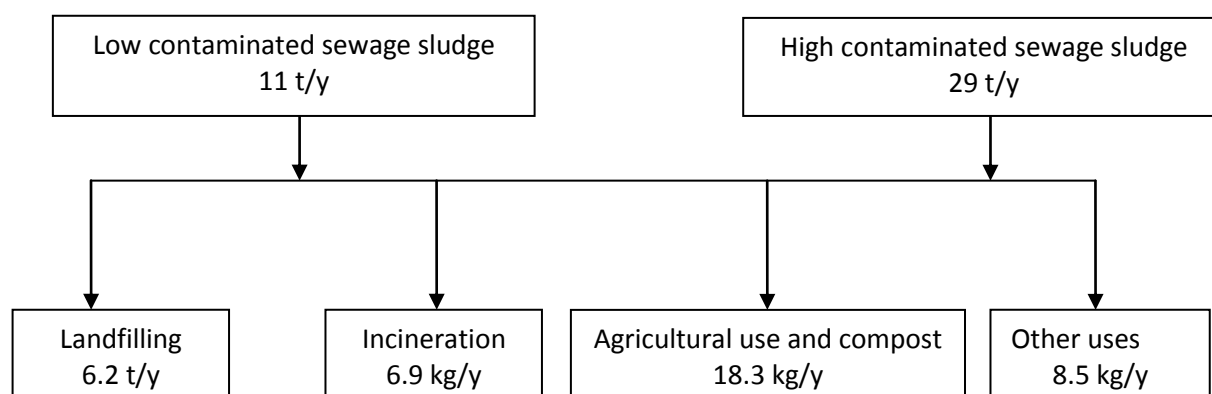


Figure 6-86: Substance flow of SCCP in sewage sludge

H) Material flow

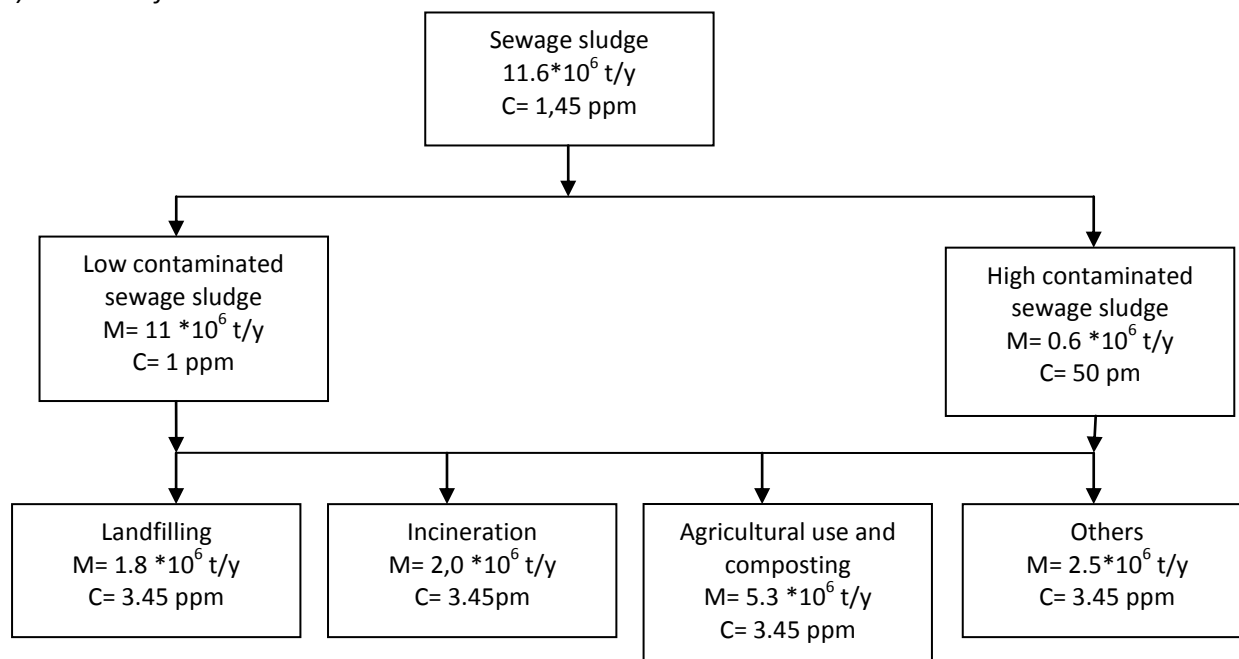


Figure 6-87: Material flow of SCCP in sewage sludge

Figure 6-87 gives a detailed overview of the mass flow of sewage sludge with the corresponding concentrations of SCCP.

6.5.12 Leather

A) Background

As a consequence of the restriction by Directive 2002/45/EC the use of SCCP in the EU has ceased in 2004. Therefore in 2004 a small use of SCCP in the leather industry was still possible. However leather products containing SCCP are still in use and are a source of current and future waste. One possible use has been shoes [SCCP VDL 2010] or upholstery [SCCP EC 2009].

SCCP is used in fat liquors [POPRC 6/INF/15]. Fat liquoring is generally carried out after tanning and dyeing. Agents may be applied in order to achieve particular properties and wear qualities, e.g. water-repellent or water proof properties, oil repelling or anti-electrostatic properties, reduced permeability to gas, flame-retardant and abrasion reduction. The surfactants used in wet processes could increase the hydrophilic properties of leather. To correct this, hydrophobic chemicals such as silicones or organic chlorofluoro-polymers dissolved in organic solvents can be applied. Therefore fat liquors are used to lubricate the leather to achieve product specific characteristics and to re-establish the fat content lost in the previous procedures [SCCP EC 2009].

The consumption of SCCP in the leather industry in the EU 27 is represented in Table 6-64 and Figure 6-88.

Until 1998 the data were only available for the EU 15. To estimate the use of SCCP in the leather industry for the EU 27 the values of Table 6-64 have been extrapolated on the basis of a per capita approach for the EU 27 (25 %; 400 million vs. 500 million). Figure 6-88 represents the extrapolated levels (488 t in 1994 and 56 t in 1998), whereas the dots represent the amounts given represented in Table 6-67 [SCCP ECHA 2008].

Table 6-64: Estimated tonnage of SCCP used in the EU (t/y) [SCCP ECHA 2008]

Use area	1994 (EU 15)	1995 (EU 15)	1998 (EU 15)	2001 (EU 15)	2003 (EU 15)	2004 (EU 25)	2007 (EU 27)
Leather	390	n.a.	45				X

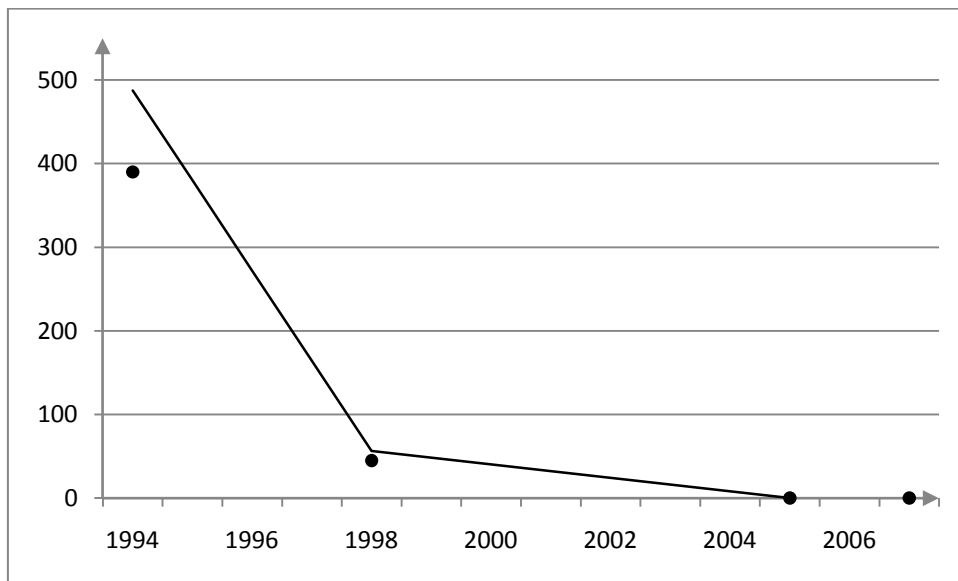


Figure 6-88: Use of SCCP in the EU leather industry.

B) Input/Output

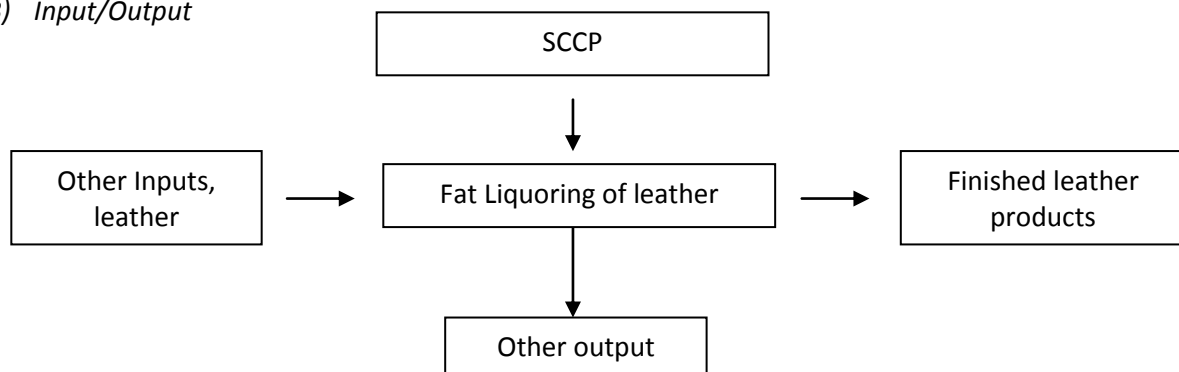


Figure 6-89: Input/Output flow of SCCP in the leather industry.

Input:

Relevant input is the SCCP contained in fat liquors. For the fat liquoring of leather a solution with a SCCP content of about 20 % is used.

Output:

Leathers treated with fat liquors including SCCP are used for upholstery as well as for shoe soles [SCCP VDL 2010] [SCCP EC 2009]. Losses due to emissions and waste generation during the tanning process as well as processing losses due to cut off are only historically as SCCP is not used in this industry anymore.

C) Generation factors

The generation factor is between 95 and 99 %. The majority of the SCCP from the fat liquoring solution becomes part of the final product. For the calculation a mean value of 97 % is used [SCCP BUWAL 2003]. 1 kg of SCCP is related to the generation of 0.97 kg of SCCP in finished leather products.

1 t of SCCP is contributes to the generation of 48.5 tonnes $(= (1t \cdot 0,97\%) / 20\%)$ of finished leather products.

Due to cut off in shoe and upholstery manufacturing (5%) a quantity of about 46 tonnes $(48.5 t - 5 \%)$ SCCPs is incorporated in the finished leather products.

D) Contamination Data

Leather products contain about 3-15 % (average about 10 %) of fat liquoring agents [SCCP EC 2000] with shoes requiring less fat liquor than upholstery [EC 2009]. Together with a SCCP concentration of 20 % in the fat liquoring mixture a mean value of 2 % SCCP for leather, which has been treated with a SCCP containing fat liquor, is considered. These 2 % are equal to 20,000 ppm.

If all leathers are collected together (see section PFOS leather for more details) the average SCCP content reduces to 13.8 ppm for all leathers which have been produced between 1998 and 2004 or to 6.9 ppm for all leathers produced after 1998.

E) Activity data

Footwear remains overall the most important outlet for EU tanners production with a share of 50%. The clothing industry takes approximately 20% of all finished leather produced in the EU. Leather for furniture and automotive upholstery represents some 17 % of EU tanners output and the leather goods sector 13% [EC 2009].

It is considered that only shoes and [VDL 2010] upholsteries have been treated with SCCP containing fat liquors. Therefore the above mentioned ratio of 50:17 of tanned leather for shoes and upholstery is used to calculate the shares of the different products regarding the use of SCCP containing fat liquors, which corresponds to about 75 % for shoes and 25 % for upholstery. As it is also stated that shoes require less fat liquors than upholstery the shares for use of SCCP containing fat liquors are shifted to 2/3 for shoes and 1/3 for upholstery to account for this differentiation.

The following table provides an overview regarding the used amount of SCCP for the different products regarding their different life time. It is considered that the average life time of upholstery is about 10 years for household upholstery and 12 years for automobile [GEO 2007]. For the calculations an average life time of 12 years for upholstery and 6 years for shoes has been chosen.

For the production step a cut off rate of 5 % has been assumed. This type of waste does not occur anymore as the use of SCCP for leather is banned.

Table 6-65: Overview of the use of SCCP in the leather industry since 1994

Year	SCCP consumption [t/y]	Cut off [t/y]	SCCP use for shoes [t/y]	SCCP use for upholstery [t/y]
1994	488	24	299	150
1995	380	18	233	117
1996	272	13	167	84

1997	164	8	101	50
1998	56	3	35	17
1999	48	2	30	15
2000	40	2	25	12
2001	32	2	20	10
2002	24	1	15	7
2003	16	1	10	5
2004	8	0	5	2
2005	0	0	0	0
2006	0	0	0	0
2007	0	0	0	0
2008	0	0	0	0
2009	0	0	0	0
2010	0	0	0	0
Total	1528	74	0	52

Due to the short life time of shoes it is expected, that the amount of shoes containing SCCP should be negligible and is not further investigated.

Regarding the upholstery products, the 52 tonnes represent the amount of SCCP which are still used in different upholstery products of which 17.3 tonnes (Production in 1998) is disposed of in 2010. With a concentration of 2 % this represents about 850 tonnes of leather. Approximately 20 % of the upholstery consists of leather. Therefore the total amount of bulky waste containing SCCP in 2010 is about 4,320 tonnes. It is assumed that this type of SCCP containing waste will be negligible after 2016. The SCCP concentration of the leather is 20,000 ppm and from the SCCP containing upholstery 4,000 ppm.

The total annual production of leather is summarized in section 6.3.11 concerning the PFOS use in the leather industry. According to this table the average European annual production of leather is about 264 million m². With an average leather weight of about 6 kg/m² the total weight is of about 1.6 million t of leather per year of which about 17 % are used for upholstery. This represents about 475,615 t/y of leather used for upholstery. The leather share of the upholstery is about 20 % resulting in a total leather upholstery weight of 2.4 million t/y, which will become bulky waste after lifetime.

If these leather upholstery bulky waste is mixed up with SCCP containing leather upholstery waste with a total SCCP amount of 17 t/y then the average SCCP contamination is about 7.1 ppm.

F) Waste Treatment

It is expected that leather products are not recycled. Therefore the majority of SCCP containing leather will end up as MSW or bulky waste and is therefore treated similar to MSW (no composting, no recycling). It is assumed that 67.1 % of the end of live textile is landfilled, 20.2 % is incinerated and 12.7 % is incinerated with energy recovery. Waste and emission which occur during production are not relevant, because in the EU feather is not treated with SCCP anymore.

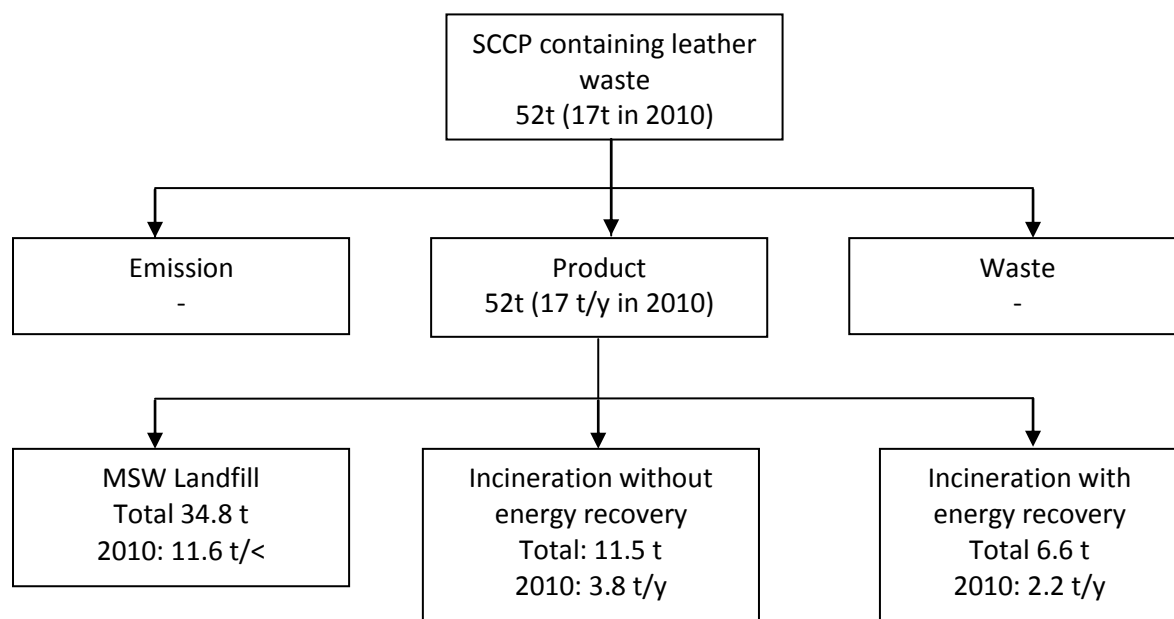
G) *Substance flow*

Figure 6-90: Substance flow of SCCP used in the leather industry

H) Material flow

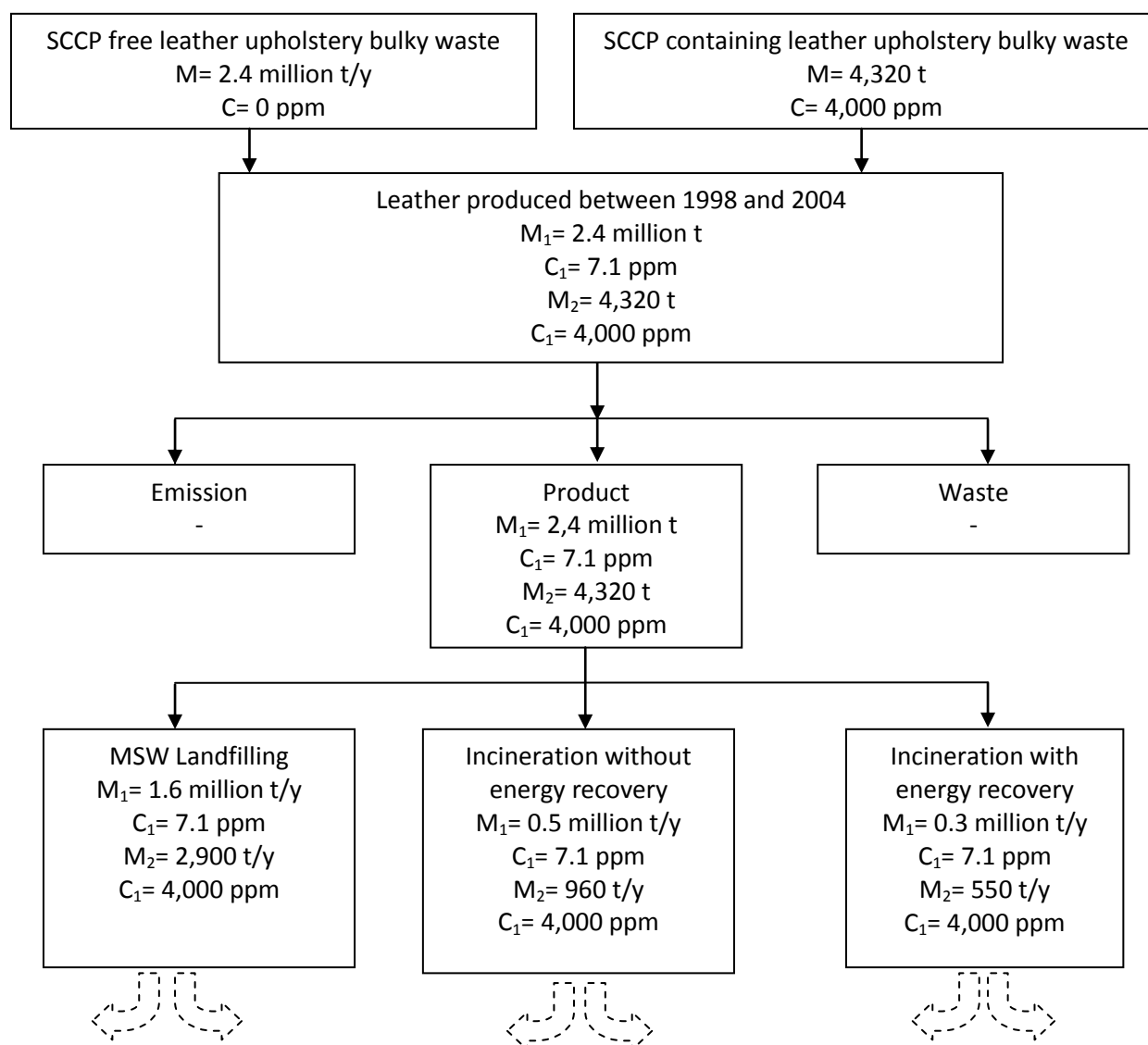


Figure 6-91: Waste flow of SCCP containing products from the leather industry

6.5.13 Medium-chain chlorinated paraffins (MCCPs)

SCCPs are present as impurities in medium-chain chlorinated paraffins. The actual levels of SCCPs in the MCCPs are as low as <1%. The annual MCCP production in Europe is in the range of 45,000 to 160,000. There are 4 companies located in Italy, UK, Slovakia and Romania thought to be main producers of SCCPs in the EU in recent years [SCCP ERA 2008]. According to the EU risk assessment report on MCCPs, medium-chain chlorinated paraffins may contain <1% of chlorinated paraffins with chain lengths other than C₁₄₋₁₇ [JRC 2005].

MCCP is used in many cases as an alternative for SCCP but the concentration of MCCP has been changed in comparison with SCCP. However it can be stated that the concentration of SCCP in products in which cases MCCP is used as an alternative is at least of a factor 100 less compared to the concentration before the exchange.

In the following table the amounts of SCCP have been used in 1994 the data from Western Europe are from a technical guidance document [SCCP ERA 2000]. It is assumed that the reduction of SCCP in the industries have been exchanged 1:1 with MCCP having a SCCP concentration of 1 %.

Table 6-66: Possible quantities of SCCPs as impurities in MCCPs in different industry sectors if former SCCP uses have been replaced by the use of MCCPs containing 1% SCCPs

Industry	Western Europe [t/y] (former SCCP now MCCP)	EU 27 [t/y] (former SCCP now MCCP)	SCCP 2010 [t/y]	% of total use
Metal working	9,380	11,256,00	112,56	71%
Rubber	1,310	1,572,00	15,72	10%
Paints	1,150	1,380,00	13,8	9%
Sealants	695	834,00	8,34	5%
Leather	390	468,00	4,68	3%
Textile Others	183	219,60	2,196	1%
Others	100	120,00	1,2	1%
Total	13,208	15,849,60	158	100%

Compared with the SCCP uses in 2010 the following additional percentage has to be considered in the sections above:

Table 6-67: Possible contributions to the use of SCCPs by impurities in MCCPs in different industry sectors if former SCCP uses have been replaced by the use of MCCPs containing 1% SCCPs

	Use of SCCP in 2010	SCCP as impurity in 2010	Percentage of impurity
Metal working	--	113	100%
Rubber	162	16	9%
Paints	101	14	12%
Sealants	237	8	3%
Leather	--	5	100%
Textile	29	2	7%
Others	--	1	100%
Total	529	158	23%

It can be seen that for the investigated current uses the additional amount of SCCP due to impurities in MCCP is up to 12 %.

A more detailed analysis of the use of MCCP and the implemented effect due to SCCP is not further investigated. Due to the low concentration of SCCP in the products it is assumed that the contents will be below the POP content levels to be set.

6.5.14 References SCCPs

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6.6 HBCD flows

6.6.1 Use and Production

Production

At present, HBCD is only produced at one site in the former EU 15, located in the Netherlands with an annual production of around 6,000 t in 2005 (100% HBCD in form of powder or granulate). Two other production sites, whereof one was situated in the UK, were closed in 1997 and 2003. Besides, HBCD is imported to and also exported from the EU as a chemical itself, in formulations and in flame retarded articles. Countries outside the EU known to produce HBCD are the USA, Japan and China ([ECB 2008a] [UNEP/POPS/POPRC.6/10]). Production of HBCD between 2003 and 2007 was fairly static at 6000 tonnes [IOM 2008]. According to industry information, current production in the Netherlands is still around 6,000 tonnes per year [PS foam 2011].

Use

HBCD has been on the world market since the 1960s. The use of HBCD in flame retarded insulation boards started in the 1980s [UNEP/POPS/POPRC.6/10]. HBCD is solely used as an additive flame retardant [IOM 2008]. At present, the main downstream uses of HBCD are in the polymer and textile industries. HBCD can be used on its own or in combination with other flame retardants e.g. antimony trioxide and decabromodiphenyl ether. HBCD 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, HBCD is typically used alone and not with other additives [PS foam 2011].

According to industry information the main use of HBCD is for polystyrene (PS) and amounts to 90%. The predominant use of PS is in rigid insulation panels/boards for building construction (EPS and XPS). About 2% of the total use of HBCD is in HIPS. Examples of EPS end-products containing HBCD are insulation boards for construction in general, for indoor and outdoor building constructions, against frost heaves of road and railway embankments and for transport vehicles as well as packaging material in general excluding food packaging (minor use). XPS end products are used for the same applications. HIPS end products are applied in electrical/electronic parts, e.g. electric housings for VCR (video cassette recorder), electrical and electronic equipment (distribution boxes for electrical lines) and video cassette housings. Further, HBCD can be used as textile coating agent in polymer dispersion on cotton or cotton/synthetic blends for upholstery fabrics, e.g. residential and commercial furniture and transportation seating, bed mattress ticking, draperies and wall coverings, interior textiles, e.g. roller blinds, and automobile interior textiles. Other minor uses of HBCD in polypropylene (PP), in styrene-acrylonitrile copolymer resins (SAN resins), in polyvinylchloride (PVC) as well as in unsaturated polyesters, adhesives and coatings have been reported, whereas it is unclear if these uses are relevant within the EU [ECB 2008]. Granulated EPS waste is also used to improve the texture of agricultural and horticultural soil ([UNEP/POPS/POPRC.6/10], [Vogdt 2009]).

Specific product categories for EPS, XPS, HIPS and polymer dispersions for textile coating are listed in IOM 2008. EPS is a rigid cellular plastic, which is used in a multitude of applications such as packaging.

EPS is mainly used for the following purposes [IOM 2008]:

- insulation panels/boards in the construction sector
- automobile cushions for children*
- rigid packaging material for fragile equipment (minor use)*
- packaging material such as “chips” and shaped EPS-boards (minor use)*
- in props for theatre and film, and in exhibitions (minor use)

* Uses marked with an asterisk are not considered relevant by the EU PS foam industry. However, if disposal is not well controlled these uses have the potential to lead to more HBCD release than through insulation boards. For these applications imported EPS might also play a role in imports from countries where HBCD is not restricted [PS foam 2011].

Extruded polystyrene (XPS) is a plastic foam based on polystyrene that is formed by extrusion. HBCD-containing XPS is mainly used for the following purposes [IOM 2008]:

- cold bridge insulation
- sandwich panels and laminates
- cavity insulation
- floors
- basement walls and foundations
- inverted roofs
- ceilings

Most flame-retarded High Impact Polystyrene (HIPS) products are used in electrical and electronic appliances. For example [IOM 2008]:

- audio visual equipment cabinets (video and stereo equipment)
- distribution boxes for electrical lines in the construction sector
- refrigerator lining

Flame retardant systems are used in textile applications to comply with flame retardant standards. HBCD is formulated to polymer-based dispersions (e.g. acrylic or latex) of variable viscosity in the polymer industry. The dispersions are then processed in the textile finishing industry. The textiles with the back-coating containing HBCD are mainly used for [IOM 2008]:

- flat and pile upholstered furniture (residential and commercial furniture)
- upholstery seating in transportation
- draperies, and wall coverings
- bed mattress ticking

- interior textiles e.g. roller blinds
- automobile interior textiles and car cushions.

The HBCD particles used for textile back-coating need to be very small and micronised (see chapter 2.1.2). Flame retarded formulations containing HBCD are prepared as water-based dispersions that can contain a binder system and HBCD as well as up to 20 other ingredients.

To conclude, the main downstream uses of HBCD are in polymer and textile industries. HBCD is used in four principal product types, which are XPS, EPS, HIPS and polymer dispersion for textiles. The main use (90 %) of HBCD is in EPS and XPS in building and construction. About 2 % of the total use of HBCD is in HIPS, in electronic products and articles. The remaining use of approximately 6% is in EPS and XPS in packaging material and insulation boards of transport vehicles. About 2% is currently used in textile coating for upholstery fabrics, bed mattress ticking, draperies and wall coverings and interior textiles (see [ECB 2008] and [CPAN 2010]).

6.6.2 Chemical Characteristics of HBCD

Commercially available HBCD is a white solid substance. The structural formula of HBCD is a cyclic ring structure with six Br-atoms attached. The molecular formula of the compound is $C_{12}H_{18}Br_6$ and its molecular weight is 641 g/mol.

Technical HBCD is manufactured in two forms, high-melting (HM) and low-melting (LM). Technical HBCD is often characterised as a mixture of three diastereomers, that are compounds that are identical except for the spatial disposition of the atoms, α - β - γ - HBCD, and the final distribution of these diastereomers in technical HBCD varies with the range of about 70-95 % γ -HBCD and 5-30 α - and β -HBCD [Swerea 2010]. The LM HBCD consists of 70-80 % γ -isomer and 20-30 of α - and β -isomers. The HM HBCD consists of 90 % or more of the γ -isomer.

Two different CAS Numbers can describe HBCD, in particular 25637-99-4 (Hexabromocyclododecane) and 3194-55-6 (1,2,5,6,9,10-Hexabromocyclododecane). However, there are no differences in molecular structure or properties between the chemicals represented by these CAS numbers. In commercial HBCD only three of several stereoisomers are commonly found. Depending on the manufacturer and the production method used, technical HBCD consists of 70-95 % γ -HBCD and 3-30 % of α - and β -HBCD. Each of these stereoisomers has its own specific CAS number, i.e. α -HBCD, CAS No: 134237-50-6; β -HBCD, CAS No: 134237-51-7; γ -HBCD, CAS No: 134237-52-8. Two other stereoisomers, δ -HBCD and ϵ -HBCD have also been found in commercial HBCD in concentrations of 0.5 % and 0.3 %, respectively.

Specific information pertaining to the chemical identity of HBCD is listed in Table 2, 3, and 4 of [UNEP/POPS/POPRC.6/10].

Commercial HBCD (i.e. CAS Nos 25637-99-4 and 3194-55-6) is proposed to be listed under the Stockholm Convention (proposal see [UNEP/POPS/POPRC.5/4]).

6.6.3 *Environmental Background Levels*

HBCD is widespread in the global environment, with high levels in the top predators. According to literature reviews the HBCD levels in the environment are generally increasing in all matrices in the environment, and seem to correlate with the increasing use of HBCD. The reviews cover over 100 published scientific studies (up to 2007) performed in North America, Europe, the Arctic, Asia and the South Pacific region. The reviews cover a variety of environmental compartments (atmosphere, indoor and outdoor air, sewage sludges, soils and sediments) and a variety of biological samples and food chains. [UNEP/POPS/POPRC.6/10]

Recent monitoring results show decreasing trends for polar bears for the period from 2005 to 2008 [MCKinney et al 2011] and for herring gull eggs from the Germany coast for the period from 2001 to 2008 [Esslinger et al. 2010].

Several sediment core analyses performed in Asia and Europe show higher levels of HBCD in the top layers and lower concentrations in the deeper layers. These findings correlate with the trend in use of HBCD. In sediment cores HBCDs first appeared in the mid-1970s and concentrations observed in the cores have increased since then. [UNEP/POPS/POPRC.6/10]

The concentrations in the European environment are often higher than those measured in biota in North America and the Asia-Pacific region. The levels in the Asia-Pacific region and North America are found to be in the lower range of the levels detected in sea mammals in Europe. The results likely reflect the substantially higher market demand for HBCD in Europe relative to other regions of the world. [UNEP/POPS/POPRC.6/10]

The supporting document for the draft risk profile on HBCD gives an overview of HBCD concentrations measured in the environment (see [UNEP/POPS/POPRC.6/INF/25]). Table 6-68 shows the concentrations measured in sediment and soil.

Table 6-68: HBCD concentrations measured in the ambient environment: compartments sediment and soil (source: [UNEP/POPS/POPRC.6/INF/25]). Unless otherwise stated concentrations are indicated in ng/g dry weight (= ppb dry weight).

SEDIMENT					
The Arctic					
Norwegian Arctic; 2001	sediment	α -HBCD 0.43 β -HBCD nd γ -HBCD 3.88	4	0.06	Evenset et al. 2007
North America					
Detroit River; 2001 upstream stations upper reaches near Belle Isle near mouth of Rouge River near head of Trenton Channel in Trenton Channel eastern side near Fighting Island western side near outflow to Lake Erie same latitude as above station but on Canadian side of river	suspended sediments	annual mean: 0.106, 0.019 0.910 ~ 0.810 ³ ~ 0.675 ³ 0.393 0.012 1.14 0.066 monthly range: α -HBCD nd – 1.85 β -HBCD nd – 0.246 γ -HBCD nd – 2.27 Σ HBCD nd – 3.65	63	0.025	Marvin et al. 2004, 2006
Lake Winnipeg; 2003	sediment	mean (range): α -HBCD nd β -HBCD nd γ -HBCD 0.05 (nd – 0.10)	4	0.04 – 0.08	Law et al. 2006a
Europe					
United Kingdom; no year upstream of Aycliffe STW	river sediments	1131	1	50	Deuchar 2002
England; 2000 – 2002	estuarine and river sediments	mean (range): 199 (nd – 1680)	22	2.4	Morris et al. 2004
Ireland; 2000 – 2002	estuarine sediments	mean (range): 3.3 (nd – 12)	8	1.7	Morris et al. 2004
Belgium; 2001	estuarine sediments	mean (range): 60 (nd – 950)	20	0.2	Morris et al. 2004
The Netherlands; no year	sediments (< 63 μ m)	25.4 - 151	ns	ns	Bouma et al. 2000
The Netherlands; 2000 Western Scheldt Rivers	estuarine sediments river sediments	mean (range): 10 (nd – 99) 3.2 (nd – 9.9)	19 9	0.6 0.8	Morris et al. 2004
The Netherlands; 2001	estuarine sediments	14 – 71	ns	ns	Verslycke et al. 2005
The Dutch North Sea; 2000	surface sediments (< 63 μ m)	0.76 – 6.9	in 9 of 10	0.20	Klamer et al. 2005
Switzerland; no year	lake sediment	< 0.1 – 0.7 ³	1	ns	Kohler et al. 2007
Switzerland; 2003 1974 1982 1989	lake sediment	0.51 0.40 1.3	1	ns	Kohler et al. 2008

1995 2001		1.8 2.5			
Sweden; 1995	river sediments	nd – 7600 ng/g ignition loss (nd – 1600 ng/g dw)	18	ns	Sellström et al. 1999 (de Wit 1999)
Sweden; textile plant; 2000	river sediments: upstream downstream	nd, 0.2 1, 4.6, 7.8, 25	2 4	0.1 ns	Remberger et al. 2004
Stockholm; 1996 – 1997 1999	sediments sediment traps	0.2 – 1.5 nd – 2.1	6 3	ns 1	
S.E. Norway; no year	lake and river sediments river and estuarine sediments	α -HBCD β -HBCD γ - HBCD nd – 0.1 nd – 7.91 nd – 0.05 nd – 10.15 nd – 0.65 nd – 3.34	15 11	0.2 – 0.4 0.08 – 0.4	Schlabach et al. 2004a,b
2003					
Spain; 2002 industrialized urban area with chemical industry	river sediments: 20 km upstream 12 km upstream just downstream 30 km downstream	nd nd 513.6 89.7	4	0.006 – 0.050	Eljarrat et al 2004
Spain; no year Sampling sites in vicinity of those used in Eljarrat et al. 2004	river sediments: 20 km upstream at urban site 27 km downstream 67 km downstream	nd 2658 76.8 48.4	4	0.0003 – 0.0015	Guerra et al. 2008
Asia					
Japan; 1987	marine and river sediments	20 – 90	in 3 of 69	ns	Watanabe and Tatsukawa 1990
Japan; 2002	estuarine sediments	0.056 – 2.3	in 9 of 9	0.01	Minh et al. 2007
SOIL					
Europe					
United Kingdom; 1999 formulator/compounder plant	Soil	mean (range): 62,800 (18,700 – 89,600)	4	ns	Dames and Moore 2000a
Sweden; 2000 XPS production facility	Soil	140, 1000, 1300	3	ns	Remberger et al. 2004
Asia					
Guangzhou, China; 2006	Soil	1.7 – 5.6	3	0.0003 – 0.0005	Yu et al. 2008

According to Covaci et al. (2006) there is a general trend to higher environmental HBCD concentrations (air, sediment, and fish) near point sources (plants producing or processing HBCDs) and in urban areas, than in locations with no obvious sources of HBCDs. Concentrations of HBCDs are often elevated by at least one order of magnitude in the vicinity of plants either producing or using HBCDs. Several hot spots have been identified in Europe: the rivers Viskan (Sweden), Tees and Skerne (U.K.), Cinca (Spain), and the Western Scheldt estuary (Netherlands) (Covaci et al. 2006). All of these sites were related to present or former production facilities for HBCDs or HBCD retarded materials. Higher HBCD concentrations are also frequently found near urban centres and industrial sites (Janak et al. 2005; Remberger et al. 2005; Petersen et al. 2005; Minh et al. 2007; Morris et al. 2004; Sellström et al. 1998; Eljarrat et al. 2009; Hoh and Hites 2005), (cited from [UNEP/POPS/POPRC.6/10]). Detected levels in corresponding sediments in Europe range up to 1680 ppb dry weight (see Table 6-68). According to the knowledge of PS foam industry there are no elevated levels of HBCD around EPS conversion plants [PS foam 2011].

In particular, soil samples collected near HBCD-processing factories are found to have high levels of HBCD. Remberger et al. (2004) and Petersen et al. (2005) measured HBCDs, ranging between 111 and 23,200 ng/g DW, in soil samples collected outside an XPS producing plant (cited from [UNEP/POPS/POPRC.6/10]). Detected levels in corresponding soils in Europe range up to 1300 ppb dry weight (soil from XPS production facility) and 89600 ppb (soil from formulator/compounder plant) (see Table 6-68).

As shown in Table 6-68 in the environment, the detected European background contamination levels in sediment are usually below 10 ppb (see results from Klamer et al. 2005, Kohler et al. 2007, Kohler et al. 2008, Remberger et al. 2004, Schlabach et al. 2004 a,b) but may range up to 71 ppb in areas with known high background contamination (see results from Verslycke et al. 2005 from analyses of sediment from the Scheldt estuary, one of the largest and most polluted estuaries in Western Europe). Background levels for soil are not available for Europe. Reported concentrations from Asia ranged from 1.7 to 5.6 ppb (see results from Yu et al. 2008).

6.6.4 Legal Situation

International level

Commercial HBCD is proposed to be listed under the Stockholm Convention (proposal see [UNEP/POPS/POPRC.5/4]). At the sixth meeting of the POPRC In October 2010 the risk profile for hexabromocyclododecane was adopted and an intersessional working group was established to prepare a draft risk management evaluation on HBCD.

Hexabromocyclododecane (HBCD) was nominated to be included in the UNECE POP protocol in 2008. The Task Force on POPs finished the Track A review at its 7th meeting, evaluating the available information on risk for the environment and human health the use of HBCD is posing. At the 26th session of the Executive Body the Parties to the POP Protocol decided that HBCD is to be considered as a POP as defined under the Protocol. At the 8th meeting of the Task Force on POPs in May 2010, the exploration of management options for HBCD [CPAN 2010] was presented and discussed.

European Level

Waste management

According to Annex II of the WEEE directive 2002/96/EC “Selective treatment for materials and components of waste electrical and electronic equipment in accordance with Article 6(1)” as a minimum [...] [e.g. plastic containing brominated flame retardants] have to be removed from any separately collected WEEE. This means that plastics containing brominated flame retardants have to be removed from any separately collected WEEE. HBCD is a brominated flame retardant. Consequently this obligation includes plastics containing HBCD.

Restrictions on marketing and use

HBCD was considered as a candidate for inclusion into the planned Annex III of the currently recasted RoHS Directive 2002/95/EC. In the current version of the Directive which was approved by the European Parliament on 24.11.2010 Annex III is deleted. However, the substances which were listed in the planned Annex III were moved to recital 10: “The annexes to this Directive should be reviewed periodically to take into account, inter alia, Annexes XIV and XVII to REACH. In particular, the risks to human health and the environment arising from the use of Hexabromocyclododecane (HBCDD), Bis (2-ethylhexyl) phthalate (DEHP), Butyl benzyl phthalate (BBP) and Dibutylphthalate (DBP) should be considered as a priority. With a view to further restriction of substances, the Commission should re-investigate the substances, which were subject to previous assessments, in accordance with the new criteria of this Directive as part of the first review.” A review and amendment of the list of restricted substances shall be considered by the Commission within 3 years after the entry into force of the Directive, and periodically thereafter on its own initiative or following the submission of a proposal by a Member State (see Article 6 of the current version of the Directive).

HBCD and all major diastereoisomers (CAS Nos 25637-99-4, 3194-55-6, 134237-50-6, 134237-51-7, 134237-52-8) has been identified as “substance of very high concern” (SVHC) and is included on the candidate list of SVHC since 28.10.2008 due to its PBT properties (Decision number ED/67/2008; see also [ECHA 2008]). As a consequence, HBCD underlies specific obligations resulting from the inclusion in the Candidate List of Substances of Very High Concern. The specific provisions on obligations linked to the substances on the Candidate List of Substances of Very High Concern are laid down in the REACH Regulation in Articles 7.2 (Notification to ECHA), 7.3 (Supply of appropriate instructions), 31.1 (Provision of Safety Data Sheet) and 33 (Duty to communicate safe use information or responding to customer requests). These obligations are in particular:

- From the date of inclusion EU or EEA suppliers of articles which contain substances on the Candidate List in a concentration above 0.1% (w/w) have to provide sufficient information to allow safe use of the article to their customers or upon request, to a consumer within 45 days of the receipt of the request. This information must contain as a minimum the name of the substance.
- From 2011 EU and EEA producers or importers of articles have to notify ECHA if their article contains a substance on the Candidate List. This obligation applies if the substance is present in those articles in quantities totalling over one tonne per producer or importer per year and if the substance is present in those articles above a concentration of 0.1% (w/w). For substances included in the Candidate List before 1 December 2010, the notifications have to be submitted not later than 1 June 2011. For substances included in the Candidate List on or after 1 December 2010, the notifications have to be submitted no later than 6 months after the inclusion. (Note: A notification is not required when (i) the producer or importer of an article can exclude exposure of humans and the environment during the use and disposal of the article. In such cases, the producer or importer shall however supply appropriate instructions to the recipient of the article (ii) the substance has already been registered for that use.
- For the substances on their own, from the date of inclusion EU and EEA suppliers of substances on the Candidate List have to provide their customers with a safety data sheet.
- For the substances in mixtures, from the date of inclusion EU and EEA suppliers of mixtures not classified as dangerous according to Directive 1999/45/EC have to provide the recipients, at their request, with a safety data sheet if the mixture contains at least one substance on the Candidate List and the individual concentration of this substance in the mixture is $\geq 0.1\%$ (w/w) for non-gaseous mixtures if the substance is persistent, bioaccumulative and toxic (PBT) or very persistent and very bioaccumulative (vPvB).
- This is without prejudice to the general obligation, applicable from 1 June 2007, for all EU and EEA suppliers of mixtures not classified as dangerous according to Directive 1999/45/EC to provide the recipients, at their request, with a safety data sheet if the mixture contains a substance with an individual concentration $\geq 1\%$ (w/w) for non-gaseous mixtures and $\geq 0.2\%$ by volume for gaseous mixtures where that substance poses human health or environmental hazards.

HBCD is listed in Annex XIV of the REACH regulation (see Regulation (EU) No 143/2011) with the sunset date 21 July 2015. A SVHC which is included into Annex XIV is subject to the authorisation regime, with Article 56 REACH Regulation setting out the basic principle that an Annex XIV substance must not be

placed on the market or used after elapsing of the sunset date, unless an authorisation is granted or one of the other exemptions apply. Note that Article 56(6) lit. (b) REACH Regulation REACH foresees that the authorisation regime does not apply for PBT substances such as HBCD, if they are present in a mixture below a concentration limit of 0.1 % w/w.

6.6.5 Occurrence of HBCD in Europe

Trade

The total volume used in the EU was estimated to amount to about 11,580 t in 2006. The demand of HBCD within the EU is bigger than the production, and a net import to the EU was expected to have been about 6,000 t in 2006. Poland reported on imports from China of about 500 t and Romania of 185 t from an unknown supplier [UNEP/POPS/POPRC.6/10]. Another source reported that approximately 5,000 t/y are imported from the USA [ECB 2008]. EU production of HBCD between 2003 and 2007 was fairly static at 6000 tonnes implying an increase in net imports of almost 50% over this period. Estimates of EU consumption are slightly different. They imply an increase of almost 15% between 2003 and 2007 with net imports increasing by almost 40%. The best estimate of imports of HBCD in 2006 is 5,580 tonnes. It is unclear whether HBCD is imported into the EU in the form of masterbatch and plastic compounds [IOM 2008].

HBCD contained in expanded polystyrene (EPS) and extruded polystyrene (XPS) for the construction industry is not likely to be transported long distances due to the bulkiness of the material. Furthermore, the extrusion and expansion processes are relatively straightforward industrial processes, which allow them to be carried out in the region where the products are needed. Transport of compounded polystyrene (PS) with HBCD (granules, masterbatch or beads) over long distances cannot be excluded, but information on this is lacking. Packaging material containing HBCD (e.g. EPS) to protect sensitive equipment is likely to be transported worldwide²⁹. High impact polystyrene (HIPS) containing HBCD is likely to be imported to the EU in electrical and electronic equipment, but no data on this has been submitted. Textile that is back-coated with a HBCD containing layer is imported from the US and other countries in unknown quantities. Import of polymer dispersions for textiles containing HBCD cannot be excluded. In summary, import (and export) of HBCD in articles is likely to occur but has not been possible to quantify. No up to date information on the export of HBCD from the EU is available. [IOM2008]

HBCD is probably also exported from the EU (no quantitative data), both as a chemical (on its own or in formulations) as well as in finished articles [ECB 2008]. Data on export has been calculated in the RAR using the production, import and consumption figures of HBCD in the EU, resulting in an export figure around 1,880 t/y. The RAR, however, does not consider imports and exports of HBCD treated articles [ECB 2008]. According to IOM 2008, it is unclear whether any HBCD is exported out of the EU as a simple substance. Given the proximity of the production plant to major port facilities, it is likely that some HBCD would be exported outwith the EU as well as transported within the EU [IOM 2008]. It is unclear whether HBCD is exported out of the EU in the form of masterbatch and plastic compounds. Some

²⁹ The PS foam industry considers that only a very limited amount of EPS Packaging material containing HBCD to protect sensitive equipment is likely to be transported worldwide. [PS foam 2011]

export to other European countries that are not part of the EU would seem likely [IOM 2008].

Due to the lack of specific data on import and export EU consumption data are taken as a basis for the calculation of the mass flows.

EU consumption

Data from BSEF shows that the highest demand and use of HBCD is in the EU [CPAN 2010]. The estimated worldwide production volume of HBCD in 2001 was 16,700 t with approximately 57% of global distribution occurring in Europe, 23% in Asia-Pacific region, 17% in North America (5% in other regions) (see [Nordic COM 2007]). According to industry information, the estimated global industrial HBCD consumption was in 1999 15,900 t, in 2001 16,700 t in 2002 21,400 and in 2003 21,900 t (see [Morf 2007]).

According to industry information, the market for HBCD in Europe was 8,900 tonnes in 1999. A figure of 9,618 t/y has been used within the HBCD Risk Assessment Report (RAR), [ECB 2008].

According to industry information the EU industrial HBCD consumption was in 2003 9,600 t, in 2004 9,750 t, in 2005 11,000, in 2006 11,580 t, in 2007 10,897 t and in 2008 8913 t (see [IOM 2008] and [VECAP 2009]).

According to the EU PS foam industry EU HBCD consumption from 2002 to 2009 was 7.376 t in 2002 8.403 t in 2003, 8.886 t in 2004, 9.812 t in 2005, 11.160 t in 2006, 11.062 t in 2007, 10.733 t in 2008 and 10.439 t in 2009 [PS foam 2011].

Based on these available data on consumption of HBCD in Europe, the consumption since the late 1980ies can be estimated and is illustrated in Figure 6-92. The underlying figures are used for the calculation of the waste flows. Specific data are available for the time frame from 1999 until 2009. For the time from the late 1980ies (the first use of HBCD in flame-retarded polystyrene foams) to 1999 a constant increase in consumption has been assumed. The use of HBCD for other uses than in EPS or XPS (i.e. polymer dispersion for textile coating and HIPS for electronic products) that occurred already since the 1960ies until the end of the 1980ies has been neglected as the majority of the corresponding products have already become waste due to their limited service lifetime. They are consequently not considered relevant for current waste streams.

The development of the consumption after 2009 depends on the market situation, changes in the legal situation and the availability of alternatives. The PS foam industry expects a significant increase for all types of insulation materials for construction due to changes in the construction sector. HBCD is expected to be listed under the POPs Protocol and the Stockholm Convention. Industry is carrying out intensive research and is confident that within 5 to 10 years at least one appropriate alternative will be available. Against this background, for the time after 2013 a continuous declining trend is assumed with a complete phase out of the consumption in 2017.

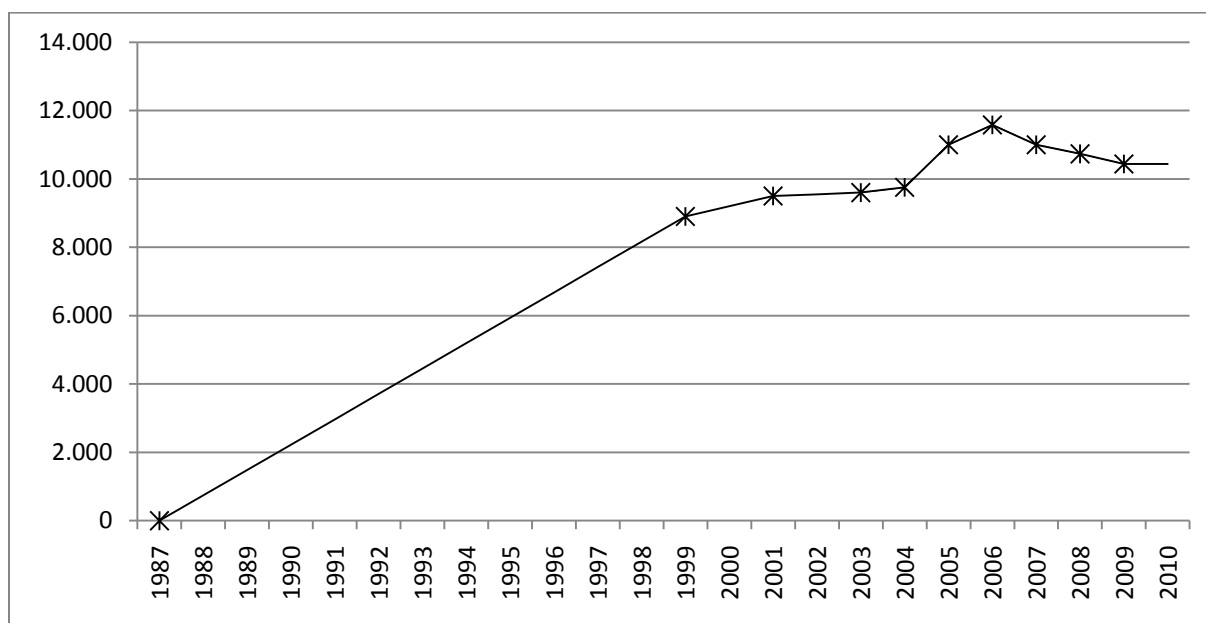


Figure 6-92: Estimation of the consumption of HBCD in tonnes in Europe since the late 1980ies until 2010 (estimation based on data from [ECB 2008], [Nordic COM 2007], [IOM 2008], [PS foam 2011])

This estimation fits comparatively well with a modeled consumption of HBCD in Europe as carried out by Morf et. al for the time from 1980 until 2005 (see [Morf 2007] and Figure 6-93). Compared to the estimation illustrated in Figure 6-92, the figures provided by Morf are particularly higher for the time before 1992. This is particularly due to the fact that at that time HBCD was not yet used in EPS/XPS and because the estimation illustrated does not consider the use of HBCD in non EPS/XPS uses because it is not relevant in the scope of the present study (as explained above).

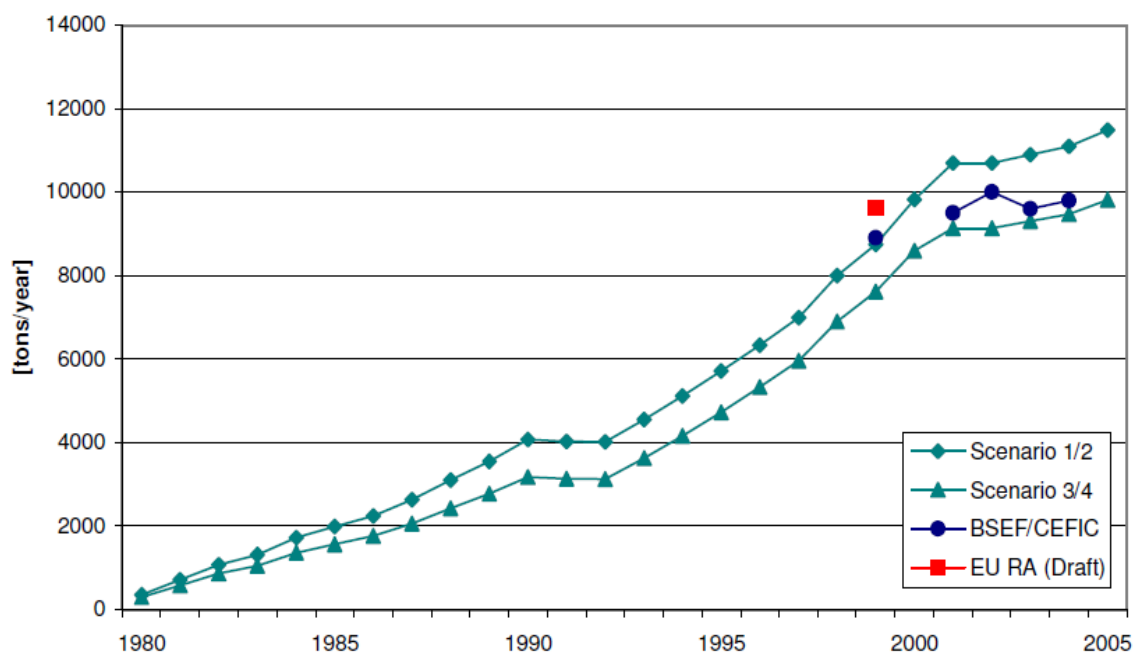


Figure 6-93: Modelled consumption of HBCD in Europe and comparison with data provided by BSEF/CEFIC and the estimate in the EU risk draft [Morf 2007]

6.6.6 Relevance of occurrence of HBCD in Europe

HBCD 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]. It has to be noted that not all products of these types are flame retarded with HBCD but only a certain share (e.g. approximately 77% of the EPS foams used for construction contains HBCD and most XPS products contain HBCD [EPS foam 2011])

Table 6-69: Product types, products and the relevance related to the use of HBCD in the EU according to different sources

Product type	EPS	XPS	EPS	XPS	HIPS	Polymer dispersion
Products	Insulation boards for construction		Other products		Electronic products and articles	Textile fabrics
ECB 2008	< 90%		n.s.		2%	~ 10% in 2003
IOM 2008	Most				2% in EU 15 2000 - 2004	2% in 2007
	48% in 2006	52% in 2006				
POPRC.6/10	Main application				2%	2%
SWEREA 2010 ³⁰	96.4%				1.8%	1.8%

The data indicate that the most important use of EPS and XPS is in the construction sector with approximately 90% of the HBCD consumption followed by the use of EPS and XPS for other uses such as insulation boards for vehicles, packaging material and other minor uses with approximately 6% of the consumption. HBCD is used in slightly higher quantities for XPS than for EPS (ratio ~ 52:48³¹). For HIPS and in textiles HBCD use represent each approximately 2% of the EU consumption.

The relative importance of the consumption of HBCD for textiles has changed over time. Available information indicates that the use since 2007 is approximately 2% of the consumption (see [IOM 2008], [POPRC.6/10], [Swerea 2010]). According to BSEF estimates formerly the consumption was about 800 tonnes per year [ECB 2008]. In the EU risk assessment the use in 2003 was estimated at 1054 tonnes [ECB 2008]. This corresponds to approximately 10% of the consumption at that time.

According to IOM 2008, the use of HBCDD in textiles is believed to have fallen substantially in recent years and industry data stated that 2% of HBCDD production went to textile manufacture in 2007, implying that a little over 200 tonnes of HBCDD was used in textiles in 2007 [IOM 2008].

Against this background it can be concluded, that the use of HBCD for textiles was approximately 10% of the consumption in the past (i.e. until 2003) and is currently (since 2007) about 2% of the EU HBCD consumption. During the time from 2003 to 2007 a decrease from 10 to 2% took place. The share for HIPS is considered constant over the considered time frame at 2% and the shares for EPS and XPS

³⁰ Calculated on the basis of Table 7 of [SWEREA 2010]

³¹ Considering the consumption data for XPS and EPS from [IOM 2008] for the years 2002 to 2006, the average ratio is 52.36% use for XPS and 47.64% use for EPS.

applications change accordingly between 2003 and 2007 from lower shares until 2003 to current shares of 43% for EPS for construction, 47% for XPS for construction, 6% EPS/XPS for non construction uses.

Taking the estimated EU HBCD consumption from of 1988 to 2010 as a basis this corresponds to the quantities for the different applications as shown in the following table:

Table 6-70: Estimated consumption in tonnes of HBCD in the EU for the relevant applications (based on data from [ECB 2008], [Nordic COM 2007], [IOM 2008], [PS foam 2011])

Year	Total (t)	EPS for construction	XPS for construction	EPS/XPS for other applications	HIPS for electrical products	Polymer dispersion for textiles
1988	742	291	321	41	15	74
1989	1,483	583	641	82	30	148
1990	2,225	874	962	122	45	223
1991	2,967	1,165	1,282	163	59	297
1992	3,708	1,456	1,603	204	74	371
1993	4,450	1,748	1,924	245	89	445
1994	5,192	2,039	2,244	286	104	519
1995	5,933	2,330	2,565	326	119	593
1996	6,675	2,621	2,886	367	134	668
1997	7,417	2,913	3,206	408	148	742
1998	8,158	3,204	3,527	449	163	816
1999	8,900	3,495	3,847	490	178	890
2000	9,200	3,613	3,977	506	184	920
2001	9,500	3,731	4,107	523	190	950
2002	9,550	3,750	4,128	525	191	955
2003	9,600	3,770	4,150	528	192	960
2004	9,750	3,916	4,311	548	195	780
2005	11,000	4,516	4,971	633	220	660
2006	11,580	4,858	5,347	680	232	463
2007	11,000	4,712	5,188	660	220	220
2008	10,733	4,598	5,062	644	215	215
2009	10,439	4,472	4,923	626	209	209
2010	10,439	4,472	4,923	626	209	209

In order to be able to estimate the relevance of HBCD containing products for the waste stream it is indispensable to assess the average lifetimes of the relevant products before they become waste. A mean lifetime has been applied and an uncertainty range has been considered within which it is expected that most of the relevant products become waste. The lifetimes have been assumed generally in line with the information given by Morf et al. 2007.

According to a recent study in Switzerland, the mean service life of EPS insulation panels is about 40 to 50 years, while in the EU risk assessment draft report, the mean service life of EPS and XPS insulation panels was estimated as 50 years [Morf et al. 2007]. In the EU risk assessment a lifetime of 30-100 years

was estimated for building insulation [ECB 2008]. According to CEN TC350, buildings are expected to have a typical lifetime of 80 – 100 years [PS foam 2011]. Industry claims lifetimes of 70 to 100 years [PS foam 2011]. As a consequence the average lifetime of insulation panels may even be longer than 50 years. For the calculation of mass flows a lifetime of 50 years (standard deviation of 25 years) is applied for HBCD containing EPS and XPS products in the construction sector. These differences in the lifetimes have no impact on the relevant conclusions of the present report.

For other EPS and XPS products (insulation boards for vehicles, automobile cushions for children, rigid packaging material for fragile equipment, packaging material such as “chips” and shaped EPS-boards, props for theatre and film, and in exhibitions) specific information on lifetimes is not available. It can be assumed that packaging material has a comparatively short average lifetime (usually less than 2 years but may be stored for longer times before disposal). Also for the other uses (such as automobile cushions for children, props for theatre and film, and in exhibitions lifetimes usually below 10 years can be assumed. The service life of passenger cars was estimated between 10 and 14 years. Other vehicle types may have slightly different service lives. Lorries, buses or airplanes have longer service lives, whereas it is shorter for e.g. motorbikes. As lorries or railway cars are the relevant transport vehicles where insulation boards are used, a lifetime between 12 and 20 years is assumed. As no specific figures are available for the individual uses a lifetime time of 10 years with an equal distribution from 0 to 20 years is applied for HBCD containing EPS and XPS products in non construction uses.

The actual lifetime of office and consumer electronics consists of a service life of 5–6 years and an additional period of 3–4 years where the appliances are stored until they are disposed. Hence, the lifetime in the use phase is approximately 9 years at present. In the mid 1990s, the mean lifetime was 11 years and a decrease to 7 years is expected in future. Large household appliances like freezers or washing machines have a lifetime of 15 years and more, whereas small household appliances like vacuum cleaners or coffee percolators are in use for about 5 to 10 years (see [Morf et al. 2007]). For the calculation of mass flows a lifetime of 10 years with a standard deviation of 5 years has been assumed for HIPS in electrical appliances.

Service lifetimes of furniture have been estimated around 10 years (see [Morf et al. 2007]). For the calculation of mass flows a lifetime of 10 years with a standard deviation of 3 years has been assumed.

Table 6-71: Lifetimes of relevant products

Application	EPS for construction	XPS for construction	EPS/XPS for other applications	HIPS	Polymer dispersion
Lifetime (years; standard deviation in brackets)	50 (+/-25)	50 (+/-25)	0 to 20 (equal distribution)	9 (+/-5)	10 (+/-3)

For the assessment of the amounts of HBCD which will finally enter the waste streams it has to be taken into consideration that not all HBCD consumed in the EU will become waste. A certain share will be released to the environment prior to the disposal of the relevant products. Therefore, the consumption quantities have to be reduced by the quantities released to the environment during use of HBCD in formulations, professional uses and releases during the service life of products. The corresponding releases to air, waste water and surface water have been estimated for the year 2006 (see [IOM 2008])

and [Swerea 2010]). On the basis of these estimates release factors for the product categories can be derived. The relative release compared to the consumed quantity for EPS and XPS represent specific shares of the consumed quantity which can be calculated by applying specific release factors³². These release factors contain releases to the environment during use of HBCD in formulations, professional uses and releases during the service life of products and based on consumption figures for 2006. The data for textile coating use estimates for 2007 (summarised in Table 8 of [Swerea 2010] and based on [IOM 2008]).

Release factors can be derived from the EU risk assessment [ECB 2008]. In [IOM 2008] it is stated that “The more recent information on the amounts of HBCDD used in textile coatings indicates a decrease over the figures from 2000-2004, by 80% for the 2007 figures. Assuming that the emissions would decrease pro rata, the revised emissions would be as in Table 2.24 and 2.25.” (see corresponding tables of [IOM 2008]). These data have been used in SWEREA 2010 (see table 8 of that report) and for the present report. In order to consider most recent data for current release estimates, the release factors are based on [Swerea 2010] and [IOM 2008] for the calculation of releases from the time after 2006. In order to correctly reflect historical releases, the releases for the time before 2007 are calculated according to the use shares of the specific applications and correspond to the release estimates made in the EU risk assessment. The release factors used are therefore considered to reflect reality as far as possible according to the most recent state of knowledge.

HBCD has now been used for several decades. The annual consumption of HBCD which is incorporated in articles made of polystyrene and in coating layers for textile back-coating for the EU market is accumulating and forming a significant stockpile. Since the lifetime of most end-products is often long – it varies between less than 1 year and more than 50 years – a huge share of the relevant products will become waste in the future. Depending on the lifetimes of the relevant products the HBCD used has already become waste or becomes waste by 2010 or will become waste after 2010.

Table 6-72 shows the estimated cumulated quantity of HBCD consumed in the EU from 1988 to 2010 for the different use categories. The consumption figures are based on the available data on consumption of HBCD in Europe (see Table 6-70). The releases are calculated by using the above mentioned release factors.

Table 6-72: Cumulated use, release and disposal of HBCD consumed from 1988 to 2010 (in tonnes)

Use category	Use (t)	Release (t)	Total HBCD becoming waste (t)
EPS	69,125	10.0	69,115
XPS	76,096	11.0	76,085
EPS&XPS	9,681	1.4	9,680
HIPS	3,413	0.5	3,412
PD	12,325	79.7	12,246
Total	170,641	102.6	170,538

³² The release factors are 0.0001389 for EPS and XPS, 0.0000028 for HIPS and 0.0001294 for textiles of the consumed quantity (calculated on the basis of [SWEREA 2010])

It can be concluded that from 1988 to 2010 about 170,000 tonnes of HBCD have been consumed within the EU. This consumption is related to the release of around 100 tonnes HBCD to the environment.

Table 6-73 shows the cumulated HBCD quantities in waste from estimated use from 1988 to 2017. The amount of HBCD becoming waste is reduced by the corresponding releases. The amount of HBCD which becomes waste by 2010 and after 2010 has been calculated using the product lifetimes and standard deviations shown in Table 6-71. The amount becoming waste after 2010 is based on the hypothetical assumption that the consumption declines continuously after 2013 and that from 2017 the consumption will be zero.

Table 6-73: Cumulated HBCD quantities in waste from estimated use from 1988 to 2017 (in tonnes)

Use category	Total HBCD in waste (t)	Will become waste after 2010 (t)	Already waste by 2010 (t)
EPS	88,344	88,228	116
XPS	97,253	97,125	128
EPS&XPS	12,498	3,273	9,225
HIPS	4,352	2,899	1,453
PD	13,179	6,033	7,146
Total	215,626	197,559	18,068

It can be concluded that the bulk of HBCD containing waste will have to be disposed off in the future. 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.

Table 6-74 gives an overview of the HBCD content for different product categories according to several sources.

Table 6-74: HBCD content for different product categories according to several sources

Product category	HBCD content (%)	Note	Source	Remark
EPS products	0.7	mean	[CeficEfra 2006]	
EPS products	0.7	max	[IOM 2008] [ECB 2008]	
EPS products	0.7		[CPAN 2010]	
EPS products	0.7 to 1.0		[Morf 2007]	
EPS products	0.51	mean		Average concentration calculated on the basis of [PS foam 2011]
XPS products	2.5	mean	[CeficEfra 2006]	
XPS products	2.0		[CPAN 2010]	
XPS products	1.5		[Morf 2007]	
XPS products	1.0 to 3.0	range	[ECB 2008]	
HIPS products	1.0 to 3.0	range	[ECB 2008] [IOM 2008]	
HIPS products	5.0 to 7.0	range	[ECB 2008] [IOM 2008]	7% used in EU RAR as a realistic worst case [ECB 2008]
Textile products	5 to 48		[ECB 2008] [IOM 2008]	Concentration in the dispersion (the concentration in the dilution of the dispersion applied is ~ 15%)
Textile products	~ 25		[ECB 2008] [IOM 2008] [Morf 2007]	Concentration in the coated layer

Product category	HBCD content (%)	Note	Source	Remark
			[Swerea 2010]	
Textile products	7 to 9	range	[Morf 2007]	Load of the coated textile
Textile products	2.2 to 4.3	range	[UNEP/POPS/POPR C.6/10]	HBCD load of the coated textile of flame retarded Japanese upholstery fabrics
Treated textile	6 to 15	range	[CeficEfra 2006]	Textile back coating

Based on the available information the mass flows will be calculated on the basis of the following best estimates as realistic average HBCD contents:

- In EPS products: 0.51 %
- in XPS products: 2 %
- in EPS and XPS without differentiation: 1.35 %
- in HIPS products: 4 %
- in Textile products: 25 % in the coated layer; 8 % load of the coated textile

Applying these concentrations to the cumulated HBCD quantities in waste from estimated use from 1988 to 2017, it can be calculated that the EU consumption of HBCD from 1988 to 2017 is related to the waste quantities shown in Table 6-75. The waste quantities by 2010 and after 2010 have been calculated using the product lifetimes and standard deviations shown in Table 6-71. The amount becoming waste after 2010 is based on the hypothetical assumption that the consumption of HBCD declines continuously after 2013 and that from 2017 the consumption will be zero.

Table 6-75: Cumulated quantities of waste containing HBCD from estimated use of HBCD from 1988 to 2017

Waste	Total HBCD in products (t)	HBCD concentration (%)	Related waste quantity (t)	Will become waste after 2010 (t)	Already waste by 2010 (t)
EPS products	88,344	0.51	17,322,428	17,299,674	22,754
XPS products	97,253	2.0	4,862,653	4,856,266	6,387
EPS&XPS products	12,498	1.35	925,788	242,463	683,326
HIPS products	4,352	4.0	108,793	72,465	36,328
Coated textiles	13,179	8.0	164,739	75,416	89,323
Total	215,626		23,384,401	22,546,283	838,118

It becomes obvious that out of a total relevant waste quantity of about 23 million tonnes, less than one million tonnes of waste has been disposed of by 2010 (~840,000 t).

According to industry representatives, the PS foam industry is carrying out promising research on chemical alternatives and is confident that within 5 to 10 years at least one appropriate alternative flame retardant for EPS and XPS foams will be available [PS foam 2011].

6.6.7 Selection of Relevant Uses, Applications and Finished Articles

All principal product types in which HBCD is used are considered relevant in the scope of the present study:

1. Expandable Polystyrene (EPS)
2. Extruded Polystyrene (XPS)
3. High Impact Polystyrene (HIPS)
4. Polymer dispersions for textiles

The following figure illustrates the relevance of the HBCD consumption for these product types for 2009 (latest data available) where the EU consumption of HBCD amounted to approximately 10,500 tonnes. Besides the releases to the environment, all HBCD consumed is relevant for disposal and hence for waste streams.

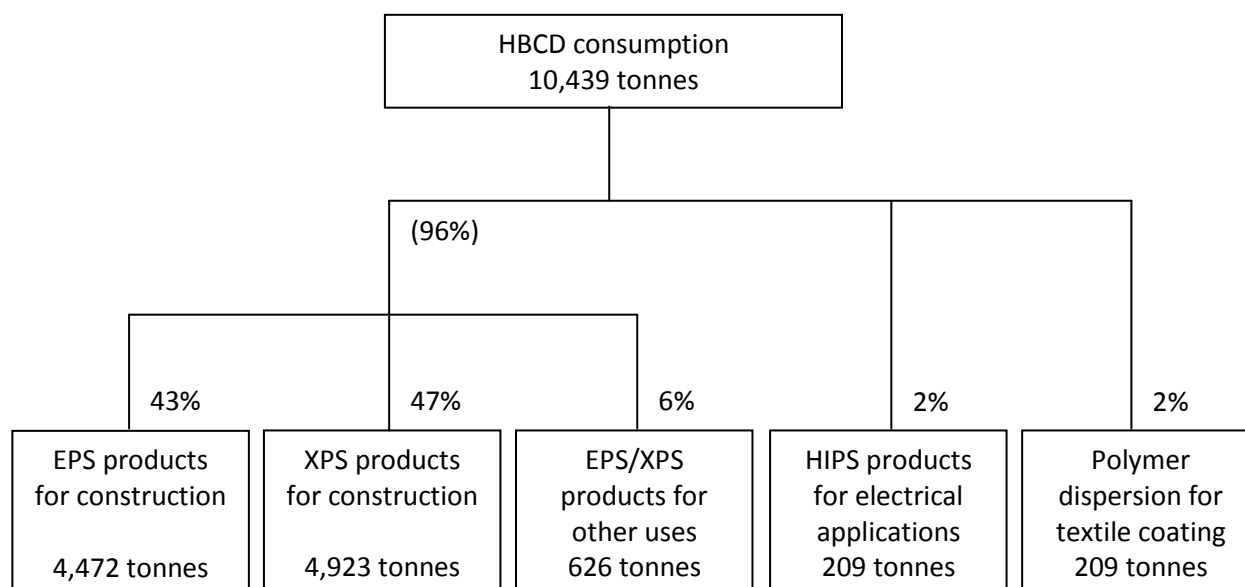


Figure 6-94: Relevance of product types with respect to the HBCD consumption for the year 2009 (Figures indicate the HBCD content)

It has to be noted that prior to the year 2004 the EU consumption of HBCD for textiles was approximately 10% of the total consumption. Since 2007 it represents about 2% of the total consumption. For the time from 2003 to 2007 a decrease from 10 to 2% took place. The share for HIPS is considered constant over the considered time frame at 2% and the shares for EPS and XPS applications change accordingly between 2003 and 2007 from lower shares until 2003 to current shares of 43% for EPS for construction, 47% for XPS for construction, 6% EPS/XPS for non construction uses.

In order to illustrate the historic situation for 2003 before, the following figure illustrates the relevance of the HBCD consumption for these product types for 2003.

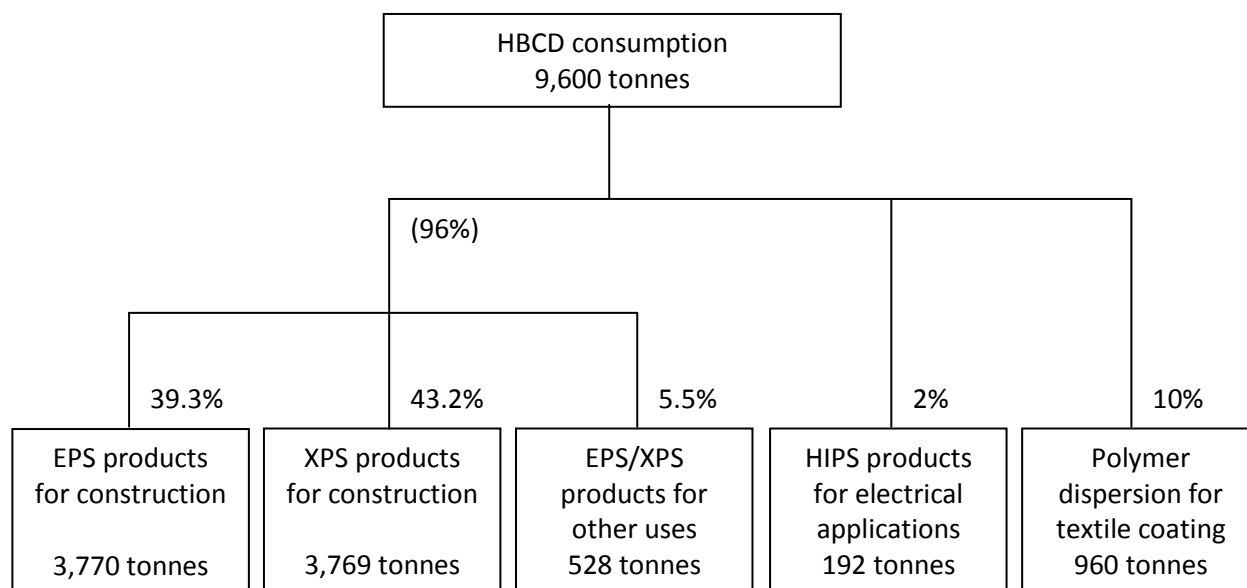


Figure 6-95: Relevance of product types with respect to the HBCD consumption for the year 2003 (Figures indicate the HBCD content)

6.6.8 Expanded Polystyrene (EPS) used for construction

A) Background

EPS is a rigid cellular plastic which is used in various applications such as insulation panels/boards for the construction sector or rigid packaging material for fragile equipment [IOM 2008].

EPS and XPS foams/boards have a long track record as insulation materials in private and public buildings. Both, the extruded and expanded foams, are widely used as rigid insulation all over the world, due to their good insulation properties, high moisture resistance, strength, performance and affordability [Building Green 2009].

In their pure form, EPS/XPS foams are easily flammable or ignitable. In several European countries, national building regulations require insulation materials to be flame resistant. In order to fulfil these fire safety requirements, EPS/XPS foams have been widely treated with HBCD flame retardants. HBCD is proven to be highly effective and durable flame retardant, ensuring highest levels of fire safety while applied in quite low quantities [BSEF 2007].

According to the Bromine Science Environmental Forum (BSEF), EPS/XPS manufacturers have conducted extensive research in order to identify appropriate alternatives to HBCD in EPS/XPS boards, but have not been able to identify technically feasible solutions yet. However, intensive research by industry continues ([BSEF 2007], [PlasticsEurope 2010], [PS foam 2011]). The PS foam industry is confident that within 5 to 10 years at least one appropriate alternative flame retardant for EPS and XPS foams will be available [PS foam 2011].

A possible solution to avoid HBCD treated PS foams is to use other insulation materials such as mineral wool (stone wool and glass wool) and foam glass, which do not require the use of flame retardants [BuildingGreen 2009]. Different insulation products have specific attributes which lead them to be selectively chosen for specific applications. This results in a range of insulation products which cannot automatically be exchanged in those applications [PS foam 2011].

B) Process input (raw material) and output (waste, air, water, product)

EPS is formed in a batch process, by suspension polymerisation of styrene which is dispersed in the form of small droplets in water. Additives are added which generally contain suspension agents, free-radical-forming initiators and HBCD flame retardants. The HBCD powder is typically delivered in 25 kg paper bags which are emptied into an intermediate storage container from where the flame retardant can be further transported via pipes. The HBCD powder is suspended at low temperatures in styrene prior to the addition of the water phase. In the reactor, styrene droplets (0.01-0.5 mm) in the continuous water phase, polymerisation occurs while the reactor content is heated and held at appropriate reaction temperature (65-140°C). An expansion agent (e.g. pentane) is added to the reactor under pressure, where it is absorbed in the polymer droplets. In the resulting EPS beads, HBCD is incorporated as an integral and encapsulated component within the polymer matrix, with consistent concentration throughout the bead. The maximum concentration of HBCD in EPS beads is assumed to

be 0.7 % (w/w), [ECB 2008, Swerea 2010].

Subsequently, the reactor is cooled down and the beads are separated from the water by means of centrifugation. The decanted water, which could possibly contain dissolved and dispersed HBCD, is reused and from time to time exchanged (e.g. on an annual basis or less frequently). The EPS beads are dried, and further classified into different size fractions and surface coated. Finally, the beads are packed in bins, bags, or transported in bulk trucks to EPS foam converters for further processing [ECB 2008, Swerea 2010].

In the first step the beads are pre-expanded in loose form with the help of dry saturated steam in pre-expanders. After expansion the beads are partly dried in fluid bed driers and subsequently stored in large permeable silos for several hours (up to 24 h). During this stage the beads can dry further and reach equilibrium with the ambient atmosphere around them. In the third step the beads are transported/blown via pipes into block or shape moulds or continuous moulding machines in which the final product gets its shape. The foam can afterwards be further formed by cutting, sawing etc. [ECB 2008, Swerea 2010].

Figure 6-7 indicates the main inputs and outputs associated with the production of EPS foams/boards containing HBCD.

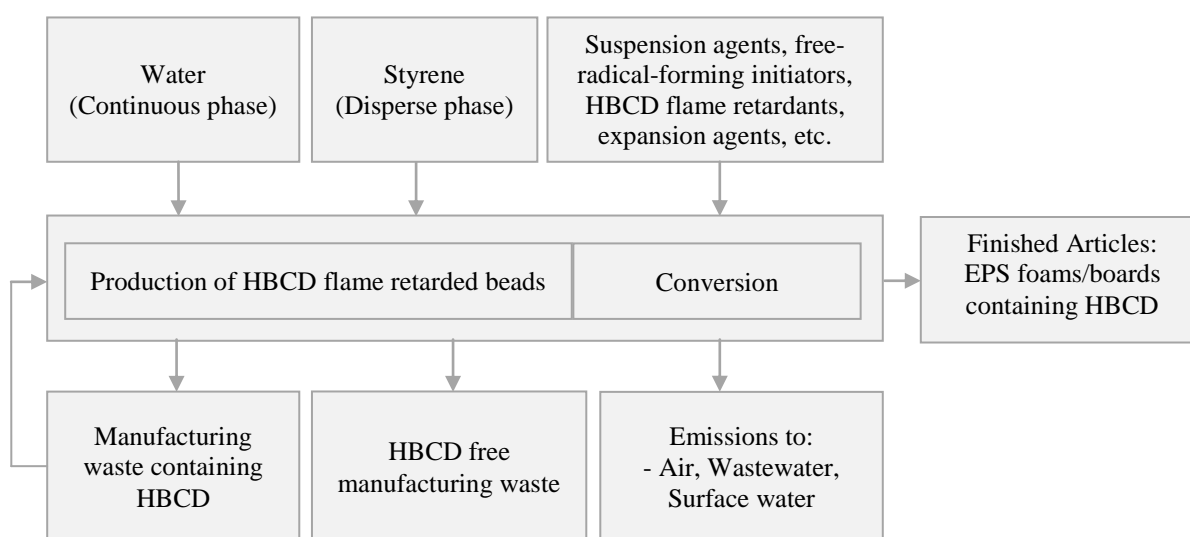


Figure 6-96: Relevant process inputs/outputs and products (HBCD treated EPS)

Inputs:

- The major input of concern is the input of HBCD flame retardant to the manufacturing process.
- Other inputs to the EPS bead respective foam production process are: water (continuous phase), styrene droplets (disperse phase), suspension agents, free-radical-foaming initiators, expansion agents and possibly other inputs.

Outputs:

- Finished articles: HBCD flame retarded EPS foams/boards

- Manufacturing waste can be recycled within the manufacturing process [ECB 2008].
- During the use of HBCD in formulations for EPS products approximately 0.000251 % of the consumed HBCD are emitted to air, 0.000619% are emitted to waste water and 0.002723 % are emitted to surface water (total release = 0.003592 %; calculated on the basis of [Swerea 2010]).

C) Generation factors for product/waste and other output (air, water)

The generation factor for EPS and XPS is $1 - 0.0001389 = 0.9998611$ as the releases from use of HBCD in formulations, professional uses and releases during service life sum up to a share of 0.01389 % of the HBCD consumption (calculated on the basis of [Swerea 2010]). This means that an EPS product that becomes waste at the end of its service life contains approximately 99.986 % of the HBCD which was consumed for its manufacturing. Potential releases during disposal operations are not considered as these are related to significant uncertainty.

Concerning releases polystyrene insulation foams are the major use of HBCD. However, this use is related to comparatively low emissions in production and during service life. Concerning releases during disposal, available information is the following:

According to industry, removal of PS foams during construction and demolition is not a dusty operation. XPS insulation foam generates low amounts of dust in installation and use and the same is likely to apply to demolition. IOM 2008 refers to manual breaking of boards tests which indicated no particulate losses for XPS and 90 g/tonne for EPS [PS foam 2011].

According to [ECB 2008], there is a big uncertainty in future releases of HBCD from demolition of buildings insulated with flame retarded polystyrene (EPS and XPS).

The uncertainties depend, among other things, on how and by whom the demolition will be performed. Current methods of demolition include; implode a structure with explosives, use a crane and wrecking ball technique, or deconstruct the structure (US EPA; www.epa.gov/epaoswer/non-hw/debris/). This handling may contribute to release of minor particles and fragments of the material spread in the environment. HBCD may then be released from these particles to the environmental compartments. Larger pieces of polystyrene board may be recycled. [ECB 2008]

[IOM 2008] summarises the information from the EU risk assessment [ECB 2008] and further elaborates on possible releases during disposal (see [IOM 2008] page 27):

There may be releases of HBCDD when buildings insulated with flame retarded EPS are demolished. Estimates of possible future emissions from insulation boards to the wider environment at disposal are included in the risk assessment. The basis for the estimates is as follows. For boards which are recovered, possible losses of particulates were based on tests on manual breaking of board samples (information not referenced in the EU risk assessment). There were no particulate losses for XPS and 90 g particulates per tonne for EPS. The proportion of demolition waste recycled was taken as 30% (from RTD information). A quantity of 4,000 tonnes per year of HBCDD in EPS was assumed (as 50% of the 8,000 tonnes of HBCDD used in insulation boards). The resulting emission was 108 kg per year.

For the 70% of boards assumed not to be recycled, an emission factor of 0.1% was used. (This is referenced to Anonymous, 1998 in the risk assessment, but this source does not appear in the reference list. In an earlier draft of the assessment, this source was given as a draft version of the OECD ESD on Plastics Additives (however, no such emission factor could be found in that source). Applying this factor to 70% of the 8,000 tonnes gives an emission of 5,600 kg per year.

The same approaches can be applied to the tonnages for 2006. For use of 5,301 tonnes of HBCDD in EPS, an emission of 143 kg per year (in EPS particulates) is estimated (assuming 30% recycling). For the use of 12,160 tonnes in EPS and XPS, emissions of 8,5112 kg per year of HBCDD can be estimated for the 70% not recycled.

These estimates assume that the equivalent of one year's production is present in demolition waste each year. The figures apply to the EU27. The estimate of 30% of material recycled has been retained. It could be foreseen that in the future a higher percentage could be recycled (no specific evidence, but based on general trends in recycling). Emissions from recycled material are lower, and hence the above estimates should be a worst case. There are considerable uncertainties in these estimates and it is likely that a substantial proportion of buildings in which flame retarded EPS has been used are less than 30 years old and not likely to be subject to demolition at present or in the near future. EC (2008) suggested that such products had not yet been in use for 30 years. Therefore emissions from this source are unlikely to be significant at present, but may grow in the future, particularly from about 2025 onwards as increasing numbers of buildings containing HBCDD-treated EPS become subject to refurbishment or demolition. There have been no studies of predicted current or future emissions.

Concerning releases at disposal, it can be concluded that there is uncertainty on the importance of releases during deconstruction and demolition. Releases will depend on how buildings with HBCD containing PS foams will be deconstructed and demolished in the future. Releases at disposal are unlikely to be significant at present, but may grow in the future due to an increasing number of buildings insulated with PS foams and an ongoing increase of the thickness of the insulation material (expected to double from 2010 to 2020).

Based on the above introduced concentration of 0.51 % (w/w) HBCD in EPS foams, it can be concluded that with 1 tonne of HBCD approximately 196 tonnes of flame retarded EPS foams/boards can be manufactured.

During manufacturing of EPS cut-offs and other EPS factory waste are recycled back into the production process. Cut-off from professional use (installation of insulation boards) is collected for recycling through a Europe-wide network of collection points, organised both by local authorities and commercial enterprises [EUMEPS 2009]. Cut offs are estimated at 2%. 1% is considered to be recycled at the conversion site and 1% is considered to become waste.

A certain share of EPS product will remain in the environment e.g. if insulation of basements or insulation against frost heaves of road and railway embankments are not completely removed during deconstruction. Quantification is not available and quantities possibly left in the environment are not taken into account in the calculation of the mass flows.

D) Contamination data for output (and input)

Several literature sources indicate the concentration of HBCD in EPS beads to be 0.7% (w/w), [CeficEfra 2006; IOM 2008; ECB 2008; Albemarle 2009; CPAN 2010; Swerea 2010]. Morf (2007) specifies HBCD concentration values in EPS products between 0.7 and 1.0 % (w/w). Based on recent information from the HBCD industry, 0.51 % (w/w) will be used for all further calculations and estimations.

E) Activity data

As already discussed, 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]. Imports and exports of HBCD in insulation materials are considered to be insignificant and will therefore not be further taken into consideration. EPS packaging material containing HBCD, for protection of sensitive equipment, is likely to be transported worldwide, but has not been possible to quantify.

Also in this case of polystyrene granules, master batch or beads, no further considerations are made in this report. Transport of HBCD treated polystyrene (in form of granules, master batch or beads) over long distances may occur, however relevant information on this is lacking [IOM 2008].

Furthermore, it is unclear whether any HBCD has been exported from the EU as a substance itself [IOM 2008]. Consequently, due to the lack of specific quantitative data, this study will not further focus on exports of HBCD from the EU. Exclusively the imports of HBCD as a substance itself have been taken into account.

At this point it should be further mentioned that fire safety regulations can vary among individual countries. Therefore, regional differences in the consumption of HBCD occur. In the construction sector, Austria, Germany and Switzerland have strict legal requirements on all building materials regarding fire safety. Even more stringent requirements have been reported for Eastern Europe, where almost all insulation materials used contain flame retardants [Morf 2007]. In contrast, Nordic Countries do not use flame retarded organic insulation material [BSEF 2007]. The PS foam industry provides specific information on flame retardancy in building in specific EU Member states [PS foam industry 2010]:

- Flame retardancy is a must in PS foams in Austria, Czech Republic, Germany, Hungary, the Netherlands, Slovakia and Slovenia
- Flame retardancy is generally used for construction applications, approvals and insurance needs in Italy, Portugal and the UK
- For specific applications and for insurance needs flame retardancy is used in PS foams in France
- In Norway and Sweden a thermal barrier approach is used, however PS foams are without flame retardancy in non-building uses such as under road and rail applications

- In Denmark, Finland and Iceland flame retardancy (class E³³) is needed if foams are used in buildings
- In the Netherlands, flame retardancy is needed to provide class E even if a thermal barrier is used
- In Spain, EPS/XPS insulation foams of Class E may be used, if they are covered by non combustible facings (thermal barrier)

On average, approximately 77% of EPS insulation boards for construction currently manufactured in the EU are flame retarded with HBCD (reference year 2009; [PS foam 2011]).

As already discussed in previous chapters the mean service life of EPS insulation panels is indicated to be about 40 to 50 years. ECB (2008a) estimates the mean service life of EPS and XPS insulation panels to be 50 years. Hence, for the calculation of mass flows a lifetime of 50 years, with a standard deviation of 25 years will be applied for both EPS and XPS products for the construction sector.

For other EPS and XPS products a lifetime time of 10 years with an equal distribution from 0 to 20 years is applied for HBCD containing EPS and XPS products in non construction uses.

Table 6-70 shows the estimated consumption in tonnes of HBCD in the EU for the relevant EPS and XPS applications. This estimation is taken as a basis for the calculation of the substance and waste flows.

F) Waste treatment

C&D waste generation in the EU increased from 510 million tonnes in 2002 to 866 million tonnes in 2004 and to 970 million tonnes in 2006. The average C&D waste generation per capita has been estimated to be approximately 2 tonnes/year/capita, based on the data available from 2006. The generation figures vary among EU Member States. For instance in Latvia only around 0.04 tonnes of C&D waste is generated per capita, whereas in France 5.9 tonnes have been reported. [Bio 2010]

Significant variations can also be observed in relation to the recycling rates of C&D waste in the EU-27. 'Recycling rates' thereby refer to 'reuse, recycling and other form of material recovery'. It has been reported that Denmark, Estonia, Germany, Ireland and the Netherlands already fulfill the Directive's (WFD) 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. Based on these figures, an average recycling rate of 47 % is estimated in the EU-27. This average of 47% recycling rate for EU-27 is a broad estimation with a high uncertainty. However, it looks rather plausible, and within the range of estimates proposed by experts and literature ranging from 30 to 60 % [Bio 2010]. ECB (2008a) estimated the overall recycling of C&D materials to be 30 %, with an increase over time.

³³ Class E fire performance according to EN13501-1

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 small. 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 [Appricode 2006].

On the other hand, according to industry, the recycling rate of EPS is high compared to other plastics [EUMEPS 2009].

According to PS foam industry, polystyrene is easily recycled and is one of the most recycled plastics. Since EPS does not degrade or deteriorate, it can be readily recycled in several ways at the end of its lifetime. It can be added back into new polystyrene foam insulation boards or moulded into new applications such as plant pots, coat hangers, park benches or fence posts. Since EPS and XPS are made from oil, the stored energy can be recovered in modern incinerators and then used for local heating and for generation of electricity. Some EPS can be recycled and used for the manufacture of bricks where the polystyrene is gasified and leaves a cavity in the centre of the brick. It can also be remelted to new granulate to reduce the volumes of raw material, which would be beneficial for new EPS production of end products [CPAN 2010]. The use of EPS granules for soil treatment has been reported by the German authorities in the Stockholm Convention [CPAN 2010]. According to the PS foam industry the use of EPS granules for soil treatment does not take place anymore since a long time [PS foam 2011].

Information from industry (APME, Sofres Conseil for the Association of Plastics Manufacturers Europe, 1995) shows that, in 1992, out of the total C&D waste, plastic waste was 0.4 % by weight. EPS and XPS represented about 3 % of the plastic waste and therefore 0.01 % of the total C&D waste. The extrapolation to 2010 shows that EPS and XPS will represent a fraction of 0.08 % of the total C&D waste. This indicated the small proportion of total waste related to EPS and XPS. [ECB 2008]

Waste containing HBCD is generated at each life cycle step. According to ECB (2008a), the so called pre-consumer waste can be recycled into the process. The HBCD 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. HBCD 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 grinded 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 [ECB 2008]. It has also been reported that granulated EPS waste is used to improve the texture of agricultural and horticultural soil (UNEP/POPS/POPRC.6/10; Vogdt 2009).

According to information provided by the PS foam industry, EPS waste from demolition and deconstruction goes to landfill (39.3%), incineration with energy recovery (R1; 53.1%), and recycling (7.6%; mainly recycled in the conversion process into new EPS insulation boards). As it is not considered realistic that the complete waste fraction of EPS foams from C&D waste is currently incinerated with energy recovery (in several MS the capacity is not available) the average share for incineration of MSW of 39% R1 and 61% D10 is applied. Therefore for the mass flows the following shares are applied for EPS

from construction: Landfill 39.3 %, R1 20.7 %, D10 32.4 % and recycling 7.6 %.

In the future, an increasing amount of HBCD containing products will leave the use stage and enter the waste management sector. As a consequence, the amounts of HBCD to be disposed of will increase. The share of HBCD flame retarded EPS was comparatively low in the past and the bulk of this waste will be disposed of in the future with an expected peak around the 2050ies or later.

Figure 6-97 shows the estimated annual quantity of HBCD in EPS waste from construction and demolition to be disposed of from 2010 until 2060, considering an average lifetime of 50 +/- 25 years of EPS insulation boards. The total estimated accumulated amount of HBCD in waste from EPS products to be disposed of by 2085 amounts to approximately 88,000 tonnes. Around 116 tonnes thereof have been disposed of in 2010. The remainder will have to be disposed of in the future.

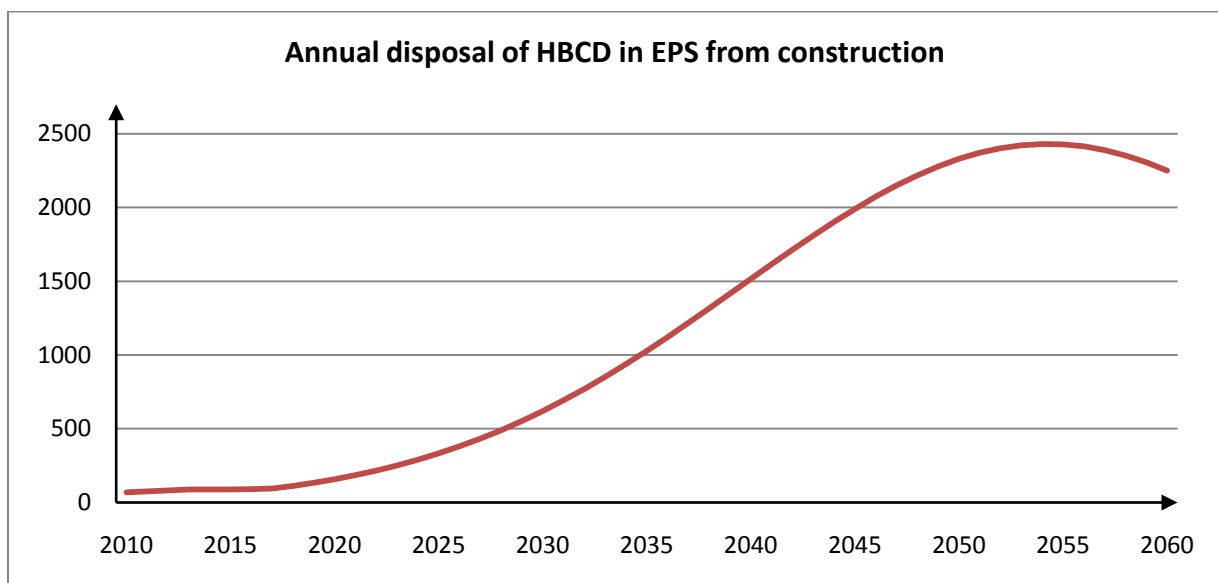


Figure 6-97: Estimated annual quantity of HBCD in EPS waste from construction and demolition that will be disposed of until 2060 (calculation based on the consumption figures provided in Table 6-70; lifetime 50 +/- 25 years; 1% cut offs becoming waste at the construction site taken into account)

To what extent end-products containing HBCD are landfilled, incinerated, left in the environment or recycled in the EU will largely depend on the waste management strategy chosen by different MS.

According to [CPAN 2010], most of flame retarded insulation material goes to landfills or incineration in the UNECE region. Landfilling is the last preferred route of the various solid waste management options that should be considered. In several countries it is prohibited to landfill combustible waste, such as EPS, because better alternatives are widely available [EUMEPS 2009].

EPS/XPS waste represents a valuable source of energy in solid waste recovery plants fully equipped to be compliant with air emissions requirements [BSEF 2007]. Since EPS and XPS are made from oil, the stored energy can be recovered in modern incinerators and then used for local heating and for generation of electricity [CPAN 2010]. 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.

Waste-to-energy incineration facilities accept EPS and are pleased to use it, due to its relatively high calorific value [EUMEPS 2009].

The properties of EPS/XPS insulation foams also enable their recycling and recovery. Recovered foams can be reprocessed as they match the properties of virgin material foams [BSEF 2007]. According to PS foam industry, PS is easily recycled and is one of the most recycled plastics. Since EPS does not degrade or deteriorate it can be readily recycled in several ways at the end of its lifetime. It can be added back into new polystyrene foam insulation boards or moulded into new applications such as plant pots, coat hangers, park benches or fence posts. Some EPS can be recycled and used for the manufacture of bricks where the PS is gasified and leaves a cavity in the centre of the brick. It can also be remelted to new granulate to reduce the volumes of raw material, which would be beneficial for new EPS production of end products (see [CPAN 2010], [EUMEPS 2009], [Swerea 2010]). It should be noted, that recycling rates may considerably increase in the future.

According to ECB (2008a), increasing quantities of EPS waste including post-consumer waste are recovered and collected via national waste management schemes, e.g. in Germany, and primarily mechanically recycled into EPS or PS products for construction applications. However, it is not clear to what extent, or if at all, a distinction is made between streams of flame retarded and non flame retarded material. There is no labelling/marketing system so that the waste streams could be kept apart. Theoretically, recycling can lead to an uncontrolled contamination of PS products, which do not need to be flame retarded, with HBCD [ECB 2008]. Hence recycling of EPS does occur in several European countries. The PS foam industry expects that recycling will increase the control of the material flow and will lead to decreased emissions. Industry is prepared to discuss any potential marking/labeling of the products to differentiate between HBCD-containing and non-HBCD-containing PS foams in order to facilitate the separation of HBCD containing waste from HBCD free waste [PS foam 2011].

EPS and XPS insulation has long lifetime in buildings. The use of EPS and XPS flame retarded with HBCD in construction started in the late 1980's and will need restoration or will be demolished in the future. Therefore, relatively little HBCD containing EPS and XPS insulation materials from the C&D sector have been treated by existing waste management infrastructure in the EU so far. Generally it is noteworthy, that insulation of buildings is becoming increasingly important in order to achieve sustainable construction. As a consequence, quantities of insulation material for construction are significantly increasing due to an increasing number of buildings insulated with PS foams and an increasing thickness of the insulation material (expected to double from 2010 to 2020; more details see [PS foam 2011]).

G) HBCD substance flow

As shown in Figure 6-98 approximately 69 t/y of HBCD have been disposed of in the EU in 2010. This quantity is comparatively low because the bulk of HBCD treated EPS has not entered the waste management stage yet. As already discussed, most of the HBCD contaminated EPS waste will be disposed of in the future, with the expected peak around the early 2050ies (more than 2,200 tonnes will have to be disposed of annually; see Figure 6-97).

In Figure 6-98, 0.65 tonnes of HBCD are indicated as emission to the environment. This quantity is

related to the consumption of approximately 4,500 tonnes of HBCD for the production of new EPS products for construction in 2010. These products will be used in the future and the main share of these products will become waste within the next 25 to 75 years i.e. between 2035 and 2085. The production and use of new HBCD treated EPS foams/boards in the construction sector is expected to decline, resulting in reduced emissions of HBCD to the environment from production and use of EPS products (release = 0.01389% of the HBCD consumption).

Scenario 1:

In the first scenario (see Figure 6-98) it is assumed that the HBCD containing EPS is separated from the non HBCD containing C&D fraction and treated separately.

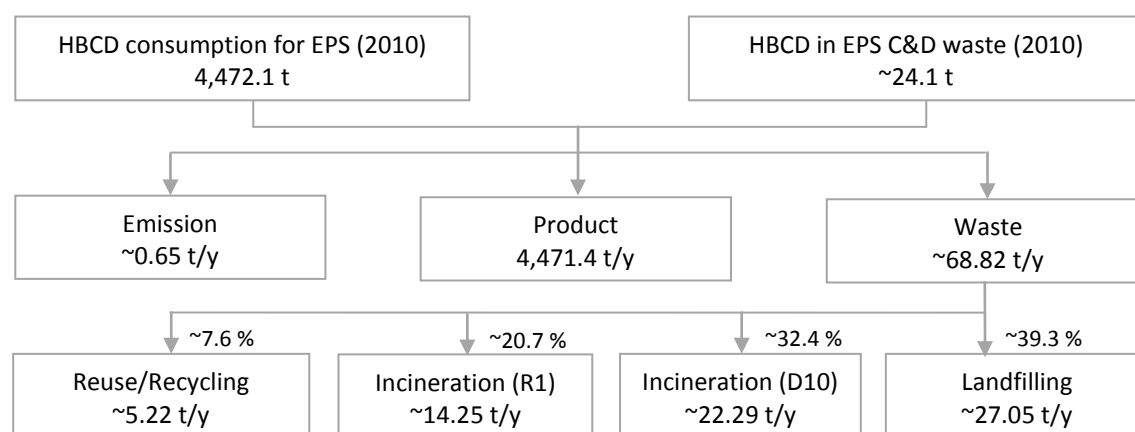


Figure 6-98: HBCD substance flow EPS products for construction (Scenario 1: HBCD containing EPS waste separated)

As shown in Figure 6-98 approximately 69 tonnes of HBCD in EPS waste from C&D have been landfilled (~27 tonnes), incinerated with (~14 tonnes)/without energy recovery (~22 tonnes) and reused/recycled (~5 tonnes). The main share of the approximately 4,500 tonnes of HBCD incorporated in new EPS products for construction will become waste between 2035 and 2085.

Scenario 2:

The second scenario is based on the assumption that the HBCD containing EPS waste will be treated together with the entire plastic fraction of the C&D waste (see Figure 6-101).

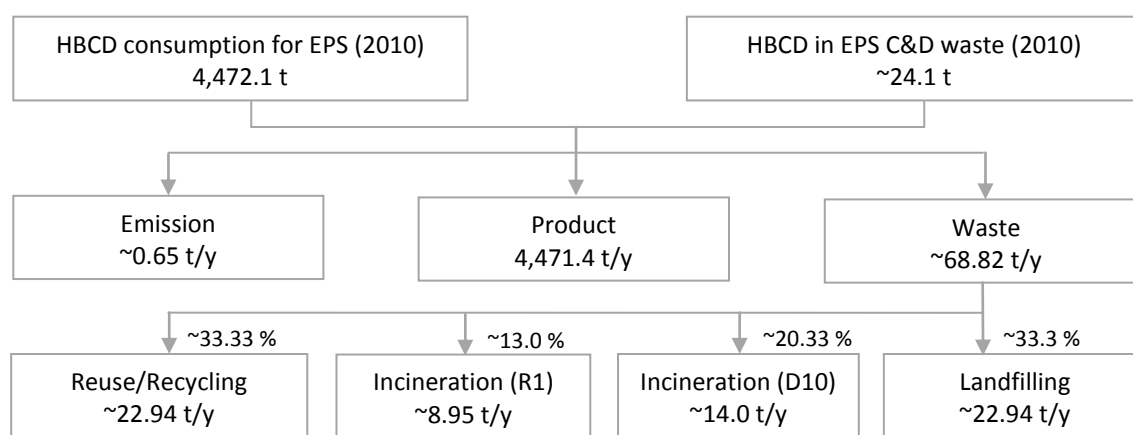


Figure 6-99: HBCD substance flow EPS products for construction (Scenario 2: HBCD containing EPS waste not separated but treated with the plastic fraction of C&D waste)

As shown in Figure 6-99 approximately 69 tonnes of HBCD in EPS waste from C&D have been landfilled (~23 tonnes), incinerated with (~9 tonnes)/without energy recovery (~14 tonnes) and reused/recycled (~23 tonnes). The main share of the approximately 4,500 tonnes of HBCD incorporated in new EPS products for construction will become waste between 2035 and 2085.

H) Waste flow for HBCD

In their pure forms, EPS and XPS foams are easily flammable or ignitable. In many European countries national building regulations therefore require EPS/ XPS foams to meet obligatory fire safety levels to ensure a high protection of homes, public and industrial buildings. Consequently, approximately 80% of all EPS boards and practically all XPS boards are processed to meet these national fire safety regulations [BSEF 2007]. As mentioned above, significant regional differences occur.

In 2005 in Europe, around 800,000 tonnes of EPS/XPS foams have been applied for energy saving applications in roofs, walls and floors of private and public buildings [BSEF 2007]. According to the quantities consumed in 2005 for EPS and XPS for construction (see Table 6-70) and considering an average content of 0.51 % for EPS products and 2 % for XPS products, the total quantity of EPS and XPS products containing HBCD used for construction amounts to around 1,134,000 tonnes (~ 885,500 tonnes for EPS and ~ 248,600 tonnes for XPS products).

As shown in Figure 6-100 approximately 12,400 t of EPS containing HBCD have been disposed of in the EU in 2010. Two different waste treatment scenarios have been calculated for the HBCD containing EPS in the EU.

Scenario 1:

In the first scenario (see Figure 6-100) it is assumed that the HBCD containing EPS is separated from the non HBCD containing C&D fraction and treated separately. It should be noted, that in practice it will usually not be possible to separate 100% of the HBCD containing EPS or XPS during deconstruction because the insulation boards are often glued and otherwise fixed to other material.

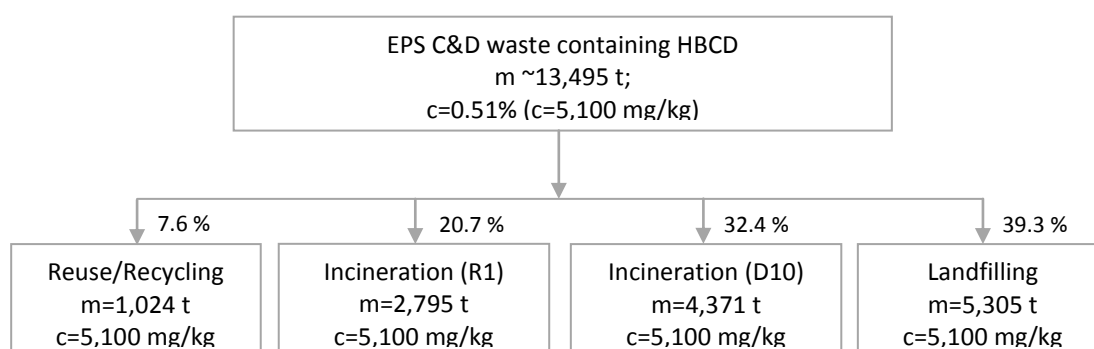


Figure 6-100: EPS waste flow (Scenario 1: HBCD containing EPS waste separated)

In this scenario, approximately 13,500 t of HBCD containing EPS from the C&D sector have been treated in the EU in 2010, with a comparatively high content of 5,100 mg/kg HBCD. The waste has been sent to landfills, incinerated with/without energy recovery (R1/D10) or reused/recycled.

Scenario 2:

The second scenario is based on the assumption that approximately 13,500 t HBCD containing EPS waste will be treated together with the whole plastic fraction of the C&D waste (see Figure 6-101). The total amount of C&D waste to be treated in the EU27 amounts to approximately 970,000,000 t (2006 data). Information on the plastic content of C&D waste ranges between 0.1 and 2.0 % [Bio 2010]. An average share of 1 % is used as best estimate for the calculation of Scenario 2. According to this estimate the total amount of the plastic fraction amounted to approximately 9,700,000 tonnes in 2006. Due to the significantly higher amount of the waste to be treated, the content figures of HBCD in this waste fraction decline to approximately 7.1 mg/kg (~7.1 ppm).

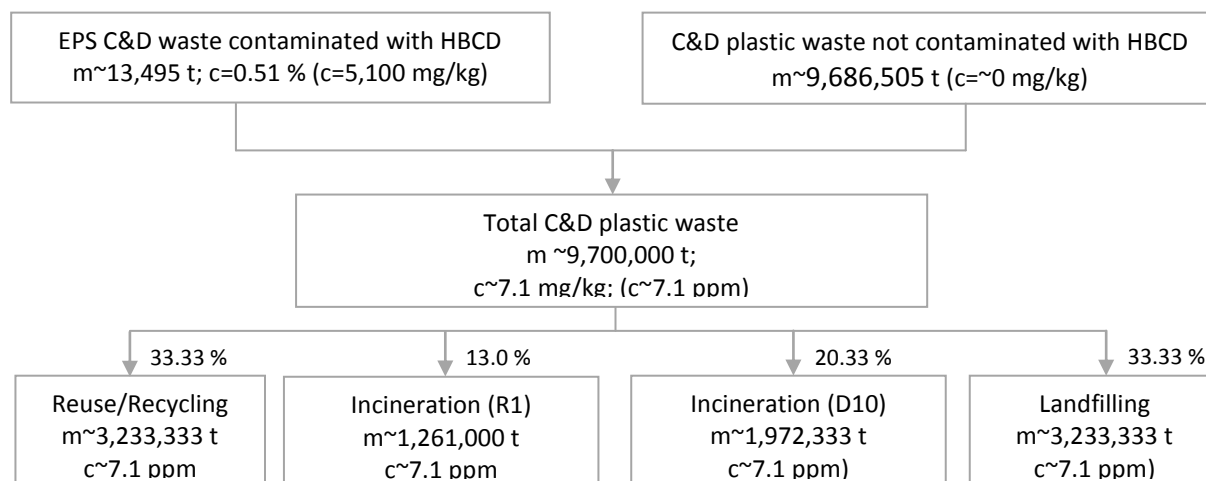


Figure 6-101: EPS waste flow (Scenario 2: HBCD containing EPS waste not separated but treated with the plastic fraction of C&D waste)

This waste will be finally disposed, incinerated (R1/D10) and reused/recycled. However, quantitative data on the relative shares is not available. For the waste flow it has therefore been assumed, that the distribution between the three relevant waste treatment options is equal, i.e. 1/3 goes to landfill, 1/3 to

incineration (61 % thereof without and 39 % with energy recovery) and 1/3 is reused/recycled.

Consequently, around 3,233,000 tonnes of EPS C&D waste with an average HBCD content of 7.1 ppm have been landfilled and the same amount reused/recycled. Approximately 1,261,000 tonnes (7.1 ppm) have been incinerated with energy recovery and around 1,972,000 tonnes (7.1 ppm) without energy recovery in 2010.

6.6.9 Extruded Polystyrene (XPS) used for construction

A) Background

Extruded polystyrene (XPS) is a plastic foam based on polystyrene that is formed by adding gas during extrusion.

There are approximately 35 sites in the EU15 area manufacturing flame retarded XPS containing HBCD. There are also 3 sites in Eastern Europe and 4 sites in Turkey [ECB 2008].

The use of XPS is as follows: Cold bridge insulation 10 %; Sandwich Panels and Laminates 4 %; Cavity Insulation 17 %; Floors 10 %; Basement Walls and Foundations 30 %; Inverted Roofs 18 %; Ceilings 5 % and Miscellaneous 6 % [ECB 2008].

More information which is common for both, EPS and XPS foams used for construction, is given in section A) Background of chapter 6.6.8.

B) Process input (raw material) and output (waste, air, water, product)

The manufacture of XPS materials is carried out in the following way (according to [ECB 2008]):

- The polystyrene, the additives such as processing aids, flame retardant, dye and blowing agent are fed continuously to an extruder.
- The polymer is melted; the blowing agent is mixed with the melted polymer and a “foamable gel” is formed.
- The gel is then cooled before it exits through an orifice called a die, where the blowing agent volatilises, causing the plastic to assume a foam structure. The blowing agent is usually a volatile, chemically stable compound, and by its introduction into the molten polymer, it reduces the density of the product by the formation of a myriad of closed cells within its structure.
- The foam is then trimmed to desired shape. The boards are packed into shrinkwrapped bundles and palletted. The pallets are stored for curing (usually 1 week) and are then ready for shipment.
- A remainder of about 25 % of the material is recycled to the extruder. This material is mainly “skin” from the surface, with higher density. The recycled material comes in contact with water in some cases.

In the XPS process HBCD can be used directly or as a HBCD master batch with Polystyrene. For the production of a HBCD master batch the HBCD is supplied either in powder or in low-dust granulated form in either 25 kg bags or in 1 ton super sacks or “big bags”. The super sacks are emptied into hoppers designed to minimize dust emissions. The HBCD is then carried to the point of mixing with screw or air driven metering equipment. The compounded polystyrene is extruded and cut into granules, and

packaged. The extrudate is either air-cooled or cooled by running in a water bath. According to industry information (personal communication) the master batch can contain approx. 40 % (w/w) of HBCD. ([ECB 2008] and [PS foam 2011])

One technology usually known as the UCI technology uses a vacuum in addition to blowing agents to produce lighter (lower density) foams. In this technology, the product comes into contact with water in a water pond directly after the extrusion. [ECB 2008]

Figure 6-102 indicates the relevant inputs and outputs associated with the manufacturing of HBCD flame retarded XPS products.

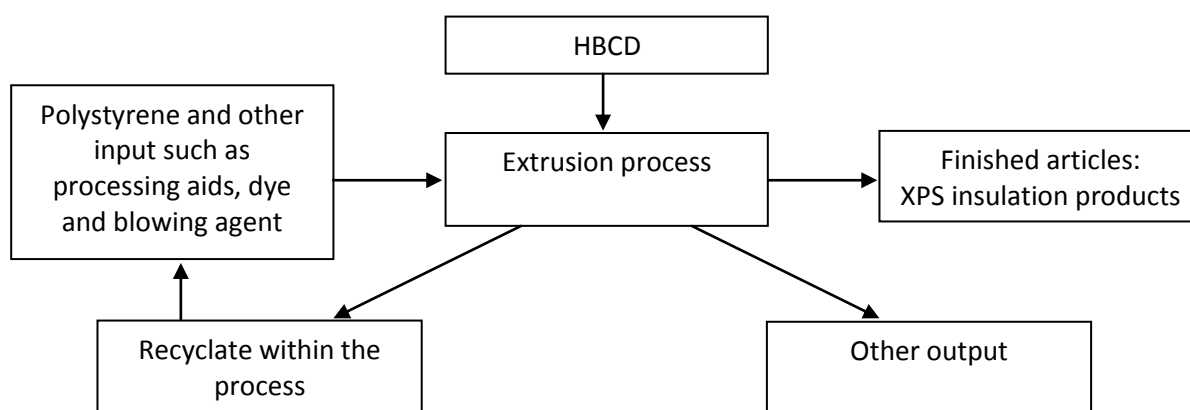


Figure 6-102: Relevant process inputs/outputs and products in XPS manufacturing

Input:

- Major input of concern is the input of HBCD as flame retardant

Outputs:

- Finished articles: HBCD containing XPS insulation products for the above mentioned uses.
- About 25 % of the input material is recycled to the extruder. This material is mainly “skin” from the surface, with higher density.
- During the use of HBCD in formulations for XPS products approximately 0.000117 % of the consumed HBCD are emitted to air, 0.000725 % are emitted to waste water and 0.000086 % are emitted to surface water (total release = 0.000928 %; calculated on the basis of [Swerea 2010]).

C) Generation factors for product/waste and other output (air, water)

Polystyrene insulation foams are the major use of HBCD but correspondingly have low emissions in production and during service life. The generation factor for EPS and XPS is $1 - 0.0001389 = 0.99986$ as the releases from use of HBCD in formulations, professional uses and releases during service life sum up to 0.01389 % of the HBCD consumption (calculated on the basis of [Swerea 2010]). This means that an XPS product that becomes waste at the end of its service life contains approximately 99.986% of the HBCD which was consumed for its manufacturing.

Based on the average content of 2 % HBCD in XPS foams, it can be concluded that with 1 tonne of HBCD approximately 50 tonnes of flame retarded XPS products can be manufactured.

Analogue to the situation for EPS, cut offs are estimated at 2%. 1% is considered to be recycled at the production site and 1% is considered to become waste.

A certain share of XPS product will remain in the environment e.g. if insulation of basements or insulation against frost heaves of road and railway embankments are not completely removed during deconstruction. Quantification is not available and quantities possibly left in the environment are not taken into account in the calculation of the mass flows.

D) Contamination data for output (and input)

Data for XPS products range between 1 and 3 % HBCD content (see Table 6-74). For the calculation of the waste flows an average content of 2 % is applied for XPS products.

E) Activity data

For XPS products the same considerations as for EPS products are valid (see chapter 6.6.8, section E).

For the calculation of mass flows a lifetime of 50 years, with a standard deviation of 25 years is applied for both EPS and XPS products for the construction sector.

Table 6-70 shows the estimated consumption in tonnes of HBCD in the EU for the relevant EPS and XPS applications. This estimation is taken as a basis for the calculation of the substance and waste flows.

F) Waste treatment

The situation for XPS waste is similar to the situation of EPS waste (see chapter 6.6.8, section F on waste treatment). HBCD containing waste is generated at each life cycle step. According to ECB (2008a), the so called pre-consumer waste can be recycled into the process. The HBCD containing end products which become waste (post-consumer waste) are either incinerated, put on landfill, left in the environment or recycled [ECB 2008].

It is not known to what extent end-products containing HBCD are landfilled, incinerated, left in the environment or recycled. Waste ending up in the municipal waste streams is likely to be put on landfill or incinerated. In the particular case of construction materials, waste could possibly also be left in the environment or used for road construction purposes or other purposes such as filling material. Another possible option is the recycling of larger pieces of EPS boards, where the EPS boards are grinded and returned to the moulding process together with virgin EPS to form new boards. This has been reported to occur in a number of European countries [ECB 2008].

According to the PS foam industry, recycling is not an obvious route for XPS products. This is because construction products have a very long lifetime. As there is no packaging application for XPS (short life) and recycling would not be the first choice even though XPS is, in principle, recyclable. XPS boards,

generated from construction and demolition waste, which remain glued and fixed after use are not readily recycled due to the intensive effort to reclaim the boards and potential process impact. At this time this is therefore not seen as cost effective. While a possible current route is landfill, a more effective route is via incineration with energy recovery. A study has shown that HBCD containing waste can be effectively destroyed in a MSWI (see [Vehlow and Mark 1995] and [Vehlow and Mark 1996]). [PS foam 2011]

In the future, an increasing amount of HBCD containing EPS and XPS waste will leave the use stage and enter the waste management sector. As a consequence, the amounts of HBCD incinerated, accumulating on landfills as well as the amounts left in the environment will increase. The share of HBCD flame retarded XPS was comparatively low in the past and the bulk of this waste will be disposed of in the future with an expected peak in the early 2050ies. Figure 6-103 shows the estimated annual quantity of HBCD in XPS waste from construction that will be disposed of until 2060 considering an average lifetime of 50 +/- 25 years of XPS insulation boards used for construction. The total estimated accumulated amount of HBCD to be disposed of by 2085 amounts to approximately 97,000 tonnes. Only around 128 tonnes thereof have already been disposed of by 2010. The remainder will have to be disposed of in the future.

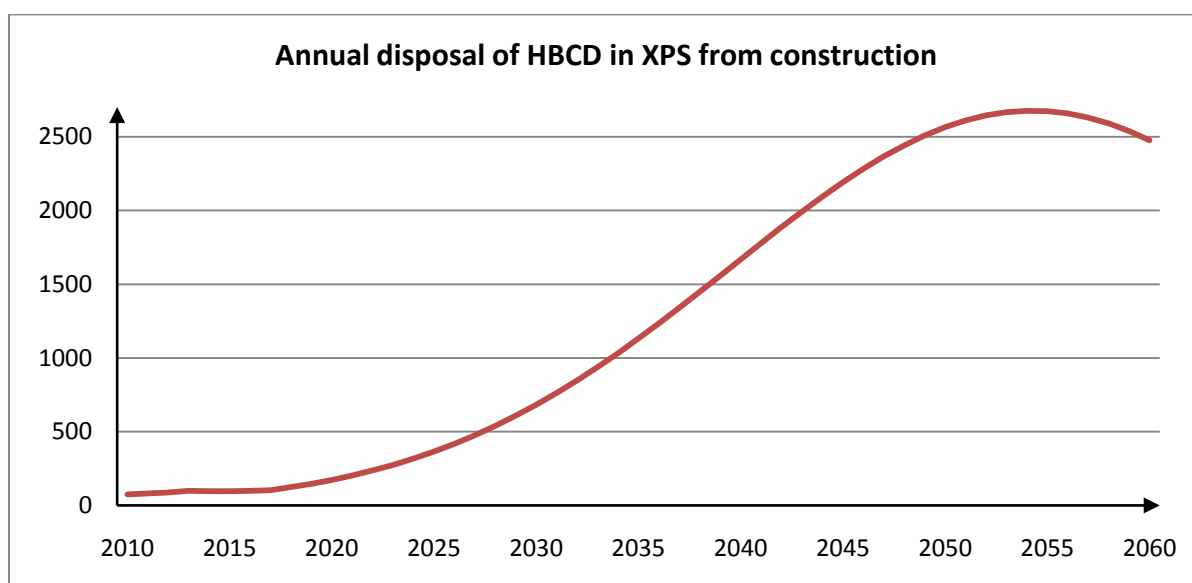


Figure 6-103: Estimated annual quantity of HBCD in XPS waste from construction that will be disposed of until 2060 (calculation based on the consumption figures provided in Table 6-70; lifetime 50 +/- 25 years; 1% cut offs becoming waste at the construction site taken into account)

The waste treatment can be assumed similar for EPS and XPS products for the construction sector. The properties of both, EPS and XPS insulation foams enable their recycling and recovery. Recovered foams can be reprocessed as they match the properties of virgin material EPS/XPS foams. Additionally, EPS and XPS waste represents a valuable source of energy in solid waste recovery plants fully equipped to be compliant with air emissions requirements [BSEF 2007].

Quantitative data on current XPS waste disposal routes for the EU is not available. For the waste flow it has been assumed, that the distribution between the relevant waste treatment options is similar to the

situation of EPS foams. However, according to the PS foam industry, there is currently no recycling of XPS from construction. Therefore for the mass flow calculations the following shares are applied for XPS from construction: Landfill 41.8 %, R1 23.2 %, D10 34.9 % and recycling 0 %.

G) HBCD substance flow

As shown in Figure 6-104 approximately 75 t/y of XPS containing HBCD have been disposed of in the EU in 2010. This quantity is comparatively low because the bulk of HBCD treated XPS has not entered the waste management stage yet. As already discussed, most of the HBCD containing XPS waste will be disposed of in the future, with the expected peak around the early 2050ies (more than 2,400 tonnes will have to be disposed of annually; see Figure 6-103).

In Figure 6-104 approximately 0.7 tonnes of HBCD are indicated as emission to the environment. This quantity is related to the consumption of approximately 4,900 tonnes of HBCD for the production of new XPS products for construction in 2010. These products will be used in the future and the main share of these products will become waste within the next 25 to 75 years i.e. between 2035 and 2085. The production and use of new HBCD treated XPS foams/boards in the construction sector is expected to decline, resulting in reduced emissions of HBCD to the environment from production and use of XPS products (release for EPS and XPS products = 0.01389 % of the HBCD consumption).

Scenario 1:

In the first scenario (see Figure 6-104) it is assumed that the HBCD containing XPS is separated from the non HBCD containing C&D fractions and treated separately.

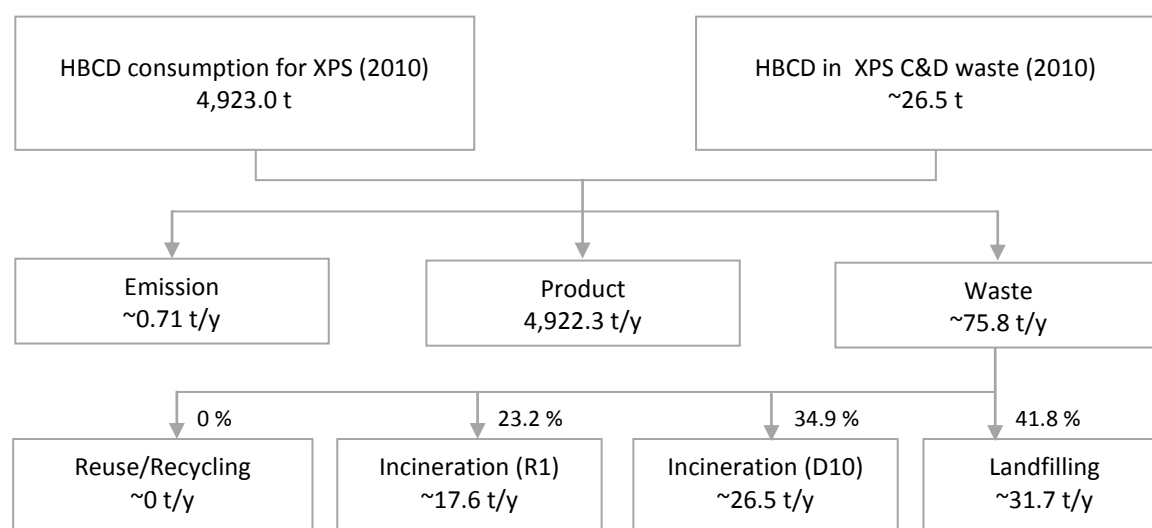


Figure 6-104: HBCD substance flow XPS products for construction (Scenario 1: HBCD containing XPS waste separated)

The approximately 75 tonnes of HBCD in XPS waste from C&D are landfilled (~32 tonnes), incinerated with (~18 tonnes) and without energy recovery (~26 tonnes). Quantitative data on current XPS waste disposal routes for the EU is not available. For the waste flow it has been assumed, that the distribution between the relevant waste treatment options is similar to EPS and taking into account that XPS is

currently not recycled. The main share of the 4,900 tonnes incorporated into new XPS products for construction in 2010 will become waste between 2035 and 2085.

Scenario 2:

The second scenario is based on the assumption that the HBCD containing XPS will be treated together with the entire plastic fraction of the C&D waste (see Figure 6-105).

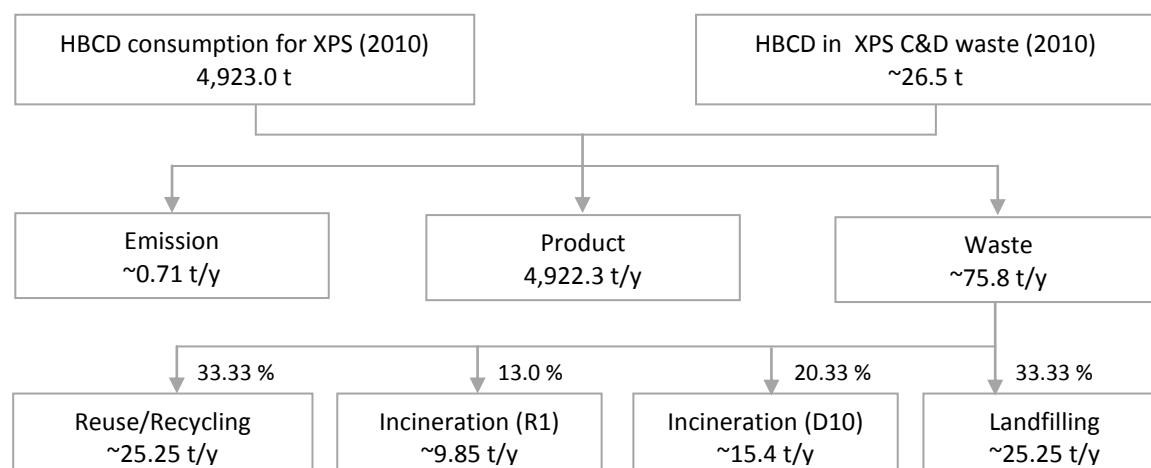


Figure 6-105: HBCD substance flow XPS products for construction (Scenario 2: HBCD containing XPS waste not separated but treated with the plastic fraction from C&D waste)

The approximately 75 tonnes of HBCD in XPS waste from C&D are landfilled (~25 tonnes), incinerated with (~9.85 tonnes) and without energy recovery (~15 tonnes).

H) Waste flow for HBCD

As already mentioned, practically all XPS foams/boards are processed to meet national fire safety regulations. In Europe in 2005, around 800,000 tonnes of EPS/XPS foams have been applied for energy saving applications in roofs, walls and floors of private and public buildings [BSEF 2007]. According to the quantities consumed in 2005 for EPS and XPS for construction and considering an average content of 0.51 % for EPS and 2 % for XPS products, the total quantity of EPS and XPS flame retarded products used for construction amounts to around 1,134,000 tonnes (~ 885,000 tonnes for EPS and ~ 249,000 tonnes for XPS products).

As shown in Figure 6-106 approximately 3,800 tonnes of XPS containing HBCD have been disposed of in the EU in 2010. Two different waste treatment scenarios have been developed for the HBCD containing XPS.

Scenario 1:

In the first scenario (see Figure 6-106) it is assumed that the HBCD containing XPS is separated from the non HBCD containing C&D fractions and treated separately. In this scenario, approximately 3,800 t of HBCD containing XPS from the C&D sector have been treated in the EU in 2010, with a comparatively high content of 20,000 mg/kg HBCD.

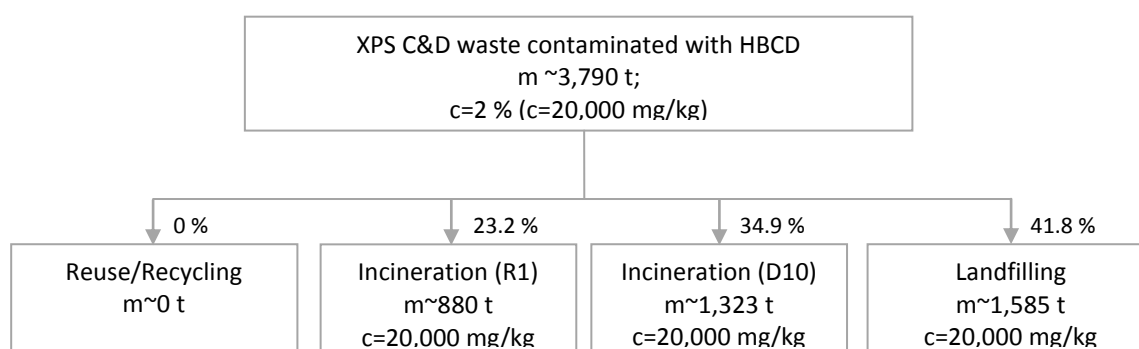


Figure 6-106: HBCD treated XPS waste flow (Scenario 1: XPS waste separated)

This waste has been sent to landfills (~1,600 tonnes), incinerated with (~900 tonnes) or without energy recovery (~1,300 tonnes). However, quantitative data on current XPS waste disposal routes for the EU is not available. For the waste flow it has been assumed, that the distribution between the three relevant waste treatment options is similar to EPS and taking into account that XPS is currently not recycled.

Scenario 2:

The second scenario is based on the assumption that the HBCD containing XPS will be treated together with the entire plastic fraction of the C&D waste (see Figure 6-107). The total amount of C&D waste in the EU27 is approximately 970,000,000 t (2006 data). Information on the plastic content of C&D waste ranges between 0.1 and 2.0 % [Bio 2010]. An average share of 1 % is used as best estimate for the calculation in Scenario 2. According to this estimate the total amount of the plastic fraction amounted to approximately 9,700,000 tonnes in 2006.

Due to the significantly higher quantity of the total waste to be treated, the content figures of HBCD decline to approximately 7.8 mg/kg (~7.8 ppm).

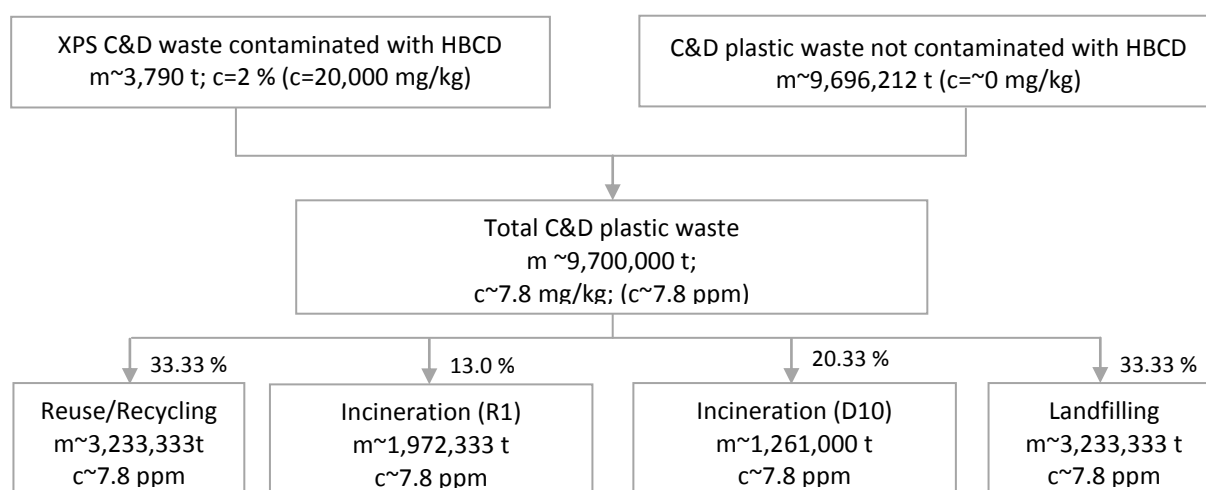


Figure 6-107: XPS waste flow (Scenario 2: XPS waste not separated)

This waste has been disposed, incinerated (R1/D10) and reused/recycled. However, quantitative data on current XPS waste disposal routes for the EU is not available. For the waste flow it has been assumed,

that the distribution between the three relevant waste treatment options is equal, i.e. 1/3 goes to landfill, 1/3 to incineration (61 % thereof without and 39 % with energy recovery) and 1/3 is reused/recycled. According to this scenario recycling of XPS waste would take place together with the recycled fractions of the construction and demolition waste.

Consequently, around 3,233,000 tonnes of XPS C&D waste with an average HBCD content of 7.8 ppm have been landfilled and approximately the same amount has been recycled. Approximately 1,261,000 tonnes have been incinerated without energy recovery and around 1,972,000 tonnes without energy recovery in 2010 (no impact on waste quantities).

Considering that the same waste streams receive also input from EPS from construction, the contamination values can be cumulated and can be considered in an aggregated waste flow with cumulated contamination values (7.1 ppm from EPS + 7.8 ppm from XPS = 14.9 ppm from EPS and XPS for construction).

6.6.10 Use of EPS and XPS outside the construction industry

A) Background

EPS and XPS are used for a wide variety of uses in other application areas than construction e.g. for transport vehicles as well as packaging material in general excluding food packaging. XPS end-products are used for the same applications as EPS end-products [UNEP/POPS/POPRC.6/10].

Predominantly EPS is used for other applications outside the construction sector [PS foam 2011].

EPS is used in many applications outside the construction industry, such as shape moulded products: industrial protective packaging (consumer electronics, appliances), agricultural applications (seed trays, soil aeration), food packaging (fish, meat, meals), insulated food containers (cups, trays), general protective packaging (building materials, furniture), flotation devices, child car seats, etc. [EUMEPS 2009]. Quantitative information on the relevance of specific uses other than for construction is not available.

B) Process input (raw material) and output (waste, air, water, product)

Figure 6-108 indicates the relevant inputs and outputs associated with the manufacturing of HBCD flame retarded EPS/XPS products.

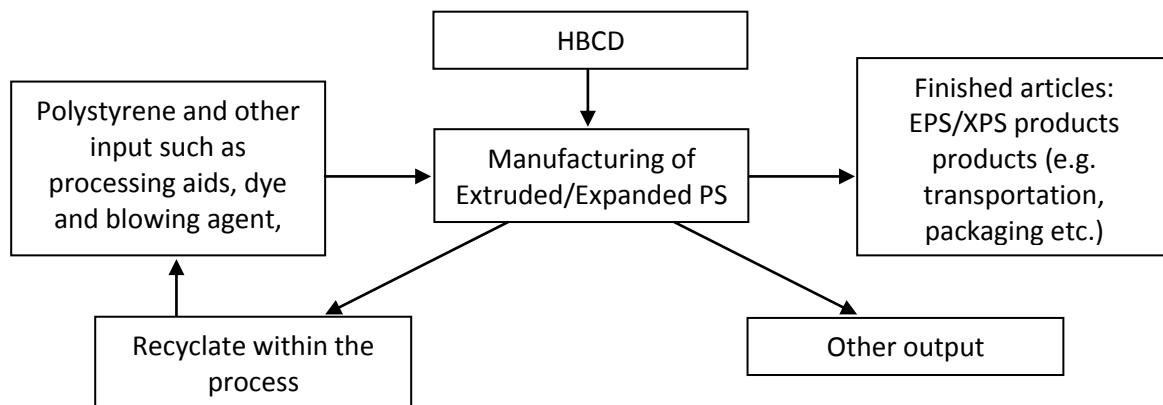


Figure 6-108: Relevant process inputs/outputs and products (EPS/XPS outside the construction sector)

Input:

- Major input of concern is the input of HBCD as flame retardant.

Outputs:

- Finished articles: HBCD containing EPS/XPS products for the above mentioned uses.
- Possibly some manufacturing waste can be recycled within the process.
- During the use of HBCD in formulations for EPS/XPS products approximately 0.000117 % of the consumed HBCD is emitted to air, 0.000725 % are emitted to waste water and 0.000086 % are emitted to surface water (total release = 0.000928 %; calculated on the basis of [Swerea 2010]).

C) Generation factors for product/waste and other output (air, water)

The generation factor for EPS/XPS is $1 - 0.0001389 = 0.99986$ as the releases from use of HBCD in formulations, professional uses and releases during service life sum up to 0.0001389 of the HBCD consumption (calculated on the basis of [Swerea 2010]). This means that an EPS/XPS product that becomes waste at the end of its service life contains approximately 99.986 % of the HBCD which was initially consumed for its manufacturing.

Based on the average content of 1.35 % HBCD in EPS/XPS products outside the construction sector, it can be concluded that with 1 tonne of HBCD approximately 74 tonnes of flame retarded EPS/XPS products can be manufactured. Losses due to cut off are not taken into account in the calculation of the mass flows.

D) Contamination data for output (and input)

For the calculation of the waste flows an average content of 1.35 % is applied for EPS/XPS products outside the construction sector.

E) Activity data

As no specific figures are available for the end-products for individual uses, a lifetime of 10 years with an equal distribution from 0 to 20 years is applied for HBCD containing EPS/XPS products in non-construction uses.

Table 6-70 shows the estimated consumption in tonnes of HBCD in the EU for the relevant EPS/XPS applications for non-construction uses. This estimation is taken as a basis for the calculation of the substance and waste flows.

According to this data, the consumption of EPS/XPS for non-construction uses amounted to 626 tonnes in 2009. According to information provided by industry the consumption of HBCD for EPS for packaging in 2009 amounted to approximately 163 tonnes³⁴. This means that about ¼ of the HBCD consumption for EPS/XPS non-construction uses was used in 2009 for packaging purposes.

F) Waste treatment

The situation for EPS/XPS waste outside the construction sector is different to the situation of EPS/XPS waste from the C&D sector. Due to the lack of specific data, related to waste treatment of EPS/XPS waste in the EU, it is assumed that most of this waste will enter the MSW stream or will be treated similar to MSW (see Table 5-1).

Taking into account that in some countries EPS/XPS waste is collected separately and landfilling for such waste is in several countries not permitted the share for incineration and/or recycling may be higher as for MSW.

³⁴ Calculated on the basis of the following data: virgin material used for EPS for packaging in 2009: 320 kt; thereof 10% flame retarded [PS foam 2011]; average HBCD content: 0.51 % HBCD

According to information provided by industry on the end of life fate of EPS for packaging the corresponding waste is currently landfilled (44.2%), incinerated (31.9%) and recycled (53.5%; recycling at conversion sites for EPS for construction and for PS extrusion and other recycling) [PS foam 2011]. Assuming that other EPS/XPS waste is treated similar if it is collected separately, the following distribution can be applied for the waste flow calculations: landfilling 44.2%, D10 19.5%, R1 12.4 %, recycling 53.5%.

Figure 6-109 shows the estimated annual quantity of HBCD in EPS/XPS waste from the non-construction sector that will be disposed of until 2040, considering an average lifetime of 0 to 20 years with an equal distribution. The total estimated accumulated amount of HBCD to be disposed of by around 2040 amounts to approximately 12,500 tonnes. Approximately 9,200 tonnes thereof have already been disposed of by 2010. The remainder (~3,300 tonnes) will have to be disposed of in the future, with the expected peak between 2025-2030 (more than 30 tonnes will have to be disposed of annually).

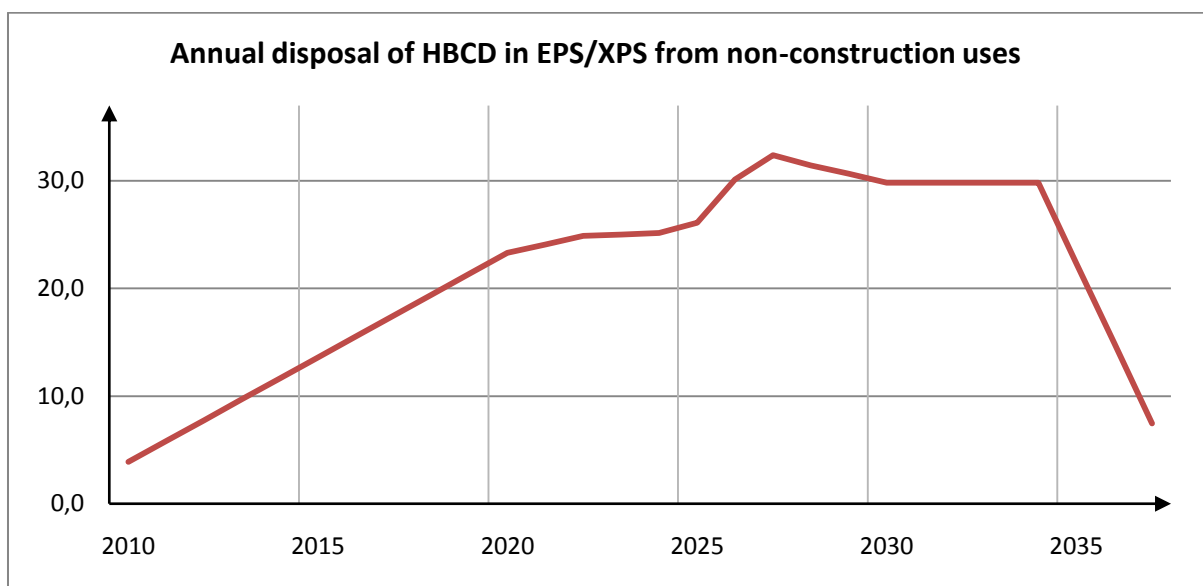


Figure 6-109: Estimated annual quantity of HBCD in EPS/XPS waste from non-construction uses that will be disposed of until around 2040 (calculation based on the consumption figures provided in Table 6-70; lifetime 0 - 20 years)

G) HBCD substance flow

As shown in Figure 6-110 approximately 3.9 t/y of HBCD in EPS/XPS waste from non-construction uses have been disposed of in the EU in 2010. This quantity is comparatively low because the bulk of HBCD treated EPS/XPS has not entered the waste management stage yet. Approximately 0.1 tonnes of HBCD are indicated as emission to the environment (see Figure 6-110). This quantity is related to the consumption of approximately 630 tonnes of HBCD for the production of new EPS/XPS products outside the construction sector in 2010. These products will be used in the future and the main share of these products will become waste within the next 20 years. The production and use of new HBCD treated EPS/XPS products is expected to decline, resulting in reduced emissions of HBCD to the environment from production and use of EPS products.

As specific data on the share of waste management operations is not available, two substance and waste flow scenarios have been developed.

Scenario 1:

In Scenario 1 it is assumed that EPS/XPS waste is collected and treated separately by landfill, incineration (R1/D10) or recycling/recovery (see Figure 6-110). Approximately 3.9 tonnes/year (2010) have been incinerated with/without energy recovery and recycled/recovered. As the exact share among these waste treatment operations is not available, the shares indicated for EPS in packaging are applied for the calculation.

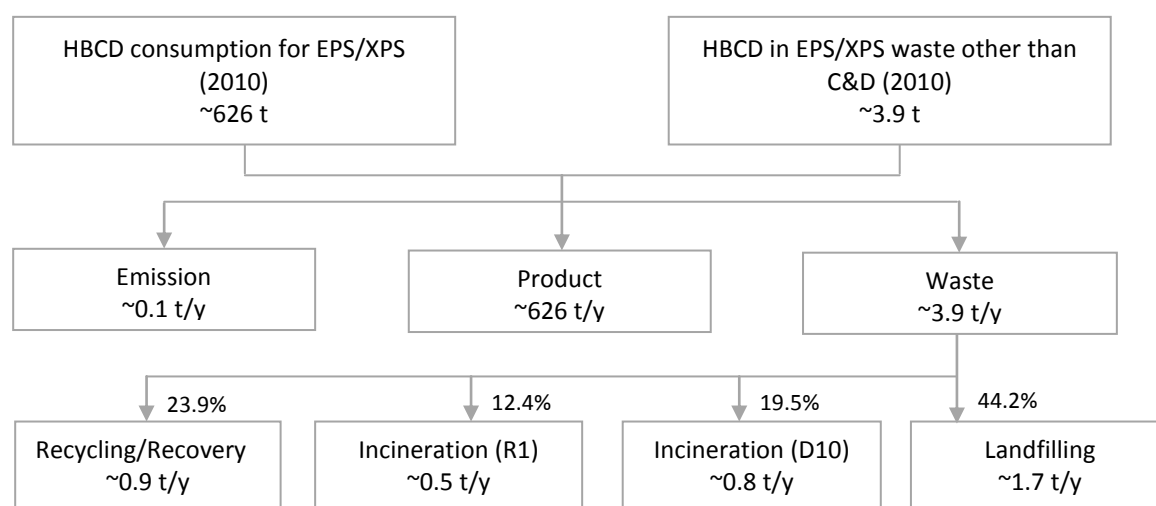


Figure 6-110: HBCD substance flow (Scenario 1: EPS/XPS products separated)

According to this scenario, around 1.7 t/y of HBCD from EPS/XPS have been landfilled. Approximately 0.8 tonnes of HBCD have been incinerated without energy recovery, 0.5 % with energy recovery and around 0.9 tonnes have been recycled/recovered in 2010.

Scenario 2:

In Scenario 2, the EPS/XPS waste is collected and treated together with MSW. Based on the available data for MSW treatment (see Table 5-1, no composting), approximately 1.9 t/y of HBCD have been landfilled, 0.6 t/y incinerated without energy recovery (D10), 0.4 t/y incinerated with energy recovery (R1) and 1.1 t/y treated in other operations (i.e. recycling/recovery operations).

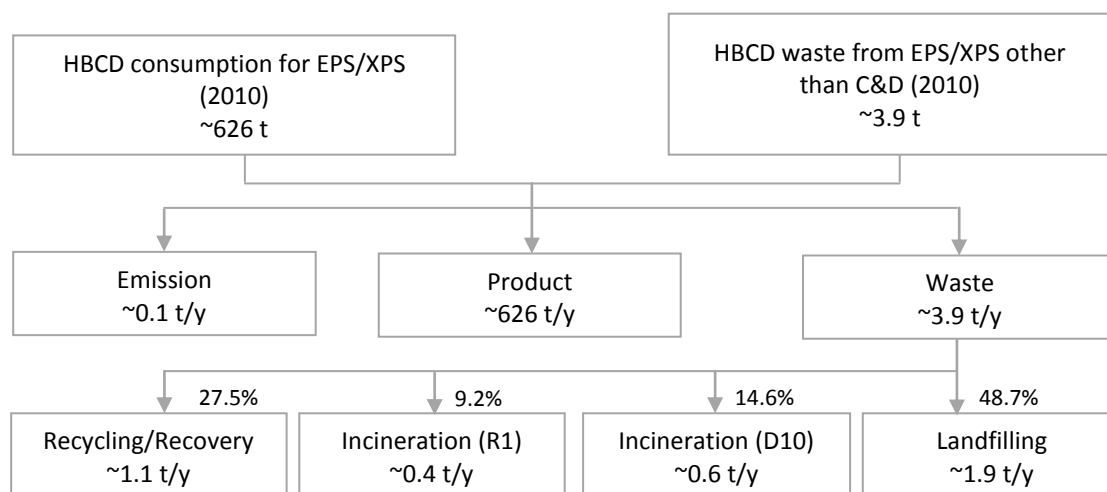


Figure 6-111: HBCD substance flow (Scenario 2: EPS/XPS products not separated)

The main share of the 626 t incorporated into new EPS/XPS products will become waste within the next 20 years in both scenarios.

H) Waste flow for HBCD

Based on the assumptions presented above the following two waste flows have been developed.

Scenario 1:

In the first scenario (see Figure 6-112) it is assumed that the HBCD containing EPS/XPS is collected and treated separately by landfilling, incineration or recycling/recovery. In this scenario, approximately 290 t of HBCD containing EPS/XPS waste (other than C&D) have been treated in the EU in 2010, with a comparatively high content of 13,500 mg/kg HBCD (13,500 ppm).

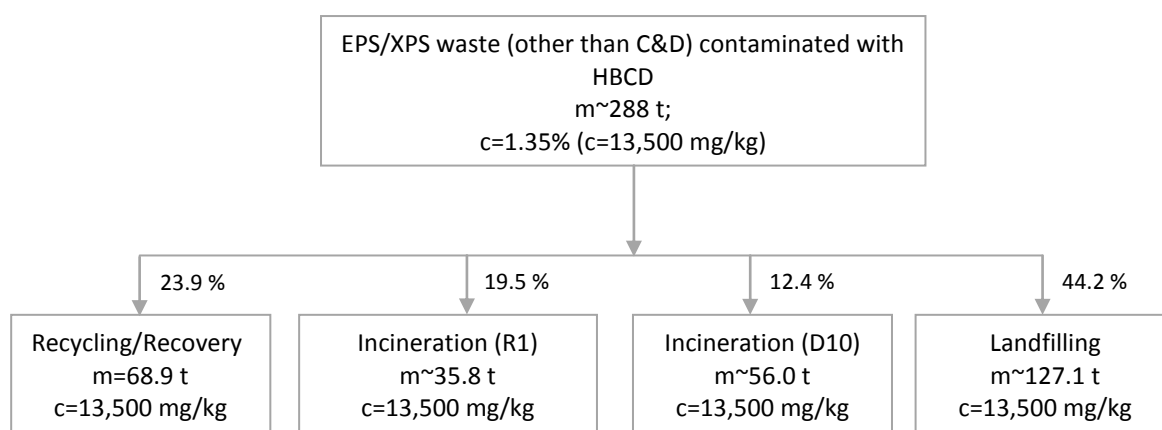


Figure 6-112: EPS/XPS other than C&D waste flow (Scenario 1: EPS/XPS collected and treated separately)

According to this scenario, 127 tonnes of HBCD containing waste has been landfilled, 56 tonnes of waste have been incinerated without energy recovery, 36 tonnes with energy recovery and 69 tonnes of waste has been recycled/recovered in 2010.

Scenario 2:

The second scenario is based on the assumption that the 288 t HBCD containing EPS/XPS will be treated together with the entire MSW in the EU (see Figure 6-113). Based on the available data for MSW treatment in the EU27 (255,371,000t treated, thereof 103,105,000 t landfilled, 50,351,000 t incinerated, 58,183,000 recycled, 43,552,000 t composted; EUROSTAT 2010) the following waste flow has been developed.

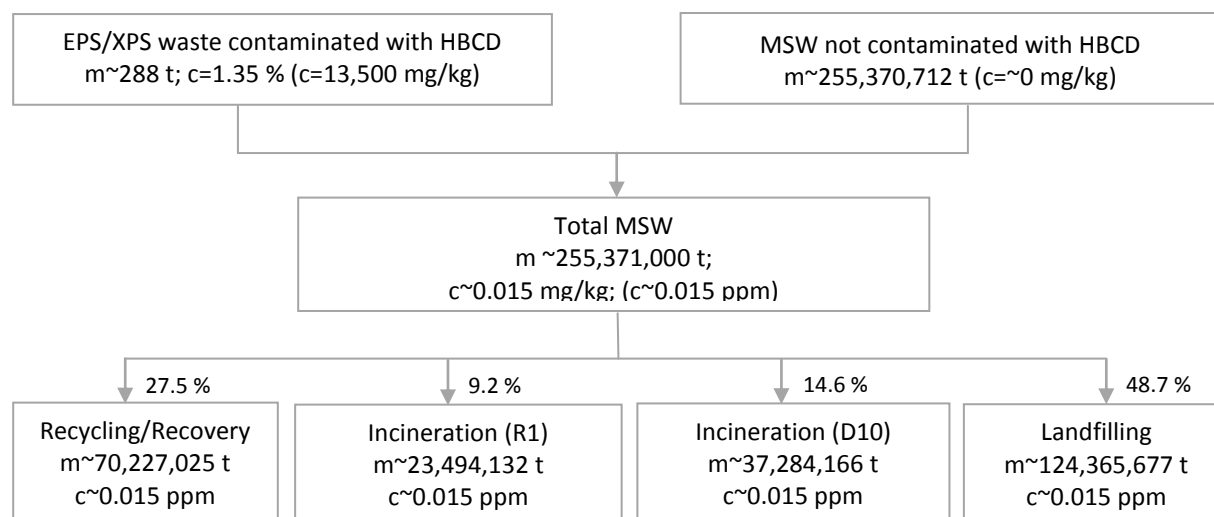


Figure 6-113: Waste flow EPS/XPS other than C&D (Scenario 2: EPS/XPS treated together with MSW)

Due to the significantly higher quantity of the total waste to be treated, the content figures of HBCD decline to approximately 0.015 mg/kg (~0.015 ppm).

This waste has been deposited in landfills (~124,366,000 t), about 37,284,000 t have been incinerated without energy recovery (D10) and another 23,494,000 t have been incinerated with energy recovery (R1). The remaining waste (~70,227,000 t) has been treated in other operations (i.e. recycling/recovery operations).

6.6.11 High Impact Polystyrene (HIPS)

A) Background

About 2 % of the total EU consumption of HBCD is for HIPS. HIPS end-products are applied in electrical/electronic parts, for instance in:

- audio visual equipment cabinets (video and stereo equipment)
- distribution boxes for electrical lines in the construction sector
- refrigerator lining

HIPS containing HBCD is likely to be imported to the EU in EEE, but no data on this has been identified [ECB 2008].

B) Process input (raw material) and output (waste, air, water, product)

Figure 6-114 indicates the relevant inputs and outputs associated with the manufacturing of HBCD flame retarded HIPS end-products.

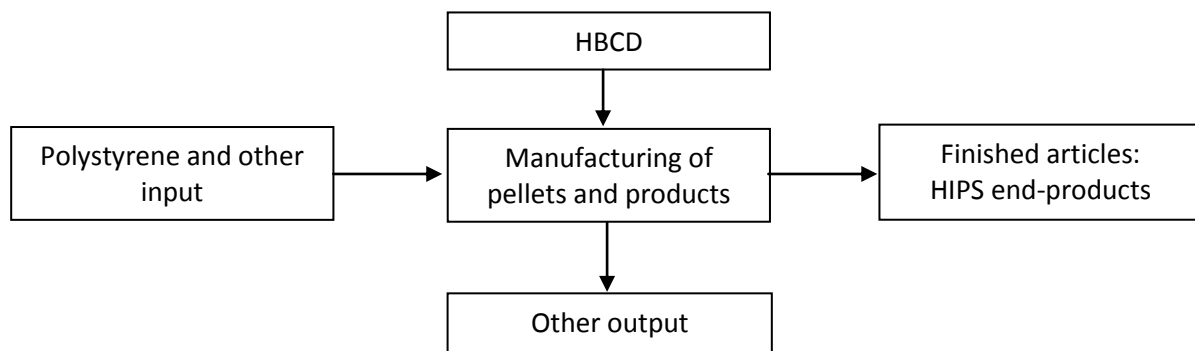


Figure 6-114: Relevant process inputs/outputs and products (HIPS)

Input:

- Major input of concern is the input of HBCD as flame retardant

Outputs:

- Finished articles: HBCD containing HIPS end products for the above mentioned uses.
- Manufacturing Waste (containing HBCD; HBCD free): Different wastes are generated during the manufacture of HBCD containing products. Possibly some of the waste can be recycled within the manufacturing process.
- Other output: During the use of HBCD in formulations for HIPS products approximately 0.000012 % of the consumed HBCD are emitted to air, 0.000029 % are emitted to waste water and 0.000127 % are emitted to surface water (total release = 0.000168 %; calculated on the basis of [Swerea 2010]).

HIPS are either produced in a batch or continuous polymerisation process. The final raw material is homogenised and extruded into HIPS pellets either strand- or face-out. These pellets are the starting

material for the production of flame-retarded HIPS. Different flame retardant additives are used of which HBCD constitutes only a small part. [ECB 2008]

The HBCD powder, delivered in plastic bags, is filled in intermediate storage containers from where the HBCD is transported to a weighing station. HBCD and other ingredients required for the particular HIPS formulation are weighed and transported further to the feeding hopper of the extrusion equipment. In the feeding hopper all ingredients, together with the HIPS pellets, are metered in the extruder for further mixing, homogenisation and granulation into pellets. [ECB 2008]

An alternative route for HIPS production is via an intermediate compounding route. First a master batch of general-purpose PS pellets and HBCD at a high concentration is prepared, followed by compounding this master batch with virgin HIPS material in a conversion step. The process of preparing the HBCD master batch is similar to that of the HIPS production but at a higher HBCD concentration. [ECB 2008]

After the molten mass at the end of the extruder is pressed through a plate with holes (die/plate), different granulation processes take place, for example [according to ECB 2008]:

- face cutting in air; a rotating knife after the plate cut the extruded “strands” into pellets cooled by air
- under water face cutting; a rotating knife in a water bath cuts the extruded strands in pellets cooled by water
- strand cutting; the molten strands are passed through a water bath to solidify and cool and are cut in a granulator

After the granulation process the HIPS pellets are dried and packed, either in bulk silos/containers or 25 kg bags, ready for conversion into HIPS products. The HBCD master batch process normally uses the strand-cutting route.

HIPS materials can be converted into HIPS products using various extrusion techniques and injection moulding. HIPS products can also be manufactured via a compounding route (i.e. mixing virgin HIPS raw material with a HBCD master batch during the extrusion or injection moulding process).

C) Generation factors for product/waste and other output (air, water)

The generation factor for HIPS is $1 - 0.0000028 = 0.99999$ as the releases from use of HBCD in formulations, professional uses and releases during service life sum up to 0.00028 % of the HBCD consumption (calculated on the basis of [Swerea 2010]). This means that a HIPS product that becomes waste at the end of its service life contains approximately 99.999% of the HBCD which was consumed for its manufacturing.

Based on the average content of 4 % HBCD in HIPS, it can be concluded that with 1 tonne of HBCD approximately 25 tonnes of flame retarded HIPS products can be manufactured.

D) Contamination data for output (and input)

Contamination data for HIPS products range between 1 and 7% HBCD content (see Table 6-74). For the calculation of the waste flows an average content of 4% is applied for HIPS products.

E) Activity data

HIPS containing HBCD is likely to be imported to the EU in EEE, however no data on this is available [ECB 2008].

For the calculation of mass flows a lifetime of 10 years with a standard deviation of 5 years has been applied for HIPS end-products.

Table 6-70 shows the estimated consumption in tonnes of HBCD in the EU for HIPS applications. This estimation is taken as a basis for the calculation of the substance and waste flows.

F) Waste treatment

At the end of their lifetime HIPS become specific parts of the waste regime according to their use in different products. Table 6-76 shows the waste categories and EWC numbers and description to which waste from HIPS could be allocated.

Table 6-76: Waste categories and EWC numbers and description for waste from HIPS end-products

Product	Waste category	EWC number	EWC description
audio visual equipment	WEEE category 4: consumer equipment	<ul style="list-style-type: none"> 20 01 35* 20 01 36 	<ul style="list-style-type: none"> discarded electrical and electronic equipment other than those mentioned in 20 01 21 and 20 01 23 containing hazardous components discarded electrical and electronic equipment other than those mentioned in 20 01 21, 20 01 23 and 20 01 35
refrigerator lining	WEEE category 1: large household appliances	<ul style="list-style-type: none"> 20 01 23* 20 01 35* 20 01 36 	<ul style="list-style-type: none"> discarded equipment containing chlorofluorocarbons discarded electrical and electronic equipment other than those mentioned in 20 01 21 and 20 01 23 containing hazardous components discarded electrical and electronic equipment other than those mentioned in 20 01 21, 20 01 23 and 20 01 35
distribution boxes for electrical lines in the construction sector	C&D waste	<ul style="list-style-type: none"> 17 17 02 03 17 02 04* 17 04 09* 17 04 10* 17 04 11 17 06 17 09 	<ul style="list-style-type: none"> Construction and demolition wastes Plastic Glass, plastic and wood containing or contaminated with dangerous substances Metal waste contaminated with dangerous substances Cables containing oil, coal tar and other dangerous substances Cables other than those mentioned in 17 04 10 Insulation materials and asbestos containing construction materials Other construction and demolition wastes

According to IOM 2008, articles containing HIPS treated with HBCD may be returned to the manufacturers under the requirements of the WEEE directive, which will increase the potential for recycling of the materials in these articles. Material which cannot be used in recycling will be disposed of to landfill or incineration. Other articles will be disposed of into MSW, and the eventual fate of this material will also be to landfill or incineration. The proportion which is disposed of by each route will depend on the MS in which this occurs. The risk assessment considers that incineration leads to only very low levels of products such as brominated dioxins, and that operated correctly it is a suitable method for the disposal of HBCD. [IOM 2008]

Based on this information it is assumed, that HIPS end-products are partly recycled (no quantitative data available) and partly disposed to landfill or incineration. As no quantitative data is available on the significance of the different waste treatment options, the shares as indicated in Table 5-1 are applied (waste treatment similar to MSW, no composting: landfilled 48.7 %, D10 14.6%, R1 9.2 %, recycled 27.5 %).

According to the WEEE directive, plastics containing brominated flame retardants have to be removed from any separately collected WEEE. HBCD is a brominated flame retardant. Consequently this obligation includes plastics containing HBCD and HIPS flame treated with HBCD have to be removed from separately collected WEEE.

Figure 6-115 shows the estimated annual quantity of HBCD in HIPS waste that will be disposed of until approximately 2027, considering an average lifetime of 10 years with a standard deviation of 5 years.

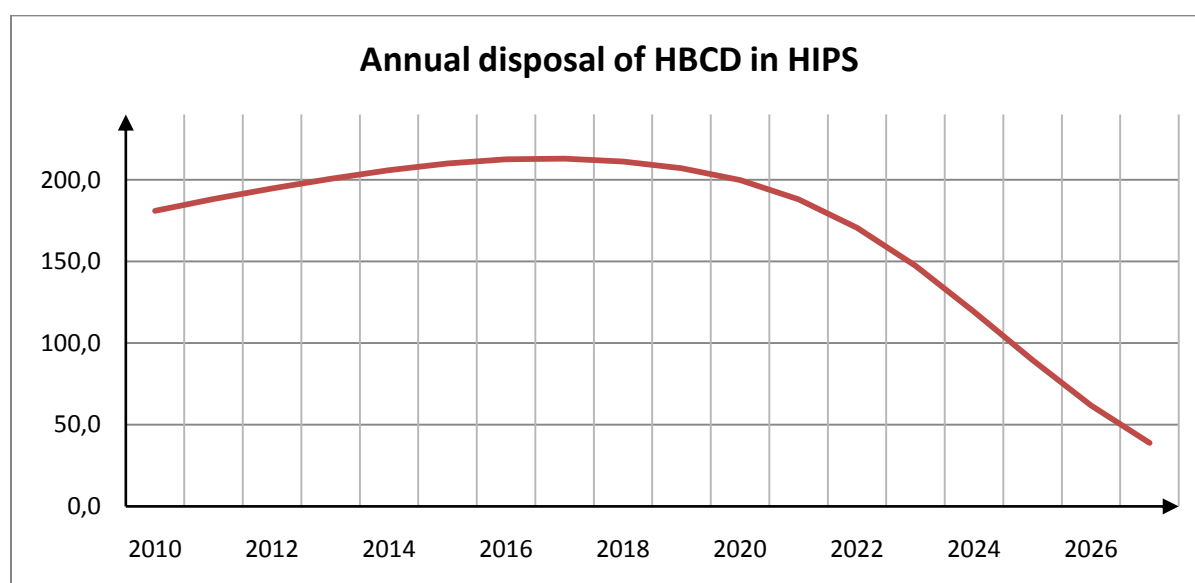


Figure 6-115: Estimated annual quantity of HBCD in HIPS waste that will be disposed of in the EU

G) HBCD substance flow

The HBCD input to the waste stream from HIPS end-products becoming waste in 2010 amounts to approximately 180 tonnes (see Figure 6-116). Approximately 0.0004 t/y are emitted to the environment.

Based on the available data within literature, the waste is recycled, incinerated or disposed of to landfills. As no specific quantitative data is available on the significance of the different waste treatment options, the shares as indicated in Table 5-1 (no composting: 48.7 % landfilled, D10 14.6%, R1 9.2 %, recycled 27.5 %) are applied.

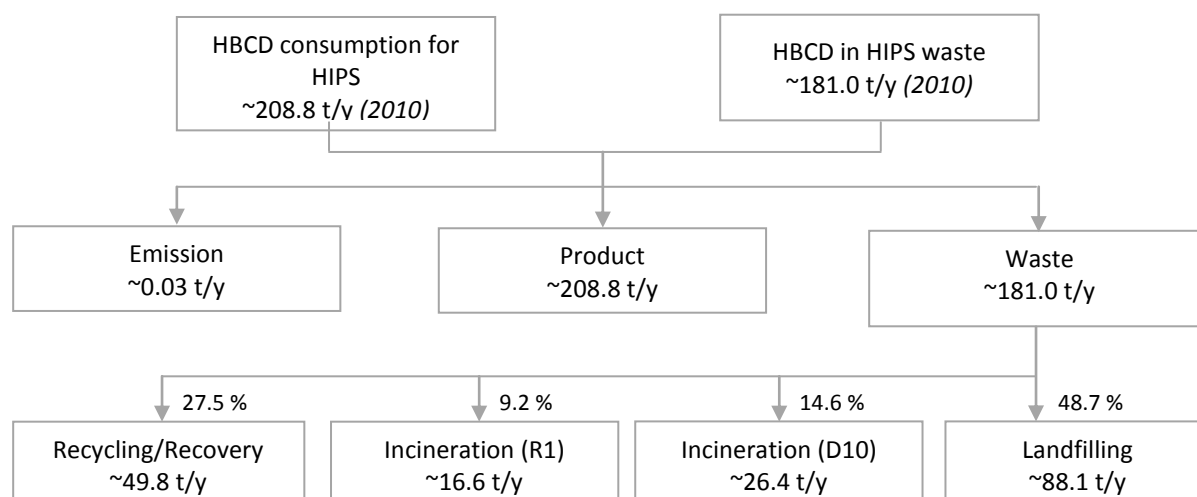


Figure 6-116: Substance flow (HBCD from HIPS)

As it can be seen in Figure 6-116, most of the HBCD containing HIPS has been landfilled in the EU (~88 t) in 2010. Approximately 26 tonnes have been incinerated without energy recovery and around 17 tonnes with energy recovery. The remaining 50 tonnes have been recycled/recovered.

H) Waste flow for HBCD

As shown in Figure 6-117 approximately 4,525 t of HBCD containing HIPS waste have been treated in the EU in 2010, with a comparatively high content of 40,000 mg/kg HBCD (40,000 ppm). Based on the assumptions presented above the following waste flow has been developed.

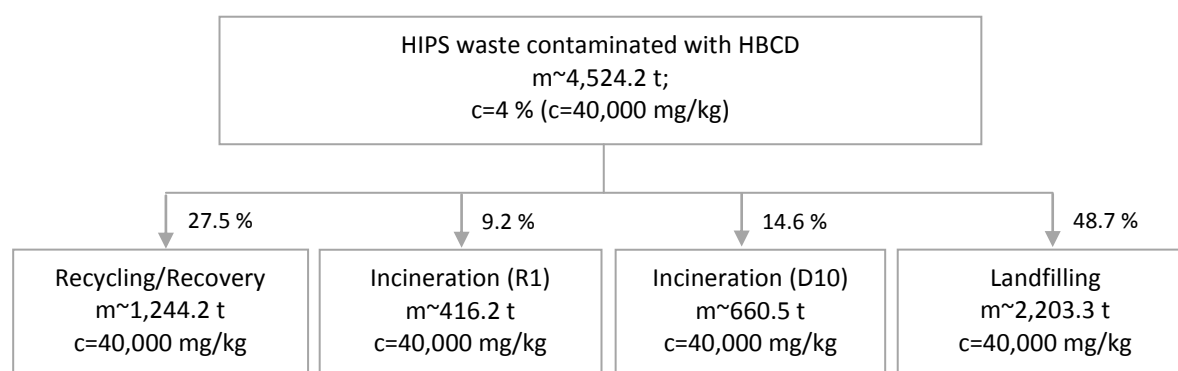


Figure 6-117: Waste flow (HIPS containing HBCD)

The waste has been disposed of to landfills (~2,200 t), about 660 tonnes have been incinerated without

energy recovery (D10) and another 420 tonnes have been incinerated with energy recovery (R1). The remaining HIPS waste (~1,240 t) has been treated in other operations (i.e. recycling/recovery operations).

6.6.12 Polymer dispersion for textiles

A) Background

Flame retardant systems are used in textile applications to comply with flame retardant standards [ECB 2008]. HBCD is formulated to polymer-based dispersions (e.g. acrylic or latex) of variable viscosity in the polymer industry. The dispersions are then processed in the textile finishing industry. The textiles with the back-coating containing HBCD are mainly used for flat and pile upholstered furniture (residential and commercial furniture), upholstery seating in transportation, draperies, and wall coverings, bed mattress ticking, interior textiles e.g. roller blinds and automobile interior textiles and car cushions [IOM 2008].

For the use of HBCD in textiles, alternative substances are commercially available. These are for instance chlorinated paraffins, decabromodiphenyl ether (DecaBDE), ammonium polyphosphates and reactive phosphorus constituents. [IOM 2008]

B) Process input (raw material) and output (waste, air, water, product)

Figure 6-118 indicates the main inputs and outputs associated with the production of articles containing HBCD treated textiles.

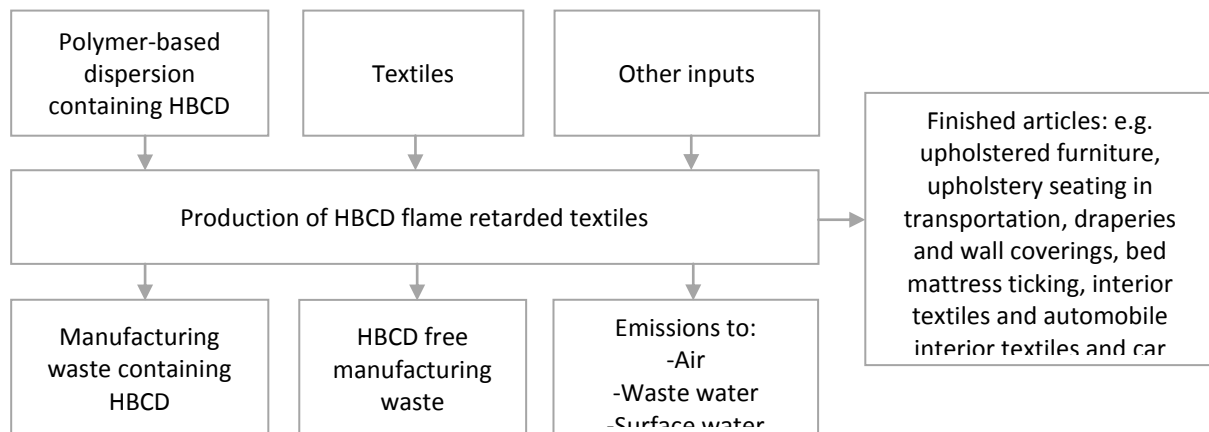


Figure 6-118: Relevant process inputs/outputs and products

Input:

- Major input of concern is the input of the polymer-based dispersion containing HBCD

Outputs:

- Finished articles: see Figure 6-118.
- Manufacturing Waste: Generated during the manufacture of HBCD containing products (e.g. cut off).
- Emissions to air, waste water and surface water: Approximately 0.000012 % of the consumed HBCD are emitted to air, 0.000380% are emitted to waste water and 0.000095 % are emitted to surface water (total release = 0.000487%; calculated on the basis of [Swerea 2010]).

HBCD is formulated to polymer-based dispersion (e.g. acrylic or latex) of variable viscosity in the polymer industry. The dispersions are further processed in the textile finishing industry.

The HBCD particles used for textile back-coating need to be very small. Therefore, micronising is performed as the initial step [Swerea 2010]. The water based dispersion used by the back-coaters; both paste as well as foam need to be stable (no precipitation and no viscosity change), and should not contain particles clogging the system. Too fine particles act as thickener, where too coarse material will lead to a non-stable dispersion (precipitation) and an applied coated film with a non acceptable rough surface [ECB 2008].

Textile formulators prepare flame retarded formulations which are water-based dispersions and can contain a binder system and HBCD as well as up to 20 other ingredients. These flame retarded formulations, mostly custom tailored, are supplied as dispersions to back-coaters. In general, formulation is carried out in an open batch system. HBCD is added to a dispersion containing water, a polymer (e.g. synthetic latex, acrylates or PVC), thickener and dispersion agent. Other BFRs such as DecaBDE can also be added to the preparation. In addition, synergists such as antimony trioxide and antimony pentoxide could be included in the final product. Water and solvents will leave the preparation when dried and concentrations of flame-retardants in the coating layer will be higher than in the preparation. Preparations with the highest concentration of HBCD are assumed to be diluted before use [ECB 2008].

Finally, the back coating can be applied to textile as paste, where a layer is 'glued' to the textile and a scratch knife defines the final thickness depending on the flame retardant standard, the textile used and the flame retardant concentration in the dispersion. The back coating can also be applied as foam, where a foam layer is pressed on the textile through a rotating screen. Once applied, the foam cells will break, resulting in a thin coating film. The coating is dried and fixated in an oven at temperatures between 140-180 °C.

C) Generation factors for product/waste and other output (air, water)

The generation factor for HBCD back-coated articles is $1 - 0.0001294 = 0.99987$ as the releases from use of HBCD in textiles, professional uses and releases during service life sum up to 0.01294 % of the current HBCD consumption (calculated on the basis of [Swerea 2010]). This means that a HBCD back-coated article that becomes waste at the end of its service life contains approximately 99.987% of the HBCD which was initially used for its manufacturing. The most relevant release occurs during the industrial use of polymer dispersions in the process of textile back coating (see [Swerea 2010]).

Based on an average content of 25 % of HBCD in back coatings, which corresponds to approximately 8 % HBCD in a back coated fabric, it can be concluded that with 1 tonne of HBCD around 12.5 tonnes of flame retarded textiles can be manufactured. Losses due to cut off are not taken into account in the calculation of the mass flow.

D) Contamination data for output (and input)

Textile back coating may contain approximately 25 % HBCD which corresponds to a load of 7 to 9 % on the back-coated fabric [Swerea 2010]. Therefore, for the calculation of the waste flow an average content of 8 % has been applied.

E) Activity data

Prior to the year 2004 the EU consumption of HBCD for textiles was approximately 10% of the total consumption. Since 2007 it represents about 2% of the total consumption. For the time from 2003 to 2007 a decrease from 10 to 2% took place. The current use of HBCD in polymer dispersions for textiles represents approximately 2 % of the total HBCD consumption in the EU (see Table 6-69).

According to [ECB 2008], import of polymer dispersions for textiles containing HBCD cannot be excluded, however, these are not possible to quantify. The same applies also in the case of export of HBCD from the EU [IOM 2008]. Variations in consumption occur among different MS. Most of the use of the flame retarded textiles is expected to be in those countries which have specific regulatory requirements for use of flame-retarded textiles, in particular the UK and Ireland [IOM 2008].

For the calculation of the mass flow a lifetime of 10 years with a standard deviation of 3 years has been applied.

F) Waste treatment

According to [IOM 2008], back-coated textiles represent a relatively low proportion of HBCD containing waste. It is understood that most of this waste goes to landfill or incineration [IOM 2008].

The largest part of textiles such as upholstered furniture, furniture covers, mattresses, curtains, carpets, flexible foam components and protective clothing treated with flame retardants are either applied in combination with furniture or material for interior decoration. At the end of life, these products are collected as bulky waste from building demolition (coated textiles for interior decoration), or by the municipal solid waste collection system (furniture upholstery and interior decoration). The textiles in the interior decoration of vehicles make part of the application area transport and go to ASR treatment plants. [Morf 2007]

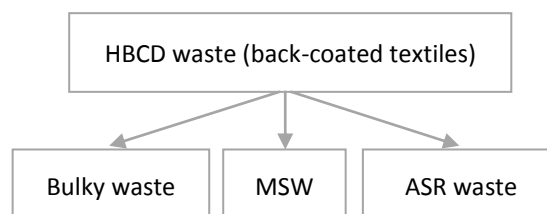


Figure 6-119: Possible routes of disposal for HBCD back-coated textiles

As noted above, most of the use of the treated materials is likely to be in the UK and Ireland. In these countries the main route of disposal for municipal solid waste is landfill. According to EUROSTAT DATA

FROM 2010 data the share of the waste treatment options for waste similar to MSW (no composting) can be calculated for Ireland and the UK to be 62.8% landfill, 10.5% incineration and 26.7% recycling in the year 2008.

Textile recycling activities that have started at the beginning of the 90's may concern protective clothing to some extent [Morf 2007]. As these applications are not of particular importance in this case, no recycling of textiles containing HBCD has been considered. Accordingly the remaining waste treatment options represent 85.7 % landfill and 14.3 % incineration in Ireland and the UK.

The accumulated HBCD quantity from 1988 to 2017 amounts to approximately 13,200 tonnes. Around 7,200 tonnes thereof have been disposed of by 2010. Around 6,000 tonnes will have to be disposed of in the future. Figure 6-120 shows the estimated annual quantity of HBCD in back-coated textiles that will be disposed of until approximately 2030, considering an average lifetime of 10 +/- 3 years of those articles. The disposal of HBCD amounts to approximately 870 tonnes in 2010.

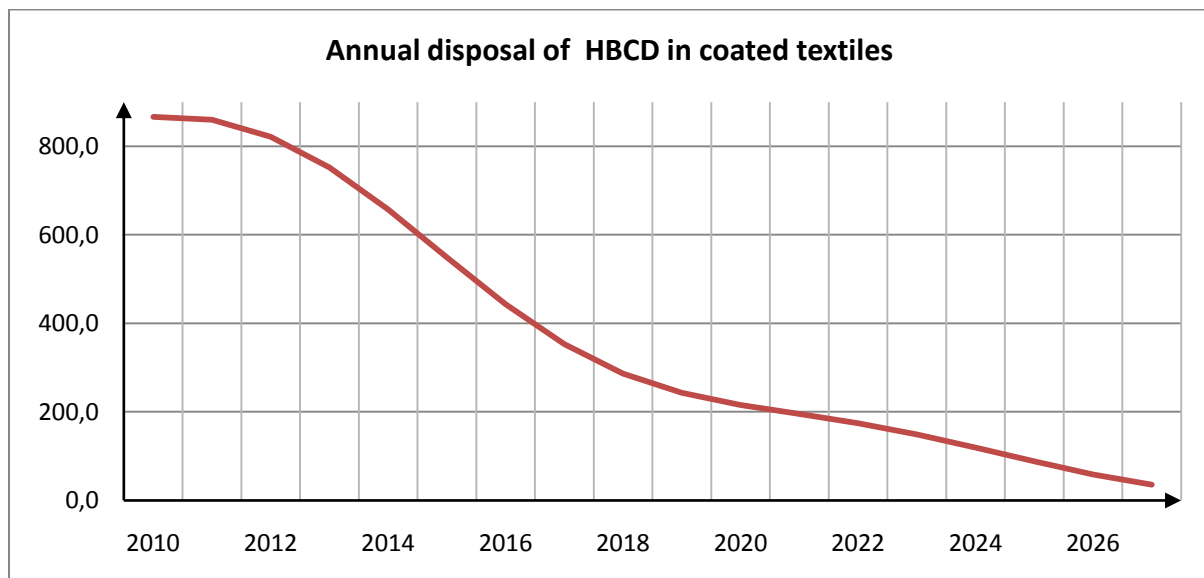


Figure 6-120: Estimated annual quantity of HBCD in back-coated textile waste that will be disposed of in the EU

G) HBCD substance flow

As shown in Figure 6-121 approximately 870 t of HBCD from flame retarded textiles have been disposed of in the EU in 2010. Most of the HBCD containing textiles will be disposed of in the near future, with an expected decline as illustrated in Figure 6-120.

Approximately 1.35 tonnes of HBCD are indicated as emission to the environment. This quantity is related to the consumption of approximately 210 tonnes of HBCD for the production of new back coated textiles in 2010 (see Figure 6-121). The production and use of new HBCD treated textiles is expected to decline, resulting in reduced emissions to the environment from production and use.

Specific data on the share among different waste management operations in the EU is not available.

However, due to the fact that most of the contaminated waste will be disposed of in the UK and Ireland as MSW, it can be assumed that a relatively high proportion of the HBCD treated textiles will be landfilled. Therefore, for the following substance flow it is assumed that 85.7 % of the HBCD containing waste will be landfilled. The remaining 14.3 % of the HBCD will be incinerated with and without energy recovery (61 % thereof without and 39 % with energy recovery).

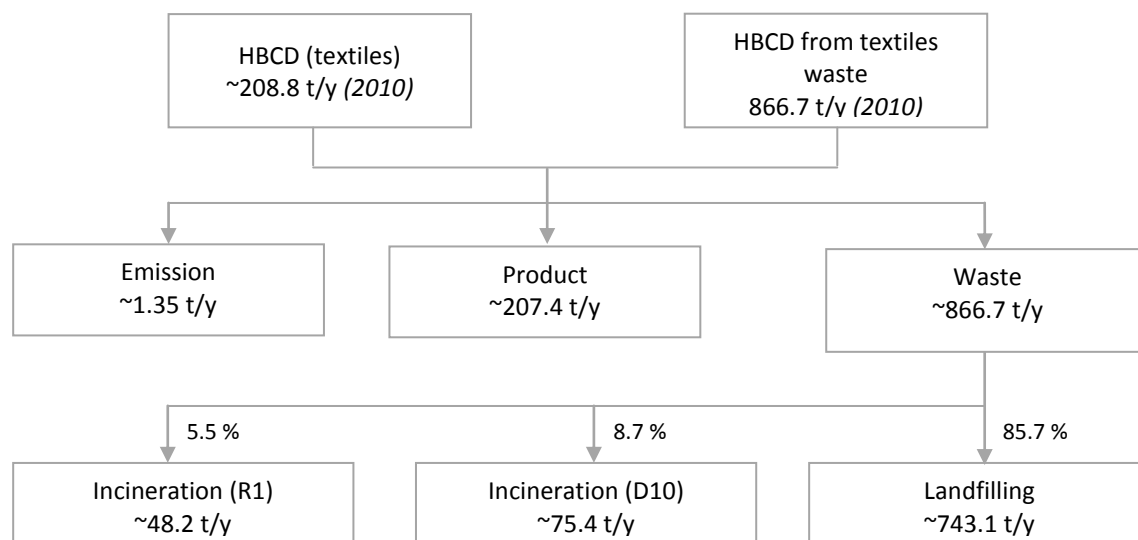


Figure 6-121: HBCD substance flow (back-coated textiles)

As shown in Figure 6-121, most of the HBCD from textiles has been landfilled in the EU (~740 t) in 2010. Approximately 75 tonnes have been incinerated without energy recovery (D10) and around 50 tonnes with energy recovery (R1). As discussed previously, no recycling of textiles has been considered.

H) Waste flow for HBCD

As indicated in Figure 6-122, approximately 11,000 tonnes HBCD containing textile waste, with an average HBCD content of 80,000 ppm, has been disposed of in the EU in 2010.

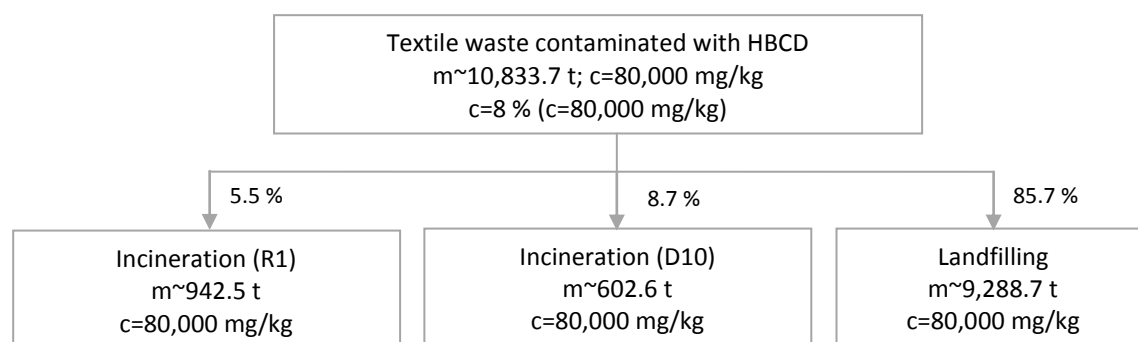


Figure 6-122: Waste flow (HBCD treated textiles)

Most of the HBCD containing waste has been landfilled (~9,300 t). Approximately 600 tonnes have been incinerated without energy recovery and around 940 tonnes with energy recovery (see Figure 6-122).

6.6.13 References HBCD

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6.7 PCP flows

6.7.1 Use and Production

PCP, NaPCP and PCPL have mainly been used for wood impregnation since the 1930's, but also in a variety of other applications as biocide, pesticide, disinfectant, defoliant, anti-sapstain agent and anti microbial agent. PCP has furthermore been used in the paper industry. NaPCP has been used for the same application as PCP whereas PCPL has been used in the textile industry including leather materials and natural fibre carpets ([UNECE 2010], [BfLU 2008]). The total worldwide amount produced in 2009 was indicated at 7,257 tonnes, but in Europe the production has stopped years ago [UNECE 2010]. The production of PCP and NaPCP ceased in many countries in the EU in 1992 and these substances were imported [IEP 2002].

In 1996 a total of 378 tonnes of NaPCP and 30 tonnes of PCP were imported to the EU. From 378 tonnes of imported NaPCP 126 tonnes were imported to France, 108 tonnes to Portugal and 144 tonnes to Spain. From the 30 tonnes PCP imported in 1996 to the EU 28 tonnes were synthesized to 46 tonnes PCPL in the UK. Of the 46 tonnes PCPL produced, 5-10 tonnes were exported to France and a similar quantity was used in the UK. The rest was exported to countries outside the EU 15. However, it could have been exported to the new EU 12 countries but no specific information is available regarding this possibility. Import of PCP and production of PCPL have slightly decreased before 2002 within the EU 15. According to the only producer of PCPL in Europe (UK) they have purchased no more than 20 tonnes of PCP in 1999 for conversion into PCPL and their production of PCPL for 2000 was less than 30 tonnes [IEP 2002]

Production of PCPL ceased in the European Union in 2000 since according to Commission Directive 1999/51/EC use of PCP and its compounds was not permitted as a synthesizing and/or processing agent in industrial processes from 1 September 2000 on [UNECE 2010]. This did not affect Spain, Portugal, France and the UK. In 2002 no PCPL imports took place [IEP 2002]. According to [OSPAR 2001] the only PCPL producing country within the EU before 2002 was the UK, but the use of PCPL was restricted in the UK in 2002 [PRTR 2010]. It is therefore assumed that at this time the production stopped in the EU 27.

6.7.2 Chemical Characteristics of PCP and its derivatives

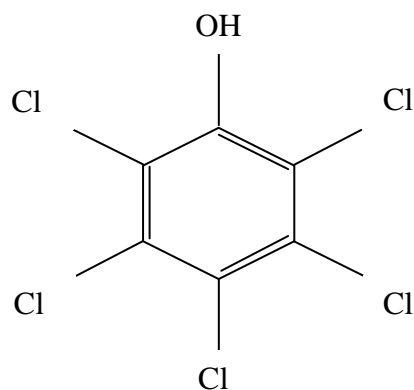
The salt sodium pentachlorophenate (Na-PCP) is used for similar purposes as PCP and readily degrades to PCP. The environmental toxicity, fate and behaviour profile of PCP and Na-PCP are quite similar. The same applies for the ester pentachlorophenyl laurate (PCPL) [UNECE 2010].

Molecular formula PCP: C_6Cl_5OH

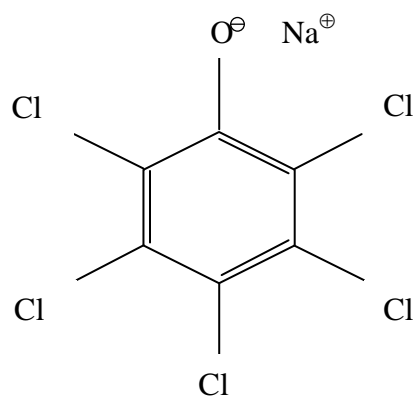
<i>CAS Registry Number:</i>	Pentachlorophenol	(CAS No.87-86-5)	(PCP)
	Sodium pentachlorophenate	(CAS No.135-52-2)	(NaPCP)
	Pentachlorophenyl laurate	(CAS No.3772-94-9)	(PCPL)

Structure:

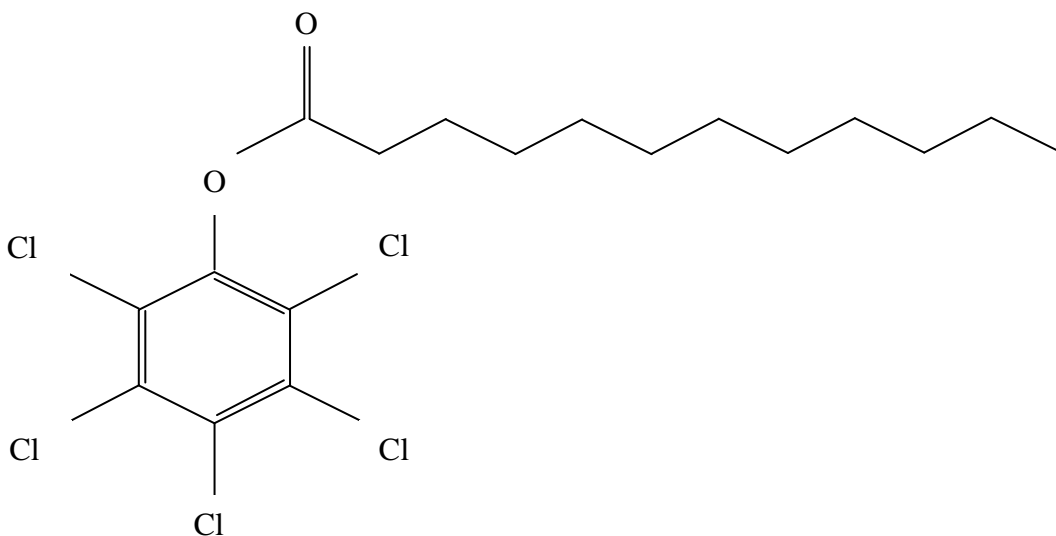
PCP:



NaPCP



PCPL



Molar Mass: 266.34 g/mol (PCP)

288.32 g/mol (NaPCP)

448.64 g/mol (PCPL)

Table 6-77: Physico-chemical properties of PCP ([UNECE 2010], [LfU 1996], [IEP 2002])

Aggregate state	Solid	[UNECE 2010]
Melting Point	191 °C	[UNECE 2010]
Density	1.987 g/ml at 22 °C	[UNECE 2010]
Boiling point	310 °C	[UNECE 2010]
Vapour pressure (at 100 °C)	2 mPa at 20 °C; 16 mPa at 100 °C	[UNECE 2010]
Partition coefficient Log Kow	5.12-5.18	[UNECE 2010]
Water solubility	Slightly soluble, (20 mg/l 20 C)	[UNECE 2010] [LfU 1996]
Henry's Law Constant	$3.4 \cdot 10^{-6}$ atm m ³ /mol	[IEP 2002]

Impurities of commercial PCP are in polychlorodibenzodioxin (PCDD), polychlorodibenzofurans (PCDF) and hexachlorobenzene (HCB) [UNECE 2010].

6.7.3 Environmental Background Levels

PCP has spread widely in the global environment. Table 6-14 and Table 6-15 give an overview of the concentration levels in sediment and sewage sludge detected in Europe in sediments, soil and sludges. 1 µg/kg is equal to 1 ppb or 0.001 ppm.

Table 6-78: Levels of PCP in sediment and soil (DW: dry weight; ww: wet weight; ND: not detected) [LfU 1996]

Country/Region	Compartment	Levels of PCP	References	Comments
Finland (1980)	Sediment	5-11 µg/kg	[LfU 1996]	Average in 0-2 cm
North Sea tideland close to discharge of a paper facility	Sediment	2.4-42 µg/kg	[LfU 1996]	
North Sea tideland reference	Sediment	2.4-7.2 µg/kg	[LfU 1996]	
North Sea tideland close to discharge (DE)	Sediment	5-16 µg/kg	[LfU 1996]	
North Sea coastal and estuarine waters (1991-1994)	Sediment	26.5 µg/kg	[OSPAR 1999]	Highest average conc., based on 77 samples analyzed
North Sea coastal and estuarine waters (1995-1997)	Sediment	<10 µg/kg	[OSPAR 1999]	All 33 samples analyzed from the same locations
Rivers (1985-1994)	Sediment	29.7 µg/kg	[OSPAR 1999]	Highest average conc. based on 244 samples
Rivers (1995-1997)	Sediment	<10 µg/kg	[OSPAR 1999]	Most of the 40 samples analyzed from the same locations
Rivers (1995-1997)	Sediment	11.25 µg/kg	[OSPAR 1999]	Highest average conc. of the 40 samples analyzed from the same locations
Germany (1995)	Soil	0.5 µg/kg dw	[BLfU 2008]	Source: Rippen 1995
Finland (1992)	Soil	heavily contaminated soils (> 4 mg/kg)	[OSPAR 2001]	Volume: 720 m ³ (of 154 soil contamination cases), immediate treatment
Finland (1992)	Soil	less contaminated soil (0.4 mg/kg to 4 mg/kg)	[OSPAR 2001]	Volume: 15 000 m ³ (of 154 soil contamination cases), no immediate treatment required

Table 6-79: Levels of PCP in sewage sludge including hot spot levels (DW: dry weight)

Country/ Region	Compartment	Levels of PCP	References	Comments
Germany (1987-1988)	Sewage sludge	Median: 28 µg/kg dw 90% Percentile: 166 µg/kg dw	[Jobst 1998]	85 sewage sludges in total west Germany
Germany (1994-1997)	Sewage sludge	Median: 19 µg/kg dw 90% Percentile: 86 µg/kg dw	[Jobst 1998]	101 sewage sludges in Rheinland-Pfalz

The data which is mainly based on sediment monitoring in the North Sea and of rivers which flow into the North Sea, indicate that a typical PCP background concentration of Sediment is between 5 to 30 µg/kg dw (= 0.005 to 0.030 ppm). It also shows that sea waters close to discharges are generally higher contaminated.

The data on soil in Finland of 1992 is based on 154 soil contamination cases, which were gathered in a Finnish register by environmental authorities [OSPAR 2001]. Two thirds of the contamination cases occurred at sawmills and 32 of the cases were in a groundwater area. The collected data shows a ratio of heavily contaminated to less contaminated soils of approximately 1:20.

In a German study a comparison of sewage sludges from two test series at the end of the eighties and the middle of nineties has been made. According to the explanations in the study, the sludges of Rheinland-Pfalz used in the second study would be representative for the rest of Germany. A decreasing trend to lower PCP concentrations in sewage sludges in Germany is explained as an effect of a ban in 1989. However it is also stipulated that using the Wilcoxon Test, the two measurement series are only different to a probability of 80 %. [Jobst 1998] Regarding this study it is assumed that PCP concentrations of sewage sludges are below 1 ppm Europe wide.

6.7.1 Legal Situation PCP

6.7.1.1 EU level

POPs related provisions

According to Directive 1999/51/EC which has been integrated into REACH Regulation 1907/2006 Annex XVII entry 22, pentachlorophenol and its salts and esters shall not be used in a concentration equal to or greater than 0.1 % by mass in substances or preparations placed on the market with effect of 1 September 2000. The impregnation of textiles and fibres with PCPL had to be ceased by 31 December 2008 in France, Ireland, Portugal Spain and the UK. In the other EU Member States it already had to be ceased by 31 August 2000 if it is not already banned by national legislation ([UNECE 2010], [1999/51/EC]).

PCP is discussed as a candidate under the POP protocol; currently, an inclusion to the scope of the Stockholm Convention is not discussed.

Classification/chemicals legislation

PCP is classified under CLP-Regulation 1272/2008 as

Carc. 2 H351
Acute Tox. 2 * H330
Acute Tox. 3 * H311
Acute Tox. 3 * H301
Eye Irrit. 2 H319
STOT SE 3 H335
Skin Irrit. 2 H315
Aquatic Acute 1 H400
Aquatic Chronic 1 H410

* Minimum classification

- H301: Toxic if swallowed,
- H311: Toxic in contact with skin,
- H315: Causes skin irritation,
- H319: Causes serious eye irritation
- H330: Fatal if inhaled
- H351: Suspected of causing cancer (state route of exposure if it is conclusively proven that no other routes of exposure cause the hazard)
- H400: Very toxic to aquatic life
- H410: Very toxic to aquatic life with long lasting effects.

Other fields (water, PRTR, food)

PCP is identified as priority hazardous substance in the Water Framework Directive 2000/60/EC. Environmental Quality Standards (EQS), both for annual average (AA-EQS) and for Maximum Allowable Concentration (MAC-EQS) for inland surface water for PCP are set in Directive 2008/105/EC as follows:

- Inland AA-EQS: 0.4 [µg/l]
- Other surface waters AA-EQS: 0.4 [µg/l].
- Inland AA-MAC: 1 [µg/l]
- Other surface waters AA-MAC: 1 [µg/l]

In the PRTR Regulation, it is laid down that PCP releases of > 10 kg/y to air, 1 kg/y to water and 1 kg/y to land have to be reported by the concerned facilities.

6.7.1.2 Member States level

Some European countries – Norway, Denmark, Germany, Netherlands and Austria – have introduced national legislation on PCP in products that is stricter than the harmonised EU legislation. EU legislation is only applicable to the use of PCP within the EU Member States. National legislation restricts the presence of PCP in products placed on the national market. In the Netherlands for example the trade and import of articles containing more than 5 mg PCP/kg is strictly prohibited since 1997. Belgium reports that PCP was never authorized as pesticide. In 1987 the authorization as biocide is withdrawn. Cyprus confirms that PCP is restricted under national legislation. ([UNEP 2010] [IEP 2008])

Austria:

PCP has a POP relevant limitation in the ordinance on waste water emissions for the production of plant protecting agents and crop spraying (Fed. Law Gaz. No. 1996/668). The limit values for PCP are 1 mg/l or 23 gram per ton installed capacity per plant.

Germany:

The limit value for PCP for different categories of soil are set in the Bundes-Bodenschutz-und Altlastenverordnung (BBodSchV), July 1999 p. 1554 (last amended 31 July 2009). The limit values for children playgrounds are set at 50 mg/kg DW, for residential areas 100 mg/kg DW, parks and industrial areas 250 mg/kg DW.

Poland:

The waste Concentration limit for PCP in treated industrial sewage is equal to 2.0 mg/l (daily average) and equal to 1.0 mg/l (monthly average) (Dz. U. Nr 27, poz. 169³⁵)

PCP in streams, creeks, rivers, canals, lakes, natural and artificial lakes, marine internal, transitional and

³⁵ Ordinance of the Minister of the Environment of 28 January 2009 amending the Regulation on conditions for the introduction of sewage into the water or soil to be met and on substances hazardous to the water environment

coastal waters equal to 1,0 µg/l (Dz. U. Nr 162, poz. 1008³⁶)

6.7.2 Occurrence and Relevance of PCP and its derivatives in Europe

According to Directive 1999/51/EC pentachlorophenol and its salts and esters shall not be used in a concentration equal to or greater than 0.1 % by mass in substances or preparations placed on the market. However, some of the products still in use can contain PCPs and are entering the waste stream even nowadays. This is especially the case for products with a long life time as e.g. impregnated wood.

6.7.2.1 Current uses

There are no current uses of PCP, NaPCP or PCPL in the EU 27 except the use below 0.1% by mass.

6.7.2.2 Past uses

The worldwide production in 1981 was estimated to be about 90,000 t/y but no recent data are available. It is likely that the worldwide production level has decreased due to restrictions. China is expected to produce about 5,000 t/y [UNEP 2010]. Commercial application of PCP in UK has been for wood impregnation, paints and as antimicrobial in paper and board. The use of NaPCP has been anti mildew agent in the wool textile industry and fungicide in adhesives. Both PCP and NaPCP have been used for agricultural applications. The use of PCPL has been as anti-mildew agent in the wool textile industry, mothproofing carried out by dyers and cleaners, wood preservation, antifungal agent in textiles other than wool (flax and jute fabric, ropes, cordage and tentage) and cable impregnation [IEP 2002].

The main application of PCP and NaPCP was for wood treatment but as stipulated in the ninth amendment on EC Directive 76/769/EEC the rot treatment is only authorised after 1 July 1992 for buildings of cultural, artistic and historical interest, or in emergencies. There is no data available of the number nor the extent of treatments carried out in the EC Member States after 1992 [IEP 2002].

In 1983, the use of PCP in Germany has been in wood treatment (61%), textile (13 %), leather (5%), mineral oil (6%) and glue (6%) industries. There has been no use of PCP in paint or pulp industry, whereas in 1974 about 3 and 7 %, respectively, were used in these industries. In case of textile the use is normally in the form of PCP esters [IEP 2002].

Large quantities of PCP and NaPCP have been used for agricultural applications. The domestic use, which was almost exclusively related to the treatment of wood, played only a minor role in the overall PCP market. It was also reported that PCP was included in health care products and disinfectants for use in the home, farms, and hospitals. PCP might have also been used in dental care products, bacterial soaps, laundry products and medical products for the skin [IEP 2002].

³⁶ Ordinance of the Minister of the Environment of 20 August 2008 on the classification of the state bodies of surface water

PCPL has been used in the preservation of textiles, which are subject to attack by fungi and bacteria during storage and use. The majority of treatment in the UK and France is carried out for textiles used by military. These include wool, cotton, flax and jute fabrics and yarns used in covers, tarpaulins, awnings, tents, webbing and netting and also sisal and manila ropes [IEP 2002].

In 1996 90% of the total EU consumption of PCP, NaPCP and PCPL was through the use of NaPCP for sapstain control in France, Portugal and Spain [IEP 2002].

Pentachlorophenyl laurate (PCPL) was developed especially for application on fabrics. About 80 % of the PCPL in the emulsion is adsorbed to the fibre or yarn during treatment in a dyeing process [IEP 2002].

According to an unpublished note submitted to the IPCS by Catomance Limited, Hertfordshire, the sole manufacturer of pentachlorophenyl laurate in the United Kingdom, the usage pattern in the United Kingdom was not changed following the cessation of production of PCP in 1978. However, most of the PCP ester used there was for domestic timber preservation; the use of PCPL for textile preservation was mainly confined to tropical or semi-tropical countries [IEP 2002].

Other reported applications of PCP included health-care products and disinfectants for use at home, farms and hospitals. PCP might also be contained in dental-care products, bactericidal soaps, laundry products and medical products for the skin [IEP 2002]. PCP, NaPCP and PCPL have been used in the past also as a preservative in oil-based paint-like products, as preservative in glues (leather, toilet paper etc.) and in adhesives, in mushroom farms for wooden trays where mushrooms are grown, in slime control, in pulp and paper production and in weed control [IEP 2002].

6.7.2.3 Unintentional production

Unintentional production of PCP is not known [BLfU 2008]. However, PCP but can be emitted from secondary sources. Former wood impregnation plants which used PCP can still be a source of PCP or NaPCP emissions. Especially the soil of the area which was used to dry the impregnated wood can be a source of emission. Another unintentional source is cotton which has been treated with PCP on the fields and remainder of these can be found in the raw product [BfLU 2008]; [IEP 2002]; [IEP 2008].

6.7.3 *Selection of Relevant Uses, Applications and Finished Articles for PCP*

PCP has been banned in most EU Member States in 2000 and in all Member States by 2008 due to EU legislation. However in many cases national stricter regulations led to a stop before these time limits. Especially products with long life times should be considered as relevant to enter the waste stream after 2010.

6.7.3.1 Current uses

Referring to Directive 1999/51/EC, there is a ban (< 0.1 %) on the use of PCP, NaPCP and PCPL in the EU 27 and therefore no relevant current use exists for these substances.

6.7.3.2 Past uses

There has been a wide range of application for PCP, NaPCP and PCPL. It was reported that PCP was used for leather and textile, as well as in the mineral oil and glue industry in 1983. Also the use of paper and pulp industry in 1974 has been reported. However, it is not reported that PCP or NaPCP have been used for these applications in the last 2 decades. As it has to be expected that the majority of the PCP and NaPCP in this applications has already entered the waste stream, they are not investigated any further.

The main application of PCP and NaPCP has been for wood treatment. The service life of such products is within a wide time range. According to [SEPA 2009] build-in woods such as panels, sheets and sole plates can have an unlimited life time. For dip treated wood in contact with ground the life time is considered to be in the order of 5-20 years and for pressure-treated wood in the order of 15-30 years [SEPA 2009]. For following calculations a service life of up to 40 years was considered. Within this service life it is assumed that the disposal rate of each year is equal and therefore set to 2.5 %. The materials can therefore still enter the waste stream and are investigated in this report.

PCPL has been used in Europe in the textile industry until 2002. Its typical applications have been for military purposes, tropical textiles and tentages. These products are considered to have a relatively long life time (15-20 years) and are therefore also considered to be a relevant waste streams.

6.7.4 Wood impregnation

A) Background

Pentachlorophenol, also known as “penta” or PCP, had its wood preserving properties discovered in the 1930s. It is usually dissolved in petroleum or other organic solvents that allows it to adequately penetrate wood. At times in the US, it is added to creosote to enhance preserving effectiveness. Most commercial uses of PCP are used to treat utility poles and crossarms for utility poles. It has no residential uses [UF 2005]. Advantages of PCP are that it can be dissolved in oils having a wide range of viscosity, vapour pressure and colour, and it is easy to handle and use. Disadvantages are that it does not provide protection as well as creosote, it is not suitable for use in living areas, and it is toxic and irritating to plants, animals and people [UF 2005].

For the use as wood preservative alternative chemicals as well as alternative materials are generally available and applicable. Which alternative performs comparably to PCP, varies upon the type of product and type of application. None of the applications of preserved wood exploration of management options for Pentachlorophenol (PCP) known is completely dependent on PCP, as reported by the US EPA. Although some of the alternative chemicals are more or less as toxic as PCP, like creosote, they do not contain the persistent microcontaminants that characterise formulated PCP products. The environment benefits even more from replacement of wood by materials like concrete, steel and fibre reinforced composite. To people who are frequently exposed to PCP-treated wood, like electrical utility linemen health risks reduce when other chemicals or alternate materials are used [UNECE 2010].

Wood poles treated with PCP or Na-PCP is hazardous waste [UNECE 2010]. In the waste phase they should be treated as such. After service life they are either incinerated, disposed of in landfills, reused or recycled. Once landfilled, treated poles still leach PCP and microcontaminants into the environment. Therefore, incineration under controlled conditions is preferred over disposal into landfills [UNECE 2010]. Also in the Swedish EPA report [SEPA 2009] it is mentioned, that landfill disposal of chlorophenol-treated wood products is a very unsatisfactory way of disposal, as it represents a risk of diffuse emissions over a very long period of time, together with loss of potential energy recovery and that in their opinion all demolition waste should be collected and combusted in incineration plants under controlled conditions. The use of PCP as wood impregnation agent was banned in the EU in general in 2000, but exceptions existed for some countries (e.g. France, UK, Portugal, Spain) until 2008. This means that in the meanwhile the use of PCP as wood impregnation agent is banned Europe wide and no PCP containing waste stream should occur from the impregnation sites. However, emissions from contaminated sites might still occur and contaminated soils at the production sites have to be supposed.

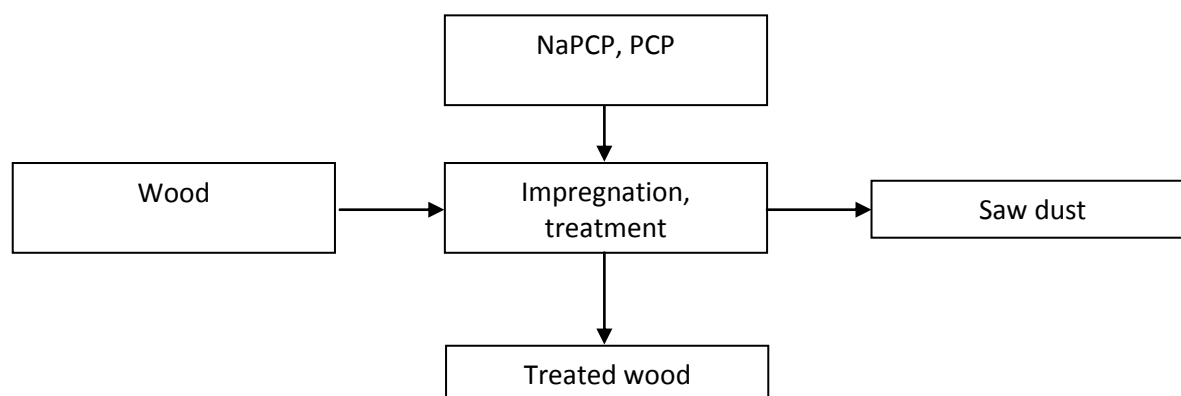
B) *Input/Output*

Figure 6-123: Input/Output of PCP for wood treatment.

Input: One of these inputs are different types of woods and the other are PCP containing solutions often with a concentration of about 2-5% [OSPAR 2001].

Output: The products are PCP treated woods and after tooling PCP containing saw dust. Due to the ban of use of PCP for wood impregnation since at least 2008 no outputs should occur anymore.

C) *Generation factors*

Losses during the product lifetime due to wear out are considered to be about 2 %. Emissions in the treatment process have not been considered as information is missing. There is also no information regarding the loss of PCP during production. Therefore the product generation factor in this industry is considered 1 as a worst case estimate. The use of 1 kg PCP results in 1 kg PCP in the impregnated wood. Considering the losses during lifetime, for the corresponding waste a generation factor of $1 - 0.02 = 0.98$ is used.

D) *Contamination Data*

The outer layers of treated wood contain up to several hundred mg/kg of PCP [UNECE 2010] or 0.15-5 kg/m³ [Wazny et.al. 1989]. It has been assumed that 15 l of 2,5 % (by weight) NaPCP solution is used to treat one cubic meter (m³) of wood by immersion and that 40 m² of wood surface area are associated with each cubic meter of treated wood [OSPAR 2001]. Other concentrations have been stated to range from 2.2-5 kg/m³ in case of pressure treated wood down to 0.1 kg PCPL/m³ in case of dip treated wood. For the dip treated wood it is further stated that the majority of PCP was already removed as the dip treated timbers were normally planed before use and the PCP penetrated area is only 1-2 mm [SEPA 2009]. In other reports a retention of 2.6 to 25 kg of NaPCP/m³ have been investigated regarding their preservative capability. However, this retention concentration might only be relevant for laboratory use as the block had only a size of 19 x 19 x 19 mm [Ahmad et.al 1982].

Volumes of NaPCP used in wood treatment annually were stated in the OSPAR report to be 450 tonnes in Portugal, 300 tonnes in France, 200 tonnes in the UK and 100 tonnes in Spain in 1997. It was further calculated that emissions to air were in Portugal 516 kg/a, in France 344 kg/a, in the UK 229 kg/y and in

Spain 115 kg/a. This corresponds to an emission factor of 0.1 % per year [OSPAR 2001]. Considering an average lifetime of 15 years this would represent a total emission of about 1.7 %. This is much less than the estimate made by WHO. Depending on the solvent, temperature, pH, and type of wood, 30-80 % of PCP may evaporate within 12 months from dip- or brush treated wood [OSPAR 2001]. According to [UNEP 2010], PCP has historically been estimated to volatilize from the surface of treated wood roughly at 2% of the total amount of the preservative applied. This value has been taken for the calculation in this report.

The ppm value of PCP substances in air-dried wood was calculated to be 625 ppm using the following data: 15 litre of a 2.5 % impregnation solution is applied to wood, resulting in 0.375 litre of NaPCP in one cubic meter. The density (ρ) of NaPCP is assumed to be 1 g/cm³, therefore the amount of NaPCP in wood is consistent with 0.375 kg/m³. The ppm is calculated by dividing the concentration of NaPCP with the average density of wood, which is considered to be in the range of 0.4-0.8 g/cm³ [Stoffdichten online]. For the calculation a mean value of 0.6 g/cm³ or 600 kg/m³ has been used. This results in a Na-PCP concentration of 625 ppm.

In 1992 and 1993, 45 samples of wooden goods/articles (kitchen equipment, children's play equipment and fruit crates) were analysed. In all samples the PCP concentrations were below 2.5 mg/kg. A Swiss campaign was carried out in 1996 to study the market situation (i.e. import) of wood, textile and leather treated with PCP; 129 samples of wood were analysed and the highest PCP levels detected were in the range of 500-1,000 ppm. A limit value of 5 ppm was exceeded in 9.3 % of the samples [OFEFP 1998]; [OSPAR 2001].

E) Activity data

PCP treated woods are considered to have a life time up to 30 years with an average life time of 15 years (crates: 5 years, fences: 10 years, construction wood: 30 years), which has to be considered for the activity data [TNO Phase 1 2005]. For the disposal range it is assumed that from one production year an equal annual share is disposed of within 25 years.

The production of PCP and NaPCP ceased in the EU in 1992 and the production of PCPL in 2000. Between 1992 and 2000 PCP was imported to the UK to produce PCPL. It is assumed that between 1992 and 2008 there has been a steady decline of the import of Na-PCP to the four countries and that all the imported NaPCP has been used for wood impregnation. For the time period between 1980 the productions in Germany and France have been considered. As far as known about 4,000 tonnes [IEP 2002] have been produced at Dynamit Nobel in Germany and another 4,500 tonnes at Rhone-Poulenc [IEP 2002] in France. Together with some smaller not further known companies it is estimated, that the annual production has been about 10,000 tonnes in the EU 27 until 1986 when the German production stopped [IEP 2002]. Together with the production stop in France (proposed in 1992 together with the other unknown small producers) the production and import volumes shown in Table 6-80 have been developed. In 1996 378 tonnes and in 1999 328 tonnes of NaPCP have been imported into the EU (France, Portugal, UK and Spain) as well as 30 and 20 t of PCP respectively into the UK [IEP 2002]. With 1 tonne of PCP about 1.53 tonnes of PCPL can be produced [TNO Phase 1 2005].

According to [UNECE 2010] PCP was produced in Spain until 2003 but no production data are available and therefore the amount is considered to be negligible compared to the import which is assumed to be mainly from the U.S. [UNECE 2010]. For simplification it is assumed, that regarding NaPCP until 1992 no imports or exports outside of the EU 27 occurred, which is also stated by [UNECE 2010], including that imports started with the stop of production in the EU. In the following table the assumed production and imports of the EU 27 are shown, which is visualised in Figure 6-124.

Table 6-80: PCP production and import in the EU 27

Year	Production [t]	Import [t]	Total [t]
1984	10,000	0	10,000
1985	10,000	0	10,000
1986	10,000	0	10,000
1987	4,500	0	4,500
1988	4,500	0	4,500
1989	4,500	0	4,500
1990	4,500	0	4,500
1991	4,500	0	4,500
1992	4,500	450	450
1993	0	432	432
1995	0	396	396
1996	0	378	378
1997	0	360	360
1998	0	342	342
1999	0	324	324
2000	0	288	288
2001	0	252	252
2002	0	216	216
2003	0	180	180
2004	0	144	144
2005	0	108	108
2006	0	72	72
2007	0	36	36
2008	0	0	0

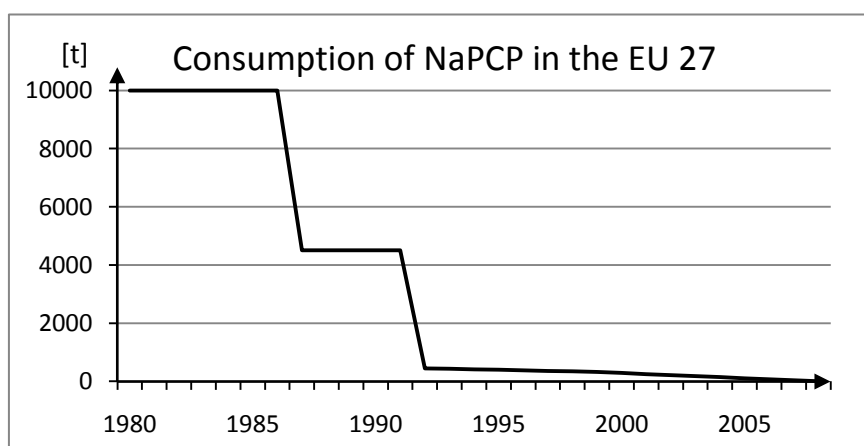


Figure 6-124: Consumption of Na PCP for wood treatment in the EU 27

However, according to this report as well as others as the OSPAR report, the vast amount of NaPCP is considered to be for wood treatment.

F) Waste Treatment

Today, almost all waste in Sweden is treated with controlled methods. Most wood and wood-based materials are combusted under controlled conditions in special incineration plants. Wood-based waste was previously also combusted, partly in the wood industry's own boilers and partly in open, completely uncontrolled fires. However, large quantities of wood waste were also sent to landfill [SEPA 2009]. In Germany all preserved wood is incinerated for other countries the final use of preserved wood after life time is unclear [UNEP 2010]. It has to be expected that the predominant amount of treated wood in the EU 27 is either incinerated or disposed of at landfill for hazardous waste. Due the limit value of total organic carbon (TOC) of 5-6 % set in the WAC Decision 2003/33/EC, wood waste is only allowed to be disposed of at inert or mere non-hazardous waste landfills (non-hazardous waste landfills accepting stable non-reactive hazardous waste are not allowed to accept this type of waste regarding to 2003/33/EC). PCP treated wood waste is in general considered to be hazardous waste and is not allowed to be disposed of at hazardous waste landfills due to the TOC content. Therefore the share of PCP treated wood is expected to be close to 100 % hazardous waste incineration.

A possible reuse of old treated wood can be neglected as this material is normally strongly affected by the environment and not usable anymore.

The disposal of PCP-containing waste should preferably involve controlled high-temperature combustion. PCP treated wood combusted at 910-1025°C resulted in no detectable chemicals in the off gas. However, the incineration temperature needs to be high enough and the resident time long enough to fully degrade the PCP, as it combusts incompletely at temperatures under 800°C [UNEP 2010]. The controlled combustion conditions under which most wood and wood based materials are incinerated today in Sweden quickly destroy dioxins and other chlorinated aromatic substances in the treated timber [SEPA 2009].

Table 6-81: Annual disposal of PCP in production and remaining amount for future disposal in the EU 27

Year	Annual disposal amount [t/y]	Future disposal amount [t]
2010	1838	6045
2011	1446	4599
2012	1054	3545
2013	878	2667
2014	701	1966
2015	525	1441
2016	349	1092
2017	172	920
2018	155	766
2019	138	628
2020	121	507
2021	106	401

2022	91	310
2023	77	233
2024	64	169
2025	51	119
2026	40	79
2027	30	49
2028	21	28
2029	14	14
2030	8	6
2031	4	1
2032	1	0

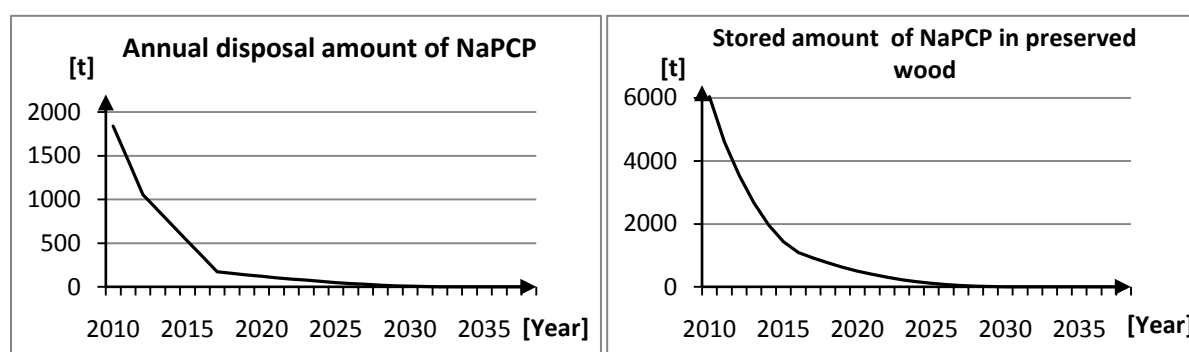


Figure 6-125: Annual disposal of PCP in production and remaining amount in use in the EU 27

Due to the long lifetime of treated wood it is assumed, that at least until 2017 there will be a considerable annually amount of PCP containing waste disposed of.

With data provided from [SEPA 2009] on the stored amount of PCP in treated wood, the following amounts could be calculated or have been transposed:

From pressure treated wood: 225 t

From dip-treated wood: 50 t

From DIY use: 55 t and

From remedial treatment products 10 t,

All sum up to 340 t of PCP still in use in Sweden. This is about 5% of the EU 27. The population of Sweden is only about 1.8 % but it was also reported that the used amounts in the report are considered to be at the high side and it has to be considered that wood is a more common building material in Sweden compared to the rest of the EU 27. Under this consideration the both estimations are considered to be close together.

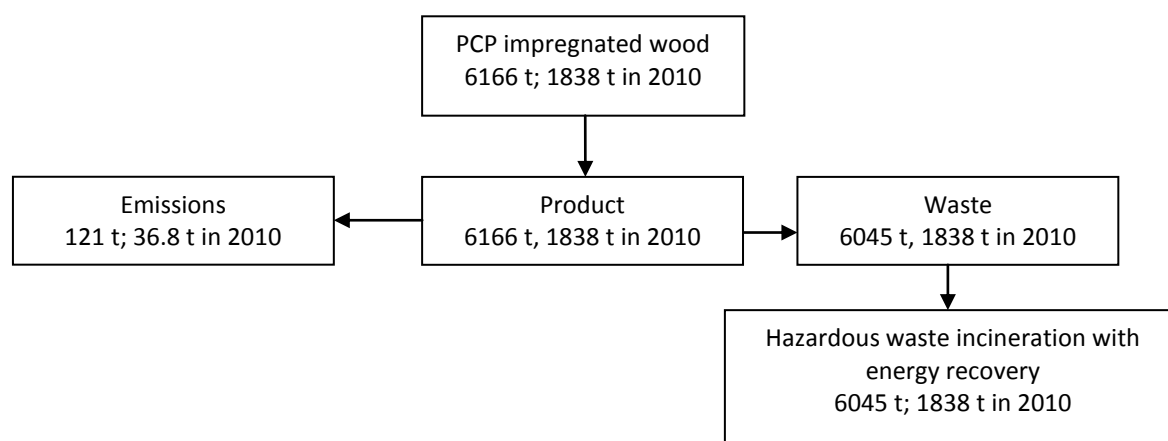
G) *Substance flow*

Figure 6-126: Substance flow of PCP for wood impregnation

Figure 6-126 gives an overview of the total amount of PCP contained in treated wood still in use as well as an annual breakdown of the waste stream. As discussed in the section waste treatment, all end of life wood containing PCP should be incinerated in a hazardous waste incineration plant. It is expected that about 2 % of the PCP in the wood is emitted to soil, water and air during lifetime.

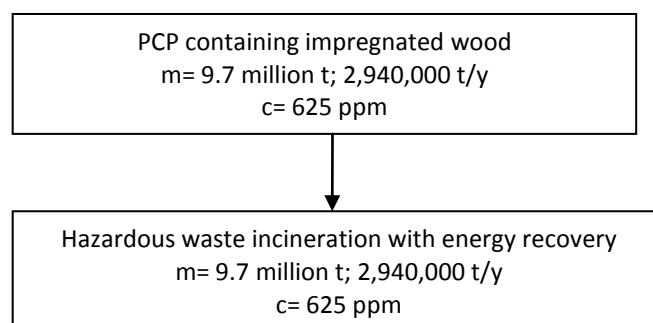
H) *Material flow*

Figure 6-127: Material flow of PCP for wood impregnation

For the calculation an average PCP concentration of 625 ppm has been considered.

6.7.5 Textiles

A) Background

Pentachlorophenyl laurate (PCPL) is used for the treatment of fabrics and webbings, but also for yarns and ropes. PCPL is insoluble in water and is therefore applied either in solvent solution or, more normally, as a concentrated aqueous solution/emulsion. Fabrics and webbings are usually passed through a batch, containing the emulsion and using a roller system in a continuous process. The fabric is then squeezed by rollers (the mangle), the pressure of which is adjusted so that the required quantity of PCPL remains in the fabric. Any excess emulsion is passed back into the treatment vessel. The fabric is then dried and other additives such as water repellents and fire retardants are applied at the same time [OSPAR 2001].

Discharges to water and emissions to air are possible from the use of PCPL (and PCP) treated textiles. Textiles which have lately been treated in Europe are heavy-duty textiles for outdoor use, where they may be in contact with rain, surface water or sea water. Textiles are used for transport lorries, public market trade stalls, army tents, awnings, ropes, sails and marine fabrics. The main use of textiles treated in Europe was for the UK armed forces. No estimates are available of the volume of treated textiles, but the volume of PCPL used for treatment ranged from 30 to 46 tonnes per year in the late 1990s. Releases during the use phase depend on the type of textile, the environmental conditions and the application. When treated tents and tarpaulins are exposed to weather, PCPL may undergo photodegradation; no studies on this degradation have been reported. Under wet conditions PCPL will slowly decompose to PCP in its ionic form. This is washed out of the textile or is lost by vaporisation. This is a very slow process; it is generally expected that it takes 10 years for the PCPL concentration to drop from 2 % to less than 1 %; at this level the preservative effect is considerably reduced. However, it is not possible to estimate the extent of emissions or discharges [OSPAR 2001].

B) Input/Output

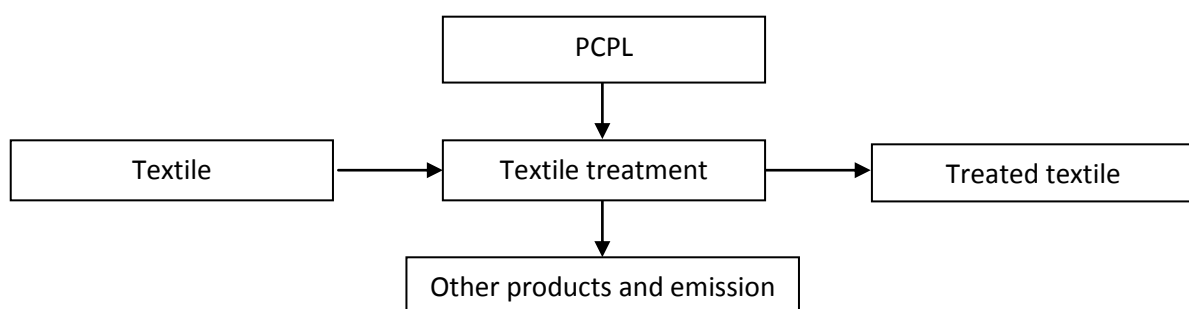


Figure 6-128: Input/Output of PCP in the textile industry

Input: textiles and PCPL solution.

Output: the product is a water proof and antifouling textile

C) Generation factors

It is reported [OSPAR 2001] that about 80 % of the PCPL will be adsorbed on the textile. From the remaining 20 % about 80 % will be removed in the effluent treatment in the plant before treatment and the remaining 20 % (4 % of the total PCPL amount) will be treated in a sewage treatment plant [OSPAR 2001]. Losses during the product lifetime due to wear out degradation and emissions in the impregnation process have not been considered. The generation factor in this industry is $1 - 0.2 = 0.8$. The use of 1 kg PCPL results in 0.8 kg PCPL in the impregnated textile. The average PCPL content is considered to be about 2.5 % and therefore 32 kg of textile can be treated with 1 kg of PCPL.

D) Contamination Data

It is assumed, that 500 kg of yarn is treated in one batch. 12.5 kg of PCPL is applied and 10 kg (80 %) is taken up by the fibre. Of the remaining 2.5 kg PCPL, 80 % is assumed to be removed in effluent treatment in the plant before discharge to waste water [OSPAR 2001]. The resulting concentration of the textile is therefore about 2.5 % or 25,000 ppm. As textile treatment with PCP does not take place in the EU anymore the emission from the treatment step is not further investigated.

In studies in Germany and Switzerland in the beginning and middle of the nineties some leather and textile products have been analysed regarding their PCP content. In 5 to 29 % of the leather products analysed in the German study a PCP level of 5 ppm was exceeded. In case of Switzerland 6.8 of the analysed textiles and 4.3 % of the leather products exceeded 5ppm [OSPAR 2001]. No current data regarding PCP concentrations in textiles sold in Europe are available.

E) Activity data

In the majority of EU countries the use of PCPL for textile has already been banned at the latest at 2000. Therefore the waste streams from these countries should be negligible compared to the countries who allowed using PCPL until 2008. In 1996 about 46 tonnes of PCPL have been produced in the UK. About 5-10 tonnes were exported to France and a similar amount was used in the UK. The rest was exported outside of the EU 15 (~30 t). This export could have been to the new EU 12 Member States (this worst case scenario has been assumed for the calculation, due to a lack of more detailed data). In 1999 about 15 tonnes of PCP have been used to produce about 23 tonnes of PCPL.

With the production of 46 t and 23 tonnes in 1996 and 1999, respectively [OSPAR 2001], this trend has been used for the timeframe between 1992 and 2000. In 1992 it is assumed that a similar break in took place as for NaPCP and PCP which was assumed to be about a factor of 10. Another drop is the sudden use and production stop in 2000.

Considering the nature of PCPL treated textiles, it is assumed that the PCP containing textile products have a life time of about 15-20 years (average: 17.5 years). For the calculation a constant disposal distribution within a lifetime of 15-20 years was used.

The following table provides the assumed values of PCPL used in the textile industry

Table 6-82: PCPL production and import in the EU 27

Year	Total amount [t]	PCPL in Textile [t]	Year	Total amount [t]	PCPL in Textile [t]
1989	767	613	1996	46	37
1990	767	613	1997	38	31
1991	767	613	1998	31	25
1992	77	61	1999	23	18
1993	69	55	2000	0	0
1994	61	49	Total	2,737	2,190
1995	54	43			

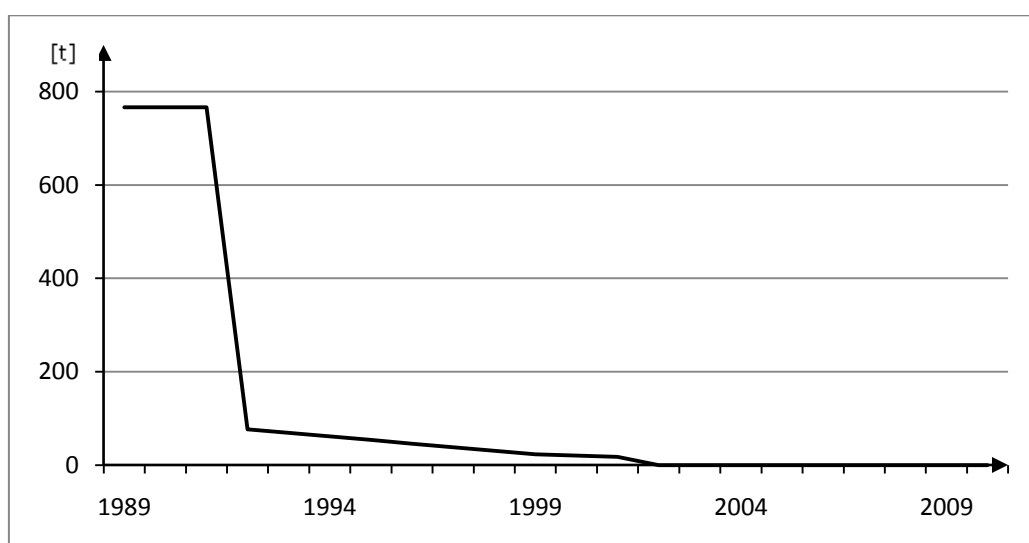


Figure 6-129: Use of PCPL for textile impregnation in the EU 27

F) Waste Treatment

As it is usually impossible without a special laboratory test to notice whether wood or textile has been treated with PCP, NaPCP or PCPL, a separate recovery and recycling of treated material seems to be not a realistic option at present. Therefore, the two most common ways of disposal of PCP, NaPCP or PCPL wastes are landfills and incineration/burning [OSPAR 2001]. In general the goal of the recycling process is to receive long textile fibres which can be used to produce new textiles [PFOS SOEX 2010]. Due to the nature of the PCP treated textile and the intensive incorporation of the fabrics with the PCP containing coating it is doubtful that it is possible to receive long fibres. These types of textiles are therefore considered not to be suitable for recycling and will end up as MSW. The treatment shares are considered to be equal to MSW without composting and recycling.

Table 6-83: Annual disposal of PCPL in production and remaining amount for future disposal in the EU 27

Year	Annual disposal amount [t/y]	Future disposal amount [t]
2010	239	360
2011	143	217
2012	46	171
2013	40	131
2014	34	97
2015	28	69
2016	24	45
2017	17	28
2018	12	16
2019	8	7
2020	5	2
2021	2	0
2022	0	0

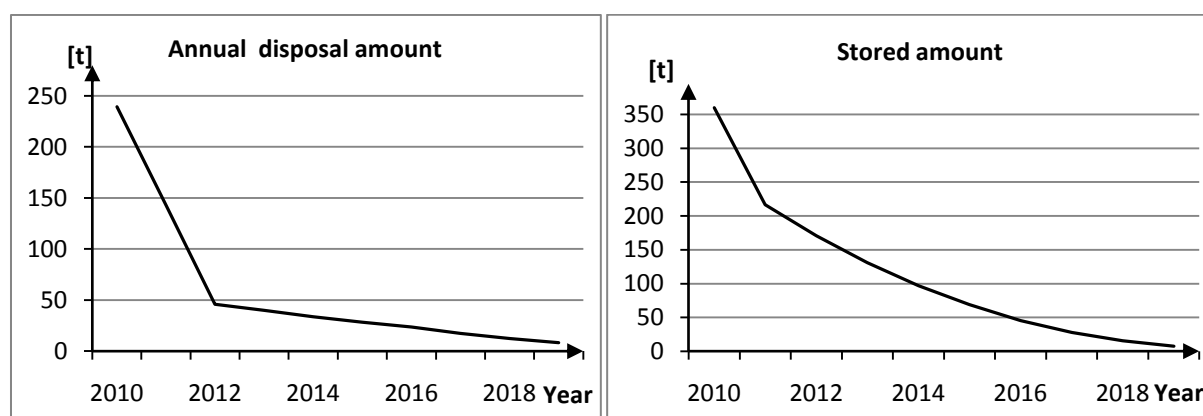


Figure 6-130: Remaining PCPL in use from the textile industry; annual disposal of PCPL from textile products

It is expected that the majority of the waste ends up as non hazardous waste. This waste goes to 67.1 % to landfill, 22.2 % to incineration without recovery and 12.7 % incineration with energy recovery. It is expected that this waste stream will continue till ~2022.

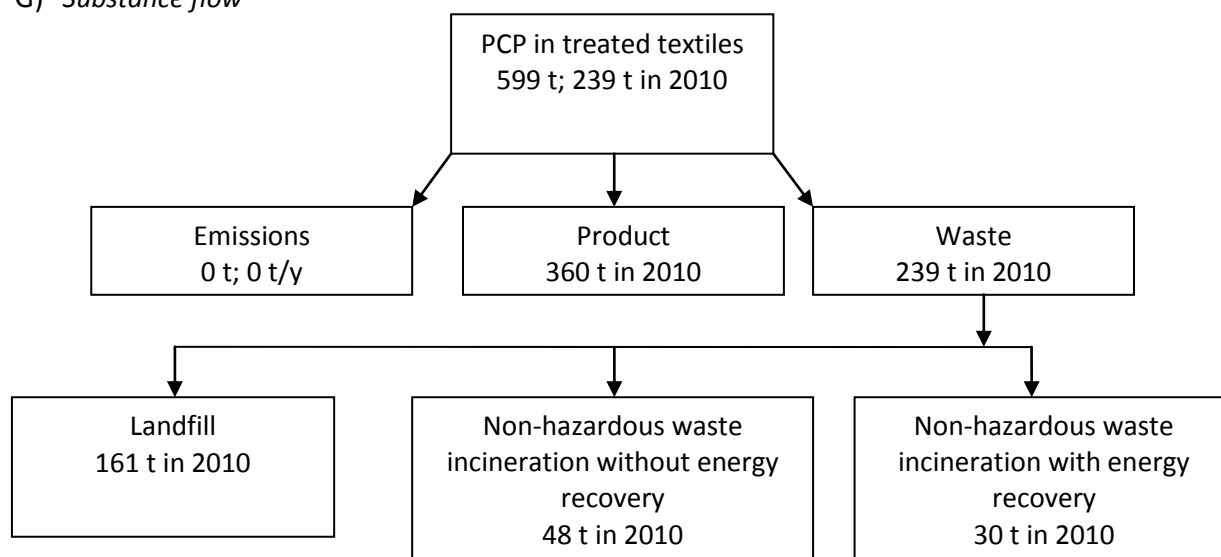
G) *Substance flow*

Figure 6-131: Substance flow of PCP for wood impregnation

As described above it is assumed that PCP containing waste will mainly be disposed of similar to MSW waste. Considering emission of about 50% as indicated in [OSPAR 2001] the amounts could be cut in half but even in that report it is stated that it is not possible to estimate the extent of emissions or discharges.

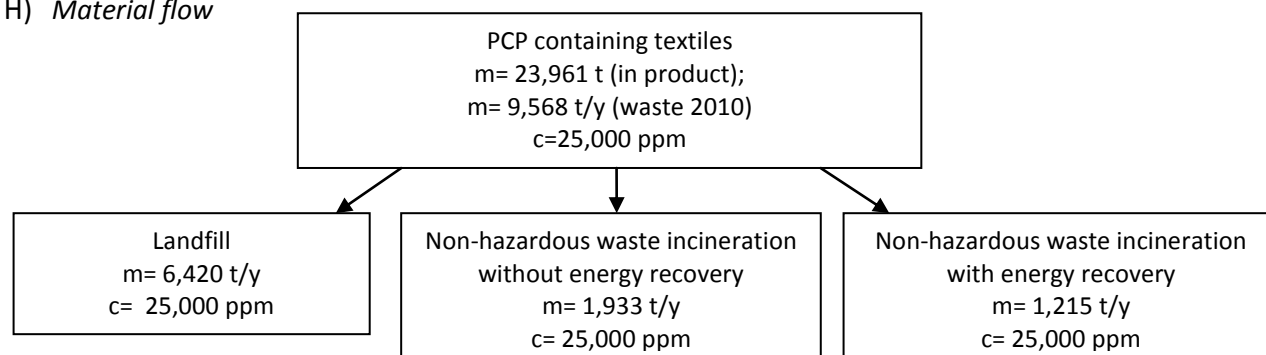
H) *Material flow*

Figure 6-132: Material flow of PCP for wood impregnation

With the average PCPL concentration of 2.5 % the total waste amounts can be calculated from the substance flow figure. In case emissions would be considered the concentration would have to be cut in half but the total masses should stay the same.

6.7.6 References PCPs

[Ahmad et.al. 1982]

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6.8 HCBd flows

6.8.1 Use and Production

Hexachlorobutadien (HCBd) has been historically used as solvent for rubber and other polymers, in heat transfer fluids, as transformer liquid, as hydraulic fluid, for the recovery of “snift” (chlorine containing gas in chlorine plants) and washing liquor for removing certain volatile organic compounds (VOCs) as e.g. hydrocarbons from gas streams. The two latter uses have been reported to represent the most significant ones prior to 1975 [UNECE 2007]. Further, HCBd has been applied as intermediate for the manufacture of fluor-containing lubricants and rubbers, as gyroscopic liquid, as biocide for prevention of algae production as well as in some EU countries as fumigant for treating grapes in vine production [UBA 2006]. EU countries reported to formerly have used HCBd as fungicide for this purpose are France, Italy, Greece and Spain. Outside the EU Argentina and Russia have been reported to use HCBd as fungicide as well, whereas it is assumed that Russia still makes use of HCBd for this purpose [UNECE 2007]. Other historical uses might include use in the production of aluminum and graphite rods [UNECE 2007; UBA 2006].

At present there is no intentional use of HCBd in Europe since many years [UNECE 2007].

HCBd is not anymore intentionally produced within Europe [UBA 2006]. Van der Gon et al. refer to information from Eurochlor according to which 50 kg of HCBd has been produced in Europe in 2000 [UBA 2006]. Another source reported that commercial production of HCBd in Europe stopped in the late 1970s [UNECE 2007]. Nowadays, HCBd is mainly formed as unintentional by-product during the production of chlorinated hydrocarbons such as tri- and tetrachloroethene and tetrachloromethane. There also may be substantial amounts of by-product formation from non-chemical facilities producing magnesium [UNECE 2007].

In summary, HCBd has been historically produced and used for several purposes, but at present there exists no intentional production or use of this substance within the EU. The only current source of HCBd is due to unintentional production in specific industrial processes.

6.8.2 Chemical Characteristics of HCBd

HCBd is an organic compound, which belongs to the group of halogenated aliphatic substances. It is completely perchlorinated and further exhibits a high reactivity due to its unsaturated character. 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 ([UNECE 2007], [GESTIS 2010], [Euro Chlor 2004]).

IUPAC Name: hexachlorobutadiene

CAS Chemical Name: 1,1,2,3,4,4-hexachloro-1,3-butadiene

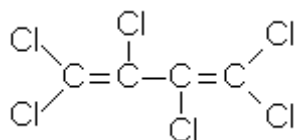
Synonyms: HCBd, perchlorobutadiene, perchloro-1,3-butadiene, 1,1,2,3,4,4-hexachloro-1,3-butadiene, hexachloro-1,3-butadiene, 1,3-hexachlorobutadiene ([UNECE 2007], [UBA 2006], [Euro Chlor 2004], [IARC 1999])

CAS Registry Number: 87-68-3

EINECS Number: 201-765-5

Previous Trade Names: Dolen-Pur, C-46, UN2279, GP-40-66:120 [UNECE 2007] [Euro Chlor 2004]

Structure:



Formula: C₄Cl₆

Molar Mass: 260.76 g/mol [GESTIS 2010]

Physico-chemical properties

Melting point	-21 °C	[GESTIS 2010], [Euro Chlor 2004], [UNE 2007]
Boiling point	215 °C	[GESTIS 2010], [Euro Chlor 2004], [UNE 2007]
Density (at 20 °C)	1.68 g/cm ³ 1.55 g/cm ³	[GESTIS 2010] [Euro Chlor 2004]
Vapour pressure (at 20, 30, 50 °C)	0.36, 0.75, 2.80 mbar	[GESTIS 2010]
Vapour pressure (at 20 °C)	0.2	[UNECE 2007]
Ignition temperature	610 °C	[GESTIS 2010], [Euro Chlor 2004]
Partition coefficient Log K _{ow} (Octanol/water)	4.78	[GESTIS 2010], [Euro Chlor 2004], [UNE 2007]
Water solubility (at 20 °C)	0.50 g/L 2.55 mg/L 3.20 mg/L	[GESTIS 2010] [Euro Chlor 2004] [UNECE 2007]
Henry's Law Constant	1044 Pa m ³ /mol	[UNECE 2007]

6.8.3 Environmental Background Levels

HCBD has spread widely in the global environment. Different concentration levels have been detected in abiotic and biotic media. Monitoring data on HCBD shows that levels of the substance were detected in atmosphere, biota and humans as well as in sediment and soil. A comprehensive overview of concentration levels in sediment and soil detected in different parts of the world is provided in Table 6-32. Besides, the identified 'hot spots' are presented.

Table 6-84: Levels of HCBD in sediment and soil including hot spots (DW: dry weight) ([Euro Chlor 2004]; [IPCS 1993, IPCS 1994]; [CEPA 1999])

Country/ Region	Compartment	Levels of HCBD	References	Comments
Sediment				
Canada	Sediment	not detected	[Staples <i>et al.</i> 1985]	As reported by the US EPA, HCBD was not detectable in any of 196 sediment samples, based on a detection limit of 0.5 mg/kg for the analyses. Sediments from the Niagara River contained 2.9-11 µg/kg HCBD. Sediments from the Great Lakes were reported to contain levels of HCBD typically ranging from 0.08-120 µg/kg.
		2.9-11 [µg/kg]	[Oliver and Bourbonniere 1985]	
		0.08-120 [µg/kg]	[McConnell <i>et al.</i> 1975]	
Canada	Sediment	550 [µg/kg]	[IPCS 1993]	Great Lakes area; older sediment layers, (~1960), contained higher concentrations of up to 550 µg/kg wet weight.
Canada	Sediment	0.01-0.23 [µg/kg]	[Mudroch <i>et al.</i> 1992]	It was found that HCBD was present at concentrations ranging from 0.01-0.23 ng/g at various sediment depths in samples taken from Great Slave Lake in 1987.
Canada	Sediment, river	9.6-37 [µg/kg]	[Fox <i>et al.</i> 1983]	Canada, Niagara River, downstream; sampled in 1981. Levels determined 9.6-37 µg/kg DW.
Canada	Sediment, lake	12-120 [µg/kg]	-	Canada, Lake Ontario; 12-120 µg/kg DW.
Canada	Sediment, lake	0.1-75 µg/kg	[Oliver 1984]	Canada, Lake Ontario
Canada	Sediment, lake	nd-550 [µg/kg]	[Durham & Oliver 1983]	Niagara Falls sediment; 1868-1981
United States	Sediment	0.05-0.40 [µg/kg]	[Abdelghani <i>et al.</i> 1995]	Sediments from sites in Louisiana.
United States	Sediment, sea harbour	<0.79 [µg/kg]	[Malins <i>et al.</i> 1985]	USA, Eagle Harbour, contaminated sediment
United States	Sediment, sea harbour	<2.0 [µg/kg]	-	USA, President Point, 1 reference site
Europe	Sediment	<0.1-1.8 [µg/kg]; 2-5 [µg/kg]	[IUCLID 1994]	Samples collected around Hamburg contained <0.1-1.8 µg/kg DW of sediment while a study conducted from 1980 to 1981 reported levels of 2-5 µg/kg DW in sediment from the Rhine. Pearson and McConnell (1975) examined the concentrations of HCBD in marine sediments and while a few samples indicated concentrations above 1 µg/kg, the majority were below 0.5 µg/kg.
		<0.5 [µg/kg]	[Pearson and McConnell 1975]	

Country/ Region	Compartment	Levels of HCBd	References	Comments
Europe	Sediment, river	0.002- 0.005 [µg/kg]	[Alberti 1983]	Germany, River Rhine, 707 km (0.002 µg/kg DW) idem, 815 km (0.005 µg/kg).
Europe	Sediment	1.1 [µg/kg] (mean); 4 [µg/kg] (90 th per.)	[Govaerts et al. 2000 and 2004]	Statistical analysis of about 500 measured concentrations indicated that the mean and the 90th percentile values of HCBd concentrations in sediments in Europe are 1.1 and 4 µg/kg, respectively. Recent measurements indicate concentrations in estuarine or coastal sediment vary between <0.2 and 3 µg/kg, with typical values close to 1µg/kg.
		<0.2-3 [µg/kg]	[EU COMMPs 1998]	
Europe	Suspended solids	<3.4-19 [µg/kg]	[Hendriks et al. 1998]	Recent data for suspended solids collected from the Rhine-Meuse river basin indicate HCBd levels ranging from <3.4 to 19 µg/kg.
Taiwan	Sediment	few µg/kg	[Lee et al. 2000]	Surface sediment samples collected from 40 stations along the Kaohsiung coast in southern Taiwan. HCBd was found at a level of a few µg/kg in the most polluted locations.
Soil				
Canada	Soil	-	[Webber and Wang, 1995]	In the only identified relevant survey of soils in Canada, HCBd was neither detected (detection limit 0.05 µg/g dry weight) in 24 samples of agricultural soils from across the country, nor in six samples from areas that had repeatedly received heavy applications of pesticides.
Hot Spot levels (sediment and soil)				
Canada;	Sediment	430 [mg/kg]	[Oliver and Pugsley 1986]	St Clair River, Ontario, Canada; samples for sediment were taken prior to 1986. Concentration is based on DW. In 1985, HCBd was also detected in the same area with lowest reported concentration being 0.1 µg/kg DW.
Canada	Sediment (top 5 cm of sediment)	310 [mg/kg]	[Farara and Burt 1997]	St Clair River, Ontario; samples taken in 1994. The highest concentration measured in 1994 was 310 mg/kg DW. Concentration ranged from <0.001-243 mg/kg DW. Bottom sediment levels up to 0.120 mg/kg DW.
United States	Sediment	2-82 [mg/kg]	[US EPA 1992; Prytula and Pavlostathis 1996; Gess and Pavlostathis 1997; Chen <i>et al.</i> 1999]	Several hot spots were identified in Baton Rouge and Lake Charles areas.

Most of the monitoring data on HCBd contamination levels in sediment and soil exists for Canada and the United States. As shown in Table 6-32, the detected contamination levels in sediments ranged from 0.01 µg/kg, measured in sediment samples taken from the Great Slave Lake, up to 550 µg/kg measured in older sediment layers in the Great Lakes area. In the United States, levels of HCBd have been reported between 0.05-2.0 µg/kg (see Table 6-32).

HCBd was found in surface sediment samples at a level of a few µg/kg in the most polluted locations

along the Kaohsiung coast in southern Taiwan (see Table 6-32).

For Europe, statistical analysis of about 500 measured concentrations indicated that the mean and the 90th percentile values of HCBd in sediments are 1.1 and 4 µg/kg, respectively. Recent measurements indicate that concentrations in estuarine or coastal sediment vary between <0.2 and 3 µg/kg, with typical values close to 1µg/kg. Specific data for Germany shows HCBd concentration values between <0.1-1.8 µg/kg, while a study conducted from 1980 to 1981 reported levels of 2-5 µg/kg in sediments from the Rhine. Even lower concentration figures from the Rhine River were reported (0.002-0.005 µg/kg). Measurements in sediments collected from the Rhine-Meuse river basin indicate HCBd levels ranging from <3.4-19 µg/kg (see Table 6-32).

Several hot spot levels were detected in the United States which ranged from 2 mg/kg-82 mg/kg in sediment samples from the Baton Rouge and Lake Charles areas. The highest concentration measured in 1994 in sediment samples from the St Clair River in Canada was 310 mg/kg. Sediments samples taken prior to 1986 show HCBd concentrations up to 430 mg/kg (see Table 6-32).

In the only identified relevant survey of soils in Canada, HCBd was neither detected (detection limit 0.05 µg/g DW) in 24 samples of agricultural soils across the country, nor in six samples from areas that had repeatedly received heavy applications of pesticides (see Table 6-32).

6.8.4 Legal Situation

6.8.4.1 EU level

POPs related provisions

HCBD is a POP candidate under the Stockholm Convention and is listed in the Annex I to the POP Protocol.

Other fields (water, PRTR, food)

HCBD is identified as priority hazardous substance in the Water Framework Directive 2000/60/EC. Environmental Quality Standards (EQS), both for annual average (AA-EQS) and for Maximum Allowable Concentration (MAC-EQS) for inland surface water for HCBD are set in Directive 2008/105/EC as follows:

- Inland AA-EQS: 0.1 [µg/l]
- Other surface waters AA-EQS: 0.1 [µg/l].
- Inland AA-MAC: 0.6 [µg/l]
- Other surface waters AA-MAC: 0.6 [µg/l]

6.8.4.2 Member States level

Poland:

The concentration limit for HCBD in treated industrial sewage is equal to 3.0 mg/l (daily average) and equal to 1.0 mg/l (monthly average) [Dz. U. Nr 27, poz. 169]³⁷.

Water:

In streams, creeks, rivers, canals, lakes, natural and artificial lakes, marine internal, transitional and coastal waters equal to 0.6 µg/l [Dz. U. Nr 162, poz. 1008]³⁸.

6.8.5 Occurrence of HCBD in Europe

According to IPCS the world annual production of HCBD in 1982 was estimated to amount to 10,000 t/y [IPCS 1993], whereas it was indicated that no commercial production of HCBD took place in the USA and Canada ([UNECE 2007], [Euro Chlor 2004]). It was noted by Euro Chlor that this value for the world annual production arises from the production of HCBD as by-product during chlorination processes involving organic compounds, where HCBD is generated in the heavy fractions [Euro Chlor 2004].

³⁷ Ordinance of the Minister of the Environment of 28 January 2009 amending the Regulation on conditions for the introduction of sewage into the water or soil to be met and on substances hazardous to the water environment

³⁸ Ordinance of the Minister of the Environment of 20 August 2008 on the classification of the state bodies of surface water

HCBD has been intentionally produced for several uses in Europe in the past, whereas it was reported that in 1988 annual production amounted to 2,000 to 4,000 t. HCBD has been generated as by-product in the production of tetrachloromethane and tetrachloroethene, which both have been produced in such large scale that enough HCBD could be generated to meet the industrial demand [Rossberg et al. 2006]. The commercial use of HCBD in Europe stopped in the late 1970s [UNECE 2007]. It was indicated that in France HCBD has been used in the past as fumigant on grapes to combat Phylloxera [INERIS 2005].

HCBD is not intentionally produced and commercially used anymore in Europe since the late 1970s according to the World Chlorine Council [UNECE 2007]. The substance is currently neither registered in the Swedish Products Register in Sweden [UNECE 2007], in the French inventory of the chemical products of the guide “Achats chimie parachemie” [INERIS 2005], nor are HCBD producers registered in the European chemical Substances Information System (ESIS) according to the German Federal Environment Agency [UBA 2006]. However, four facilities have been registered within ESIS as producing/importing HCBD as low production volume chemical [ESIS 2010].

Current sources and releases of HCBD in Europe result from past intentional uses as well as unintentional production during different processes. These are described in more detail in chapters 6.8.5.1 and 6.8.5.2.

It has been indicated by several sources that HCBD has been used as laboratory reagent ([Haskoning 2002], [ATSDR 1994], [WWF 2005], [INERIS 2005]). However, it is unknown if quantities of HCBD are still being used for this purpose [Haskoning 2002].

6.8.5.1 Past intentional uses

Past intentional uses in Europe include uses as pesticide/biocide, especially in the agricultural sector, as chemical intermediate in the manufacture of different products and purification of gas streams, and as liquid in (electrical) equipment. These historic uses are described in more detail below.

Agricultural sector

The use of HCBD in the agricultural sector has been the most recent one reported. It was indicated that in France HCBD has been used for specific phytosanitary purposes in the past prior to 2003 [INERIS 2005]. No information is available on whether stockpiles do exist within the EU.

According to information available, HCBD has been applied in a wide range in order to combat soil pests including use as fumigant ([WWF 2005], [UNECE 2006], [UNECE 2007], [Haskoning 2002], [ATSDR 1994]), as seed dressing, as fungicide ([UNECE 2006], [UNECE 2007], [INERIS 2005]) and - in more general terms - as pesticide ([WHO 2003], [IARC 1999]) as well as biocide [UBA 2006].

Generally, information in this sector is scarce. The application of HCBD as fumigant in viticulture has been mentioned by several sources, where HCBD has been used for treatment of grapes to combat the pest Phylloxera. It was indicated that HCBD has been used for this purpose within Europe in France, Greece, Spain and Italy, as well as outside the EU in Russia and Argentina. Russia has been reported to

be one of the major users of HCBd for this purpose, using application rates of 100-350 kg/ha [UNECE 2007]. In France it has been used in large quantities as fungicide for treatment of grapes, but since 2003 use has stopped. It is not anymore included in phytosanitary products listed in the French inventory ACTA and neither is produced as sub-/by-product in the manufacture of other phytosanitary products [INERIS 2005]. No data is available on the specific situation in Greece, Spain and Italy. HCBd is not on the Community list of authorized active substances according to Directive 91/414/EEC and is therefore not authorised for use as plant protection product in the EU.

According to the German Federal Environment Agency, HCBd has been applied in the past as biocide for prevention of algae production in industrial reservoirs and cooling water systems [UBA 2006].

Chemical intermediate in industrial manufacture

An important application of HCBd has been its use as chemical intermediate in several industrial manufacturing processes.

HCBd has been applied as intermediate in the synthesis of rubber compounds ([WWF 2005], [UBA 2006], [Euro Chlor 2004], [ATSDR 1994], [Haskoning 2002]). Several sources also indicate that HCBd was used – in small amounts – as solvent for rubber [Euro Chlor 2004], other polymers [UNECE 2006/7] and elastomeric plastics [UBA 2006]. One source indicated that HCBd still can be present in rubber compounds in marginal amounts according to the national association on rubber and polymers in France (Syndicat National du Caoutchouc et des Polymères, SNCP 2004) [INERIS 2005].

Further, HCBd has been applied as chemical intermediate during the production of lubricants ([ATSDR 1994], [Haskoning 2002], [WWF 2005], [INERIS 2005], [UBA 2006]).

HCBd has also been used in small amounts in the manufacture of chlorofluorocarbons ([WWF 2005], [Euro Chlor 2004]). Chlorofluorocarbons contribute to ozone depletion. The manufacture of such compounds is being phased out in accordance to the Montreal Protocol.

The application of HCBd in the manufacture of aluminum and graphite rods has been indicated by several sources ([INERIS 2005], [UBA 2006]). It was noted that this could also have been a former use of the chemical compound hexachlorobenzene (HCB) [UNECE 2007].

Purification of gas streams

One of the main applications of HCBd was the recovery of “snift”, a chlorine containing gas in chlorine plants, which has been the largest use of HCBd in the USA prior to 1975 ([WWF 2005], [Euro Chlor 2004], [UNECE 2006/7]). It has been assumed that this has also been the main application outside the USA [UNECE 2006/7].

HCBd has been applied as washing liquor for purifying gas streams by removing hydrocarbons [UNECE 2006/7]. According to Euro Chlor hydrocarbons consisting of four carbon atoms and higher carbon content (C4 and higher hydrocarbons) have been removed [Euro Chlor 2004]. Other sources report in a more general way that certain volatile organic components have been removed by using HCBd as

washing liquor [UNECE 2006/7], that HCBd was used as adsorption agent for gaseous contaminants [UBA 2006] as well as solvent in chlorine gas production [WHO 2003].

Electrical equipment

HCBd has been historically used for different technical applications in electrical equipment. It has been applied as heat-transfer liquid ([Haskoning 2002], [WWF 2005], [UNECE 2006/7], [Euro Chlor 2004]) particularly in transformers ([INERIS 2005], [UBA 2006], [IARC 1999]). HCBd has also been used as hydraulic fluid and liquid in gyroscopes ([WWF 2005], [UBA 2006], [INERIS 2005], [UNECE 2007], [Euro Chlor 2004]) and as insulating liquid [INERIS 2005].

6.8.5.2 Current unintentional production & releases

At the present the most relevant source of HCBd - besides releases due to former uses - is assumed to be unintentional production during different processes. HCBd still can be produced unintentionally as by-product during manufacture of chlorinated chemicals, from where it can enter the waste stream or be released to the environment. Other processes reported during which HCBd can be unintentionally produced and released is the manufacture of Magnesium as well as other fields in the non-chemical industry. It was also reported that HCBd can be produced during the manufacture of plastics and released with motor vehicle emissions [WWF 2005]. Once unintentionally produced and not adequately handled by available waste management options, there have been several pathways identified through which HCBd can be released to the environment, which will be explained in more detail below. Therefore, it is essential that once HCBd has entered the waste stream, it is appropriately managed in order to prevent environmental releases.

Industrial processes

By-product formation during the manufacture of chlorinated chemicals, from where HCBd can enter the waste stream or released to the environment, has been cited most often. HCBd can be produced during the manufacture of chlorinated solvents as trichloroethylene (trichloroethene), tetrachloroethylene (tetrachloroethene, perchloroethylene or perchloroethene) and tetrachloromethane (carbontetrachloride), which have been mentioned as the most relevant chemicals ([EuroChlor 2004], [INERIS 2005], [WWF 2005], [UBA 2006], [UNECE 2006], [UNECE 2007]). The production of tri-, tetrachloroethene and tetrachloromethane is performed via chlorolysis. It can be distinguished between high- and the low pressure chlorolysis. According to Euro Chlor, the only remaining significant source is the low pressure chlorolysis for the combined production of tetrachloroethylene and tetrachloromethane. The residues of the low pressure chlorolysis contain 0.2-0.5 % HCBd in the raw material, of which after further distillation the residue contains 7-10 % HCBd. The HCBd containing residues are generally destroyed on-site at high temperatures by incineration at 1,200 °C or internally recycled through high temperature chlorolysis to tetrachloromethane and tetrachloroethylene [Euro Chlor 2004]. It was reported that high pressure chlorolysis used for production of tetrachloromethane results in no HCBd contained within the raw material [Haskoning 2002]. Nowadays, this process is used for the combined synthesis of tetrachloroethylene and tetrachloromethane.

The possibility of HCBd by-product formation during the production of other substances as indicated in other reports ([WWF 2005], [INERIS 2005]), e.g. vinyl chloride, allyl chloride and epichlorohydrin, is extremely unlikely due to prevalent process conditions during their production (according to [Euro Chlor 2004]).

In 2002 eight European countries indicated to produce chlorinated solvents, among them Belgium, France, Germany, Norway, Poland, Slovakia, Switzerland and the UK [Haskoning 2002]. However, the demand of chlorinated solvents in Western Europe has been generally decreasing the past years [INERIS 2005].

It has also been indicated by several sources that production of Magnesium is related to the generation of HCBd due to unintentional by-product formation ([Euro Chlor 2004], [UBA 2006], [UNECE 2006], [UNECE 2007]. According to the German Federal Environment Agency 15 to 20 g HCBd is produced per tonne Magnesium [UBA 2006]. Deutscher et al. reported on organochlorine formation in Magnesium electrowinning cells with HCBd being one of the major components, which can be considered as similar to electrolytic processes taking place in large-scale magnesium production plants [Deutscher et al. 2001]. They indicated that depending on the current density applied during electrolysis up to approximately 3 g HCBd per tonne Magnesium was formed on a laboratory scale.

Another industrial branch, which has been reported to unintentionally produce HCBd as by-product, is the plastic industry. It was stated that unintentional production of HCBd can occur during the manufacture of plastics [WWF 2005]. One source indicated that HCBd still can be present in rubber compounds in marginal amounts according to the national association on rubber and polymers in France (Syndicat National du Caoutchouc et des Polymères, SNCP) [INERIS 2005].

Waste and waste management related sources and others

It was further reported that in other branches not related to the chemical industry unintentional production of HCBd might take place [Euro Chlor 2004]. They refer to Lenoir et al. 2001, who observed by-product formation of organochlorine compounds including HCBd from incineration processes of acetylene, which was indicated as being present in flames of all incineration processes [Lenoir et al. 2001]. Releases from incinerators have been reported by [WWF 2005] and [INERIS 2005]. According to the Association of Plastic Producers (Syndicat des Producteurs de Matières Plastiques SPMP) in France in 2003, HCBd has been detected in the effluents of an incinerator eliminating chlorine residues [INERIS 2005]. It was also reported that HCBd releases during refuse combustion can take place and that combustion sources of HCBd are similar to those of dioxins, furans and hexachlorobenzene (HCB) [CEPA 1999]. Also the German Federal Environment Agency indicated that HCBd can be formed as by-product and/or waste product during incineration processes [UBA 2006]. Also emissions from motor vehicles have been indicated as source for HCBd [WWF 2005].

According to available literature, these sources have been identified as sources of HCBd due to unintentional production. In summary, these sources are generally related to specific industrial activities in the chlorine, plastic and magnesium industry but also incineration processes have been identified to be a possible source for HCBd generation and release. Related to these processes, especially the

industrial processes, HCBd formed during manufacture of certain substances may enter the waste stream. Appropriate management is required in order to avoid adverse effects to humans and environment. Therefore, BREF documents are available, clearly laying down the conditions which have to be met from industrial site in order to prevent/minimise HCBd production and releases. However, HCBd has been detected in samples taken at urban waste water treatment plants in Germany with a concentration of 0.20 µg/L [UBA 2006]. Poland reported on existing concentration limits of HCBd in industrial sewage of 3.0 mg/L [PL Questionnaire 2010]. Also emissions from former waste dumps have been reported. In the UK landfill storage of heavy fractions from the production of chlorinated organic substances resulted in leachates and secondary emissions leading to adverse effects in regard to human health and environment. HCBd has been detected in the underlying strata, groundwater and indoor air in properties close to the disposal site [COT 2000].

6.8.6 Relevance of the Occurrence of HCBd in Europe

Within Europe data on the amount of HCBd releases is available in the European Pollutant Release Transfer Register (E-PRTR) to which specific European facilities are obliged to report releases of HCBd exceeding 1 kg/y to air or to land. According to the data reported under E-PRTR, HCBd releases occurred only to water and amounted to 140 kg in 2008 (149 kg in 2007). The industrial sectors responsible for these releases are the chemical industry producing basic organic materials as nitrogenous hydrocarbons (60 kg) and basic plastic materials (24 kg), together amounting to 84 kg HCBd released, as well as facilities in the field of waste and waste water management, i.e. waste water treatment plants (~29 kg) and hazardous waste disposal and/or recovery facilities (~27 kg) [E-PRTR 2008]. An overview of the distribution is shown in Figure 6-133.

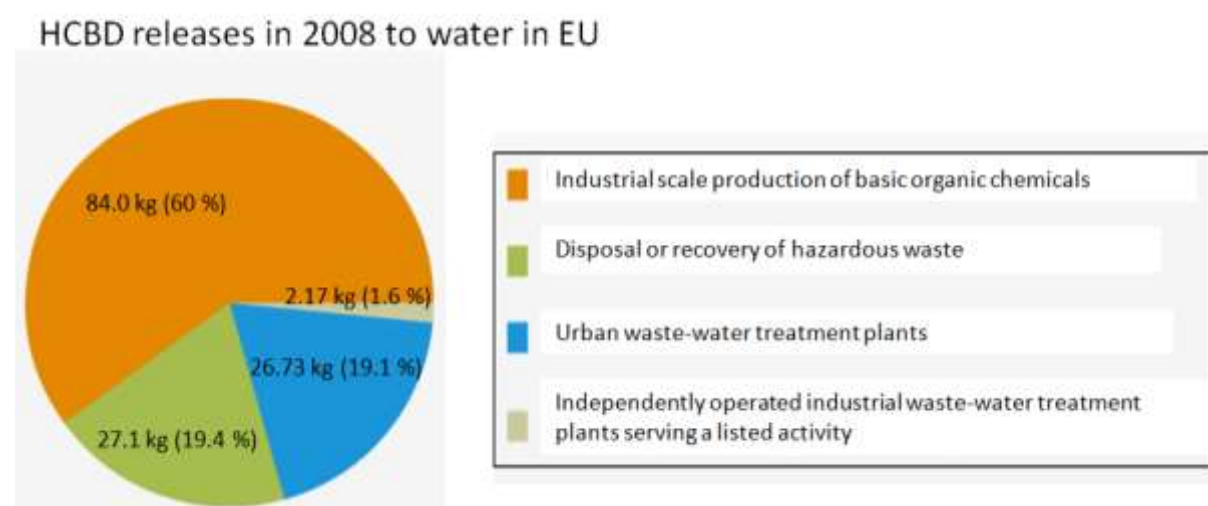


Figure 6-133: Overview of HCBd releases from facilities in Europe in 2008 [E-PRTR 2008]

6.8.7 Selection of Relevant Processes

Intentional production and use of HCBd is not relevant anymore within Europe. From the former intentional uses of HCBd only uses in electrical equipment, e.g. transformers and gyroscopes, and

existence of stockpiles of HCBd containing compositions intended for agricultural purposes might be of relevance depending on remaining amounts of the commercial products still in use. However, they have been assumed to be insignificant, since regulations for electrical equipment leading to their decrease are already in place. Correspondingly, the German Federal Environment Agency assumed already in 2002 that disposal of HCBd containing products as hydraulic liquids, cooling and adsorption liquids only play a minor role in regard to HCBd emissions [UBA 2002]. It can be considered that the quantities of HCBd contained in products have decreased since then, because most of them are already disposed of by now.

The focus in the present report is therefore on processes leading to unintentional production of HCBd. Industrial processes are considered the most important sources of unintentional production of HCBd. Identified unintentional industrial HCBd sources are the plastic industry, magnesium industry, hazardous waste incineration chlor chemical industry and waste water treatment plants and the corresponding sewage sludges.

Plastic industry

HCBd can be unintentionally produced as by-product during the manufacturing of plastics ([INERIS 2005], [WWF 2005], [UBA 2006]). It has been reported by the French Association on Rubber and Polymers (Syndicat National du Caoutchouc et des Polymeres, SNCP) that HCBd can be present in some rubbers but amounts are marginal [INERIS 2005]. Furthermore, it has been reported by the chemical industry to the European PRTR that emission to water have been registered from the production of basic plastics materials (polymers, synthetic fibres and cellulose based fibres) amounting to 24 kg in 2008 [E-PRTR 2008]. No information is available on which specific basic plastic materials are connected to the unintentional production of HCBd and further during which processes related to their production HCBd might be produced. Consequently, mass- and waste flow of HCBd related to the plastic industry cannot be specifically allocated.

Regarding the available information on plastic production statistics within Europe, it could be assumed that HCBd production takes place in manufacturing process of plastics in which the use of chlorine is involved, e.g. Polyvinylchloride (PVC).

As available information on HCBd related to plastics manufacturing cannot be allocated to specific materials and production processes within the plastic industry sector and information on potential arising of waste containing HCBd is lacking, it is not possible to perform reasonable quantitative assessments of substance or waste flows. However, it cannot be ruled out that certain waste fractions from the plastic industry sector may have relevant HCBd contents. As no further and more detailed information of the plastic industries regarding HCBd is available this sector is not further investigated in this report and might therefore be underestimated.

Magnesium industry

Magnesium can be produced either by electrolysis or a thermal reduction process [BREF NFM 2001]. Organochlorine formation, among them HCBd, have been reported by Deutscher et al. during

magnesium production using the electrolytic process [Deutscher et al. 2001]. For this manufacturing process a high temperature melt of 650-700 °C usually containing a high amount of chlorides is used as well as graphite anodes. The high temperature enables the carbon and gaseous chlorine to directly react and form a wide range of organochlorines, among them HCBd. In the study conducted by Deutscher et al. the organochlorines generated in the process were identified. HCBd has been identified as one major component [Deutscher et al. 2001].

Although magnesium is produced by thermal processes in a number of small factories, electrolytic processes have been favoured for large scale production plants. The relevant processes and aspects in magnesium production are described in [BREF NFM 2009 draft] and [UNEP BAT-BEP 2004].

The formation of HCBd regarding to the produced magnesium amount varies from 1.5-2.7 g HCBd/t magnesium (electrolytic process) [Deutscher et.al.2001], 15-20 g HCBd/ tonne magnesium [UBA 2006] or an hourly HCBd formation at the anode (off gas) at 20.9 µg/h [Wendt et al. 1991].

To give an overview of the production quantities in European countries including Eastern Europe, the production quantities have been summarised in the following table, whereas it is not sure which manufacturing method is applied by the other countries

Table 6-85: Overview of magnesium production quantities in European Countries [IMA 2009]

Country	Total Mg production [t/y]	Source
France	0	[IMA 2009]
Kazakhstan	20,000	[IMA 2009]
Norway	0	[IMA 2009]
Russia	30,000	[IMA 2009]
Ukraine	3,000	[IMA 2009]
Serbia	2,000	[IMA 2009]
total	55,000	

However, it can be concluded that there is currently no primary production of magnesium neither based on the electrolysis of magnesium nor on other processes within EU 27. Therefore, no unintentional production of HCBd is expected via the relevant process of magnesium electrolysis. Related emissions and waste arising due to production of magnesium do not occur.

Hazardous Waste Incineration (HWI)

Hazardous waste incineration is usually carried out in rotary kiln incinerators or in static furnaces. The relevant processes and aspects of waste incineration are described in detail in the corresponding BAT reference document [BREF WI 2006].

No information has been found in the literature that HCBd is contained in solid residues resulting from hazardous waste incineration.

Under the E-PRTR releases of HCBd from hazardous waste treatment has been reported by two facilities in 2008. The released quantity amounts to 27.1 kg which was completely released to water from two companies [E-PRTR 2008]. HCBd contamination data for residues from hazardous waste incineration are not available. Related emissions and waste arising due to hazardous waste incineration cannot be established.

During waste water treatment HCBd probably accumulates in sewage sludge. (see chapter 6.8.10 on sewage sludge)

Chemical Industry – Production of chlorinated solvents

From the industrial processes, production of chlorinated chemicals, especially of tri- and tetrachloroethene as well as tetrachloromethane is a potential source of HCBd and is therefore investigated further in the corresponding chapter. It has been reported that HCBd containing wastes, e.g. residues obtained during low-pressure chlorolysis in the manufacture of chlorinated solvents, can be almost completely destroyed (> 99.9 %) due to incineration at high temperature, i.e. 1,200 °C, which has been reported by several sources [Euro Chlor 2004; INERIS 2005].

Waste water treatment plants – sewage sludge

Since waste water treatment plants are likely to be a secondary source for HCBd emissions due to accumulation of unquantified amounts of HCBd, generated sewage sludge will be investigated as well.

6.8.8 Substance Flow of HCBd

Based on the available data the following schematic substance flow diagram for HCBd has been established. Figure 6-52 gives an overview of the major sources for HCBd containing waste in the EU. Since there is no intentional use or production of HCBd within the EU, the flow diagram is related to unintentional production and related releases of HCBd within the EU.

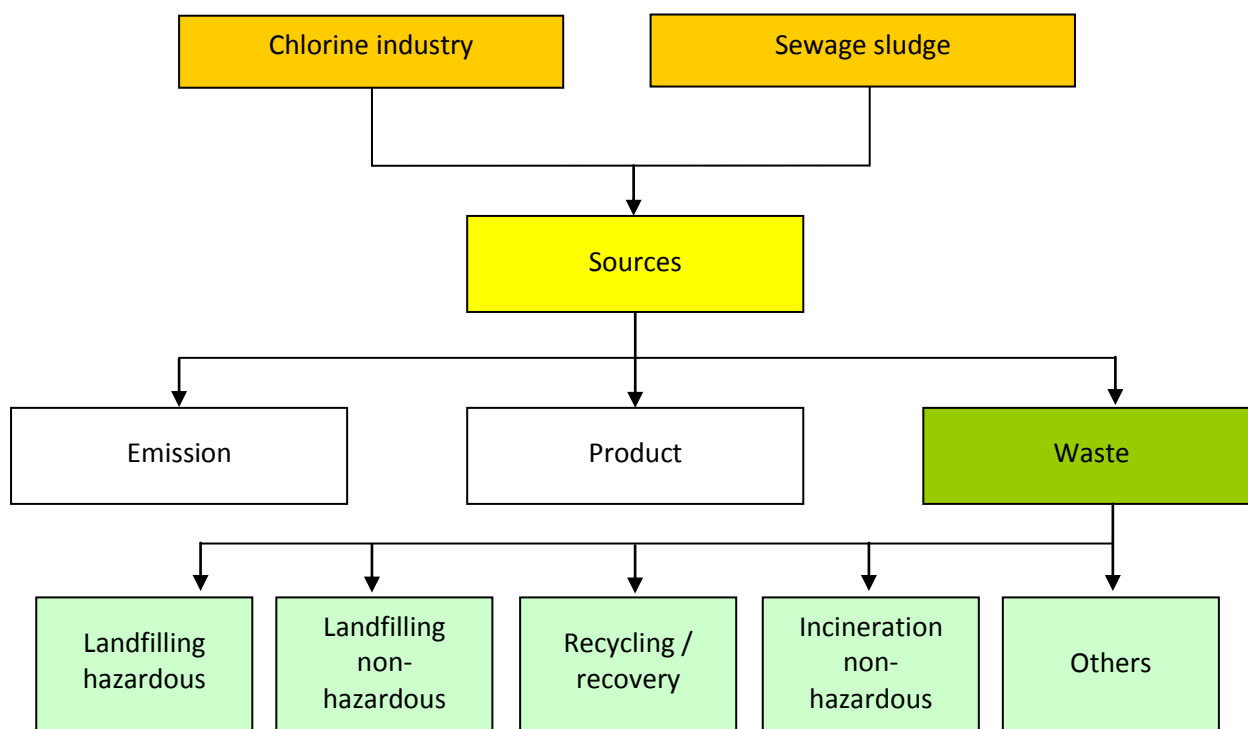


Figure 6-134: Schematic HCBd substance flow

The following sub-chapters summarise relevant information on the selected sources of HCBd containing waste.

6.8.9 Chemical Industry – Production of chlorinated solvents

A) Background

Chemical production has been deemed to be one of the most important HCBP sources at present. By-product formation during the manufacture of chlorinated chemicals has been cited the most often. HCBP can be produced during the manufacture of chlorinated solvents as trichloroethylene (trichloroethene), tetrachloroethylene (tetrachloroethene, perchloroethylene or perchloroethene) and tetrachloromethane (carbon tetrachloride), which have been mentioned as the most relevant chemicals ([Euro Chlor 2004], [INERIS 2005], [WWF 2005], [UBA 2006], [UNECE 2006], [UNECE 2007]). The production of tri-, tetrachloroethene and tetrachloromethane is carried out via chlorolysis.

Chlorolysis, also referred to as chlorinolysis, is the process of inducing a rupture of carbon–carbon bonds through a combination of saturation or near saturation of the organic reactant with chlorine at elevated temperature. Reaction of C2 and C3 hydrocarbons with excess chlorine at high temperatures can cleave the C–C bonds of the hydrocarbon to give chlorinated derivatives of shorter chain length. Aromatic and aliphatic chlorohydrocarbons containing up to six carbons can be converted to carbon tetrachloride by chlorinolysis, though presaturation of double bonds in the feedstock with chlorine is usually required in these cases. Generally, because these reactions are carried out in a chlorine-rich environment, the products are carbon tetrachloride and perchloroethylene. These two products are in equilibrium and their ratio is determined by the reaction conditions as illustrated in following equation [ECT 2004].



It can be distinguished between high- and the low pressure chlorolysis. According to Euro Chlor, the only remaining significant source is the low pressure chlorolysis for the combined production of tetrachloroethylene and tetrachloromethane. Low pressure chlorolysis results in 0.2-0.5 % HCBP being contained in the raw product, of which after further distillation the residue contains **7-10 %** HCBP. In total, hydrocarbon production wastes typically contain **33-80 %** HCBP according to the US EPA. These wastes are disposed of by various methods, whereas over the last decade, disposal practices have shifted from landfill to incineration. Incineration reportedly achieves about 99.9% destruction efficiency of HCBP [Euro Chlor 2004]. Based on information by DOW the HCBP containing residues are generally disposed of by incineration at 1,200 °C [Haskoning 2002].

It was reported that high pressure chlorolysis used for production of tetrachloromethane results in no HCBP contained within the raw material [Haskoning 2002]. It has to be noted that tetrachloromethane production has been significantly decreasing since CFC11 (FCKW trichlorofluoromethane) and CFC12 (FCKW dichlorodifluoromethane) are being phased out, for which tetrachloromethane has been used as feedstock [Haskoning 2002].

Relevant technologies for waste water and waste gas treatment are described in the BREF documents on Best Available Techniques in “Common Waste Water and Waste Gas Treatment / Management Systems in the Chemical Sector” [BREF WWCS 2003] and in the Large Volume Organic Chemical Industry [BREF LVOC 2003].

Due to the improved production processes it has been noted by Euro Chlor as unlikely that HCBd can be found as contaminant in tri- and tetrachloroethylene [INERIS 2005].

According to Lecloux, the possibility of HCBd by-product formation during the production of other substances as indicated in several other reports [WWF 2005] [INERIS 2005], e.g. vinyl chloride, allyl chloride and epichlorohydrin, is extremely unlikely due to prevalent process conditions during their production [Euro Chlor 2004].

B) Process input and output

The following flow chart shows schematically where HCBd may arise in chemical production processes.

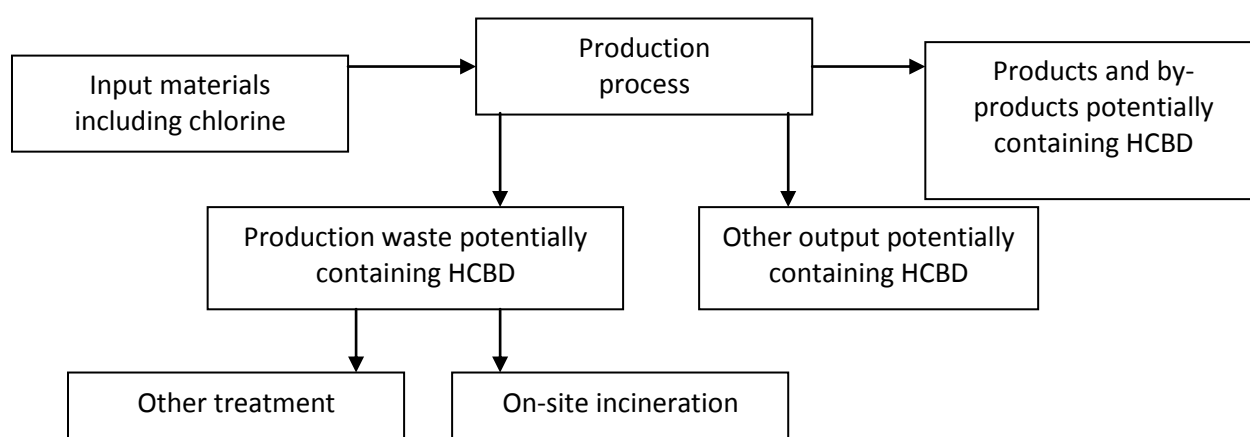


Figure 6-135: Input and output scheme for chemical industry

Output directly resulting from production processes which is not incinerated as hazardous waste is expected to be very low. The formation of HCBd as unintentional by-products is very limited under present production conditions.

However, it has to be noted that the information available is very scarce. Only marginal information on the technical process of the manufacture of the chlorinated solvents tri-, tetrachloroethylene and tetrachloromethane has been identified. Consequently, assumptions have been made based on the literature available in order to allocate a model for the generation of the HCBd mass flow and waste stream as realistic as possible. According to literature available, the following output fractions have been assumed to be potentially relevant for the HCBd flow:

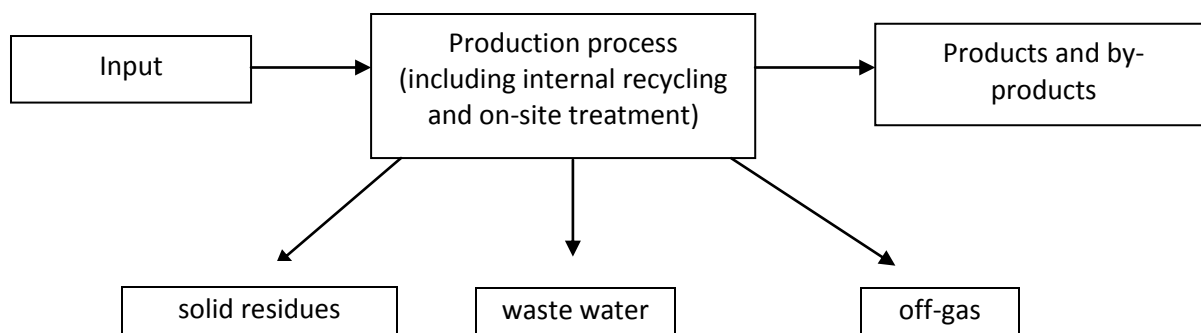


Figure 6-136: Relevant input and output of chlorolysis process

Input:

- Aromatic and aliphatic chlorohydrocarbons containing up to six carbons (in case of unsaturated compounds in the feedstock pre-saturation is required), e.g. organic residues from 1,2 ethylene dichloride production (EDC) [ECT 2004] [BREF LVOC 2003]
- Chlorine
- Temperature/pressure [ECT 2004]

Output:

- Products: tetrachloromethane and tetrachloroethylene (equilibrium; ratio is determined by reaction conditions)
- Solid residues (from hazardous waste incineration)
- Off-gas (from hazardous waste incineration)
- Waste water and residues from waste water treatment

C) Generation factors for product/waste and other output (air, water)

According to Markovec et al. the reaction yield of the chlorolysis for the combined production of tetrachloroethylene and tetrachloromethane can be as high as 96 % in regard to the chlorine consumption, i.e. 4% are subject to losses during the manufacture and purification steps [Markovec et al. 1984]. Assuming that these 4 % contribute to the distillation residues, inter alia containing HCBd, which are obtained after the purifying distillation step, this leads to ~9.97 kt distillation residues per year. These are usually either internally recycled as chlorolysis feedstock or incinerated on-site at 1,200 °C. Information on the ratio between internal recycling and incineration is not available.

According to the E-PRTR, in 2008 60 kg HCBd have been emitted to water from one chemical plant in France (Perstop Tolonate France) manufacturing nitrogenous hydrocarbons [E-PRTR 2008]. It is unclear in which way the production of nitrogenous hydrocarbons is related to the unintentional production of HCBd. According to the E-PRTR, during the manufacture of nitrogenous compounds substances as tri-, tetrachloroethene and tetrachloromethane have been released in 2008 besides HCBd. This leads to the assumption that these chlorinated solvents are either involved or produced (intentionally or unintentionally) during the production of the nitrogenous hydrocarbons.

D) Contamination data for output

It has been indicated by several sources that HCBd is contained in the heavy fraction of the purifying distillation step of the raw material obtained after the chlorolysis process in amounts varying from 7-10 % according to the BUA ('Beratergremium für umweltrelevante Altstoffe', Advisory Committee on Existing Chemicals of Environmental Relevance of the German Chemical Society (GDCh)) [INERIS 2005; Haskoning 2002].

No data have been identified on HCBd amounts in solid residues.

No data have been identified on HCBd amounts in off-gas.

Several sources indicated that HCBd can be found in effluents from chemical manufacturing plants ([WHO 2003], [California EPA 2000], [INERIS 2005], [Euro Chlor 2004]). According to the WHO, HCBd concentrations in effluents from chemical manufacturing plants of 6 µg/L have been reported [WHO 2003]. In Belgium the threshold limit for HCBd in effluents may not exceed 2.0 µg/L [BE Staatsblatt 2003]. In Poland the threshold value for HCBd in industrial sewage has been set equal to 3.0 and 1.0 mg/L on a daily and monthly average, respectively [PL Questionnaire 2010]. It has been reported that HCBd could not be detected in samples taken at Greek sewage treatment plants [Katsoyannis et al. 2007]. However, it has been reported by the industry producing basic organic chemicals to the European PRTR that 84 kg have been emitted to water in 2008 [E-PRTR 2010].

E) Activity data

According to the European chemical Substances Information System (ESIS) production of tri- (TrCE), tetrachloroethylene (TeCE) and tetrachloromethane (TCM) takes place in the following countries as indicated in the following table [ESIS 2010].

Table 6-86: Production sites of tri- (TrCE), tetrachloroethylene (TeCE) and tetrachloromethane (TCM) in Europe [ESIS 2010]

Country	Company	Town	TrCE	TeCE	TCM	Source
AT	NEUBER GES.M.B.H.	1060 WIEN	x	x	x	[ESIS 2010]
BE	SOLVAY S.A.	1050 BRUXELLES	x	x	x	[ESIS 2010]
FR	ATOCHEM	92080 PARIS LA DEFENSE	x	x	x	[ESIS 2010]
DE	DOW DEUTSCHLAND INC., WERK STADE	21683 STADE		x	x	[ESIS 2010]
	BASF AG	67056 LUDWIGSHAFEN	x			[ESIS 2010]
	HUELS AG	45764 MARL		x	x	[ESIS 2010]
	CELANESE GMBH	65926 FRANKFURT AM MAIN			x	[ESIS 2010]
	CHEMIEWERK NUENCHRITZ GMBH	01612 NUENCHRITZ			x	[ESIS 2010]
	HOECHST AG	65903 FRANKFURT/MAIN			x	[ESIS 2010]
	BRENNTAG INTERNATIONAL CHEMICALS GMBH	45472 MUELHEIM	x	x		[ESIS 2010]
	BRENNTAG AG	45427 MUELHEIM A. D. RUHR	x			[ESIS 2010]
	HELM AG	20097 HAMBURG		x		[ESIS 2010]
	WACKER - CHEMIE GMBH	84480 BURGHAUSEN	x	x		[ESIS 2010]
IT	ENICHEM S.P.A.	20124 MILAN	x	x	x	[ESIS 2010]
	AUSIMONT	20021 BOLLATE (MILANO)			x	[ESIS 2010]
	S.A.I. SOCIETA' APPROVV. INDUSTRIALI SPA	20090 PIEVE EMANUELE (MI)	x			[ESIS 2010]
	BRENNTAG SPA	20152 MILANO		x		[ESIS 2010]
NL	VOS B.V.	2404 HM ALPHEN AAN DEN RIJN	x	x	x	[ESIS 2010]
	AKZO NOBEL CHEMICALS B.V.	3800AE AMERSFOORT			x	[ESIS 2010]
	DOW BENELUX N.V. (BOTLEK)	3197 AA XA BOTLEK RT	x			[ESIS 2010]
	DE MONCHY INTERNATIONAL BV	3011JR ROTTERDAM			x	[ESIS 2010]
	NED. BENZOL MIJ. B.V.	3161 GR RHOON	x			[ESIS 2010]
	LEDUC CHEMIE B.V.	3821 AP AMERSFOORT	x	x		[ESIS 2010]
	PETRASOL B.V.	4200 AE GORINCHEM	x	x		[ESIS 2010]
ES	ERKIMIA,S.A.	08007 BARCELONA		x	x	[ESIS 2010]
	ARAGONESAS INDUSTRIAS Y ENERGÍA S.A.	28004 MADRID			x	[ESIS 2010]
UK	ICI CHEMICALS & POLYMERS LIMITED	WA7 4QF RUNCORN, CHESHIRE	x	x	x	[ESIS 2010]
	BRE – BUILDING RESEARCH ESTABLISHMENT	WD2 7 JR GARSTON, WATFORD	x			[ESIS 2010]
	INSTITUTE FOR TERRESTRIAL ECOLOGY	PE17 2LS CAMBRIDGESHIRE	x			[ESIS 2010]
	RHONE-POULENC CHEMICALS LIMITED	WD1 1QH WATFORD, HERTFORDSHIRE			x	[ESIS 2010]

It has not been indicated to which year the data refer. According to the E-PRTR ~70% of the total HCBd emissions originate from French facilities, which correspond to 60 kg emitted by Perstorp Tolonate France during production of nitrogenous hydrocarbons. (The other ~30% related to 24 kg emitted by Solvay during manufacture of plastic materials and 14.9 kg emitted by Arkema due to waste operations) [E-PRTR 2010]. Perstorp produces isocyanates, but also chlorine (besides soda) which could be relevant for the unintentional production and emission of HCBd reported to the E-PRTR. According to ESIS, the only French company producing the relevant chlorinated chemicals is Atochem, which is now Atofina. This is partly in agreement with information from [INERIS 2005], who indicate Atofina and Solvay to be the only French companies producing the chlorinated solvents. Further investigation confirmed that Solvay produces relevant solvents as tetrachloromethane and perchloroethylene in the facility located in

Taveaux³⁹. Concluding, it can be assumed that data according to ESIS is old, while information submitted by facilities to the E-PRTR should be treated with caution in regard to emission sources.

Based on data of the European Statistics Database EUROSTAT, production in the EU 27 of tetrachloromethane decreased from 2006 to 2009 from 39,864 t to 23,481 t and of tri- and tetrachloroethylene from 308,526 t to 215,781 t, making a total of 239,262 t/y in 2009.

According to Markovec et al. 1983 the production yield in the industry is about 96% and the remaining 4% end up as distillation residue. For the year 2009 this represents about 9,9692 kt of distillation residue.

F) Waste treatment

Waste streams resulting from the inadvertent production of hexachlorobutadiene as a byproduct of certain chlorinated hydrocarbons typically contain 33-80% hexachlorobutadiene. These wastes are disposed of by various methods. Over the last decade, disposal practices have shifted from landfilling to incineration. Incineration, which is considered the preferred method of disposal, reportedly achieves greater than 99.9% destruction efficiency. In 1982, approximately 68% of an estimated 27 million pounds of hexachlorobutadiene wastes were disposed of by incineration, 32% by deep well injection, and less than 0.2% by hazardous waste landfill operations according to the US EPA ([ATSDR 1994], [Euro Chlor 2004]).

Statistical data on different waste treatment options for hazardous waste from chemical industry are available at MS level for EU 27 [EUROSTAT 2010].

A certain share of the distillation residues containing HCBd is internally recycled. The remaining share is incinerated on-site. Information on the ratio between internal recycling and incineration is not available. Transfer of waste containing HCBd from facilities manufacturing basic organic chemicals are not reported under E-PRTR for 2008 [E-PRTR 2010]. The destruction efficiency of incineration is > 99.9%. As all releases of HCBd reported from waste management facilities under the E-PRTR are completely to water, it is assumed that the remaining share of up to 0.1% is released to water (see chapter sewage sludge 6.8.10). Information on HCBd present in solid residues from waste incineration is not available. An assessment of possible waste arising from HWI contaminated with HCBd is therefore not possible.

G) HCBd flow

In regard to the scarcity of information available different scenarios have been elaborated. These are listed in the following table.

³⁹ www.solvaychemicals.com

Table 6-87: HCBd amounts resulting considering different recycling/incineration ratios (1, 10 and 50%) and different destruction efficiencies for incineration (99.90, 99.95 and 99.99 %)

	Option I 1% incinerated	Option II 10% incinerated	Option III 50% incinerated
Amount distillation residues incinerated [t]	99.69	996.92	4,984.62
Destruction efficiency 99.9%			
Amount distillation residues destroyed [t]	99.59	995.93	4979.64
Amount distillation residues remaining [t]	0.10	0.99	4.98
7-10% HCBd contamination [kg]	6.98-9.97	69.78-99.69	348.92-498.46
Destruction efficiency 99.95%			
Amount distillation residues destroyed [t]	99.64	996.43	4,982.13
Amount distillation residues remaining [t]	0.05	0.49	2.49
7-10% HCBd contamination [kg]	3.49-4.98	34.89-49.85	174.46-249.23
Destruction efficiency 99.99%			
Amount distillation residues destroyed [t]	99.68	996.83	4,984.13
Amount distillation residues remaining [t]	0.01	0.09	0.49
7-10% HCBd contamination [kg]	0.70-1.00	6.98-9.97	34.89-49.85

Solid residue [t/y]			9969.25	
Contamination [%]			min	max
			7%	10%
HCDB in solid residue [t/y]			698	997
Option I 1 % incinerated			6.98	9.97
incineration efficiency	99.90%	destroyed [t/y]	6.97	9.96
		remained [t/y]	0.007	0.010
	99.95%	destroyed [t/y]	6.97	9.96
		remained [t/y]	0.003	0.005
	99.99%	destroyed [t/y]	6.98	9.97
		remained [t/y]	0.0007	0.001
Option II 10 % incinerated			69.8	99.7
incineration efficiency	99.90%	destroyed [t/y]	69.7	99.6
		remained [t/y]	0.070	0.100
	99.95%	destroyed [t/y]	69.7	99.6
		remained [t/y]	0.035	0.050
	99.99%	destroyed [t/y]	69.8	99.7
		remained [t/y]	0.007	0.010
Option III 50 % incinerated			349	498
incineration efficiency	99.90%	destroyed [t/y]	349	498
		remained [t/y]	0.349	0.498
	99.95%	destroyed [t/y]	349	498
		remained [t/y]	0.174	0.249
	99.99%	destroyed [t/y]	349	498
		remained [t/y]	0.035	0.050

At a European scale the estimated unintentional production of HCBd due to unintentional manufacture of chlorinated solvents as tri-, tetrachloroethylene as well as tetrachloromethane is difficult to quantify. Therefore, different scenarios have been elaborated with option I, II and III indicating that 1, 10 and 50% of the distillation residues are incinerated, respectively, with the rest being recycled by using it as feedstock for the chlorolysis process. The distillation residue waste is subject to hazardous waste incineration, whereof more than 99.9 % of the HCBd is destroyed. In above table, for each option destruction efficiencies of 99.90, 99.95 and 99.99 % have been taken into account. With a contamination value of the distillation residues from 7-10 % indicated in literature, for the different options and destruction efficiencies a range of possible remaining HCBd amounts with a minimum of 0.7 kg/y up to possible ~500 kg/y arise. The geometric mean is 18.7 kg/y.

It is assumed that the amount of HCBd not destroyed via incineration reaches the waste water, which is treated before emission. HCBd contained in sewage sludge is discussed in the corresponding chapter 6.8.10.

H) Waste flow for HCBd

On the basis of the literature available, no relevant wastes types contaminated with HCBd could be identified, except HCBd remaining in the heavy fraction of the distillation residues resulting from purification of raw products. No information on the ratio between the subsequent recycling and hazardous waste incineration process, to which distillation residues are subject, has been identified. HCBd contaminated distillation residues destined to hazardous waste incineration leads to > 99.9 % destruction of HCBd. HCBd not eliminated during incineration has been assumed to be completely emitted to water, since no indicators on HCBd present in solid residues from hazardous waste incineration, e.g. fly ash or bottom ash, or off-gas have been found in the literature. Further, no off-site transfers of HCBd contaminated wastes have been reported to the E-PRTR [E-PRTR 2010].

To conclude, no relevant waste types contaminated with HCBd have been identified arising during production of the chlorinated solvents tri-, tetrachloroethene and tetrachloromethane. Therefore, no waste flow has been established. However, it cannot be excluded that there exist relevant amounts and types of HCBd contaminated wastes entering the waste flow.

6.8.10 Sewage Sludge

A) Background

HCBd has been reported to be released to water in effluents from independently operated industrial waste water treatment plants (2.17 kg in 2008) as well as from urban waste water treatment plants (26.7 kg in 2008) [E-PRTR 2010]. The releases from industrial waste water treatment plants have been assumed to be related to industrial plants manufacturing nitrogenous hydrocarbons (see chapter 6.8.9 on chemical industry).

For Europe no data has been identified on contamination values of sewage sludge, but from other sources it has been indicated that HCBd can be found in sewage sludge ([Cai et al. 2007], [IPCS 1994]). HCBd tends to adsorb on particles. A certain share of the HCBd content of waste water accumulates in sewage sludge during waste water treatment. Since it has been reported that HCBd can not only be detected in effluents from industrial waste water treatment plants but also in sewage generated from urban waste water treatment plants [E-PRTR 2010], the total amount sewage sludge produced within the EU 27 has been taken into account for the allocation of HCBd mass- and waste flow.

B) Process input (raw material) and output (waste, air, water, product)

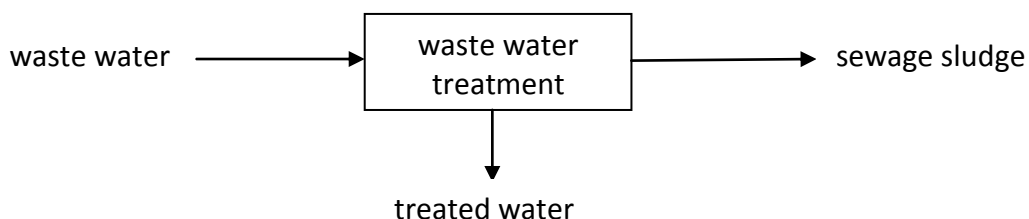


Figure 6-137: Relevant input and output of waste water treatment

Input: waste water

Output: sewage sludge and treated water

C) Generation factors for waste and other output (air, water)

No generation factor is used because the amount of sewage sludge produced from the wastewater depends on the composition of the wastewater and the treatment processes.

D) Contamination data for output (and input)

No European contamination values for HCBd in sewage sludge have been identified. Based on data available at international level, sewage sludge contamination in China from waste water treatment plants of averagely 39 µg/kg sludge (dry weight) has been reported. For most of the samples it has not been specified if the sludge originated from domestic or industrial sewage. Only for two sample locations ratios were indicated (domestic: industrial – Beijing: 1:1 with 61 µg/kg; Zhuhai: ~2:1 with

11 µg/kg) [Cai et al. 2007].

In the USA 15 µg/kg, for industrial sewage sludge has been reported [IPCS 1994], which has been taken as a best estimate for the mass flow calculations. For sewage sludge resulting from urban waste water treatment a value of 0.6 µg HCBd/kg urban sewage sludge has been used [IPCS 1994].

According to recent information received from the Environmental Protection Agency in Ireland (from a recent sampling study), no HCBd was measured above the limit of detection (<0.1 mg/kg) in any of the samples [EPA IRE 2011a].

E) Activity data

In all 27 Member States a total amount of approximately 11.5 Mio t of sewage sludge from waste water treatment has been calculated from EUROSTAT data (values derived from amount of sewage sludge from urban waste water treatment, from other waste water treatment and independently operated waste water treatment; see chapter 6.4.12 sewage sludge PeCB). For the calculations used for the allocation of the mass- and waste flow of HCBd only the production values for sewage sludge resulting from urban waste water treatment indicated in table Table 6-88 have been taken into account (total 9.82 mio t/y). For sewage sludge resulting from industrial waste water treatment generated during the manufacture of chlorinated solvents no information on quantities is available and therefore could not be taken into account.

Table 6-88: Annual amount of urban sewage sludge generated in the EU 27 with relevance for the HCBd mass- and waste flow

Country	Amount sewage sludge resulting from urban waste water treatment [mio t/y]	Reference
AT	0.26	[EUROSTAT 2010]
BE	0.11	[EUROSTAT 2010]
BG	0.04	[EUROSTAT 2010]
CY	0.01	[EUROSTAT 2010]
CZ	0.17	[EUROSTAT 2010]
DE	2.05	[EUROSTAT 2010]
DK	0.00	[EUROSTAT 2010]
EE	0.03	[EUROSTAT 2010]
ES	1.07	[EUROSTAT 2010]
FI	0.16	[EUROSTAT 2010]
FR	1.06	[EUROSTAT 2010]
GR	0.13	[EUROSTAT 2010]
HU	0.18	[EUROSTAT 2010]
IE	0.09	[EUROSTAT 2010]
IT	1.06	[EUROSTAT 2010]
LT	0.08	[EUROSTAT 2010]
LU	0.01	[EUROSTAT 2010]
LV	0.02	[EUROSTAT 2010]
MT	0.00	[EUROSTAT 2010]

Country	Amount sewage sludge resulting from urban waste water treatment [mio t/y]	Reference
NL	0.37	[EUROSTAT 2010]
PL	0.53	[EUROSTAT 2010]
PT	0.24	[EUROSTAT 2010]
RO	0.10	[EUROSTAT 2010]
SE	0.21	[EUROSTAT 2010]
SI	0.02	[EUROSTAT 2010]
SK	0.06	[EUROSTAT 2010]
UK	1.77	[EUROSTAT 2010]
EU 27	9.82	

F) Waste treatment

The relevant treatment of sewage sludge includes land filling, incineration, application on land (agricultural use and compost) and other options. The amount of sewage sludge undergoing the different treatments is specific for each country and for each type of sewage sludge. In Table 6-43 values in % for each Member State are shown for the treatment of urban sewage sludge, which have been derived from EUROSTAT data for each country as far as available (not available for PT and DK).

Table 6-89: Treatment of urban sewage sludge in Member States (PT, DK: no data; MT: no production)

Country	Landfill [%]	Incineration [%]	Application to land [%]	Others [%]	Reference
AT	10	38	44	7	[EUROSTAT 2010]
BE	8	32	19	41	[EUROSTAT 2010]
BG	53	0	15	33	[EUROSTAT 2010]
CY	0	0	63	38	[EUROSTAT 2010]
CZ	5	0	78	16	[EUROSTAT 2010]
DE	0	47	53	0	[EUROSTAT 2010]
EE	17	0	14	69	[EUROSTAT 2010]
ES	16	4	65	16	[EUROSTAT 2010]
FI	6	0	92	2	[EUROSTAT 2010]
FR	21	17	60	3	[EUROSTAT 2010]
GR	98	0	0	2	[EUROSTAT 2010]
HU	29	0	68	3	[EUROSTAT 2010]
IE	6	0	72	23	[EUROSTAT 2010]
IT	42	3	44	11	[EUROSTAT 2010]
LT	12	0	42	46	[EUROSTAT 2010]
LU	33	0	67	0	[EUROSTAT 2010]
LV	0	0	43	57	[EUROSTAT 2010]
NL	4	68	13	15	[EUROSTAT 2010]
PL	23	0	23	53	[EUROSTAT 2010]
RO	44	0	0	56	[EUROSTAT 2010]
SE	11	0	50	39	[EUROSTAT 2010]
SI	43	24	19	14	[EUROSTAT 2010]
SK	16	0	70	14	[EUROSTAT 2010]
UK	5	16	70	9	[EUROSTAT 2010]
EU 27	21	10	45	24	[EUROSTAT 2010]

Country	Landfill [%]	Incineration [%]	Application to land [%]	Others [%]	Reference
(average)					

G) HCBd flow

Member State specific percent values for urban sewage sludge have been used for the estimation of HCBd transferred to the different treatment stations from the total amount of sewage sludge produced in each country relevant for the HCBd mass- and waste flow. For Portugal the European average value for the sludge treatment has been applied. For sewage sludge resulting from the treatment of industrial waste water generated during the manufacture of chlorinated solvents no calculations were conducted since no information on sewage sludge quantities is available. The values derived for sewage sludge resulting from urban waste water treatment are given in Table 6-90.

Table 6-90: HCBd mass flow for sewage sludge in EU 27

Country	Total mount sewage sludge resulting from urban waste water treatment [mio t/y]	Amount HCBd [kg/y]	HCBd landfilled [kg/y]	HCBd incinerated [kg/y]	HCBd applied to land [kg/y]	HCBd treated in other ways [kg/y]
AT	0.26	0.15	0.02	0.06	0.07	0.01
BE	0.11	0.07	0.01	0.02	0.01	0.03
BG	0.04	0.02	0.01	0.00	0.00	0.01
CY	0.01	0.00	0.00	0.00	0.00	0.00
CZ	0.17	0.10	0.01	0.00	0.08	0.02
DE	2.05	1.23	0.00	0.58	0.65	0.00
DK	0.00	0.00	0.00	0.00	0.00	0.00
EE	0.03	0.02	0.00	0.00	0.00	0.01
ES	1.07	0.64	0.10	0.02	0.41	0.10
FI	0.16	0.10	0.01	0.00	0.09	0.00
FR	1.06	0.64	0.13	0.11	0.38	0.02
GR	0.13	0.08	0.07	0.00	0.00	0.00
HU	0.18	0.11	0.03	0.00	0.08	0.00
IE	0.09	0.05	0.00	0.00	0.04	0.01
IT	1.06	0.63	0.26	0.02	0.28	0.07
LT	0.08	0.05	0.01	0.00	0.02	0.02
LU	0.01	0.01	0.00	0.00	0.00	0.00
LV	0.02	0.01	0.00	0.00	0.01	0.01
MT	0.00	0.00	0.00	0.00	0.00	0.00
NL	0.37	0.22	0.01	0.15	0.03	0.03
PL	0.53	0.32	0.08	0.00	0.07	0.17
PT	0.24	0.14	0.00	0.00	0.00	0.14
RO	0.10	0.06	0.03	0.00	0.00	0.03
SE	0.21	0.13	0.01	0.00	0.06	0.05
SI	0.02	0.01	0.01	0.00	0.00	0.00
SK	0.06	0.03	0.01	0.00	0.02	0.00
UK	1.77	1.06	0.06	0.17	0.74	0.09
EU 27	9.82	5.89	0.86	1.13	3.05	0.85

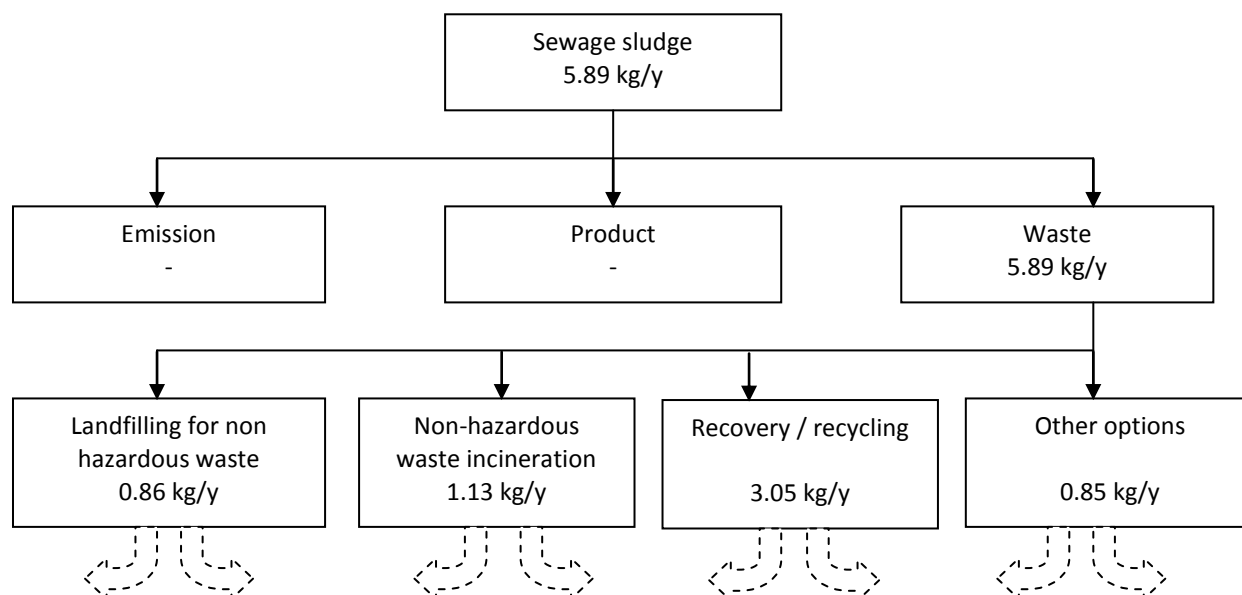


Figure 6-138: Assessment of the HCBd flow related to sewage sludge

H) Waste flow for sewage sludge

In order to illustrate the amounts of residues generated in EU 27 and the importance of currently used disposal/recovery operations which are of high relevance in the evaluation of impacts of proposed limit values, a waste flow has been drawn up in the following figure as far as possible. Mean contaminations and concentration ranges for each of the residues have been included for better interpretation. It has to be noted that due to the lack of information on quantities of industrial sewage sludge generated during waste water treatment from production of chlorinated solvents the indicated waste flow represents a provisional

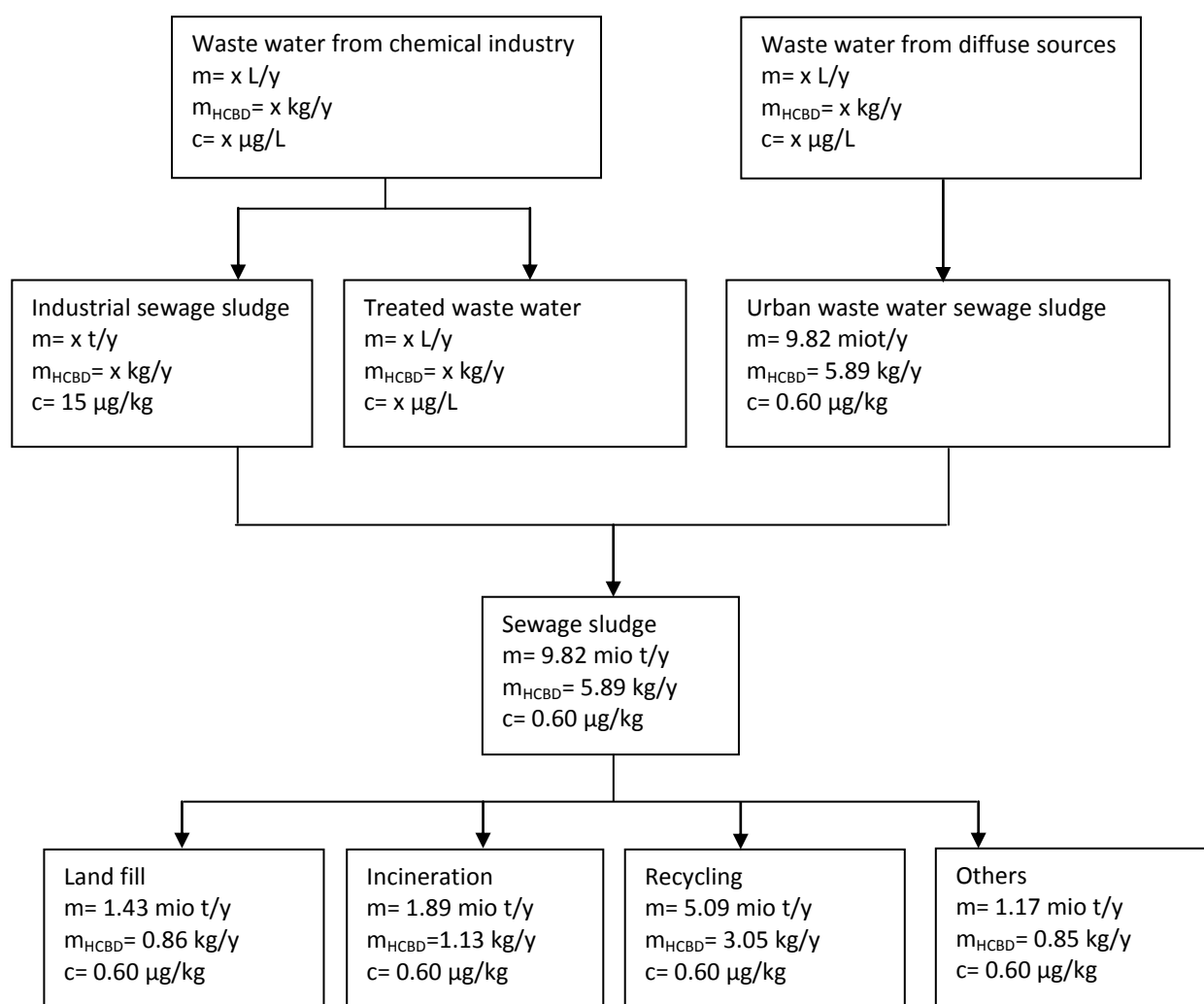


Figure 6-139: Waste flow for sewage sludge in EU 27

6.8.11 References HCBd

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6.9 PCN flows

6.9.1 Use and Production

Polychlorinated naphthalenes (PCNs) have historically been used for several purposes, which are cable insulation, wood preservation, as additive in engine oils, electroplating masking compounds, feedstock for dye productions, dye carriers, capacitors and refracting index oils. Thereof, application in wood preservation, as additive in paints and engine oils, for cable insulation and in capacitors have been the most important ones. They were also applied in technical PCB formulations. PCNs were mainly used between 1920 and 1980. Afterwards use declined considerably. Last data on use available have been reported for Germany and former Yugoslavia, where small amounts were used as casting material until 1989.

PCNs were commercially produced as mixtures of several congeners with different product names. Until the 1970s they were high volume chemicals with global production quantities estimated to amount to 9,000 t/y in the 1920s and 150,000 t/y in the 1970s. Production ceased significantly after 1977 within the UNECE regions. In the USA production stopped in 1980, though small amounts of 15 t/y were used in 1981 as refractive index testing oils and capacitor dielectrics. In the UK production stopped in the mid 1960s, although it was reported that in 1970 small amounts were still produced. In Germany production in 1984 amounted to 300 t for use as dye intermediate. In Europe, Bayer produced PCNs in ranges of 100-200 t/y between 1980 and 1983, with production ceasing after 1983.

No information is available on production and use of PCNs at present and therefore it is assumed that no intentional production or uses are occurring within Europe. However, it has been reported that PCN can be formed unintentionally during various thermal processes, e.g. waste incineration and other combustion processes taking place domestically or in the industry [UNECE 2007; Haskoning 2002].

6.9.2 Chemical Characteristics of PCN

PCNs are a group of compounds based on the naphthalene ring system, where one or more hydrogen atoms have been replaced by chlorine. There are 75 possible congeners of PCN in eight congener groups (or groups of homologue congeners) with one to eight hydrogens substituted with chlorine around the planar aromatic naphthalene molecule. While PCNs are considered as a class, physical-chemical and toxicological properties vary by congener and homolog group [IPCS 2001; UNECE 2007].

They are hydrophobic, have high chemical and thermal stability, good weather resistance, good electrical insulating properties and low flammability and are compatible with other materials. Most industrially produced PCNs were mixtures of several isomers of PCNs. The commercial products ranged from low viscosity oils to high melting point solids with intermediate wax like solids varying in crystallinity and melting point [NICNAS 2002]. Liquid PCNs are soluble in most organic solvents, whereas the waxy or solid PCNs are soluble in chlorinated solvents, aromatic solvents and petroleum naphthas, and can be mixed with petroleum waxes, chlorinated paraffins, polyisobutylates, and plasticizers. PCNs have low flammability and are of medium to low volatility, whereas volatility decreases with increasing chlorination [IPCS 2001].

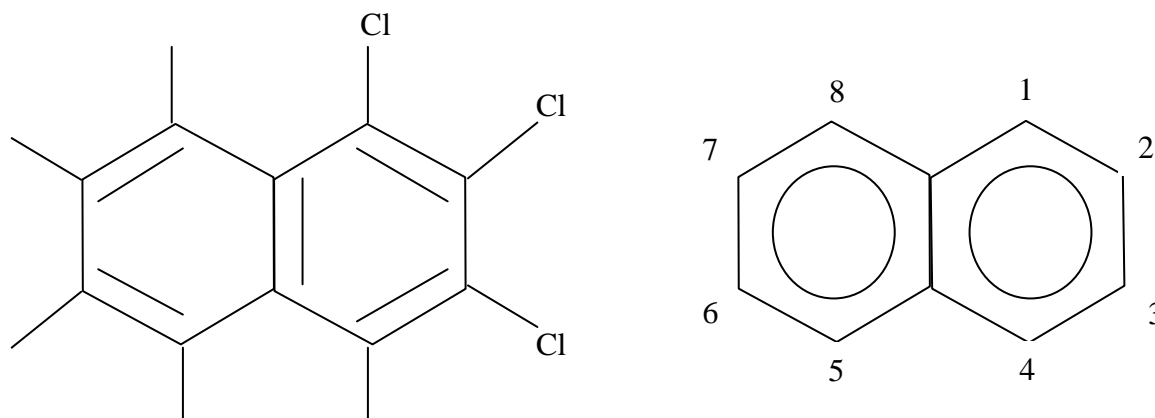
Names and CAS numbers:

Table 6-91: Names and CAS numbers for PCN groups

PCN Name	CAS Number	Molecular Formula
Monochloronaphthalene	25586-43-0	C ₁₀ H ₇ Cl
Dichloronaphthalene	28699-88-9	C ₁₀ H ₆ Cl ₂
Trichloronaphthalene	1321-65-9	C ₁₀ H ₅ Cl ₃
Tetrachloronaphthalene	1335-88-2	C ₁₀ H ₄ Cl ₄
Pentachloronaphthalene	1321-64-8	C ₁₀ H ₃ Cl ₅
Hexachloronaphthalene	1335-87-1	C ₁₀ H ₂ Cl ₆
Heptachloronaphthalene	32241-08-0	C ₁₀ H ₁ Cl ₇
Octachloronaphthalene	2234-13-1	C ₁₀ Cl ₈

Previous Trade Names: Halowax, Nibren Waxes, Seekay Waxes and Cerifal Materials, N-Oil

[NICNAS 2002; UNECE 2007]

Structure:

Generic molecular Formula: C₁₀H_{8-n}Cl_n, where n = 1–8

Physico-chemical properties:

Table 6-92: Chemical and physical properties of polychlorinated naphthalenes [NICNAS 2002]

Group	Molecular weight [g/mol]	Melting point [C°]	Boiling point [C°]	Water solubility [mg/L]	Vapour pressure [kPa]	Henry's Law Constant [Pa m ³ /mol]	LogK _{ow} partition coefficient (octanol/water)
Mono	162.61	-2.3-60	260	0.28-9.2 x 10 ⁻²	2.1 x 10 ⁻³ -3.9 x 10 ⁻³	36	3.9
Di	197	37-138	285-298	8.5 x 10 ⁻³ -8.6 x 10 ⁻²	1.7 x 10 ⁻³	n.a.	4.19-6.39
Tri	231.5	68-133	ca. 274	1.6 x 10 ⁻² -6.7 x 10 ⁻²	1.3 x 10 ⁻⁴	n.a.	5.35-7.56
Tetra	266	115-198	n.a.	3.7 x 10 ⁻² -8.0 x 10 ⁻²	n.a.	n.a.	5.50-8.58
Penta	300.4	147-171	ca. 313	7.3 x 10 ⁻³	4.2 x 10 ⁻⁶	11.9	8.73-9.06
Hexa	335	194	ca. 331	1.1 x 10 ⁻⁴	9.5 x 10 ⁻⁷ -3.0 x 10 ⁻⁹	8.8	6.98-10.37
Hepta	369.5	194	ca. 348	4.0 x 10 ⁻⁵	3.7 x 10 ⁻⁷	n.a.	7.63-8.3
Octa	404	192	440	8.0 x 10 ⁻⁵	1.3 x 10 ⁻⁷	4.8	n.a.
ca. = circa: denotes an estimated or calculated value n. a. = not available							

6.9.3 Environmental Background Levels

In common with many other persistent organochlorine chemical groups, PCNs have spread widely in the global environment. Limitations to analytical determination of these contaminants were a barrier to more detailed description of their environmental distribution and fate, and hence to the identification of likely sources and their relative contributions. However, since the development of reliable congener-specific analytical techniques, the occurrence of PCNs as contaminants in abiotic and biotic media has been increasingly well described [GRL 2004].

A number of studies in the early to mid 1990s focused on the Baltic Sea and its catchment. For instance in the Gdansk region, between 44 and 48 different PCN congeners were reported in sediments and biota (Falandysz et al. 1996). In the Gulf of Bothnia and in the Baltic proper, Jarnberg et al. (1997) described the presence of PCNs in marine sediments and fish from various locations, as well as identified significant point sources of PCNs to Swedish lakes. More recent work by Lundgren et al. (2003) indicates that a consistent congener pattern of PCN contamination exists across much of the Baltic Sea, although absolute concentrations vary and there are some important point sources to coastal regions. A useful review of many of the early studies in the Baltic region is provided by Falandysz (1998) [GRL 2004].

PCNs have also been recognised as significant contaminants in other regions, related to both point and diffuse sources. Eljarrat et al. (1999) for instance, recorded PCNs in sediments from the Venice lagoon, potentially arising from a number of industrial point sources in that region. Both Kannan et al. (1998, 1999) and Brack et al. (2003) have described localised contamination of sediments downstream from specific industrial sources, namely old chlor-alkali facilities, in the USA and in the Bitterfeld region of Germany respectively [GRL 2004].

Besides, PCNs are capable of undergoing long-range transport to remote regions. Their presence as contaminants in air from urban areas or downwind of industrial facilities has been recognised for some time (Dörr et al. 1996, Harner & Bidleman 1997), in patterns reflected also in soil contamination (Krauss & Wilcke 2003) [GRL 2004].

Only a few of the available data sets provide reliable indications of trends in PCN inputs and concentrations. One of the studies, using sediment profiles collected from Esthwaite water (in the Lake District, UK), indicates a subsurface peak of PCN contamination corresponding to the period of major manufacture and use in the first half of the 20th Century (Gevao et al. 2000). A similar trend was reported by Horii et al. (2002) for Lake Kitaura in Japan [GRL 2004].

The chloralkali industry, at least in the form of certain specific facilities, has been recognised as a substantial point source of PCN contamination for many years. PCN contamination has been recorded in the vicinity of chlor-alkali plants in Sweden (Jarnberg et al. 1997), Germany (Brack et al. 2003) and the USA (Kanan et al. 1998) [GRL 2004].

A comprehensive overview of concentration levels in sediment and soil detected in different parts of the world is provided in Table 6-93. Besides, the identified 'hot spots' are presented.

Table 6-93: Levels of PCNs in sediment and soil including hot spots (DW: dry weight) ([GRL 2004]; [Haskoning 2002]; [IPCS 2001]; [ITM 1991]; [Bidleman et al. 2010]; [Brack 2001, 2008]; [Castells 2008]; [Krauss and Wilcke 2003]; [Bogdal et al. 2006]; [Wyrzykowska 2007])

Country/ Region	Compartment	Levels of PCN [ng/g] [ppb]	References	Comments
Sediment				
Europe Sweden	Sediment	0.23 (background concentration)	[Järnberg et al. 1997]	PCN concentrations in sediments of Swedish lakes; A concentration of 0.23 ng/g DW was the lowest of all samples. This concentration was concluded to be the background concentration for Swedish sediments. The concentrations in the other sediment samples varied from 0.62-270 ng/g. The highest concentrations were measured in PCB polluted areas and near chlor-alkali industries.
		0.62		
		260-270		
Europe Sweden	Sediment	1.3	[Asplund et al. 1990]	Pentachloronaphthalenes have been detected at levels of 1.3 µg/kg wet weight in sediments from Lake Järnsjön on the River Emån-Sweden
Europe Baltic Sea	Sediment	0.088-1.9	[Lundgren et al. 2002, 2003]	Average concentrations ΣPCNs were 0.088–1.9 ng/g in multiple samples of surface sediment (0–1 cm) collected at six stations in Bothnian Bay and Bothnian Sea, northern Baltic Sea (in 1991–1992), and 0.3–1.4 ng/g in suspended particulate matter from the same region.
		0.3-1.4		
Europe Baltic Sea	Sediment	6.7	[Falandysz et al. 1996; Ishaq et al. 2003]	ΣPCN in sediments of lakes and rivers in Sweden ranged 6.7 ng/g in the Gdańsk Basin of the southern Baltic, and ~0.5–100 ng/g in settled particulate matter from Lake Mälaren and the Stockholm Archipelago.
		0.5-100		
Europe Germany	Sediment	0.03-13	[Brack 2008]	Concentrations of 30 ng/kg DW up to 13 µg/kg DW were found in Elbe River sediments downstream of Bitterfeld (Germany).
Europe Italy	Sediment	0.034-1.5	[Eljarrat et al. 1999]	In sediments of two lakes; Concentration ranging from 34-1,515 pg/g DW. The lowest concentration was measured at a control site far from industrial areas. The highest concentration was measured at a sewage treatment plant using chlorine.
Europe Spain	Sediment	0.17-3.27	[Castells et al. 2008]	In sediment samples collected from the coastal area of Barcelona (Spain) concentration levels ranged from 0.17-3.27 ng/g dw; for samples collected near submarine emissary values ranged from 2.02 to 6.56 ng/g dw.
		2.02-6.56		
Europe Switzerland	Sediment	0.1-2.1	[Bogdal et al. 2006]	Lake Thun; very low concentrations of 0.1 ng/g dw in deeper, pre-industrial, sediments; followed by a maximum in the late 1920s (2.1 ng/g dw) and a lower maximum around 1957 (1.0 ng/g dw). Since the 1960s, concentrations gradually decreased to 0.3 ng/g dw, representing the current Σ(PCN) concentration in the surface sediments.
USA	Sediment	0.08-187	[Kannan et al. 2000]	Total PCN levels ranging from 0.08 to 187 ng/g dry weight for surface sediments from the Detroit and Rouge rivers, Michigan, USA, with penta and hexachloronaphthalenes the predominant congeners.
Soil				

Country/ Region	Compartment	Levels of PCN [ng/g] [ppb]	References	Comments
USA	Soil	0.13-2.3	[US EPA 1977; Erickson et al. 1978]	Levels of PCNs have been measured in soils near manufacturing plants in the USA where PCNs are thought to have been used. Levels of 0.130–2.3 ng/g were measured near a PCN manufacturing plant. PCN levels between not detected (nd) and 21 ng/g and between nd and 470 ng/g were measured near two capacitor manufacturing facilities, and levels ranging from nd-34 ng/g were measured near a paper manufacturing plant.
		nd-21 nd-470		
		nd-34		
Europe United Kingdom	Soil	6,000-8,640	[Meijer et al. 2001]	In the Broadbalk soil the Σ PCN concentration varied between 6,000 ng/g dw in 1944 to 8,640 ng/g dw in 1956, subsequently declining to 0.317 ng/g dw in 1986. The peak from 1980 of ~16,000 ng/g dw was considered to be a remarkable outlier. In the Luddington soils the Σ PCN concentrations declined as well varying from 6,000 ng/g dw to 0.417 ng/g dw in the control soils and from 8,900 ng/g dw to 2,500 ng/g dw in the sludge amended soils.
		0.317		
		0.417-6,000		
		2,500-8,900		
Europe United Kingdom	Soil	12	[Harner et al. 2000]	Rural soils in the UK. Peak levels of 12 ng/g DW in the 1960s, falling to 0.5–1 ng/g in 1990.
		0-5-1		
Europe Germany	Soil	0.1-15.4	[Krauss and Wilcke 2003]	The sums of concentrations of 35 PCNs were <0.1–15.4 μ g/kg in urban soils and <0.1 to 0.82 ng/g in rural soils.
		<0.1-0.82		
Europe Poland	Soil	0.35-1.1	[Wyrzykowska 2007]	In soil, PCNs concentrations ranged from 0.35 to 1.1 ng/g dry weight.
Hot Spot levels (sediment and soil)				
-	Sediment	100,000 in the past	[IPCS 2001]	Sediment levels of up to 100,000 ng/g have been recorded in the past.
USA	Sediment	104,000	[Rostad & Pereira 1989]	Octachloronaphthalene has been detected in estuarine sediment samples from the USA at levels of 104,000 ng/g DW.
USA	Sediment	nd-61,000	[Furlong et al. 1988]	Levels of PCNs in 33 sediment samples from the Trenton Channel of the Detroit River, Michigan, USA, have been measured. The channel receives waste discharges from several chemical manufacturers. Total PCN levels between nd and 61 ng/g DW were reported.
USA	Sediment	23,000	[Kannan et al. 1998]	Bayou d'Inde (USA), near an industrial outfall; PCN concentrations of up to 23,000 ng/g DW were found in sediments collected in an area contaminated by disposal of wastes from the chlor-alkali process.
USA	Sediment	5,000-20,000	[Elder et al. 1981]	Levels of 5,000-20,000 ng chloronaphthalene/g, 8,000-10,000 g dichloronaphthalene/g, and 4,400-6,000 ng trichloronaphthalene/g have been measured in sediments from the Niagara River, New York, USA, near a dump site.
		8,000-10,000		
		4,400-6,000		
Europe Germany	Sediment	880-1,120	[Brack 2001]	Bitterfeld (Germany) was a major site of chemical production in the former German Democratic Republic, with chlor-alkali electrolysis as the basic process. Concentrations of 880, 543 and 1120 ng/g DW were found, respectively.
Europe Norway	Sediment	1,000	[Naes 1999]	PCNs – emitted by a magnesium plant - were measured in Norway in the Grenland fjords in

Country/ Region	Compartment	Levels of PCN [ng/g] [ppb]	References	Comments
				1997. Tetra- to octa-CN's were measured in concentrations lower than 1,000 ng/g Σ PCN.
Europe France	Sediment	100	[Milano et al. 1985; Milano & Vernet 1988]	Levels of 1-chloronaphthalene ranging up to 100 ng/g dry weight have been measured in marine sediments of Cortiou Creek in the sewage outfall area of Marseilles, France.
-	Soil	1,300,000	[IPCS 2001]	Soil levels of up to 1,300,000 ng/g were measured at contaminated sites in the early 1980s compared with a more recent value for a former chlor-alkali plant of 18,000 ng/g DW.
Europe Netherlands	Soil	31,000-38,000	[De Kok et al. 1983]	Soil samples from areas in the Netherlands that have been used for municipal waste disposal. PCN levels of 31,000–38,000 ng/g dry soil and 1,180,000–1,290,000 ng/g dry soil were measured in two soils.
		1,180,000-1,290,000		
United States	Soil	17,900	[Kannan et al. 1998]	PCN concentrations of 17,900 ng/g DW in soil near a former chlor-alkali plant.

As shown in Table 6-32, the detected contamination levels in sediment ranged from 0.03 ng/g measured in Elbe River sediments to >1,000 ng/g in Germany and Norway at contaminated sites (e.g. PCB polluted areas and near chlor-alkali industries). 'Hot spot' levels of up to 104,000 ng/g were detected in the United States. Highest contamination levels in Europe were identified in Sweden (270 ng/g), Germany (880-1,120 ng/g) and Norway (1,000 ng/g) at polluted sites. According to IPCS (2001), sediment levels of up to 100 mg/kg (100,000 ng/g) have been recorded in the past; however, recent results show levels of 0.2 µg/kg at unpolluted sites and 250 ng/g at polluted sites.

The detected contamination levels in soil ranged from <0.1 ng/g in urban and rural soils in Germany to approximately 1,300,000 ng/g in soil samples taken from areas in the Netherlands that have been used for municipal waste disposal. 'Hot spot' levels were identified in the Netherlands and the United States. According to IPCS (2001), soil levels of up to 1,300 mg/kg (1,300,000 ng/g) were measured at contaminated sites in the early 1980s, compared with a more recent value for a former chlor-alkali plant of 18 mg/kg (18,000 ng/g) dry weight.

6.9.4 Legal Situation PCN

6.9.4.1 EU level

POPs related provisions

PCN is listed in Annex 1 to the POP Protocol, but not in the Stockholm Convention.

Waste management legislation

Following Annex V, Part 1 of EU Waste shipment Regulation 1013/2006 (stemming from the EU obligations of the Basel Convention), wastes, substances and articles containing, consisting of or contaminated with polychlorinated naphthalene (PCN) at a concentration level of 50 mg/kg or more are included in the “amber” list for waste shipments.

The corresponding legal background is discussed in detail in section 4.1.

6.9.5 Occurrence of PCN in Europe

PCNs were commercially produced as mixtures of several congeners with different product names e.g. Halowax, Nibren Waxes, Seekay Waxes and Cerifal Materials. Until the 1970s PCNs were high volume chemicals, e.g. in the 1920s the worldwide production was approximately 9,000 t/y [NICNAS 2002]. In Europe, the production of Bayer - besides production by Imperial Industries - amounted to 100 to 200 t/y between 1980 to 1983 ceasing in 1983 ([UNECE 2007], [Haskoning 2002], [UNECE 2007]).

In Germany around 300 t were produced in 1984, mainly for the use as dye intermediates. It was reported that there was use as casting materials until 1989 in Germany and former Yugoslavia [UNECE 2007]. Production in Germany stopped in 1989 [NICNAS 2002].

In the UK the production stopped in the mid 1960s, although it was reported that in 1970 small amounts of PCNs were still produced [GRL 2004].

Reports from Japan have suggested that PCN formulations may have been imported very recently from suppliers in North America and Europe as recently as the late 1990s. It is not known whether these reported illegal activities represent isolated incidents or hint at more widespread illegal trade and use of PCN stockpiles ([UNECE 2007], [GRL 2004]).

At present, there is no intentional production and use of PCNs within the UNECE region anymore [UNECE 2009]. As there is no production of PCN anymore there are no intentional emissions [UNECE 2007]. Only unintentional emissions of PCNs in UNECE regions originating from unintentional production from thermal processes, of which waste incineration was indicated to be the most important one, have been reported ([UNECE 2007], [NICNAS 2002]).

6.9.5.1 Past intentional uses

Historical use of PCNs has been diverse due to their various properties, as already indicated in chapter 6.9.1 on use and production. Historic uses are described in more detail below.

Electrical industry

Polychlorinated naphthalenes (PCNs) have been commercially produced and used mainly in devices in the electrical industry ([GRL 2004], [Jakobsson et al. 2000]). Reported uses include application as separator in storage batteries, capacitor impregnates, binders for electrical grade ceramics and sintered metals, cable covering compositions [Jakobsson et al. 2000], electroplating [GRL 2004], transformer and capacitor fluids due to their dielectric properties, whereas it has also been reported that PCNs have been present in trace amounts in technical PCB formulations [UNECE 2007].

Not much information is available for this field, but it was reported that especially the higher chlorinated PCN mixtures were used for capacitor impregnation and electrical insulation due to their flame retardant properties [GRL 2004]. They have been applied as additive in plastic and rubber production ([GRL 2004], [Jakobsson et al. 2000]). According to the US EPA, former use of PCN in this field belongs to the more recent uses before production and use stopped. They reported that PCN was applied as intermediate for polymers and as flame-retardants in plastics (Crookes and Howe, 1993) [Jakobsson et al. 2000].

Most data is available for the use of PCNs in PCB formulations. PCNs generally represent a minor component of commercial PCB mixtures. Analysed formulations of Aroclor and Clophen were reported to contain up to 870 ppm PCNs with a median of 67 ppm (Falandysz 1998). Delor formulations from the Czech Republic have been considered to contain relatively high PCN amounts ranging from 82 to 445 ppm (Taniyasu et al. 2003) [GRL 2004]. Solvol formulations from Russia contained PCN concentrations of 730.8 ppm [Haskoning 2002].

Summarizing, several authors measured the PCN content in commercial PCBs leading to a potential releases of PCNs of 100-169 tons. This amount is <0.1% of the former total global production of PCNs of 150,000 tons estimated by Falandysz in 1998 [Haskoning 2002].

Table 6-94 shows the estimated production of PCNs from technical PCB mixtures.

Table 6-94: Estimated production of PCNs from Technical PCB mixtures based on [Yamashita et al. 2000] and [Taniyasu et al. 2003]

Country	PCB mixture	PCB production (tonnes)	Mean PCN concentration (mg/kg)	Amount of PCNs in PCBs (kg)	Reference
US	Aroclors	435,100	39	16,969	[Yamashita et al. 2000]
UK	Aroclors	66,748	39	2,603	[Yamashita et al. 2000]
Japan	Kanechlors	59,119	84	4,966	[Yamashita et al. 2000]
Germany	Clophens	123,552	95	11,737	[Yamashita et al. 2000]

France	Phenochlors	201,679	298	60,100	[Yamashita et al. 2000]
Former USSR	Sovol	100,000	730	73,000	[Yamashita et al. 2000]
CZ	Delors	21,500	171	3,677	[Taniyasu et al. 2003]
Total		1,007,698		173,052	

It has been reported that some PCN-containing electrical equipment was still in use in Europe as late as the start of the 1990s (Weistrand et al. 1992) [GRL 2004]. Given the reported phase-out of production and use of PCNs almost two decades ago, and the parallel phase-out of PCB use, much of the equipment and material may since then have entered the waste stream [GRL 2004]. However, the quantity of PCNs contained in products still in use is highly uncertain based on the very limited existing knowledge [GRL 2004].

According to Directive 96/59/EC PCBs should be disposed by 2010 but there is a derogation for fluids containing less than 500 ppm which can be disposed of at the end of their useful lives. For example in the Czech Republic about 95 % of the relevant PCBs are already destroyed. Remaining fluids containing PCBs may also contain PCN. During the TAC meeting on waste on 14.03.2011 some Member States acknowledged difficulties to comply the disposal obligation by 2010. It can be concluded that relevant quantities of PCB containing fluids may be in use either because they were not duly disposed of by 2010 or because they contain PCBs at levels below 500 ppm. In order to get an overview of the situation of PCBs in the MS it is foreseen to investigate this issue. Based on the outcome it would also be possible to estimate remaining PCN quantities in PCB containing fluids.

Wood impregnate and preservative / fungicide

Several sources report that PCNs have also been used for impregnation of wood in order to attain water-proofness ([GRL 2004], [Jakobsson et al. 2000]). Lower chlorinated naphthalenes have been further used as glues for plywood in Finnish plywood plants and as a product for wood preservation against bugs in Sweden until 1987. In 1987 use in Sweden stopped as it was shown that higher chlorinated naphthalenes than declared were contained [Jakobsson et al. 2000]. Only marginal data is available on the use of PCNs as protection against insects, molds and fungi [Jakobsson et al. 2000].

Textile and paper industry

As for the same reason as in wood, PCNs have also been applied for coating/impregnation of paper and textiles leading to water proofness ([Haskoning 2002], [Jakobsson et al. 2000]). Also Marti et al. report on application as sealants and in finishes for certain textiles and papers [GRL 2004].

Oil additives and lubricants

Several sources report on application of PCNs as oil ([Haskoning 2002], [GRL 2004]). They have been used as engine oil additive, refracting index testing oils ([Haskoning 2002], [GRL 2004]), as additives in

oils for lubrication in gear and cutting oils [Jakobsson et al. 2000]. Especially the lower chlorinated mixtures were predominately used as lubricants [GRL 2004].

Dyes/Dye carrying agents

One major application of PCNs has been settled in the production of dyes, where they have been used as raw material/feedstock and as dye carriers ([GRL 2004], [ITM 1991]). Further, application in lacquers and underwater paints has been reported [Jakobsson et al. 2000]. Around 300 t/y were reported to be produced in Germany in 1984 mainly as dye intermediate until 1989 when production was stopped [NICNAS 2002].

6.9.5.2 Current unintentional production & releases

Besides possible releases due to former uses, several industrial processes have been identified as potential sources for unintentional production of PCNs taking place at present. Identified processes especially include thermal processes, as e.g. waste incineration processes ([NICNAS 2002], [UNECE 2007]), as well as industrial processes (chloralkali processes), whereas in particular metallurgical processes as production of magnesium, copper and aluminium have been mentioned [Jakobsson et al. 2000]. The major current source of unintentional PCN production is likely to result from municipal and special waste incineration [NICNAS 2002]. According to [UNECE 2007] only unintentional emissions still contribute to PCN levels in the environment, whereas PCN emissions for the year 2000 have been calculated by TNO to amount to 1.03 t/y for UNECE Europe based on data from individual countries and expert estimates where detailed data were missing [UNECE 2007].

Incineration processes

Jakobsson et al. report on presence of PCNs in fly ash samples from municipal solid waste incinerators. In fly ashes mono- to octaCNs have been identified with di- to pentaCNs being the most abundant. The lower chlorinated PCNs (mono- to triCNs) have been indicated to be the dominating substances in the flue gas. The formation of PCNs has also been shown in several laboratory combustion and incineration experiments [Jakobsson et al. 2000]. Besides municipal solid waste incineration, medical waste incineration has been indicated as source of unintentional PCN production, too ([GRL 2004], [EGU 2010]). According to [GRL 2004] existing amounts of the chlorinated plastic PVC has been considered as significant waste content in municipal and hospital waste relevant as source for production of PCNs during incineration (Wang et al. 2003). Also domestic/residential burning/combustion of wood and coal have been identified as source of unintentionally formed PCNs ([UNECE 2007], [Lee et al. 2005]). Further, combustion plants providing energy have been considered as source of unintentionally produced PCNs [UNECE 2007].

Metallurgical processes

Unintentional production of PCNs in the metal refining sector has been reported by various sources. The identified sectors include magnesium production ([Jakobsson et al. 2000], [Falandysz 1998], [Baumann Ofstad et al. 1978]), aluminium production ([Jakobsson et al. 2000], [GRL 2004], [Aittola et al. 1994], [NICNAS 2002], [Haskoning 2002]), copper production ([GRL 2004], [Jakobsson et al. 2000], [Theisen et

al. 1993], [NICNAS 2002], [Haskoning 2002]) as well as iron and steel production ([TNO 2006b], [EGU 2010]).

The relevant processes responsible for unintentional production of PCNs have been mostly identified as thermal processes.

In Germany, copper roasting processes have been shown to release PCNs [Theisen et al. 1993] and emission of octaCN has been detected from an aluminum refinery according to Vogelgesang [Jakobsson et al. 2000]. Aittola et al. report on smelting of aluminum scrap as industrial source of unintentional PCN emissions [Aittola et al. 1994].

According to Baumann PCNs were released in the effluents from the production of magnesium [Jakobsson et al. 2000].

Jakobsson et al. indicate that emission levels of PCNs are generally lower compared to e.g. polychlorinated biphenyls (PCBs), but that high levels have been observed near point sources such as manufacturers of chlorine/soda, magnesium, copper and aluminum [Jakobsson et al. 2000].

Iron ore sintering and electric arc furnace steel making units have been reported as possible sources for PCNs [EGU 2010]. Also TNO reports on identification of the secondary iron and steel industry as relevant source [TNO 2006b]. Helm et al. (2000a) analysed ashes from an iron sintering plant in Canada. The samples contained 1.3-2.0 ng PCN/g ash for iron sintering ash (besides ashes from municipal solid waste incinerators and cement kilns) [Helm et al. 2000].

Other processes

The chlor-alkali industry was mentioned as a possible source of unintentional release of PCNs, but there is only little information available ([UNECE 2007], [Haskoning 2002], [Jakobsson et al. 2000], [GRL 2004]). PCN contamination has been recorded in the vicinity of chloralkali plants in Sweden (Jarnberg et al. 1997), Germany (Brack et al. 2003) and the USA (Kanan et al. 1998), with recorded levels of 150 ppb in graphite electrode sludge arising in case of the Swedish facility. However, there appears to be disagreement regarding the origin of PCNs in the sludge wastes. Some authors suggest historic uses of PCNs as solvent or binder used in the manufacture of the graphite electrodes as origin, while others suggest that de novo synthesis of PCNs at the graphite surface during electrolysis is responsible for PCN generation [GRL 2004]. According to Falandysz PCNs in graphite sludge can be related to use of formulations containing PCNs as lubricant and less to formation due to reactions of released chlorine [Falandysz 1998]. However, it can be assumed that PCNs are not anymore used as binder or lubricant for graphite electrodes in the chlor-alkali industry since PCNs are nowadays not used anymore at all. According to [GRL 2004], it is unknown whether any of the existing or decommissioned chlor-alkali plants remain potential sources of PCNs to local environment [GRL 2004].

Other thermal processes associated to unintentional production of PCNs have been reported for coking processes [EGU 2010]. Further, cement kilns have been considered as source of unintentionally produced PCNs ([Helm et al. 2000], [Kucklick et al. 2006], [Falandysz 1998], [UNECE 2007]). Only marginal data is available for these sources.

Besides indicated thermal processes, PCNs have been detected in sewage sludge from waste water treatment plants according to a study in UK as well as in Sweden ([Stevens et al. 2003], [Nylund et al. 1992]).

6.9.6 *Relevance of the Occurrence of PCN in Europe*

Regarding the former uses of PCNs, little information is available, except for the use as dielectric fluid in electrical equipment, in particular in PCB formulations. Since the phase-out of production and use of PCNs almost two decades ago as well as the parallel phase-out of PCB use, much of the equipment and material may since then have entered the waste stream [GRL 2004]. However, it cannot be excluded that due to the broad range of former applications and the differing life span of relevant products PCNs are not anymore contained in products presently in use in Europe.

Besides PCNs contained in products from historic uses, unintentional production sources have been reported, which at the moment are estimated to be the most important sources of PCN emissions. These unintentional sources are related to thermal processes of which combustion/incineration processes of wastes have been indicated as the most important. Total emissions of PCN from thermal processes and incineration were estimated to be 74 % of the total PCN emissions in Europe in the year 2000 ([UNECE 2007], [TNO 2006b]). Incineration of waste is probably the most important source of remaining emissions of PCNs ([UNECE 2007], [NICNAS 2002]). According to more recent investigations conducted by Te Ba et al., emissions due to secondary metal smelting have been indicated to be higher than emissions based on waste incineration [Te Ba et al. 2010]. Due to high activity concerning these processes, they are considered as relevant for the occurrence of PCNs in Europe. Sewage sludge generated during waste water treatment as well has to be taken into account for Europe.

There are no international regulations with respect to PCNs. However, wastes containing PCN are characterised as hazardous waste under category A3180 under the new Annex VIII of the Basel Convention ([TNO 2006b], [UNECE 2007]).

6.9.7 *Selection of Relevant Processes*

Since the phase-out of production and use of PCNs almost two decades ago as well as the parallel phase-out of PCB use, it is estimated that most of the equipment and material may since then have entered the waste stream [GRL 2004]. Although it cannot be excluded that due to the broad range of applications and the differing life span of relevant products PCNs are not anymore contained in products in use at present, these products have not been regarded for allocation of the PCN mass flow and waste flow of PCN containing waste.

The focus in this study for PCN has been set on processes leading to unintentional production of PCNs. Incineration/combustion processes and thermal industrial processes, as e.g. in the metal refining sector, have been assumed as the most important production sources of PCNs. From these processes, incineration of municipal solid waste as well as hospital waste, residential combustion of coal and wood, production of iron and steel, aluminum and copper have been considered as the most relevant processes based on literature identified. Further, waste water treatment has been taken into account

since contamination of sewage sludge has been reported in literature [Stevens et al. 2003]. As already mentioned in the chapter focussing on HCBd, magnesium production is not relevant anymore within the European Union since the amount of primary magnesium produced in plants located in France and Norway have been indicated to amount to zero according to the International Magnesium Association [IMA 2009]. Further, unintentional production arising from combustion processes during energy recovery, chlor-alkali processes, coking processes and cement kilns have either been considered to play a secondary role or have not been further elaborated due to lack of information.

6.9.8 Substance Flow of PCN

Based on the available data the project team has developed the following initial substance flow diagram for PCNs. Since there is no intentional use or production of PCN within the EU, the substance flow diagram is related to unintentional production and related releases of PCN within the EU.

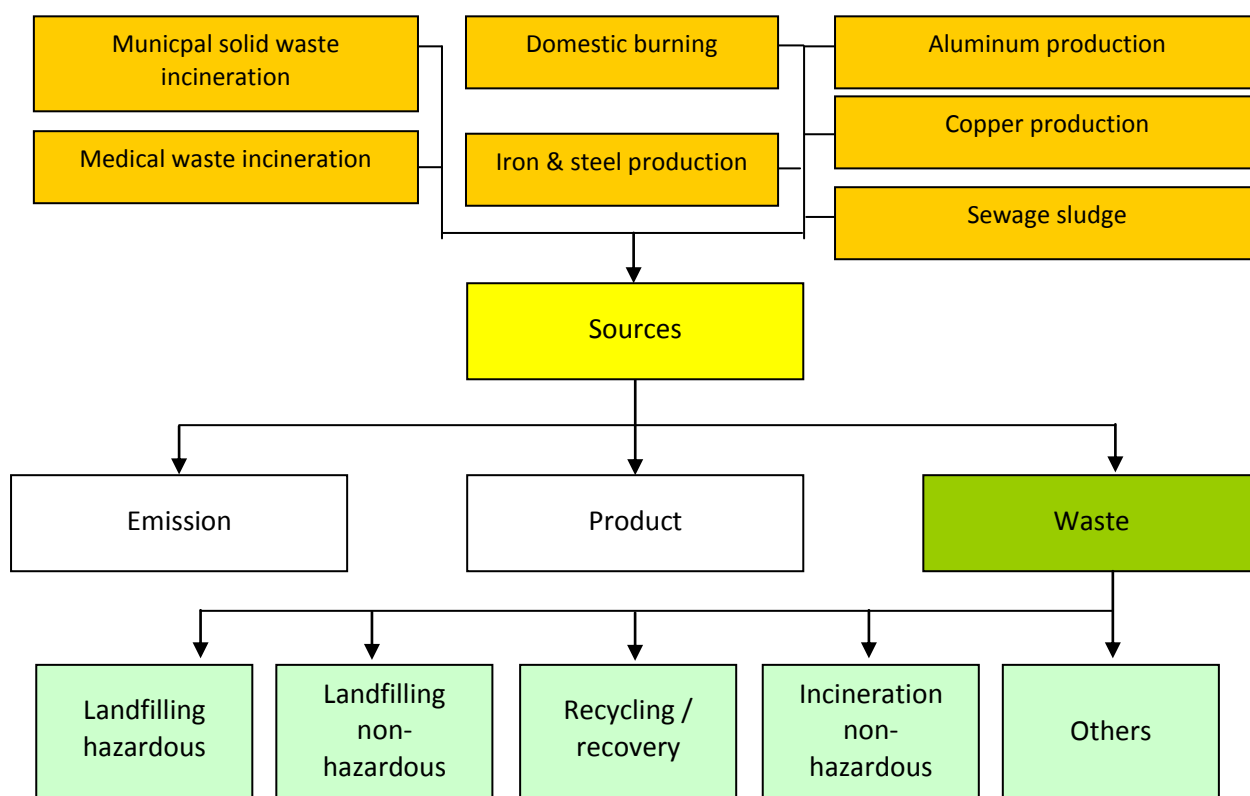


Figure 6-140: Overview of initial PCN substance flow

Figure 6-52 gives an initial overview of the major sources for PCN in the EU. At this stage of the project the main purpose of this figure is to identify and summarise relevant unintentional production processes and related releases of PCN in order to enable an assessment of the relevance of the mass flows of materials containing PCN.

6.9.9 Municipal solid waste incineration (MSWI)

A) Background

The objective of waste incineration is to generate energy and to treat wastes so as to reduce their volume and hazard, whilst capturing (and thus concentrating) or destroying potentially harmful substances that may be released from incineration. It is likely that municipal solid waste as input contains relevant amounts of PCN entering the waste stream due to products reaching their end-of-life-time, but however, it has been shown that residues resulting from the incineration process contain increased amounts of PCN, which have been considered to stand in close relation with PCDF/PCDD formation.

Operating conditions are laid down in Article 6 of the Waste Incineration Directive⁴⁰. The relevant processes and technical aspects in waste incineration are described in detail in the corresponding BAT reference document [BREF WI 2006].

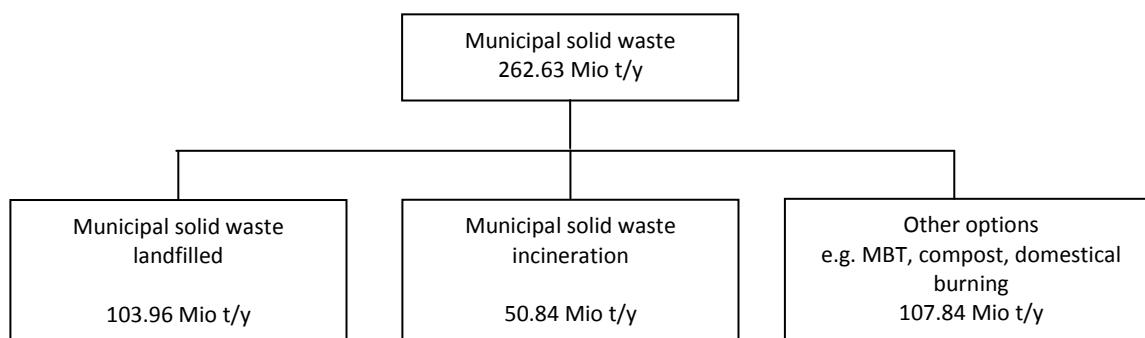
Basically, incineration is the chemical reaction (oxidation) of a combustible material with oxygen. Waste is generally a highly heterogeneous material, consisting essentially of organic substances, minerals, metals and water. During incineration, flue-gases are created that will contain the majority of the available fuel energy as heat. The organic fuel substances in the waste will burn when they have reached the necessary ignition temperature and come into contact with oxygen. The actual combustion process takes place in the gas phase within fractions of seconds and simultaneously releases energy. If the calorific value of the waste and oxygen supply is sufficient, it can lead to a thermal chain reaction and self-supporting combustion, i.e. there is no need for the addition of other fuels.

The main stages of incineration can be found in chapter 6.4.9 on PeCB.

It has to be noted that municipal solid waste in Europe is either treated by municipal solid waste incineration or sent to other management options such as non-hazardous waste landfills, mechanical biological treatment (MBT) or composting. Based on average figures for generation of municipal solid waste of 262.63 Mio t/y within EU 27 [EUROSTAT 2010] and a contamination level of MSW which is difficult to estimate, but has been assumed to amount to 34 ng/g [Noma et al. 2004], the PCN input via MSW is estimated to account to ~8,930 kg/y.

Based on the average of waste incineration in EU 27 the following average flow for municipal solid waste has been calculated for EU 27:

⁴⁰ DIRECTIVE 2000/76/EC OF THE EUROPEAN PARLIAMENT AND OF THE COUNCIL of 4 December 2000 on the incineration of waste



Approximately 20 % of municipal waste is incinerated, 40 % is landfilled and 40 % is treated with other options [EUROSTAT 2010].

In the following, only the path of municipal solid waste incineration is investigated.

B) Process input (raw material) and output (waste, air, water, product)

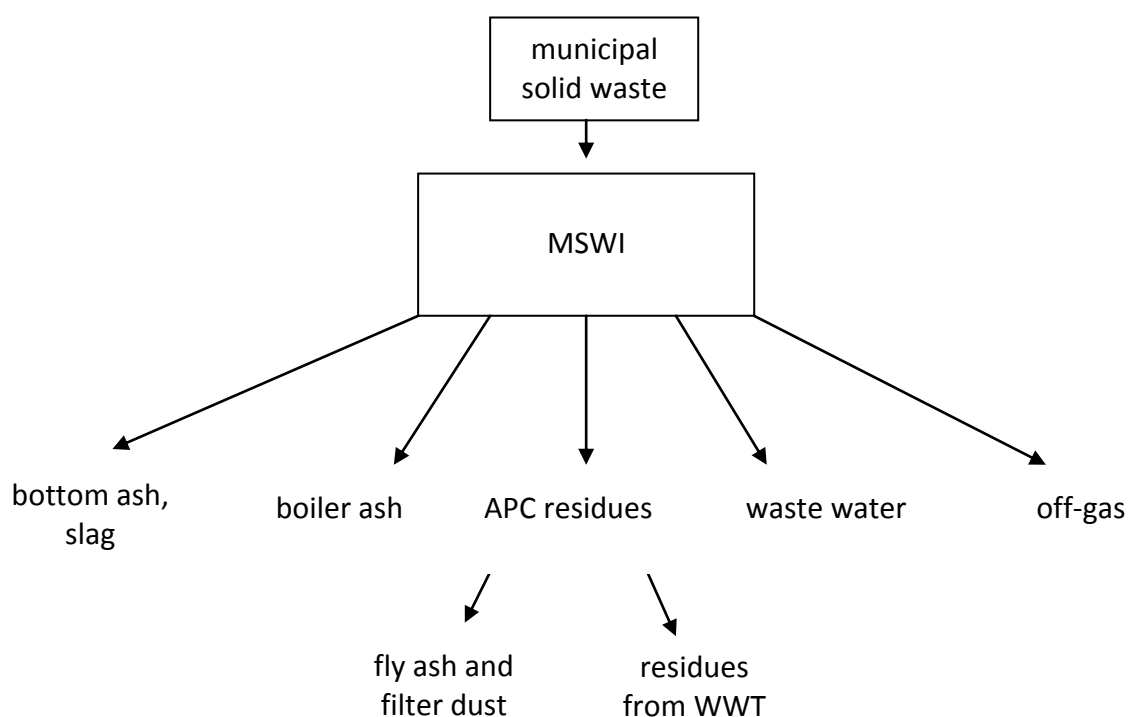


Figure 6-141: Relevant input and output from municipal solid waste incineration

Details on input and output are the same as described in the PeCB chapter 6.4.9 in subchapter B).

C) Generation factors for product/waste and other output (air, water)

Details on generation factors can be found in chapter 6.4.9 on PeCB in subchapter C) and B).

D) Contamination data for input and output

Municipal solid waste	It is likely that PCNs still can be present in products from former uses, although they are not intentionally used or produced anymore. Uncertainty on quantities containing PCNs already having entered the waste stream due to reaching their end of life stadium exists, but according to Noma et al. an input amount of 34 ng/g (34 ppb) waste has been estimated [Noma et al. 2004].
Bottom ash	It has been reported in literature that PCNs are contained in bottom ash with contamination values of 0.95, 1.20 and 1.70 ng/g bottom ash [Noma et al. 2004]. For following calculations an average value of 1.28 ng/g bottom ash (1.28 ppb) has been applied.
Boiler ash	It has not been reported in literature that PCNs are contained in boiler ash.
APC residues:	The most significant amounts of PCNs have been found in fly-ash samples. Schneider et al. report on fly ash contamination values ranging from 0.40-87.0 ng/g fly ash [Schneider et al. 1998]. Noma et al. analysed relatively low concentrations between 0.17-0.96 ng/g [Noma et al. 2004], while Benfenati et al. report on values between 0.05-75 ng/g fly ash [Benfenati et al. 1991]. For calculations an average of 19.95 ng/g (19.95 ppb) fly ash has been used.
Off gas	Several sources reported that off-gas released by a municipal solid waste incinerator still included PCNs in an amount ranging between values as high as 443 ng/Nm ³ [EGU 2010] and as low as 0.002 ng/Nm ³ [Abad et al. 1999], whereas it has to be noted that the majority of values lie between 1-10 ng/Nm ³ . An average has been used based on the values reported by Abad et al. from Spain incinerators (1.77 ng/Nm ³), of values reported by Noma et al. (4.63 ng/Nm ³), Benfenati et al. (49.95 ng/Nm ³) and by the Royal Haskoning (3.03-11.22 ng/Nm ³) amounting to 14.12 ng/Nm ³ ([Abad et al. 1999]; [Noma et al. 2004]; [Benfenati et al. 1991]; [Haskoning 2002]).

E) Activity data

Data on municipal solid waste are mainly derived from European statistics [EUROSTAT 2010]. The following table shows amounts of municipal solid waste incinerated in Europe according to the EUROSTAT database:

Table 6-95: Quantities of MSW generated in EU 27 and amounts of MSW incinerated in EU 27 in 2008

Country	MSW generated [mio t/y]	MSW incinerated [mio t]	Reference year	Reference for amount of MSW incinerated
AT	5.03	1.37	2008	[EUROSTAT 2010]
BE	5.34	1.79	2008	[EUROSTAT 2010]
BG	3.53	0.00	2008	[EUROSTAT 2010]
CY	0.61	0.00	2008	[EUROSTAT 2010]
CZ	3.22	0.36	2008	[EUROSTAT 2010]
DE	47.53	15.79	2008	[EUROSTAT 2010]
DK	4.44	2.40	2008	[EUROSTAT 2010]
EE	0.69	0.00	2008	[EUROSTAT 2010]
ES	26.44	2.44	2008	[EUROSTAT 2010]
FI	2.79	0.48	2008	[EUROSTAT 2010]
FR	35.14	11.13	2008	[EUROSTAT 2010]
GB	35.03	3.41	2008	[EUROSTAT 2010]
GR	5.12	0.00	2008	[EUROSTAT 2010]
HU	4.54	0.39	2008	[EUROSTAT 2010]
IE	3.27	0.08	2008	[EUROSTAT 2010]
IT	33.85	4.16	2008	[EUROSTAT 2010]
LT	1.35	0.00	2008	[EUROSTAT 2010]
LU	0.35	0.12	2008	[EUROSTAT 2010]
LV	0.74	0.00	2008	[EUROSTAT 2010]
MT	0.29	0.00	2008	[EUROSTAT 2010]
NL	10.31	3.37	2008	[EUROSTAT 2010]
PL	12.21	0.08	2008	[EUROSTAT 2010]
PT	5.07	0.97	2008	[EUROSTAT 2010]
RO	8.20	0.00	2008	[EUROSTAT 2010]
SE	4.81	2.34	2008	[EUROSTAT 2010]
SI	0.94	0.01	2008	[EUROSTAT 2010]
SK	1.78	0.16	2008	[EUROSTAT 2010]
EU-27	262.63	50.84		

F) Waste treatment

The high mineral content of incineration ash residues can make them suitable for use as road or other construction material. Use is possible if the material complies with a set of environmental and technical criteria. This requires an optimisation of the ash quality through primary (e.g. optimised combustion process) or secondary measures. Due to its large production volume, treatment for recycling is mainly applied to MSW bottom ash. Bottom ash use is promoted in the Netherlands (>90% used), Denmark (90%), Germany (80%), France (>70%). Recovery of bottom ash in Belgium and the UK is 21% [BiPRO 2005]. Based on these figures it is assumed that more than 50% of bottom ash is used for construction. The remainder is usually directly land filled or land filled after certain treatment. Bottom ash is not mixed with APC residues and is used as secondary raw material in a number of processes. Filter and boiler ash treatment is performed in only a few installations in Europe.

According to the BAT reference document on waste incineration [BREF WI 2006], the following measures are available for pre-treatment improving waste characteristics prior to disposal or recovery:

- solidification
- thermal treatment (vitrification, melting, sintering)
- extraction and separation
- chemical stabilisation
- other methods

The treatment shall improve the leaching properties or reduce the contamination level in the residues. After pre-treatment the residues are normally disposed of at landfills for hazardous waste or are stored underground.

The most important pre-treatment method for wastes intended for landfilling is solidification. Solidification can principally be achieved by addition of several, mostly inorganic, binder reagents (cement, lime and other pozzolanic materials such as fly ash from coal fired power plants, blast furnace bottom ash or cement kiln dust) or organic binders such as bitumen/asphalt, paraffin and polyethylene. However, immobilisation in cement blocks is the prevalent technique.

No information is available concerning the relevance of the other different treatment options. It can be expected that the amount of PCNs contained in the FGT residues is finally landfilled as hazardous waste or stored underground.

In the Netherlands about 36% of fly ash is recovered in asphalt production [BiPRO 2005].

G) PCN flow

The available data and assumptions allow an estimation of PCN emissions to air and discharge to waste via solid residues resulting from MSWI. The following table shows a country specific estimation on emitted amounts:

Table 6-96: PCN mass flow for MSWI in EU 27

Country	MSW incinerated [mio t]	PCNs emitted through fly ash [kg/y]	PCNs emitted through bottom ash [kg/y]	PCNs emitted as off- gas [kg/y]
AT	1.37	0.61	0.41	0.12
BE	1.79	0.79	0.53	0.15
BG	0.00	0.00	0.00	0.00
CY	0.00	0.00	0.00	0.00
CZ	0.36	0.16	0.11	0.03
DE	15.79	7.01	4.72	1.34
DK	2.40	1.06	0.72	0.20
EE	0.00	0.00	0.00	0.00
ES	2.44	1.08	0.73	0.21
FI	0.48	0.21	0.14	0.04
FR	11.13	4.94	3.33	0.94
GB	3.41	1.51	1.02	0.29
GR	0.00	0.00	0.00	0.00
HU	0.39	0.17	0.12	0.03
IE	0.08	0.04	0.03	0.01

Country	MSW incinerated [mio t]	PCNs emitted through fly ash [kg/y]	PCNs emitted through bottom ash [kg/y]	PCNs emitted as off- gas [kg/y]
IT	4.16	1.85	1.24	0.35
LT	0.00	0.00	0.00	0.00
LU	0.12	0.06	0.04	0.01
LV	0.00	0.00	0.00	0.00
MT	0.00	0.00	0.00	0.00
NL	3.37	1.49	1.01	0.29
PL	0.08	0.03	0.02	0.01
PT	0.97	0.43	0.29	0.08
RO	0.00	0.00	0.00	0.00
SE	2.34	1.04	0.70	0.20
SI	0.01	0.01	0.00	0.00
SK	0.16	0.07	0.05	0.01
EU-27	50.84	22.57	15.20	4.31

At a European scale the estimated emissions of PCNs due to MSWI amount to 42.08 kg/y. Thereof 4.31 kg/y are emitted to air, 15.20 kg/y as bottom ash to waste and 22.57 kg/y as fly ash to waste.

Of the bottom ash waste 90 % have been assumed to be recovered/recycled for construction purposes in the Netherlands and Denmark, 80 % in Germany, 70 % in France and 21 % in Belgium and the UK. For the other Member States a value of 50 % has been applied [BIPRO 2005]. The remainder is usually land filled as hazardous waste. Of the fly ash waste approximately 36% have been assumed to be recovered (e.g. construction) in the Netherlands, i.e. 0.54 kg/y, and the rest (22.03 kg/y) disposed off in landfills as hazardous waste. The following PCN flow results:

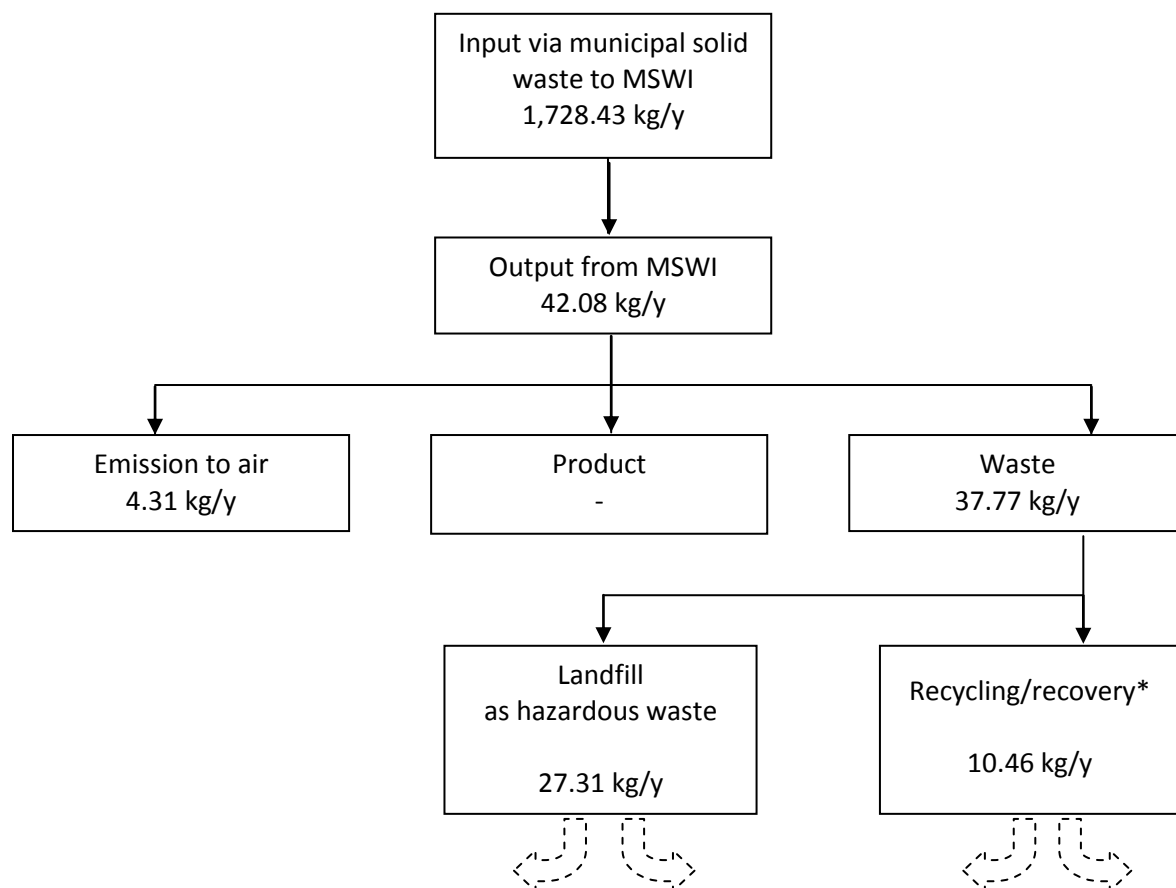


Figure 6-142: Assessment of the PCN flow related to MSWI

H) MSWI waste flow for PCN

In order to illustrate the amounts of residues generated in EU 27 and the importance of currently used disposal/recovery operations which are of high relevance in the evaluation of impacts of proposed limit values, a waste flow has been drawn up in the following figure. Mean contaminations and concentration ranges for each of the residues have been included for better interpretation as far as available in literature.

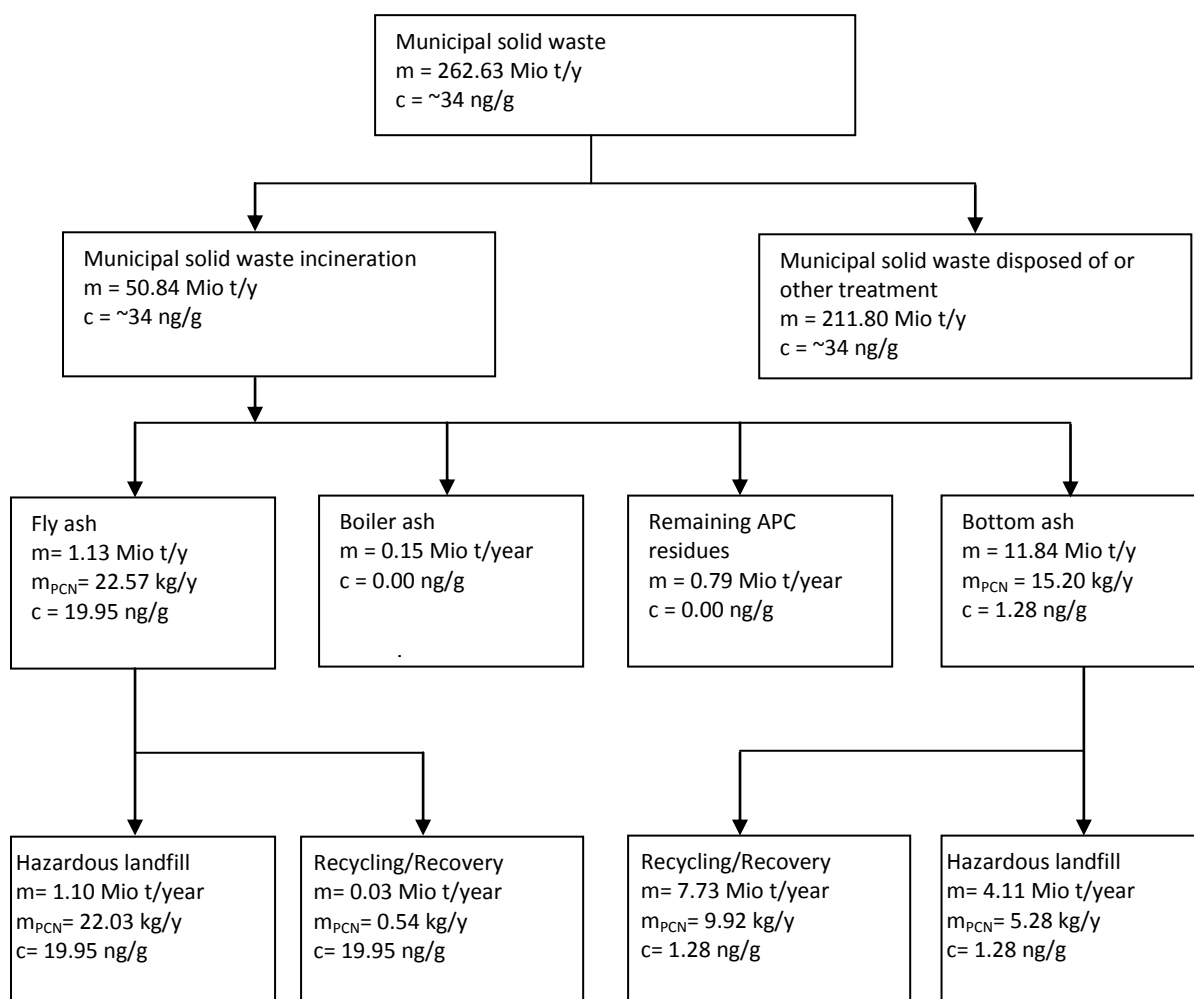


Figure 6-143: Detailed waste flow for MSWI in EU 27

6.9.10 Hospital waste incineration

A) Background

The relevant processes and aspects in clinical waste incineration are described in detail in the corresponding BAT reference document [BREF WI 2006]. For incineration of medical waste rotary kilns are most commonly used, but grate incinerators (including co-firing with other wastes) are also sometimes applied. Clinical waste is also incinerated together with municipal solid waste, e.g. in Denmark. National regulations sometimes limit the ratio of clinical waste that may be treated in combined incineration (e.g. in France <10% thermal load). Dedicated incineration of hospital waste occurs mainly among the new Member States.

B) Process Input and Output

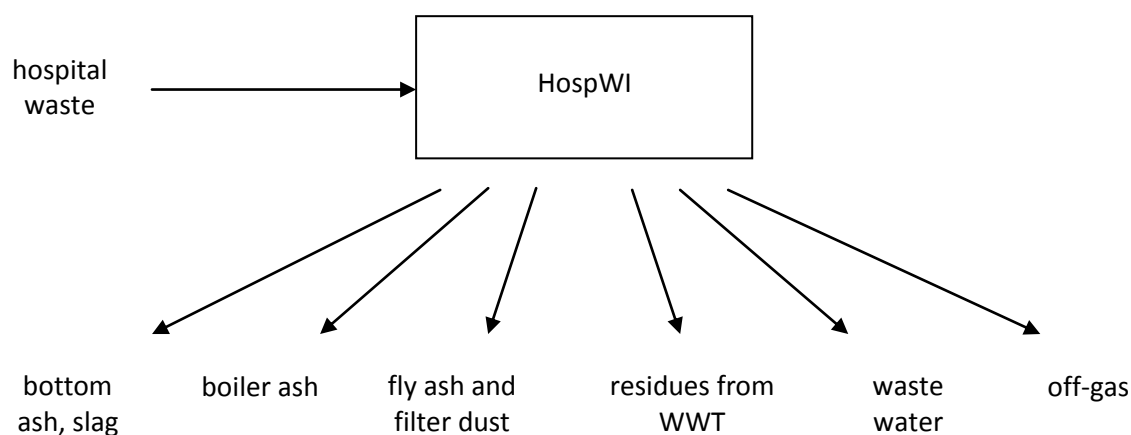


Figure 6-144: Relevant input and output of hospital waste incineration

Input:

Similar to hazardous wastes, the composition of specific hospital or clinical wastes varies greatly. According to the BREF on waste incineration clinical waste may include to varying degrees infectious agents, contaminated clothing/wipes and swabs, pharmaceutical substances, sharp materials such as hypodermic needles, veterinary wastes, body parts, used medical equipment, packaging materials, laboratory wastes and also radioactive contaminated materials. In some cases a distinction is made between the incineration routes for pathological (potentially infectious waste) and non-pathological waste. The treatment of pathological waste is sometimes restricted to dedicated incinerators, while non-pathological waste is, in some cases, incinerated with other wastes in non-dedicated incinerators e.g. MSWI. Clinical waste is also incinerated in hazardous waste incineration plants [BREF WI 2006].

Output:

The output from hospital waste incineration is not fundamentally different from that from municipal solid waste incineration of hazardous waste incineration (see above).

C) Generation factors for relevant waste and other output (air, water)

Waste generation factors are not fundamentally different from those in MSWI. For the PCN flow assessment analogue values as in MSWI have been applied.

D) Contamination data

If no specific values for hospital waste incineration were available, analogue values as in MSWI have been applied for the PCN flow assessment. This has been the case for contamination values for bottom ash (1.28 ng/g fly ash).

Contamination levels for fly ash from an medical waste incinerator in Canada has been reported by Helm et al. to amount to 5439 ng/g fly ash, which is much higher as values that have been reported for fly ashes from municipal solid waste incinerators [Helm et al. 2003].

Off gas contamination from clinical waste incineration has been reported to amount to approximately 45 ng/Nm³ [EGU 2010]. Contamination of other resulting output materials has not been reported in literature.

E) Activity Data

According to EUROSTAT data, waste generated in the health care sector, including biological wastes, in the EU 27 amounted to approximately 1,900 kt in 2008. Thereof, ~950 kt were treated by incineration or incineration for energy recovery, whereas ~550 kt were classified as hazardous waste and ~400 kt as non-hazardous waste. For the calculations it has been assumed that the non-hazardous waste is covered by municipal solid waste incineration, and that only medical waste classified as hazardous is dedicated to hospital waste incineration.

According to data from former reports, dedicated incineration of hospital waste occurs mainly in the new Member States as well as in Greece, Italy, Portugal and Spain [BiPRO 2005].

The following table shows amounts of health care waste (including biological waste) generated and treated by incineration in EU 27 according to data from EUROSTAT.

Table 6-97: Health care waste, including biological waste, generated and treated by incineration in the EU 27 in 2008 [EUROSTAT 2010]

Country	Health care waste generated [t/y]	Hazardous health care waste treated [t/y]	Hazardous health care waste treated by incineration [t/y]	Hazardous health care waste treated by energy recovery [t/y]
AT	57,299	1,038	1,038	0
BE	84,569	34,248	34,248	0
BG	2,796	81	81	0
CY	500	0	0	0
CZ	26,362	22,310	19,206	3,104
DE	242,958	10,287	7,756	2,531
DK	4,134	1,748	0	1,748
EE	2,496	40	0	40
ES	378,500	1,943	366	1,577
FI	3	345	345	0
FR	134,000	123,538	81,617	41,921
GB	528,954	143,985	143,984	1
GR	73,000	13,350	13,350	0
HU	15,516	5,121	5,090	31
IE	0	0	0	0
IT	138,227	109,948	109,946	2
LT	904	673	673	0
LU	396	0	0	0
LV	2,399	11	11	0
MT	365	260	260	0
NL	12,639	2,374	2,374	0
PL	48,276	24,290	23,584	705
PT	86,782	1,811	1,811	0
RO	43,090	3,912	3,912	0
SE	12,227	4,137	619	3,518
SI	3,260	3,873	21	3,852
SK	8,291	39,957	39,943	14
EU-27	1,907,943	549,280	490,235	59,044

F) Waste treatment

It is assumed that treatment and recycling of solid residues from hospital waste incineration follows the same principles as treatment of residues from MSWI. Of the bottom ash waste 90 % have been assumed to be recovered/recycled for construction purposes in the Netherlands and Denmark, 80 % in Germany, 70 % in France and 21 % in Belgium and the UK. For the other Member States a value of 50 % has been applied [BIPRO 2005]. The remainder is usually land filled as hazardous waste. Of the fly ash waste approximately 36% have been assumed to be recovered (e.g. construction) in the Netherlands, and the rest disposed off in landfills as hazardous waste.

G) PCN Waste flow

The available data and assumptions allow an estimation of PCN emissions to air and to waste via exhaust air and solid residues resulting from clinical waste incineration. The following table shows a country specific estimation of emitted amounts:

Table 6-98: PCN mass flow from hospital waste incineration (HospWI) in EU 27

Country	HospWI (hazardous) incinerated [t/y]	PCNs emitted thorough fly ash [kg/y]	PCNs emitted through bottom ash [g/y]	PCNs emitted as off-gas [kg/y]
AT	1,038	0.13	0.31	0.28
BE	34,248	4.14	10.24	9.25
BG	81	0.01	0.02	0.02
CY	0	0.00	0.00	0.00
CZ	22,310	2.70	6.67	6.02
DE	10,287	1.24	3.08	2.78
DK	1,748	0.21	0.52	0.47
EE	40	0.00	0.01	0.01
ES	1,943	0.24	0.58	0.52
FI	345	0.04	0.10	0.09
FR	123,538	14.95	36.94	33.36
GB	143,985	17.42	43.05	38.88
GR	13,350	1.62	3.99	3.60
HU	5,121	0.62	1.53	1.38
IE	0	0.00	0.00	0.00
IT	109,948	13.31	32.88	29.69
LT	673	0.08	0.20	0.18
LU	0	0.00	0.00	0.00
LV	11	0.00	0.00	0.00
MT	260	0.03	0.08	0.07
NL	2,374	0.29	0.71	0.64
PL	24,290	2.94	7.26	6.56
PT	1,811	0.22	0.54	0.49
RO	3,912	0.47	1.17	1.06
SE	4,137	0.50	1.24	1.12
SI	3,873	0.47	1.16	1.05
SK	39,957	4.84	11.95	10.79
EU-27	549,280	66.47	164.24	148.31

According to this estimation the emissions amount to ~66.79 kg/y. Thereof ~150 g are emitted to air and ~66.64 kg are emitted to waste.

The following PCN flow results:

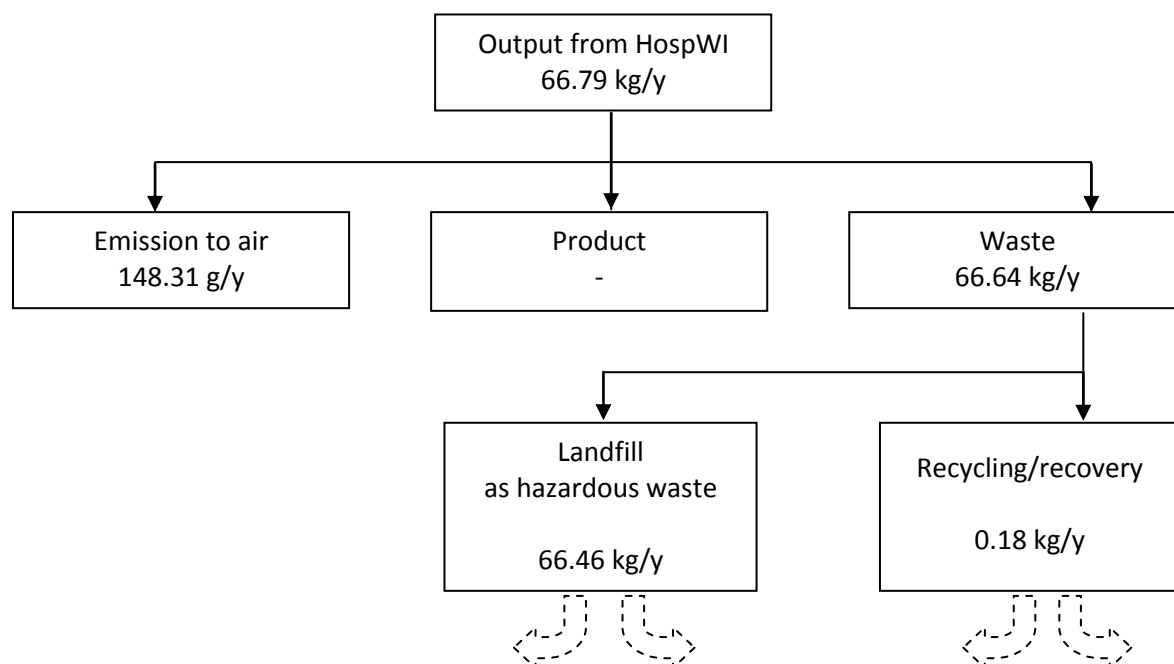


Figure 6-145: Assessment of the PCN flow related to HospWI

H) Waste flow

In order to illustrate the amounts of residues generated in EU 27 and the importance of currently used disposal/recovery operations which are of high relevance in the evaluation of impacts of proposed limit values, a waste flow has been drawn up in the following figure. Mean contaminations and concentration ranges for each of the residues have been included for better interpretation.

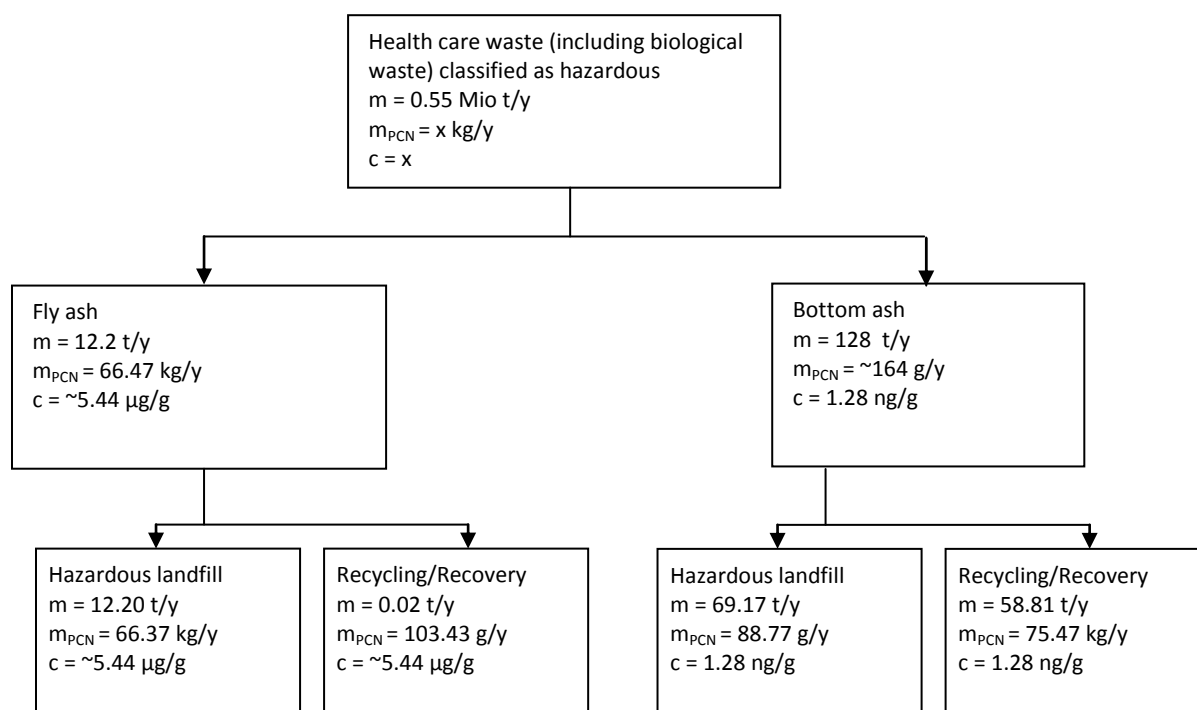


Figure 6-146: Detailed waste flow for medical waste incineration in EU 27

6.9.11 Domestic burning

A) Background

Domestic burning of wood, fossil fuels and mixed wastes accounts to private burning in single stoves or open burning places. Domestic burning is associated with high air emissions as flue gas treatment is not performed. In addition process conditions can strongly vary due to specific oven characteristics and the properties of the used fuel. As a consequence measured contamination and emission data and derived emission factor are highly inhomogeneous to a large range of possible results. In general it can be assumed that emissions exceed the discharge via residues by far and that burning of treated wood or co-combustion of waste significantly increases formation and discharge of PCNs.

B) Process input (raw material) and output (waste, air, water, product)

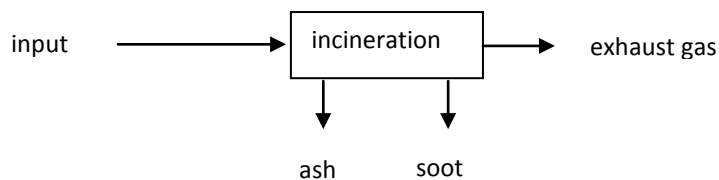


Figure 6-147: Relevant input and output of domestic burning

Input	Wood and wood waste, solid fuels (coal, lignite and their derivatives) and mixed fuels or wastes
Output:	exhaust gas, ashes and soot

C) Generation factors

Exhaust gas:	Generation factors for the amount of resulting exhaust gas are not available. Given contaminations are specific for the materials burned.
Ash:	For coal, a waste generation factor of 0.1 t ash/t coal, for wood 0.017 t ash/t wood and for domestic waste the generation factor of fly ash from municipal solid waste of 22.25 kg/t waste has been used for the calculations [BiPRO 2005].
Soot:	The generation factor for wood is 0.0003 t soot/t wood and for coal 0.0006 t soot/t coal [BiPRO 2005].

D) Contamination data for output (and input)

Exhaust gas:	Domestic burning of coal was calculated to amount to 89.0 ng PCNs/kg house coal combusted according to Lee et al. [Lee et al. 2005].
	Combustion of wood has been assumed to amount to 25 ng PCNs/kg wood

burned according to a study from Lee et al. in 2005 [Lee et al. 2005].

According to Noma et al. burning of municipal waste resulted in emissions of PCNs of 322 ng/kg municipal waste burned. Samples were directly taken after cooling at the exit of the combustion chamber [Noma et al. 2004].

Ash: For burning of solid fuels, the contamination factor of the resulting fly ash ranged from 2.5 ng PCN/g ash for coke and 8.3 ng PCN/g ash for hard coal [Wyrzykowska et al. 2009]. An average value of 5.4 ng/g has been applied for burning of solid fuels.

For wood, a contamination factor of 7.6 ng/g for the resulting ash has been used according to Wyrzykowska et al. [Wyrzykowska et al. 2009].

For domestic waste, a contamination factor for ashes of 6.0 ng/g has been applied [Wyrzykowska et al. 2009].

Soot: No information has been available of the contamination of PCNs in soot. Since the generation factor of soot for wood and coal are relatively small, they are considered as insignificant.

E) Activity data

The final energy consumption in EU 27 of solid fuels in the domestic sector in 2008 amounts to 17.03 Mio t and for wood and wood waste 32.26 Mio t [EUROSTAT 2010]. For the amount of mixed solid fuels burned domestically a share of 0.5% from a total of 262.63 Mio t MSW generated has been assumed.

Table 6-99: Annual consumption of solid fuels, wood and mixed solid fuels for domestic burning in EU 27 in 2008

Country	Solid fuels [Mio t/y]	Wood & wood waste [Mio t/y]	Reference	Mixed solid fuels (assumption) [Mio t/y]
AT	0.13	1.63	[EUROSTAT 2010]	0.03
BE	0.26	0.22	[EUROSTAT 2010]	0.03
BG	0.40	0.64	[EUROSTAT 2010]	0.02
CY	0.00	0.01	[EUROSTAT 2010]	0.00
CZ	1.30	1.06	[EUROSTAT 2010]	0.02
DE	1.75	5.35	[EUROSTAT 2010]	0.24
DK	0.00	0.92	[EUROSTAT 2010]	0.02
EE	0.02	0.39	[EUROSTAT 2010]	0.00
ES	0.30	2.06	[EUROSTAT 2010]	0.13
FI	0.05	1.13	[EUROSTAT 2010]	0.01
FR	0.55	6.67	[EUROSTAT 2010]	0.18
GR	0.04	0.60	[EUROSTAT 2010]	0.18
HU	0.41	0.47	[EUROSTAT 2010]	0.03
IE	1.13	0.02	[EUROSTAT 2010]	0.02
IT	0.01	1.42	[EUROSTAT 2010]	0.02
LT	0.09	0.40	[EUROSTAT 2010]	0.17
LU	0.00	0.02	[EUROSTAT 2010]	0.01

Country	Solid fuels [Mio t/y]	Wood & wood waste [Mio t/y]	Reference	Mixed solid fuels (assumption) [Mio t/y]
LV	0.03	0.73	[EUROSTAT 2010]	0.00
MT	0.00	0.00	[EUROSTAT 2010]	0.00
NL	0.01	0.23	[EUROSTAT 2010]	0.00
PL	9.29	2.45	[EUROSTAT 2010]	0.05
PT	0.00	1.16	[EUROSTAT 2010]	0.06
RO	0.08	3.42	[EUROSTAT 2010]	0.03
SE	0.00	0.62	[EUROSTAT 2010]	0.04
SI	0.00	0.32	[EUROSTAT 2010]	0.02
SK	0.20	0.04	[EUROSTAT 2010]	0.00
UK	0.99	0.30	[EUROSTAT 2010]	0.01
EU 27	17.03	32.26		1.31

F) Waste treatment

There is no official information available on the further treatment of the solid residues from domestic burning. However, according to experience and expert information it can be assumed that the residues are either added to the municipal solid waste or applied to land. As a first approach a share of application to land versus discharge to municipal solid waste of 1:3 has been assumed. According to chimney sweeper information removed soot is completely disposed of with municipal solid waste. Based on the data from the mass flow on MSW a distribution of 20% directed to MSWI, 40% disposed of at non-hazardous waste landfills and 40% directed to other treatment options as MBT can be used as European average for the calculation of the mass flow [BiPRO 2005].

G) PCN mass flow

The available data and assumptions allow an estimation of PCN emissions to air and discharge to ashes. The following table shows a country specific estimation on emitted amounts:

Table 6-100: PCN mass flow for domestic burning of wood and solid fuels in EU 27

Country	Fuel consumption (Mio t/y)			Emission to air [g/y]			Discharge to ash [kg/y]			Total to waste [kg/y]
	Solid fuel	Wood	MSW	Solid fuel	Wood	MSW	Solid fuel	Wood	MSW	
AT	0.13	1.63	0.03	11.57	40.8	8.10	0.07	0.21	0.003	0.28
BE	0.26	0.22	0.03	23.14	5.45	8.59	0.14	0.03	0.004	0.17
BG	0.40	0.64	0.02	35.6	15.975	5.69	0.22	0.08	0.002	0.30
CY	0.00	0.01	0.00	0	0.2	0.99	0.00	0.00	0.000	0.00
CZ	1.30	1.06	0.02	115.7	26.375	5.18	0.70	0.14	0.002	0.84
DE	1.75	5.35	0.24	155.75	133.75	76.52	0.95	0.69	0.032	1.67
DK	0.00	0.92	0.02	0	22.95	7.15	0.00	0.12	0.003	0.12
EE	0.02	0.39	0.00	1.78	9.7	1.11	0.01	0.05	0.000	0.06
ES	0.30	2.06	0.13	26.7	51.375	42.57	0.16	0.27	0.018	0.45
FI	0.05	1.13	0.01	4.45	28.15	4.50	0.03	0.15	0.002	0.17
FR	0.55	6.67	0.18	48.95	166.775	56.57	0.30	0.86	0.023	1.18
GR	0.04	0.60	0.18	3.56	14.975	56.41	0.02	0.08	0.023	0.12
HU	0.41	0.47	0.03	36.49	11.625	8.24	0.22	0.06	0.003	0.28
IE	1.13	0.02	0.02	100.57	0.575	7.30	0.61	0.00	0.003	0.62
IT	0.01	1.42	0.02	0.89	35.55	5.26	0.01	0.18	0.002	0.19
LT	0.09	0.40	0.17	8.01	10.075	54.50	0.05	0.05	0.023	0.12
LU	0.00	0.02	0.01	0	0.4	2.18	0.00	0.00	0.001	0.00
LV	0.03	0.73	0.00	2.67	18.3	0.57	0.02	0.09	0.000	0.11
MT	0.00	0.00	0.00	0	0	1.20	0.00	0.00	0.000	0.00
NL	0.01	0.23	0.00	0.89	5.725	0.46	0.01	0.03	0.000	0.04
PL	9.29	2.45	0.05	826.81	61.2	16.60	5.02	0.32	0.007	5.34
PT	0.00	1.16	0.06	0	29.025	19.66	0.00	0.15	0.008	0.16
RO	0.08	3.42	0.03	7.12	85.575	8.17	0.04	0.44	0.003	0.49
SE	0.00	0.62	0.04	0	15.45	13.20	0.00	0.08	0.005	0.09
SI	0.00	0.32	0.02	0	8.1	7.74	0.00	0.04	0.003	0.05
SK	0.20	0.04	0.00	17.8	0.875	1.51	0.11	0.00	0.001	0.11
UK	0.99	0.30	0.01	88.11	7.525	2.86	0.53	0.04	0.001	0.57
EU 27	17.03	32.26	1.31	1516.56	806.48	422.84	9.20	4.17	0.175	13.54

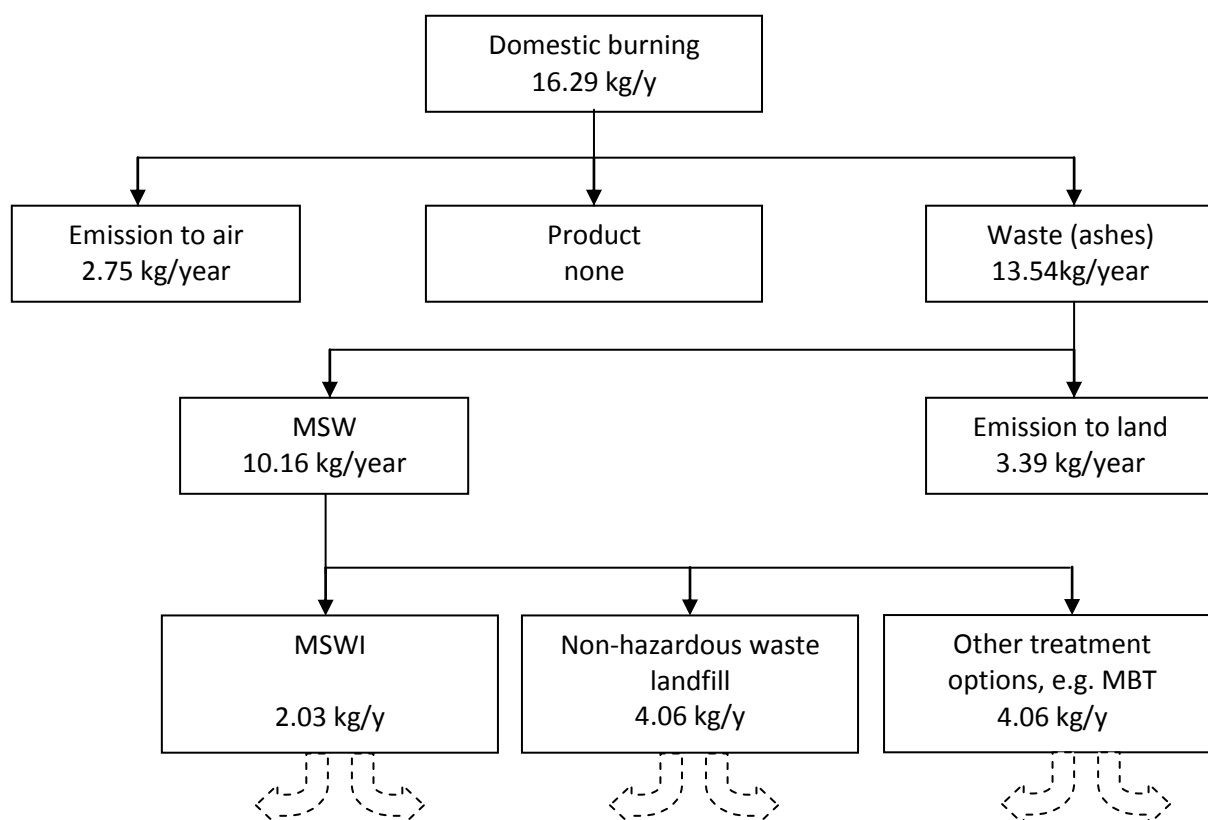


Figure 6-148: Assessment of the PCN flow related to domestic burning

H) Waste flow for domestic burning of wood, fossil fuels and MSW

In order to illustrate the amounts of residues generated in EU 27 and the importance of currently used disposal/recovery operations which are of high relevance in the evaluation of impacts of proposed limit values, a waste flow has been drawn up in the following figure. Mean contaminations and concentration ranges for each of the residues have been included for better interpretation.

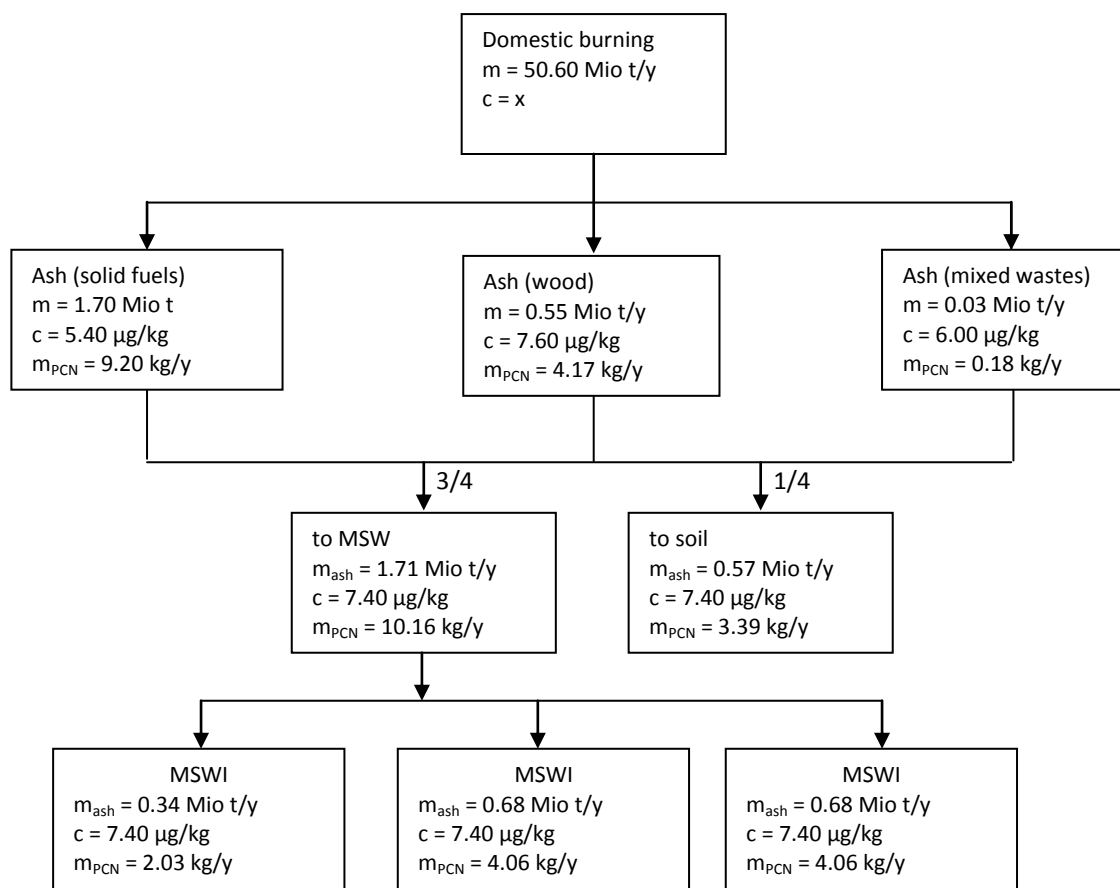


Figure 6-149: Detailed waste flow for domestic burning of wood, solid fossil fuels (coal and derivatives) and mixed wastes in EU 27

Since 25% of the ashes are emitted to land the remaining 75% enter the MSW stream, which in return is incinerated to 20%, landfilled to 40% and treated with other options to 40%. The waste flow for MSWI is described in detail in the chapter MSWI.

6.9.12 Sewage Sludge

A) Background

As PCNs tends to adsorb on particles, a relevant share of the PCN content of waste water accumulates in sewage sludge during waste water treatment. The sewage sludge mentioned in this report is urban sewage sludge as well as sewage sludge resulting from independent and other waste water treatment facilities.

B) Process input (raw material) and output (waste, air, water, product)

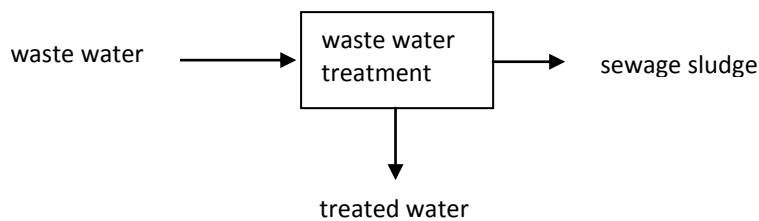


Figure 6-150: Relevant input and output of waste water treatment

Input: waste water

Output: sewage sludge and treated water

C) Generation factors for waste and other output (air, water)

No generation factor is used because the amount of sewage sludge produced from the wastewater depends on the composition of the wastewater and the treatment processes. Following the low solubility of PCNs in water the resulting treated water is not taken into further account.

D) Contamination data for output (and input)

Sewage sludge: As contamination data a median of 76 µg PCN/kg sewage sludge has been used for the UK according to a study of Stevens et al. for sewage sludge samples analysed in the UK [Stevens et al. 2003]. For Sweden the average contamination value analysed by Nylund et al. of 4.50 ng/g has been used [Nylund et al. 1992]. For the other EU countries an average value of 40.25 ng/g has been used for the calculations. The Environmental Protection Agency (EPA) in Ireland has recently undertaken sampling of wastes from sewage treatment to determine concentration levels of the new and candidate POPs in those wastes. According to [EPA IRE 2011a] no PCNs were measured above the limit of detection (<100 µg/kg) in any of the samples. For PCN analysis, the extraction method was solvent extraction and analysis standard is apparently comparable to ISO22032:2006.

E) Activity data

In all 27 Member States a total amount of approximately 11.5 Mio t of sewage sludge from waste water treatment has been calculated from EUROSTAT data (values derived from amount of sewage sludge from urban waste water treatment, from other waste water treatment and independently operated waste water treatment). The Portuguese value originates from another source [UN-Habitat 2008]. For Denmark no value has been available. The amounts for the countries are given in Table 6-42.

Table 6-101: Annual amount of sewage sludge in the EU 27

Country	Sewage sludge produced [Mio t/y]			Total amount of sewage sludge [Mio t]	Reference
	Urban waste water treatment	Other waste water treatment	Independent waste water treatment		
AT	0.26	0.00	0.00	0.26	[EUROSTAT 2010]
BE	0.11	0.00	0.00	0.11	[EUROSTAT 2010]
BG	0.04	0.00	0.00	0.04	[EUROSTAT 2010]
CY	0.01	0.00	0.00	0.01	[EUROSTAT 2010]
CZ	0.17	0.00	0.00	0.17	[EUROSTAT 2010]
DE	2.05	0.00	0.00	2.05	[EUROSTAT 2010]
EE	0.03	0.00	0.00	0.03	[EUROSTAT 2010]
ES	1.07	0.00	0.00	1.07	[EUROSTAT 2010]
FI	0.16	0.00	0.00	0.16	[EUROSTAT 2010]
FR	1.06	0.00	0.00	1.06	[EUROSTAT 2010]
GR	0.13	0.00	0.00	0.13	[EUROSTAT 2010]
HU	0.18	0.10	0.00	0.29	[EUROSTAT 2010]
IE	0.09	0.00	0.00	0.09	[EUROSTAT 2010]
IT	1.06	0.00	0.00	1.06	[EUROSTAT 2010]
LT	0.08	0.00	0.00	0.08	[EUROSTAT 2010]
LU	0.01	0.00	0.00	0.01	[EUROSTAT 2010]
LV	0.02	0.00	0.00	0.02	[EUROSTAT 2010]
MT	0.00	0.00	0.00	0.00	[EUROSTAT 2010]
NL	0.37	0.17	0.00	0.54	[EUROSTAT 2010]
PL	0.53	0.56	0.00	1.09	[EUROSTAT 2010]
PT	0.24	0.00	0.00	0.24	[EUROSTAT 2010]
RO	0.10	0.66	0.002	0.76	[EUROSTAT 2010]
SE	0.21	0.00	0.00	0.21	[EUROSTAT 2010]
SI	0.02	0.00	0.00	0.02	[EUROSTAT 2010]
SK	0.06	0.28	0.00	0.33	[EUROSTAT 2010]
UK	1.77	0.00	0.00	1.77	[EUROSTAT 2010]
EU 27	9.82	1.76	0.002	11.58	[EUROSTAT 2010]

F) Waste treatment

The relevant treatment of sewage sludge includes land filling, incineration, application on land (agricultural use and compost) and other options. The amount of sewage sludge undergoing the different treatments is specific for each country and for each type of sewage sludge. In Table 6-43 values

in % for each Member State are shown for the treatment of urban sewage sludge, which have been derived from EUROSTAT data for each country as far as available (not available for PT and DK).

Table 6-102: Treatment of urban sewage sludge in Member States (PT, DK: no data; MT: no production)

Country	Landfill [%]	Incineration [%]	Application to land [%]	Others [%]	Reference
AT	10	38	44	7	[EUROSTAT 2010]
BE	8	32	19	41	[EUROSTAT 2010]
BG	53	0	15	33	[EUROSTAT 2010]
CY	0	0	63	38	[EUROSTAT 2010]
CZ	5	0	78	16	[EUROSTAT 2010]
DE	0	47	53	0	[EUROSTAT 2010]
EE	17	0	14	69	[EUROSTAT 2010]
ES	16	4	65	16	[EUROSTAT 2010]
FI	6	0	92	2	[EUROSTAT 2010]
FR	21	17	60	3	[EUROSTAT 2010]
GR	98	0	0	2	[EUROSTAT 2010]
HU	29	0	68	3	[EUROSTAT 2010]
IE	6	0	72	23	[EUROSTAT 2010]
IT	42	3	44	11	[EUROSTAT 2010]
LT	12	0	42	46	[EUROSTAT 2010]
LU	33	0	67	0	[EUROSTAT 2010]
LV	0	0	43	57	[EUROSTAT 2010]
NL	4	68	13	15	[EUROSTAT 2010]
PL	23	0	23	53	[EUROSTAT 2010]
RO	44	0	0	56	[EUROSTAT 2010]
SE	11	0	50	39	[EUROSTAT 2010]
SI	43	24	19	14	[EUROSTAT 2010]
SK	16	0	70	14	[EUROSTAT 2010]
UK	5	16	70	9	[EUROSTAT 2010]
EU 27 (average)	21	10	45	24	[EUROSTAT 2010]

G) PCN flow

Member State specific percent values for urban sewage sludge have been used for the estimation of PCN transferred to the different treatment stations from the total amount of sewage sludge produced in each country. For Portugal the European average value for the sludge treatment has been applied. The values are given in Table 6-44.

Table 6-103: PCN mass flow for sewage sludge in EU 27

Country	Amount sewage sludge [Mio t/y]	Amount PCN [kg/y]	Landfilled [kg/y]	Incineration [kg/y]	Application to land [kg/y]	Others [kg/y]
AT	0.26	10.26	1.01	3.94	4.55	0.76
BE	0.11	4.51	0.36	1.45	0.85	1.85
BG	0.04	1.61	0.85	0.00	0.24	0.52
CY	0.01	0.36	0.00	0.00	0.23	0.14
CZ	0.17	6.92	0.36	0.00	5.43	1.13

Country	Amount sewage sludge [Mio t/y]	Amount PCN [kg/y]	Landfilled [kg/y]	Incineration [kg/y]	Application to land [kg/y]	Others [kg/y]
DE	2.05	82.47	0.20	38.84	43.43	0.00
DK	0.00	0.00	0.00	0.00	0.00	0.00
EE	0.03	1.25	0.22	0.00	0.17	0.86
ES	1.07	42.87	6.76	1.65	27.65	6.80
FI	0.16	6.44	0.40	0.00	5.92	0.12
FR	1.06	42.67	8.94	7.16	25.44	1.13
GR	0.13	5.07	4.95	0.00	0.00	0.12
HU	0.29	11.51	3.32	0.00	7.82	0.38
IE	0.09	3.54	0.20	0.00	2.54	0.81
IT	1.06	42.50	17.71	1.25	18.76	4.79
LT	0.08	3.06	0.36	0.00	1.29	1.41
LU	0.01	0.48	0.16	0.00	0.32	0.00
LV	0.02	0.93	0.00	0.00	0.40	0.52
MT	0.00	0.00	0.00	0.00	0.00	0.00
NL	0.54	21.78	0.88	14.77	2.80	3.33
PL	1.09	43.79	10.27	0.16	10.11	23.25
PT	0.24	9.54	2.00	0.99	4.30	2.25
RO	0.76	30.51	13.42	0.00	0.00	17.09
SE	0.21	0.95	0.11	0.00	0.47	0.37
SI	0.02	0.85	0.36	0.20	0.16	0.12
SK	0.33	13.36	2.15	0.00	9.31	1.91
UK	1.77	134.60	7.30	21.43	93.86	12.01
EU 27	11.58	521.82	82.27	91.86	266.03	81.66

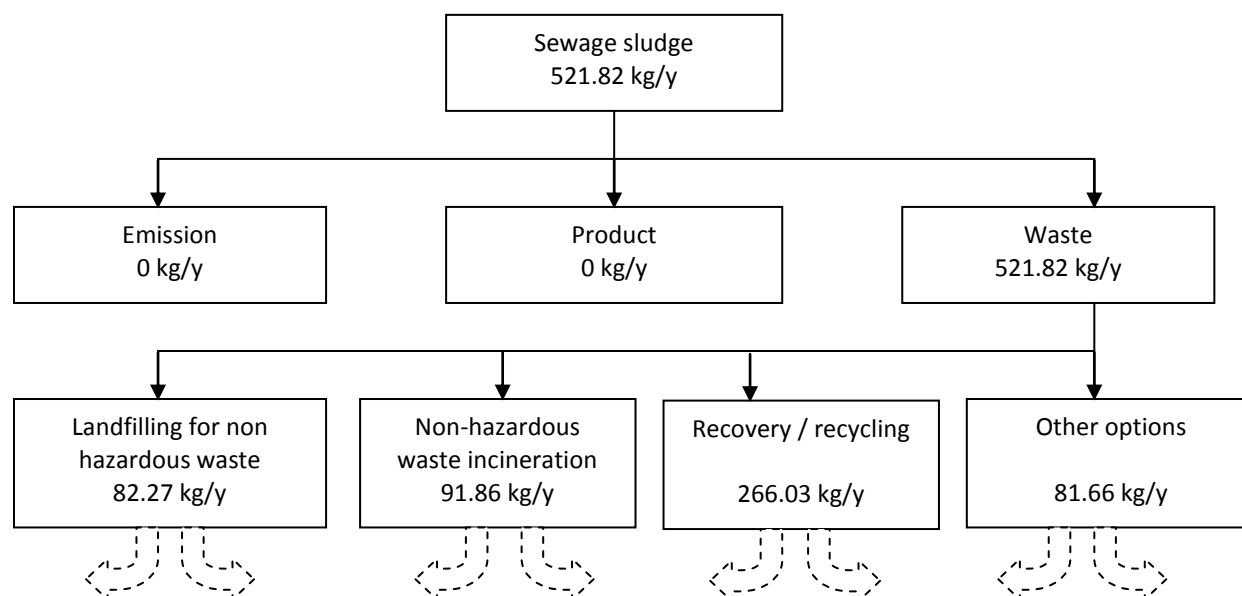


Figure 6-151: Assessment of the PCN flow related to sewage sludge

H) Waste flow for sewage sludge

In order to illustrate the amounts of residues generated in EU 27 and the importance of currently used disposal/recovery operations which are of high relevance in the evaluation of impacts of proposed limit values, a waste flow has been drawn up in the following figure. Mean contaminations and concentration ranges for each of the residues have been included for better interpretation.

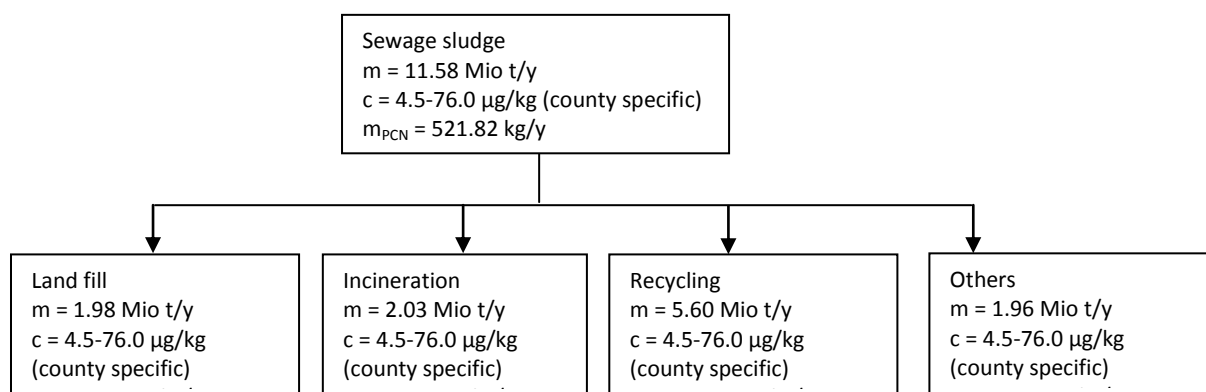


Figure 6-152: Waste flow for sewage sludge in EU 27

6.9.13 Non-ferrous metal industry – secondary copper production

A) Background

Potential emissions of PCNs from secondary copper production have been reported by several sources ([UNECE 2007], [GRL 2004], [Haskoning 2002]). Amounts of PCN have been reported to be contained in “Kieselrot”, a red siliceous slag residue generated during secondary copper production, which has been used as covering material for sports grounds and playgrounds as well as in road and path construction [Theisen et al. 1993]. Further, since PCN formation occurs under similar conditions as formations of PCDD/PCDF formation, it is crucial to examine this sector due to the fact that copper has been considered as the most efficient metal to catalyse PCDD/PCDF formation. The formation is due to the presence of chlorine from plastics and trace oils in the feed material. In a study from Iino et al. [Iino et al. 1999] it was shown that PCNs can directly be formed from PAHs when heated at 400°C for two hours in the presence of copper(I)chloride [Haskoning 2002]. The relevant processes and aspects in secondary copper production are described in [NFM BREF 2001] and [UNEP BAT-BEP 2004].

Secondary copper production is divided into four separate operations: scrap pre-treatment, smelting, alloying and casting. Pre-treatment includes the cleaning and consolidation of scrap in preparation for smelting. Smelting consists of heating and treating the scrap for separation and purification of specific metals. Alloying involves the addition of one or more other metals to copper to obtain desirable qualities characteristic of the combination of metals.

Scrap pre-treatment may be achieved through manual, mechanical, pyrometallurgical, or hydrometallurgical methods. Manual and mechanical methods include sorting, stripping, shredding, and magnetic separation. Pyrometallurgical pre-treatment may include sweating (the separation of different metals by slowly staging furnace air temperatures to liquefy each metal separately), burning insulation from copper wire, and drying in rotary kilns to volatilise oil and other organic compounds. Hydrometallurgical pre-treatment methods include flotation and leaching to recover copper from slag. Leaching with sulphuric acid is used to recover copper from slime, a by-product of electrolytic refining.

Smelting of low-grade copper scrap begins with melting in either a blast or a rotary furnace, resulting in slag and impure copper. If a blast furnace is used, this copper is charged to a converter, where the purity is increased to about 80 to 90 percent, and then to a reverberatory furnace, where copper of about 99 percent purity is obtained. In these fire-refining furnaces, flux is added to the copper and air is blown upward through the mixture to oxidize impurities. These impurities are then removed as slag. Then, by reducing the furnace atmosphere, cuprous oxide (Cu₂O) is converted to copper. Fire-refined copper is cast into anodes, which are used during electrolysis. The anodes are submerged in a sulphuric acid solution containing copper sulphate. As copper is dissolved from the anodes, it deposits on the cathode. Then the cathode copper, which is as much as 99.99 percent pure, is extracted and recast. The blast furnace and converter may be omitted from the process if average copper content of the scrap being used is greater than about 90 percent.

In alloying, copper-containing scrap is charged to a melting furnace along with one or more other metals such as tin, zinc, silver, lead, aluminium, or nickel. Fluxes are added to remove impurities and to protect

the melt against oxidation by air. Air or pure oxygen may be blown through the melt to adjust the composition by oxidizing excess zinc. The alloying process is, to some extent, mutually exclusive of the smelting and refining processes described above that lead to relatively pure copper.

The final recovery process step is the casting of alloyed or refined metal products. The molten metal is poured into moulds from ladles or small pots serving as surge hoppers and flow regulators. The resulting products include shot, wire bar, anodes, cathodes, ingots, or other cast shapes.

B) Process input (raw material) and output (waste, air, water, product)

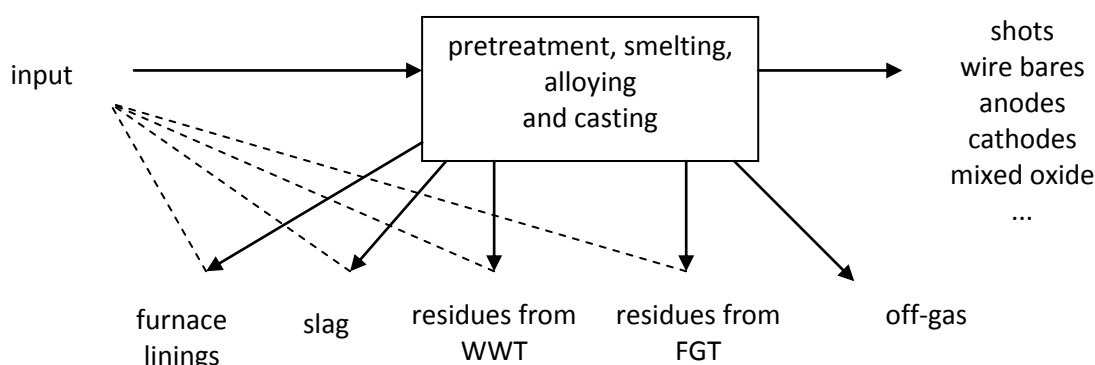


Figure 6-153: Relevant input and output of secondary copper production (dotted lines: internal recycling possible)

Input: feed source are copper scrap, sludge, computer scrap, drosses from refineries and semi-finished products and internally recycled materials such as dusts, ashes, drosses or sludge. These materials may contain organic matter like coatings or oil, and installations take this into account by using de-oiling and de-coating methods or in an appropriate design of the furnace and abatement system.

Output: Final slag (EWC 100601): In fire-refining furnaces, slags are partly recycled within the smelting process. Not recyclable “final slag” is used for road construction.

Sludges from waste water treatment (EWC 100607*): Depending on the site specific production process, waste water may arise. Process, surface and cooling water can be contaminated by suspended solids, metal compounds and oils. Most process and cooling water is recycled. By-products and residues from waste water treatment are recycled in the process as these contain recoverable quantities of copper and other non-ferrous metals. If waste arises, it generally consists of acid sludge which is disposed of.

Furnace linings (EWC 161101*, 161102, 161103*, 161104) are removed in intervals (e.g. each 14 to 15 months) from the inner surface of involved furnaces. In processing the linings are exposed to high temperature and a contamination with PCNs is not expected.

Residues from FGT (EWC 100603*, 100606*): Airborne emissions contain among other dust and metal compounds and organic carbon compounds. Particulate matter is removed from collected and cooled combustion gases by electrostatic precipitators (ESPs) or fabric filters.

Filter dusts are partly used as input for secondary production (see “products”), partly internally recycled and partly disposed of (due to their high content of arsenic).

Off-gas may contain significant amounts of PCNs.

Products: shot, wire bar, anodes, cathodes, ingots, or other cast shapes. Specific filter dust from flue gas treatment consist of zinc/lead mixed oxide (“furnace-oxide”) which is used as raw material for secondary zinc and lead production.

The relevant material flows with respect to the PCN flow are in particular slags and off-gas as far as evident from literature.

C) Generation factors for waste and other output (air, water)

Emission per tonne of pure copper produced [BiPRO 2005]:

- Slag: ~ 620 kg
- Filter dust: ~ 6 kg
- Furnace linings: ~ 6 kg
- Furnace oxide: ~ 96 kg
- WWT residues (acid sludge): no information available

D) Contamination data for output (and input)

Typical contamination:

- Slag: The most common type of slag in secondary copper production is iron silicate slag that is produced at temperatures ranging from 1200 to 1300°C. The only PCN contamination value for slag found in literature has been indicated for Kieselrot, a red siliceous slag residue, which is obtained during copper ore smelting [Theisen et al. 1993]. Theisen et al. reported a contamination value of 4.224 mg/kg for this slag type, which has been used for the following calculations.
- Filter dust: No contamination values have been reported
- Refractory bricks: No values are reported for contamination of furnace linings
- Furnace-oxide: No contamination values have been reported
- Filter cakes: No contamination values of filter cakes from WWT are available
- Off-gas: Te Ba et al. reported of stack gas emissions in China ranging from 41.3 to 1107 ng/Nm³ taken from five secondary copper facilities. One of these plants with an off-gas value of 79 ng/Nm³ has been indicated to possess an advanced air pollution control technology system considered as comparable to European systems. For this reason this value has been used for the following calculations [Te Ba et al. 2010].

E) Activity data

According to the European Copper Institute, the five main producing countries of copper are Germany, Poland, Belgium and Spain, besides Russia [ECI 2009]. In Europe, secondary copper production amounts to ~900,000 t/y [BREF NMR 2001]. The following table shows production data of secondary copper for Member States for 1997 (Table 6-42).

Table 6-104: European production of secondary copper for 1997 [BREF NFM 2001].

Country	production of secondary copper [t]	Country	production of secondary copper [t]
AT	77,000	IT	80,000
BE	183,000	LT	0
CY	0	LU	0
CZ	12,000	LV	0
DE	378,000	MT	0
DK	0	NL	0
EE	0	PL	29,000
ES	63,000	PT	0
FI	0	SE	34,000
FR	29,000	SI	0
GR	0	SK	17,000
HU	30,000	UK	58,000
IE	0	EU-25	990,000

F) Waste treatment

Filter dust is partly internally recycled (up to 100%) and partly disposed of as hazardous waste (due to its high content of arsenic). The relation between recycling and disposal is not available. "Furnace-oxide" is used as a raw material for secondary zinc and lead production. Furnace linings may be partly recycled externally or internally, or disposed of. The relation between recycling and disposal is not available. Acid sludge from waste water treatment is usually disposed of on site. Slag is used for road construction. Treated off gas is released to the atmosphere.

G) PCN flow

The available data and assumptions allow only a very rough estimation of PCN emissions to waste via slag. No information has been found in literature for contamination of filter dust, sludge, waste water, furnace linings and other waste types. No generation factor for stack gas for calculation of emitted amounts of PCN via stack gas is available; consequently no calculation could be conducted. The following tables show a country specific estimation on emitted amounts based on the available data (Table 6-44).

Table 6-105: PCN mass flow for secondary copper production in EU

Country	Production of secondary copper [t]	Amount slag generated [t/y]	Amount PCN in slag generated [kg/y]
AT	77,000	47,740	201.65
BE	183,000	113,460	479.26
CY	0	0	0.00
CZ	12,000	7,440	31.43
DE	378,000	234,360	989.94
DK	0	0	0.00
EE	0	0	0.00
ES	63,000	39,060	164.99
FI	0	0	0.00
FR	29,000	17,980	75.95
GR	0	0	0.00
HU	30,000	18,600	78.57
IE	0	0	0.00
IT	80,000	49,600	209.51
LT	0	0	0.00
LU	0	0	0.00
LV	0	0	0.00
MT	0	0	0.00
NL	0	0	0.00
PL	29,000	17,980	75.95
PT	0	0	0.00
SE	34,000	21,080	89.04
SI	0	0	0.00
SK	17,000	10,540	44.52
UK	58,000	35,960	151.90
EU-25	990,000	613,800	2592.69

At a European scale the estimated emissions to waste due to slag amount to ~2,600 kg/y. The following overall picture results:

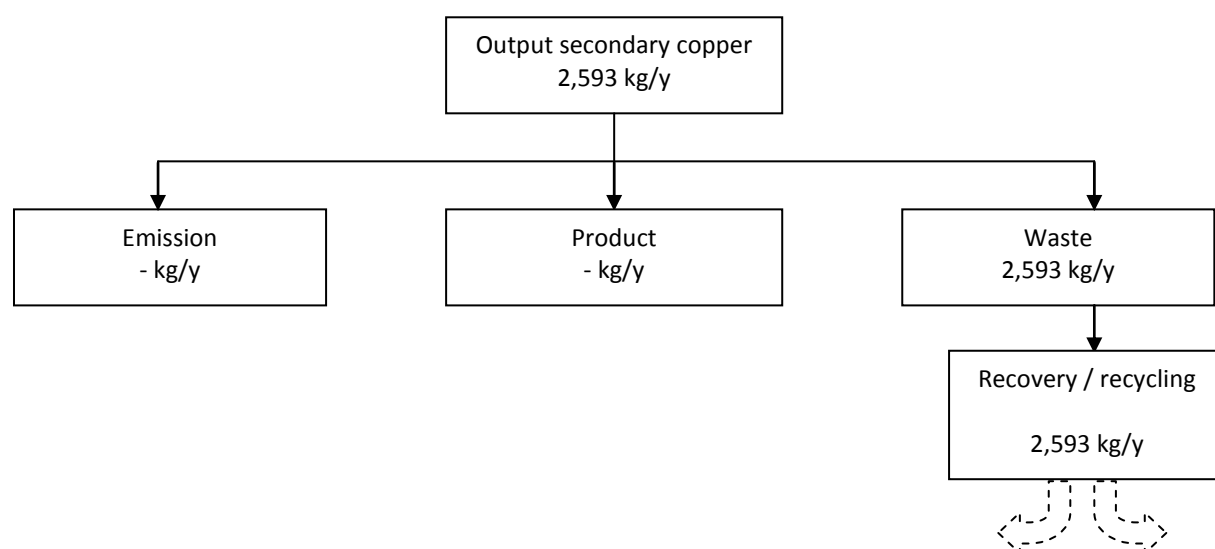


Figure 6-154: Assessment of the PCN flow related to secondary copper production

H) Waste flow for secondary copper production

On the basis of the available data, no detailed waste flow could be allocated. However, only for slag calculations were conducted. In order to elaborate a detailed waste flow further investigations are crucial.

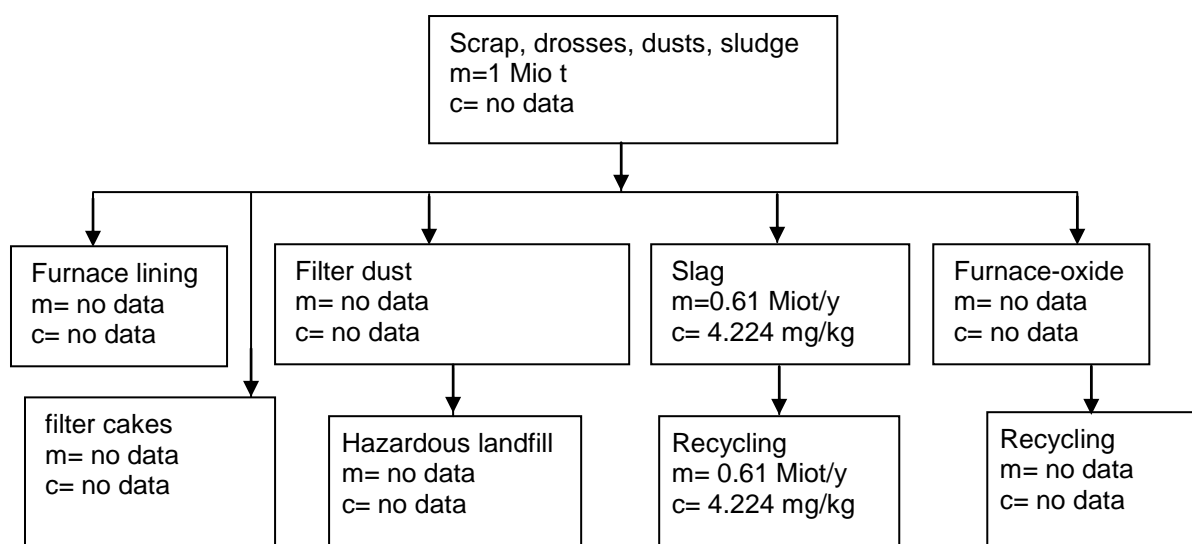


Figure 6-155: Rough waste flow secondary copper production in EU 27

6.9.14 Non-ferrous metal industry –Secondary aluminium production

A) Background

Potential emissions from secondary aluminium production are due to the presence of organic contaminants and plastics in the feed material with the addition of chlorine and chlorides during the smelting process.

The relevant processes and technical aspects in secondary aluminium production are described in [NFM BREF 2001] and [UNEP BAT-BEP 2004].

B) Process input (raw material) and output (waste, air, water, product)

The important inputs and outputs are illustrated in the following schematic flow chart.

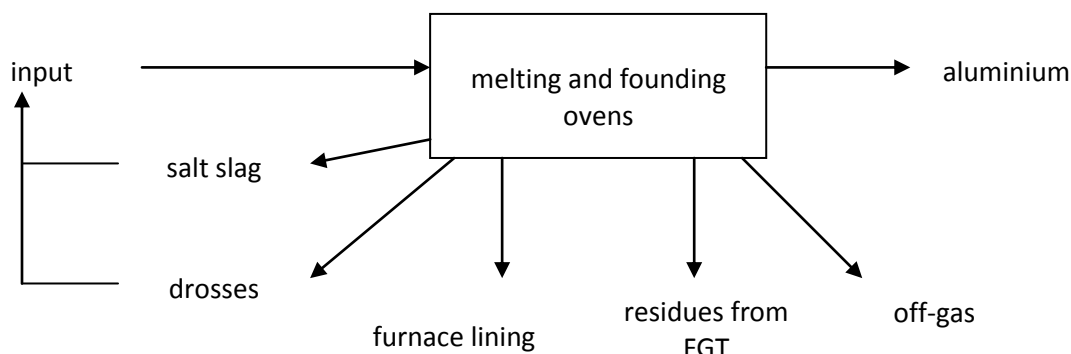


Figure 6-156: Relevant input and output of secondary aluminium production

Input:

- input: aluminium scrap, drosses, solid metal, alloy metals, adjuvants (melting salt, azote-chlorine mixture)

Output:

- salt slag (EWC 100308*) and drosses (100309*) are internally recycled and re-used as input for the production process
- furnace linings (EWC 161101*, 161102, 161103*, 161104)
- dry and wet residues from flue gas treatment (EWC 100323*, 100324, 100325*, 100326): filter dust (EWC 100319*);
- off-gas

The relevant material flows with respect to the PCN flow are in particular filter dust and off-gas.

C) Generation factors for waste and other output (air, water)

Emissions per tonne aluminium produced:

- furnace linings: 4.7 kg (range 3.6 to 7.7)
- filter dust: 20.9 kg (range 18 to 47)
- sludge from WWT 2.7 kg (range 1.0 to 3.6)
- salt slag 310 kg (range 152 to 444)
- off-gas ~ 3,500 Nm³

D) Contamination data for output (and input)

Contamination levels:

- off-gas For the assessment of the PCN emissions to air an average value of 205.63 ng/Nm³ has been applied according to the study of Te Ba et al. 2010 [Te Ba et al. 2010].
- furnace linings: No data has been indicated in the literature for contamination with PCNs.
- filter dust: Te Ba et al. analysed fly ash values of 6.9, 91.6, 16.8 and 6000 ng/g from 4 different facilities; an average of the first three values of 38.43 ng/g has been used for the calculations, while the value of plant 4, which is exceptionally high, has been not taken into account due to the great difference compared to the other values [Te Ba et al. 2010].
- sludge from WWT: No data has been indicated in the literature for contamination with PCNs. However, sludges arising from waste water treatment has been considered in the chapter on sewage sludge.
- salt slag No data has been indicated in the literature for contamination with PCNs.

E) Activity data

Production data are based on NFM BREF for the year 1997. According to the OEV at present 481,705 tonnes of aluminium are annually produced in secondary aluminium production in AT, DE, DK NL and SE.

Table 6-106: European production of secondary aluminium (reference years: [BREF NFM 2001] for 1997 and EUROSTAT, annual PRODCOM for 2001)

Country	Production of secondary aluminium [t]	Reference
AT	98,000	[BREF NFM 2001]
BE	0	[BREF NFM 2001]
BG	0	No information available
CY	0	No information available
CZ	35,000	[TNO 2005]
DE	433,000	[BREF NFM 2001]
DK	14,000	[BREF NFM 2001]
EE	0	No information available
ES	154,000	[BREF NFM 2001]
FI	33,000	[BREF NFM 2001]
FR	233,000	[BREF NFM 2001]
GR	10,000	[BREF NFM 2001]
HU	32,761	[ESTAT, PRODCOM data]
IE	0	[BREF NFM 2001]
IT	443,000	[BREF NFM 2001]
LT	0	No information available
LU	0	[BREF NFM 2001]
LV	28,138	[ESTAT, PRODCOM data]
MT	0	No information available
NL	150,000	[BREF NFM 2001]
PL	123,000	[TNO 2005]
PT	3,000	[BREF NFM 2001]
RO	0	No information available
SE	26,000	[BREF NFM 2001]
SI	0	No information available
SK	1,400	[TNO 2005]
UK	257,000	[BREF NFM 2001]
EU-27	2,074,299	

F) Waste treatment

Salt slag and drosses are internally recycled and re-used as a process input. Filter dust and furnace linings are treated for further use.

Disposal or recycling operations for filter dust according to the OEV (particular situation in Germany) include 18% mine filling, 53% underground disposal and 29% re-use as building material (cover of above ground landfills).

Off gas including remaining dust is emitted after flue gas treatment

G) PCN flow

The available data and assumptions allow an estimation of PCN emissions to air and to waste via filter dust and air. However, it is assumed that PCN is also contained in the other relevant waste types, but due to the lack of information no calculations could be conducted. The following table show a rough country specific estimation on emitted amounts based on the above mentioned data.

Table 6-107: PCN mass flow for secondary aluminium production in EU 27

Country	Production of secondary aluminium [t]	PCN emissions to air [g/y]	filter dust [g/y]	total PCN produced [g/y]
AT	98,000	70.53	78.72	149.25
BE	0	0.00	0.00	0.00
BG	0	0.00	0.00	0.00
CY	0	0.00	0.00	0.00
CZ	35,000	25.19	28.11	53.30
DE	433,000	311.64	347.81	659.45
DK	14,000	10.08	11.25	21.32
EE	0	0.00	0.00	0.00
ES	154,000	110.84	123.70	234.54
FI	33,000	23.75	26.51	50.26
FR	233,000	167.69	187.16	354.85
GR	10,000	7.20	8.03	15.23
HU	32,761	23.58	26.32	49.89
IE	0	0.00	0.00	0.00
IT	443,000	318.83	355.84	674.68
LT	0	0.00	0.00	0.00
LU	0	0.00	0.00	0.00
LV	28,138	20.25	22.60	42.85
MT	0	0.00	0.00	0.00
NL	150,000	107.96	120.49	228.45
PL	123,000	88.53	98.80	187.33
PT	3,000	2.16	2.41	4.57
RO	0	0.00	0.00	0.00
SE	26,000	18.71	20.88	39.60
SI	0	0.00	0.00	0.00
SK	1,400	1.01	1.12	2.13
UK	257,000	184.97	206.44	391.40
EU-27	2,074,299	1492.91	1666.19	3159.10

At a European scale the estimated emissions amount to ~ 3.16 kg/y. Thereof ~ 1.5kg is emitted to air and ~ 1.7 kg is emitted to waste. The following overall picture results:

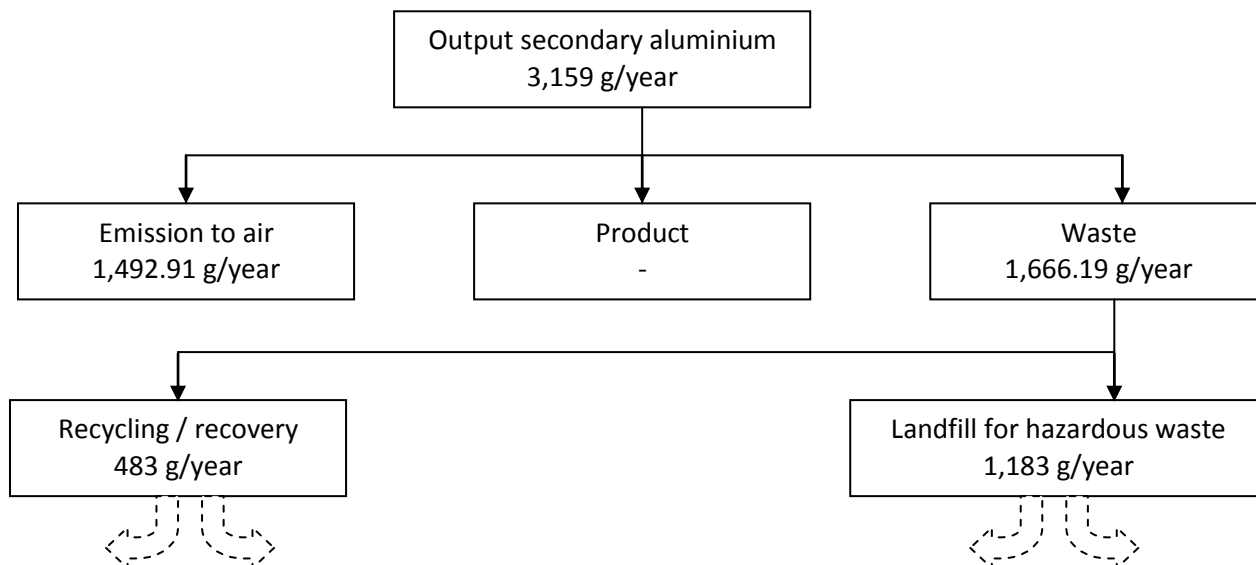


Figure 6-157: Assessment of the PCN flow related to secondary aluminium production

H) Waste flow for secondary aluminium production

In order to illustrate the amounts of residues generated in EU 27 and the importance of currently used disposal/recovery operations which are of high relevance in the evaluation of impacts of proposed limit values, a waste flow has been drawn up in the following figure. Mean contaminations and concentration ranges for each of the residues have been included for better interpretation.

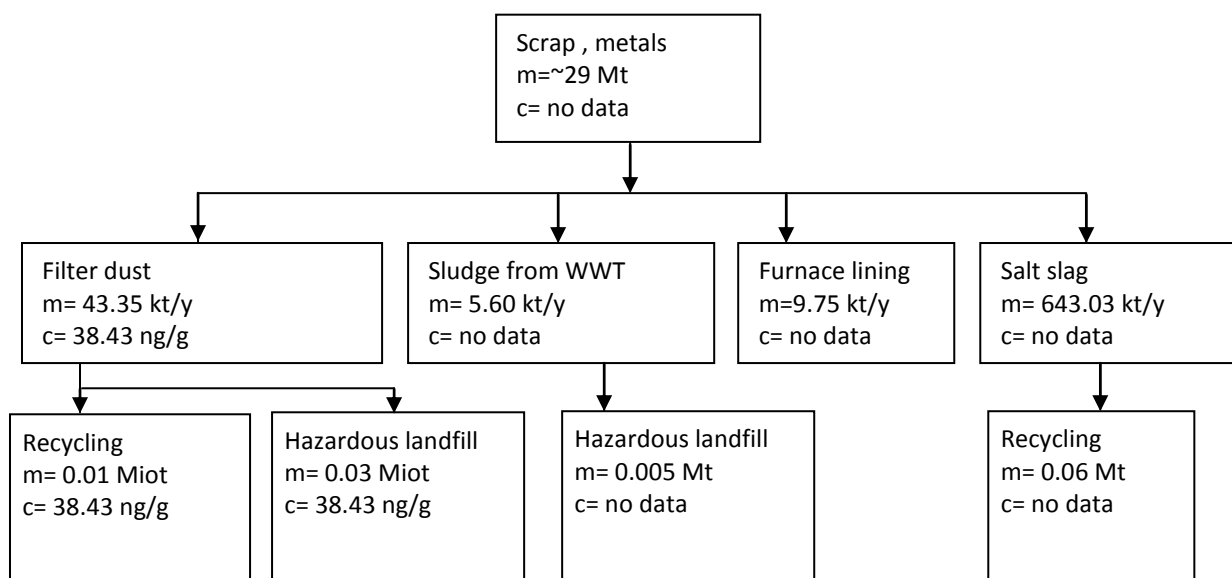


Figure 6-158: Detailed waste flow secondary aluminium production in EU 27

6.9.15 Iron and steel production – Sinter plants and electric arc furnace

Four routes are currently used for the production of steel: the classic blast furnace/basic-oxygen furnace route, direct melting of scrap (electric arc furnace), smelting reduction and direct reduction.

The EU steel production was based on the blast furnace/basic-oxygen route (approximately 61%) and the electric arc furnace (EAF) route (approximately 39%), [EUROFER 2005]. In Europe smelting reduction and direct reduction are of very limited importance [IS BREF 2001]. Consequently, the relevant emissions from European iron and steel production originate from the blast furnace/basic oxygen route and electric arc furnace.

(A) Background

In integrated steelworks applying the blast furnace/basic oxygen route, sinter plants dominate the overall emissions for atmospheric pollutants including emissions of PCN. Consequently, sinter, as a product of an agglomeration process of iron-containing materials, represents a major part of the emissions from blast furnaces. The most relevant environmental issues are exhaust air from the sinter strand, which contains a wide range of pollutants including PCN and emissions via solid waste as PCN contaminated dust resulting from filtering processes.

The relevant processes and technical aspects in iron sintering and electric arc furnace are described in detail in the corresponding BAT reference document [IS BREF 2001].

(B) Process input (raw material) and output (waste, air, water, product)

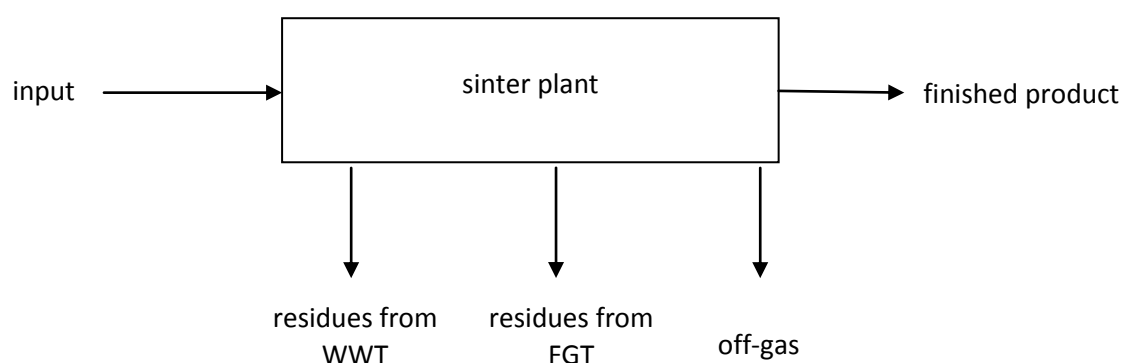


Figure 6-159: Relevant input and output of the iron sintering process

Solid residues management in an integrated steelwork aims at extracting value from the various types of slag and recycling of most of the residues in the sinter plant, which can be considered the 'digester of an integrated steelworks' [IS BREF 2001]. Thus, besides the sintering process itself, the sinter plant plays this important role of recycling residues for which no adequate alternatives exist.

Only small parts of the overall quantity of residues are landfilled. These often consist of fine dust from

blast-furnace gas cleaning, rubble, fine dust from basic oxygen furnace gas scrubbing (if wet cleaning is used) and, in some cases, the high alkali chlorides and heavy metal chlorides from the last field of electrostatic precipitators treating the off-gas from sinter strands.

Input:

Main input are iron ores, other iron materials, lime and limestone, additives, dust from blast furnace process, recycled materials and internally recycled sinter

Output:

Potential waste type outputs includes ashes, treated off-gas, sinter product and in case of the electrical arc furnace process.

(C) Generation factors for relevant waste and other output (air, water)

Dust amounts that arise in a sinter plant range from 0.9 to 15 kg/t liquid steel. According to information from EUROFER (personal communication Anna Utsi) 0.2 to 0.5 kg/t liquid steel is disposed of to landfills. An average of 0.5 kg/t liquid steel is assumed for the calculation of the PCN flow.

The average exhaust air volume per tonne liquid steel is approximately 2,300 Nm³ [BiPRO 2005]. This value is used for the PCN emission calculation from the sintering process as well as for the electric arc furnace.

(D) Contamination data for output (and input)

The emission factor of PCN in sinter plants is given at 42 µg/tonne and in electric arc furnaces at 2980 µg/tonne [EGU 2010].

For off gas from sinter plants 65 ng/Nm³ and from electric arc furnaces 720 ng/Nm³ have been reported [EGU 2010].

PCDD and PCDF are found in flue gas residues with an average of 1.1 ng TEQ/g (range <0.01-20 ppb) for sinter plants. The same value is used for filter dust for external treatment from the electric arc furnace [BiPRO 2005].

No data are available regarding the PCN concentration in wastes as dust or ashes from the iron industry. Regarding [Chemosphere 2001] who investigated the correlation of PCN and PCDD/PCDF in the fly ash from incineration of municipal waste, it was found that both components are within the same order of magnitude. Even so that this comparison is very vague this comparison is used in general, this would mean that in case of the sinter plant about 64,000 t of waste include about 70 g of PCN (~ 1.1 ng/g) [BiPRO POP waste] and in case of the electrical arc furnace some 70 Mt of waste consisting of slag, filter dust and refractory bricks contain about 1.2 kg of PCN. From the describe waste from the electric arc furnace about 6 Mt are recycled, 5 Mt are landfilled (hazardous or non-hazardous), 0.5 Mt are temporary stored and an undefined amount of the refractory bricks is reused or recycled.

(E) Activity data

The following table shows the current capacity for sinter production in Europe [EUROFER 2010]:

Table 6-108: Production of crude steel in EU 27

Cou ntry	2004 [kt/y]	2005 [kt/y]	2006 [kt/y]	2007 [kt/y]	2008 [kt/y]	2009 [kt/y]	Jan- Sept 2010 [kt/y]	extrapo lated 2010 [kt/y]	Mean [kt/y]
AT	6,529	7,031	7,129	7,577	7,593	5,662	5,293	7,057	6,940
BE	11,697	10,420	11,630	10,691	10,672	5,635	6,011	8,015	9,823
BG	2,090	1,948	2,102	1,909	1,330	725	559	745	1,550
CZ	7,033	6,188	6,860	7,057	6,386	4,593	3,931	5,241	6,194
FI	4,833	4,739	5,051	4,431	4,417	3,078	2,987	3,983	4,362
FR	20,770	19,480	19,852	19,249	17,879	12,840	11,621	15,495	17,938
GE	46,374	44,524	47,223	48,550	45,832	32,670	32,964	43,952	44,161
GR	1,967	2,266	2,416	2,554	2,477	2,000	1,383	1,844	2,218
HU	1,952	1,958	2,083	2,241	2,097	1,403	1,252	1,669	1,915
IT	28,603	29,349	31,624	31,552	30,589	19,847	19,065	25,420	28,141
LV	662	688	690	695	635	692	519	692	679
LU	2,683	2,194	2,802	2,857	2,581	2,141	1,932	2,576	2,548
NL	6,848	6,918	6,372	7,368	6,853	5,193	4,893	6,524	6,582
PL	10,592	8,444	10,007	10,631	9,727	7,129	6,089	8,119	9,236
PT	1,250	1,338	1,338	1,847	1,630	1,587	1,014	1,352	1,477
RO	5,930	6,150	6,170	6,136	4,917	2,686	2,810	3,747	5,105
SK	4,438	4,484	5,093	5,089	4,489	3,747	3,475	4,633	4,568
SL	565	582	628	638	640	430	446	595	583
ES	17,684	17,825	18,397	18,998	18,640	14,357	12,518	16,691	17,513
SE	5,948	5,691	5,434	5,635	5,163	2,778	3,563	4,751	5,057
UK	13,765	13,226	13,885	14,316	13,519	10,078	7,301	9,735	12,646
EU 27	202,220	195,452	206,787	210,029	198,076	139,278	129,697	172,929	189,253

For the calculation an annual mean value of 190,000 t is used. With the shares of 61 % sinter production and 39 % electrical arc furnace the relevant crude steel amounts are 116,000 tonnes from the sinter process and 74,000 t from the electrical arc furnace process.

(F) Waste treatment

Residues from flue gas treatment are to a large degree internally recycled. However, a certain share of dust has to be disposed of. This is in particular fine dust from the last field of electrostatic precipitators and, if a fine scrubber system is applied, sludge from flue gas treatment. It is assumed that all dust and sludge which is not internally recycled is externally disposed of. The share of disposal of hazardous and non-hazardous FGT residues (EWC numbers 10 02 07* and 10 02 08) is not known.

(G) PCN flow

The available data and assumptions allow an estimation of PCN emissions to via exhaust air. The following table shows a country specific estimation on emitted amounts in the year 2004 to 2010:

Table 6-109: PCN mass flow for iron sintering in EU 27 (average values between 2004 and 2010)

Country	Production of crude steel [kt/y]	Emission to air sinter plant [kg/y]	Emission to air EAF [kg/y]	Emission to air total [kg/y]
AT	6,940	0.63	4.48	5.11
BE	9,823	0.90	6.34	7.24
BG	1,550	0.14	1.00	1.14
CZ	6,194	0.56	4.00	4.57
FI	4,362	0.40	2.82	3.21
FR	17,938	1.64	11.58	13.22
GE	44,161	4.03	28.52	32.55
GR	2,218	0.20	1.43	1.63
HU	1,915	0.17	1.24	1.41
IT	28,141	2.57	18.17	20.74
LV	679	0.06	0.44	0.50
LU	2,548	0.23	1.65	1.88
NL	6,582	0.60	4.25	4.85
PL	9,236	0.84	5.96	6.81
PT	1,477	0.13	0.95	1.09
RO	5,105	0.47	3.30	3.76
SK	4,568	0.42	2.95	3.37
SL	583	0.05	0.38	0.43
ES	17,513	1.60	11.31	12.91
SE	5,057	0.46	3.27	3.73
UK	12,646	1.15	8.17	9.32
EU 27	189,253	17.26	122.23	139.49

From the calculation it can be seen that about 140 kg of PCN are emitted from the iron industry and that 122 kg of these are coming from the electric arc furnace process and has therefore a share of about 88 % and the PCN emission from the sinter plants only have a share of about 12 %.

When calculating the general emission by using the emission factor of 42 µg/t and 2980 µg/t for sinter plants and EAF, respectively, an emission of 7.95 kg/y for sinter plants and 583 kg/y for EAF is reached.

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6.10 Remediation methods for contaminated sites

Specific information on sites contaminated with new or candidate POPs is particularly available from the Czech Republic, where sites contaminated with PeCB exist. Furthermore, fire training sites and fire emergency sites where PFOS containing fire fighting foam has been applied are suspicious to be contaminated with relevant quantities of PFOS. Close to relevant sources certain sites may be contaminated with all relevant new and candidate POPs.

Remediation methods can be differentiated in (1) in-situ (on-site) methods vs. (2) methods where the contaminated soil or sediment is excavated and subsequently treated in an appropriate manner or disposed of under safe conditions (off-site).

According to a German engineering company carrying out remediation of contaminated sites, the usual costs for off-site remediation methods is typically around 80€ per cube meter soil (range ~60 to 140€/m³)⁴¹. These costs include the costs for excavation, the engineering services and the costs for the disposal or off-site treatment.

The costs for on-site methods start in a comparable range but show a high variation and can be significantly higher. The specific costs depend on several factors e.g. on (1) the individual remediation method applied, (2) the specific local conditions such as soil properties, ground water situation and (3) target values of the remediation etc. High-tech on-site methods are usually more expensive than off-site methods.

The methods evaluated in Table 6-110 have been selected from a compilation of innovative on-site remediation methods (see [ITVA 2010]) because they can be considered appropriate for individual POPs.

⁴¹ Personal communication, January 2011

Table 6-110: Selected innovative remediation methods considered to be appropriate for the remediation of sites contaminated with POPs (based on [ITVA 2010])

Method	Application Area	Limitations	Remark
Thermally supported soil-air extraction by vapour-air injection (TUBA)	Organic pollutants	Low volatile pollutants; Specific local conditions	Confirmed to be applicable in practice according to pilot tests; Can provide reliable results, good control, fast, cost efficient if local conditions are appropriate
Thermal on-site remediation with installed heat source (THERIS)	High to low volatile organic pollutants	Specific local conditions	Confirmed to be applicable in practice according to pilot tests; Can provide reliable results, good control, cost efficient if local conditions are appropriate
Alcohol scavenging	Organic pollutants	Specific local conditions; hydraulic safeguarding required, risk for dilution of pollutants	Method is developed at technikum scale; pilot field application with scientific attendance not yet realized
Iron particles	High to low volatile halogenated organic pollutants including chlorinated pesticides, PCP etc.	Specific local conditions; low level of experience concerning the transport behavior of the iron particles in groundwater; potential health risks	Several pilot projects ongoing (e.g. in Italy, Check Republic and Germany); still under development; limited expectations
Permeable reactive barriers (PRB)	Not specified; high variability depending on the reactive material (e.g. activated carbon, elemental iron) used.	Specific local conditions; often limited cost efficiency	Often very good remediation results; particularly PRBs with activated carbon can be considered a reliable technology

7 Measurement methods

7.1 Introduction

The needs for POPs analysis under the Stockholm Convention mainly arise from provisional limit values for POP contents in wastes. The Stockholm Convention was implemented in Community law by Regulation (EC) No. 850/2004 (POP Regulation), which was subsequently amended in 2006. The purpose of the amendment was to define concentration limits for 14 POPs substances and substance classes, above which the POPs content in waste shall be subject to destruction or irreversible transformation (lower POP content or lower POP limit) as well as threshold limits (upper POP content) above which no derogation from the destruction obligation can be granted. In 2009 a number of new substances have been added to the annexes of the Stockholm Convention by decision of the fourth conference of the Parties (COP-4), and the EU is requested to amend the POP Regulation by 26 August 2010.

Concerning concentration limits for “new POPs” as to be stipulated by the POP regulation, reliable and comparable analytical methods have to be available. The same applies to the “candidate POPs” substances proposed to be added to the list of POPs under the UNECE POP Protocol in the course of the next years.

In this respect some basic aspects related to sampling and analysis of POPs and other pollutants have to be taken into account. To establish reliable and internationally comparable analytical methods is the only prerequisite for implementation of control systems regarding concentrations of POPs in waste.

Before going into details of the challenges of analysing POP contents of wastes, some general remarks regarding analysis of POPs should be addressed. A UNEP guidance document has been developed, defining a universally valid framework for developing reliable and comparable analytical methods for POPs [UNEP 2007]. These guidelines can assist to generate results of comparable quality that form the basis for the mutual acceptance of data among countries. If customers can rely on analytical results developed in other countries or other laboratories, duplicate works and testing can be avoided thereby saving time and money.

In most aspects, POPs analysis does not differ from any other chemical analysis for determination of the mass concentration of an analyte of interest in a given matrix. Therefore, elements of common sense practices such as aspects of the “Principles of Good Laboratory Practice” [OECD 1998] and other accepted guidelines regarding quality control system should be taken into account when developing or implementing methods for performing analysis to serve the multilateral agreements such as the Stockholm, Rotterdam or Basel Conventions. To generate comparable results, it is essential to not only establish standardised analytical methods but to make sure that sampling, transport and storage of the samples and analysis (extraction, purification, separation, identification, quantification and reporting) meet high standards.

- no fixed method for matrix „waste“ from an analytical point of view
- no fixed analytical sensitivity (quality criteria) as no fixed matrix

Thus, it can be stated that from the variety of available analytical methods including pre-treatment, extraction, clean-up, and measurement not every method may be applicable for every waste matrix. On this basis it is clear that there hardly can be only one single measurement standard (reference method) covering all analytical challenges of the matrix “waste”.

On the other hand, POPs analysis shall provide reliable and comparable data regardless the individual waste matrix. Especially for the control of regulatory limits this means that analysis should be – as far as possible - based on methods that meet certain quality criteria and – on this basis – provide comparable results.

Laboratories may adopt EU / EPA / CEN methods or other published methods for sample extraction, clean up, and analysis, and have to validate them within the laboratory. The most basic requirements are:

- The laboratory must be able to prove competence for infrastructure, instrumentation, and well-trained staff to conduct specific analyses;
- Validation of the analytical methods including in-house methods;
- Standard operating procedures (SOPs) for the validated methods, including all the laboratory equipment and consumables;
- Quality criteria for QA/QC described in the SOPs, e.g., analysis of blank samples, use of reference materials, signal/noise ratio, and sensitivity of the analytical system.

It is primarily the task of standardisation bodies such as CEN, ISO and national standardisation bodies to provide standardised and validated methods. For the scope of the European Union CEN as the Community standardisation organisation constituted by national standardisation bodies is the relevant institution for this task. CEN has already been mandated by the Commission to develop important waste related standards. It can be recommended to continue along this line.

Usually, standard analytical methods cover only a special or a limited set of matrices. CEN has started a process on development of standards covering more matrices which lead to equivalent results as far as technically feasible – so called Horizontal Standards. One specific area of production of horizontal standards is in the field of bio waste, sludge and soil, where there is a need for consistency of standards in regard to the existing and upcoming EU regulation. A Task Force (CEN BT TF 151 – Horizontal standards in the field of sludge, soil and bio waste), with the task to take the relevant steps for production of horizontal standard in the given area has been established. The experiences obtained in this project will be an important support for the development of tailor-made waste standards based on CEN-work and already provide important information with respect to sampling and analysis of POPs, e.g. PBDEs.

7.3 Basic principles of standard development

Scientific literature provides a variety of methods or method modifications for certain questions. However, a suggestion for reference methods in context with the control of regulatory limits has to be based on broadly accepted and validated methods which are primarily represented by methods from standardisation bodies.

Standards for monitoring of POPs shall be any relevant CEN/ISO standards or such other national or international standards, which will ensure the provision of data of an equivalent scientific quality and comparability. The strengths of such methods are [Leprom et al., 2009]:

- (1) they have been developed on basis of established “in-house methods” by experienced laboratory experts,
- (2) they are well-established, widely accepted and easily available,
- (3) they mostly have been subjected to collaborative trials to demonstrate their interlaboratory comparability and applicability,
- (4) there is little effort to implement them as only verification and no full validation is required,
- (5) it is possible to refer to them in legislation.

On the other hand, standard methods also feature certain drawbacks, which should be considered when selecting a method for a particular purpose. These include:

- (1) standardisation takes quite a long time and hence, such methods may not always represent the current state of the art,
- (2) they often represent a compromise in performance that is tailored to a number of different users’ goals and operational needs and have not been developed specifically for waste monitoring (in most cases a special kind of waste like plastics is addressed),
- (3) usually they offer little flexibility to the user to choose from different options depending on the nature of samples (different sort of wastes) and on what equipment and experience are available, and
- (4) they do not take account of recent technical developments.

Anyway, the use of standard methods in routine monitoring of environmental pollutants is already widespread across Europe although the procedures have often been slightly modified and adjusted to the specific measurement requirements, to the nature of the sample or according to the available equipment and the existing know-how. Standards are often used as reference methods or as vantage

point for further method development. An equivalent status has to be achieved in the field of POPs analysis in waste.

Besides the recommendation of well defined individual methods it is an accepted strategy used by authorities and standardisation bodies to define critical performance or quality criteria to be met by analytical methods in order to obtain accepted results. The advantage of this strategy is that it admits highest flexibility in method development (e.g. inclusion of cost efficient techniques; adaptation of methods to complex waste matrices) without neglecting quality related issues.

Methods applied for POPs monitoring should clearly be described, properly validated and give laboratories the flexibility to select from several options when possible and meaningful. Irrespective of what method is applied in chemical monitoring, certain minimum performance criteria have to be met as discussed before. In other words, any fully validated method meeting those criteria may be used.

Following this approach of critical performance criteria, in a first step available standardised methods and literature methods for POPs in waste materials have to be reviewed on a world-wide basis including the identification and evaluation of their performance/quality criteria. By identifying consensus or minimum criteria a suggestion for criteria for reference methods could then be performed. However, this is a complicated process considering the complexity of the matrix waste and the variety of individual analytical processes to be regarded (sampling, pre-treatment, extraction, clean-up, measurement, QA/QC).

As currently comprehensive activities are on the way concerning the harmonisation of analysis in this field, so the project team supposes not to anticipate upcoming results and recommendations of ongoing harmonisation activities to which - among others - also standardisation bodies are involved (e.g. considering the methodical validation effort necessary to support a suggestion for reference methods).

Therefore, the project team has to confine itself on collecting currently available information on standard methods as well as literature methods and try to illustrate relevant performance/quality criteria in view of a proposal of such criteria for POPs analysis in wastes. In the case of no existing standard methods for POPs substances covered by this report, a literature overview of methods used to determine the substances in other matrices than waste will be provided.

Final definition of reference methods and performance/quality requirements has to be based on outputs from standardisation bodies in order to achieve accepted methods. The proposals as presented by this project have therefore to be updated in accordance with up-coming results of the official standardisation and harmonisation process.

7.4 International Overview of Measurement Methods for new POPs and candidate POPs

In the following chapter the results of a world-wide investigation on measurement methods for detection of POP concentrations in different matrices are compiled and discussed separately for the new and candidate POPs.

Critical performance/quality

Today individual analytical methods are able to detect the target compounds at quite low levels. Great progress has been made in the last years regarding detector systems like Mass Spectrometers. GC-MS and LC-MS systems are in most cases able to detect analytes in concentrations in the range of ppb or even ppt. However, these levels are not per se achievable for complex waste matrices with the original standard method. The most critical step in analysis of contaminants in different matrices however is the extraction of the target compounds from the respective matrix. Efficiency of extraction is highly depending from the individual properties of the relevant matrix. In case of interfering substances being co extracted, sample clean-up has to be performed which might cause loss of target compounds. These difficulties can be overcome by the use of internal standards having a similar structure and thus similar properties as the relevant target compound. Major problems that hamper the development of reference standard methods are in many cases the current lack of pure calibrants or traceability to them as a basic prerequisite for requested comparable and reliable measurement data and the method validation.

For many POPs specific standard methods for analysis in waste are not yet available. Nevertheless, in most cases analytical methods suitable to detect the target compounds in environmental matrices like soil or sludge at quite low levels have been reported in scientific literature. Critical performance/quality criteria taken from these methods include quantification method (internal / external standard method, isotope dilution), recovery rates, tolerable retention time deviations, chromatographic separation efficiencies. Nevertheless, again, most of these criteria are directly linked to the chosen sample separation steps and the chosen analytical technique and thus to the type of matrix (waste). Therefore, standard guidelines can and should only define limits for the performance/quality criteria which have to be met when analysing POPs in waste matrices. Suitable quality test by means of interlaboratory (round robin) tests with use of certified reference materials may be of higher importance as the method alone. This QC is especially essential when different analysis methods will be allowed, which might be necessary, as not all laboratories own the latest highly sophisticated equipment. In general, trying to improve sensitivity needs extra effort (e.g. repeat standard clean-up, include further clean-up steps and turn to more specific measurement (e.g. HRMS) with increased costs. Nevertheless, analytical methods to control compliance with limit values regarding concentrations of POPs in waste matrices have to be practicable with standard laboratory equipment, which means that critical performance criteria have to be achievable with common analytical equipment.

With respect to method validation there are further problems with the harmonisation of quality criteria and control (QA/QC). Although issues like LOD and LOQ are addressed in accredited laboratories in all of the Member States there is no consensus on how to calculate these and other methods performance criteria.

Laboratory capacities

Current extraction techniques (such as ASE) and increasing automation of clean-up steps have led to reduced analyses periods and increased capacities. From this point of view, a high analytical capacity in routine laboratories is already available in Europe even if laboratory capacity is not homogeneously distributed all over Europe. Lack of analytical capacity in almost all developing countries and access to POPs laboratories has been considered one of the crucial issues for countries to fulfil their obligations of the Stockholm Convention. In order to assess the existing capacity worldwide and the capacity building

needs in developing countries to analyse persistent organic pollutants (POPs), the Global Environment Facility (GEF) approved the project "Assessment of Existing Capacity and Capacity Building Needs to Analyse Persistent Organic Pollutants (POPs) in Developing Countries". The outcomes of this UNEP/GEF project include a databank of operational laboratories worldwide according to their capabilities to analyse classes of POPs in different matrices and recommended criteria for sampling, identification, quantification of POPs.

Based on the experience of the rapid development of appropriate analytical capacity after the implementation of maximum levels for POPs in waste, it can be expected that routine laboratory capacity will be rapidly adapted to the requirements of the analytical market.

According to the comments on the draft final report from JRC-IRMM this is a too optimistic evaluation of the real situation within the EU but also in the rest of the world⁴². According to the comments, recently JRC-IRMM has conducted a proficiency test for the determination of brominated flame retardants in plastic, (IMEP-26, the report of this exercise will be available before summer). The outcome of the exercise is far from being positive. Only 23 laboratories from all around the world took part in the exercise, 12 from the EU and 11 from outside the EU. Accreditation Bodies from Europe and the Asia Pacific region were invited to nominate accredited laboratories in this type of analyses to take part in IMEP 26. IRMM was informed by the European Cooperation for Accreditation that it has been difficult to find accredited laboratories in this field. In several European countries there is no accredited laboratory to analyse polybrominated flame retardants in plastic. Only 11 of the 23 participants were accredited, 8 of them from the EU. In general, only about 50 % of the participants reported satisfactory results for most of the measurands covered in the exercise.

In this context it is noteworthy that the Irish EPA pointed out that it is not aware of any laboratory in Ireland at this point in time that are providing a commercial service to undertake analysis of PBDEs e.g. in shredder residues ([EPA IRE 2011b]; see chapter 6.1.9. section F).

7.4.1 PBDEs

The most common use of C-PentaBDE in Europe was in flexible polyurethane (PUR) foams, accounting for approximately 95 % of the total use of C-PentaBDE. The treated PUR foams were in turn mainly used for the production of automotive and upholstery applications (e.g. automotive seating, head rests, sofas, mattresses etc.). Other applications and finished articles containing or reported to be possibly contaminated with C-PentaBDE are Polyvinylchloride (PVC), Epoxy Resins, Unsaturated Polyesters (UPE), Rubber, Paint/Lacquers, Textiles, Hydraulic oils and possibly other.

This study exclusively focuses on C-PentaBDE treated PUR foams and associated applications and finished articles (i.e. automotive and upholstery applications).

The main historic use of C-OctaBDE was in Acrylonitrilebutadiene-styrene (ABS) polymers. Around 95 % of C-OctaBDE supplied in the EU was used in ABS. The ABS in turn was mainly used for housings/casings of Electrical and Electronic Equipment (EEE), typically office equipment and

⁴² Comments from JRC-IRMM related to the draft final report. Comments received on 31.03.2011.

business machines. Other minor uses (5 %) were High Impact Polystyrene (HIPS), Polybutylene Terephthalate (PBT), Polyamide Polymers, with typical concentrations between 12-15 %. Other possible uses found in literature were nylon, low density polyethylene, polycarbonate, phenol-formaldehyde resins, unsaturated polyesters, adhesives and coatings.

The main focus of this study is on the use of C-OctaBDE in EEE, in particular on the ABS housings/casings of office equipment and business machines.

7.4.1.1 Existing Methods

Besides already existing standard analytical methods, various different methods have been identified for quantitative and qualitative detection of PBDEs in different matrices (e.g. plastics, sewage sludge, etc). The identified standard and literature methods are listed in Table 7-3 and Table 7-5. Some of the listed methods require sampling, chemical preparation (e.g. extraction of the brominated compound by different techniques) and a quite well-equipped laboratory. Different chromatographic techniques coupled with mass spectrometry are usually used for detection and analysing this type of compounds. However, PBDEs are difficult to identify without a thorough lab analysis, which is time-consuming and not feasible to apply in every sorting facility or landfill.

7.4.1.1.1 Screening methods

A study to develop a process to extract brominated flame retardants (BFRs) from WEEE polymers has been carried out under the Waste Resources Action Programme [WRAP 2005]. The report reviews different techniques available for identification of polymer types and whether or not polymers contain BFRs. In a second step techniques were assessed for their suitability for use on automated sorting lines. It was concluded that because of the way flame retardants are incorporated into plastics and the concentrations in which they are present the methods for their detection are limited to techniques based on spectroscopy. BFR detection in recycling is in its infancy and many of these techniques have been developed for the laboratory rather than a recycling line. This means they are generally expensive, relatively fragile and not particularly user-friendly. Of the reviewed techniques, surface ablation sliding spark and FT-mid IR appeared to be the most suitable for BFR detection with X-ray fluorescence in third place. The instruments in question are similar in many ways and currently appear to set the benchmark for this application. The following summarises relevant information from the WRAP report.

X-Ray Fluorescence

When the presence of BFRs, such as PBDEs, is to be detected without physical alteration of the material (for example in monitoring of plastic toys), and especially in situ measurements for plastics, XRF (X-Ray Fluorescence) seems to be the method of choice. This method is relatively cheap, does not need special training and can be done on site. XRF can be used in plastic waste sorting facilities for control of the accuracy of the applied sorting methods. As with any other measurement method there are pros- and cons related to the accuracy. XRF analysis is limited to the detection of bromine in the material, without having the possibility to identify the type of compound. False positive detects for the presence of PBDEs may occur due to the presence of BFRs other than PBDEs. Furthermore, PBDE congeners cannot be distinguished. Quantitative estimation of PBDEs is possible after calibration of the XRF device for a range of PBDEs with the help of GC-MS or similar techniques used otherwise for the specific lab analysis [SEPA

2010]. Detection limits of 0.4 mg/g have been reported for penta-BDE in polymers [Bergmann 2007]. The results of a study on the applicability of handheld XRF systems for waste characterisation initiated by the German states working group on waste (LAGA) demonstrated that handheld XRF instruments are useful tools for the screening of waste. It was concluded that reliable information can be gathered with this method regarding the presence or absence of elements, and in most cases also in terms of magnitude of the concentration levels of the elements present. The easiness of operation of the handheld XRF instruments and the low cost of the sample preparation are the major advantages compared to alternative analysis methods. Handheld XRF analysers are the detection method of choice for the initial analysis of elements in waste [LAGA 2008]. Summarising the above, XRF can be used as screening method to detect presence of brominated flame retardants in waste samples. In the case of positive results a deeper analysis with standardised analytical methods (e.g. GC-MS) will be necessary to determine the exact PBDE content of the waste samples.

Drinks bottles are commonly made of PVC or PET. These items are often collected under recycling schemes. The requirement for very low levels of PVC in a PET product led to the development of automated sorting systems based on identification of PVC using XRF in the 1980s. When x-rays bombard the surface of a material secondary x-rays are generated which have wavelengths characteristic of the elements contained within the material. The production of x-rays from excited chlorine atoms is used to identify PVC. Following identification a computer controlled air ejection system can be used to remove the PVC bottles from a mixed plastic waste stream. The separators, with multiple analyses (up to 200 per second) on each bottle, are capable of highly efficient removal of PVC. The detector must be close to the bottle because of the attenuation of x-rays in air [WRAP 2005]. Similar sorting systems may perhaps be suitable to sort out bromine containing parts in recycling of plastic waste although further investigations are necessary regarding this issue.

Sliding-Spark technique

The basic principle of the Sliding-Spark technique method is the thermal vaporization of a small amount of the material surface using a train of defined high-current sliding sparks. The material components in the spark by plasma are vaporized, atomized and, activated to emit radiation. Paints or contamination on the surface can be automatically removed by pre-sparking. Additive detection is performed by measuring simultaneously the characteristic emission lines of the elements of additives in the spectral region of approx. 210-500 nm. The intensities of selected spectral lines are compared with preset threshold values. After calibration with known samples, the system enables semi-quantitative analysis of inorganic contents down to the sub-%-region. A measurement time of 1 second is claimed.

Fourier Transform Infrared Spectroscopy (FT-IR)

FT-IR is less specific in the way it detects BFR. FT-IR instruments have already been used on IT recycling lines and have successfully segregated manually-dismantled computer casings into streams with and without BFR. FT-IR gives an identification of unknown plastic sample by type of polymer. The nature of its operating methodology means that the result is a 'percentage fit' of the spectrum collected from the unknown sample, to the spectra stored in its library of reference samples. It is thus susceptible to making wrong ID's (or giving 'no match') if the level of IR reflectance is low (as with dark colours or coated surfaces). It is dependant however on having reliable known samples in its library for comparison

of spectra. It is proven that this technique can detect the presence of BFR's because they produce small differences in the reflected spectra. Most new infra-red spectrometers use a specular reflectance technique in which infrared light that is reflected from the surface is analysed. The surface of the object should therefore be relatively flat (a potential disadvantage in some systems but unlikely to be a problem in manual WEEE recycling). Analysis times greater than 1 second are commonly used. As the object has to be held stationary at the detector the technique does not lend itself perfectly to automated sorting.

Laser Induced Breakdown Spectroscopy (LIBS)

The LIBS technique identifies the presence of BFR by detecting Antimony (Sb) in the Br synergist compound Sb_2O_3 . Laser Induced Breakdown Spectroscopy (LIBS) (also known as Laser Induced Plasma Spectroscopy (LIPS)) and Laser Induced Multi Emission Spectroscopy (LIMES) are simple, relatively inexpensive analytical techniques to determine the elemental composition of a sample, regardless of whether the sample is a solid, liquid or gas. A powerful laser is focused onto the sample and hot plasma is formed. Light emitted by the plasma is composed of spectral lines characterising the element present in the sample. This technique is sensitive to all elements, with typical limits of detection between 0.1-200 parts per million (depending on the sample and the element of interest). Minimal sample preparation is needed making it quick and easily adaptable to automated chemical monitoring equipment or portable units. Variation in the laser operating conditions, such as increased power, will cause a variation in the type of molecules generated at the surface and the 'signature' for a particular polymer will change. The technique therefore needs well-controlled operating conditions for consistent identification. The time taken for spectrum acquisition of approximately 0.1 to 1 second suggests the technique is applicable for the identification of plastics on a recycling line. Stepputat & Noll ran a conveyor line at 0.5m/s and LIBS with an auto-focusing system identified the metals Cr, Hg, Cd and Pb in plastic samples with an accuracy of 84-95%. They had more difficulty with Br relying on determining the presence of BFR's in the plastic by identifying Sb in the synergist compound Sb_2O_3 . In the laboratory, they could detect Br directly at levels down to 1.5% by weight of plastic. This is sufficient to detect BFR in plastics which normally occur in concentrations exceeding 3% wt. However, LIBS has been regarded as not being perfectly accurate and the equipment is very expensive. This method has been designed for lab rather than industrial use.

Comparison

Following the conclusions from the review of available screening techniques for polymer types and BFR content, a practical, working comparison of FT-Infra Red Reflectance, X-Ray Fluorescence and Sliding Spark spectral analysis has been performed (Table 7-1).

Table 7-1: Comparison of screening methods [WRAP 2005]

Instrument Type	FT-Infra Red Reflectance	X-Ray Fluorescence	Sliding Spark spectral analysis
Prime Function	Detect polymer type + some additives	Detect and quantify additives and bromine	Detect + quantify bromine and chlorine

Instrument Type	FT-Infra Red Reflectance	X-Ray Fluorescence	Sliding Spark spectral analysis
Method of operation	Measures absorption of range of IR light wavelengths on sample surface and compares with library of known reference samples to give % fit to ID type.	Low power X-ray penetrates ~10mm into sample, detector measures distinct energy peaks from fluorescence of a range of elements to give ID and % conc. of additives	High voltage spark on surface creates plasma of vaporised material. Light spectra analysed for known peaks at Br and Cl wavelength to estimate % conc.
Approx Cost €	26,500	30,000	3,900
Size	Benchtop box approx 1000 x 500 x 200mm	Handheld, bulky gun	Hand-held detector with small bench-top box
Weight approx	25 kilos	1.7 kilos	0.75 kilo (gun)
Portability	Zero – needs firm stable worktop + separate laptop PC	Excellent – battery powered no cables.	Okay – light to carry, but needs mains power.
Ruggedness	Low – treat as laboratory equipment	Excellent – built for on-site use in scrap-yards	Good – but care needed for fibre optic cable
Start-up speed	Slow, 15 minutes to calibrate etc	Okay – 2-3 min	Fast - < 1min
Sample presentation & speed	Difficult with big samples and must keep still for 30 seconds	Easy – hold gun on sample for 15 – 30 seconds	Very easy – 1 second to 'fire' spark
Adjustment	Tricky – need background measurement and to re-align laser unit	Not a problem	Easy to adjust base level reading to remove noise.
Sample preparation	Flat non-shiny surfaces best. Dark colours more difficult, surface coatings a problem	Thicker samples better (> 5mm). Will detect surface contamination & coatings. Good for granular plastics.	Clean surface required and flat area for good spark contact. Will detect dirt and coatings on surface.
Measurement - - Accuracy	Depends on closeness of match to library samples	Very good – ppm levels of elements	Sufficient – to nearest 1% on Br / Cl concentration
- Repeatability	Good if high match level	Excellent	Good – some noise around 0-1% level
- Reliability	Average	Very good	Very good
- Speed	Slow – 30 secs	Slow 15-30 secs	Fast – 1 sec
- User confidence	Average – needs degree of judgement	Good on primary elements, lower on chlorine.	High – except at 1% conc. level
Operator Skill level required	Good technical / laboratory person	Technical operator to interpret results	Factory operator with basic training

7.4.1.1.2 Methods to determine the PBDEs content of waste samples

Summary of analytical strategies for quantitative analysis of PBDEs

Commercially produced PBDE mixtures contain a limited number of congeners. PBDEs are typically produced at three different degrees of bromination, i.e., Penta-BDE, Octa-BDE and Deca-BDE. Penta-BDE formulation consists of 41 % to 42 % tetra-BDEs (mainly BDE- 47) and 44 % to 45 % penta-BDEs (predominantly BDE-99 and BDE-100), whereas Deca-BDE formulation consists mainly of BDE-209 (97 % to 98 %), with a small amount of nona-BDEs (0.3 % to 3 %). Hepta-BDE-183 is often taken as indicative of the presence of the Octa-BDE formulations. As technical products show different congener patterns, calculation of the amount of a technical PBDE preparation in a waste matrix on the basis of one congener however, does not seem reasonable. Furthermore, the matrix waste is not a homogenous

matrix from a single origin and thus may contain PBDEs resulting from different technical products used in the production of the single parts of the matrix. Therefore, all four congener groups listed in the POP Regulation should be analysed to calculate the PBDE content of waste matrices. Most of the available methods analyse the PBDE congeners which are dominating the commercial PBDE products. Various isotope-labelled and unlabeled PBDE standards are available. PBDE congeners used as standards in the individual analytical methods are listed in Table 7-3 and Table 7-5. Table 7-2 shows the congeners used as “indicator isomers” for tetra- to deca-PBDE in most standard and literature methods.

Table 7-2: Standard substances used in reviewed analytical methods as “Indicator” isomers

Congener	Formula	Numbering analogue to IUPAC nomenclature for PCB
2,2',4,4' Tetrabromodiphenylether	C ₁₂ H ₆ Br ₄ O	BDE -47
2,2',4,4',5 Pentabromodiphenylether	C ₁₂ H ₅ Br ₅ O	BDE -99
2,2',4,4',6-Pentabromodiphenylether	C ₁₂ H ₅ Br ₅ O	BDE-100
2,2',4,4',5,6'-Hexabromodiphenylether	C ₁₂ H ₄ Br ₆ O	BDE-154
2,2',4,4',5,5'-Hexabromodiphenylether	C ₁₂ H ₄ Br ₆ O	BDE-153
2,2',3,4,4',5',6-Heptabromodiphenylether	C ₁₂ H ₃ Br ₇ O	BDE-183
2,2',3,3',4,4',5',6-Octabromodiphenylether	C ₁₂ H ₂ Br ₈ O	BDE-196
2,2',3,3',4,4',6,6'-Octabromodiphenylether	C ₁₂ H ₂ Br ₈ O	BDE-197
2,2',3,4,4',5,5',6-Octabromodiphenylether	C ₁₂ H ₂ Br ₈ O	BDE-203
Decabromodiphenylether	C ₁₂ Br ₁₀ O	BDE-209

Extraction of PBDEs from waste matrices and separation from interfering substances extracted with PBDEs is considered to be the most challenging part in waste analysis. As already mentioned waste defines several matrices requiring different extraction and clean up strategies.

PBDEs are typically separated from plastic matrices, biological and environmental media by extraction with organic solvents. Liquid-solid extraction (e.g., Soxhlet apparatus) remains a widely used technique for solid samples despite recent advances in other extraction techniques. Typical solvents are hexane, toluene, hexane/acetone mixtures, or dichloromethane. Soluble polymer samples are completely solubilised in suitable solvents to extract PBDEs.

New extractions techniques, such as accelerated solvent extraction (ASE) or microwave-assisted extraction (MAE), are also used by a number of laboratories. The advantage of these techniques is lower solvent consumption and reduced extraction time. Supercritical fluid extraction (SFE) with solid-phase trapping has been used for the extraction of brominated flame retardants from sediment with CO₂ as the supercritical fluid. Extraction with pressurized hot water (PHWE) has been used for the analysis of brominated analytes from sediment. Liquid-liquid extraction has been applied for river and seawater samples, using hexane/acetone mixtures. Nevertheless, Soxhlet extraction despite being time consuming and requiring higher amounts of solvent seems to be the most popular extraction method. It does not require special, expensive equipment, is easy to handle and seems to be the extraction method with the highest recovery rates.

Clean-up has to be used if compounds are present that can interfere (co-elute) with the PBDE congeners of interest. Depending on the different matrices, a variety of sample clean-up procedures are suitable. The complexity of waste matrices implies the use of multi-step purification methods. Waste extracts may contain relatively large amounts of elemental sulphur, which would disturb the GC analysis, and must be removed. Typical methods for sulphur removal are treatment with copper powder, silica modified with AgNO_3 in a multi-layer silica column, or desulfuration with mercury. Optionally, elimination of elemental sulphur could be done simultaneously with the extraction procedure, by addition of copper powder to the Soxhlet beaker or to PLE extraction cell. Waste samples may also contain high level of organic matter that disturbs the well operation of purification via column chromatographic procedure. For this reason, in most cases, it may be useful to start the purification step by a treatment with concentrated sulphuric acid to remove organic matter. Clean-up stage may then be performed with the aim of purification as well as fractionation of the extract, using a great variety of adsorbents (silica, Florisil, alumina ...). Extracts from polymer samples have been analysed without further clean up. Nevertheless, it is important to consider the effects of the clean-up procedures on recovery and sensitivity of the complete analytical method.

GC coupled to either negative chemical ionisation (NCI) or electron ionisation (EI) low resolution mass spectrometry (LRMS) are the alternatives more frequently used for PBDE analyses. NCI-LRMS is by far the most popular technique due to high level of sensitivity provided. However, if high specificity and accuracy in quantification is needed (for instance, by using isotopic dilution method), EI-LRMS shows better performance, at the expense of lower sensitivity. Another MS detection technique that may be used is the quadrupole ion trap (QIT)-MS in a tandem mode, which has been shown to have good selectivity and to be close to NCI in sensitivity. GC-high resolution mass spectrometry (HRMS) based analytical methods are also available for determination of congener specific PBDE compounds, providing the most selective and sensitive method. However sophisticated and expensive instruments which require a trained personnel and frequent maintenance are needed for HRMS. GC-ECD has also been used due to the high sensitivity of this detector for PBDEs. LC-MS methods have also been reported for the determination of PBDE levels in soluble polymers. An overview of analytical methods reported in literature and the respective performance criteria can be found in Table 7-3.

Table 7-3: Summary of the results of a survey on analytical methods to be considered for PBDE analysis reported in literature

Method	Matrix	Extraction	Clean-up	Detection	Congeners/Standards	LOD and LOQ	Reference
Literature: Flame retarded polymers	Polymers	Soxhlet extraction with toluene	Colum chromatographic purification H ₂ SO ₄ /silica	HRGC-HRMS	PBDE not further specified	Not applicable	[Riess et al., 2000]
Literature : BFRs in plastic waste	Waste of electrical and electronic equipment	Ultrasonic extraction with toluene	-	GC-ECD/MS	penta-BDE octa-BDE deca-BDE not further specified	Not applicable	[Morf et al., 2000]
Literature: Leaching test/PBDEs in plastic products	Plastics	Ultrasonic with toluene Liquid/liquid with dichlormethane from leachate	Silica gel column chromatography	HRGC/HRMS	Mono-BDE (BDE-3) di-BDE (BDE 15) tri-BDE (BDE 28) tetra-BDE (BDE 77) penta-BDE (BDE 100) hexa-BDE BDE 154) hepta-BDE (BDE 183) octa-BDE nona-BDE (207) deca-BDE (BDE-209)	0.05 µg/g (LOQ) 0.05 µg/g (LOQ) 0.05 µg/g (LOQ) 0.05 µg/g (LOQ) 0.1 µg/g (LOQ) 0.1 µg/g (LOQ) 0.2 µg/g (LOQ) 0.2 µg/g (LOQ) 0.3 µg/g (LOQ) n.a.	[Kim et al., 2006]
Literature: PBDEs i styrenic polymers	Styrenic polymers	Ultrasonic-supported dissolution	Precipitation (USDP)	GC-MS/GC-ECD	BDE congeners 28, 47, 49,66, 85, 99, 100, 139, 138, 153, 154, 171, 182, 183, 196, 197, 201, 203, 206, 209 and 207	2.4-6.9 mg/kg (LODs) GC-MS 1.4-9.0 mg/kg (LODs) GC-ECD	[Pöhlein et al., 2008]
Literature: PBDEs and metabolites in environmental samples	Sediment, fish, milk	PLE extraction hexane/C ₂ Cl ₂ (1 :1)	GPC clean-up; Florisil chromatography	GC-HRMS	32 different PBDEs (PBDE 7 – 209) identified in the samples + hydroxylated and methoxylated metabolites	3.28 pg/g dry weight (LOQ sediment) 20.5 pg/g lipid weight (LOQ milk) 41.4 pg/g lipid weight (LOQ fish)	[Lacorte et al., 2010]
Literature: PBDEs in sediment	Sediment	SPME	-	GC-MS/MS	tetra-BDE (BDE 47) penta-BDE (BDE 85) penta-BDE (BDE 99) penta-BDE (BDE 100) hexa-BDE BDE 153) hexa-BDE BDE 154)	0.05 ng/ml (LOQ) 0.40 ng/ml (LOQ) 0.10 ng/ml (LOQ) 0.15 ng/ml (LOQ) 0.40 ng/ml (LOQ) 0.50 ng/ml (LOQ)	[Montes et al., 2010]

Certified Reference Materials (CRMs) are cornerstones of modern analytical quality assurance because they allow calibration of instruments, validation of analytical methods, and quality control of methods and laboratories based on traceability and comparability of measurement results. The Institute for Reference Materials and Measurements (IRMM) provides certified reference materials, produced by the EC-JRC-IRMM. At the moment there are reference materials for PBDEs in Polyethylene (LDPE) (ERM-EC590) and in Polypropylene (PP) (ERM-EC591) available. Detailed composition of the reference materials provided by IRMM is given in Table 7-4 [IRMM 2010].

Table 7-4: Certified reference materials commercially available from IRMM. Values in brackets are not certified. Uncertainties are expanded uncertainties corresponding to a level of confidence of about 95 %.

Substance	ERM-EC590 Polyethylene (LDPE) g/kg	ERM-EC591 Polypropylene (PP) g/kg
Br	2.13 ± 0.09	2.08 ± 0.07
2,4,4'-TriBDE (BDE-28)		0.0025 ± 0.0006
2,2',4,4'-TetraBDE (BDE-47)	0.23 ± 0.04	0.245 ± 0.023
2,2',3,4,4'-PentaBDE (BDE-99)	0.302 ± 0.060	0.32 ± 0.04
2,2',4,4',6-PentaBDE (BDE-100)	0.063 ± 0.016	0.066 ± 0.007
2,2',4,4',5,5'-HexaBDE (BDE-153)	0.047 ± 0.012	0.044 ± 0.010
2,2',4,4',5,6'-HexaBDE (BDE-154)	0.0257 ± 0.007	0.026 ± 0.006
2,2',3,4,4',5,6'-HeptaBDE (BDE-183)	0.132 ± 0.012	0.087 ± 0.008
2,2',3,3',4,4',6,6'-OctaBDE (BDE-197) + 2,2',3,4,4',5,6,6'-OctaBDE (BDE-204)	0.076 ± 0.015	0.052 ± 0.009
DecaBDE (BDE-209)	0.65 ± 0.010	0.78 ± 0.09
DecaBB (BB-209)	0.63 ± 0.10	0.74 ± 0.08
Sb	(0.756 ± 0.025)	(0.713 ± 0.022)

Standard methods for qualitative and quantitative analysis of PBDEs

Regarding determination of PBDEs in waste, there is no standard analytical methods available covering waste in general. Nevertheless, different standard methods concerning PBDE analysis in waste relevant matrices or recycling materials are available or are under preparation. Furthermore reported standardised methods can easily be adapted for relevant waste categories. A standard method for determination of PBDEs in polymers might e.g. be suitable for determination of PBDE content in plastic waste or corresponding recycling matrices. Identified standardized methods and projects to establish standardised methods are listed in Table 7-5.

ISO 22032:2006 specifies a method for the determination of selected PBDEs in sediment and sludge using gas chromatography/mass spectrometry (GC-MS). When using GC-EI-MS, the method is applicable to samples containing 0.05 to 25 µg/kg of tetra- to octabromo congeners and 0.3 to 100 µg/kg of decabromo diphenyl ether (BDE-209), respectively. At the moment the Ministry of Agriculture, the Environment and Consumer Protection of Mecklenburg-Vorpommern (Germany) as managing country for the German federal states financing the programme "Water, Soil and Waste", in cooperation with

the Task Group of WG 5 of CEN/TC 292 „Characterization of waste“, has started a project with the objective of adaption of ISO 22032:2006 to waste matrices and following validation by at least 20 laboratories to create a new European standard method. First results will be available in 2011 [ISO 22032].

Standard method IEC 62321, Edition 1 provides test procedures for determining the levels of lead (Pb), mercury (Hg), cadmium (Cd), hexavalent chromium (Cr VI) and their compounds, and two types of brominated flame retardants, polybrominated biphenyls (PBB) and polybrominated diphenyl ethers (PBDE) (except decabrominated diphenyl ether, DecaBDE), contained in electrotechnical products. The International Electrotechnical Commission (IEC) is a worldwide organization for standardization comprising all national electrotechnical committees (IEC National Committees). The proposed method follows the UBA approach and is suitable to determine PBDEs in plastic waste and recycling material as well as in electronic products or polymer articles [IEC 62321].

A proposal for a standard analytical method for the determination of PBDEs in sediment, sludge (sewage and industrial), bio-waste and soil (BT WI CSS99044:2007) has been prepared by Technical Committee CEN/BT 151 “Horizontal”. The guideline is the result of a desk study "Horizontal Standardisation of Brominated Flame Retardants (BFRs)" in the “Horizontal” project and aims at evaluation of the latest developments in assessing PBDE in sludge, soil, treated bio waste and neighbouring fields. After an evaluation study, in which e.g. the ruggedness of the method was studied, a European wide validation of the draft standard has taken place. The results of the desk studies as well as the evaluation and validation studies have been subject to discussions with all parties concerned in CEN. The guideline provides different extraction (Soxhlet, PLE, SPME) and clean up (GPC, silica gel chromatography) procedures to be used depending from matrix characteristics. Furthermore, different GC-columns suitable to overcome several co-elution problems are provided. The final standard analytical method will be suitable for determination of PBDEs in some sorts of waste [HORIZONTAL 2007].

The German Federal Institute for Materials Research and Testing (BAM), on behalf of the Federal Environmental Agency (UBA) of Germany performed a study to optimise the individual steps (sample homogenization, extraction, GC/MS) of published analytical methods for determining penta- and octa-BDE in articles. The samples were prepared by means of Soxhlet extraction using toluene as an extraction agent. Identification and quantitative determination of PBDEs took place with the help of gas chromatography and mass spectrometry (GC/MS) in single ion monitoring (SIM) mode using three typical ion traces per isomer. The selection of the penta and octabrominated diphenylether isomers to be tested took place using the main isomers of the PENTA-BDE and OCTA-BDE technical products. The limit of detection of the developed method was 0.02 to 0.04 mg/g and the limit of quantitation was 0.06 to 0.12 mg/g. An international interlaboratory comparison was carried out. It has been shown that the developed standard method provides precise measurement results for the determination of the sum penta- and sum octa-BDE isomers respectively, in particular within the range of 0.1 % by mass (1 mg/g) under the specified conditions. However, the BAM method was developed to provide a method to verify the threshold value set by Directive 2003/11/EC for pentabromo and octabromo diphenylethers (0.1 %) and has not been optimised for detecting small amounts of PBDEs in plastic waste. The relative high LOQs reported for the BAM method are considered not to be suitable as limit values for PBDEs in plastic waste [UBA 2005].

Table 7-5: Summary of the results of a survey on standardised analytical methods to be considered for PBDE analysis.

Method	Matrix	Extraction	Clean-up	Detection	Congeners/Standards	LOD and LOQ	Reference
Standard methods							
EN 15309:2007 XRF-screening	Solid matrices (e.g. waste, soil)	Not necessary	Not necessary-	XRF	No, only Br content	Depending on matrix 0.4 mg/g for penta-BDE	[SEPA 2010] [LAGA 2008] [Riess et al., 2000] [Morf et al., 2005] [CEN 15309]
BT WI CSS99044:2007	Sediment Sludge - Sewage sludge - Industrial sludge Bio-waste Soil	Soxhlet extraction PLE SPME	Matrix dependent (GPC, multi layer silica, ...)	GC-MS/MDI	tetra-BDE (BDE 47) penta-BDE (BDE 99/100) hexa-BDE (BDE 153/154) hepta-BDE BDE 183) octa-BDE (BDE 203) nona-BDE deca-BDE (BDE-209)	Dependent from setup (LOQ, Soxhlet-SPE-GC-NCI-MS in sewage sludge) 90-1480 pg/g _{dw} 610-970 pg/g _{dw} 530-830 pg/g _{dw} 1280 pg/g _{dw} 660-1370 pg/g _{dw} 540-1140 pg/g _{dw} 4100 pg/g _{dw}	[HORIZONTAL 2007]
ISO 22032:2006	Sediment, sewage sludge	Not applicable	Not applicable	GC-MS/EI GC-MS/NCI	tetra-BDE (BDE 47) penta-BDE (BDE 99/100) hexa-BDE (BDE 153/154) hepta-BDE BDE 183) octa-BDE (BDE 205) deca-BDE (BDE-209)	0.05 – 25 µg/kg (LOQ) 0.05 – 25 µg/kg (LOQ) 0.05 – 25 µg/kg (LOQ) 0.05 – 25 µg/kg (LOQ) 0.05 – 25 µg/kg (LOQ) 0.3 – 100 µg/kg (LOQ)	[ISO 22032]
EPA Method 1614	Aqueous, solid, tissue, and multi-phase matrices	Solid phase separatory funnel and continuous liquid/liquid for aqueous liquids; Soxhlet/Dean-Stark for solids and filters; Soxhlet extraction for tissues.	Acid/base back-extraction is used for initial cleanup of extracts; Gel permeation chromatography for soil and sediment extracts; Acid, neutral, and basic silica gel, alumina and Florisil to remove non-polar and polar interferences.	HRGC/HRMS	tetra-BDE (BDE 47) penta-BDE (BDE 100) penta-BDE (BDE 99) hexa-BDE BDE 153) hexa-BDE BDE 154) hepta-BDE (BDE 183) octa-BDE deca-BDE (BDE-209)	100 pg/L (ML-W) 5 ng/kg (ML-other) 50 pg/L (ML-W) 5 ng/kg (ML-other) 100 pg/L (ML-W) 10 ng/kg (ML-other) 50 pg/L (ML-W) 5 ng/kg (ML-other) 50 pg/L (ML-W) 5 ng/kg (ML-other) 100 pg/L (ML-W) 10 ng/kg (ML-other) n.a. 2000 pg/L (ML-W) 200 ng/kg (ML-other)	[USEPA 1614]

Method	Matrix	Extraction	Clean-up	Detection	Congeners/Standards	LOD and LOQ	Reference
EPA Method 527	Water	Solid phase extraction (SPE) disk containing polystyrene-divinylbenzene (SDVB)	-	GC_MS	tetra-BDE (BDE 47) penta-BDE (BDE 99) penta-BDE (BDE 100) hexa-BDE (BDE 153)	0.028 µg/L (DL) 0.097 µg/L (DL) 0.051 µg/L (DL) 0.140 µg/L (DL)	[USEPA 527]
Method which were basis for standard methods or standard methods under elaboration							
BAM method (resulted in IEC 62321:2008)	Polymers	Soxhlet extraction with toluene	-	GC-MS	penta-BDE (BDE 85) penta-BDE (BDE 99) penta-BDE (BDE 100) octa-BDE (BDE 203) octa-BDE (BDE 196) octa-BDE (BDE 197)	0.06 mg/g (LOQ) 0.06 mg/g (LOQ) 0.06 mg/g (LOQ) 0.12 mg/g (LOQ) 0.12 mg/g (LOQ) 0.12 mg/g (LOQ)	[UBA 2005]
IEC 62321:2008 DIN EN 62321 EN 62321:2009	Electrotechnical products (polymers,) E-waste, plastics	Soxhlet extraction with - Toluene for ABS (Acrylonitrile Butadiene Styrene), HIPS (High Impact Polystyrene) and PC/ABS (Polycarbonate / Acrylonitrile Butadiene Styrene) - Propanol for Polyamides and Polyesters - 5/25 Dichloromethane/cyclohexane (B.P. 40 C/81 C) for Polyolefins Ultrasonic solvent extraction for soluble polymers	-	Screening by XRF GC/MS	Mono-BDE (BDE-3) di-BDE (BDE 15) tri-BDE (BDE 28) tri-BDE (BDE 33) tetra-BDE (BDE 47) penta-BDE (BDE 100) penta-BDE (BDE 99) hexa-BDE (BDE 153) hexa-BDE (BDE 154) hepta-BDE (BDE 183) octa-BDE (BDE 203) nona-BDE (206) deca-BDE (BDE-209)	Not yet available, but MDL < 100 mg/kg required.	[IEC 62321]

The Institute for Reference Materials and Measurements (IRMM) of the European Commission organised a proficiency test IMEP-26 for the "Determination of brominated flame retardants in plastic". This exercise aimed to assess the analytical performance of laboratories involved in the compliance with the European Directive on the "Restriction of the use of certain hazardous substances in electrical and electronic equipment" (RoHS) (Directive 2002/95/EC) and with the Regulation (EC) No 850/2004 on persistent organic pollutants implementing the Stockholm Convention provisions in the Union legislation. The concerned measurands were: total bromine, total sum of polybrominated biphenyls (PBB), total sum of polybrominated diphenylethers (PBDE), brominated diphenylethers (BDE-47, BDE-99, BDE-183 and BDE-209) and decabrominated biphenyl (BB-209). Deadline for submission of results was the 28th January 2011 so that first results should be available in 2011.

7.4.1.2 Conclusion

Based on the results of the world-wide investigation of reported measurement methods (see Table 7-3 Table 7-5) it can be stated that GC/MS techniques are state-of-the-art in the field of confirmatory analysis for PBDEs. Performance criteria are quite comparable and only selected extraction and clean-up procedures are used for analysing a broad spectrum of matrices. Although waste matrices are not extensively listed in the scope of the currently available methods, the listed methods are applied successfully to different matrices occurring in waste in routine practice. Table 7-6 provides an assessment of the suitability of the different analytical approaches to detect PBDEs in waste and recycling matrices:

Table 7-6: First assessment of the suitability of the different analytical methods to detect PBDEs

Method	Detection limit	Estimated costs	Onsite use possible (y/n)	Time until results	Training of staff needed	Waste matrix suitable for	Likelihood of method being applied onsite
XRF	0.4 mg/g	30,000 € once	y	30 sec.	low	Solid matrices	high
GC-MS	< 0.1mg/g dependent from matrix	400 € per sample if service laboratory	n/y	min. 3 days	very high	Extraction from every matrix possible	low
(HR)GC/HRMS	< 0.1 ng/g	Approximately 700 € per sample if service laboratory	n	min. 3 days	very high	Extraction from every matrix possible	very low

From this point of view, the available techniques are sufficient to detect PBDEs in quite low concentrations. Furthermore, activities are on the way to harmonise these methods in view of analysing certain type of wastes. Nevertheless extraction and clean-up procedures have to be adapted to the relevant types of waste (matrix).

Special attention must be paid on the potential co-elutions with other PBDE congeners, as well as with other brominated flame retardants, such as hexabromocyclododecane (HBCD), dimethyl tetrabromobisphenol A (MeTBBPA) and polybrominated biphenyls (PBB), which may also be present in waste samples. Furthermore, PBDEs are sensitive for UV light and sunlight and undergo coating solution

degradation if exposed. In general, the coating solution reaction rate increases with increasing number of bromine substituents, and thus are the perbrominated BDE-209 the most sensitive. Great care must be taken to avoid coating solution degradation of PBDEs throughout the analysis and the samples should be as kept in dark as far as possible. Especially for the analysis of deca-BDE-209, the exposure of the samples to high temperatures for long periods of times during the injection and separation stages have to be avoided, because of the thermal degradation of BDE-209 at temperatures higher than 300 °C [EFSA 2008].

7.4.2 PFOS and its derivatives

Perfluoroalkylated substances (PFAS) is the collective name for a vast group of fluorinated compounds, including oligomers and polymers, which consist of neutral and anionic surface active compounds with high thermal, chemical and biological inertness. Perfluorinated compounds are generally hydrophobic but also lipophobic and will therefore not accumulate in fatty tissues as is usually the case with other persistent halogenated compounds. An important subset is the (per)fluorinated organic surfactants, to which perfluorooctane sulfonate (PFOS) and perfluorooctanoic acid (PFOA) belong.

At present, PFOS is used in the metal plating industry, as content in hydraulic fluids in the aviation industry, in the photographic industry and the semiconductor industry. Another relevant source of PFOS is sewage sludge. The major waste stream from the different sources are metal plating, fire fighting foams, carpets and leather.

7.4.2.1 Existing Methods

Summary of analytical strategies for quantitative analysis of PFOS

Due to their relatively low volatility, good solubility in water and lack of chromospheres the analysis of per fluorinated alkyl substances is a challenging task. The analytical problems associated with the determination of neutral and anionic PFAS are multiple, and include diverse aspects such as unique physicochemical properties, lack of reliable standards, impurities, complicated mixtures of isomers and congeners, ion suppression, and contamination during all stages of the analytical procedure, including instrumental sources.

Analytical standards for per- and polyfluorinated alkylated substances are available from several manufacturers of fine chemicals. However, the purity of the non-isotopically labelled standards can vary considerably and may lead to systematic errors, as has been pointed out by Martin et al. [Martin et al., 2004]. For example, standards of alkanoic acids often contain short chain analogues. Moreover, the isomeric composition of these standards may also vary as a result of the production process used [Hekster et al., 2003]. Electrochemical fluorination will generally produce branched isomers next to the linear one and e.g. up to nine isomers have been shown to be present in a commercial PFOS standard [Martin et al., 2004; Langlois and Oehme, 2004]. An estimation of the overall composition of commercial PFOS is still not possible because of the different fragmentation patterns and their probably varying response factors [Langlois and Oehme, 2004].

As already mentioned earlier, extraction of POPs from waste matrices and separation from interfering substances extracted together with PFOS is considered to be the main prerequisite for PFOS analysis in waste matrices. Considering the different types of matrices covered under the term waste, it is not astonishing that various extraction techniques for different matrices exist.

The anionic perfluorinated alkyl compounds (notably PFOS and PFOA) can be extracted from environmental media by conventional methods using either acidification or ion pairing in order to obtain a neutral form of the analyte. Extraction with pressurized liquids (PLE) has been used for the analysis of perfluorinated alkyl substances including PFOS from sediment and sewage sludge. Solid-liquid extraction has been applied for dust, soil, sediment and sludge samples, using methanol or acetonitrile mixtures. Schröder compared three different extraction methods for PFAS including PFOS from STP sludge, i.e. Soxhlet extraction, hot vapour extraction and pressurised liquid extraction (PLE) with a range of solvents and solvent mixtures. PLE with sequential use of ethyl acetate (EtOAc): dimethylformamide (8:2) and MeOH: phosphoric acid (99:1) resulted in the most efficient analyte extraction [Schröder et al., 2003].

Published cleanup methods are relatively simple and straightforward and involve normal phase adsorption chromatography with e.g. silica, or C18 materials in a SPE set up or, alternatively, use of graphitised carbon. Clean-up has to be used if compounds are present that can interfere (co-elute) with PFOS. Depending on the different matrices, a variety of sample clean-up procedures are suitable. The complexity of waste matrices implies the use of multi-step purification methods. Waste samples may also contain high level of organic matter that disturbs the well operation of purification via column chromatographic procedure. For this reason, in most cases, it may be useful to start the purification step by a treatment with concentrated sulphuric acid to remove organic matter. Clean-up stage may then be performed with the aim of purification as well as fractionation of the extract, using a great variety of adsorbents (silica, Florisil, alumina ...). SPE (C18, Oasis HLB) clean-up has often been used prior to HPLC-MS/MS. Nevertheless, it is important to consider the effects of the clean-up procedures on recovery and sensitivity of the complete analytical method.

The analytical detection method of choice for PFAS is currently LC-MS or LC-MS/MS for the anionic compounds (including PFOS and PFOA), whereas both LC-MS(MS) and GC-MS can be used for the determination of the neutral per- and poly-fluorinated alkylated substances including several precursors of PFOS. Quadrupole tandem mass spectrometry applying negative electrospray ionisation interfaces (HPLC/ESI-MS/MS) seems to be the preferred instrumental method for the determination of ionic PFAS. In LC-MS of anionic PFAS, usually the dissociated acid (pseudo molecular) ion $[M-H]^-$ is observed, which can be used for quantitative purposes in LC-single quad MS, or as the precursor ion for multiple ion reaction monitoring in LC-MS/MS. Detection limits of LC-MS(MS) and GC-MS methods are sufficiently low to allow in principle for the determination of environmental levels of PFAS in drinking water and in food samples. Berger et al. compared three types of mass spectrometers for PFAS analysis, namely ion-trap MS(MS), ToF-HRMS and quadrupole MS(/MS). ToF-HRMS was found to be the superior method, combining high selectivity with optimal sensitivity. However, due to the low distribution of this type of instrument in analytical laboratories, quadrupole MS/MS is used most frequently [Berger et al., 2004].

The total fluorine content can be determined applying the non-destructive X-ray fluorescence, which has already been discussed in the previous chapter on PBDEs. XRF might also be useful as screening

method for fluorinated substances in waste or products [Gisey and Kannan, 2002]. The known limitations of the method are low sensitivity and lack of structure-specific information.

Literature methods for qualitative and quantitative analysis of PFOS

A lot of different analytical methods for determination of per fluorinated alkyl substances (PFAS) including PFOS in various matrices have been developed and reported in scientific literature in the last years. Most methods focus on environmental (water, soil, air, wildlife) and human matrices due to the concerns which arose with discovery of the ubiquitous occurrence of PFAS in the global environment. Jahnke and Berger [Jahnke and Berger, 2009] provided an extensive review on analytical methods applicable for various matrices. Based on their review and latest publications in scientific literature an overview of reported analytical methods, which are assumed to be applicable for waste matrices is provided in Table 7-7.

Environmental samples

Sinclair & Kannan published a method for waste water samples analysis reaching LOQs of 2.5 ng/L for both PFOA and PFOS [Sinclair & Kannan 2006]. Later, a large-volume-injection LC-ESI-MS/MS was successfully used for waste water samples analysis (LOQ = 0.5 ng/L for both analytes) [Schulz et al. 2006]. Determination of PFSSs in waste- and river water samples by mixed hemimicelle-based SPE before HPLC-ESI-MS/MS was presented [Zhao et al. 2007]. LOD was 0.2 ng/L for PFOS, and the method was found to be appropriate for PFSSs analysis. A study reporting an analytical method for PFOA and PFOS in solid matrices other than biological ones was published [Moriwaki et al. 2003]. Dust samples collected in Japanese homes were extracted with MeOH by ultrasonic agitation, and the extract was analysed by HPLC-ESI-MS/MS. LOD was 10 µg/kg for PFOS. A method, comprising liquid solvent extraction (aqueous acetic acid and MeOH), cleanup via SPE, and injection of the extracts with added internal standards into HPLC-ESI-MS/MS, enabling quantitative determination of PFASs, including PFOS, in sediment and sludge was reported [Higgins et al. 2005]. LODs of the method were analyte and matrix dependent coming to 0.1 (sediment) and 0.9 µg/kg (sludge) for PFOS.

Consumer products

Most analytical protocols for analysis of PFAS in consumer products are designed to estimate possible human exposure resulting from certain product groups treated with per- or polyfluorinated compounds. There have only been few reports of analytical methods for determination of PFOS in leather, textiles and paper. Most studies performed on polyfluorinated alkyl substances in consumer products or articles do not concern PFOS but perfluorinated alkyl substances present in articles due to material characteristics.

In a Danish study the PFAS contents of several consumer products were analysed, including floor polish waxes and impregnating agents for shoes and textiles. Waxes and liquids from aerosol cans were diluted with MeOH, dichloromethane or acetone, and analysed by LC MS/MS. Limits of detection for all analytes were less than 1 mg/L of product [Vejrup and Lindblom, 2002]. Larsen et al. compared pressurised solvent and reflux extraction methods employing five different solvents (AcN, chloroform, ethanol, MeOH, and water) for the determination of perfluorooctanoate (PFOA) in polytetrafluoroethylene (PTFE)

polymers using HPLC-ESI-MS/MS. Pressurised solvent extraction with ethanol, water, or MeOH turned out to be the best choice for this purpose (recoveries: 80 – 120 %)[Larsen et al., 2005]. Powley et al. extracted PFOA from the surface of commercial cookware heated up with water under simulated cooking conditions followed by a SPE step. Additionally, rectangular pieces of pans (1.5 x 7.5 cm) were extracted by ASE with a water/ethanol mixture. All extracts were analysed by HPLC-ESI-MS/MS [Powley et al., 2005]. Begley et al. extracted PFOA from materials being in direct contact with food such as PTFE-coated cookware or paper by either shaking with MeOH (50 °C, PTFE) or sonication with ethanol/water mixture (50:50 %)[Begley et al., 2005]. Extracts were subjected to HPLC-ESI-MS/MS analysis. A similar procedure was applied by Stadallius et al. to extract PFOA from paper and textile samples and a LOD of 1 µg/kg for paper and of 2 µg/kg for textile samples was achieved [Stadallius et al., 2006]. A method for the analysis of extractable PFOA was developed to evaluate leaching of PFOA from treated textiles and carpet [Mawn et al., 2005]. The method compared extraction efficiencies of water, sweat simulant and saliva simulants with that of MeOH using LCMS/MS. Limits of detection of between 1 and 3 µg/kg of sample were reported.

Although PFOS has not been addressed in these studies, the used extraction procedures and HPLC-ESI-MS/MS procedures should be suitable for determination of PFOS in articles. HPLC-ESI-MS/MS seems to be the method of choice to detect and determine PFOS contents in solvent extracts of articles.

Standard methods for qualitative and quantitative analysis of PFOS

Currently there is no standardised analytical method for determination of PFOS-related substances in waste matrices, preparations or articles available. However, in 2006 the Commission mandated CEN to develop such a method. The technical specification of the standard method “Determination of extractable perfluorooctanesulfonate (PFOS) in coated and impregnated solid articles, liquids and fire fighting foams - Method for sampling, extraction and analysis by LC-MS/MS or LC-MS” (FprCEN/TS 15968) has been already been prepared by CEN/TC 382. It will describe a method applicable for analyses and it can already be applied today. The analytical methods in the process of standardisation by CEN were Liquid Chromatography/Mass Spectroscopy (LC/MS) for anionic PFOS species, and Gas Chromatography/ Mass Spectroscopy (GC/MS) for non-ionic PFOS species. As both methods require liquid samples, for semi-finished products and articles, solvent extraction of PFOS will be required. The extracted PFOS species are expected to be mainly anionic, but non-ionic species may also be present in the liquid samples. In practice, samples may contain several PFOS species and they must all be taken into account in calculating the total PFOS concentration. However, the molecular weights of the various PFOS species in a sample can cover a wide range of values. Aggregation of the amounts of the different species is therefore best achieved by first calculating the corresponding amounts of a reference species, i.e. PFOS acid $C_8F_{17}SO_3H$. CEN has confirmed that the analytical method described in the technical specification can be used for the proposed lower threshold of 0.001%. However, before the method is an official standard it must still be tested in an inter-laboratory comparison which is expected to take some years.

Table 7-7: Summary of the results of a survey on standardised analytical methods and methods reported in literature to be considered for PFOS analysis.

Method	Matrix	Extraction	Clean-up	Detection	Substance	LOD and LOQ	Reference
Standard methods for determination of polyfluorinated substances in water							
E DIN 38407-42:2010-05 (D)	Water	SPE	No details available	LC-MS/MS	Polyfluorinated substances	No details available	[DIN 2010]
EPA 537	Drinking water	SPE	-	LC-MS/MS	perfluorinated alkyl acids (incl. PFOS)	1.4 µg/L (DL) 6.5 µg/L (lowest concentration minimum reporting level)	[EPA 2009]
ISO 25101:2009	Water	SPE	-	LC-MS/MS	PFOA and PFOS	2,0 ng/l to 10 000 ng/l	[ISO 25101]
Standard method for determination of PFOS under development							
FprCEN/TS 15968	Coated and impregnated solid articles, liquids, fire fighting foams	Depending from matrix – no details available	Depending from matrix – no details available	LC-MS/MS; LC-MS	PFOS	0,5 ng/ml to 50 ng/ml in extract	[CEN 15968]
Methods for determination of PFOS its salts and PFOA in different matrices reported in scientific literature							
Literature	Water	On-line extraction (turbulent flow chromatography)	-Column wash	LC-APPI/MS	PFOS	18 ng/L (LOQ)	[Takino et al., 2003]
Literature	Waste water STP sludge	Sedimentation, decantation Solvent liquid extraction after drying and grinding (aqueous acetic acid and MeOH)	SPE (OASIS HLB)	LC-APPI/MS	PFOS	2.5 ng/L (LOQ) 10 – 25 ng/g (LOQ)	[Sinclair and Kannan., 2006]
Literature	Water, waste water	Filtration	SPE mixed hemimicelle-based	HPLC-ESI-MS/MS	PFA's incl. PFOS	0.2 ng/L (LOD)	[Zhao et al., 2007]
Literature	Sediment, sludge	Solvent liquid extraction (aqueous acetic acid and MeOH)	SPE	HPLC-ESI-MS/MS	PFA's incl. PFOS	0.1 µg/kg (LOD sediment) and 0.9 µg/kg (LOD sludge)	[Higgins et al., 2005]
Literature	Sewage sludge	Freeze drying, Soxhlet extraction, hot vapour extr., PLE	-	HPLC-ESI-MS/MS GC—NCI-MS	PFSAs incl. PFOS	6.000 – 10.000 ng/g (LOD) 10.000 – 20.000 ng/g (LOQ)	[Schröder 2003]
Literature	Dust	Solvent liquid extraction (MeOH) sonication	Centrifugation, filtration	HPLC-ESI-MS/MS	PFSAs incl. PFOS	10 – 50 ng/g (LOQ)	[Moriwaki et al.,2003]
Literature	Dust	Solvent liquid extraction (AcN) sonication	Centrifugation, SPE (C18)	HPLC-ESI-MS/MS	PFSAs incl. PFOS	0.99 – 4.56 ng/g (MDL)	[Kubwabo et al.,2005]

Method	Matrix	Extraction	Clean-up	Detection	Substance	LOD and LOQ	Reference
Literature	Fabrics and leathers	Sonication with 0.1 M HCl and MeOH	Zorbax SB-C18 column	LC-MS/MS	PFOS	1.5 mg/kg (LOD)	[Huang et al.,2007]
Literature	Packaging materials and textiles	Pressurized liquid extraction	-	GC-MS after silylation	PFOA and PFOS	13.9 ng/mL (LOD)	[Wang et al.,2009]
Literature	Paper products	Methanol by the accelerated solvent extractor	Purification, film-filtration	LC-MS/MS	PFOS	0.10 mg/kg (LOD)	[Ma et al.,2009]
Literature	Textile/carpet	SLE (water, MeOH) shaking PLE (MeOH)	Protein precipitation/centrifugation	LC-ESI-MS/MS	PFCAS	1-3 ng/g (LOQ)	[Mawn et al., 2005]
Literature	Paper/textile	SLE (MeOH)	Centrifugation	LC-ESI-MS/MS	PFCAS	1-2 ng/g (LOD)	[Stadillus et al., 2006]

7.4.2.2 Conclusion

Based on the results of the world-wide investigation of measurement methods and methods reported in literature (Table 7-7) it can be stated that LC-MS/MS techniques are state-of-the-art in the field of confirmatory analysis for PFOS. Performance criteria of standard and literature methods are quite comparable and only selected extraction and clean-up procedures are used for analysing a broad spectrum of matrices. Although waste matrices are not extensively listed in the scope of the currently available methods, the listed methods are applied successfully to different waste matrices in routine practice. Therefore, standard methods can be developed on the basis of the reported analytical methods. From this point of view, the available techniques are sufficient to detect PFOS and its salts in waste matrices at quite low levels. Furthermore, activities are on the way to harmonise these methods in view of analysing certain type of wastes with one standard method. Nevertheless extraction and clean-up procedures have to be adapted to the relevant types of waste (matrix).

Interlaboratory comparisons have revealed that until now large between laboratory variabilities can be observed in the analysis of water or food samples. Hence, much work remains to be done before the analysis of this group of analytes will be fully understood and controlled.

Against this background it seems to be very challenging to develop one standard method for PFOS analysis in waste. In view of the variety of waste matrices it is suggested to provide a list of possible methods (e.g. clean-up methods) instead of strictly focusing on a single reference standard for certain types of waste. Although recommendation should be given to the analyst, which methods are adequate for which kind of matrix and which quality criteria and standards have to be achieved, the analyst should be free to take the final decision depending on his individual target.

7.4.3 PeCB

PeCB is not being intentionally produced or used anymore within the EU. The most relevant source of PeCB production and emission is unintentional production and connected releases. As main components incineration and combustion processes of different wastes and materials have been identified, i.e. municipal solid waste incineration, hazardous waste incineration, coal combustion and domestic burning of fuels and mixed waste. Further, also waste water treatment has been considered as relevant.

7.4.3.1 Existing Methods

Table 7-8 summarises the results of a survey on standardised analytical methods to be considered for PeCB analysis. It lists standardised multiple component methods which include PeCB analysis. Only few standard analytical methods dealing with PeCB analysis in environmental samples and no standard methods especially developed for PeCB detection in waste have been identified. However, standard methods suitable to determine the amount of “aromatic hydrocarbons” or “semivolatile organic compounds” in environmental samples are available. These methods may be applicable to other compounds not explicitly covered (e.g. PeCB) or to other types of matrices. However, it is necessary to verify the applicability of these methods for these special cases. A definitive judgement is not possible based on the information available. Therefore, additionally methods for quantitative detection of PeCB

in different matrices (e.g. fly ash, sewage sludge, etc) reported in literature have been evaluated. Some of the listed literature methods require sampling, chemical preparation (e.g. extraction by different techniques) and a quite well-equipped laboratory.

The philosophy of the recent published methods for the determination of chlororganic compounds in environmental samples is concentrated mainly on the development of universal, rapid and high capacity methods capable of analysing as many different pollutants in as many matrices as possible (multiresidue–multimatrix techniques). In addition to this, advanced highly reliable techniques for confirming positive results found with the screening methods are required.

Summary of analytical strategies for quantitative analysis of PeCB

Depending on the type of matrix, different extraction techniques are used to separate chlorinated benzenes or pentachlorobenzene from samples. The frequently used isolation technique for recovering chlorinated benzenes from aquatic environmental samples is a simple liquid–liquid extraction. Pentane, hexane, and a 1:1 mixture of diethyl ether or petroleum ether and cyclohexane have been found to be the most effective extraction solvents for chlorinated benzenes [Veningerova et al., 1997]. The extraction of chlorobenzenes from aquatic sediments or soil can be achieved by Soxhlet extraction. Sewage sludges were Soxhlet extracted for 6 h with a mixture of hexane–acetone (2:1) in a full glass system [Wang et al. 1992]. The recoveries of chlorinated benzenes were 80–98%. Thermal extraction can be used to analyse fly ash and other solid matrices. This technique provides a rapid quantitative and simultaneous determination of a wide range of volatile and semi-volatile organic compounds without laborious sample preparation or any chemical pre-treatment. Its amenability to automation and coupling with on-line detection methods makes thermal extraction a promising technique, not only in laboratory analysis, but also for in situ emission monitoring. The PeCB amounts determined with thermal extraction under optimized conditions corresponded well with the results of conventional solvent extraction [Tsytsik et al. 2008]. Solid-phase microextraction (SPME) was examined as a possible alternative to Soxhlet extraction in the analysis of chlorobenzenes in soils by Santos et al. [Santos et al., 1997]. For headspace SPME good sensitivity was obtained by adding 200 ml of water to the soil, at a sampling temperature of 30°C and absorption times of 40 and 60 min, respectively. SPME can integrate sampling, extraction, concentration and sample introduction in a single step and is a fast, inexpensive and easily automated technique not requiring organic solvents. SPME uses coated fused-silica fibres to extract analytes from gaseous and liquid phases. After equilibrium is reached or after a well-defined extraction time, the absorbed compounds are thermally desorbed by exposing the fibre in the injection port of a gas chromatograph, or redissolved in an organic solvent if coupled to HPLC.

Gas chromatographic techniques coupled with mass spectrometry or electron capture detection are usually used to analyse samples for their PeCB content. DB-5 MS (5% phenyl-, 95% methylpolysiloxane) fused-silica capillary columns have been used for separation of analytes by most authors. Alternatively, HP-1 (methyl silicone) fused-silica capillary columns, CPsil8MS (5% phenyl, 95% polydimethylsiloxane) capillary column or DB-17 (50% phenyl-, 50% methylpolysiloxane) columns have been used.

Table 7-8: Summary of the results of a survey on standardised analytical methods to be considered for PeCB analysis.

Method	Matrix	Extraction	Clean-up	Detection	LOD and LOQ	Reference
Standard methods to determine PeCB in environmental samples and water						
DIN 38407-F9/F3	Water Waste water, Sludge	Solid phase extraction	-	GC-ECD GC-MS	Depending on matrix and detector. LOD: 0.025 µg/L	[Böhm et al., 2002]
DIN 38414-S20	Solid matrices (sludge, sediment)	Soxhlet extraction	-	GC-MS	LOD 2µg/kg	[DIN 38414]
ISO 6468:1996	Drinking water, Ground water, Surface water and Waste water	Liquid-liquid extraction	Matrix dependent	GC-MS/MDI	Applicable to samples containing up to 0,05 g/l of suspended solids. LOD depending on matrix and detector.	[ISO 6468]
US EPA 8270D	Air, Water, Soil/sediment, Waste matrices	US EPA 3524 US EPA 3510 US EPA 3540, 3541, 3545, 3546, 3550, 3561 3540, 3541, 3545, 3546, 3550, 3560, 3580	US EPA 3600	GC-MS	LOQ: 10 µg/L for ground water LOQ: 660 µg/kg (wet weight) for soil/sediment LOQ: 1-200 mg/kg for wastes (dependent on matrix and method of preparation)	[NEMI 8270]
US EPA 1625	Waste water	Liquid-liquid extraction	Depending on matrix	GC-MS	n.a.	[NEMI 1625]
US EPA 625	Waste water	Liquid-liquid extraction	Depending on matrix	GC-MS	n.a.	[NEMI 625]
Methods to determine PeCB in waste relevant matrices reported in scientific literature						
Literature	Water, Soil and Vegetables	Solid phase extraction Soxhlet extraction	- Florisol column	GC-ECD	LOQ 0.001 µg/l LOQ 0.001 mg/kg	[Veningerova et al., 1997]
Literature	Soil	Solid-phase microextraction (SPME)	-	GC-IT-MS	Applicable to samples containing chlorobenzenes up to 65 µg/g. LOD: 30 pg/g (headspace analysis) LOD: 100 pg/g (direct extraction)	[Sarrionet et al., 1998] [Santos et al., 1997]
Literature	Fly ash	Thermal extraction	-	GC-MS	Not applicable Linearity of 0.9860 and mean recovery of 107% in the range of 2–32 ng on GC column	[Tsytisik et al., 2008]
Literature	Sewage sludge	Soxhlet extraction	Florisol column	GC-ECD	LOD: 0.07 µg/kg, Recovery: 80-98%	[Wang et al., 1992]
Literature	Fly ash and waste ash	Solvent extraction with toluene	Silica column	GC-ECD	Lowest reported content 0,004 ng/g	[Schreiner et al., 1986]
Literature	Fly ash	Solvent and water extraction	Filtration	LVI-GC-MS	Lowest reported content 13 ng/g	[Korenkova et al., 2006]

Standard methods for quantitative analysis of PeCB

Regarding determination of PeCB contents in waste matrices, only one standard analytical method has been identified. Nevertheless, different standard methods dealing with analysis of chlorinated benzenes in different matrices comparable to waste are available. These methods can be adapted for the respective waste categories. A standard method for determination of PeCB in soil or sludge might also be suitable for determination of PeCB content in sewage sludge or bio waste for example.

US EPA METHOD 8270D is used to determine the concentration of semivolatile organic compounds in extracts prepared from many types of solid waste matrices, soils, air sampling media and water samples. The method has been proven to be suitable to detect PeCB and further presents an overview of extraction methods applicable for different analyte-matrix combinations. This method can be used to quantitate most neutral, acidic, and basic organic compounds that are soluble in methylene chloride (or other suitable solvents provided that the desired performance data can be generated) and are capable of being eluted, without derivatization, as sharp peaks from a gas chromatographic fused-silica capillary column coated with a slightly polar silicone. The samples are prepared for analysis by gas chromatography/mass spectrometry (GC/MS) using the appropriate sample preparation and, if necessary, sample cleanup procedures. Identification of target analytes is accomplished by comparing their mass spectra with the electron impact (or electron impact-like) spectra of authentic standards. Quantitation is accomplished by comparing the response of a major (quantitation) ion relative to an internal standard using an appropriate calibration curve for the intended application. The lower limits of quantitation for this method when determining an individual compound are approximately 660 µg/kg (wet weight) for soil/sediment samples, 1-200 mg/kg for wastes (dependent on matrix and method of preparation), and 10 µg/L for ground water samples.

US EPA METHOD 1625 is designed to determine the semivolatile toxic organic pollutants amenable to extraction and analysis by capillary column gas chromatography-mass spectrometry (GC/MS). The covered chemical compounds may be determined in municipal and industrial discharges by this method. Stable isotopically labelled analogs of the compounds of interest are added to a one litre wastewater sample. The sample is extracted with methylene chloride using continuous extraction techniques. The extract is dried over sodium sulfate. An internal standard is added to the extract, and the extract is injected into the gas chromatograph (GC). The compounds are separated by GC and detected by a mass spectrometer (MS). The labelled compounds serve to correct the variability of the analytical technique. The detection limit of this method is usually dependent on the level of interferences rather than instrumental limitations.

DIN 38407 describes German standard methods for the examination of water, waste water and sludge. Subsection F9 relates to the determination of benzene and some of its derivatives by gas chromatography. Subsection 2 covers determination of low volatile halogenated hydrocarbons by gas chromatography. Methods reported in these documents have been used to determine PeCB contents in environmental samples. Methods according to DIN 38414-20 have also been adjusted to allow detection of PeCB in solid matrices like sludge and sediment by GC-MS. Limits of these methods strongly depend on the kind of matrix and the used detectors. Nevertheless, LODs of 2 µg/kg for solid matrices and 0.025 µg/L for water samples have been reported.

ISO 6468:1996 describes a method for determining certain organochlorine insecticides, polychlorinated biphenyls (PCBs) and chlorobenzenes (except the mono- and dichlorobenzenes) in drinking water, ground water, surface waters and waste waters. The method is applicable to samples containing up to 0,05 g/l of suspended solids.

7.4.3.2 Conclusion

Based on the results of the world-wide investigation of standardised measurement methods (see Table 7-8) it can be stated that GC/MS and GC-ECD techniques are state-of-the-art in the field of standardised confirmatory analysis for PeCB. Performance criteria are quite comparable and different extraction and clean-up procedures are available and sufficient for analysing a broad spectrum of matrices. Although waste matrices are not extensively listed in the scope of the currently available methods, the listed methods are applied successfully to different matrices occurring in waste in routine practice.

Standard methods are available for some matrices like waste water or soil and sludge. Standard methods for other waste matrices can be developed on the basis of the reported analytical methods. Nevertheless extraction and clean-up procedures have to be adapted to the relevant types of waste (matrix). From this point of view, the available techniques are sufficient to detect PeCB in waste matrices at quite very low levels.

Due to the inhomogeneity of the matrix waste, it seems to be very challenging to develop one standard method for PeCB analysis or analysis of chlorinated benzenes in all types of waste. In view of the variety of waste matrices it is suggested to provide a list of possible methods (e.g. clean-up methods) instead of strictly focusing on a single reference standard for certain types of waste. Although recommendation should be given to the analyst, which methods are adequate for which kind of matrix and which quality criteria and standards have to be achieved, the analyst should be free to take the final decision depending on his individual target. This approach has partly been followed by the US EPA methods, which provide suitable extraction methods for analyte-matrix combinations.

7.4.4 PCP

Pentachlorophenol (PCP) is investigated together with the substances sodium pentachlorophenoxide (NaPCP) and Pentachlorophenyl laureate (PCPL). In the past the PCP was used for many applications however, due to long product life times waste containing PCP can still enter the waste stream. The relevant sources of waste regarding PCP are nowadays only from the wood impregnation industry and the textile industry.

7.4.4.1 Existing Methods

Various analytical methods are available for detecting, and/or measuring, and/or monitoring pentachlorophenol and its metabolites. The intent is not to provide an exhaustive list of analytical methods. Rather, the intention is to identify well-established methods that are used as the standard methods of analysis. Many of the analytical methods used for environmental samples are the methods approved by federal agencies and organizations such as EPA and the National Institute for Occupational Safety and Health (NIOSH). Other methods presented in this chapter are those that are approved by

standardization bodies like CEN. Additionally, analytical methods are included that modify previously used methods to obtain lower detection limits, and/or to improve accuracy and precision. Evaluation of analytical methods has been limited to approved standard methods suitable to determine the mass content of pentachlorophenol in relevant waste matrices like wood or textiles.

Summary of analytical strategies for quantitative analysis of PCP

Regarding biological samples exposure to pentachlorophenol is most commonly evaluated by analysis of urine, blood, feces, or adipose or other tissues, using gas chromatography (GC) combined with electron capture detection (ECD) or high-performance liquid chromatography (HPLC) combined with ultraviolet (UV) detection. Recovery is generally high and sensitivity using GC/ECD and HPLC/UV is in the parts per billion (ppb) range.

Many purification schemes take advantage of the fact that pentachlorophenol is a weak organic acid. These methods involve extracting the compound into the organic phase under acidic conditions, and/or extracting into alkaline solution as phenolate salts [Chou and Bailey 1986; EPA 1986b]. Thus, the standard methods involve multiple extractions, with potential for sample loss. Some of these methods derivatize pentachlorophenol prior to analysis [EPA 1980b; NIOSH 1984b]. Derivatization often involves diazomethane or diazoethane, which are toxic substances [Bevenue et al. 1968; Holler et al. 1989; Morgade et al. 1980; Shafik 1973; Wagner et al. 1991]. Recent methods have tried to simplify the purification scheme and avoid using toxic chemicals for derivatization [Maris et al. 1988]. In an effort to use less toxic materials, blood and urine samples were derivatized with acetic anhydride [Needham et al. 1981]. The detection limit was 1–2 ppb using GC/ECD. Penta- and tetrachlorophenols were analyzed simultaneously in urine using HPLC [Pekari and Aito 1982]. A rapid extraction method followed by GC/ECD had a detection limit of 0.5 ppb [Kalman 1984].

Concerns about contamination of environmental media, plants, and animals with pentachlorophenol have led to the need for more rapid, sensitive, and selective methods of analysis. As with biological samples, the most common methods of analysis are GC/ECD, high resolution gas chromatography (HRGC)/ECD, and HPLC/UV detection. Methods are available that detect pentachlorophenol in water or sediment at the 1–10-ppb range. Pentachlorophenol could be detected in marine water at concentrations ranging from 0.2 to 200 ppb in volumes as small as 5 mL using a simplified monitoring procedure with HPLC/UV detection [Giam et al. 1980]. This method reduces costs and analysis time, and can also be used in other aquatic toxicity studies. Automated HPLC is 10 times faster than wet chemical techniques. Once the method for analysis has been established and tested thoroughly, the HPLC method requires neither extensive pretreatment nor highly trained laboratory personnel [Ervin and McGinnis 1980]. For relatively clean water samples, HPLC offers a rapid and sensitive method, but its advantages are lost when a complex matrix such as municipal waste water has to be analyzed [Buisson et al. 1984]. The resolution possible with capillary gas chromatography and the selectivity of the ECD towards halogenated compounds make HRGC/ECD the method of choice for the detection and quantification of chlorinated phenols at trace levels in complex matrices. Derivatization with a halogen-containing reagent enhances the ECD response. For measuring pentachlorophenol in waste water using HRGC/ECD, sensitivity is in the ppt range. Recoveries are adequate and precision is good.

Similar results were obtained in a comparison of HPLC and GC techniques for determination of pentachlorophenol in animal materials [Mundy and Machin 1981]. Pentachlorophenol could be separated from acidic pesticides and other organic acids possibly present in a mill effluent by extraction with an acetylating agent [Rudling 1970]. A similar single step extraction and acetylation procedure was used to determine several chlorinated phenolic compounds in paper mill effluent without interference [Lee et al. 1989]. GC/MS has been used to measure pentachlorophenol in honey [Muiño and Lozano 1991]. This method is simple, accurate, and rapid. Sensitivity is in the low-ppb range. Good recoveries (84–102%) and precision (2.8–6.3% relative standard deviation ([RSD])) were obtained. A study comparing several methods for rapidly extracting pentachlorophenol from water or soil reported high recovery from all methods using HPLC/UV [Wall and Stratton 1991]. A method combining extraction with derivatization by acetic anhydride had a detection limit of about 0.1 ppb for GC/ECD [Xie 1983].

Regarding waste matrices, a solid-phase microextraction (SPME) method has been developed for the determination of pentachlorophenol (PCP) in paper and board samples. The analytical procedure involves direct extraction of PCP from paper and board samples and determination by gas chromatography with electron capture detection (GC–ECD). The optimized procedure was applied to the analysis of pentachlorophenol (PCP) in five samples of virgin and recycled paper and board. To evaluate the effectiveness of the proposed method, it was compared with conventional extraction method with liquid–liquid extraction and derivatization. Detection limit of 0.015 mg/kg for PCP in paper was achieved with a RSD of 14% [Domeno 2005].

Becker et al. compared two independently developed and validated procedures for the determination of pentachlorophenol (PCP) in waste wood by means of a collaborative trial. Both methods had foreseen quantification of PCP by gas chromatography (GC–ECD) after acetylation and differed with regard to the use of methanol or toluene/sulphuric acid, respectively, as solvent in the sonication extraction step. Test samples with established analyte homogeneity were prepared from a ground “real life” starting material. A total of 23 participating laboratories with experience in wood preservative analysis were instructed to apply both methods to three levels of content in the range of 0.5–20 mg PCP/kg. In case of the toluene/sulphuric acid extraction, lower recoveries and higher interlaboratory dispersion of results at the higher PCP contents were observed. Thus, the sonication extraction procedure with methanol has been annexed as a reference method to the German waste wood regulation [Becker et al. 2006].

Mou et al (1999) developed a method for determination of pentachlorophenol residues in textiles and leather and leather products by gas chromatography/mass spectrometry. The pentachlorophenol residues in samples were acidified with sulphuric acid and extracted with n-hexane. Determination was performed by means of gas chromatograph equipped with MSD, using external standard calibration. The detection limit was 0.02 mg/kg. The average recovery ranged from 86.7% to 93.1%. The method has been used to determine the mass concentration of pentachlorophenol residues in raw cotton, towel, shirt, leather gloves and leather watchguard.

Standard methods for quantitative analysis of PCP in relevant waste matrices

CEN ISO 15320:2004 specifies a test method for the determination of pentachlorophenol (PCP) in an aqueous extract in pulp, paper and board. Though it is developed for paper and board intended to come

into contact with foodstuffs, it is applicable to all kinds of pulp, paper and board. Determination of the mass concentration of PCP in the extract is performed by means of GC-ECD or GC-MS. The working range of the method is 0.05 to 0.5 mg/kg.

The German Federal Institute for Materials Research and Testing (BAM) has elaborated a reference procedure for the Determination of Pentachlorophenol (PCP) in Wood. The method is based on the extraction of the crashed wood and wood materials (particle size 1mm) with methanol, using the "accelerated solvent extraction" ASE-apparatus. PCP in the extract is derivatised with acetic anhydride to pentachlorophenol acetate. Pentachlorophenol is analysed and quantified by means of GC-MS or GC-ECD. The method works in a concentration range of 0.1 µg/g - 100 µg/g. CEN provides standard

CEN/TR 14823:2003 which specifies a laboratory method of determining the pentachlorophenol content of wood. The method is applicable to all types of PCP-treated wood and wood-based materials as well as for the analysis of waste timber with respect to its PCP content. The method has a quantification limit corresponding to 100 µg PCP per kilogram of wood material expressed as dry matter. The method described has a measurement range up to PCP contents of 25 mg/kg of dry matter. These figures refer to the given example (where an aliquot of 1 ml of the extract is used for acetylation. If lower quantification limits are required, a higher volume of extract aliquot can be used for derivatisation.

A further standard EN ISO 17070:2006 specifies a method for determining the content of pentachlorophenol (PCP), its salts and esters in leather. After extraction into n-hexane PCP is acetylated by acetic anhydride and PCP acetates are analysed by gas-chromatography with an electron capture detector (ECD) or mass selective detector (MSD). Quantification is performed by an external standard and correction made with an internal standard.

7.4.4.2 Conclusion

Based on the results of a literature search on standardised measurement methods and methods reported in scientific articles it can be stated that GC/MS and GC-ECD techniques are state-of-the-art in the field of standardised confirmatory analysis for PCP. Performance criteria are quite comparable and different extraction and clean-up procedures are available and sufficient for analysing a broad spectrum of matrices. Extraction and clean-up procedures have been adapted to the relevant types of waste (matrix). From this point of view, the available techniques are sufficient to allow analytical control of limit values.

Due to the inhomogeneity of the matrix waste, it seems to be very challenging to develop one standard method for PCP analysis in all types of waste. In view of the limited number of relevant waste matrices it is suggested to use the available internationally accepted standard methods for analysis of the certain types of waste instead of trying to develop a single standard method covering all kinds of waste. Standardised methods are available for most of the waste matrices which have been identified to be relevant.

7.4.5 HCB

At present there is no intentional use of HCB in Europe since many years. HCB is mainly formed as

unintentional by-product during several industrial processes, but the available information basis is very scarce. However, production of tri- and tetrachloroethene and tetrachloromethane has been indicated as the most important source. Production of primary magnesium has been identified to belong to the historic uses by now, since its production within EU27 has stopped recently. Unintentional production and releases from the plastic industry may be relevant, but specific data is not available. The same case has been identified for the sector of hazardous waste incineration and sewage sludge resulting from treatment of industrial waste water.

7.4.5.1 Existing Methods

The purpose of this chapter is to describe the analytical methods that are available for detecting, and/or measuring, and/or monitoring hexachlorobutadiene (HCBD). The intent is to identify well established methods that are used as the standard methods of analysis. Standard methods to detect HCBD in relevant waste matrices like sludges or ashes certified by international standardisation bodies like ISO or CEN are not available. Therefore, analytical methods are included that modify previously used methods to obtain lower detection limits, and/or to improve accuracy and precision.

Summary of analytical strategies for quantitative analysis of HCBD [ASTDR, 1994]

Gas chromatography (GC) with an electron-capture detector (ECD) and/or GC with detection by mass spectrometry (MS) have been used to measure hexachlorobutadiene concentrations in human blood and adipose tissue [Bristol et al. 1982; LeBel and Williams 1986; Mes et al. 1985] and in rat liver tissue [Wang et al. 1991]. In gas chromatography, samples dissolved in a volatile solvent are injected into a heated column with a stationary phase consisting of silica coated with a liquid phase. An inert gas carries the sample through the column, and the partitioning of hexachlorobutadiene between the mobile and stationary phases gives it a characteristic retention time which is used to identify it. Electron-capture detectors use a radioactive source such as ^{63}Ni to generate electrons that are captured by the chlorine atoms in hexachlorobutadiene. Reduction in electron flow by this capture produces a characteristic signal for hexachlorobutadiene. Identity of hexachlorobutadiene is confirmed by detection by mass spectroscopy, which provides specific identification by a characteristic ion fragmentation pattern.

Biological samples are prepared for analysis by extraction with organic solvents. This extract from blood may be used directly [Bristol et al. 1982; Kastl and Hermann 1983], but extracts from adipose or liver tissue are cleaned up by gel permeation chromatography (GPC), which separates hexachlorobutadiene from higher molecular weight lipids, and/or by passage through a Florisil column which retains lipids and other contaminants [LeBel and Williams 1986; Mes et al. 1985]. These methods provide 42- 122 % recovery and can detect < 1 µg/L hexachlorobutadiene in blood and 1 µg/kg hexachlorobutadiene in fat [Bristol et al. 1982; LeBel and Williams 1986; Mes et al. 1985; Wang et al. 1991]. No information was located on methods for detection of hexachlorobutadiene metabolites or other biomarkers of hexachlorobutadiene exposure or effect.

Hexachlorobutadiene in environmental samples is also measured using GC coupled with ECD, MS, a halogen electrolytic conductivity detector (HECD), or a photoionization detector (PID) [APHA 1992a, 1992b; EPA 1982a, 1982c, 1986, 1989c, 1989d, 1990b, 1990d, 1990e]. Several methods have been used

for extraction of hexachlorobutadiene from environmental samples. Standard methods for analysis of air involve pumping the air through a material that will adsorb hexachlorobutadiene or through a cold trap to condense the hexachlorobutadiene [EPA 1990b; NIOSH 1990]. Purge-and-trap methods are used to extract hexachlorobutadiene from water, soil, or solid waste [APHA 1992b; EPA 1989c, 1989d, 1989e, 1990e]. Purge-and-trap methods involve bubbling an inert gas through the sample, trapping the hexachlorobutadiene in a tube containing a sorbent material, and then heating the sorbent tube and flushing the hexachlorobutadiene into a GC. Soil, sediment, and waste samples are mixed with water prior to purging [EPA 1990e]. An alternative way to prepare water, soil, or solid waste samples for GC analysis is to extract with methylene chloride or some other organic solvent; for waste water, soil, and solid waste samples, the organic extracts are cleaned up by gel permeation chromatography (GPC) or Florisil adsorption chromatography (FAC) [APHA 1992a; EPA 1982a, 1982c, 1986]. Purge-and-trap methods generally provide > 90% recovery, while organic extraction may have lower and more variable recovery rates [APHA 1992a, 1992b; EPA 1982a, 1982c, 1989c, 1990e]. Gas chromatographic methods with ECD and other detectors have a detection limit for hexachlorobutadiene of 0.02-0.05 µg/L in water [EPA 1982a, 1989c, 1989d, 1989e]. Detection limits for soil and solid waste are usually higher, depending on matrix interferences, extraction, and clean up procedures. LODs between 0.66 and 50 mg/kg wet weight (ppm) [EPA 1986] and between 0.05 and 2.5 mg/kg wet weight [EPA 1990e] have been reported for soil and solid waste. Detection by MS is most specific because identification is based on the characteristic mass ion as well as the retention time. Newer MS methods can achieve detection limits of 0.04-0.11 µg/L in water, comparable to ECD [EPA 1982a, 1989e]. A report regarding organic contaminants in soil published by the Federal Environment Agency of Austria reported a method using soxhlet extraction with Hexane followed by clean up with column chromatography and measurement of HCBd content by means of GC-MS achieving a LOD of 170 ng/kg dry weight and a LOQ of 340 ng/kg dry weight [UBA, 2008].

Standard methods for quantitative analysis of HCBd in relevant waste matrices

Standard methods for quantitative analysis of HCBd in relevant waste matrices like sludges or ashes have not been identified. Analytical methods reported in literature will therefore be assessed to identify a suitable LOQ to identify HCBd in waste matrices. As already shown above, there are several EPA methods available in the National Environmental Methods Index⁴³ covering HCBd analysis in water and soil samples.

7.4.5.2 Conclusion

Based on the results of the world-wide literature search for standardised measurement methods and methods reported in literature, it can be stated that GC/MS and GC-ECD techniques are state-of-the-art in the field of confirmatory analysis for HCBd. Performance criteria of the reported methods are quite comparable and different extraction and clean-up procedures are available and sufficient for analysing a broad spectrum of matrices. Extraction and clean-up procedures have to be adapted to the relevant type of waste (matrix). Against this background, achievable LODs for waste matrices might be higher than for matrices like water or biological samples, which have to be taken into account when defining a LOD for sludge or ash. However, the relevant waste matrices regarding HCBd analysis seem to have

⁴³ <https://www.nemi.gov/apex/f?p=237:1:4377134365110964>

comparable properties as environmental samples like sediment or soil. From this point of view, the available techniques are sufficient to allow analytical control of limit values.

The listed literature methods are designed to detect HCBd at quite low levels in the corresponding matrices, however, it has to be emphasised that the achievable levels can vary and strongly depend on the individual matrix to be analysed, especially if waste matrices are analysed for their HCBd content. Although standard methods have not been identified, there are analytical methods available to determine the mass concentration of HCBd in environmental samples which define critical performance criteria like LOD or recovery.

7.4.6 PCN

PCN can be formed unintentionally during various thermal processes. As PCNs exhibit similar formation properties as PCDD/Fs, unintentional production during incineration processes such as municipal solid waste incineration, hospital waste incineration, domestic burning, different metal processing steps such as secondary copper production, secondary aluminium production, magnesium production as well as iron sintering and electrical arc furnace processes for iron production are of relevance. Further, the accumulation in sewage sludge from diffuse sources has been taken into account.

7.4.6.1 Existing Methods

Theoretically, there are 75 possible planar congeners with 1-8 chlorine atoms substituted to the naphthalene skeleton. In technical PCN products, up to 67 congeners were reported. The analysis of organohalogen substances (OHS) in biological samples includes extraction of the OHS and the lipids from the matrix, removal of the lipids, separation of various classes of OHS and finally, detection and quantification. PCNs have physical and chemical properties similar to the PCBs and the methods used to analyze PCBs and PCNs are therefore rather similar. After extraction, sulfuric acid may be used to degrade the lipids in the sample. The PCNs should thereafter be separated from other OHS such as PCBs, which are generally present at much higher levels, otherwise these will interfere with the PCNs in most chromatographic systems. More recently, liquid chromatographic methods have been developed to isolate PCNs from other OHS.

Summary of analytical strategies for quantitative analysis of PCN [Jacobsson and Asplund, 2000]

Traditional extraction methods which extract OHS from biological matrixes with high reproducibility can also be used for PCN analysis [Jensen et al., 1983, Blingh et al., 1959]. Since PCNs are planar compounds and may adsorb rather strongly to carbon particles, methods that are especially effective at extracting planar OHS from the matrix are recommended, e.g., soxhlet extraction with toluene, for samples with a high carbon content [Järnberg et al., 1997].

Gel permeation chromatography (GPC) on styrene-divinylbenzene copolymer columns has been found to separate PCN from e.g., PCB. The method was first applied to the isolation of PCNs in commercial PCB products [Haglund et al., 1993]. Charcoal has frequently been used to separate planar and non-planar substances and also to separate PCNs and PCBs [Stalling et al., 1979; Jensen et al., 1974; Kuehl et al.,

1980 and Jansson et al., 1991]. Generally, the sample is dissolved in a non-aromatic solvent and transferred to an open column filled with charcoal. Using hexane as the mobile phase, non-planar compounds such as most of the PCBs will elute from the column while PCNs and other planar compounds are retained on the column. The mobile phase is then changed to an aromatic solvent such as toluene, which liberates the planar compounds from the charcoal [Jansson et al., 1991; Falconer et al., 1995 and Harner et al., 1997]. Activated carbon dispersed on a LiChrospher RP-18 phase has been used in high-performance liquid chromatography (HPLC) for PCN analysis [Lundgren et al., 1992 and Falandysz et al., 1996].

Järnberg et al. (1994) tested several GC stationary phases for separation of the individual CN congeners in the Halowax mixtures [Järnberg et al., 1994]. On phenyl-methylpolysiloxane (5%) columns, frequently used in PCN analysis, the CN congeners elute in homologue groups. In general, more polar stationary phases give wider retention time intervals for each homologue group. On the most polar stationary phases the different chlorination degrees overlap each other and the elution order is sometimes altered [Järnberg, 1994 and 1997]. To completely resolve the CN congeners, different types of gas chromatographic stationary phases need to be used. Using three types of cyclodextrine columns and a liquid crystalline column eight of the ten hexaCN congeners have been separated [Imgawa et al., 1997]. The PCN profiles of Halowax and fly ash seen in gas chromatograms are very different. When the PCNs in an unknown sample are to be identified, it is advisable to compare the PCN profile in the unknown sample with chromatograms of both Halowax 1014 and fly ash.

In order to avoid interferences from coeluting OHS, PCNs are preferably analyzed by gas chromatography-mass spectrometry (GC-MS). The response in electron-capture negative ionization (ECNI-MS) is high for penta- up to octa-CN, but varies considerably between congeners. Electron ionization (EI) is therefore the advantageous method when several PCNs are included in the analysis. For quantitative analysis, selected ion monitoring (SIM) detecting molecular ions should be used to obtain highest sensitivity. For samples with high levels of PCNs, low-resolution mass spectrometry (LR-MS) could be sufficient [Järnberg et al., 1993] but for low levels, or when high selectivity is necessary, high-resolution mass spectrometry (HR-MS) is the preferred method [Asplund et al., 1994 and Falanysz et al., 1996].

A troublesome circumstance in PCN analysis has been the limited commercial availability of authentic standards. There are currently no commercially available ¹³C-labeled PCN congeners and therefore ¹³C-labeled PCBs are frequently used as internal standards also for PCN analysis. Quantification has to be performed with few individual congeners and should therefore preferably be performed with GCMS and EI, which gives similar responses for all CN congeners of the same chlorination degree [Järnberg et al., 1993]. Wiedmann and Ballschmiter (1993) have suggested a method for quantification of PCNs based on only one or two CN congeners [Wiedmann et al., 1993]. They found that the molar response for CNs in EI increased with increasing degree of chlorination. The various methods influence the result of the quantification and this has to be considered when levels and profiles from different publications are compared.

One study presented concentrations of PCNs in a dated sediment core from a Swiss lake. GC/MS analysis was carried out on a high resolution mass spectrometer. Limits of detection (LOD) based on a signal-to-

noise ratio > 3 in the chromatogram were 0.002-0.008 ng/g dw for each PCN congener [Bogdal et al., 2006]. Krauss and Wilcke (2003) quantified 35 di- to octachlorinated PCNs in urban topsoils. Soils were extracted with hexane:acetone 2:1 in an Accelerated Solvent Extractor. The extracts were purified by solid phase extraction with aluminum oxide. GC-MS was used to detect PCNs. The lower limit of determination of PCNs was about 2–10 ng/kg for each PCN congenere [Krauss and Wilcke, 2003]. Chloronaphthalenes (CNs) were detected in the emissions of polyvinylchloride (PVC) combustion by Wang et al. CNs were analyzed with an GC-MSD with electron impact ionization (EI) in selected ion monitoring (SIM) mode. The limits of detection (LODs), defined as a signal-to-noise of 3:1 in blank samples, were in a range of 0.1–5.1 µg/kg PVC. A sensitive and rapid method for the determination of polybrominated diphenyl ethers (PBDEs), polybrominated biphenyls (PBBs) and polychlorinated naphthalenes (PCNs) in sediment samples has been proposed by Yusa et al. (2006). The method involving microwave-assisted extraction (MAE), GPC clean-up and large-volume injection (LVI) gas chromatography (GC)–ion trap mass spectrometry (ITMS). The limit of detection (LOD) ranged between 4 and 20 pg/g dry weight (dw) when 5 g of sediment were analysed [Yusa et al., 2006].

Standard methods for quantitative analysis of PCN

ISO 15680:2003 specifies a general method for the determination of volatile organic compounds (VOCs) in water by purge-and-trap isolation and gas chromatography (GC). Annexes A, B and C provide examples of analytes that can be determined using ISO 15680:2003. They range from difluorodichloromethane (R-12) up to trichlorobenzene, including all non-polar organic compounds of intermediate volatility. Detection is preferably carried out by mass spectrometry in the electron impact mode (EI), but other detectors may be applied as well. The limit of detection largely depends on the detector in use and the operational parameters. Typically detection limits as low as 10 ng/l can be achieved. The working range typically is up to 100 micrograms per litre. ISO 15680:2003 is applicable to drinking water, ground water, surface water, seawater and to (diluted) waste water.

7.4.6.2 Conclusion

The listed literature methods are designed to detect PCN at quite low levels in the corresponding matrices, however, it has to be emphasised that the achievable levels can vary and strongly depending on the individual matrix to be analysed, especially in the case of a matrix like waste which covers different. Although standard methods have only been identified for water analysis, there are analytical methods available to determine the mass concentration of PCN in environmental samples which define critical performance criteria like LOD or recovery.

Based on the results of the world-wide investigation of standardised measurement methods it can be stated that GC/MS techniques are state-of-the-art in the field of confirmatory analysis for PCN. Performance criteria are quite comparable and different extraction and clean-up procedures are available and sufficient for analysing a broad spectrum of matrices. Extraction and clean-up procedures have to be adapted to the relevant type of waste (matrix) and have to consider coelution of PCBs which can interfere the GC-MS measurement. Against this background, achievable LODs for waste matrices might be higher than for other matrices like water or biological samples, which has to be taken into account when defining a LOD for sludge or ash. However, the relevant waste matrices regarding PCN

analysis seem to have comparable properties as environmental samples like sediment or soil. From this point of view, the available techniques are sufficient to allow analytical control of limit values.

7.4.7 SCCPs

At the moment neither natural occurrence nor unintentional production of SCCP are known. The rubber industry is currently the main application of SCCP followed by the sealants and adhesives sector and paints and varnishes. The textile industry covers only a small fraction of the overall used amount of SCCP. Also leathers which have been impregnated with SCCP in the past can enter the waste stream due to their long lifetime.

7.4.7.1 Existing Methods

Analytical methodology for the reliable determination of SCCPs is scarce. This is mainly due to the number of isomers (up to 6,300) of which this class of compounds consists, and the lack of pure solutions of individual SCCPs for calibration as well as of matrix reference materials [Shojania, 1999]. The chromatograms of SCCP mixtures have a characteristic broad profile corresponding to a large number of coeluting compounds because the separation of the different isomers is not possible, even when using several stationary phases with different characteristics.

Summary of analytical strategies for quantitative analysis of SCCPs

Extraction usually does not represent a problem, because the same procedures used for the determination of other organochlorine compounds are suitable and applicable also to the extraction of SCCPs. Well-known more or less routine applicable methodologies can be applied such as Soxhlet, accelerated solvent extraction (ASE), solid-phase micro-extraction (SPME), or solid-phase extraction (SPE). The clean-up step can be regarded as more critical because other organic compounds, potentially co-extracted together with SCCPs (pesticides, PCBs, toxaphenes and chlordanes), might represent a problem in the following steps of the analytical process. The choice of the clean-up procedure mainly depends on the selectivity of the detection system applied: electron capture detector (ECD), low resolution mass spectrometer (LRMS) or high resolution mass spectrometer (HRMS). A strict clean-up procedure, including e.g. fractionation of the extracts and gel permeation chromatography (GPC), could be necessary. This implies long and more expensive analytical conditions. At present, the clean-up is one of the steps that make the analysis of SCCPs so challenging.

At present, no gas chromatographic technique is able to separate SCCPs, partly or completely, into single isomers, even when applying lengthy and expensive clean-up procedures, and when using several stationary phases of different polarity. Most quantitative SCCP methods are based on gas chromatography (GC) combined with electron capture detection (ECD) as well as high or low resolution mass spectrometry (HRMS or LRMS) [Nicholls et al. 2001, Tomy et al. 1999]. Detection of SCCPs by mass spectrometry often involves electron capture negative ion detection (ECNI) using methane as reagent gas due to its high selectivity and sensitivity [Coelhan 1999, Froescheis and Ballschmiter 1998, Gjos and Gustavsen 1982, Schmid and Müller 1985, Tomy et al. 1999b]. Other techniques were developed by employing electron ionization (EI) MS or tandem MS [Castells et al. 2004, Junk and Meisch 1993, Moore et al. 2003, Zencak et al. 2003, Zencak et al. 2004]. The use of comprehensive two-dimensional gas

chromatography (GC) has recently shown that an improvement in the separation of SCCPs is possible [Korytar et al., 2005 a and b], although neither routinely nor quantitatively applicable. The use of mass spectrometry (MS) in the electronic ionisation mode generates an extensive fragmentation of SCCPs with unspecific patterns difficult to interpret. Therefore, analysis is currently mostly performed in the electron capture negative ionisation mode.

Recently, GC×GC with electron capture negative ion detection time-of-flight mass spectrometry (ECNI-TOF-MS) was used to study the composition of SCCP, MCCP and LCCP mixtures [Korytar et al. 2005b]. The applicability of this method was shown for dust samples. Zencak et al. and Castells et al. compared different mass spectrometric methods for the determination of chloroparaffins. The results of the investigated methods were comparable for the analysis of standards and real samples (fish or sediment) and showed a satisfying accuracy including those based on low resolution MS (LRMS) [Zencak et al., 2005 and Castells et al., 2004]. An analytical routine procedure to classify chlorinated paraffins in technical products such as cutting fluids and lubricants has been presented by Randegger-Vollrath. After sample clean up with solid phase extraction over silica, screening was performed with gas chromatography and electron capture detection (ECD). Positive identification and quantitation was then performed with gas chromatography/mass spectrometry using negative chemical ionisation (NCI). Both methods showed good reproducibility and repeatability and the average recovery of the chlorinated paraffins with the NCI method is 98%. The detection limits for restricted paraffins in samples ranged between 0.02–0.08% (w/w) for the ECD method and between 0.2–2.6% (w/w) for the NCI method [Randegger-Vollrath et al., 1997].

Kemmlein et al. reported a HRGC-LRMS method based on the dechlorination of CPs in the injector [Kemmlein et al., 2002]. It is based on the carbon skeleton GC [Cooke and Roberts, 1980], where SCCPs are catalytically hydrodechlorinated to the corresponding n-alkanes. By that the complexity of the chromatogram is enormously reduced and only four peaks of n-alkanes have to be quantified, which simplifies the calibration step. Information on the chlorine content is lost, but accurate quantification of the sum of SCCPs is possible. Koh et al. applied this approach for the determination of chlorinated paraffins in cutting fluids and sealing materials [Koh et al., 2003]. Pellizzato et al. have been working in applying this approach to the analysis of environmental samples. Their method for the accurate determination of SCCPs in soil and sediment samples is based on carbon skeleton GC. The procedure includes the following steps: extraction from the soil or sediment using an accelerated solvent extractor (ASE), clean up and fractionation by column chromatography with Florisil and quantification by GC–MS equipped with a heated palladium catalyst placed in the GC injector. The linear range of calibration using n-alkanes covered concentrations from 0.04 to 18 mg/L and LODs ranged between 0.2 and 0.7 nmol/g. This method could be a candidate to the standardisation procedure for a fit-for-regulatory-purposes approach for the determination of SCCPs [Pellizzato et al., 2009].

The use of high-performance liquid chromatography combined with chloride-enhanced atmospheric pressure chemical ionization for the determination of chlorinated paraffins was described as alternative to gas chromatographic methods [Zencak and Oehme 2004].

Standard methods for quantitative analysis of SCCPs

ISO/DIS 12010 Water quality - Determination of short-chain polychlorinated alkanes (SCCP) in water - Method using gas chromatography/mass spectrometry (GC/MS) and negative-ion chemical ionization (NCI) is currently under development. Further standards regarding determination of SCCPs have not been identified.

Table 7-9: Summary of the results of a survey on standardised analytical methods and methods reported in literature to be considered for SCCP analysis.

Method	Matrix	Extraction/Clean-up	Detection	LOD and LOQ	Reference
Sealing material	C10-C29-CPs	Soxhlet, Silica gel minicolumn	GC/ECD or GC/FID	0.6 mg/kg	[Koh et al., 2002]
Polyurethane foam (PUF)	SCCPs	Soxhlet	HRGC/ECNI-HRMS	60 µg/L	[Peters et al., 2000]
Polyurethane foam (PUF)	SCCPs, MCCPs	PFE, Florisil column	HRGC/ECNI-HRMS	n.a.	[Barber et al., 2005]
Sediment	SCCPs	PFE, Copper treatment/ Florisil column	HRGC/ECNI-HRMS	MDL=11 µg/kg	[Tomy et al., 1999a]
Sediment	SCCPs	Hot solvent extraction, H ₂ SO ₄ treatment/copper treatment/silica gel column/GPC short-column	GC/ECNI-LRMS	100 pg/µL MDL=0.5 µg/kg	[Stejnarova et al., 2005]
Sediment	SCCPs, MCCPs	Soxhlet, Silica gel column	HRGC/ECNI-LRMS	1 ng/µL MQL=200 µg/kg	[Nicholls et al., 2001]
Sediment	SCCPs, MCCPs	Soxhlet×2, Sulfuric acid treatment/GPC	HRGC/ECNI-HRMS	n.a.	[Borgen et al., 2003]
Sediment	SCCPs	MAE Florisil column	HRGC/ECNI-LRMS	MDL=1.5 µg/kg	[Parera et al., 2004]
Sediment	SCCPs	Soxhlet Florisil column	HRGC/ECNI- MS	MDL=20–25 ng	[Castells et al., 2002]
Sediment	CPs	Soxhlet Silica gel column/SPE	GC×GC/ECNI-TOF-MS	n.a.	[Korytar et al., 2005c]
Sediment	SCCPs	Extract with acetone:hexane (1:1, v:v); purify by alumina chromatography	GC/MS	5 ng	[Schmid & Müller, 1985]
Sediment	SCCPs	Soxhlet extraction with hexane, silica gel and aluminium oxide column chromatography	GC/ MS-NCI	1 µg/L	[Steele et al., 1988]
Dust	CPs	Soxhlet GPC/H ₂ SO ₄ treatment/ silica gel column	GC×GC/µECD GC×GC/ECNI-TOF-MS	n.a.	[Korytar et al., 2005a]
Sewage sludge	SCCPs, MCCPs	Soxhlet H ₂ SO ₄ treatment/ silica gel column	HRGC/NICI-LRMS	1 ng/µL	[Nicholls et al., 2001]
Sewage sludge	SCCPs	Homogenize in acetone; extract with pentan; purify by alumina chromatography	GC/MS	5 ng	[Schmid & Müller, 1985]

7.4.7.2 Conclusion

Based on the results of the performed investigation of published measurement methods it can be stated that GC/MS techniques are state-of-the-art in the field of SCCPs analysis. Performance criteria depend on the kind of matrix which has been analysed. Soxhlet extraction followed by clean-up using column chromatography has been used for analysing a broad spectrum of matrices. This extraction and clean-up procedure seems to be suitable for most of the relevant types of waste (matrix) and can also be applied to rubber or plastic samples. Against this background, taking into account that extraction procedures are the most critical step in analysis of contaminants, achievable LODs for different waste matrices seem to be in the same range as for other matrices like environmental samples or PFU foams. However, the relevant waste matrices regarding SCCP analysis may have different properties regarding sample extraction compared to environmental samples like sediment or soil which has to be taken into account when defining a suitable LOD for waste matrices. Nevertheless, the available techniques are sufficient to allow analytical control of limit values at a quite low ppm level.

7.4.8 HBCD

At present, the main uses of HBCD are in the polymer and textile industries. HBCD 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. The focus of this study is on waste of EPS and XPS for the construction sector, EPS/XPS outside the construction sector, HIPS for EEE and on polymer dispersions for textile products. The relevant waste quantities are expected to significantly increase, due to huge amounts of HBCD incorporated into new EPS/XPS products for construction purposes and the relatively long life times of the products. The quantity of relevant wastes is expected to reach its maximum around 2050.

7.4.8.1 Existing Methods

Standard methods for determination of HBCD in waste matrices or related materials have not been identified. Nevertheless, in scientific literature there are numerous reports available describing analytical procedures suitable to quantify HBCD in different environmental and waste matrices. In general, analytical methods for determination of HBCD are quite similar compared to methods for analyzing PBDEs, which have already been summarized in this chapter. Most of the methods listed in this chapter are suitable to simultaneously detect several brominated flame retardants. Furthermore, all screening methods listed in the PBDEs subchapter can also be used for determination of HBCD as the common principle of all these methods is the detection of bromine in the relevant matrix.

Summary of analytical strategies for quantitative analysis of HBCD

As these screening methods have already been intensively discussed in the PBDEs chapter, detailed information can be found there. Suitability of XRF for HBCD analysis has for example been demonstrated by Allen et al. who reported a strong correlation between bromine concentrations in consumer products as measured by a handheld XRF and GC/MS. Kajiwar et al. computed the concentrations of LC-MS/MS-measured and HRGC/HRMS-measured bromine in textile samples based on bromine contents in HBCD and BDE 209 molecules, respectively and could demonstrate that bromine contents in

textile samples measured by XRF were highly correlated with MS-measured bromine levels, indicating that bromine sources for textiles were technical HBCD and Deca-BDE mixtures, respectively.

The analysis of flame retardants in polymers has been realised by gas or liquid chromatographic techniques. GC favourably coupled to MS detection allows both a high-resolution chromatographic separation and a high specific (and sensitive) detection. However, some flame retardants are characterised by high boiling points and require high injection and elution temperatures, which may lead to analytical artefacts due to thermal degradation. Because the thermal rearrangement of HBCD diastereomers and the decomposition of HBCDs occur at temperatures above 160 and 240°C, respectively, gas chromatography–mass spectrometry is not considered an optimal technique for the analysis of HBCD diastereomers (Peled et al., 1995; Covaci et al., 2007; Köppen et al., 2008). Presently, liquid-chromatography-coupled tandem mass spectrometry (LC–MS/MS) is used frequently for HBCD analysis, because it can resolve HBCD diastereomers and because no thermal degradation or analyte composition transmutation occurs during analysis. HBCD consisting of three diastereoisomers, can be separated and determined successfully by using LC–MS or LC–MS/MS. These techniques are preferred for environmental analyses, but they can also be used for material analysis.

A high-performance liquid chromatography coupled with electrospray ionization tandem mass spectrometry (LC–MS/MS) was developed for the determination of HBCD diastereoisomers by Yu et al. A detailed study was carried out to optimize the composition of the mobile phase involving methanol/acetonitrile/water, and the values of MS/MS parameters. The instrumental limits of detection (LODs) on column in this study were 0.5, 0.3 and 0.3 pg for α -HBCD, β -HBCD and γ -HBCD, respectively. The method reported was further applied to air and soil samples. Three HBCD diastereoisomers were detected in all the air and soil samples, with levels ranging from 1.2 to 1.8 pg/m³ and 1.7 to 5.6 µg/kg dry weight, respectively [Yu et al., 2008]. A method to determine HBCD in water and sediments using solid phase extraction (SPE) and/or solvent extraction was presented by Suzuki et al. Recoveries from sediments were approximately 100% for all the chemicals. Technical difficulties in HBCD measurement were discussed in terms of the physico-chemical properties of HBCD isomers. The method was applied to landfill samples and marine sediment. In applications to landfill leachates, the limit of detection for SIM using APCI was 2 ng/L for each HBCD diastereoisomer. Marine sediment from an urban area contained up to 860 ng/L γ -HBCD [Suzuki et al., 2006].

Schlummer et al. developed a HPLC–UV/MS method to identify and quantify flame retardants in post-consumer plastics from waste of electric and electronic equipment (WEEE). The method was applied to detect flame retardant additives in polymer extracts obtained from pressurised liquid extraction of solid polymers (ASE) using iso-octane. In addition, a screening method was developed for soluble styrene polymers to isolate a flame retardant fraction through the application of gel permeation chromatography (GPC). This fraction was transferred to an online-coupled HPLC column and detected by UV spectroscopy, which allowed a reliable qualitative and quantitative analysis of brominated flame retardants in the polymer solutions. For UV detection, the computed detection limits of the reference materials including HBCD were in the range between 0.5 and 100 ppm. Significantly reduced detection limits were found with MS detection. They were in the range between 0.01 and 1 ppm [Schlummer et al., 2005]. The extraction efficiency of pressurised liquid extraction (PLE), microwave-assisted extraction (MAE), and ultrasonic-assisted extraction (UAE) under different conditions has been compared for the

recovery of the most commonly employed brominated flame retardants (BFRs) from styrenic polymeric matrixes by Vilaplana et al. A HPLC–MS/MS method has been proposed for the simultaneous separation and quantification of tetrabromobisphenol A (TBBPA), hexabromocyclododecane (HBCD) diastereomers, and decabromodiphenyl ether (deca-BDE) in the polymeric extracts. PLE results in complete extraction of TBBPA and HBCD (95–100% recovery), and intermediate recovery rates for deca-BDE (50%). MAE, on the other hand, gives comparable performance to PLE for HBCD, but lower extraction yields for TBBPA and mainly deca-BDE. Ultrasonication, finally, offers relatively low extraction recoveries (10–50%). Following LODs have been reported α -HBCD 2.7 $\mu\text{g/ml}$, β -HBCD 1.2 $\mu\text{g/ml}$ and γ -HBCD 5.0 $\mu\text{g/ml}$ [Vilaplana et al., 2008]. Pöhlein et al. developed two versatile and fast methods to identify and quantify brominated flame retardants (BrFRs) in styrenic polymers. Gas chromatography/mass spectrometry (GC/MS) as well as gas chromatography with electron-capture detection (GC/ECD), both following ultrasonic-supported dissolution and precipitation (USDP) were applied. The substance range includes poly-brominated biphenyls (PBBs) and diphenyl ethers (PBDEs), as well as other commonly used flameretardants (FRs), including HBCD. The methods were verified using congener standards and flame-retardant polymer samples. Good recoveries were found. LODs for HBCD were 6.1 ng/cm^3 (GC-MS standard solution), 92 mg/kg (GC-MS polymer reference), 0.03 ng/cm^3 (GC-ECD standard solution) and 0.8 mg/kg (GC-ECD polymer reference) [Pöhlein et al., 2008]. Thirteen laboratories in Japan took part in an intercalibration study intended to evaluate the accuracy and reliability of the analysis of organobromine compounds, including PBDE, TBBP-A, TBP, and HBCD, in waste TV cabinets. In-house validated methods based on GC–MS or GC–HRMS were used for the determination step. The relative standard deviations for PBDE ranged from 10 to 39% [Takahashi et al., 2005].

Kajiwarra et al. established a concise and rapid procedure to analyze hexabromocyclododecane (HBCD) diastereomers in flame-retarded textiles. They compared three different methods of extraction—Soxhlet, ultrasonic, and soaking extractions with toluene and dichloromethane (DCM). During Soxhlet extraction using toluene, the percent contribution of a-HBCD to total HBCDs increased slightly and that of c-HBCD decreased, indicating that c-HBCD was isomerized to some extent at the boiling point of toluene. For ultrasonic extraction, the temperature of the water bath can easily increase over time during the procedure, which might lead to undesirable effects. Therefore, soaking extraction with DCM was considered to be the most facile procedure to analyze HBCD diastereomers in textiles. Identification and quantification of HBCD diastereomers were carried out by using a liquid chromatograph equipped with a triple quadrupole mass spectrometer. The instrumental detection limit was 3–4 pg , which was calculated from standard deviations of five replicate analyses of 5 ng/mL standard solution [Kajiwarra et al., 2009].

7.4.8.2 Conclusion

Based on the results of a literature search on standardised measurement methods and methods reported in scientific articles it can be stated that LC/MS and LC-MS/MS techniques are state-of-the-art in the field of standardised confirmatory analysis for HBCD. Analytical methods for simultaneously determination of several brominated flame retardants covering PBDEs besides HBCD have been developed. Performance criteria are quite comparable and different extraction and clean-up procedures are available and sufficient for analysing a broad spectrum of matrices. From this point of view, the available techniques are sufficient to allow analytical control of limit values.

Due to the inhomogeneity of the matrix waste, it seems to be very challenging to develop one standard method for HBCD analysis in all types of waste. In view of the big differences between the identified relevant waste matrices, different extraction procedures have to be used when analysing waste matrices. As extraction in most cases is the most sensitive part of an analytical method, it has to be taken into account that quality criteria like LOD or recovery might be different for different waste matrices. This has to be taken into account when defining a suitable LOD for analytical methods to determine the HBCD content (content of brominated flame retardants) in waste as a basis for establishing threshold limit values.

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8 Scenarios and Prognosis on future Development

Chapter 5 and 6 of this report describe the mass flows of new and candidate POPs. Chapter 9 presents a methodology to establish low POP concentration limit and maximum POP concentration limit values and results in proposals for feasible limits for the different substances. Results from current mass and waste flows are a major input for the development of limit values in chapter 9. Limit values on the other hand may have an important impact on future mass flows. Therefore in chapter 0 different scenarios are presented and discussed in order to evaluate the consequences of different possible limit values.

8.1 Function of Limit Values under the EU POP Regulation

The POP regulation (EC) No 850/2004 has the purpose to clearly define the management regime for wastes consisting of, containing or contaminated with POPs (further referred to as POP wastes) in accordance with the provisions of the Stockholm Convention.

To this purpose, besides other, the two following sets of limit values for POP concentration in wastes have been established

1. A "concentration limit referred to in article 7 (4)(a)" in Annex IV hereafter referred to as **"low POP concentration limit"** (LPCL) for the classification as POP waste
2. A "maximum concentration limit of substances listed in Annex IV" (Annex V) hereafter referred to as **"maximum POP concentration limit"** (MPCL) concerning derogation from destruction

The low concentration limit values (LPCL) define the threshold concentration above which wastes are classified POP waste and subject to the management regime of the POP regulation. These limits define whether a waste, due to its POP content, will have to be treated by the destruction or irreversible transformation methods defined in Annex V, part 1⁴⁴ to the regulation. The LPCL corresponds to the low POP content defined under the Stockholm Convention. The LPCL is quantitatively stipulated in the General technical guidelines for the environmentally sound management of wastes consisting of, containing or contaminated with persistent organic pollutants.⁴⁵

By way of derogation from article 7, paragraph 2, alternative treatment options –as defined in Annex V⁴⁶ - may be permitted by Member States for single waste codes listed in Annex V⁴⁷ if destruction and/or irreversible transformation do not represent the environmentally preferable option and if the maximum

⁴⁴ D9, D10, R1, R4

⁴⁵ See paragraph 29 [BC TG 2007]. In these guidelines LPCLs are currently only determined for PCBs, PCDD/Fs, Aldrin, chlordane, DDT, dieldrin, endrin, heptachlor, HCB, mirex and toxaphene.

⁴⁶ permanent storage in safe, deep, underground hard rock formations, permanent storage in salt mines, permanent storage on a landfill for hazardous waste

⁴⁷ 10 01 14 *, 10 01 16 *, 10 02 07 *, 10 03 04 *, 10 03 08 *, 10 03 09 *, 10 03 19 *, 10 03 21 *, 10 03 29 *, 10 04 01 *, 10 04 02 *, 10 04 04 *, 10 04 05 *, 10 04 06 *, 10 05 03 *, 10 05 05 *, 10 06 03 *, 10 06 06 *, 10 08 08 *, 10 08 15 *, 10 09 09 *, 16 11 01 *, 16 11 03 *, 17 01 06 *, 17 05 03 *, 17 09 02 *, 17 09 03 *, 19 01 07 *, 19 01 11 *, 19 01 13 *, 19 01 15 *, 19 04 02 *, 19 04 03 *,

POP concentration limit is not exceeded. In practice this means for example that combustion residues as well as non-combustible POP wastes may be disposed of underground or on hazardous waste landfills up to the MPCL. The MPCL in Annex V of the POP regulation does not have a correspondence in any of the international Conventions on POPs.

8.2 General Implications of Limit Values

8.2.1 Implications of low POP concentration limits

The stricter the low POP concentration limit for POP waste will be, the more wastes will be classified as POP waste and consequently will require incineration (D10), physico-chemical treatment (D9), energy recovery/use as secondary fuel (R1), recycling /reclamation of metals and metal compounds (R4) or (by way of derogation for specific EWC codes) disposal in safe, deep, underground hard rock formations, salt mines or hazardous waste landfill sites.

This means that existing mass flows will change in the following way in case of a stricter low POP concentration limit (↑ = increase expected; ↓ decrease expected).

hazardous waste incineration: ↑

hazardous waste landfilling: ↑

municipal solid waste incineration: ↑

co-incineration: ↑

hazardous waste landfilling and underground disposal: ↑

non-hazardous waste landfilling: ↓

inert waste landfilling: ↓

recycling operations: ↓

Waste incineration, energy recovery operations, high temperature metal reclamation (where relevant) as well as hazardous waste landfilling and underground storage will receive all waste volumes that will be classified as POP waste and thus increase in volume in relation to the decrease in volume of non-hazardous waste landfilling, inert waste landfilling and recycling operations.

8.2.2 Implications of maximum POP concentration limits

The stricter the MPCL for POP waste will be, the more of the specific waste codes concerned (ashes and slags, C&D wastes) will require (physico-chemical treatment D9 or metal reclamation (R4) to reduce the POP content. Incineration, energy recovery and would be allowed in theory, but are generally no valid options in reality (due to low calorific value).

This means that existing mass flows will change in the following way in case of a stricter maximum POP concentration limit:

hazardous waste landfilling: ↓

Underground disposal: ↓

(if no MPCL will be established for underground storage the effect will be: ↑)

8.3 Waste and Mass Flow Scenarios in Relation to Limit Values

8.3.1 Scenario for C-PentaBDE

As explained above the impacts of the low POP concentration limit on the overall mass flow of C-PentaBDE are a function of the limit value, the contamination level (average and range) of relevant waste types and the volume of waste streams concerned. Thus it is necessary to compile and evaluate the waste streams investigated in the mass flow chapters. An overview of amounts concerned at different low POP content limits (LPCL) is given in the figure below.

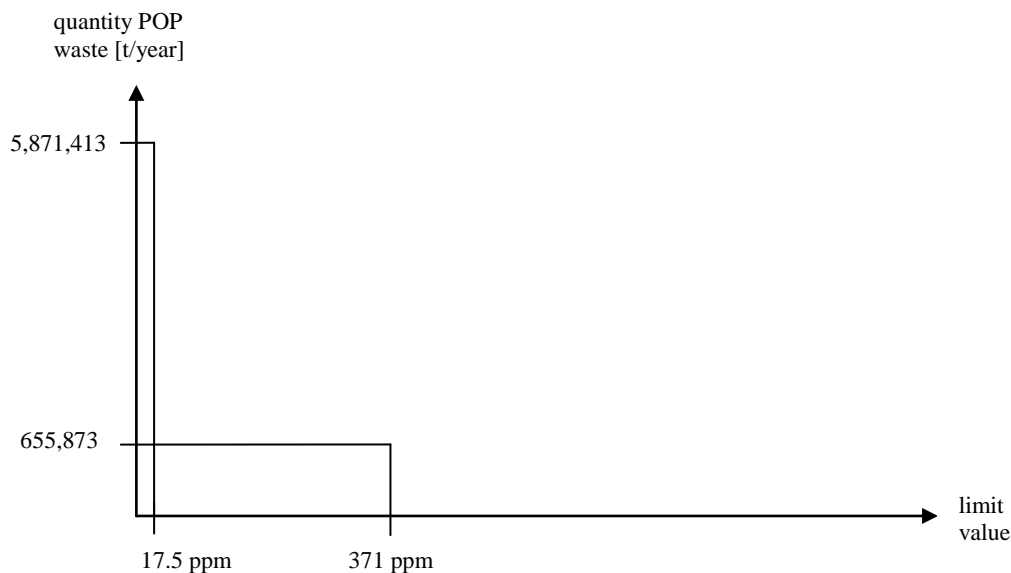


Figure 8-1: Estimated annual quantities of waste classified POP waste due to its C-PentaBDE concentration in relation to different low POP concentration limits

Based on the available data the following effects of different potential limits can be observed:

The first impact on waste treatment can be observed at 371 ppm, and is related to ~655,873 t of automotive PUR foam waste. It comprises around 265,629 t of automotive waste which is currently landfilled, about 265,629 t of automotive waste which is currently incinerated without energy recovery (D10), approximately 45,911 t of automotive PUR foam waste which is currently incinerated with energy recovery (R1) and ~78,049 t of automotive PUR foam waste which is currently recycled.

If classified as POP waste approximately 343,678 t will have to be directed to incineration (D10) and/or incineration with energy recovery (R1). R4 (metal reclamation) and D9 (physico-chemical treatment) are other potential treatment methods, which however, do not seem to be primary options, due to the characteristics and the composition of the waste.

By setting the LPCL at 17.5 ppm around 5,871,413 t of contaminated PUR foam waste from automotive and upholstery applications will be covered. Of the total 5,871,413 t approximately ~3,765,256 t are currently landfilled and around 78,049 t of automotive waste is recycled. This waste would have to be directed to incineration (D10) and/or incineration with energy recovery (R1). For the remaining 2,027,453 t incineration with and/or without energy recovery remains the preferred waste management option.

8.3.2 Scenario for C-OctaBDE

Electrical and Electronic Equipment has been identified as the only relevant source of C-OctaBDE in Europe, accounting to date for approximately 732 t/y of contaminated ABS plastic waste. This waste typically contains 17.5 % (175,000 mg/kg = 175,000 ppm) of C-OctaBDE. An overview of amounts concerned at different low POP content limits (LPCL) is given in the figure below.

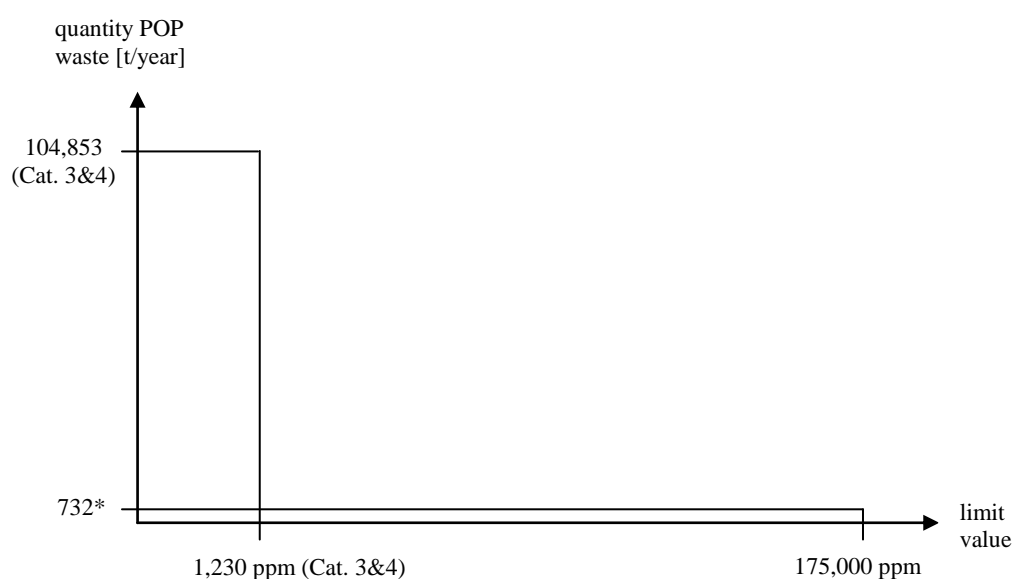


Figure 8-2: Estimated annual quantities of waste classified POP waste due to its C-OctaBDE concentration in relation to different low POP concentration limits (*C-OctaBDE contaminated plastics separated)

As regards the mass flow of C-OctaBDE two different scenarios are discussed in chapter 6.

The first scenario is based on the assumption that the C-OctaBDE contaminated ABS plastics are treated together with other non contaminated plastics of the WEEE category 3 (IT and communication) and WEEE category 4 (consumer electronics). This leads to a total figure of around 104,853 t/y waste to be treated with an average (mixed) contamination of approximately 1,230 ppm (see Figure 8-2). This waste will be finally landfilled (~32,242 t/y), incinerated without energy recovery (~32,242 t/y), incinerated

with energy recovery (~20,971 t/y) and recycled (~18,874 t/y). Around 524 t/y are possibly reused as a whole appliance and hence do not enter the waste regime.

Due to the provisions of Annex V changes in treatment are expected for the 32,242 t/y which are currently landfilled and the 18,874 t/y which are currently recycled. For the remaining waste, incineration with/without energy recovery remains the preferred waste management option.

The second scenario is based on the assumption that the 732 t/y of C-OctaBDE contaminated plastics (~175,000 ppm) can be successfully separated from non contaminated WEEE fractions as requested by the WEEE Directive. Most of this contaminated plastic fraction will then be incinerated with and without energy recovery (D10, R1), which are permitted treatment option also if classified as POP waste. In case incineration capacities are not available in a particular MS, underground disposal according to Annex V, Part II would be an alternative waste treatment operation, if the MPCL is not exceeded.

The discussions provided above are related to the C-PentaBDE and C-OctaBDE technical mixtures. For the congener groups of C-PentaBDE and C-OctaBDE possibly lower LPCL could be set. This will be discussed in the following.

8.3.3 Scenario for congener groups (Tetra-, Penta-, Hexa-, and Heptabromodiphenyl ether)

The scenarios by PBDE congener group as listed in the POP Regulation comprises all potential occurrences of the congeners comprising C-Penta- and C-OctaBDE material flows where relevant. In this context the relative average concentrations in the commercial mixtures are used for derivation of concentrations in the different waste fractions.

TetraBDE

An overview of the amounts concerned at different potential low POP content limits (LPCL) is given in Figure 8-3.

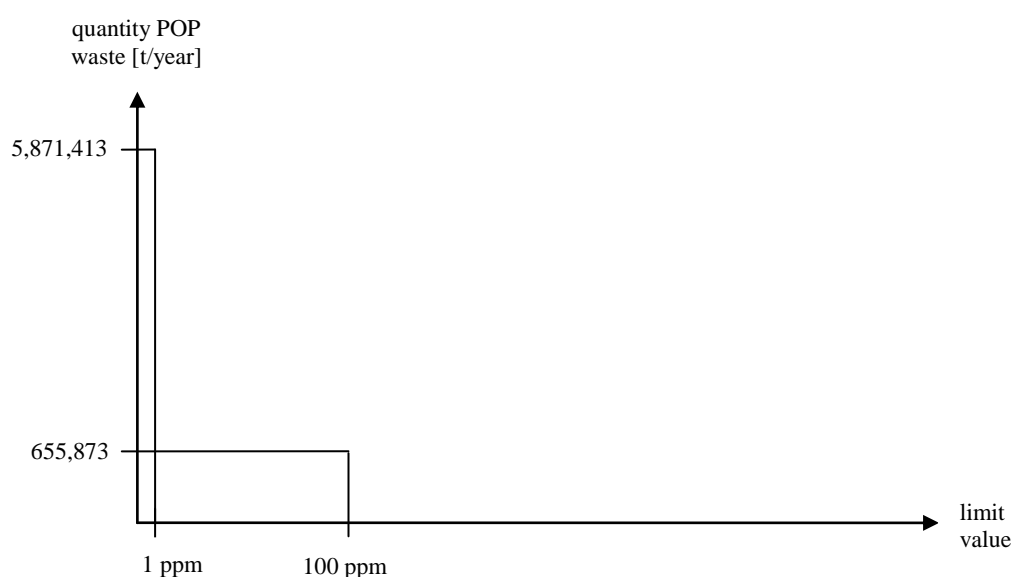


Figure 8-3: Overview of the waste amounts concerned at different LPCL (TetraBDE)

At a LPCL of 100 ppm approximately 655,873 t of PUR waste would be affected. This figure comprises around 265,629 t of automotive waste which is currently landfilled, the same amount of automotive waste which is currently incinerated without energy recovery (D10), around 45,911 t of automotive PUR foam waste which is currently incinerated with energy recovery and ~78,049 t of automotive waste which is currently recycled.

If classified as POP waste ~265,629 t of automotive waste which is currently landfilled and around 78,049 t of automotive waste which is currently recycled will have to be directed to incineration and/or incineration with energy recovery. R4 (metal reclamation) and D9 (physico-chemical treatment) are other potential treatment methods, which however, do not seem to be primary options, due to the characteristics and the composition of the waste. For the remaining (~312,195 t) incineration with/without energy recovery remains the preferred waste management option.

By setting the LPCL at 1 ppm around 5,871,413 t of contaminated PUR foam waste from automotive and upholstery applications will be covered. Of the total approximately ~3,765,256 t is currently landfilled and around 78,049 t recycled. This waste would have to be directed to incineration (D10) and/or incineration with energy recovery (R1) if classified as POP waste. The 3,765,256 t encompass ~3,499,627 t of landfilled upholstery waste and around 265,629 t of landfilled automotive waste. For the remaining ~2,028,108 t of automotive and upholstery waste, incineration with/without energy recovery remains the preferred waste management option.

PentaBDE

An overview of the amounts concerned at different potential low POP content limits (LPCL) is given in Figure 8-4.

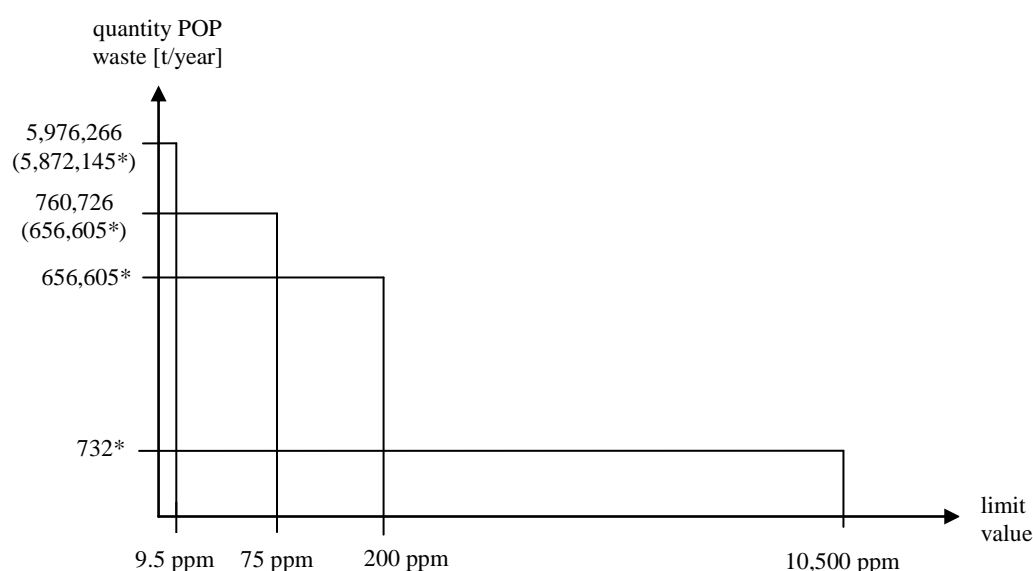


Figure 8-4: Overview of the waste amounts concerned at different LPCL (PentaBDE)

The first impact on waste treatment can be observed at 10,500 ppm, and is related to ~732 t* of WEEE. This is, however, only valid under the assumption that the WEEE contaminated plastics can be successfully separated from non-contaminated waste streams. Therefore, this waste stream has been highlighted with an asterisk.

By setting the LPCL at 200 ppm a significantly higher amount of waste will be considered as POP waste (~656,605 t*). It will be made up of ~655,873 t automotive waste and around 732 t* of WEEE (*WEEE contaminated plastics separated from the non-contaminated plastics). If classified as POP waste the ~265,629 t of landfilled and the 78,049 t of recycled automotive waste will have to be directed to incineration (D10) and/or incineration with energy recovery (R1). For the 732 t* of WEEE incineration with/without energy recovery remains the preferred waste management option.

By setting the LPCL at 75 ppm approximately 760,726 t (656,605 t*) of waste will be affected, dependant on whether the WEEE contaminate fraction is successfully separated or not. Again a large proportion of this waste stream will have to be treated alternatively (~394,794 t/y). The same amounts and types of waste would also be considered as POP waste at an LPCL of 50 ppm.

Additional waste streams would be considered as POP waste at an LPCL of 9.5 ppm. Such a limit would affect ~655,873 t/y automotive waste, 5,215,540 t/y of upholstery waste and ~104,853 t (732 t/y*) of WEEE, leading to a total figure of 5,976,266 t/y (5,872,145 t/y*).

If classified as POP waste, around 3,894,421 t (consisting of ~343,678 t/y landfilled and recycled automotive waste, ~3,499,627 t/y landfilled upholstery applications, ~18,874 t/y of reused/recycled and ~32,242 t/y of landfilled WEEE) will require alternative treatment.

HexaBDE

An overview of the amounts concerned at different potential low POP content limits (LPCL) is given in Figure 8-5.

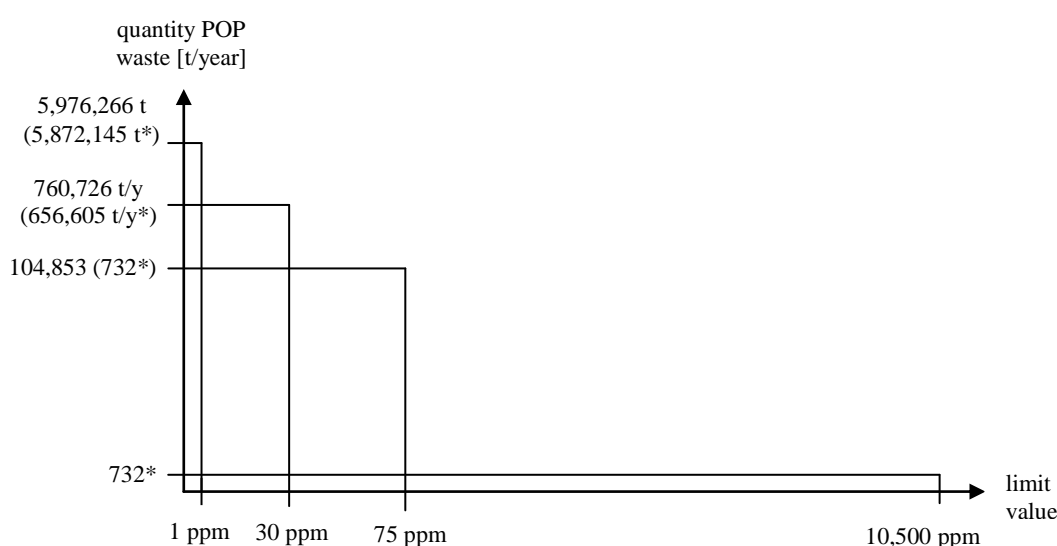


Figure 8-5: Overview of the waste amounts concerned at different LPCL (HexaBDE)

Similar to the PentaBDE, the first impact on waste treatment can be observed at 10,500 ppm, and is related to ~732 t* of WEEE (in case the contaminated plastic fraction can be successfully separated from non-contaminated WEEE plastics). This waste stream is currently incinerated with/without energy recovery to equal amounts. This will also remain the preferred waste management option.

At a LPCL of 75 ppm around 104,853 of WEEE will be affected (reflects the assumption that there is no successful separation of contaminated and non contaminated plastics before shredding for WEEE category 3 and 4 items). This amount would be only ~732 t* in case of complete separation of contaminated WEEE plastics. Around 32,242 t/y of WEEE which is currently landfilled and about 18,874 t/y which is currently recycled will require a changed treatment. The remaining waste can still be incinerated with and/or without energy recovery.

By setting the LPCL at 30 ppm approximately 760,726 t/y (656,605 t/y*) of waste will be covered. It will be made up of ~655,873 t automotive waste and around 104,853 t/y (732 t/y*) of WEEE. If classified as POP waste ~265,629 t/y of landfilled and ~78,049 t/y of recycled automotive waste, ~18,874 t/y of reused/recycled WEEE and ~32,242 t/y of landfilled WEEE will have to be directed to incineration (D10) and/or incineration with energy recovery (R1).

By setting the LPCL at 1 ppm a significantly higher amount 5,976,266 t (5,872,145 t*) of waste will be considered as POP waste as this LPCL also affects a large amount of upholstery waste. Again a large proportion of this waste stream will have to be treated alternatively (i.e. ~343,678 t/y landfilled and recycled automotive waste, ~3,499,627 t/y landfilled upholstery, ~18,874 t/y of reused/recycled and 32,242 t/y of landfilled WEEE).

HeptaBDE

An overview of the amounts concerned at different potential low POP content limits (LPCL) is given in Figure 8-6.

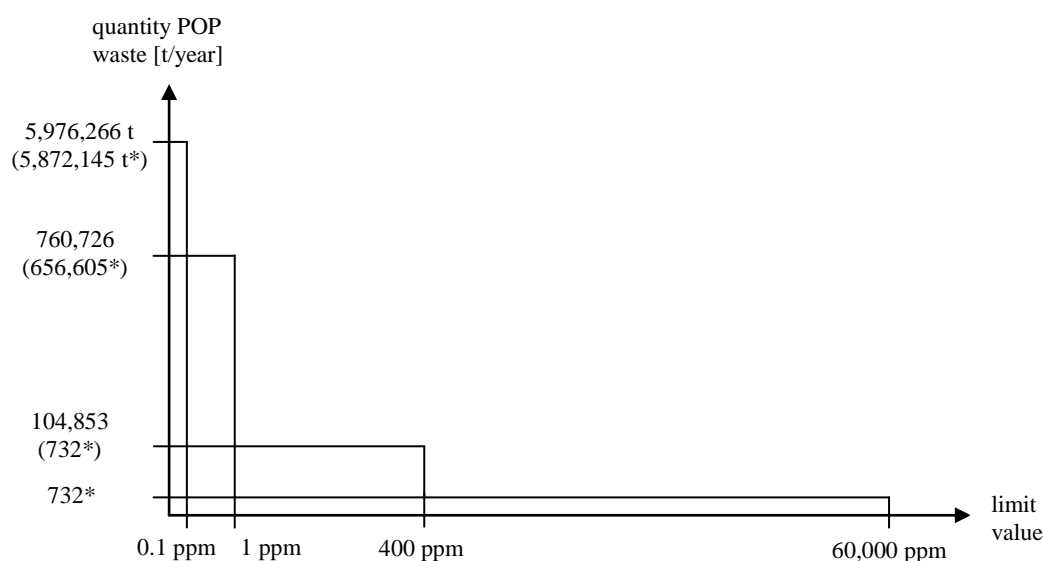


Figure 8-6: Overview of the waste amounts concerned at different LPCL (HeptaBDE)

The first impact on waste treatment can be observed at 60,000 ppm and is related to ~732 t* of WEEE. Incineration with/without energy recovery will remain the preferred waste management option.

At a LPCL of 400 ppm around 104,853 t of WEEE would be covered. This amount would be only ~732 t* in case of complete separation of contaminated WEEE plastics. Around 32,242 t/y of WEEE which is currently landfilled and about 18,874 t/y which is currently recycled will require a changed treatment. The remaining waste can still be incinerated with and/or without energy recovery.

By setting the LPCL at 1 ppm a higher amount of ~760,726 t (~656,605 t*) of automotive waste and WEEE will be covered. Again a relatively large proportion of this waste stream will have to be treated alternatively (~394,794 t). This waste comprises approximately 265,629 t/y of currently landfilled and ~78,049 t/y of recycled automotive waste, approximately 32,242 t/y of landfilled and about 18,874 t/y reused/recycled WEEE.

By setting the LPCL at 0.1 ppm a significantly higher amount 5,976,266 t (5,872,145 t*) of waste will be covered. In this case, approximately 3,894,421 t/y will have to be treated alternatively.

LPCL [ppm]	0.1	1	9.5	30	75	100	200	400	10,500	60,000
Waste amounts concerned at different LPCL [t/y]										
TetraBDE (c ₁)		5,871,413				655,873				
PentaBDE (c ₂)			5,976,266		760,726					
			5,872,145*		656,605*		656,605*		732*	
HexaBDE (c ₃)		5,976,266		760,726	104,853					
		5,872,145*		656,605*	732*				732*	
HeptaBDE (c ₄)	5,976,266	760,726						104,853		
	5,872,145*	656,605*						732*		732*
Waste amounts requiring alternative treatment if classified as POP waste [t/y]										
TetraBDE (c ₁)		3,843,305				343,678				
PentaBDE (c ₂)			3,894,421		394,794		343,678		-	
HexaBDE (c ₃)		3,894,421		395,794	51,116				-	
HeptaBDE (c ₄)	3,894,421	394,794						51,116		-

*PBDE contaminated WEEE plastics successfully separated from the non-contaminated plastic fractions

Table 8-1: Overview of the waste amounts concerned at different potential low POP content limits (LPCL)

8.3.4 Scenario for PFOS and its derivatives

As explained above the impacts of the low POP content limit on the overall mass flow of PFOS are a function of the limit value, the contamination level (average) of relevant waste types and the volume of waste streams concerned. Thus it is necessary to compile and evaluate the waste streams investigated in the mass flow chapters.

For PFOS a wide range of contamination values for different wastes have been derived from available literature. These range from 0.05 to 410,000 ppm. For PFOS and derivatives the proposed LPCL of 1000 ppm for semi-finished products or articles or parts thereof and 10 ppm in substances or in preparations (Article 4(1) (b))⁴⁸ do lie in the range identified and might have an effect on the PFOS waste streams.

Figure 8-7 gives an overview of the amounts of wastes concerned at different low POP concentration limits (LPCL) for PFOS. Setting a LPCL of 0.05 ppm would cover all wastes, which are generated from different industrial sectors (such as metal plating, photographic industry, semiconductor industry, aviation industry, waste water treatment) and would have impacts on articles in use such as fire fighting foams, leather upholstery and carpets.

⁴⁸ Draft COMMISSION REGULATION (EC) No .../.. of; amending Regulation (EC) No 850/2004 of the European Parliament and of the Council on persistent organic pollutants as regards Annexes IV and V (Text with EEA relevance); April 2010

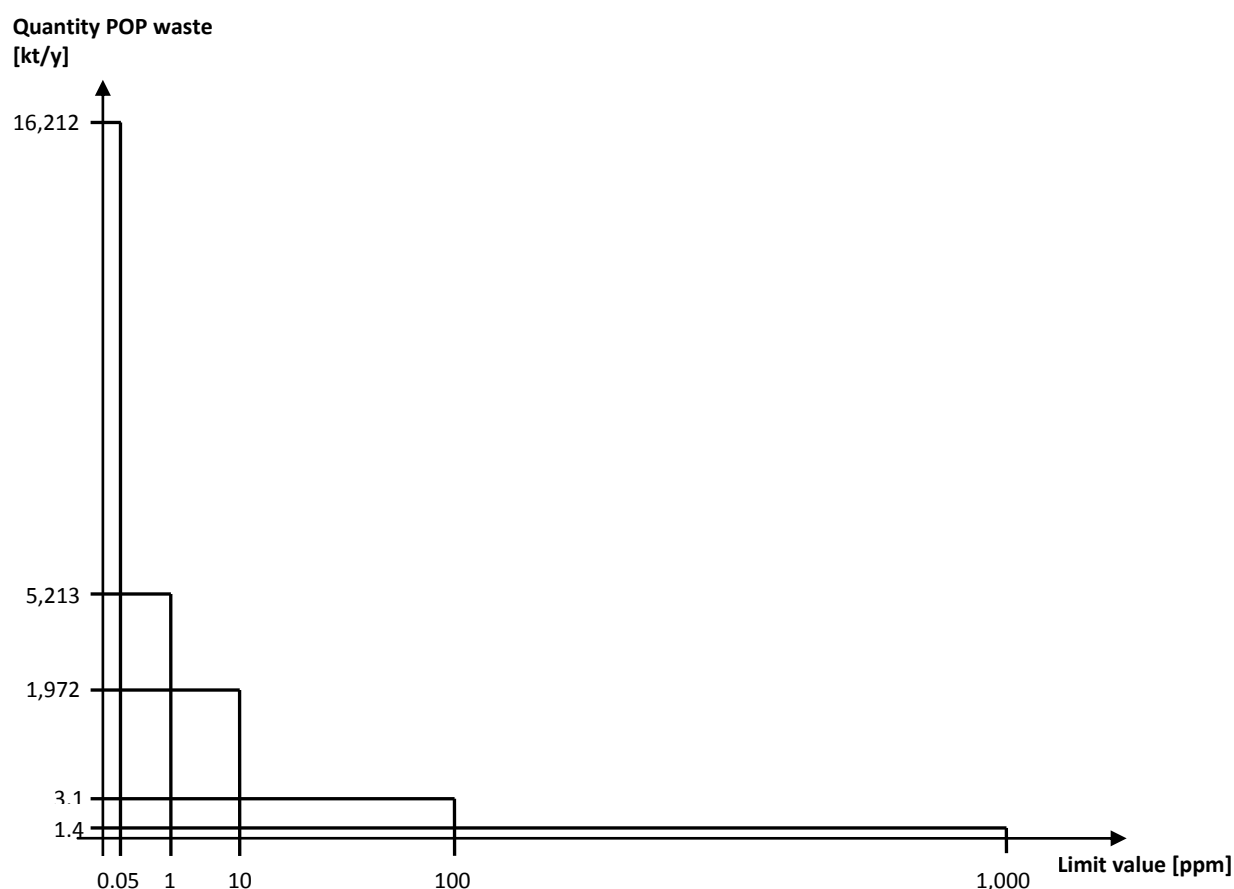


Figure 8-7: Estimated annual quantities in (kt/y) of waste classified POP waste due to its PFOS concentration (mg/kg = ppm) in relation to different low POP content limits

A more detailed compilation of the relation between limit value and amounts of wastes exceeding them, which illustrates the contribution of individual waste types, is given in Figure 8-8 and the related Table 8-2 below. The compilations indicate the exact annual amounts of waste in kt/y contributed by the individual source sectors in relation to specific potential LPCL.

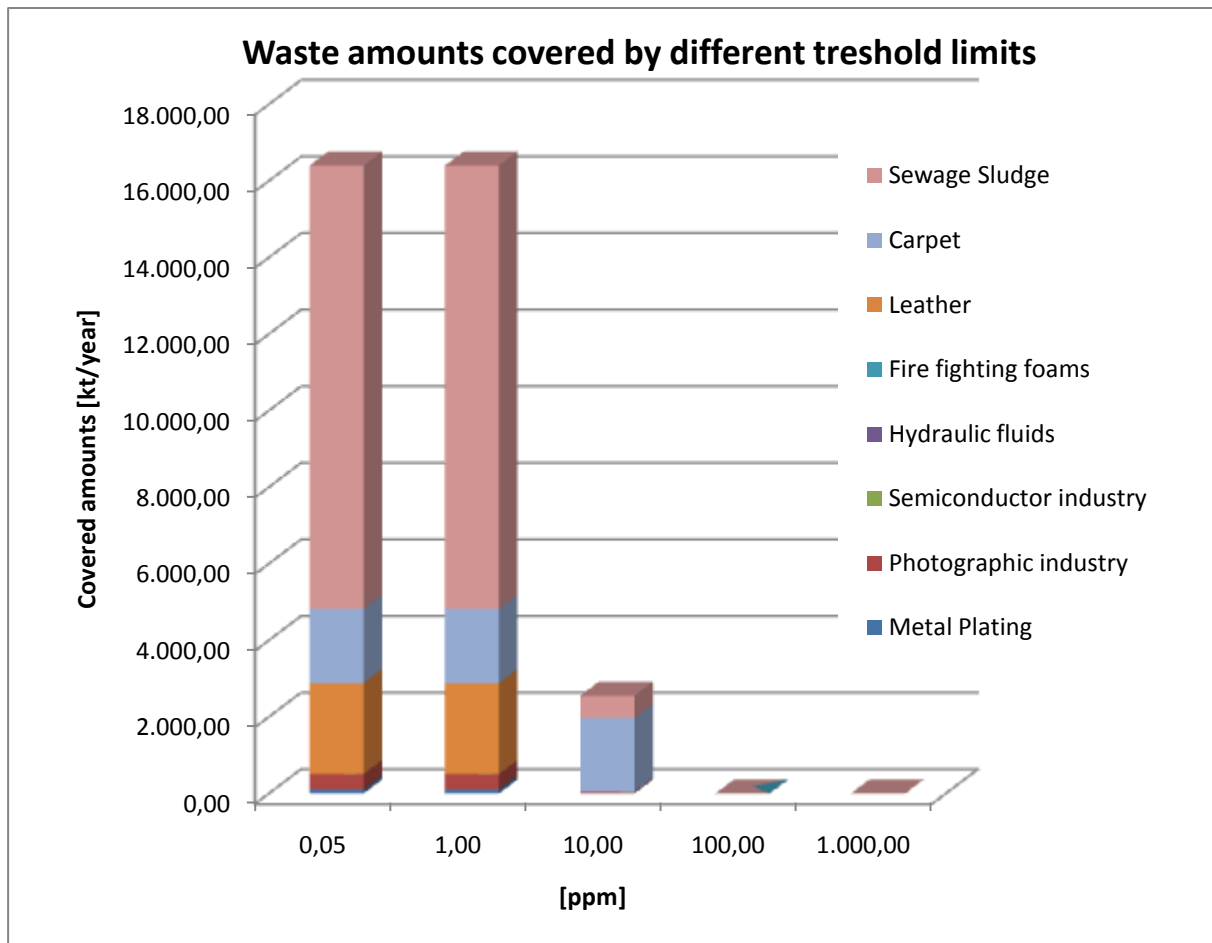


Figure 8-8: Contribution of specific wastes types to amounts exceeding different potential low POP content limits for PFOS in EU 27

Table 8-2: Overview of contribution of waste amounts contaminated with PFOS from specific sectors at different LPCL set

Sector ft. amount waste [kt/y] resulting from setting of a specific LPCL [ppm]	LPCL 0.05	LPCL 1.0	LPCL 10	LPCL 50	LPCL 100	LPCL 1,000
Metal plating	101.12	101.12	1.62	1.62	1.62	0.02
Photographic industry	214.98	214.98	30.96	0,02	0.02	0.02
Hydraulic fluids	0.72	0.72	0.60	0.60	0.60	0.60
Fire fighting foams	0.60	0.60	0.00	0.00	0.00	0.00
Leather upholstery	2,378.08	2,378.08	1,938.02	1,938.02	0.00	0.00
Carpet	1,938.02	1,938.02	0.00	0.00	0.00	0.00
Sewage sludge	11,578.00	578.90	1,971,915	1,940,975	2,957	1,358
Total	16,211,511	5,212,411	1,971,915	1,940,974	2,957	1,358
Share in %	100%	32%	12%	12%	0.009%	0.008%

As illustrated in Figure 8-8 and described in Table 8-2 the effect on wastes containing PFOS already starts at relatively high concentration levels above **1,000 ppm**. The maximum contamination concentration of 410,000 ppm and 127,000 ppm can be found for liquid photographic fluid wastes and solid residue from the x-ray recycling respectively. Both wastes are from the photographic industry (permitted use according to Annex I). This liquid photographic fluid waste occurs in small amounts of 166.5 kg/y and is subject to hazardous waste incineration. Further waste streams concerned at a limit of

1,000 ppm are the small amounts of resin wastes from ion exchange ($c=19,000$ ppm; $m=17.70$ kg,) waste stream in the metal plating sector. More important (permitted) are activated carbon filters from x-ray wastes recycling with a contamination concentration of 1,000 ppm amounting to 19,500 kg generated in the photographic sector, hydraulic fluids ($c=1,000$ ppm, $m=715,400$ kg) in the aviation industry and fire fighting foams ($c=10,000$ ppm, $m=0.6$ kt/y and 8.4 kt stored till 2011). All of these wastes are currently treated by hazardous waste incineration and are exempted from the ban on marketing and use according to Annex I under the provision that Member States report in a four year period on the progress to eliminate PFOS. All of the fire fighting foams will have to be eliminated by June 2011. Incineration capacities should be checked.

At **100 ppm** additional wastes from the metal plating sector will be concerned. This comprises an overall amount of 1.62 kt/y consisting of vacuum distillation concentrate ($c\sim 240$ ppm; currently recycled/recovered), and activated carbon filter wastes ($c\sim 354$ ppm; hazardous waste incineration). No capacity problems should arise for these wastes.

At a LPCL of 50 ppm impregnated carpets ($\sim 1,938$ kt) would need to be classified POP waste because a contamination level of 75 ppm has been derived from available literature. This also applies for upholstery in the leather industry (~ 214 kt) with a contamination level of 80 ppm. It has to be mentioned that at the current state the majority of PFOS containing carpets are still in use, representing about 11.5 times of the annual amount of disposed of carpets. at present, i.e. still in use, and will enter the waste stream at the end of their life time, which will occur peu à peu., No capacity problems are expected in regard to the large waste amounts contributed by these two sectors. Currently this waste stream is treated by non hazardous waste incineration (~ 775 kt/y), non hazardous waste landfill ($\sim 1,163$ kt/y) and temporary storage ($\sim 21,705$ kt/y).

If the LPCL would be set at **10 ppm** additional significant amounts of waste from the photographic industry (~ 31 kt/y) consisting of x-ray wastes ($c\sim 13.5$ ppm, subject to non-hazardous waste landfill and non hazardous waste incineration) and desilvered-solution waste ($c\sim 20$ ppm destined for hazardous waste incineration).

A LPCL of **1 ppm** would lead to additional waste amounts arising from upholstery in the leather industry ($\sim 2,378$ kt) with a contamination level of 2.4 ppm. About twice of the amount of PFOS containing upholstery is temporary still in use (4,756 kt). _From the annual leather upholstery waste about 951 kt are treated by non-hazardous waste incineration and 1,162 kt are disposed of at non-hazardous waste incinerations. Further effected industries would be the metal plating sector (~ 100 kt) consisting of chromium sludges ($c\sim 4$ ppm, subject to recycling/recovery operations or 7ppm considering the activated carbon from the activated carbon treatment), from the photographic industry (~ 371 kt) consisting of film and picture wastes ($c\sim 4.5$ ppm, destined for non hazardous waste landfill and non hazardous waste incineration) and photo plate wastes ($c\sim 1$ ppm, recycling/recovery). In addition sewage sludges with "high" PFOS contamination (~ 579 kt, $c\sim 1$ ppm) would be affected. The amounts currently used for agricultural purposes ($\sim 7,377$ kt/y) and landfilled as non hazardous waste ($\sim 1,715$ kt/y) would need to be changed to incineration/co-incineration for current treatment as non hazardous waste landfill if classified as POP waste. This might cause capacity shortcomings due to the quantities involved.

Lowering the LPCL to **0.05 ppm** would cover all waste types contaminated with PFOS. This would additionally include sewage sludge wastes with low contamination (~11,000 kt, c=0.05 ppm). Latter one is responsible for a second drastic increase of the total waste amount affected by introduction of a LPCL of 0.05 ppm.

In conclusion, above a LPCL of 10 ppm no drastic effects will be observed since affected wastes are mostly destined for hazardous waste incineration and are likely to be allowed for extended use until appropriate alternatives are in place.

For a LPCL of 50 ppm carpet wastes might cause problems regarding capacity aspects, though it is assumed that due to their gradual entering of the waste stream at the end of their service life no capacity problems will occur.

Setting the LPCL lower than 10 ppm is not recommended since only little waste contributions from the industrial sector, which PFOS application allowance is likely to be extended, but an significant amount of sewage sludge wastes and leather upholstery would be affected, which might lead to capacity problems (see chapter 9.2.1.)

8.3.5 Scenario for PeCB

As explained above the impacts of the low POP content limit on the overall mass flow of PeCB are a function of the limit value, the contamination level (average and range) of relevant waste types and the volume of waste streams concerned. Thus it is necessary to compile and evaluate the waste streams investigated in the mass flow chapters.

For PeCB relatively low contamination values have been indicated in literature. These range from approximately 76.0 to 0.2 ppb. For PeCB the proposed LPCL of 50 ppm⁴⁹ and any other limit in the dimension of ppm does not have any effect on the PeCB waste streams.

Impacts on PeCB containing waste streams will only occur with a LPCL limit in the dimension of ppb. Therefore, for the PeCB scenario a ppb range has been applied as alternative LPCL limit.

Figure 8-9 gives an overview of the amounts of wastes concerned at different low POP concentration limits (LPCL) for PeCB. Setting a LPCL of 0.1 ppb would cover all wastes, which generally are ashes generated from different sectors such as domestic burning of solid fuels and waste incineration or sludges from waste water treatment.

⁴⁹ Draft COMMISSION REGULATION (EC) No .../.. of; amending Regulation (EC) No 850/2004 of the European Parliament and of the Council on persistent organic pollutants as regards Annexes IV and V (Text with EEA relevance); April 2010

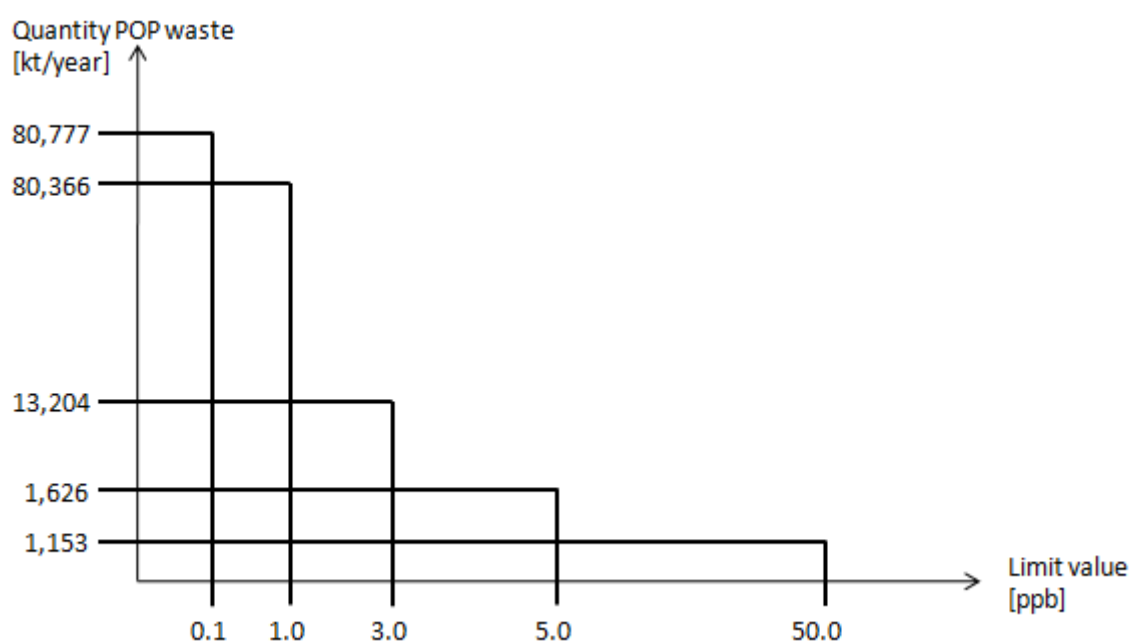


Figure 8-9: Estimated annual quantities in (kt/y) of waste classified POP waste due to its PeCB concentration (mg/g = ppb) in relation to different low POP content limits

A more detailed compilation of the relation between limit value and amounts of wastes exceeding them, which illustrates the contribution of individual waste types, is given in Figure 8-10 below:

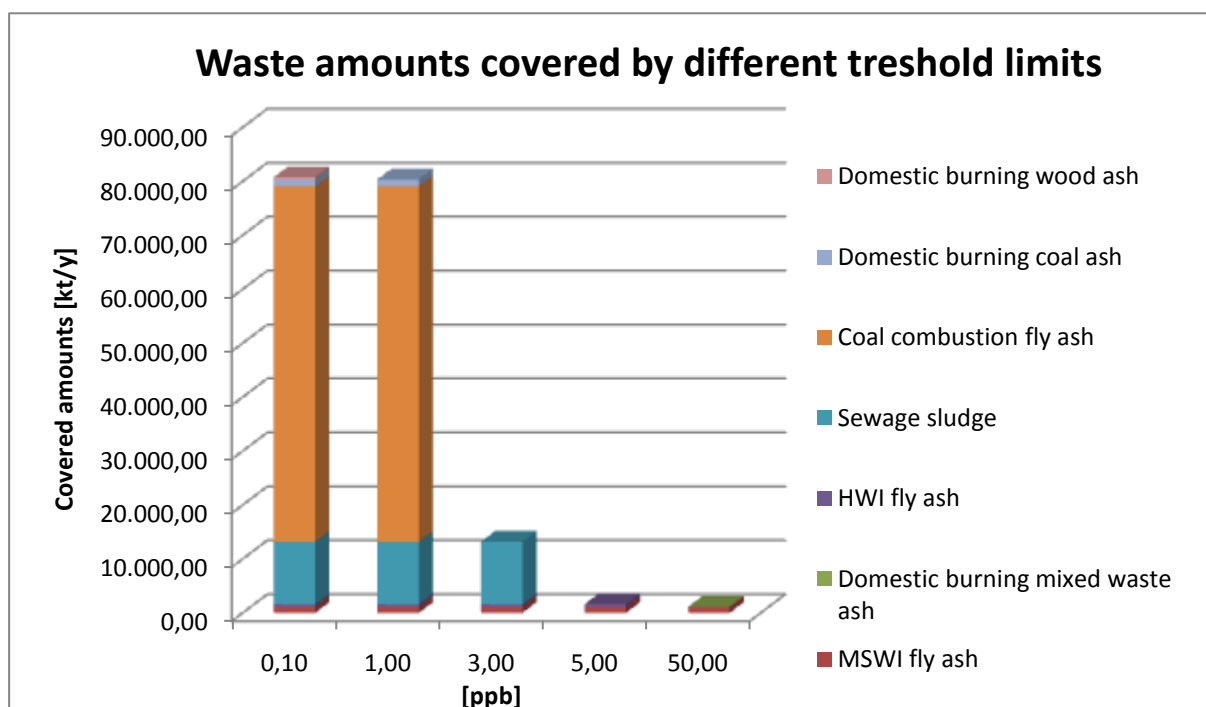


Figure 8-10: Contribution of specific wastes types to amounts exceeding different potential low POP content limits for PeCB in EU 27

As illustrated in Figure 8-10 the effect on wastes containing PeCB starts between 100 and 50 ppb. Within this range fly ashes generated during municipal solid waste incineration as well as ashes obtained from domestic burning of mixed household wastes are concerned. Below 50 ppb, to be precise between 5 and 10 ppb, fly ashes generated from hazardous waste incineration are affected. None of these wastes account for large quantities. In addition, due to hazardous properties from other sources, the majority of these wastes are generally disposed off as hazardous waste.

This situation changes for a LPCL of beneath 5 ppb, in which sewage sludges and (fly) ashes resulting from coal combustion processes in the industry play a significant role. Waste quantities resulting from these two sectors are responsible for about 95 % of wastes contaminated with PeCB.

Currently, sewage sludge is either landfilled (21%), incinerated (10), used for composting or agricultural purposes (45%) or treated under other options (24 %) in average calculated on the basis of EUROSTAT data. It has to be mentioned that treatment of sewage sludge differs strongly among Member States. However, considering the average ratios setting of a LPCL lower than 5 ppb would strongly affect sewage sludge treatment by prohibiting landfilling as non-hazardous waste. This would result in an increase of sewage sludge wastes subject to hazardous waste incineration and therefore might lead to a capacity problem for incineration plants (for assessment see chapter 9.2).

Another aspect to be considered at a LPCL of <5 ppb are the large quantities of (fly) ashes resulting from coal combustion in power production. Waste ashes from this sector account for approximately 80 % of wastes contaminated with PeCB, i.e. ~65 Mio t/y.

Ashes do not contain a high energetic value and therefore are not adequate for thermal treatment (D10, R1). The metal content can be expected to be too low for R4 treatment. Therefore the ashes could either be destined to D9 (physico-chemical treatment) or by way of derogation from article 7 would need to be either landfilled after stabilisation at hazardous waste landfill sites or transported to underground storage.

At present, the majority of ashes (88 %) generated are subject to recycling/recovery options as e.g. construction, underground mining as well as restoration of mines, quarries and pits as explained in chapter 6.4.11. Only 10 % are landfilled as hazardous (1 %) or non-hazardous waste (9 %) and a minority of 2 % is temporarily stored. Consequently, for ~55 Mio t/year the treatment would need to change upon introduction of a LPCL < 5 ppb. (For assessment whether this might lead to capacity problems see chapter 9.2).

Ashes resulting from domestic burning of wood and coal, have reported contamination values < 3 ppb. These waste would only be affected by a LPCL of 1 ppb. Due to the fact that they account for small quantities of PeCB contaminated wastes, they are not considered as relevant for the determination of a LPCL for PeCB.

To conclude, in consideration of amounts of wastes contaminated with PeCB, ashes from coal combustion as well as sewage sludge account for the largest waste quantities and for this reason might

lead to the most significant problems concerning capacity in case the lower concentration limit for PeCB should be set below 5 ppb.

8.3.6 Scenarios for SCCP

As explained above the impacts of the low POP content limit on the overall mass flow of SCCP are a function of the limit value, the contamination level (average and range) of relevant waste types and the volume of waste streams concerned. Thus it is necessary to compile and evaluate the waste streams investigated in the mass flow chapters.

For SCCP relatively high contamination values have been indicated in literature. These range from approximately 1 to 30,000 ppm.

Figure 8-9 gives an overview of the amounts of wastes concerned at different low POP concentration limits (LPCL) for SCCP. Setting a LPCL of 30,000 ppb would cover all wastes, which is mainly low contaminated sewage sludge.

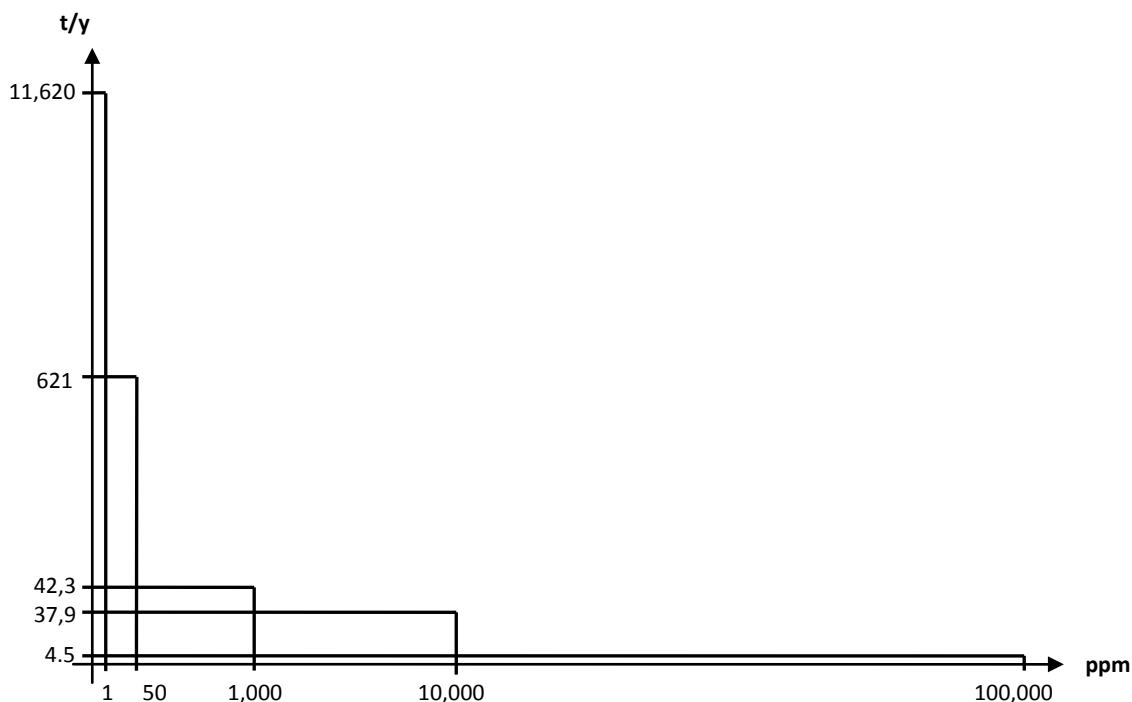


Figure 8-11: Estimated annual quantities in (kt/y) of waste classified POP waste due to its SCCP concentration (mg/g = ppb) in relation to different low POP content limits

Currently, sewage sludge is either landfilled (21%), incinerated (10), used for composting or agricultural purposes (45%) or treated under other options (24 %) in average calculated on the basis of EUROSTAT data. It has to be mentioned that treatment of sewage sludge differs strongly among Member States. However, considering the average ratios setting of a LPCL lower than 50 ppm would strongly affect sewage sludge treatment by prohibiting landfilling as non-hazardous waste. This would result in an increase of sewage sludge wastes subject to hazardous waste incineration and therefore might lead to a capacity problem for incineration plants (for assessment see chapter 9.2).

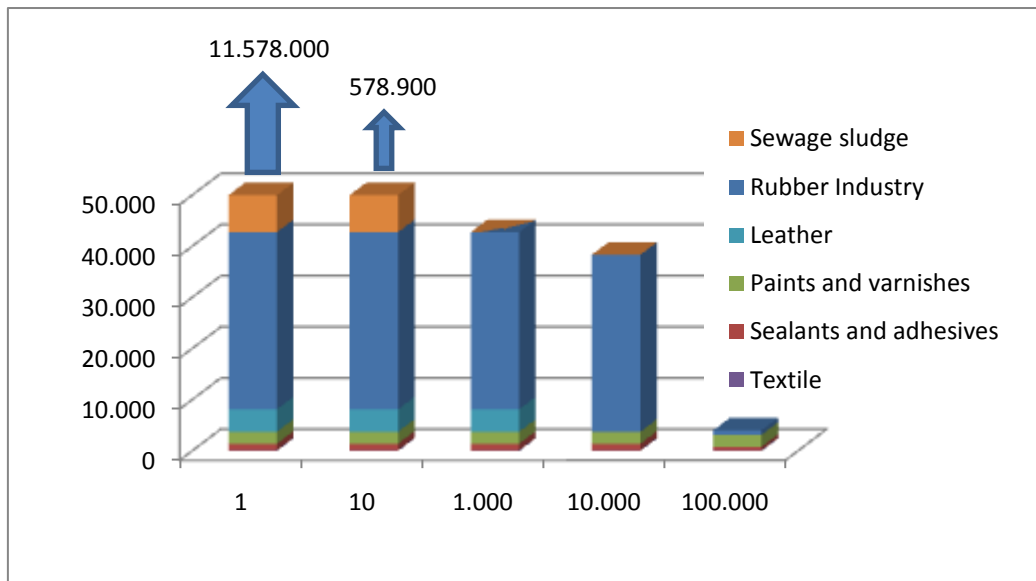


Figure 8-12: Contribution of specific wastes types to amounts exceeding different potential low POP content limits for SCCP in EU 27

As illustrated in Figure 8-10 the effect on wastes containing SCCP starts between 300,000 ppm and 33,000 ppm. Within this range are wastes from textiles, waste from rubber products others than conveyor belts, paints and varnishes as well as sealants and adhesives. None of these wastes account for large quantities.

This situation changes for a LPCL between 4,000 and 33,000 ppm. In this interval waste from the rubber industry is included which effects a much higher amount than all the other wastes types mentioned before. The next waste type is affected if the LCPL is set between 50 and 4,000 ppm which also includes wastes from leather products. This amount is relatively small compared to the waste arising from the rubber industry and it is expected that waste from the leather industry will vanish by 2012. At this LPCL the waste from the rubber industry has a share of about 80 %.

At a LPCL of 50 ppm high contaminated sewage sludge and at a LPCL of 1 ppm high contaminated sewage sludges are affected. This leads to an enormous increase of waste amount of annually ~ 580,000 and ~ 11,600,000 t respectively.

Currently, sewage sludge is either landfilled (21%), incinerated (10), used for composting or agricultural purposes (45%) or treated under other options (24 %) in average calculated on the basis of EUROSTAT data. It has to be mentioned that treatment of sewage sludge differs strongly among Member States. However, considering the average ratios setting of a LPCL lower than 5 ppb would strongly affect sewage sludge treatment by prohibiting landfilling as non-hazardous waste. This would result in an increase of sewage sludge wastes subject to hazardous waste incineration and therefore might lead to a capacity problem for incineration plants (for assessment see chapter 9.2).

At present, the majority of the waste except sewage sludges end up as non-hazardous waste and is treated as such with about 67.1 % being landfilled, 20.2 % incinerated without energy recovery and about 12.7 % being incinerated with energy recovery.

To conclude, in consideration of amounts of wastes contaminated with SCCP, rubber products as well as sewage sludge account for the largest waste quantities and for this reason might lead to the most significant problems concerning capacity in case the lower concentration limit for PeCB should be set below 1 ppm. Table 8-3 gives an overview of the affected waste amounts regarding to the LCPL.

Table 8-3: Overview of contribution of waste amounts contaminated with PFOS from specific sectors at different LPCL set

Sector ft. amount waste [kt/y] resulting from setting of a specific LPCL [ppm]	LPCL 1	LPCL 50	LPCL 4,000	LPCL 33,000	LPCL 300,000
Rubber Industry	33,606	33,606	33,606	33,606	844
Sealants and adhesives	1,270	1,270	1,270	1,270	677
Paints and varnishes	2,330	2,330	2,330	2,330	2,330
Textile	134	134	134	134	134
Leather	4,406	4,406	4,406	0	0
Sewage sludge	11,578,000	578,900	0	0	0
Total	11,619,745	620,645	41,745	37,339	3,986
Share in %	100%	5%	0%	0%	0%

8.3.7 Scenarios for HBCD

The impacts of the low POP concentration limit (LPCL) on the overall mass flow of HBCD are a function of the limit value, the contamination level (average and range) of relevant waste types and the volume of waste streams concerned. Thus, it is necessary to compile and evaluate the waste streams investigated in the mass flow chapters. An overview of amounts concerned at different LPCL is given in the Figure 8-13.

Based on the available data the following effects of different potential limits can be observed:

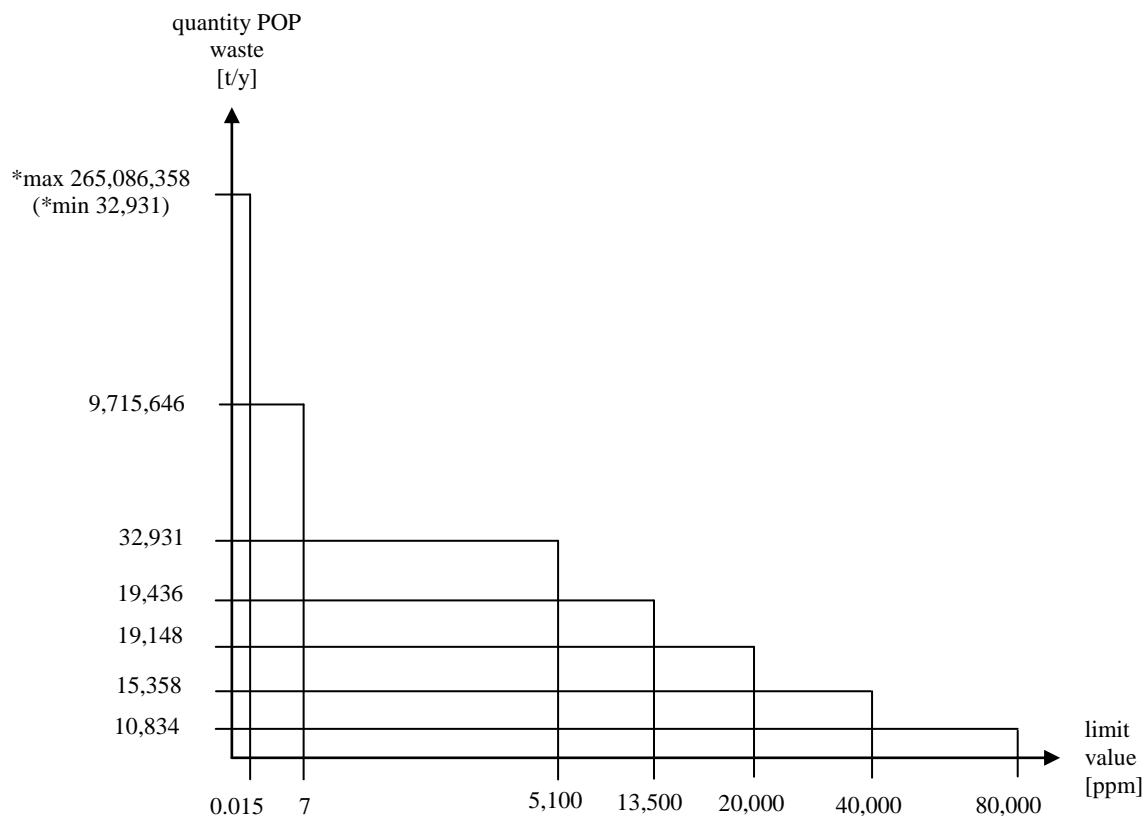


Figure 8-13: Estimated annual quantities of waste classified POP waste due to its HBCD concentration in relation to different low POP content limits (*min: EPS and XPS C&D waste and EPS/XPS other than C&D waste is separately treated; *max: EPS and XPS C&D waste and EPS/XPS other than C&D waste is treated together with non-contaminated waste fractions)

A more detailed compilation, showing the contribution of individual waste types, is provided in Figure 8-14 and the corresponding Table 8-4. In addition, Table 8-4 indicates the amount of waste which will require alternative treatment in case of being classified as POP waste.

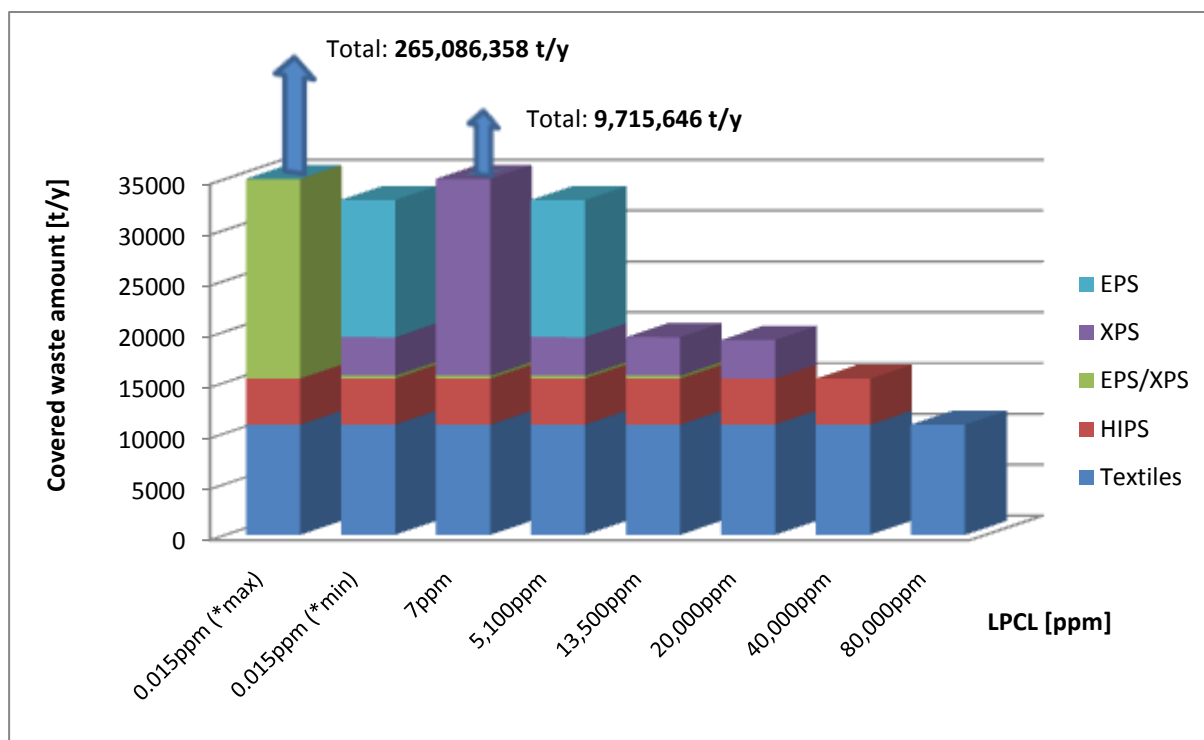


Figure 8-14: Contribution of specific wastes types to amounts exceeding different potential low POP content limits for HBCD in EU (*min: EPS and XPS C&D waste and EPS/XPS other than C&D waste is separately treated; *max: EPS and XPS C&D waste and EPS/XPS other than C&D waste is treated together with non-contaminated waste fractions)

LPCL [ppm]	Textiles	HIPS	EPS/XPS (other than C&D)	XPS (C&D)	EPS (C&D)	Total
Waste covered at different LPCL [t]						
80,000	10,834					10,834
40,000	10,834	4,524				15,358
20,000	10,834	4,524		3,790		19,148
13,500	10,834	4,524	288	3,790		19,436
5,100	10,834	4,524	288	3,790	13,495	32,931
7.0	10,834	4,524	288	9,700,000		9,715,646
0.015 (min*)	10,834	4,524	288	3,790	13,495	32,931
0.015 (max*)	10,834	4,524	255,371,000	9,700,000		265,086,358
Changes in waste treatment required [t]						
80,000	9,289					9,289
40,000	9,289	3,447				12,736
20,000	9,289	3,447		1,585		14,321
13,500	9,289	3,447	196	1,585		14,517
5,100	9,289	3,447	196	1,585	6,329	20,846
7.0	9,289	3,447	196	6,466,020		6,478,952
0.015 (min*)	9,289	3,447	196	1,585	6,329	20,846
0.015 (max*)	9,289	3,447	194,592,702	6,466,020		201,071,458

Table 8-4: Waste covered by different low POP concentration limits (*min: EPS and XPS C&D waste and EPS/XPS other than C&D waste is separately treated; *max: EPS and XPS C&D waste and EPS/XPS other than C&D waste is treated together with non-contaminated waste fractions)

Based on the available data the following effects of different potential limits can be observed:

The first impact on waste treatment can be observed at 80,000 ppm, and is related to ~10,834 t of HBCD contaminated textile waste which is currently incinerated (R1/D10) and to a large extent landfilled (~9,289 t). If classified as POP waste the ~9,289 t will have to be directed to incineration (D10) and/or incineration with energy recovery (R1). R4 (metal reclamation) and D9 (physico-chemical treatment) are other potential treatment methods, which however, do not seem to be primary options, due to the characteristics and the composition of the waste.

At a LPCL of 40,000 ppm a higher amount (~15,358 t) of contaminated waste would be affected. It comprises around 10,834 t of textiles waste and about 4,524 t of HIPS waste. If classified as POP waste approximately 12,736 t will have to be directed to incineration (D10) and/or incineration with energy recovery (R1). The 12,736 t encompass ~9,289 t of landfilled textile waste, about 2,203 t of landfilled HIPS waste and approximately 1,244 t of recycled/recovered HIPS waste. For the remaining 2,622 t incineration with and/or without energy recovery remains the preferred waste management option.

By setting the LPCL at 20,000 ppm around 19,148 t of contaminated waste from textiles, HIPS and XPS (scenario 1: XPS treated separately) will be covered. Of the total 19,148 t approximately 13,077 t are currently landfilled and about 1,244 t are recycled/recovered. This waste would have to be directed to incineration (D10) and/or incineration with energy recovery (R1). For the residual 4,827 t incineration with and/or without energy recovery remains the preferred waste treatment option.

At a LPCL of 13,500 ppm a slightly higher amount of waste (~19,436 t) would be affected, comprising additional 288 t of EPS/XPS waste other than C&D which is assumed to be collected and treated separately in the EU (scenario 1: EPS/XPS other than C&D waste collected and treated separately). If classified as POP waste approximately 14,517 t will have to be directed to incineration (D10) and/or incineration with energy recovery (R1). For the remaining 4,919 t incineration with and/or without energy recovery remains the preferred waste management option.

By setting the LPCL at 5,100 ppm around 32,931 t of HBCD contaminated waste will be covered, comprising additional 13,495 t of EPS C&D waste which is assumed to be collected and treated separately (scenario 1: EPS from C&D collected and treated separately). Of the total 32,931 t approximately 20,846 t are currently landfilled and recycled/recovered. This waste would have to be directed to incineration (D10) and/or incineration with energy recovery (R1). For the residual 12,085 t incineration with and/or without energy recovery remains the preferred waste treatment option.

By setting the LPCL at 7 ppm a significantly higher amount of HBCD contaminated waste will be covered (~9,715,646 t). It is assumed that both EPS C&D as well as XPS C&D are treated together with non-contaminated C&D plastic waste. Besides, it is assumed that EPS/XPS other than C&D is treated separately. In order to avoid double counting of waste streams it is important to notice that both waste streams (EPS and XPS C&D) contribute to the total C&D waste amount of approximately 9,700,000 t in the EU. Of the total 9,715,646 t around 6,478,952 t are currently landfilled and recycled/recovered. This waste would have to be directed to incineration (D10) and/or incineration with energy recovery (R1).

For the remaining 3,236,694 t incineration with and/or without energy recovery remains the preferred waste treatment option.

At a LPCL of 0.015 ppm at least 32,931 t of waste would be affected (*min: in case EPS and XPS C&D waste and EPS/XPS other than C&D waste is separately treated). In case EPS and XPS C&D is treated together with non-contaminated waste fractions and the EPS/XPS waste with non contaminated MSW this figure increases to a maximum of 265,086,358 t (*max). Besides, it is possible that the EPS and XPS C&D is separated, whereas the EPS/XPS other than C&D waste is treated together with non-contaminated MSW and vice versa (see LPCL 7 ppm). However, for reasons of simplification only the above indicated minimum (*min) and maximum (*max) figures will be elaborated (scenario 1/scenario 2 for all relevant waste streams). If classified as POP waste approximately 20, 846 t (*min: scenario 1 for all relevant waste streams) will have to be directed to incineration (D10) and/or incineration with energy recovery (R1). For the residual 12,085 t incineration with and/or without energy recovery remains the preferred waste management option. Alternatively, approximately 201,071,458 t (*max: scenario 2 for all relevant waste streams) will have to be directed to incineration (D10) and/or incineration with energy recovery (R1). For the remaining 64,014,900 t incineration with and/or without energy recovery remains the preferred waste management option.

8.3.8 Scenarios for PCP

As explained above the impacts of the low POP content limit on the overall mass flow of PCP are a function of the limit value, the contamination level (average and range) of relevant waste types and the volume of waste streams concerned. Thus it is necessary to compile and evaluate the waste streams investigated in the mass flow chapters.

Figure 8-21 gives an overview of the amounts of wastes concerned at different low POP concentration limits (LPCL) for SCCP. Setting a LPCL of 25,000 ppm would cover all wastes, which is mainly (~ 99 %) impregnated wood.

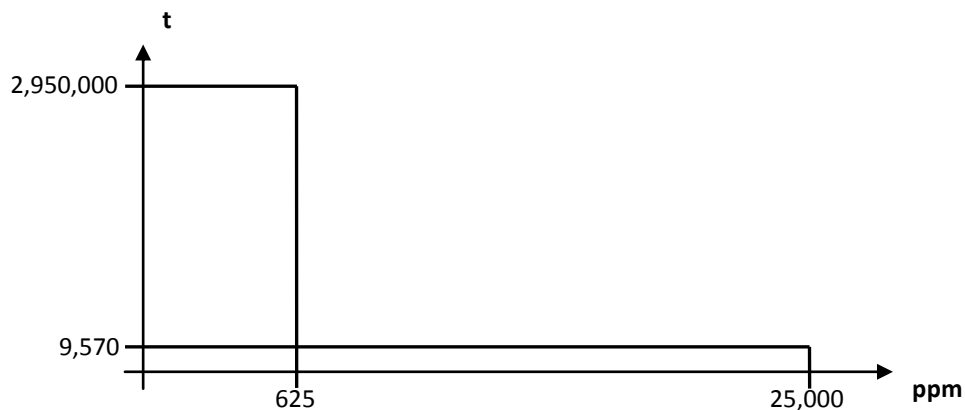


Figure 8-15: Contribution of specific wastes types to amounts exceeding different potential low POP content limits for PCP in EU 27

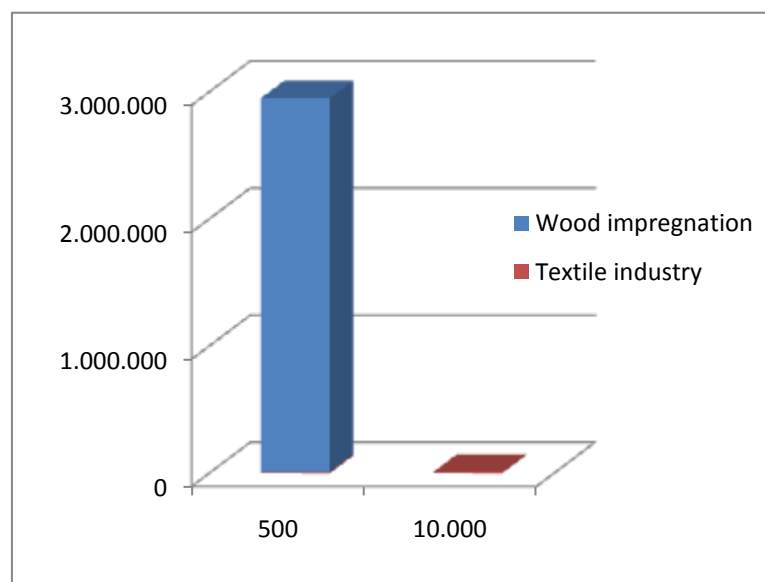


Figure 8-16: Contribution of specific wastes types to amounts exceeding different potential low POP content limits for PCP in EU 27

As illustrated in Figure 8-22 the effect on wastes containing PCP starts at 25,000 ppm. At that LPCL textile impregnated with PCP are affected. At a LPCL of about 625 ppm also impregnated woods have to

be considered. The waste stream from impregnated wood is much larger than from the textile industry and represents about 99 % of the total waste flow.

8.3.9 Scenarios for HCBd

The only data available regarding the concentration of HCBd in waste is data on sewage sludge contamination, with an average value of 0.0006 ppm, which would affect about 10 million tonnes of sewage sludge.

Due to the lack of information, i.e. in the sections chemical chlorine industry, plastic industry and hazardous waste incineration, it is recommended to choose a relatively high limit value in a first step. This limit value can be adjusted in case more information regarding typical waste concentrations is available, e.g. after inspection controls have taken place and/or further literature on contamination levels based on scientific analysis is available.

8.3.10 Scenarios for PCNs

As explained above the impacts of the low POP content limit on the overall mass flow of PCNs are a function of the limit value, the contamination level (average and range) of relevant waste types and the volume of waste streams concerned. Thus it is necessary to compile and evaluate the waste streams investigated in the mass flow chapters.

For PCNs relatively low contamination values have been indicated in literature with exceptions for slag from secondary copper production and fly ashes from hospital waste incineration. Generally, the contamination values range from 1.28 to 76 ppb. For slag from secondary copper production and fly ash from hospital waste incineration contamination values are much higher, amounting to 4.23 and 5.44 ppm, respectively. Impacts on PCN containing waste streams will therefore already occur with a LPCL limit set in the dimension of ppm (4 to 5 ppm), although the arising waste quantities are low (12 to 626 kt/y). Most of PCN contaminated waste will arise for a LPCL set in a dimension of ppb (1 to 50 ppb). Therefore, for the PCN scenario a ppb range has been applied as alternative LPCL limit.

Figure 8-17 gives an overview of the amounts of wastes concerned at different low POP concentration limits (LPCL) for PCN. Setting a LPCL of 1 ppb would cover all wastes, which generally are ashes generated from different sectors such as domestic burning of wood, solid fuels and household wastes, hospital waste incineration and municipal solid waste incineration, sludges from waste water treatment and slag and dust from secondary metal production of copper and aluminium.

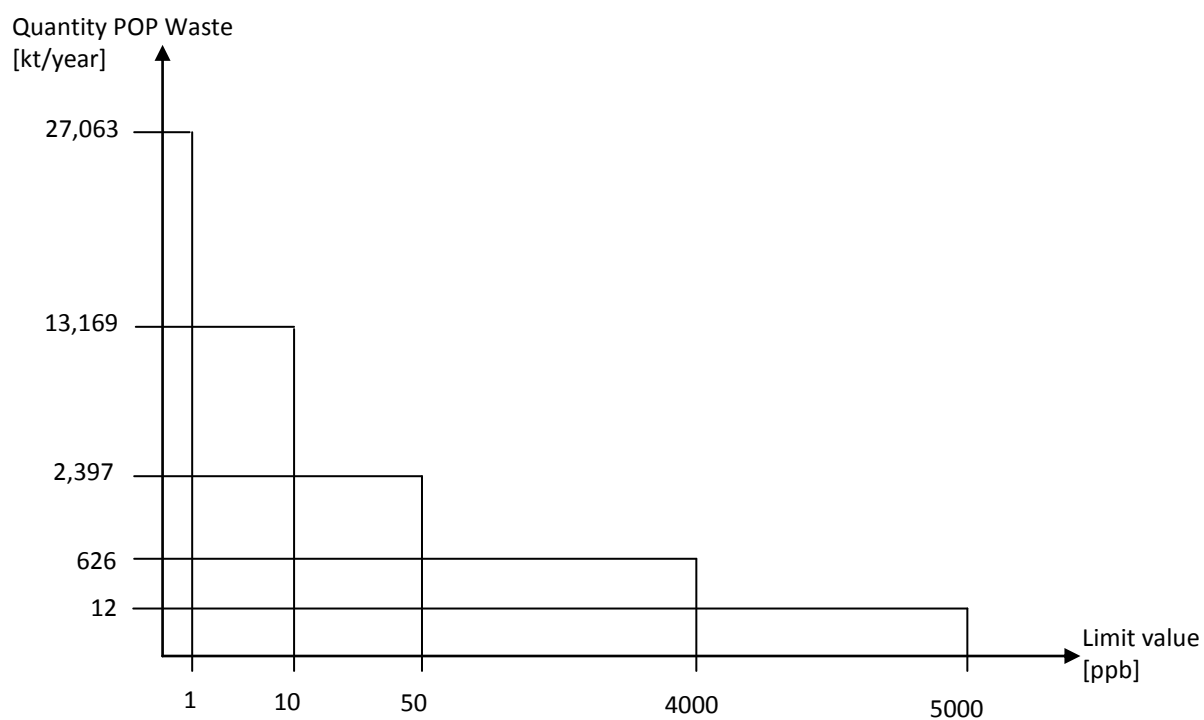


Figure 8-17: Estimated annual quantities in (kt/y) of waste classified POP waste due to its PCN concentration (mg/g = ppb) in relation to different low POP content limits

A more detailed compilation of the relation between limit value and amounts of wastes exceeding them, which illustrates the contribution of individual waste types, is given in Figure 8-18 below:

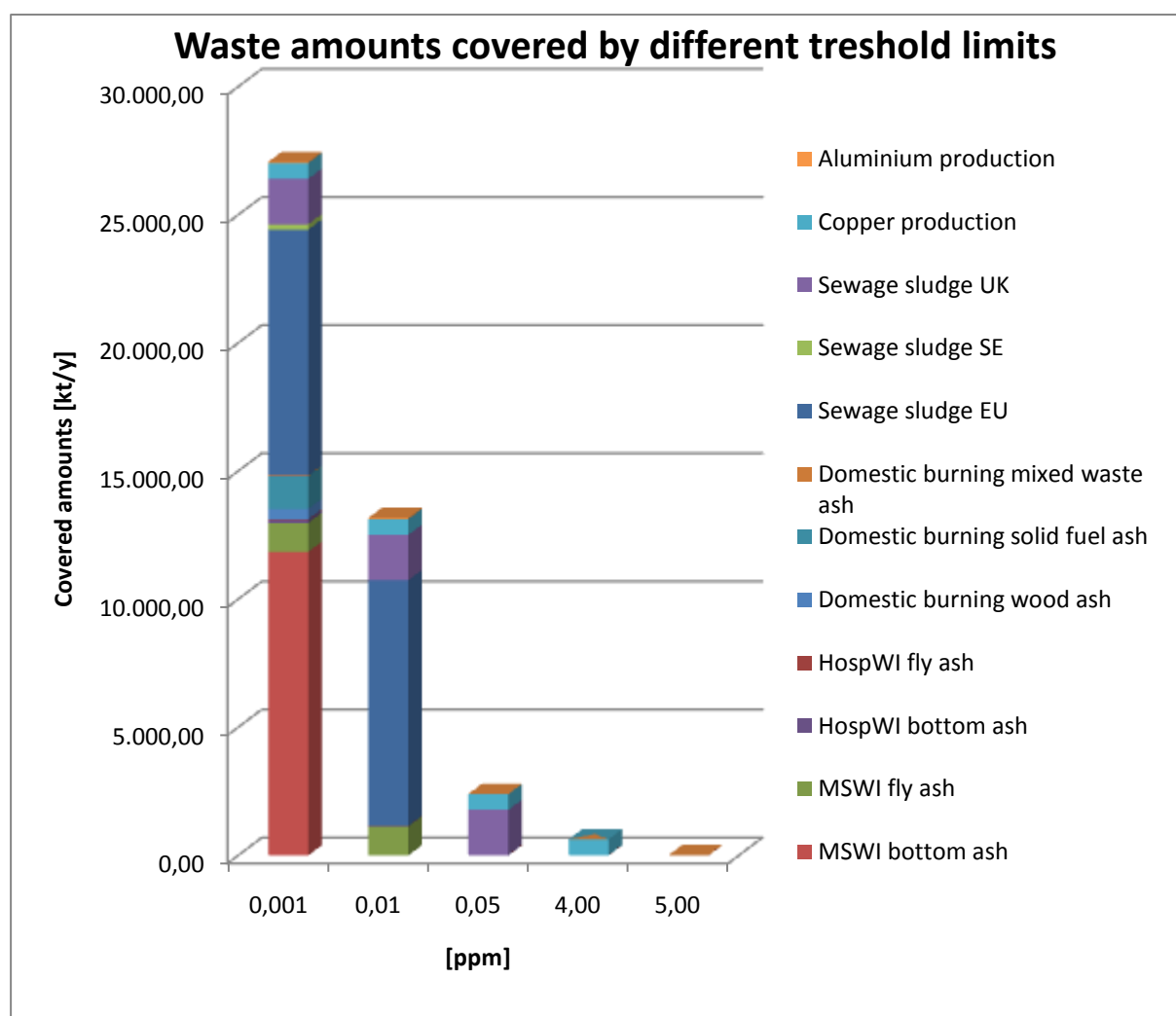


Figure 8-18: Contribution of specific wastes types to amounts exceeding different potential low POP content limits for PCNs in EU 27

As illustrated in Figure 8-18 the effect on wastes containing PCNs starts at a limit value of 5 ppm (5,000 ppb), where 12.22 kt fly ash wastes from hospital waste incineration are affected.

Between a range from 4 to 5 ppm, slag generated during secondary copper production is concerned, amounting to ~614 kt. These two waste types represent a relatively low waste amount and therefore, no big waste amounts would have to be additionally treated setting a limit value of 4 ppm (or even lower, down to a value of 0.1 ppm (100 ppb)).

Introducing a limit value of 0.05 ppm (50 ppb) further waste of 1,771 kt/y composed of sewage sludge from urban waste water treatment (specific for the UK) would arise, summing up to a total affected waste amount of approximately 2,400 kt/y.

Setting the LPCL even lower to 0.01 ppm (10 ppb), additional wastes as sewage sludge (from urban waste water treatment in the EU, except UK and SE), filter dust from secondary aluminium production and fly ashes from municipal solid waste incineration would have to be treated additionally, whereas

the amount waste to be treated in total would drastically increase from ~2,400 kt/y to about 13,150 kt/y. Sewage sludge (EU) would build the main bulk (~9,600 kt/y).

A limit value of 0.001 ppm (1 ppb) would cover all wastes contaminated with PCNs, additionally including bottom ash waste resulting from hospital waste incineration (~128 kt/y) and municipal solid waste incineration (~11,850 kt/y), ashes from domestic burning of wood, solid fuels and mixed wastes (1,711 kt/y) as well as sewage sludge from Sweden (210 kt/y), whereas bottom ash from municipal waste incineration would represent the dominating waste type leading to a drastical waste increase.

Currently, sewage sludge is either landfilled as non-hazardous waste (21%), incinerated as non-hazardous waste (10%), used for composting or agricultural purposes (45%) or treated under other options (24 %) in average calculated on the basis of EUROSTAT data. It has to be mentioned that treatment of sewage sludge differs strongly among Member States. In the UK 5% are landfilled, 16% incinerated, 70% recycled/recovered for agricultural purposes and/or composting, while 9% are treated in other ways. In Sweden, values amount to 11%, 0%, 50% and 39%, respectively.

However, considering the average ratios, setting of a LPCL at 10 ppb or lower would have the strongest effect on sewage sludge treatment by prohibiting landfilling as non-hazardous waste. This would result in an increase of sewage sludge wastes subject to hazardous waste incineration by ~1,956 kt/y and therefore might lead to a capacity problem for incineration plants (for assessment see chapter 9.2). Also sewage sludge wastes destined to recycling/recovery operations (~5,500 kt/y) would have to be treated differently.

Application of a LPCL of 1 ppb would additionally affect bottom ash waste generated during municipal solid waste incineration amounting in total to ~11,850 kt/y. Thereof, ~4,114 kt/y are dedicated to landfill as hazardous waste, whereas almost twice the amount (~7,731 kt/y) is used for recycling/recovery operations.

Ashes do not contain a high energetic value and therefore are not adequate for thermal treatment (D10, R1). The metal content can be expected to be too low for R4 treatment. Therefore, the ashes could either be destined to D9 (physico-chemical treatment) or by way of derogation from article 7 would need to be either landfilled after stabilisation at hazardous waste landfill sites or transported to underground storage.

At present, half of the amount ashes (53 %) generated are subject to recycling/recovery options as e.g. construction, underground mining as well as restoration of mines, quarries and pits as explained in chapter 6.4.11. About 41 % are landfilled as hazardous (36 %) or non-hazardous waste (5 %), an amount of ~5% is treated differently and a minority of ~2% from the domestic burning sector enters the MSWI stream and is incinerated as non-hazardous waste. For assessment whether this might lead to capacity problems see chapter 9.2.

Ashes resulting from domestic burning have reported contamination values < 10 ppb. These waste would only be affected by a LPCL of 1 ppb. Due to the fact that they account for small quantities of PCN

contaminated waste (~1,711 kt/y), they are not considered as relevant for the determination of a LPCL for PCNs.

To conclude, in consideration of amounts of wastes contaminated with PCNs, ashes from municipal waste incineration, especially bottom ashes, as well as sewage sludge account for the largest waste quantities and for this reason might lead to the most significant problems concerning capacity in case the lower concentration limit for PCNs should be set below 10 ppb.

8.4 Prognosis on future development

8.4.1 Prognosis C-PentaBDE Flow for 2015 and 2020 (status quo)

With the estimated yearly consumption of C-PentaBDE in automotive applications, the export data and the average life time of cars (~12 years) the substance flow of C-PentaBDE in automotive products is expected to phase out by-2016 (see Chapter 6). Similar estimations have been also performed for C-PentaBDE in upholstery applications. It can be summarized that with the assumed lifetime of 10 years all C-PentaBDE in upholstery applications will be disposed of by 2014.

The resulting mass flow of C-PentaBDE in the EU in 2015 is illustrated in the figure below.

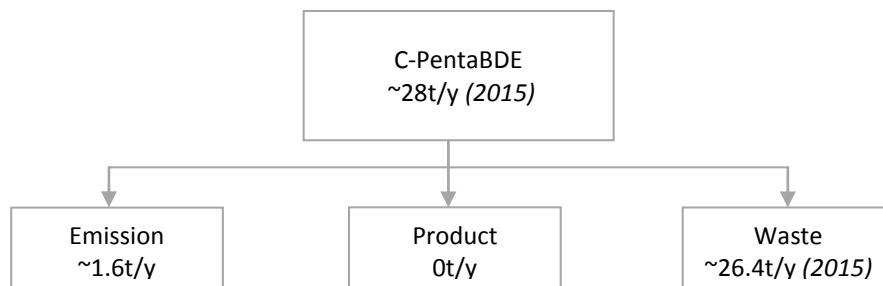


Figure 8-19: Prognosis mass flow of C-PentaBDE in the year 2015

The resulting waste flow is illustrated in Figure 8-19. As shown, the concentrations in the final waste fractions would have dropped by a factor of almost 10 compared to 2010 (~40 ppm instead of 371 ppm for SLF).

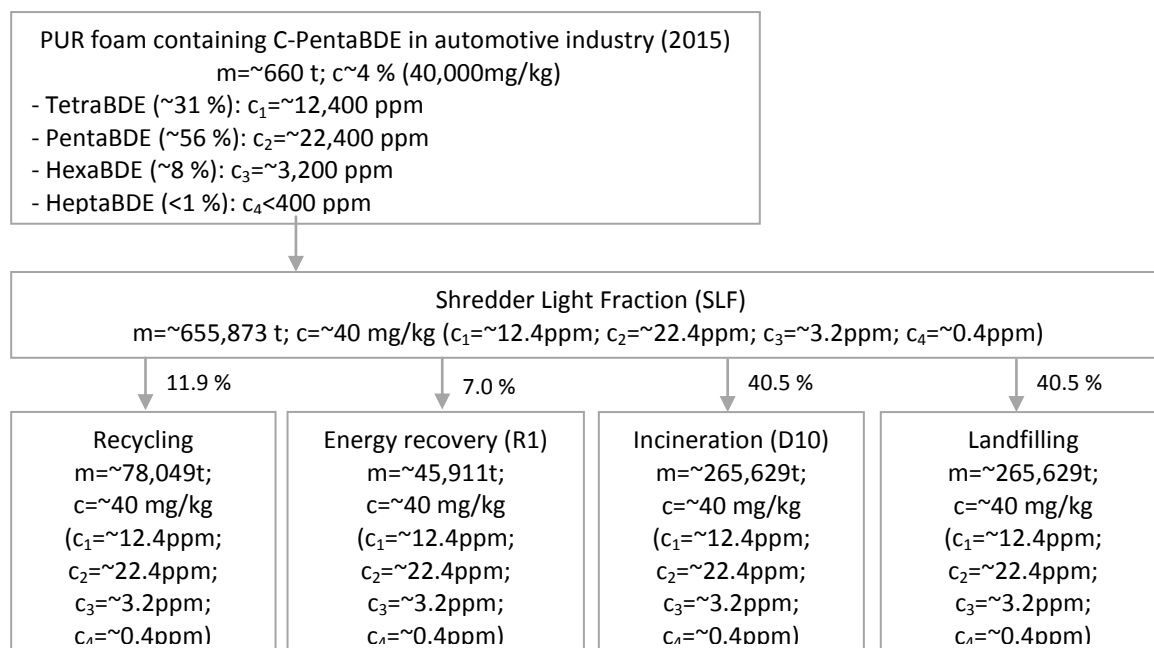


Figure 8-20: Prognosis waste flow of C-PentaBDE in 2015

In 2020 there is no expected mass flow of C-PentaBDE anymore under status quo conditions (no new C-PentaBDE input).

8.4.2 Prognosis C-OctaBDE Flow for 2015 and 2020 (status quo)

Due to the relatively short lifetime of products containing C-OctaBDE, (between 7 and 9 years) it has been estimated that all C-OctaBDE will be disposed of during the next two years (~2012).

8.4.3 Prognosis PBDE Flow for 2015 and 2020 (unintentional trace contamination and derogation as indicated in Annex I of the POP Regulation)

The prognosis above does not take into consideration that PBDE could continue to enter the product stream at unintentional trace contamination level of 10 ppm, nor does it take into account that articles partly or fully produced from recycled material could contain congeners of individual congener groups in concentrations up to 1,000 ppm.

If this derogation shall also apply for material produced within the EU it is interpreted that also shredder output as basis material would be allowed for the recycling up to concentrations of 1,000 ppm⁵⁰.

The remaining stocks of PBDEs in PUR foam and WEEE plastics is roughly 470 t/y for commercial products, corresponding to TetraBDE 103.73 t/y, PentaBDE 195.38 t/y, HexaBDE 34.64 and HeptaBDE 48.65 t/y.

Given that the derogation in Annex I applies for shredder fractions intended for recycling the complete remaining stocks of PBDEs in PUR foam and WEEE plastics (roughly 470 t/y for commercial products corresponding to TetraBDE 103.73 t/y, PentaBDE 195.38 t/y, HexaBDE 34.64 t/y, HeptaBDE 48.65 t/y) could in theory be kept in the recycling cycle, as all concentrations for single congener groups in shredder fractions (if mixed with non contaminated items as currently practiced) are well below 1,000 ppm.

The unintentional trace contamination level of 10 ppm (continuous input into the mass flow) on the other hand has to be compared to current concentration levels in the original products of around 40,000 ppm. The permissible contamination in new products hence appears to be negligible compared to the historic load, and will lead to maximum continuous substance flows in a dimension of TetraBDE 25 kg/y, PentaBDE 50 kg/y, HexaBDE 10 kg/y and HeptaBDE 12 kg/y.

⁵⁰ If this would be desired it would be necessary to add recycling of the relevant wastes to the permitted disposal and recovery operations specified in Annex V, Part 1 of the POP regulation.

8.4.4 Prognosis for PFOS and its derivatives

For the ten year prognosis of the PFOS mass flow up to the year 2020 it is expected that several changes will occur.

The ~84 t of PFOS in fire fighting foams will have to be eliminated by means of hazardous waste incineration by June 2011. The stock of ~11.4 t of PFOS in leather upholstery will be eliminated by 2012 (~4,756,000 kt waste) and the PFOS stocks in carpets of ~1,600 kt will be eliminated by 2018 at the latest (~200 kt/y).

The use of PFOS in hydraulic fluids in the aviation industry (~730 kg) is expected to remain more or less stable (exemption of ban), whereas the amount used in metal plating is expected to further decline to ~2 t. The same is expected for the photographic industry where a complete elimination of stocks is expected by around 2016. The PFOS contamination in sewage sludge should slowly reduce to lower input from primary source.

Based on the expected reduction of PFOS used in the industrial sector due to substitution with alternatives, the mandatory elimination of PFOS contained in fire fighting foams, the elimination of treated carpets and upholstery, and the assumption that PFOS concentration in sewage sludges will slowly decrease, a reduced mass flow as illustrated in Figure 8-21 is expected for 2020.

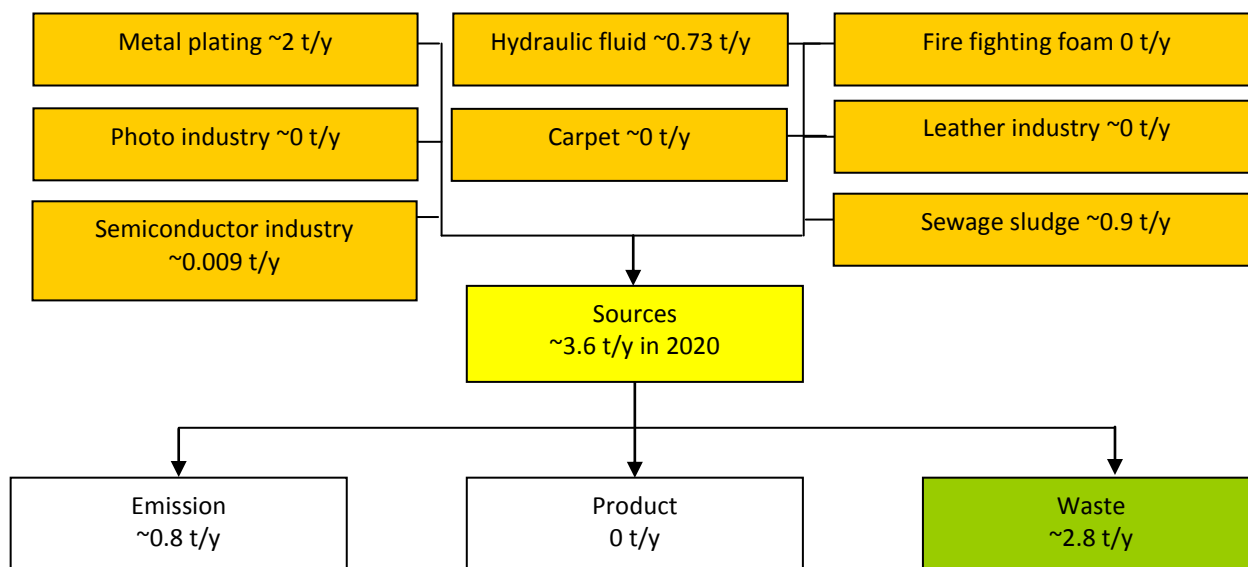


Figure 8-21: Prognosis for the PFOS mass flow in the year 2020

The waste prognosted mass flow does not take into consideration unintentional trace contamination levels of PFOS. If comparing such a unintentional trace contamination (0.1 g/m²) in e.g. carpet and leather with previously used amounts of roughly 2.4 g/m² for leather and 370 mg/m² for textiles, a continued trace contamination threshold would result in maximum annual discharge via waste of 5g/y via leather upholstery and 1 kg/y via impregnated textiles, which seems to be negligible.

8.4.5 Prognosis for PeCB

For the ten year prognosis of the PeCB mass flow in the year 2020 it is expected that relatively small changes will occur. PeCB is produced unintentionally due to incineration and combustion processes in the industry as well as in the domestic sector. The amounts of PeCB produced are relatively low and also the contamination concentration of affected wastes with PeCB is according to several literature sources located in a rather low area, i.e. ppb, compared to the other new POPs, which exhibit rather high contamination concentrations in the ppm range.

However, it is not excluded that improvements in technical standards might lead to further decrease in PeCB emissions to the environment and discharges as waste. Especially PeCB amounts released to the environment account for a relatively high contribution of PeCB emissions, compared to PeCB discharges to waste. The industrial coal combustion sector has been identified to play a major role concerning the PeCB emissions to air (83 %), which is also in accordance with data reported to the European PRTR from the steel and iron production sector based on the assumption that this sector accounts for over 50 % of the final coal consumption according to EUROSTAT data. Improvement of technical standards in this sector as well as in the other investigated sectors might lead to further PeCB decreases.

Air emissions as well as emissions to soil resulting from domestic burning of coal, wood and mixed household wastes is assumed to stay rather constant or even increase than to decline since it is estimated that especially burning of wood and other solid fuels as coal are a lucrative alternative to electricity.

This prognosis does not yet take into account an expected shift from landfill of municipal solid waste to an increased share of incineration. Also impacts of future legislation have not been taken into consideration.

With respect to technical improvements in the industrial sector a reduced mass flow with PeCB quantities from sources reduced by maximum 40 % and resulting in similar reduction for air emissions as illustrated in Figure 8-22 could be expected for 2010.

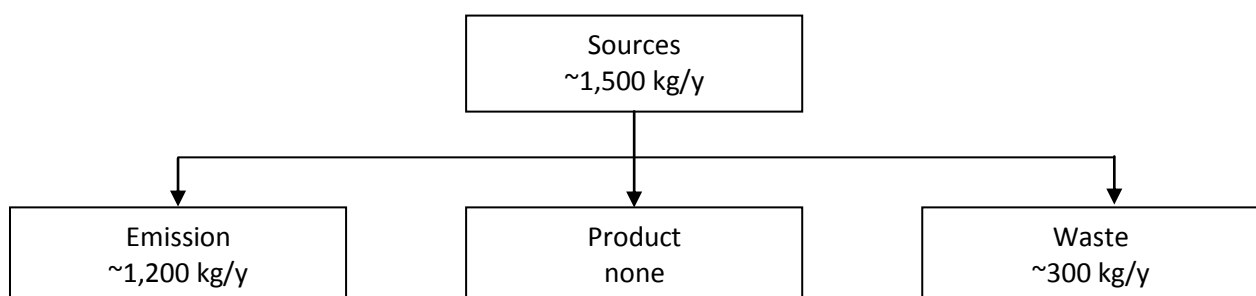


Figure 8-22: Prognosis for the PeCB mass flow in the year 2020

However, a reduction by 40 % is only a maximum estimation. It is more likely that reduction will rather account to approximately 10 to 20 %.

8.4.6 Prognosis for SCCP

For the ten year prognosis of the SCCP mass flow up to the year 2020 it is expected that several changes will occur.

Waste from the rubber industry consists of production waste, waste streams from gaskets and hoses (which are both considered to change stay constant for the next 10 years) as well as waste resulting from the disposal of used conveyor belts. The waste from the end of life conveyor belts represents about 97.5 % of the current total waste stream from the rubber industry and is expected to reduce significantly due to the drop of use of SCCP since the mid ninetieth. In 2010 about 33,800 t of old conveyor belts reach their end of life time. By 2015 this will reduce to about 21,000 t and by 2020 it should be down to about 13,400 t. The corresponding amounts of involved SCCP are 1,126 t for 2010, 690 t for 2015 and 447 t for 2020.

In case of sealants and adhesive it is considered that the estimated current annual waste amount of about 2,000 t will reduce in the following 10 years. Regarding the SCCP inputs in the last years it is estimated that in a best case the waste amount will drop by 50 %. Following a similar approach for the paint and varnishing industry and considering the drop of used amounts of SCCP the last years, the waste amount might drop in a best case from 290 t/y in 2010 by 75% in 2020 down to 66 t/y.

Considering the data of the used amounts of SCCP in the textile industry in the last 15 years it has to be expected, that no significant change will happen in the next 10 years and that therefore the annual disposal amount of 116 t will be more or less constant. For the leather industry on the other hand it is expected that the waste stream from this sector will vanish within the next 2 years, considering an average life time of 6 to 12 year depending on the leather product.

All the other sources, including sewage sludge and the use of SCCP are considered to stay constant. In case of the MCCC waste stream only the SCCP amount and not the waste amount can be given.

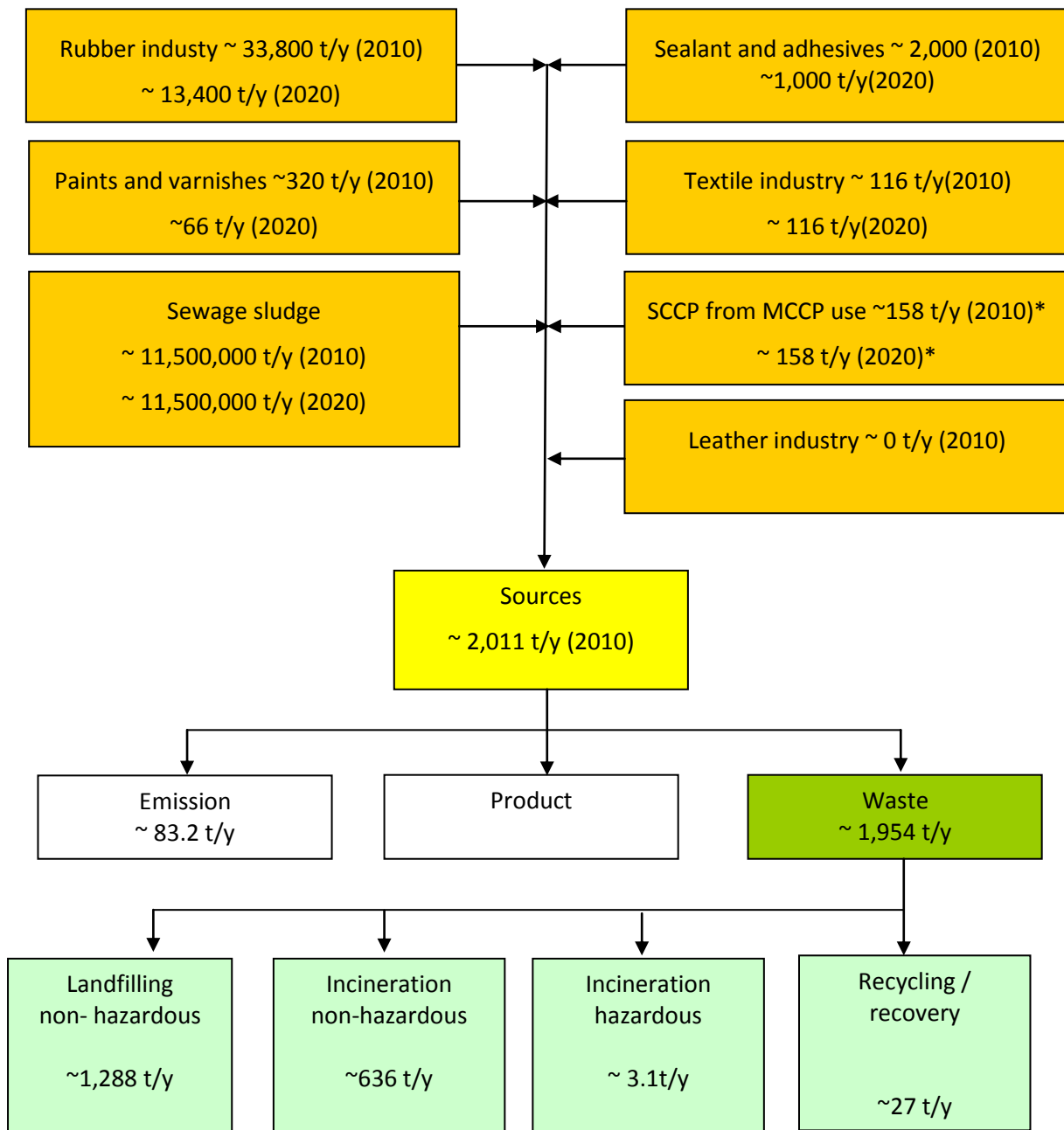


Figure 8-23: Overall mass flow from SCCP from sources to current disposal/recovery operations in the EU

* the value only represent the total SCCP amount not the waste amount

8.4.7 Prognosis for HBCD

As already discussed in Chapter 5, the discharge situation of HBCD is expected to dramatically change in the future. The main share of HBCD is at the present incorporated into new products which will become waste in the future. Besides, it has been estimated that in long terms the C&D sector will be the only relevant sector, contributing to the entire HBCD discharge in the EU (see future estimations in Chapter 5).

Figure 8-24 summarizes the main sources of HBCD and their estimated contribution to the HBCD substance flow in 2015. These are the EPS C&D waste (~88.0 t/y), the XPS C&D waste (~97.0 t/y), EPS/XPS other than C&D waste (~13.6 t/y), HIPS waste (~210.0 t/y) and HBCD from textile waste (~549.1 t/y). Besides, Figure 8-24 indicates that approximately 4,874 t/y HBCD are expected to be consumed for the manufacture of new HBCD flame retarded products in 2015.

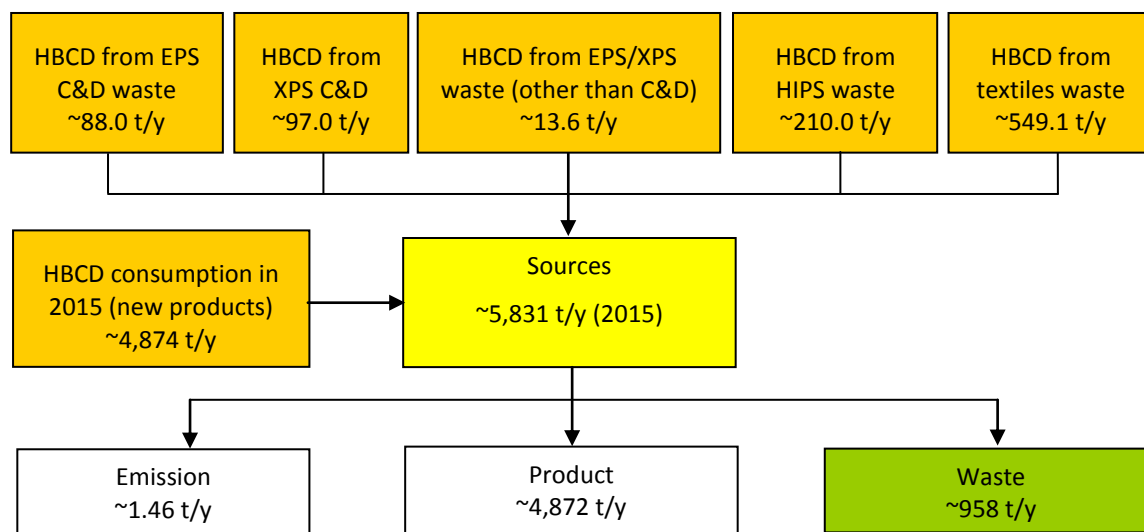


Figure 8-24: Prognosis mass flow of HBCD in 2015

In 2015, around 958 t/y of HBCD are expected to be disposed of (see Figure 8-24). This figure is ~238 t/y lower than the current HBCD discharge figure of ~1,196 t/y. The most significant change, however, is that approximately 4,874 t/y HBCD will be consumed for the manufacture of new HBCD flame retarded products, which is considerably lower when compared to the current HBCD consumption figure of ~10,431 t/y. The total emissions are expected to decline from approximately 2.84 t/y to 1.46 t/y as indicated in Figure 8-24.

Figure 8-25 summarizes the main sources of HBCD and their estimated contribution to the overall HBCD substance flow in 2020. These are the EPS C&D waste (~158 t/y), the XPS C&D waste (~174 t/y), EPS/XPS other than C&D waste (~23.3 t/y), HIPS waste (~199.7 t/y) and HBCD from textile waste (~215.3 t/y).

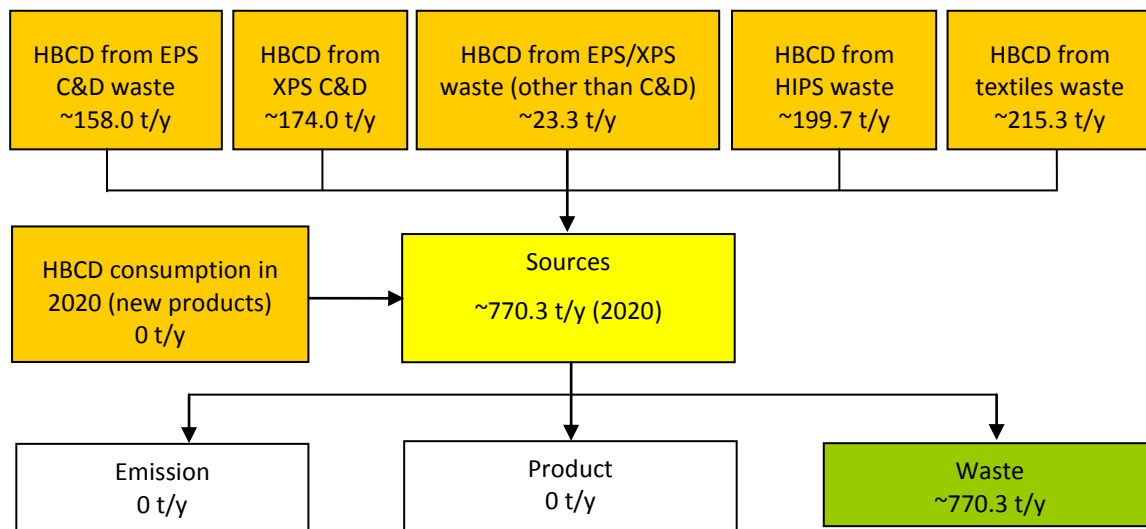


Figure 8-25: Prognosis mass flow of HBCD in 2020

Compared to the mass flow of HBCD in 2015, it can be observed that a slightly lower amount of HBCD is expected to be disposed of in 2020, whereas it is assumed that no HBCD will be used for the manufacture of new flame retarded products (see Figure 8-25).

However, the discharge figures are expected to increase, especially when considering that significant amounts of HBCD have been incorporated into new EPS/XPS products for construction purposes and the relatively high life times of those products. Estimations presented in Chapter 5 show that the discharge figures will increase to ~1,335 t/y in 2030 and 3,185 in 2040. In 2030, the most relevant contribution to the HBCD discharge to waste will be from the C&D sector (~98 %; EPS and XPS). In addition a minor contribution to the total discharge is expected from EPS/XPS products outside the C&D sector (~2 %). From ~2040, the C&D sector is expected to be the only relevant sector contributing to the total HBCD discharge (~48 % from EPS and ~52 % from XPS C&D). The highest annual discharge of HBCD is expected around 2050 (2054), with more than 5,100 t/y of HBCD in EPS and XPS C&D waste.

The resulting waste flows for 2015, 2020, 2030 and 2040 are summarized in Table 8-5.

Sector	Waste	Amount [t/y]	Contamination [%]			HBCD discharge [t/y]	HBCD to waste/ products [t/y]	Share of HBCD to waste/products [%]
			min	ea*	max			
HBCD waste flow in 2010								
C&D	EPS	13,495	0.7	0.51	1.0	69.5	68.82/4,471.4	6/43
	XPS	3,790	1.0	2.0	3.0	76.5	75.8/4,922.3	6/47
Other than C&D (e.g. packaging)	EPS/XPS	288	0.7	1.35	3.0	4.0	3.9/626	<1/6
Electronic	HIPS	4,524	1.0	4.0	7.0	>181	181/208.8	15/2
Various (e.g. automotive)	Textiles	10,834	7.0	8.0	9.0	868	866/207.4	73/2
TOTAL		~32,931				~1,199	~1,196/~10,435	100/100
Prognosis HBCD waste flow for 2015								
C&D	EPS	17,255	0.7	0.51	1.0	>88.0	88/2,213	9/45.4
	XPS	4,850	1.0	2.0	3.0	>97.0	97/2,436	10/50

Sector	Waste	Amount [t/y]	Contamination [%]			HBCD discharge [t/y]	HBCD to waste/products [t/y]	Share of HBCD to waste/products [%]
			min	ea*	max			
Other than C&D (e.g. packaging)	EPS/XPS	1,007	0.7	1.35	3.0	>13.6	13.6/14.9	2/0.3
Electronic	HIPS	5,250	1.0	4.0	7.0	>210.0	210.0/104.4	22/2
Various (e.g. automotive)	Textiles	6,864	7.0	8.0	9.0	>549.1	549.1/103.7	57/2
TOTAL		~35,226				~957.7	~957.7/4,872	100/100
Prognosis HBCD waste flow for 2020								
C&D	EPS	30,980	0.7	0.51	1.0	~158	158/0	20.5/0
	XPS	8,700	1.0	2.0	3.0	~174	174/0	22.5/0
Other than C&D (e.g. packaging)	EPS/XPS	1,726	0.7	1.35	3.0	~23.3	23.3/0	3/0
Electronic	HIPS	4,993	1.0	4.0	7.0	~199.7	199.7/0	26/0
Various (e.g. automotive)	Textiles	2,691	7.0	8.0	9.0	~215.3	215.3/0	28/0
TOTAL		49,090				~770.3	~770.3/0	100/0
Prognosis HBCD waste flow for 2030								
C&D	EPS	121,765	0.7	0.51	1.0	~621	621/0	47/0
	XPS	34,200	1.0	2.0	3.0	~684	684/0	51/0
Other than C&D (e.g. packaging)	EPS/XPS	2,207	0.7	1.35	3.0	~29.8	29.8/0	2/0
Electronic	HIPS	0	1.0	4.0	7.0	0	0/0	0/0
Various (e.g. automotive)	Textiles	0	7.0	8.0	9.0	0	0/0	0/0
TOTAL		~158,172				>1,335	~1,335/0	100/0
Prognosis HBCD waste flow for 2040								
C&D	EPS	297,254	0.7	0.51	1.0	~1,516	1,516/0	48/0
	XPS	83,450	1.0	2.0	3.0	~1,669	1,669/0	52/0
Other than C&D (e.g. packaging)	EPS/XPS	0	0.7	1.35	3.0	0	0/0	0/0
Electronic	HIPS	0	1.0	4.0	7.0	0	0/0	0/0
Various (e.g. automotive)	Textiles	0	7.0	8.0	9.0	0	0/0	0/0
TOTAL		~380,704				~3,185	3,185/0	100/0

Table 8-5: Prognosis HBCD waste flows for 2015, 2020, 2030 and 2040

As shown in Table 8-5 the HBCD waste flow is expected to significantly increase from ~32,931 tonnes in 2010 to more than 380,000 tonnes in 2040, reaching its maximum around 2050. Currently, the HBCD contaminated waste is directed to different waste management operations including disposal and recovery operations. Due to the high life time of HBCD treated products it is not possible to predict the relevant waste management operations in 2040 or even 2050. Possibly, innovative recycling methods (e.g. CreaSolv) will evolve, allowing a separation of HBCD from EPS/XPS with subsequent recycling of this waste.

8.4.8 Prognosis for PCP

For the ten year prognosis of the PCP mass flow up to the year 2020 a significant decrease of PCP contaminated waste is expected.

In case of wood impregnation which is with > 99 % of the major source of PCP containing waste a decline of about 90 % is expected in the following 10 years, due to a strong reduction of PCP for wood impregnation in the mid eighties and the beginning of the ninetieth of the last century. It was calculated the waste amount of about 2,940,000 t/y in 2010 will drop to 194,000 t/y in 2020. A further drop is calculated to take place in 2033 down to zero.

In case of the textile industry the stop of use of PCP by 2001 leads to a continuous decline of disposed of PCP impregnates textiles. It was calculated the current amount of about 9,600 t/y in 2010 will decline to 1,130 t/y in 2015 and further down to 204 t/y in 2020. By 2022 it is expected that the waste amount containing PCP from the textile industry will be negligible.

8.4.9 Prognosis for HCB

No ten year prognosis could be elaborated for the HCB mass flow due to the lack of information. Further investigations and assessments have to be carried out in order to enable future prognosis estimations.

8.4.10 Prognosis for PCN

For the ten year prognosis of the PCN mass flow in the year 2020 it is expected that relatively small changes will occur. PCNs are produced unintentionally mainly due to waste incineration, metal manufacturing processes in the industry as well as in the domestic sector. The amounts of PCNs produced are dominated by secondary copper production, but are relatively low for the other sectors. Also the contamination concentration of affected wastes with PCNs is according to several literature sources located in a rather low area, i.e. ppb, compared to the other new POPs, which exhibit rather high contamination concentrations in the ppm range.

PCN amounts released to the environment are very low and are considered not to change within the next ten years. However, it is not excluded that improvements in technical standards might lead to decrease in PCN discharges as waste. The secondary copper production sector has been identified to play a major role concerning the PCN emissions to waste (~80%), since copper is considered to act as catalyst for the formation of specific substances, including PCNs. Improvement of technical standards in this sector might lead to further PCN decreases.

Air emissions as well as emissions to soil resulting from domestic burning of coal, wood and mixed household wastes is assumed to stay rather constant.

This prognosis does not yet take into account an expected shift from landfill of municipal solid waste to an increased share of incineration. Also impacts of future legislation have not been taken into consideration.

With respect to technical improvements in the industrial sector of secondary copper production a reduced mass flow with PCN quantities from this source reduced by maximum 50 % as illustrated in Figure 8-22 could be expected for 2020, leading to an overall reduction of ~40%.

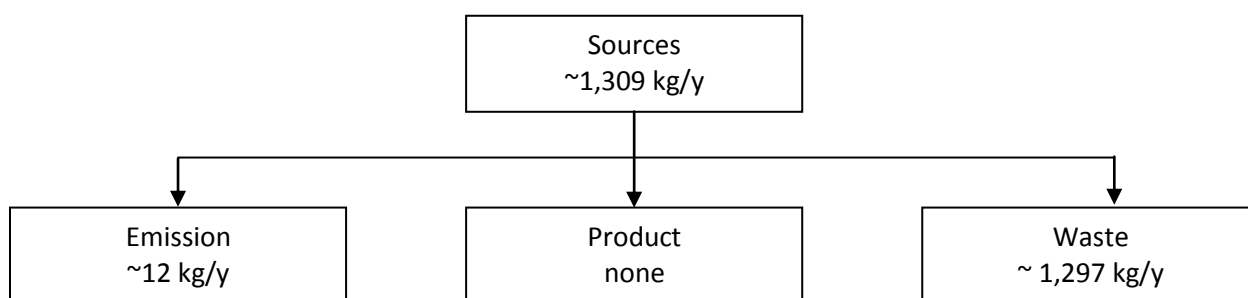


Figure 8-26: Prognosis for the PCN mass flow in the year 2020

However, a reduction by 40 % in total is only a maximum estimation. It is more likely that reduction will be much lower.

9 Low POP concentration limit and maximum POP concentration limit for POP waste

9.1 Methodology to assess and suggest low POP concentration limit and maximum POP concentration limits

9.1.1 Overview and objective of the methodology

For the requirements laid down in the annexes to the European POP regulation (EC) No 850/2004, two limit values are of major importance:

The **low POP concentration limit (LPCL)**⁵¹ serves to classify whether a waste is a POP waste or not. It therefore defines the "entry into the POP waste regime" of the Regulation. The low POP concentration limit can be set at different limit values for different POPs.

As stipulated in Annex V part 1 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 (recycling/reclamation of metals and metal compounds)

However, permanent storage (D12) in safe, deep hard rock formations, salt mines or landfill sites for hazardous waste may be permitted according to Article 7 (4)(b) by national authorities in exceptional cases for wastes listed in part 2 of Annex V.⁵²

This exemption is possible under two conditions. First, it has to be demonstrated that the selected operation is environmentally preferable. Second, the relevant waste must not exceed a certain maximum POP concentration limit (MPCL) to be defined in Annex V. These limits apply exclusively to a landfill site for hazardous waste and do not apply to permanent underground storage facilities for hazardous wastes, including salt mines.

⁵¹ Concentration limit referred to in Article 7(4)(a) of Regulation (EC) No 850/2004 and specified in its Annex IV

⁵² From power production, aluminium, lead, copper, zinc and other metallurgical processes, C&D and thermal waste treatment: 100114*, 100116*, 100207*, 100304*, 100308*, 100309*, 100319*, 100321*, 100329*, 100401*, 100402*, 100404*, 100405*, 100406*, 100503*, 100505*, 100603*, 100606*, 100808*, 100815*, 100909*, 161101*, 161103*, 170106*, 170503*, 170902*, 170903*, 190107*, 190111*, 190113*, 190115*, 190402*, 190403*

If POP concentrations in EWCs listed in Annex V part 2 exceed the **(MPCL)**⁵³, destruction or irreversible transformation of the POP content by means of D9 (physico-chemical treatment), D10 (incineration on land), R1 (use as a fuel) or R4 (recycling/reclamation of metals and metal compounds) operations will be obligatory, as for all other wastes exceeding the LPCL set in Annex IV to the POP regulation.

Figure 9-1 illustrates which limit values have to be proposed for the new POPs and candidate POPs in the course of the present study, how they are related to Annex IV and V of Regulation (EC) No 850/2004 and the methodological approach for establishing proposals for LPCLs and MPCLs.

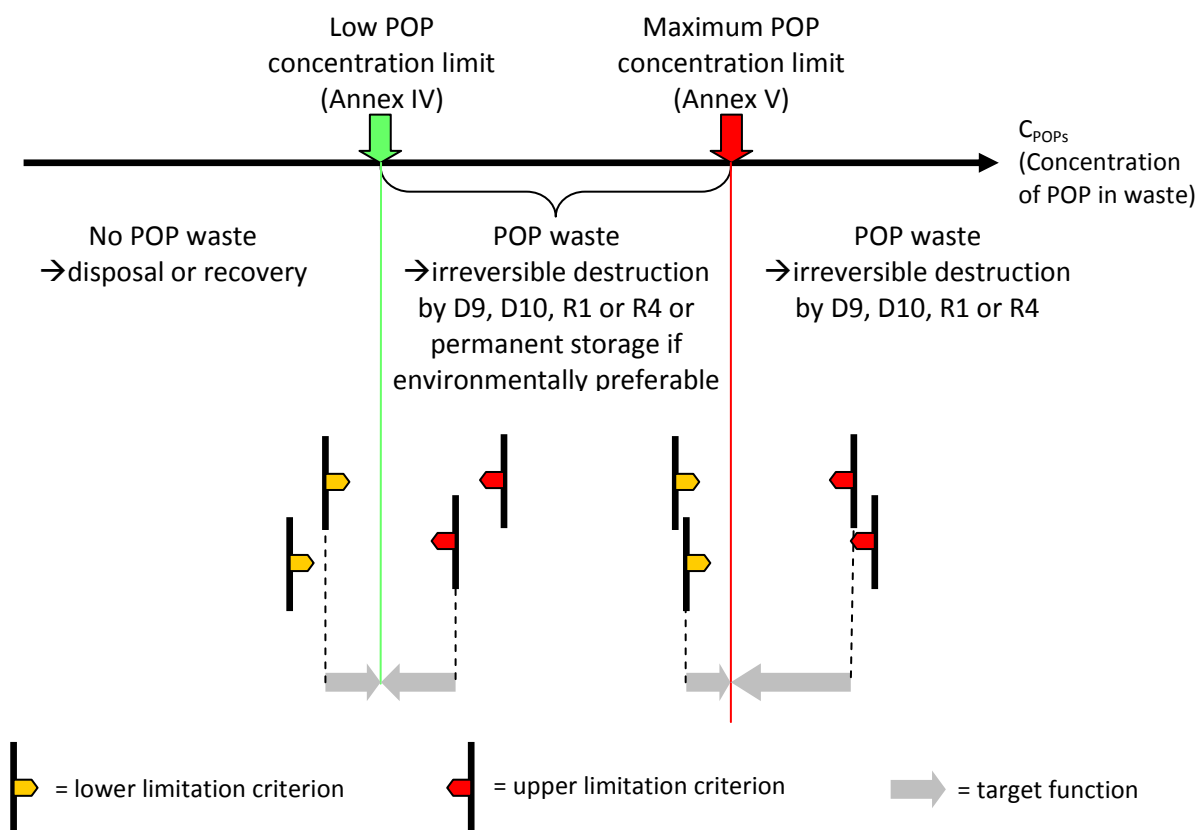


Figure 9-1: Methodological approach to derive proposals for POP concentration limits

For each substance a set of lower and upper limitation criteria is applied in order to determine a possible range for the limit values. Subsequently, the range is narrowed to a specific value by means of specific target functions in order to enable a specific proposal for a limit value (see Figure 9-1). The method is explained in this chapter.

9.1.2 Basic principles of the method and its assessment criteria




According to the Commission Communication on the precautionary principle [EC 2000] with respect to the Stockholm Convention “measures based on the precautionary principle should be, inter alia

⁵³ Maximum concentration limits of substances listed in Annex IV of Regulation (EC) No 850/2004 and specified in its Annex V

proportional to the chosen level of protection, ... based on an examination of the potential benefits and costs of action or lack of action (including where feasible an economic cost-benefit analysis)” Besides this, “examining costs and benefits entails not simply a cost-benefit analysis: its scope is much broader, and includes non-economic considerations, such as the efficacy of possible options and their acceptability to the public.”

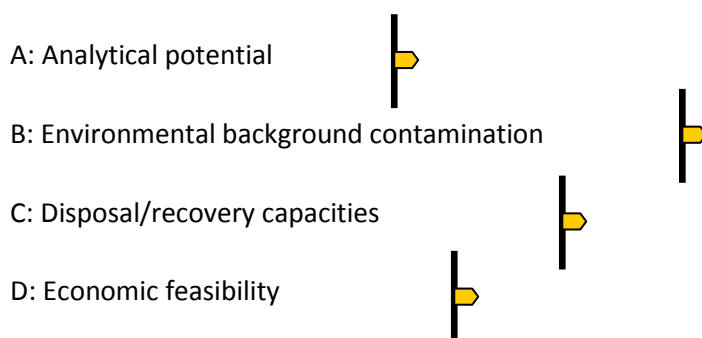
Thus the derivation of LPCLs for the classification of wastes as POP wastes takes into account a number of criteria relevant for the evaluation of impacts and implications of measures to take. These shall include environmental and health aspects as well as technical feasibility, economic impacts and existing commitments. Furthermore the relative importance of waste streams in terms of amounts of waste and amounts of POPs, as well as the relation between annual contribution and environmental load from other sources has to be taken into account for the suggestion of appropriate LPCLs.

The basic principles of the developed methodology are:

- I  A set of **lower limitation criteria** for the low POP concentration limit indicate POP concentrations below which limit values should not be set.
- II  A set of **upper limitation criteria** for the low POP concentration limit indicate POP concentrations above which potential limit values should not be set.
- III  According to **target functions** a resulting range is reduced to a specific POP concentration limit.
- IV In case of contradictions between lower and upper limitation criteria additional requirements have to be established to dissolve the contradiction.

Based on the requirements for the methodology mentioned above the following limitation criteria for low POP concentration limits have been used:

Lower limitation criteria:

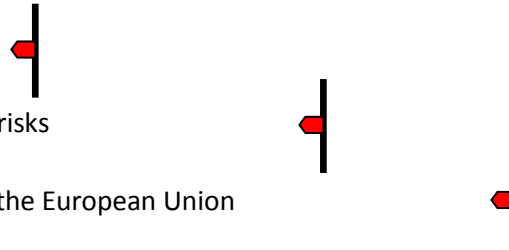


Upper limitation criteria:

X: Precautionary principle

Y: Worst case scenario for human health risks

Z: Existing limit values already agreed by the European Union



The assessment also includes the proposed threshold below which a substance is considered to be occurring as an unintentional trace contaminant (referred to in Article 4.1 (b) of the POP Regulation⁵⁴) for PBDEs (10 mg/kg) and PFOS (10 mg/kg, and 1,000 mg/kg)

It has to be stated that the limitation criteria are not of equal relevance and weight with respect to health and environmental protection, however they are important for the implementation and acceptability of the intended measure and are important to assess the efficacy and proportionality of a potential limit value. However the methodology has been designed to be flexible to up-coming knowledge and technical development and thus will enable to take new information into consideration.

Consequently the proposals for limit values in chapter 9.2.3 reflect implementable limits based on current knowledge and present technical status. The results however may change with up-coming knowledge and technical changes, so that a review process and a flexible adaptation of the limits in the Annexes to the POP regulation should be foreseen.

An exemplary possible distribution resulting from the evaluation of the lower and upper limitation criteria is illustrated in Figure 9-2.

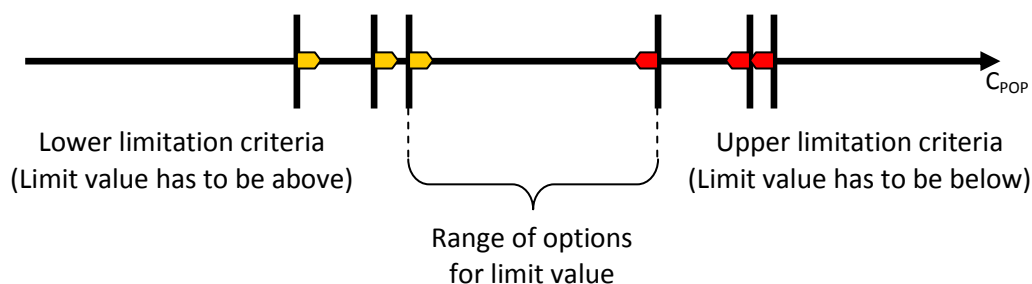


Figure 9-2: Resulting range of options for a limit value after application of lower and upper limitation criteria

The resulting range of feasible limit values differs for different POPs and different results can be expected for different waste codes. Thus it will not be possible to derive specific proposals for limit values and it is therefore necessary to reduce the range of possible options for a limit value to an

⁵⁴ According to Article 4(1)(b), control measures shall not apply in the case of a substance occurring as an unintentional trace contaminant in substances, preparations or articles.

implementable proposal for one specific value. For this purpose the methodology foresees the use of target functions as standard decision tool in decision theory in order to reduce the range of potential options. The following target functions are applied:

The **target function I "Reduce results for different waste matrices to the most unfavourable waste matrix"** can be applied to Criterion A (analytical potential) in order to reduce the large variety of achievable limits of quantification to an implementable value for all relevant waste matrices.

As a second tool criterion X (precautionary principle) as defined under the Stockholm Convention (Articles 5-7) "Each party shall ...take ...measures to reduce the total releases...with the goal of their continuing minimization and, where feasible, ultimate elimination " serves as **target function II** "reduce the limit value to the lowest limit value within the feasible range of options" in the final decision on a limit proposal in the range between upper and lower limitation criteria.

An exemplary illustration for the effect of target function I is given in Figure 9-3.

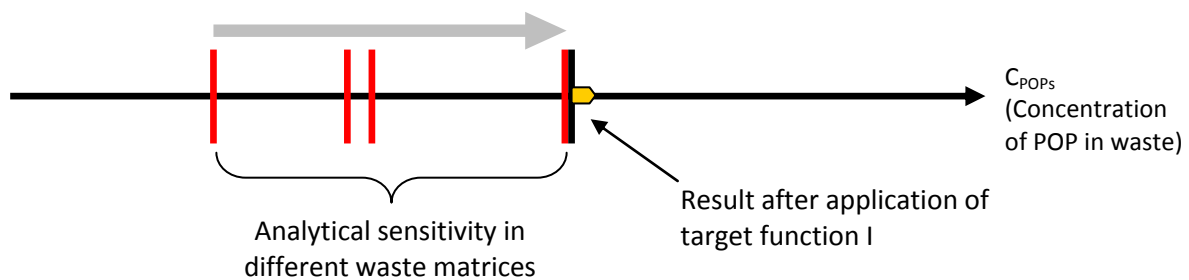


Figure 9-3: Target function I for reduction of varying analytical sensitivity in different waste matrices

The effect of target function II is exemplarily illustrated in Figure 9-4.

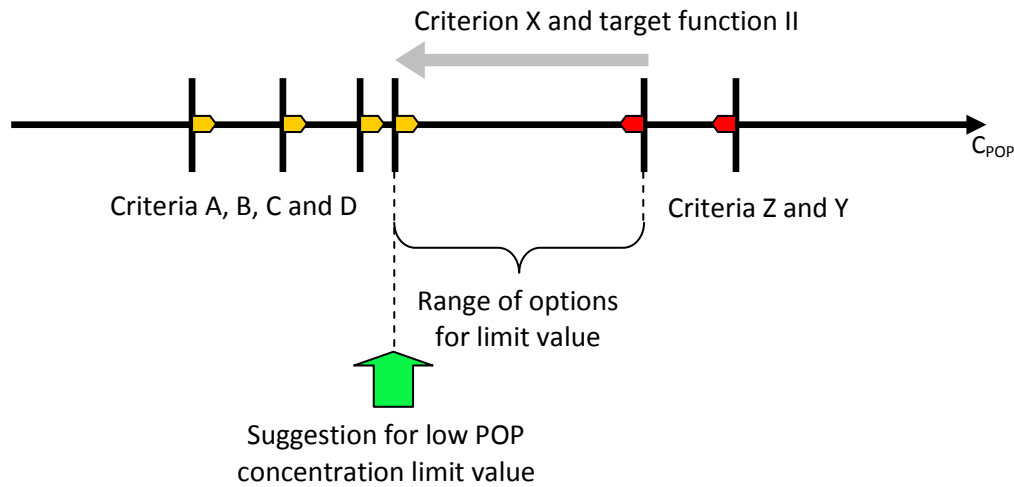


Figure 9-4: Target function II for decision on low POP concentration limit in the range between upper and lower limitations

If target function II is not applied the two edges of the range can be taken as two options for suggestions of limit values.

The application of lower and upper limitation criteria may result in controversial and contradictory requirements for limit values. Such a case is exemplarily illustrated in Figure 9-5.

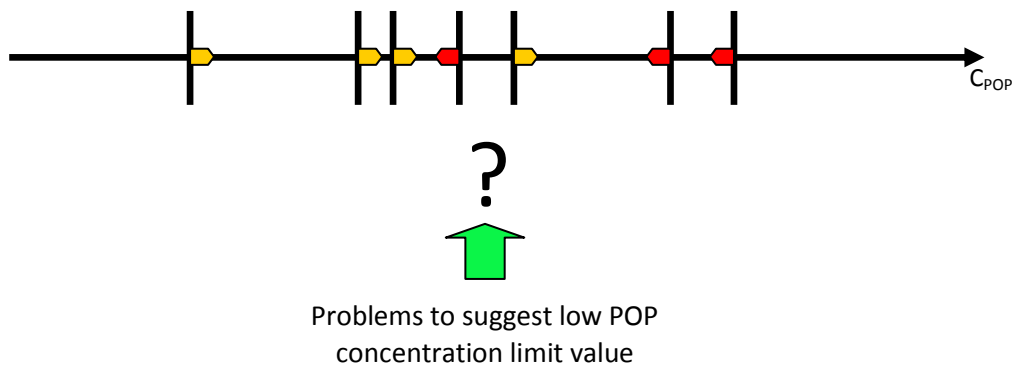


Figure 9-5: Conflict of interest between lower and upper limitation criteria

In such cases there is a contradiction between the results of the criteria because lower limitation criteria result in limit values above upper limitation criteria. As a consequence no range of possible options to set limit values remains. In such cases, expert discussion is necessary and additional requirements have to be established to dissolve the contradiction.

For the derivation of MPCLs, in principle the same lower limitation criteria can be applied. However, according to the POP Regulation, the MPCLs cannot be established below the low POP concentration

limit value. Therefore, once the LPCLs are derived, the lower limitation criteria for MPCLs can be concentrated into one single lower limitation criterion: "LPCLs".

In application of upper limitation criteria for the derivation of MPCLs, the following considerations can be made: Following the requirement of the POP Regulation to prove "environmental preferability", it is not necessary to apply criterion X (precautionary principle), because it is already taken into account when the decision on environmental preferability is taken. Furthermore, criterion Z on existing limit values is not applicable as limit values are not yet agreed.

As a consequence 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? This corresponds to the application of Criterion Y (worst case scenario for human health risks). The limit value must be below concentrations where corresponding risks may occur according to a worst case scenario.

9.1.3 Lower and upper limitation Criteria of Method 1 (assessment of minimum concentration limits)

Lower limitation criteria

Criterion A: Analytical potential

From the economic and technical point of view the lower edge of possible low POP concentration limits is marked by the limitations imposed to the system in terms of analytical accuracy and related costs. A limit which is not measurable in all Member States at reasonable economic conditions cannot be implemented. Thus the criterion of laboratory capacity and the relation between detection limit and costs of different analytical methods is a major criterion for the finding of the lower boundary of low POP concentration limit options. The methodological approach is demonstrated below.

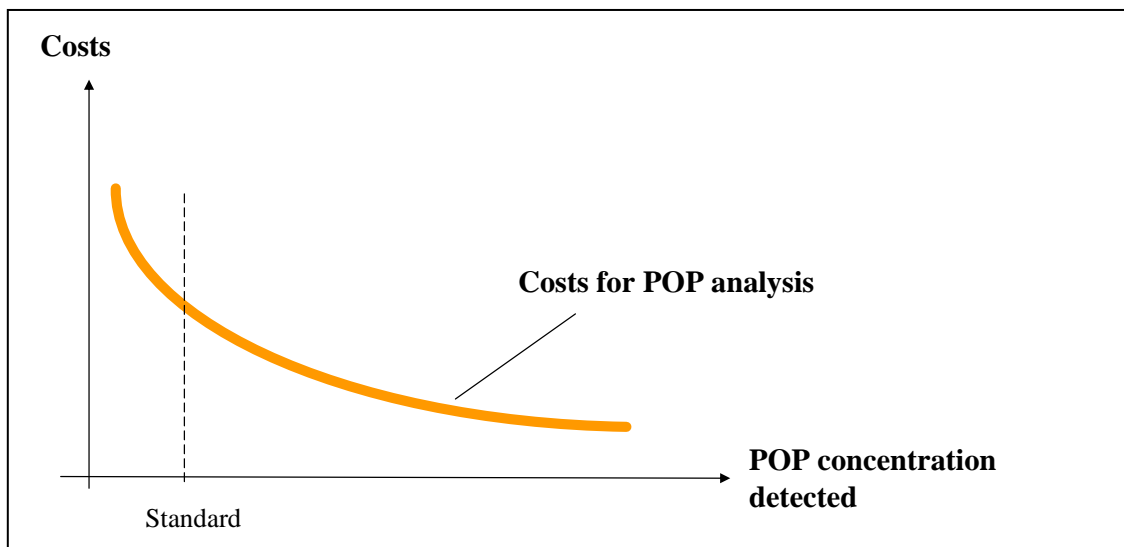


Figure 9-6: Feasible level of quantification as a function of costs and analytical sensitivity (schematic curve)

The “standard” LOD (limit of detection) for a pollutant reflects the performance of a widely available measurement technique with the POP substance enclosed in a usually prevailing matrix. Standard LOD do not represent minimum LOD achievable for the corresponding POP substance, but reflect LODs for established analysis standards for POP containing matrices, as far as available. In case that no established measurement method for analysis in waste is currently available for the corresponding POP, extrapolations have been used for the assessment of reduction in sensitivity or additional requirements with respect to sampling, pre-treatment, extraction and clean-up which limit the overall sensitivity achievable.

In order to be implementable applied standards will have to be applicable on-site for a quick control of delivered waste fractions at landfill sites or recovery installations. Thus standards have to comply especially with the provisions set up in Directive 1999/31/EC on the landfill of waste Article 11 (waste acceptance procedure) specified in Annex II (3, Level 2⁵⁵ and Level 3⁵⁶). Measurement techniques will have to be validated and issued as CEN standards in order to be added to the “sampling and test methods” as required in Article 3 of Council Decision 2003/33/EC on acceptance criteria for waste at landfills.

Standard LODs might be different for different matrices and wastes. Furthermore, established measurement methods are not available for various wastes with possible POP contamination. Related costs are a function of analytical effort related to the desired level of sensitivity. The relation between costs for POP analysis and related detectable concentrations is schematically illustrated in Figure 9-7.

⁵⁵ Compliance testing. This constitutes periodical testing by simpler standardised analysis and behaviour testing methods.

⁵⁶ On-site verification: This constitutes rapid check methods to confirm that a waste is the same as that which has been subjected to compliance testing and that which is described in the accompanying documents

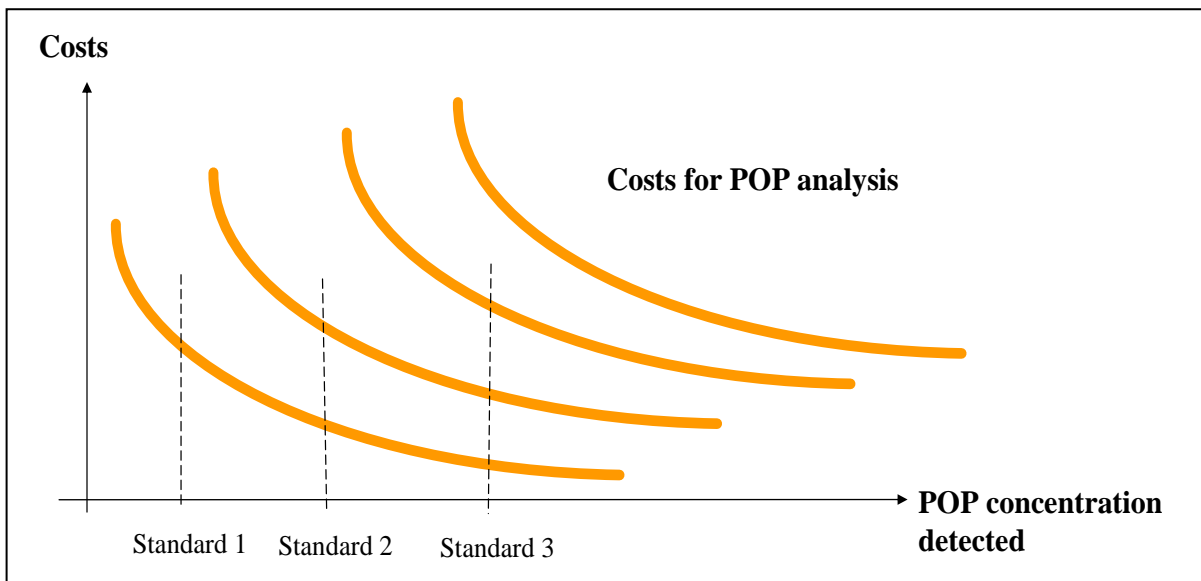


Figure 9-7: Series of curves for different measurement methods and waste matrices illustrating the relation between costs for analysis and detectable concentrations (LODs) (schematic curves)

As mentioned above the POP concentrations that can be measured in wastes respecting all parameters stated above and containing sufficient statistical strength have to be reduced by application of the target function I in order to allow a proposal feasible for implementation and enforcement.

As the minimum concentration limit values have to be operational for all relevant waste types, the limitation on potential limit values arising from Criterion A have to be based on the results achievable for the most unfavourable but relevant (waste) matrix.

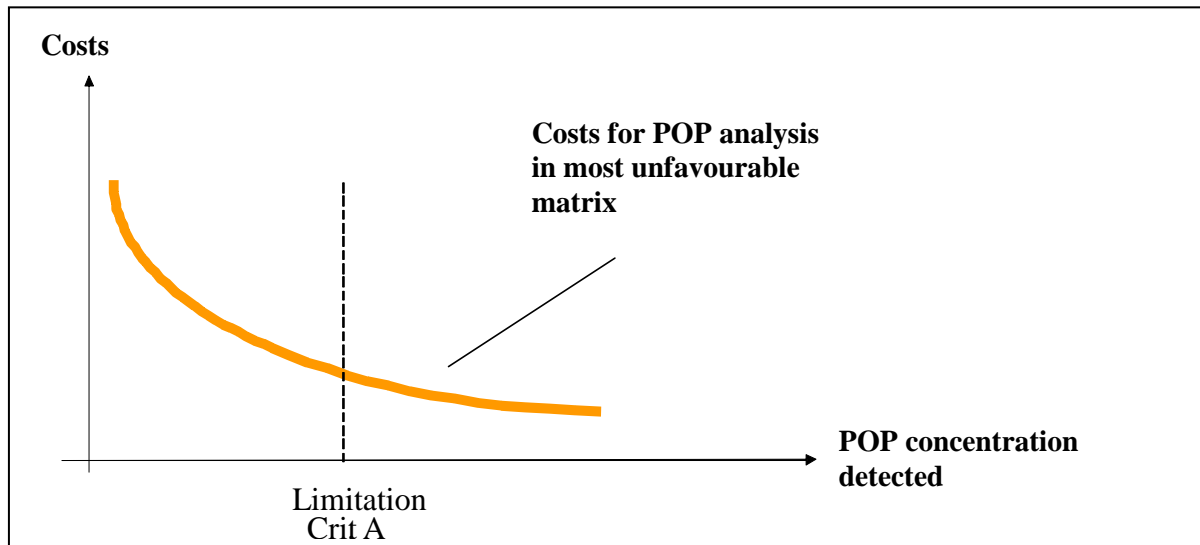


Figure 9-8: Limitation for most unfavourable matrix (schematic curve)

Besides quality requirements that have to be achievable in all EU 27 Member States availability of capacities depends on the measurement cost incurred by the applicants of a technique. A limit value giving rise to unacceptable cost cannot be enforced. In the course of time, though, the use of more sophisticated technologies can become affordable.

Such development corresponds to a shift of Limitation Criterion A in the direction of lower concentrations. Likewise, the “standard” mark can shift to smaller values with technical progress or extra effort at laboratory level, e.g. for additional cleaning steps.

The described procedure can be applied to all new and candidate POPs. The current results are presented in chapter 9.2.3.

Criterion B: Environmental 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. Any limit value should be significantly higher than average or background levels observed in the environment. A low POP concentration limit value below environmental background concentrations would cause severe economic problems and problems of acceptability if e.g. a soil with usual background concentration would have to be treated as POP waste as soon as it is excavated and the owner intends to get rid of it.

Therefore background contamination levels need to be taken into consideration. For this assessment environmental contamination levels have been compiled for all relevant substances within the respective sections in chapter 5 (detailed mass flows). “Hot spot” data indicate the factor by which common values are likely to be exceeded. Combining both types of information and the number of data available, a level of contamination and an “uncertainty factor” (that expresses the uncertainty related to the availability of representative data) are derived that lead to a value for the lower limitation criteria.

For future revisions of the limit values it should be taken into account that background levels are expected to decline.

Criterion C: Disposal/recovery capacities

Proposals for limit values should take into account that with a large number of waste types classified as POP wastes, considerable waste amounts might be directed to specific disposal/recovery operations. Therefore the proposals for a low POP concentration limit should take into account whether sufficient disposal/recovery/recycling capacities exist or can/must be established on national and community level for the management of the resulting POP waste streams. This process uses the results from the material flow analysis and is carried out according to the scheme illustrated in Figure 9-9.

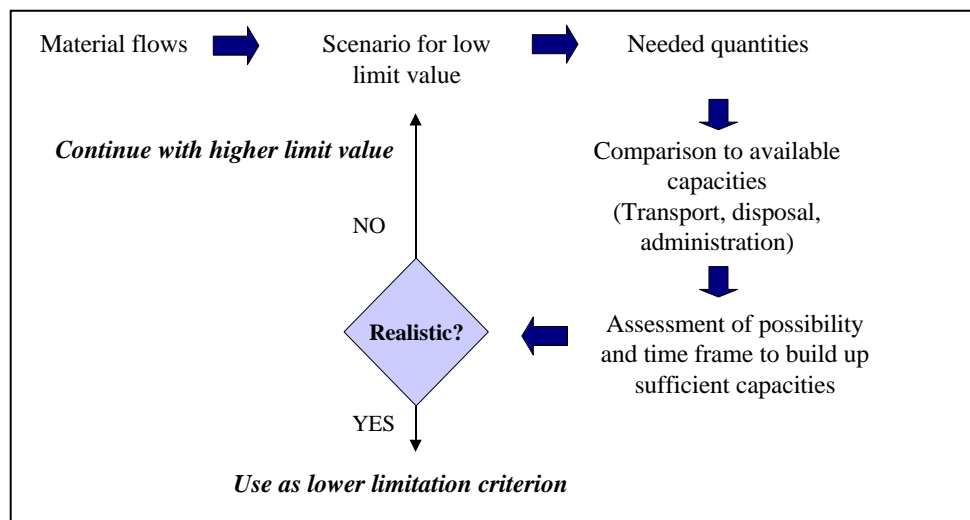


Figure 9-9: Impact on mass flow and disposal/recovery capacity as a limitation criterion for the definition of the low POP concentration limit

It may be difficult to come to consensual results with respect to "realistic" capacities as the involved stakeholders have different interests. Therefore scenarios have been developed in chapter 7 to identify the relation between potential limit values and resulting waste quantities becoming subject to the management provisions set in Annex V to the POP regulation. This approach is exemplarily demonstrated in Figure 9-10

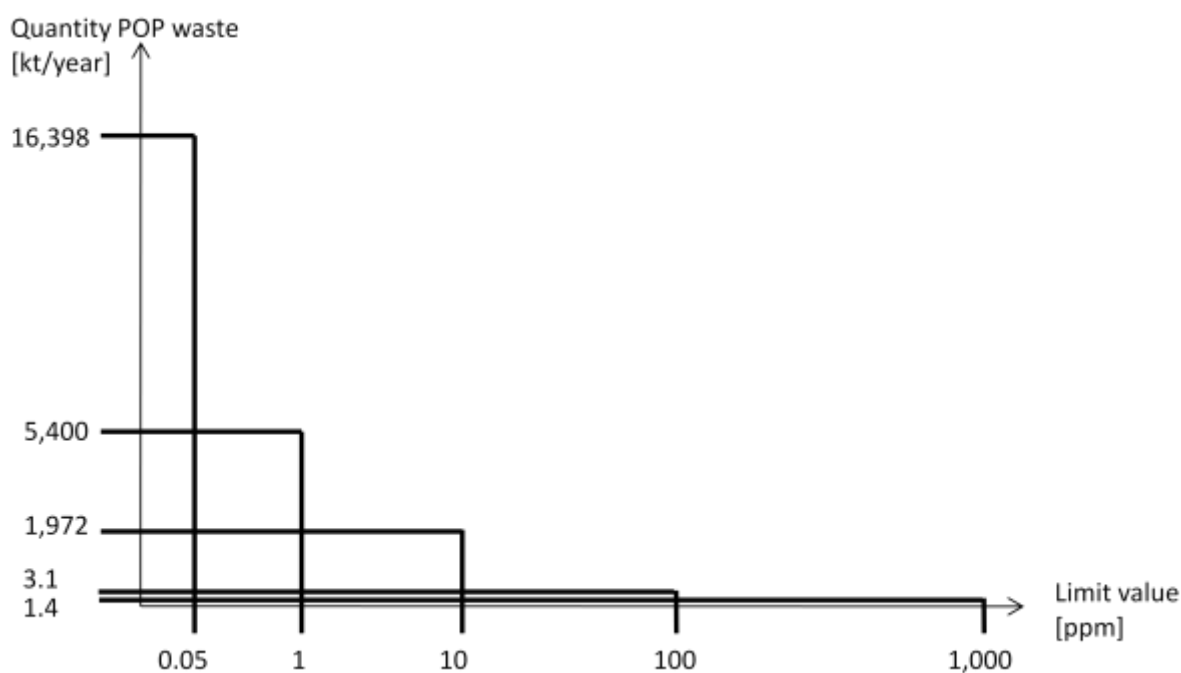


Figure 9-10: Exemplary correlation of low POP concentration limit and quantities waste exceeding the concentration

The results of these scenarios are also used for the assessment of economic impacts (criterion D).

It has to be taken into account that results from criterion C will change with technical development and political decisions. Thus it may not be used as finally limiting factor for the decision on limit values. However it is important for the assessment of the effects and the implementability of potential limit values.

Criterion D: Economic feasibility

The evaluation of a possible low POP concentration limit against the economic feasibility depends on the induced POP waste quantities as well. Costs arise when material classified as POP waste requires specific treatment, or is excluded from profitable recovery operations. As for criterion C, iteration along a simple scheme is applied, which is illustrated in Figure 9-11.

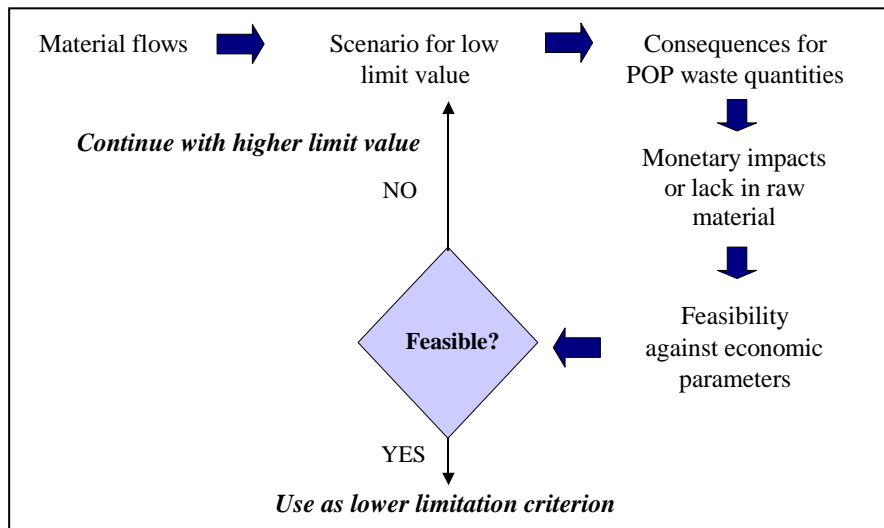


Figure 9-11: Economic impacts as a limitation criterion for the definition of low POP concentration limits

Wider cost-benefit aspects and environmental impact from operations potentially concerned have to be taken into account in a complete economic impact assessment. The scope of this project however, does not allow a complete economic and environmental risk assessment as part of an economic impact assessment.

Upper limitation Criteria

Criterion Z: Existing limit values agreed by the European Union

Proposed low POP concentration limits should not exceed existing limits agreed by the Community or by international conventions. To define the upper limitation of the limit range, existing European legislation or international agreement in the field of waste management have been taken into account. Stricter regulation in single Member States must not pose a constraint for the upper limitation of the range of possible options for concentration limits.

The analysis is supported by results of chapter 4 where existing concentration limits are compiled. A distinction was made between

- International conventions
- EU legislation

- Member States legislation

was made.

Related legislation and established limit values were structured into the fields of

- waste management
- chemicals – classification, production, and use
- atmospheric pollution – air emissions and ambient air
- water protection
- food and feed safety
- specific legislation on POPs implementing international conventions

Criterion Y: 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. It is therefore strongly correlated to its specific bioavailability and potential to enter the food chain. The risk resulting from hazardous substances in waste is basically a function of physico-chemical properties of the waste in combination with the specific waste management operations.

Criterion Y targets the following risk components: the possible events of damage that can occur along a waste's life cycle, their severity, and the realistic probability of such an event. Consequently, the assessment methodology with respect to criterion Y combines elements of Life Cycle Assessment, conventional Risk Assessment and Impact Assessment.

Starting from the toxicological properties and exposure as the two key components of a Risk Assessment, it is possible to assess the extent to which the probability of adverse health and environmental effects can be influenced by the established LPCLs.

Generally spoken, risks to humans occur via inhalative, dermal and oral exposure. Occupational and residential exposure settings have to be taken into consideration for a risk assessment. However, a number of factors can be excluded from in depth investigation due to a low probability to generate relevant impacts or because the risk potential is not influenced by the classification of waste under the POP regulation.

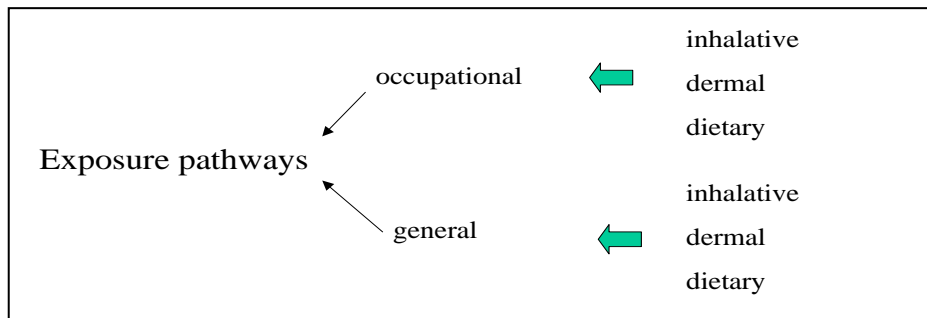


Figure 9-12: Exposure model according to the Risk Assessment methodology

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. Instead existing information from literature is used and investigations are targeted to identify those risks that can be mitigated by means of a limit value concept, and that are not covered by the Basel convention. An approach in the style of a life cycle analysis helps to systematise possible sources of risk.

As illustrated in Figure 9-13 relevant stages of the waste life cycle to be investigated are waste generation, treatment, transport, and finally disposal/recovery.

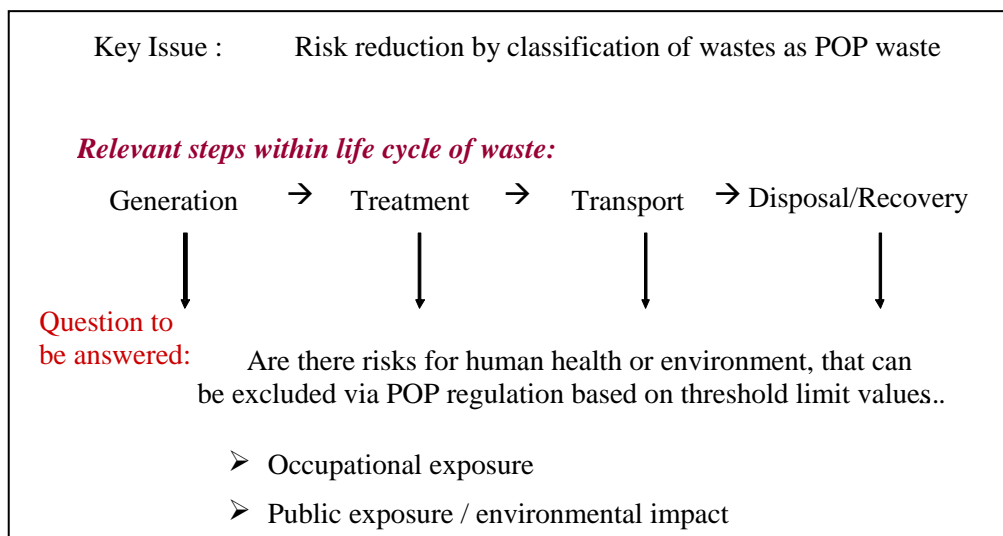


Figure 9-13: Life cycle of waste in relation to Regulation (EC) No 850/2004

When the relevant sectors have been identified, bioavailability and toxicological properties of the substance in question define the potential health effects in case of exposure, and limit the acceptable exposure levels for humans or wildlife.

In the course of a risk assessment regarding new and candidate POPs in the scope of the POP regulation and waste management it can be stated that for the three stages (1) waste generation, (2) storage and (3) transport, legislation in place does not require differing management relating to whether or not the waste is classified POP waste. Thus only the risks related to waste disposal/recovery is in the scope of

this project. Thus the potential impacts from waste disposal/recovery on environment and humans have to be investigated. For this purpose literature has been reviewed for TDIs, MRLs, as well as for information on environmental transport and fate and risk assessment for the general population and for professional exposure where relevant. After identification of the most critical pathways for environmental and human exposure existing legislation was cross-checked for potential restrictions/limitations imposed to it.

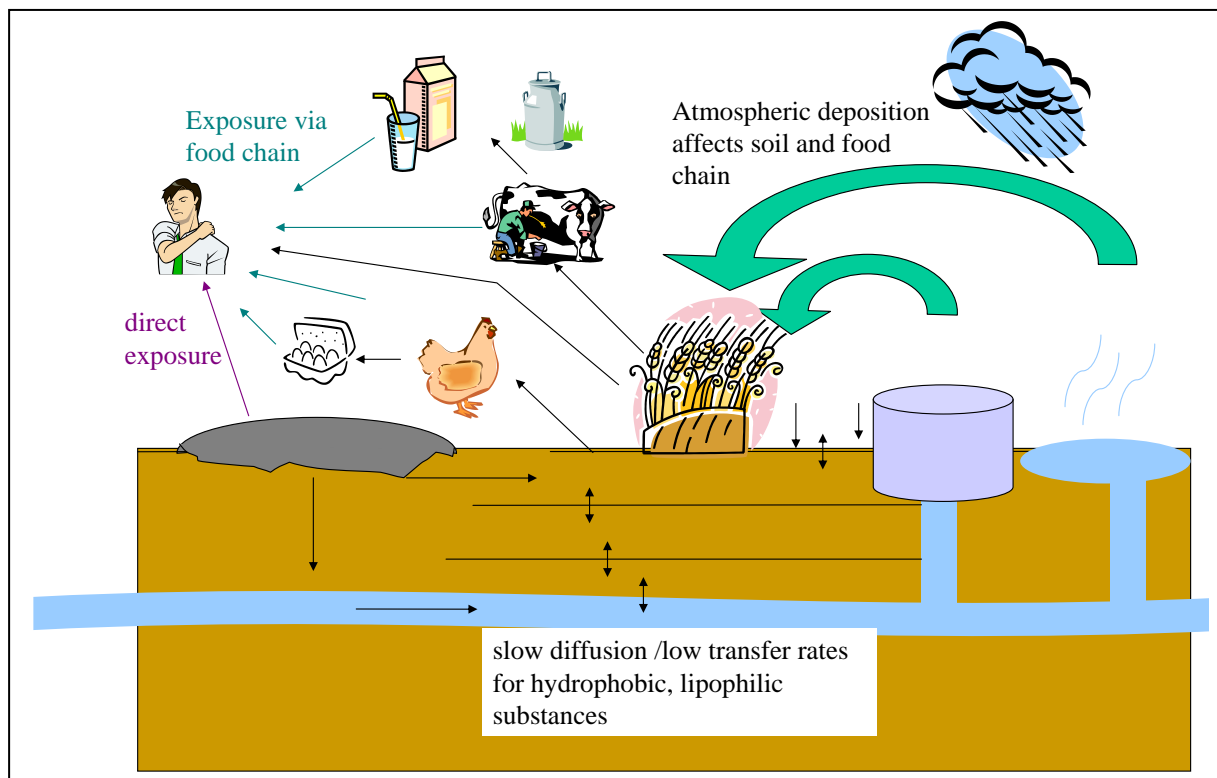


Figure 9-14: Pathways to different environmental compartments and to humans from new POPs and POP candidates released into the environment

Criterion X: Precautionary principle

According to the provisions set in the Stockholm Convention and in the POP Regulation (EC) no 850/2004, the precautionary principle shall be applied to all POP related issues. This aims at reducing emissions and discharge to the highest possible extent, in order to protect the environment and human health. Thus, due to the high persistency and the potential long-term effects, the precautionary principle demands to opt for the lowest of concentration limits which can be accepted considering the before mentioned limitation criteria A-D, Y and Z.

Criterion X therefore is used in the decision making process as target function II in order to reduce the range of potential concentration limits to its lowest value.

9.2 Proposals for low POP concentration limits

The proposals for low POP content limit values presented in this chapter reflect implementable limits based on current knowledge and the present technical status. The results however will change with up-coming knowledge and technical changes, so that a review process and a flexible adaptation of the limits in the Annexes to the POP regulation should be foreseen. As explained in chapter 9.1 the final proposals are the result of the application of the criteria A-D (lower limitation criteria) and X-Z (upper limitation criteria). A lower limit of 50 ppm for new POPs – in analogy with existing limits for POP pesticides and other POPs is included into the investigation on request of the Commission Services.

The investigation is restricted to effects due to the POP content of wastes. Classification as hazardous or restriction in use due to other pollutants is not in the scope of this study. However current disposal/recovery/recycling practice reflecting this is taken into consideration for the assessment of disposal/recycling capacity and economic impacts.

9.2.1 Results for lower limitation Criteria

The requirements with respect to acceptable concentration limits for POPs in waste discussed in this chapter result from an investigation of the current status and state of the knowledge in all investigated issues in order to provide a technical basis for the political discussion. It has however to be pointed out, that requirements and limitations for most of the criteria are flexible and will change with up-coming knowledge and technical development. Thus a review process and further investigations in a number of topics should be foreseen.

Results for criterion A: Analytical potential

This chapter compiles the results of investigations on measurement methods and achievable analytical sensitivity. A more in depth discussion of basic measurement requirements, available standards is presented in chapter 0.

The assessment is not based on lowest achievable limits of detection (LOD) but based on well justified limits of quantification (LOQ) in currently used standard methods - as far as available - including state of the art and expert knowledge on limitations and restrictions due to matrix properties. In case no standard methods are available, measurement methods and achievable analytical sensitivity reported in scientific literature have been analysed for their suitability to determine POP concentrations in waste matrices.

As standardised measurement methods for POP analysis do not yet exist for most of the relevant waste matrices (which have been detected in the chapters relating to mass flows), in many cases an estimation of the achievable LOQ using LODs reported for a different matrix and a factor taking into account the characteristics of the respective matrix had to be performed. In case there does not exist a standard method for analysis of a relevant POP substance, LODs reported in scientific literature for analytical methods suitable to detect the respective substances have been used to calculate a LOQ reflecting the state of the art in the respective field of POPs analytics.

Based on internationally agreed standard methods or the methods reported recently scientific literature, and using the most unfavourable relevant matrix as a reference⁵⁷, the following limits of quantification (LOQ) are achievable for the different new POP and candidate POP homologues/substance groups.

Table 9-1: Limits of quantification for new POPs and candidate POPs in most unfavourable relevant matrices

Substance	LOQ (Limitation criterion A)	Example for most unfavourable relevant waste matrix
TetraBDE	10 ppm (mg/kg) or 100 ppb	Plastics
PentaBDE	10 ppm (mg/kg) or 100ppb	Plastics
HexaBDE	10 ppm (mg/kg) or 100 ppb	Plastics
HeptaBDE	10 ppm (mg/kg) or 100 ppb	Plastics
PFOS	100 ppb (µg/kg)/100 ppb	Carpets, leather, textiles
PeCB	1 ppb (µg/kg)	Sludges / Ashes
SCCP	10 ppm	Rubber, sealant and adhesives, paint and varnishes, textiles, leather, sewage sludge
HBCD	10 ppm	Polymers, textiles
PCP	0.1 ppm	Impregnation, textiles
HCBD	0.05 ppm	Sludges/Ashes
PCN	0.1 ppm	Sludges/Ashes

The concentrations indicated as limitation level are derived from the assessment of the analytical potential of a established standard/method suited best for the substance in question and most appropriate for waste matrices, taking into consideration also applicability for the intended purposes – mass screening.

PBDEs

According to chapter 5, use of PUR foams in upholstery, furniture, automotive applications have been identified as predominant source for C-PentaBDE. Use in Acrylonitrilebutadiene-styrene (ABS) polymers for housings/casings of EEE has been identified as major source for C-OctaBDE.

Within a BAM (Bundesanstalt für Materialforschung und –prüfung) project, which resulted in method IEC (International Electrotechnical Commission) 62321, it has been shown by interlaboratory studies that the developed standard measurement method provides precise measurement results for the determination of the sum penta- and sum octa-BDE isomers in articles respectively, in particular within the range of 0.1 % by mass (1 mg/g = 1,000 ppm). The LOQ for penta- and octa-BDE congeners has been reported to be 0.06 to 0.12 mg/g (60-120 ppm) and was established according to DIN 32 645 by comparison of interlaboratory measurement results with standard materials. These standard matrices included epoxy resin, polyurethane, polystyrene and ABS (acrylonitrile-butadiene-styrene-copolymer). However, the BAM method was developed to provide a method to verify the threshold value set by Directive 2003/11/EC for pentabromo and octabromo diphenylethers (0.1 %) and has not been

⁵⁷ being well aware that the different POPs can usually be measured at concentrations far below the concentrations that are measurable in unfavourable matrices

optimised for detecting small amounts of PBDEs in plastic waste. The relative high LOQs reported for the BAM method are considered not to be suitable as limit values for PBDEs in plastic waste. The precision and accuracy of the IEC method and the method detection limit will be available once the ongoing interlaboratory study is finished.

LOQs of standard methods for other matrices like sewage sludge (estimated LOQ < 130 µg/kg = ppb) have been reported to be much lower than the LOQs reported for the BAM method. Available standard methods are able to detect PBDEs in soil and sludge at levels down to ppt (pg/g) depending on the analytical effort applied. However these levels are not per se achievable for complex waste matrices with the original standard. Considering decreased sensitivity due to very difficult measurable matrices the achievable LOQ will be higher when analysing waste matrices with the standard method to determine PBDEs in soil and sludge compared to the reported figure. HeptaBDEs are the congeners with the highest LOQ. Therefore a LOQ of 1,280 pg/g dry weight has been used as a basis for the calculation. Trying to improve sensitivity needs extra effort (e.g. repeat standard clean-up, include further clean-up steps, turn to more specific measurement e.g. HRMS) and increased costs. An LOQ of 13 ppb (ng/g) seems to be achievable for single PBDE congeners at reasonable conditions in soils or sludge.

Methods to determine PBDEs in polymer matrices reported in literature are suitable to detect PBDEs at a level of ppm (µg/kg) or even down to ppb. Pöhlein developed a gas chromatography/mass spectrometry (GC/MS) method as well as gas chromatography with electron-capture detection (GC/ECD) method to detect PBDEs in styrenic polymers. LODs between 2.4 mg/kg (BDE-138) and 9.5 mg/kg (BDE 154) have been reported for the determination of penta- and octaBDE congeners in styrene reference materials.

Basic GC-MS systems are sufficient to achieve LOQs in the dimension of 10 ppm (for single congeners) and it is assumed that it will therefore be possible to detect PBDEs in plastic waste matrices following the IEC method in the same range. Unreasonably expensive analytical equipment is not required to detect PBDEs in the ppm range. Lower LOQs (100 ppb – 1 ppm for individual congeners) can be reached with analytical equipment with enhanced sensitivity (GC/HRMS) as demonstrated e.g. by Kim et al. 2006, or by Danon-Schaffer, 2010, or by more extensive extraction and clean-up effort. This results in higher costs which can be expected to reach up to a similar dimension as for PCDD/PCDF.

A LOQ of 100 µg/kg (ppb) – 1 (mg/kg) ppm for individual congeners is considered to be achievable in all EU 27 Member States regarding measurement costs incurred by the application of the required GC-MS technique.

As an alternative with several advantages compared to the time consuming determination of PBDE content in wastes by GC-MS, the screening for Br by XRF has intensively been discussed in chapter 0. XRF can be easily used for on-site verification on landfills (hand-hold systems), is very quick and comparably cheap (costs for equipment and trained personnel). On the other hand, XRF only detects Br and it is impossible to detect PBDE congeners with this method. An LOD for PentaBDE of 0.4 mg/g (400 ppm) has been reported. If XRF is considered as an alternative method, a shift of Limitation Criterion A in the direction of higher concentrations would be mandatory.

PFOS

Carpets and leather as well as fire fighting foams and hydraulic liquids have been identified to be the most important waste matrices containing relevant amounts of PFOS (chapter 6.3.5).

The technical specification of the standard method "Determination of extractable perfluorooctanesulfonate (PFOS) in coated and impregnated solid articles, liquids and fire fighting foams - Method for sampling, extraction and analysis by LC-MS/MS or LC-MS" (FprCEN/TS 15968) has already been prepared by CEN/TC 382 and is expected to be adopted in 2010. This standard method is assumed to cover practically all relevant matrices. LOQs achievable with this method are not yet available. Regarding determination of PFOS in extracts, LODs between 0.5 ng/ml and 50 ng/ml will be possible with the proposed LC-MS or LC-MS/MS methods. CEN has confirmed that the analytical method described in the technical specification can be used for the proposed threshold of 0.001% pursuant to Annex I to the POP Regulation (10 µg/kg = 10 ppb).

A method for the analysis of extractable PFOA was developed to evaluate leaching of PFOA from treated textiles and carpet. The method compared extraction efficiencies of water, sweat simulant and saliva simulants with that of MeOH using LC-MS/MS. Limits of detection of between 1 and 3 µg/kg of sample were reported [Mawn et al., 2005]. Stadallius et al. extracted PFOA from paper and textile samples achieving LODs of 1 µg/kg for paper and of 2 µg/kg for textile samples [Stadallius et al., 2006].

Although quality criteria of standard methods are not yet available, expected LOQs of standard methods (CEN) and LODs reported in literature are in the same range (1-10 µg/kg). Nevertheless, there are only limited data available which can serve as a basis for the estimation of a limit value. Considering the different matrices relevant for PFOS, it can be concluded from the literature search that methods for the determination of PFOS in environmental samples have lower LOQs than methods for PFOS in articles or textiles. The reason therefore is the less complicated extraction and clean up. Against this background, the limit value for PFOS in waste is calculated based on LODs reported in literature and the expected limits of method FprCEN/TS 15968 assuming that leather and carpets (textiles) are the most unfavourable matrices relevant for PFOS. Taking into account the limited set of information, it is difficult to estimate an achievable LOQ from reported LODs. However, a LOQ of 100 µg/kg (ppb) for textiles and leather is considered to be achievable with reasonable analytical effort. This also covers matrices like fire-fighting foams, soil or sludge. As soon as data for FprCEN/TS 15968 will be available, the suggested limit value may be adapted.

The LOQ of 100 ppb is considered to be suitable for PFOS in relevant waste matrices, as the limit of standard method FprCEN/TS 15968 will be in the same range and as LOQs of methods for other matrices like sludge (estimated LOQ < 10-50 µg/kg) are much lower. Furthermore, these limits seem to be achievable in all EU 27 Member States regarding measurement costs incurred by the application of the required LC-MS/MS technique. Unreasonably expensive analytical equipment is not required to apply the FprCEN/TS 15968 method and achieve the reported LOQs.

PeCB

Ash from different burning processes and sewage sludge has been identified as relevant matrices for PeCB (chapter 6.4). Available standard methods are able to detect PeCB at levels down to below $\mu\text{g/kg}$ values depending on the analytical effort applied. However these levels are not per se achievable for complex waste matrices with the original standard methods. LODs for the determination of PeCB in solid matrices like soil or sludge using Soxhlet extraction and GC-MS have been reported to be in the range of $2 \mu\text{g/kg}$ (DIN 38414-S20).

LODs reported for literature methods are in the same range or lower. Veningerova and Sarrion used GC-ECD and GC-MS after Soxhlet or SPME extraction to analyse sewage sludge for their PeCB content. LODs have been reported to be $0.07 \mu\text{g/kg}$ for sewage sludge. Fly ash from a municipal solid waste incinerator has been analysed by GC-MS after solvent extraction (Tsytisik, 2008) with $\text{LOD} < 3.7 \mu\text{g/kg}$. Korenkova et al. (2006) analysed fly ashes with large volume injection-gas chromatography–mass spectrometry and reported PeCB levels down to 1.1 ng/g ($\mu\text{g/kg} = \text{ppb}$). Schreiner et al. (1986) reported PeCB contents of fly ashes and waste ashes from waste incinerators from 0.004 ng/g to 150 ng/g .

Taking into account the limited set of information, it is difficult to estimate an achievable LOQ from available LODs. However, a limit of quantitation (LOQ) of $1 \mu\text{g/kg}$ (ppb) is considered to be suitable as limit value for PeCB in relevant waste matrices, as this LOQ corresponding to standard method DIN 38414-S20 (sludges) and estimated LOQs for most relevant matrices like fly ash ($< 0.1 \mu\text{g/kg}$) are in the same range. Furthermore, these limits seem to be achievable in all EU 27 Member States regarding measurement costs incurred by the application of the required GC-MS techniques. Unreasonably expensive analytical equipment is not required to apply standard methods and achieve the reported LOQs.

PeCB can be analysed together with other (semivolatile) chlorinated organic compounds (e.g. chlorinated biphenyls and benzenes) in screening methods (EPA 8270D/DIN 38414-S20 and others). Analytical equipment established on-site at landfills, waste incineration plants or recycling facilities to determine chlorinated organic compounds in waste matrices allow detection of PeCB with adequate sensitivity to achieve the proposed LOQ of 1 ppb.

To establish methods with lower LOQs is possible but requires analytical equipment with enhanced sensitivity or more extensive extraction and clean-up effort which might result in higher costs.

SCCP

At the moment neither natural occurrence nor unintentional production of SCCP are known. All existing SCCP is therefore anthropogenic and regarding to the UNECE CLRTAP POPs protocol shall be eliminated except for the use as fire retardant in conveyor belts in the mining industry and in dam sealants. On the EU level the SCCP concentration in leather tanning and in the metal industry shall not exceed 1 % of SCCP, which is almost equal to a ban. For all other uses such as rubbers, sealants, adhesives, paints, varnishes and textiles there are no restrictions. Today, the rubber industry is the main application of SCCP followed by the sealants and adhesives section and by the paints and varnishes section. The textile industry covers only a small fraction of the overall used amount of SCCP. Regarding the waste stream all of the mentioned sections are included in a similar ratio as their current use. Also leathers which have been impregnated with SCCP in the past can enter the waste stream due to their long lifetime.

An overview of analytical methods which have been reported in literature for SCCP analysis in matrices with properties comparable to relevant waste matrices can be found in Table 7-9. Although not for all methods LODs have been reported it has been shown that GC-MS systems in combination with adequate sample extraction and sample clean up is suitable to detect SCCPs in matrices like sediment or sludge down to low $\mu\text{g/kg}$ values. There are a lot of reports regarding SCCP contents in sediments available. Parera et al. reported SCCP levels ranging from 0.25 to 3.04 mg/kg dry weight (DW) in river sediments in Barcelona, Spain. Stejnarova et al. detected <0.5 to 180.75 $\mu\text{g/kg}$ SCCP dw in top layer river and pond sediments in the Czech Republic and , Tomy et al. reported 1.6 to 257 $\mu\text{g/kg}$ in surface sediments at various latitudes and Arctic lakes in Canada [Parera et al., 2004; Stejnarova et al., 2005 and Tomy et al. 1997b]. These results demonstrate that available analytical methods are suitable to determine the SCCP content of sediment samples in the area of low $\mu\text{g/kg}$ values. Koh et al. reported a detection limit of 0.6 mg/kg (ppm) for single SCCP compounds in sealing materials. Another analytical procedure was able to classify chlorinated paraffins in technical products such as cutting fluids and lubricants with detection limits for restricted paraffins in samples between 0.02–0.08 % (w/w).

Against this background it is clear that achievable LODs are highly correlated to the kind of matrix which has to be analysed. LODs of analytical methods to detect HBCD in sediment or soil samples seem to be in a range of 1 to 10 $\mu\text{g/kg}$ (ppb). Higher LODs have to be expected when analysing matrices like sealing materials or fluids and lubricants 10-100 ppm. As no reports regarding analytical methods to determine the SCCP content of rubber or textiles have been identified, there are no figures representing achievable LODs available. However, LODs for these matrices are expected to be higher than LODs for sediment samples but lower or in the same range as for sealing materials and cutting fluids containing a lot of substances interfering with SCCPs during the analytical procedures.

Taking into account the high uncertainty in this estimation due to limited data regarding analytical procedures for relevant waste matrices, a LOQ of 10 mg/kg (ppm) seems to be achievable with standard laboratory equipment with reasonable analytical effort.

HBCD

HBCD is used as an additive flame retardant. Currently, the main downstream uses of HBCD are in the polymer and textile industries. HBCD 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. Further, HBCD can be used as textile coating agent in polymer dispersion on cotton or cotton/synthetic blends for upholstery fabrics, e.g. residential and commercial furniture and transportation seating, bed mattress ticking, draperies and wall coverings, interior textiles, e.g. roller blinds, and automobile interior textiles.

It can be concluded from the performed literature review that analytical methods which have been reported in literature for HBCD analysis in matrices with properties comparable to relevant waste matrices are able to detect brominated flame retardants in quite low levels. Although not for all methods LODs have been reported it has been shown that a combination of LC-MS systems with adequate sample extraction and sample clean up is suitable to detect HBCD in matrices like soil down to low $\mu\text{g}/\text{kg}$ values. Comparable results have been found for PBDEs which have been determined in relevant waste matrices with similar methods.

There are a lot of reports regarding contents in soil and sediments available. Yu et al reported HBCD concentrations in soil samples between 1.7 to 5.6 $\mu\text{g}/\text{kg}$ dry weight (ppb) [Yu et al., 2008]. Schlummer et al. developed a HPLC–UV/MS method to identify and quantify flame retardants in post-consumer plastics from waste of electric and electronic equipment (WEEE) with LODs in the range between 0.01 and 1 ppm [Schlummer et al., 2005]. Pöhlein et al. developed methods to identify and quantify brominated flame retardants (BrFRs) in styrenic polymers by GC-ECD. LODs for HBCD were 0.03 ng/cm^3 (standard solution) and 0.8 mg/kg (ppm) (GC-ECD polymer reference) [Pöhlein et al., 2008]. These results demonstrate that available analytical methods are suitable to determine the HBCD content of sediment samples in the area of low $\mu\text{g}/\text{kg}$ value.

It can be concluded that reported LODs for different matrices only slightly differ within a range of 0.01 and 1 ppm. A LOQ of 10 ppm should be achievable with standard laboratory equipment when establishing analytical procedures for determination of HBCD in relevant waste matrices.

PCP

In the past, PCP was used for many applications as anti-mildew mothproofing and antifungal agent for wood impregnation and in different industries such as leather, textile, mineral oil, glue and paint or pulp industry. Due to national legislation, EU legislations as well as conventions the use of PCP in the EU 27 has stopped. The relevant sources of waste regarding PCP are nowadays only from the wood impregnation industry and the textile industry. About 71 % of all PCP in the waste is from impregnated wood and the rest (29 %) from the textile industry.

The listed individual standard and literature methods are designed to detect PCP at quite low levels in the corresponding matrices, however, it has to be emphasised that the achievable levels can vary and strongly depend on the individual matrix to be analysed. Nevertheless, there are standardized methods

available to determine the mass concentration of PCP in relevant waste matrices like wood, paper and textiles which define critical performance criteria like LOD or recovery. CEN standard protocols allow analysis of relevant waste samples in a comparable way in all countries affected.

Performance of suitable quality tests by means of interlaboratory tests using certified reference materials is of higher importance than the method itself. Such QC is essential when different analysis methods will be allowed. Some of the listed standard methods have already been validated by international interlaboratory tests.

Available standard methods are able to detect PCP at levels down to low $\mu\text{g/kg}$ values depending on the analytical effort applied. These levels are achievable for the relevant waste matrices (wood, paper and textiles) with the original standard methods and with regular laboratory equipment.

Reported standard methods achieve LOQs in the range of 0.01-0.05 mg/kg in wood, paper and textile samples. A realistic LOQ easily achievable with standard laboratory equipment when following CEN standard methods for waste matrices is assumed to be 0.1 mg/kg (ppm).

HCBD

At present there is no intentional use of HCBD in Europe since many years [UNECE 2007], and also intentional production has stopped [UBA 2006]. HCBD is mainly formed as unintentional by-product during several industrial processes, but the available information basis is very scarce. However, production of tri- and tetrachloroethene and tetrachloromethane has been indicated as the most important source. Production of primary magnesium has been identified to belong to the historic uses by now, since its production within EU27 has stopped recently. Unintentional production and releases from the plastic industry have to be considered relevant, but no data is available. The same case has been identified for the sector of hazardous waste incineration and sewage sludge resulting from treatment of industrial waste water.

The most important waste matrices, as far as can be identified, are sewage sludges arising from waste water treatment. For the other investigated sectors no indications can be made since further investigations have to be conducted, e.g. on HCBD contamination values in diverse waste types arising from relevant sectors.

Available methods reported in scientific literature are able to detect HCBD at levels down to low $\mu\text{g/kg}$ values depending on the analytical effort applied. Gas chromatographic methods with ECD and other detectors have a detection limit for hexachlorobutadiene of 0.02-0.05 $\mu\text{g/L}$ in water [EPA 1982a, 1989c, 1989d, 1989e]. Detection limits for soil and solid waste are usually higher, depending on matrix interferences, extraction, and clean up procedures. LODs between 0.66 and 50 mg/kg wet weight (ppm) [EPA 1986] and between 0.05 and 2.5 mg/kg wet weight [EPA 1990e] have been reported for soil and solid waste. Measurement of HCBD content by means of GC-MS achieving a LOD of 170 ng/kg dry weight and a LOQ of 340 ng/kg dry weight has been reported for topsoil [UBA, 2008]. These levels seem to be achievable for the relevant waste matrices (sludges and ashes) with regular laboratory equipment. The critical step in analysis of contaminants is the extraction of the analyte from the sample matrix and following clean-up procedure, where different matrices may require different methodological

approaches. Against this background, achievable LODs for waste matrices might be higher than for other matrices like water or biological samples, which have to be taken into account when defining a LOD for sludge or ash. Nevertheless, the above reported levels seem to be achievable for the relevant waste matrices (sludges and ashes) with regular laboratory equipment as sludges and ashes are considered to show comparable characteristics as sediment or soil samples during sample extraction. From this point of view, the available techniques are sufficient to allow analytical control of limit values.

Reported methods achieve LOQs in the range of 0.05 µg/L in water, 0.1 µg/kg (ppb) in fat samples and around 0.5 µg/kg dry weight (ppb) for soil samples. A realistic LOQ achievable with standard laboratory equipment without unreasonable analytical effort when following reported literature methods using CG-MS for waste matrices is assumed to be 50 µg/kg (ppb).

PCN

Unintentional production during various thermal processes have been identified as the most relevant sources of PCN discharges to waste. As PCNs exhibit similar formation properties as PCDD/Fs, unintentional production during incineration processes as municipal solid waste incineration, hospital waste incineration, domestic burning, or different metal processing steps such as secondary copper production, secondary aluminium production, (former) magnesium production as well as iron sintering and electrical arc furnace processes for iron production are of relevance. Further, the accumulation in sewage sludge from diffuse sources has been identified and taken into account. Therefore, ashes from incineration processes (municipal solid waste incineration, hospital waste incineration and domestic burning), sludges from waste water treatment, slag from secondary copper production as well as dust (filter dust, similar to ashes) from secondary aluminium production have been considered as the most relevant PCN containing waste matrices.

The literature review about analytical methods for PCN analysis in matrices with properties comparable to relevant waste matrices lead to the conclusion that standardised procedures (methods) are not available but scientific literature provides procedures suitable to detect PCN in different matrices. Although not for all methods LODs have been reported it has been shown that GC-MS systems in combination with adequate sample extraction and sample clean up is suitable to detect PCN in matrices like sediment or sludge down to low ng/kg (ppb) values. There are few reports regarding PCN contents in sediments available. Bogdal et al reported LODs for PCN congeners in sediments between 0.002-0.008 ng/g DW (ppm) [Bogdal et al., 2006]. Lower limit of determination of PCN between 2 and 10 ng/kg (ppb) for each PCN congener have been reported for soil samples [Krauss and Wilcke, 2003]. Available methods are able to detect PCN in sediment and soil at levels down to low ng/kg values depending on the analytical effort applied. The critical step in analysis of PCN is the extraction of the analyte from the sample matrix and following clean-up procedure, where different matrices may require different methodological approaches. Against this background, achievable LODs for waste matrices might be higher than for other matrices like water or biological samples, which have to be taken into account when defining a LOD for sludge or ash. Nevertheless, the above reported levels seem to be achievable for the relevant waste matrices (sludges and ashes) with regular laboratory equipment as sludges and ashes are considered to show comparable characteristics as sediment or soil samples during sample

extraction. From this point of view, the available techniques are sufficient to allow analytical control of limit values.

Reported methods achieve LODs in the range of 10 ppb in sediment and soils. A realistic LOQ achievable with standard laboratory equipment without unreasonable analytical effort when following reported literature methods using CG-MS for waste matrices is assumed to be 100 ppb or 0.1 ppm.

Results for criterion B: Environmental background contamination

For the derivation of limitation criterion B, soil has been chosen as the corresponding environmental compartment as it resembles best by property to solid residues. Hot spot levels are not included in the specification of the background level but rural and urban background is contained where possible. Furthermore, the data background is rather different for the different substances.

In order to consider the limited data availability, an uncertainty factor of 10 has been applied if not specified otherwise. For particular substances, where the available representative data background is particularly limited, the uncertainty factor can be adjusted. This shall enable to express the uncertainty related to the availability of representative data and shall take into account the likelihood that common background levels may be exceeded.

Table 9-2: Results for limitation criterion B

Substance	Highest reported background contamination	Background concentration	Uncertainty factor	Results for criterion B	
TetraBDE		< 0.1 mg/kg	10	1 mg/kg	1 ppm
PentaBDE	51.4 µg/kg	< 0.1 mg/kg	10	1 mg/kg	1 ppm
HexaBDE		< 0.1 mg/kg	10	1 mg/kg	1 ppm
HeptaBDE		< 0.1 mg/kg	10	1 mg/kg	1 ppm
C-octaBDE sum BDEs 47, 99, 100, 153 and 154	22 µg/kg*	< 0.1 mg/kg	10	1 mg/kg	1 ppm
PFOS	50 µg/kg	< 0.1 mg /kg	10	1 mg/kg	1 ppm
PeCB	76 µg/kg	< 0.1 mg/kg	10	1 mg/kg	1 ppm
SCCP	484 µg/kg	< 1 mg/kg	10	10 mg/kg	10 ppm
HBCD	71 µg/kg	< 10 µg/kg	10	100 µg/kg	100 ppb
PCP	4 mg/kg	< 0.1 mg/kg	10	1 mg/kg	1 ppm
HCBd	550 µg/kg	< 0.1 mg/kg	10	1 mg/kg	1 ppm
PCN	8,640 µg/kg	< 1 mg/kg	10	10 mg/kg	10 ppm

* Concentration of C-OctaBDE

Results for C-PentaBDE

As shown in Table 6-10 the detected (background) contamination levels in sediment and soil for PentaBDE (99) range from less than 0.08 ng/g in Western Europe to 51.4 ng/g in Sweden with a background level in soil of 3 ng/g. Hot spot levels were detected in sediment, soil, sludge and sewage sludges from waste water treatment plants. The samples for the hot spot levels were taken from areas with high pollution e.g. due to recycling activities of e-waste. However, it has to be noted that in general total BDEs or just PBDE was indicated in the context of hot spot levels. Therefore also other congeners as DecaBDE might be the main congener for the relevant studies. This was at least stated for the study of [Leung et al. 2007].

Limit values for PentaBDE in waste should not be below the observed background values in the environment. As the highest congener specific contamination in Europe was 51.4 ng/g (= µg/kg = ppb) and comparable results from the USA are in a similar dimension, it is assumed that 100 µg/kg DW generally will not be exceeded under conditions excluding specific contamination with chemicals. Taking into consideration the relatively large quantity of existing monitoring data, an uncertainty factor of 10 is applied. Therefore, a minimum concentration limit value should not be below 1000 µg/kg DW or 1ppm DW.

Results for C-OctaBDE

As shown in Table 6-11 the detected background contamination levels in sediment and soil ranged from 0.065 µg/kg to 12.0 µg/kg DW in latitudinal transect through soils in UK and Norway. In this context the median of the sum BDEs 47, 99, 100, 153 and 154 (dominating the total concentrations) ranged from 0.44 to 1.8 µg/kg DW. In sediment samples highest background level were 22 µg/kg in Japan.

Hot spot levels of C-OctaBDE were detected inter alia in UK and Spain in sediment, soil and sewage sludges. The hot spot levels reached up to 3,030 µg/kg DW in sediment in UK. In the related study, the highest levels were detected in sediments downstream from a warehouse where C-DecaBDE was stored. All reported contamination however did not relate to single congeners but to the commercial products, so that results cannot be used directly.

Limit values for OctaBDE in waste should not be below the observed background values in the environment. As the highest detected background contamination value amounts to 22 µg/kg, it is assumed that 100 µg/kg DW generally will not be exceeded under conditions excluding specific contamination with chemicals. Taking into consideration the relatively large quantity of existing monitoring data, an uncertainty factor of 10 is applied. Therefore, a minimum concentration limit value should not be below 1,000 µg/kg DW or 1ppm DW.

Results for PFOS

As shown in Table 6-14, in the environment, the detected background contamination levels of PFOS in sediment ranged from less than 0.03 µg/kg to 1 µg/kg in marine sediments in Europe and from ND - 53.1 µg/kg in lake sediments in Canada. In general the determined concentration levels in sediment were below 1 µg/kg DW. The highest hot spot levels were detected in sewage sludges (maximum levels of 1,900 µg/kg and 6,300 µg/kg) and in PFOS contaminated soils after using fire fighting foams in Düsseldorf (Germany) reaching 6,410 µg/kg.

Limit values for PFOS in waste should not be below the observed maximum values in the environment. As the highest limit value detected in Canada was 50 µg/kg DW, it is assumed that 100 µg/kg DW generally will not be exceeded under conditions excluding specific contamination with chemicals. Taking into consideration the relatively large quantity of existing monitoring data and on the basis of the above given additional background information, an uncertainty factor of 10 is applied. Therefore, a minimum concentration limit value should not be below 1,000 µg/kg DW or 1 ppm DW.

Results for PeCB

As shown in Table 6-32, in the environment, the detected (background) contamination levels for PeCB in sediments range from less than 0.01 µg/kg in remote arctic regions to 76 µg/kg in Europe (Germany). Background soil levels range from 0.4-1.3 µg/kg DW. Hot spot levels were detected inter alia in the Danube Region in sediment and ranged up to 3,500 µg/kg. From records of PeCB polluted sites in the US the maximum concentration levels reach up to 5,100 mg/kg in sediment and up to 2,040 mg/kg in soil.

Limit values for PeCB in waste should not be below the observed maximum values in the environment. As the highest limit value in Europe was 76 µg/kg, it is assumed that 100 µg/kg DW generally will not be exceeded under conditions excluding specific contamination with chemicals. Taking into consideration the relatively large quantity of existing monitoring data, an uncertainty factor of 10 is applied. Therefore, a minimum concentration limit value should not be below 1000 µg/kg DW or 1ppm DW.

Results for SCCP

The detected background contamination levels of SCCP in sediment ranged from the LOD to a maximum concentration level of 484 µg/kg (0.484 ppm) [POPRC 6/11] DW in sediments determined in Japan. In general the determined concentration levels in sediment were below 300 µg/kg (0.3 ppm) DW. The highest hot spot levels were detected in sewage sludge (maximum levels of 200,000 µg/kg (200 ppm) [POPRC 6/11]) and in SCCP contaminated soils in the UK reaching to 10,300 µg/kg (10.3 ppm) [POPRC 6/11]. In comparison to river and lake sediments concentrations, SCCP contents in sewage sludge are much higher, especially from WWTPs serving industrial areas. It is reported that the high differences of the values between 500 and 80,000 µg/kg (0.5 and 80 ppm) from Switzerland seems to be due to different shares of waste water from industries as metal working industry. It is assumed that especially the improper disposal of lube oil might be a reason for the higher concentrations. It should be considered that these measurements took place in the 80's, at which time the use of SCCP was much higher [BUWAL 2003].

To conclude, background levels in soil and sediment are expected to be usually below 1 ppm and are not expected to exceed levels of 10 ppm.

Results for HBCD

The detected European background contamination levels in sediment are usually below 10 ppb (see results from Klamer et al. 2005, Kohler et al. 2007, Kohler et al. 2008, Remberger et al. 2004, Schlabach et al. 2004 a, b) but may range up to 71 ppb in areas with known high background contamination (see results from Verslycke et al. 2005 from analyses of sediment from the Scheldt estuary, one of the largest and most polluted estuaries in Western Europe). Background levels for soil are not available for Europe. Reported concentrations from Asia ranged from 1.7 to 5.6 ppb (see results from Yu et al. 2008).

Results for PCP

The data which is mainly based on sediment monitoring in the North Sea and of rivers which flow into the North Sea, indicate a typical PCP background concentration of Sediment between 5 to 30 µg/kg DW (0.005 to 0.030 ppm). It also shows that sea waters close to discharges are generally higher contaminated.

The data on soil in Finland of 1992 is based on 154 soil contamination cases, which were gathered in a Finnish register by environmental authorities [OSPAR 2001]. Two thirds of the contamination cases occurred at sawmills and 32 of the cases were in a groundwater area. The collected data shows a ratio of heavily contaminated to less contaminated soils of approximately 1:20.

In a German study a comparison of sewage sludges from two test series at the end of the eighties and the middle of nineties has been made. According to the explanations in the study, the sludges of Rheinland-Pfalz used in the second study would be representative for the rest of Germany. A decreasing trend to lower PCP concentrations in sewage sludges in Germany is explained as an effect of a ban in 1989. However it is also stipulated that using the Wilcoxon Test, the two measurement series are only different to a probability of 80 %. [Jobst 1998] Regarding this study it is assumed that PCP concentrations of sewage sludges are below 1 ppm Europe wide.

Results for HCBd

Most of the monitoring data on HCBd contamination levels in sediment and soil exists for Canada and the United States. As shown in Table 6-32, the detected contamination levels in sediments ranged from 0.01 µg/kg (measured in sediment samples taken from the Great Slave Lake), up to 550 µg/kg measured in older sediment layers in the Great Lakes area. In the United States, levels of HCBd have been reported between 0.05-2.0 µg/kg (see Table 6-32).

HCBd was found in surface sediment samples at a level of a few µg/kg in the most polluted locations along the Kaohsiung coast in southern Taiwan (see Table 6-32).

For Europe, statistical analysis of about 500 measured concentrations indicated that the mean and the

90th percentile values of HCBd in sediments are 1.1 and 4 µg/kg, respectively. Recent measurements indicate that concentrations in estuarine or coastal sediment vary between <0.2 and 3 µg/kg, with typical values close to 1 µg/kg. Specific data for Germany shows HCBd concentration values between <0.1-1.8 µg/kg, while a study conducted from 1980 to 1981 reported levels of 2-5 µg/kg in sediments from the Rhine. Even lower concentration figures from the Rhine River were reported (0.002-0.005 µg/kg). Measurements in sediments collected from the Rhine-Meuse river basin indicate HCBd levels ranging from <3.4-19 µg/kg (see Table 6-32).

Several hot spot levels were detected in the United States which ranged from 2 mg/kg-82 mg/kg in sediment samples from the Baton Rouge and Lake Charles areas. The highest concentration measured in 1994 in sediment samples from the St Clair River in Canada was 310 mg/kg. Sediments samples taken prior to 1986 show HCBd concentrations up to 430 mg/kg (see Table 6-32).

In the only identified relevant survey of soils in Canada, HCBd was neither detected (detection limit 0.05 µg/g DW) in 24 samples of agricultural soils across the country, nor in six samples from areas that had repeatedly received heavy applications of pesticides (see Table 6-32).

Results for PCN

As shown in Table 6-32, the detected contamination levels in sediment ranged from 0.03 µg/kg measured in Elbe River sediments to >1,000 µg/kg in Germany and Norway at contaminated sites (e.g. PCB polluted areas and near chlor-alkali industries). 'Hot spot' levels of up to 104,000 µg/kg were detected in the United States. Highest contamination levels in Europe were identified in Sweden (270 µg/kg), Germany (880-1,120 µg/kg) and Norway (1,000 µg/kg) at polluted sites. According to IPCS (2001), sediment levels of up to 100 mg/kg (100,000 µg/kg) have been recorded in the past, however, recent results show levels of 0.2 µg/kg at unpolluted sites and 250 µg/kg at polluted sites.

The detected contamination levels in soil ranged from <0.1 µg/kg in urban and rural soils in Germany to approximately 1,300,000 µg/kg in soil samples taken from areas in the Netherlands that have been used for municipal waste disposal. 'Hot spot' levels were identified in the Netherlands and the United States. According to IPCS (2001), soil levels of up to 1,300 mg/kg (1,300,000 µg/kg) were measured at contaminated sites in the early 1980s, compared with a more recent value for a former chlor-alkali plant of 18 mg/kg (18,000 µg/kg) dry weight.

Following the reported data the following conclusions can be drawn for lower limitation criterion B:

PBDEs (single congener groups):	1 ppm
PFOS:	1 ppm
PeCB:	1 ppm
SCCP:	10 ppm

HBBD:	100 ppb
PCP:	1 ppm
HCBD:	1 ppm
PCN:	10 ppm

Results for criterion C: Disposal/recovery capacities

The applied low POP concentration limit value highly influences the quantities of wastes to be managed as POP wastes. The quantities of waste which have to be destroyed or irreversibly transformed applying the disposal and recovery operations listed in Annex V part 1 and 2 of the POP regulation will increase, depending on the strictness of the applied low POP concentration limits.

Therefore, the future amount of POP wastes has to be estimated and the remaining capacities for accepted disposal and recovery operation have to be checked throughout EU 27 to assess possibly future shortage of waste management capacities. Additionally, factors like infrastructure (e.g. transport) and administrative capacities might also cause further shortage. However, those factors can only be discussed, but cannot be estimated within the assessment.

- Estimation of quantities of future generation of POP waste

Based on the information derived from the waste flows (see chapter 6) and the scenarios in chapter 7, the expected quantities of waste to be dealt with as POP waste is estimated in dependence of the applied low POP concentration limits. The correlation of low POP concentration limits with estimated quantities of POP waste is given in Table 9-3 for the relevant substances.

Table 9-3: Correlation of low POP concentration limit quantities of POP waste

Substance	Expected quantity of waste (kt/y) classified as POP waste at low POP concentration level of:					
C-PentaBDE	0.1 ppm	1 ppm	10 ppm	50 ppm	100 ppm	1,000 ppm
	5,871	5,871	5,871	656	656	-
C-OctaBDE	0.1 ppm	1 ppm	10 ppm	50 ppm	100 ppm	1,000 ppm
	105 (0.732*)	105 (0.732*)	105 (0.732*)	105 (0.732*)	105 (0.732*)	105 (0.732*)
TetraBDE	0.05 ppm	1 ppm	10 ppm	50 ppm	100 ppm	1,000 ppm
	5,871	5,871	656	656	656	-
PentaBDE	0.05 ppm	1 ppm	10 ppm	50 ppm	100 ppm	1,000 ppm
	5,976 (5,872*)	5,976 (5,872*)	761 (657*)	761 (657*)	657*	0.732*
HexaBDE	0.05 ppm	1 ppm	10 ppm	50 ppm	100 ppm	1,000 ppm
	5,976 (5,872*)	5,976 (5,872*)	761 (657*)	105 (0.732*)	0.732*	0.732*
HeptaBDE	0.05 ppm	1 ppm	10 ppm	50 ppm	100 ppm	1,000 ppm
	5,976 (5,872*)	761 (657*)	105 (0.732*)	105 (0.732*)	105 (0.732*)	0.732*
PFOS	0.05 ppm	1 ppm	10 ppm	50 ppm	100 ppm	1,000 ppm
	16,211	5,212	1,972	1,941	3.1	1.4
PeCB	0.1 ppb	1.0 ppb	3.0 ppb		5.0 ppb	50 ppb
	80,777	80,366	13,203		1,625	1,153
SCCP	1 ppm	10 ppm	100 ppm	1,000 ppm	10,000 ppm	100,000 ppm
	11,621	622	42,8	42,8	38,4	3,99
HBCD	14 ppm	5,100 ppm	13,500 ppm	20,000 ppm	40,000 ppm	80,000 ppm
	6,479,402	20,846	14,520	14,324	12,739	9,289
PCP	-	-	500 ppm	1000 ppm	10,000 ppm	50,000 ppm
	-	-	2,950,000	9,568	9,568	0
HCBd	0.1 ppb	1 ppb	1 ppm	1,000 ppm	10,000 ppm	100,000 ppm
[t]	> 9,816	n.a.	n.a.	n.a.	n.a.	n.a.
PCN	1 ppb	10 ppb	50 ppb	4,000 ppb	5,000 ppb	-
[kt]	27,062.50	13,168.48	2,397.02	626.02	12.22	-

Chapter 6 describes in detail the waste management options currently performed for the products and processes containing the substances of concern. In short, the disposal and recovery options and quantities of the current waste management practice are summarised in Table 9-4.

Table 9-4: Overview of current waste management options (status quo)

Substance	Disposal/ Recovery option	Estimated quantities (t/y)
C-PentaBDE	Hazardous landfill	0
	Non-hazardous landfill	3,765,256
	Inert landfill	0
	Hazardous waste incineration	0
	Non-hazardous waste incineration	1,319,168
	Energy recovery	708,285
	Substance recovery	78,049
C-OctaBDE	Hazardous landfill	0
	Non-hazardous landfill	32,242
	Inert landfill	0
	Hazardous waste incineration	0
	Non-hazardous waste incineration	32,242 (366*)
	Energy recovery	20,971 (366*)
	Substance recovery	18,874
TetraBDE	Hazardous landfill	0
	Non-hazardous landfill	3,765,256
	Inert landfill	0
	Hazardous waste incineration	0
	Non-hazardous waste incineration	1,319,168
	Energy recovery	708,285
	Substance recovery	78,049
PentaBDE/ HexaBDE/ HeptaBDE	Hazardous landfill	0
	Non-hazardous landfill	3,797,498
	Inert landfill	0
	Hazardous waste incineration	0
	Non-hazardous waste incineration	1,351,410 (1,319,534*)
	Energy recovery	729,256 (729,601*)
	Substance recovery	96,923
PFOS	Hazardous landfill	0.00
	Non-hazardous landfill	4,519,217
	Inert landfill	0.00
	Hazardous waste incineration	9,602
	Non-hazardous waste incineration	3,946,045
	Energy recovery	8,055,130
	Substance recovery	0.00
PeCB	Hazardous landfill	2,881,641
	Non-hazardous landfill	3,263,546
	Inert landfill	0
	Hazardous waste incineration	0
	Non-hazardous waste incineration	2,373,884
	Energy recovery	63,622,928
	Substance recovery	0
SCCP	Others	8,635,316
	Hazardous landfill	0
	Non-hazardous landfill	1,831
	Inert landfill	0
	Hazardous waste incineration	0.015
	Non-hazardous waste incineration	2,015

Substance	Disposal/ Recovery option	Estimated quantities (t/y)
	Energy recovery	5.36
	Substance recovery	7,766
HBCD	Hazardous landfill	0
	Non-hazardous landfill	18,509
	Inert landfill	0
	Hazardous waste incineration	0
	Non-hazardous waste incineration	7,013
	Energy recovery	5,070
	Substance recovery	2,337
PCP	Hazardous landfill	0
	Non-hazardous landfill	6,420
	Inert landfill	0
	Hazardous waste incineration	2,941,066
	Non-hazardous waste incineration	1,933
	Energy recovery	1,215
	Substance recovery	0
HCBD	Hazardous landfill	0
	Non-hazardous landfill	0
	Inert landfill	1,430,000
	Hazardous waste incineration	0
	Non-hazardous waste incineration	0
	Energy recovery	1,891,000
	Substance recovery	0
PCNs	Hazardous landfill	5,299,867
	Non-hazardous landfill	2,664,599
	Inert landfill	0
	Hazardous waste incineration	0
	Non-hazardous waste incineration	2,373,884
	Energy recovery	0
	Substance recovery	14,078,306
	Others	2,645,848

*Scenario 2: in case C-OctaBDE contaminated plastics can be successfully separated from the non-contaminated WEEE plastic fraction (category 3 and 4)

Depending on selected LPCL in general only waste incineration D10, energy recovery R1, physico-chemical treatment (D9) and metal reclamation (R4) will be possible for all wastes exceeding the set limits. Only certain ashes or (slags and C&D wastes not relevant for new POPs) may be authorised for other waste operations according to Annex V, part 2, if environmentally preferable.

Change of the status quo for PBDE

By introduction of a specific LPCL, PBDE containing wastes (ABS plastics in WEEE and PUR foams in ELV) which are currently landfilled and recycled would need to be sent to incineration or energy recovery if exceeding the set LPCL. Based on current treatment shares the amounts as compiled in Table 9-5 to Table 9-8 can be expected to arise until all existing stocks of the relevant PBDEs in EEE and used cars have been eliminated.

Table 9-5: Overview of new status quo on waste amounts for specific LPCL for TetraBDE

TetraBDE	Status Quo [kt/y]	New Status Quo [kt/y]	Net change [kt]	Net change [%]
100.0 ppm				

TetraBDE	Status Quo [kt/y]	New Status Quo [kt/y]	Net change [kt]	Net change [%]
Hazardous landfill	0	0	0	0
Non-hazardous landfill	3,765	3,500	-265	-7
Inert landfill	0	0	0	0
Hazardous waste incineration	0	0	0	0
Non-hazardous waste incineration	1,319	1,662	343	26
Energy recovery	708	708	0	0
Substance recovery	78	0	-78	-100
Others (temporarily stored or not specified)	0	0	0	0
50.0 ppm				
Hazardous landfill	0	0	0	0
Non-hazardous landfill	3,765	3,500	-265	-7
Inert landfill	0	0	0	0
Hazardous waste incineration	0	0	0	0
Non-hazardous waste incineration	1,319	1,662	343	26
Energy recovery	708	708	0	0
Substance recovery	78	0	-78	-100
Others (temporarily stored or not specified)	0	0	0	0
10.0 ppm				
Hazardous landfill	0	0	0	0
Non-hazardous landfill	3,765	3,500	-265	-7
Inert landfill	0	0	0	0
Hazardous waste incineration	0	0	0	0
Non-hazardous waste incineration	1,319	1,662	343	26
Energy recovery	708	708	0	0
Substance recovery	78	0	-78	-100
Others (temporarily stored or not specified)	0	0	0	0
1.0 ppm				
Hazardous landfill	0	0	0	0
Non-hazardous landfill	3,765	0	-3,765	-100
Inert landfill	0	0	0	0
Hazardous waste incineration	0	0	0	0
Non-hazardous waste incineration	1,319	5162	3,843	291
Energy recovery	708	708	0	0
Substance recovery	78	0	-78	-100
Others (temporarily stored or not specified)	0	0	0	0

Table 9-6: Overview of new status quo on waste amounts for specific LPCL for PentaBDE

PentaBDE	Status Quo [kt/y]	New Status Quo [kt/y]	Net change [kt]	Net change [%]
100.0 ppm				
Hazardous landfill	0	0	0	0
Non-hazardous landfill	3,797	3,532	-265	-7
Inert landfill	0	0	0	0
Hazardous waste incineration	0	0	0	0
Non-hazardous waste incineration	1,351	1,694	343	25.4
Energy recovery	729	729	0	0
Substance recovery	97	19	-78	-81
Others (temporarily stored or not specified)	0	0	0	0
50.0 ppm				
Hazardous landfill	0	0	0	0
Non-hazardous landfill	3,797	3,500	-297	-8

PentaBDE	Status Quo [kt/y]	New Status Quo [kt/y]	Net change [kt]	Net change [%]
Inert landfill	0	0	0	0
Hazardous waste incineration	0	0	0	0
Non-hazardous waste incineration	1,351	1,745	394	29.2
Energy recovery	729	729	0	0
Substance recovery	97	0	-97	-100
Others (temporarily stored or not specified)	0	0	0	0
10.0 ppm				
Hazardous landfill	0	0	0	0
Non-hazardous landfill	3,797	3,500	-297	-8
Inert landfill	0	0	0	0
Hazardous waste incineration	0	0	0	0
Non-hazardous waste incineration	1,351	1,745	394	0
Energy recovery	729	0	0	0
Substance recovery	97	0	-97	-100
Others (temporarily stored or not specified)	0	0	0	0
1.0 ppm				
Hazardous landfill	0	0	0	0
Non-hazardous landfill	3,797		-3,797	-100
Inert landfill	0	0	0	0
Hazardous waste incineration	0	0	0	0
Non-hazardous waste incineration	1,351	6,245	4,894	362
Energy recovery	729	729	0	0
Substance recovery	97	0	-97	-100
Others (temporarily stored or not specified)	0	0	0	0

Table 9-7: Overview of new status quo on waste amounts for specific LPCL for HexaBDE

HexaBDE	Status Quo [kt/y]	New Status Quo [kt/y]	Net change [kt]	Net change [%]
100.0 ppm				
Hazardous landfill	0	0	0	0
Non-hazardous landfill	3,797	3,797	0	0
Inert landfill	0	0	0	0
Hazardous waste incineration	0	0	0	0
Non-hazardous waste incineration	1,351	0	0	0
Energy recovery	729	729	0	0
Substance recovery	97	97	0	0
Others (temporarily stored or not specified)	0	0	0	0
50.0 ppm				
Hazardous landfill	0	0	0	0
Non-hazardous landfill	3,797	3,765	-32	-0.8
Inert landfill	0	0	0	0
Hazardous waste incineration	0	0	0	0
Non-hazardous waste incineration	1,351	1,4020	-51	3.70
Energy recovery	729	729	0	0
Substance recovery	97	78	-19	-20
Others (temporarily stored or not specified)	0	0	0	0
10.0 ppm				
Hazardous landfill	0	0	0	0
Non-hazardous landfill	3,797	3,797	-32	0
Inert landfill	0	0	0	0

HexaBDE	Status Quo [kt/y]	New Status Quo [kt/y]	Net change [kt]	Net change [%]
Hazardous waste incineration	0	0	0	0
Non-hazardous waste incineration	1,351	1,480	129	9.50
Energy recovery	729	729	0	0
Substance recovery	97	0	-97	-100
Others (temporarily stored or not specified)	0	0	0	0
1.0 ppm				
Hazardous landfill	0	0	0	0
Non-hazardous landfill	3,797	0	-3,797	-100
Inert landfill	0	0	0	0
Hazardous waste incineration	0	0	0	0
Non-hazardous waste incineration	1,351	5,245	3,894	288.3
Energy recovery	729	729	0	0
Substance recovery	97	0	-97	-100
Others (temporarily stored or not specified)	0	0	0	0

Table 9-8: Overview of new status quo on waste amounts for specific LPCL for HeptaBDE

HeptaBDE	Status Quo [kt/y]	New Status Quo [kt/y]	Net change [kt]	Net change [%]
100.0 ppm				
Hazardous landfill	0	0	0	0
Non-hazardous landfill	3,797	3,765	-32	-0.8
Inert landfill	0	0	0	0
Hazardous waste incineration	0	0	0	0
Non-hazardous waste incineration	1,351	1,402	51	3.8
Energy recovery	729	729	0	0
Substance recovery	97	78	-19	-20
Others (temporarily stored or not specified)	0	0	0	0
50.0 ppm				
Hazardous landfill	0	0	0	0
Non-hazardous landfill	3,797	3,765	-32	-0.8
Inert landfill	0	0	0	0
Hazardous waste incineration	0	0	0	0
Non-hazardous waste incineration	1,351	1,402	51	3.8
Energy recovery	729	729	0	0
Substance recovery	97	78	-19	-20
Others (temporarily stored or not specified)	0	0	0	0
10.0 ppm				
Hazardous landfill	0	0	0	0
Non-hazardous landfill	3,797	3,765	-32	-0.8
Inert landfill	0	0	0	0
Hazardous waste incineration	0	0	0	0
Non-hazardous waste incineration	1,351	1,402	51	3.8
Energy recovery	729	729	0	0
Substance recovery	97	78	-19	-20
Others (temporarily stored or not specified)	0	0	0	0
1.0 ppm				
Hazardous landfill	0	0	0	0
Non-hazardous landfill	3,797	3,500	-297	-8
Inert landfill	0	0	0	0
Hazardous waste incineration	0	0	0	0
Non-hazardous waste incineration	1,351	1,745	394	29

HeptaBDE	Status Quo [kt/y]	New Status Quo [kt/y]	Net change [kt]	Net change [%]
Energy recovery	729	729	0	0
Substance recovery	97	0	-97	-100
Others (temporarily stored or not specified)	0	0	0	0

Change of the status quo for PFOS

In Table 9-9 the current situation on waste amounts contaminated with PFOS is described. By introduction of a specific LPCL, changes in waste amounts destined for a specific treatment option will occur since treatment options originally applied will not be available anymore and affected waste amounts considered as POP waste will have to be subjected to other appropriate treatment options (D9, D10, R1, R4). In the following table an overview of these shifts of waste amounts considered as POP waste due to introduction of a specific LPCL is given. The LPCL values, which have been selected as relevant for PFOS, lie in the ppb range and are 0.05, 1.0, 10.0, 50.0, 100 and 1,000 ppb.

Table 9-9: Overview of new status quo on waste amounts for specific LPCL for PFOS

PFOS	Status Quo [kt/y]	New Status Quo [kt/y]	Net change [kt]	Net change [%]
50.0 ppm				
Hazardous landfill	0.00	0.00	0.00	0.00
Non-hazardous landfill	4,840.37	3,539.96	-1,300.41	-26.87
Inert landfill	0.00	0.00	0.00	0.00
Hazardous waste incineration	9.60	1,310.01	1,300.41	13,542.12
Non-hazardous waste incineration	3,624.89	3,624.89	0.00	0.00
Energy recovery	101.88	101.88	0.00	0.00
Substance recovery	7,766.05	7,766.05	0.00	0.00
Others (temporarily stored or not specified)	26,470.37	26,470.37	0.00	0.00
10.0 ppm				
Hazardous landfill	0.00	0.00	0.00	0.00%
Non-hazardous landfill	4,840.37	3,524.39	-1,315.980	-27.19%
Inert landfill	0.00	0.00	0.00	0.00%
Hazardous waste incineration	9.60	1,325.58	1,315.98	13704.27%
Non-hazardous waste incineration	3,624.89	3,624.89	0.00	0.00%
Energy recovery	101.88	101.88	0.00	0.00%
Substance recovery	7,766.05	7,766.05	0.00	0.00%
Others (temporarily stored or not specified)	26,470.37	26,470.37	0.00	0.00%
1.0 ppm				
Hazardous landfill	0.00	0.00	0.00	0.00%
Non-hazardous landfill	4,840.37	1,714.97	-3,125.40	-64.57%
Inert landfill	0.00	0.00	0.00	0.00%
Hazardous waste incineration	9.60	3,523.31	3,513.71	36590.82%
Non-hazardous waste incineration	3,624.89	3,624.89	0.00	0.00%
Energy recovery	101.88	101.88	0.00	0.00%
Substance recovery	7,766.05	7,377.75	-388.303	-5.00%
Others (temporarily stored or not specified)	26,470.37	26,470.37	0.00	0.00%
0.05 ppm				
Hazardous landfill	0.00	0.00	0.00	0.00%
Non-hazardous landfill	4,840.37	0.00	-4,840.373	-100.00%
Inert landfill	0.00	0.00	0.00	0.00%
Hazardous waste incineration	9.60	12,616.03	12,606.42	131279.96%
Non-hazardous waste incineration	3,624.89	3,624.89	0.00	0.00%
Energy recovery	101.88	101.88	0.00	0.00%
Substance recovery	7,766.05	0.00	-7,766.05	-100.00%
Others (temporarily stored or not specified)	26,470.37	26,470.37	0.00	0.00%

An increase of amounts of PFOS contaminated wastes to be treated under a different treatment option can be expected for hazardous waste incineration, connected with reduction in waste amounts destined for non-hazardous landfill and other treatment options.

Change of the status quo for PeCB

In Table 9-10 the current situation on waste amounts contaminated with PeCB is described. By introduction of a specific LPCL, changes in waste amounts destined for a specific treatment option will occur since treatment options originally applied will not be available anymore and affected waste amounts considered as POP waste will have to be subjected to other appropriate treatment options (D9, D10, R1, R4). In the following table an overview of these shifts of waste amounts considered as POP waste due to introduction of a specific LPCL is given. The LPCL values, which have been selected as relevant for PeCB, lie in the ppb range and are 0.1, 1.0, 3.0, 5.0 and 50 ppb.

Table 9-10: Overview of new status quo on waste amounts for specific LPCL for PeCB

PeCB	Status Quo [kt/y]	New Status Quo [kt/y]	Net change [kt]	Net change [%]
50.0 ppb				
Hazardous landfill	2,881.64	2,926.13	44.49	1.54
Non-hazardous landfill	3,263.55	3,254.78	-8.77	-0.27
Inert landfill	0.00	0.00	0.00	0.00
Hazardous waste incineration	0.00	0.00	0.00	0.00
Non-hazardous waste incineration	2,373.88	2,373.88	0.00	0.00
Energy recovery	63,622.93	63,595.97	-26.96	-0.04
Substance recovery	0.00	0.00	0.00	0.00
Others (temporarily stored or not specified)	8,635.32	8,626.55	-8.77	-0.10
5.0 ppb				
Hazardous landfill	2,881.64	2,939.17	57.53	2.00
Non-hazardous landfill	3,263.55	3,254.78	-8.77	-0.27
Inert landfill	0.00	0.00	0.00	0.00
Hazardous waste incineration	0.00	0.00	0.00	0.00
Non-hazardous waste incineration	2,373.88	2,373.88	0.00	0.00
Energy recovery	63,622.93	63,582.93	-40.00	-0.06
Substance recovery	0.00	0.00	0.00	0.00
Others (temporarily stored or not specified)	8,635.32	8,626.55	-8.77	-0.10
3.0 ppb				
Hazardous landfill	2,881.64	2,939.17	57.53	2.00
Non-hazardous landfill	3,263.55	1,274.67	-1,988.88	-60.94
Inert landfill	0.00	0.00	0.00	0.00
Hazardous waste incineration	0.00	9,546.36	9,546.36	954635.87
Non-hazardous waste incineration	2,373.88	2,373.88	0.00	0.00
Energy recovery	63,622.93	57,978.04	-5,644.88	-8.87
Substance recovery	0.00	0.00	0.00	0.00
Others (temporarily stored or not specified)	8,635.32	6,665.19	-1,970.13	-22.81
1.0 ppb				
Hazardous landfill	2,881.64	68,528.03	65,646.39	2278.09
Non-hazardous landfill	3,263.55	164.52	-3,099.02	-94.96
Inert landfill	0.00	0.00	0.00	0.00
Hazardous waste incineration	0.00	9,546.36	9,546.36	954635.87
Non-hazardous waste incineration	2,373.88	2,373.88	0.00	0.00
Energy recovery	63,622.93	0.00	-63,622.93	-100.00
Substance recovery	0.00	0.00	0.00	0.00

PeCB	Status Quo [kt/y]	New Status Quo [kt/y]	Net change [kt]	Net change [%]
Others (temporarily stored or not specified)	8,635.32	164.52	-8,470.79	-98.09
0.1 ppb				
Hazardous landfill	2,881.64	68,857.07	65,975.43	2289.51
Non-hazardous landfill	3,263.55	0.00	-3,263.55	-100.00
Inert landfill	0.00	0.00	0.00	0.00
Hazardous waste incineration	0.00	9,546.36	9,546.36	954635.87
Non-hazardous waste incineration	2,373.88	2,373.88	0.00	0.00
Energy recovery	63,622.93	0.00	-63,622.93	-100.00
Substance recovery	0.00	0.00	0.00	0.00
Others (temporarily stored or not specified)	8,635.32	0.00	-8,635.32	-100.00

An increase of amounts of PeCB contaminated wastes to be treated under a different treatment option for a specific LPCL set can be expected for hazardous waste incineration as well as hazardous landfilling. This is connected with reduction in waste amounts destined for non-hazardous landfill and other treatment options.

Change of the status quo for SCCP

In Table 9-9 the current situation on waste amounts contaminated with SCCP is described. By introduction of a specific LPCL, changes in waste amounts destined for a specific treatment option will occur since treatment options originally applied will not be available anymore and affected waste amounts considered as POP waste will have to be subjected to other appropriate treatment options (D9, D10, R1, R4). In the following table an overview of these shifts of waste amounts considered as POP waste due to introduction of a specific LPCL is given. The LPCL values, which have been selected as relevant for SCCP, lie in the ppm range and are 1, 10, 1,000, 10,000 and 100,000 ppm.

Table 9-11: Overview of new status quo on waste amounts for specific LPCL for SCCP

SCCP	Status Quo [kt/y]	New Status Quo [kt/y]	Net change [kt]	Net change [%]
100,000 ppm				
Hazardous landfill	0	0	0	0
Non-hazardous landfill	1,835	1,831	-3.6	0.2
Inert landfill	0	0	0	0
Hazardous waste incineration	0.02	0.02	0	0
Non-hazardous waste incineration	2,018	2,021	3.6	0
Energy recovery	6	6	0	0
Substance recovery	7,766	7,766	0	0
Others (temporarily stored or not specified)	716,706	716,706	0	0
10,000 ppm				
Hazardous landfill	0	0	0	0
Non-hazardous landfill	1,835	1,808	-27	-1.45
Inert landfill	0	0	0	0
Hazardous waste incineration	0.02	0.02	0	0
Non-hazardous waste incineration	2,018	2,044	27	1
Energy recovery	6	6	0	0
Substance recovery	7,766	7,766	0	0
Others (temporarily stored or not specified)	716,706	716,706	0	0
1,000 ppm				
Hazardous landfill	0	0	0	0
Non-hazardous landfill	1,835	1,805	-30	-1.61

SCCP	Status Quo [kt/y]	New Status Quo [kt/y]	Net change [kt]	Net change [%]
Inert landfill	0	0	0	0
Hazardous waste incineration	0.02	0.02	0	0
Non-hazardous waste incineration	2,018	2,047	30	1.5
Energy recovery	6	6	0	0
Substance recovery	7,766	7,766	0	0
Others (temporarily stored or not specified)	716,706	716,706	0	0
10 ppm				
Hazardous landfill	0	0	0	0
Non-hazardous landfill	1,835	1,715	-120	-6.53
Inert landfill	0	0	0	0
Hazardous waste incineration	0.02	0.02	0	0
Non-hazardous waste incineration	2,018	2,526	508	25.2
Energy recovery	6	6	0	0
Substance recovery	7,766	7,378	-388	-4.9
Others (temporarily stored or not specified)	716,706	716,706	0	0
1.0 ppm				
Hazardous landfill	0	0	0	0
Non-hazardous landfill	1,835	0	-1,835	-100
Inert landfill	0	0	0	0
Hazardous waste incineration	0.02	0	0	0
Non-hazardous waste incineration	2,018	11,619	9,601	4758
Energy recovery	6	6	0	0
Substance recovery	7,766	0	-7,766	-100
Others (temporarily stored or not specified)	716,706	716,706	0	0

An increase of amounts of SCCP contaminated wastes to be treated under a different treatment option for a specific LPCL set can be expected for hazardous waste incineration as well as hazardous landfilling. This is connected with reduction in waste amounts destined for non-hazardous landfill and other treatment options.

Change of the status quo for HBCD

By introduction of a specific LPCL, HBCD containing wastes which are currently landfilled and recycled would need to be sent to incineration or energy recovery if exceeding the set LPCL. Based on current treatment shares the amounts as compiled in Table 9-12 can be expected to arise.

Table 9-12: Overview of new status quo on waste amounts for specific LPCL for HBCD

HBCD	Status Quo [kt/y]	New Status Quo [kt/y]	Net change [kt]	Net change [%]
40,000 ppm				
Hazardous landfill	0	0	0	0
Non-hazardous landfill	18	7	-11	-61
Inert landfill	0	0	0	0
Hazardous waste incineration	0	0	0	0
Non-hazardous waste incineration	7.0	19.2	12.2	1,743
Energy recovery	5.1	5.1	0	0
Substance recovery	2.3	1.1	-1.2	-52
Others (temporarily stored or not specified)	0	0	0	0
5,100 ppm				
Hazardous landfill	0	0	0	0
Non-hazardous landfill	18	0	-18.5	-100

HBCD	Status Quo [kt/y]	New Status Quo [kt/y]	Net change [kt]	Net change [%]
Inert landfill	0	0	0	0
Hazardous waste incineration	0	0	0	0
Non-hazardous waste incineration	7	27.3	20.8	2971
Energy recovery	5	5	0	0
Substance recovery	2.3	0	-2.3	-100
Others (temporarily stored or not specified)	0	0	0	0

Change of the status quo for PCP

In Table 9-9 the current situation on waste amounts contaminated with PCP is described. By introduction of a specific LPCL, changes in waste amounts destined for a specific treatment option will occur since treatment options originally applied will not be available anymore and affected waste amounts considered as POP waste will have to be subjected to other appropriate treatment options (D9, D10, R1, R4). In the following table an overview of these shifts of waste amounts considered as POP waste due to introduction of a specific LPCL is given. The LPCL value, which has been selected as relevant for PCP, is 25,000 ppm. The two industries where PCP containing wastes arise are wood treatment and textiles. Treated wood is already incinerated and is therefore not relevant for this analysis. The remaining PCP containing waste types are textiles, which are the only one which have to be considered in this evaluation. PCP treated textiles normally contain about 2.5 % (25,000 ppm). LPCL levels above this value do not have an effect on them and values below contain all PCP containing wastes which need different treatment. For this reason only one LPCL level is analysed.

Table 9-13: Overview of new status quo on waste amounts for specific LPCL for PCP

PCP	Status Quo [kt/y]	New Status Quo [t/y]	Net change [t]	Net change [%]
25,000.0 ppm				
Non-hazardous landfill	6,420	0	-6,420	-100
Hazardous waste incineration	2,941,066	2,950,634	6,420	0.22
Non-hazardous waste incineration	1,933	1,933	0	0
Energy recovery	1,215	1,215	0	0
Others (temporarily stored or not specified)	9,686,820	9,686,820	0	0

An increase of amounts of PCP contaminated wastes to be treated under a different treatment option for a specific LPCL set can be expected for hazardous waste incineration as well as hazardous landfilling. This is connected with reduction in waste amounts destined for non-hazardous landfill and other treatment options.

Change of the status quo for HCBd

In Table 9-14 the current situation on waste amounts contaminated with HCBd is described. By introduction of a specific LPCL, changes in waste amounts destined for a specific treatment option will occur since treatment options originally applied will not be available anymore and affected waste amounts considered as POP waste will have to be subjected to other appropriate treatment options (D9, D10, R1, R4). In the following table an overview of these shifts of waste amounts considered as POP

waste due to the introduction of a specific LPCL is given. The LPCL value, which has been selected as relevant for HCBd, is 0.6 ppb. It is the only information available for one specific waste stream.

Table 9-14: Overview of new status quo on waste amounts for specific LPCL for HCBd

HCBd	Status Quo [kt/y]	New Status Quo [kt/y]	Net change [kt]	Net change [%]
0.6 ppb				
Landfill	1,430	0	-1,430	-100%
incineration	1,891	9,816	7,925	419%
application to land	5,086	0	-5,086	-100%
Others (temporarily stored or not specified)	1,409	0	-1,409	-100%

Change of the status quo for PCN

In Table 9-15 the current situation on waste amounts contaminated with PCNs is described. By introduction of a specific LPCL, changes in waste amounts destined for a specific treatment option will occur since treatment options originally applied will not be available anymore and affected waste amounts considered as POP waste will have to be subjected to other appropriate treatment options (D9, D10, R1, R4). In the following table an overview of these shifts of waste amounts considered as POP waste due to introduction of a specific LPCL is given. The LPCL values, which have been selected as relevant for PCNs, lie mostly in the ppb range with values of 1 ppb, 10 ppb and 50 ppb (relevant for resulting high volume waste streams), but also higher LPCL values of 5,000 ppb have been taken into consideration.

Table 9-15: Overview of new status quo on waste amounts for specific LPCL for PCNs

PCN	Status Quo [kt/y]	New Status Quo [kt/y]	Net change [kt]	Net change [%]
5,000 ppb				
Hazardous landfill	5,299.87	5,299.89	0.02	0.0004
Non-hazardous landfill	2,664.60	2,664.60	0.00	0
Inert landfill	0.00	0.00	0.00	0
Hazardous waste incineration	0.00	0.00	0.00	0
Non-hazardous waste incineration	2,373.88	2,373.88	0.00	0
Energy recovery	0.00	0.00	0.00	0
Substance recovery	14,078.31	14,078.29	-0.02	-0.0001
Others (temporarily stored or not specified)	2,645.85	2,645.85	0.00	0
100 ppb				
Hazardous landfill	5,299.87	5,299.89	0.02	0.0004
Non-hazardous landfill	2,664.60	2,664.60	0.00	0
Inert landfill	0.00	0.00	0.00	0
Hazardous waste incineration	0.00	613.80	613.80	100
Non-hazardous waste incineration	2,373.88	2,373.88	0.00	0
Energy recovery	0.00	0.00	0.00	0
Substance recovery	14,078.31	13,464.49	-613.82	-4
Others (temporarily stored or not specified)	2,645.85	2,645.85	0.00	0
50 ppb				
Hazardous landfill	5,299.87	5,299.89	0.02	0.0004
Non-hazardous landfill	2,664.60	2,568.60	-96.00	-4
Inert landfill	0.00	0.00	0.00	0
Hazardous waste incineration	0.00	2,102.80	2,102.80	100
Non-hazardous waste incineration	2,373.88	2,373.88	0.00	0
Energy recovery	0.00	0.00	0.00	0

PCN	Status Quo [kt/y]	New Status Quo [kt/y]	Net change [kt]	Net change [%]
Substance recovery	14,078.31	12,229.49	-1,848.82	-13
Others (temporarily stored or not specified)	2,645.85	2,487.85	-158.00	-6
10 ppb				
Hazardous landfill	5,299.87	5,370.19	70.33	1
Non-hazardous landfill	2,664.60	708.49	-1,956.11	-73
Inert landfill	0.00	0.00	0.00	0
Hazardous waste incineration	0.00	9,950.16	9,950.16	100
Non-hazardous waste incineration	2,373.88	2,373.88	0.00	0
Energy recovery	0.00	0.00	0.00	0
Substance recovery	14,078.31	7,893.29	-6,185.01	-44
Others (temporarily stored or not specified)	2,645.85	766.49	-1,879.36	-71
1 ppb				
Hazardous landfill	5,299.87	14,528.46	9,228.60	174
Non-hazardous landfill	2,664.60	0.00	-2,664.60	-100
Inert landfill	0.00	0.00	0.00	0
Hazardous waste incineration	0.00	10,160.16	10,160.16	100
Non-hazardous waste incineration	2,373.88	2,373.88	0.00	0
Energy recovery	0.00	0.00	0.00	0
Substance recovery	14,078.31	0.00	-14,078.31	-100
Others (temporarily stored or not specified)	2,645.85	0.00	-2,645.85	-100

Based on the data compiled above the following conclusions can be drawn as regards additional treatment capacity in D10, R1 and hazardous waste landfill needed dependent on specific LPCLs according to current waste amounts.

Table 9-16: Summary overview of capacity needs in incineration/energy recovery at LPCL in the dimension of ppm

	Waste requiring incineration at set LPCL (kt/y)					
	0.1 ppm	1 ppm	10 ppm	50 ppm	100 ppm	1,000* ppm
TetraBDE	3,843	3,843	344	344	344	-
PentaBDE	3,894	3,894	395	395	344	-
HexaBDE	3,894	3,894	395	51	-	-
HeptaBDE	3,894	395	51	51	51	-
PFOS	12,606	3,500	1,300	1,300	-	-
PeCB	See Table 9-17	See Table 9-17	See Table 9-17	See Table 9-17	See Table 9-17	See Table 9-17
SCCP	9,601	9,601	508	508	30	30
HBCD	6,479	6,479				
PCP	9.6	9.6	9.6	9.6	9.6	9.6
HCBD	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.
PCN	See Table 9-17	See Table 9-17	See Table 9-17	See Table 9-17	See Table 9-17	See Table 9-17
D10 or R1 total	27,273	18,181	2,728	2,233	404	60

To calculate the total waste amount which has to be incinerated via D10 or R1 in Table 9-16 it has to be considered that the same waste might be presented in different lines and would therefore be counted

several times. This is especially the case for tetra-, penta-, hexa- and hepta BDE which are all present in automotive parts and WEEE. Therefore only the highest waste amount is taken presented by one of these substances to calculate the summed up amount.

It has to be noted that the quantities of PFOS contaminated waste are largely dominated by impregnated carpets, which are expected to be eliminated by 2018. The quantities for PBDE contaminated wastes are dominated by PUR foams in upholstery currently often destined to class II landfills.

For PeCB and PCN there are several waste types which cannot be incinerated (e.g. fly ashes). These waste types need another treatment than incineration. The corresponding waste amounts at different LPCL are listed in Table 9-17. Impacts on waste streams contaminated with PeCB occur only at potential LPCLs below 50 ppb for PCN below 10,000 ppb.

Table 9-17: Summary overview of PeCB and PCNs containing waste on capacity needs in incineration and landfilling at LPCL in the dimension of ppb.

Waste requiring incineration at set LPCL (kt/y)							
PeCB		0.1 ppb	1 ppb	3 ppb	5 ppb	50 ppb	1 ppm
	Class III/IV landfill	65,975	65,646	57.5	57.5	44.5	0
	D10 or R1 total	9,546	9,546	9546	0	0	0
PCN		1 ppb	10 ppb	50 ppb	100 ppb	5,000 ppb	10,000 ppm
	Class III/IV landfill	9,229	70.3	0.02	0.02	0.02	0
	D10 or R1 total	10,160	9,950	2,103	613	0	0
Total	Class III/IV landfill	75,204	65,716	57.5	57.5	44.5	0
	D10 or R1 total	19,706	19,496	11,649	613	0	0

In Table 9-17 the resulting maximum total waste amount which can be disposed of at landfills for hazardous waste or in underground storage systems is only ~ 75,000,000 tonnes per year of additional waste. This is much lower than the European capacity for hazardous waste landfills and underground storage sites.

From Table 9-16 and Table 9-17 the LPCL levels resulting in huge waste amount, needing other treatments are provided and summarized in Table 9-18.

Table 9-18: Overview of affected waste amounts for other treatment by setting the LPCL.

Substance	Critical LPCL level	Non-critical LPCL level
TetraBDE	1 ppm	10 ppm
[kt/y]	3,843	344
PentaBDE	1 ppm	10 ppm
[kt/y]	3,894	395
HexaBDE	1 ppm	10 ppm
[kt/y]	3,894	395
HeptaBDE	0.1 ppm	1 ppm
[kt/y]	3,894	395
PFOS	1 ppm	10 ppm
[kt/y]	3,500	1,300

PeCB	1 ppb	3 ppb
[kt/y]	65,646	57.5
SCCP	1 ppm	10 ppm
[kt/y]	9,601	508
HBCD	14 ppm	5,100 ppm
[kt/y]	6,479	20.8
PCP	> 1 ppm	
[kt/y]	9.6	
HCBd	0.6 ppb	5 ppb
[kt/y]	7,925	n.a.
PCN	50 ppb	100 ppb
[kt/y]	2,103	614
Total waste amount for incineration	49,083	5,023

In Table 9-18 it has to be taken care that the same waste is not counted several times. A LPCL level is considered critical if the waste amount exceeds 2,000 kt /y which is less than 2 % of the total capacity.

These annual amounts need to be compared with available treatment capacity in EU 27.

Latest available data about waste treatment facilities in EUROSTAT⁵⁸ cover the year 2006 and are restricted to:

- Incineration with energy recovery (R1)
- Other incineration (D10)
- Landfilling (D1, D3 - 5, D12)

Data on capacity level is missing for some Member States, therefore the actual capacity of the facilities can be assumed to be higher than presented in the table.

The full data set by Member States is available in Annex V in chapter 12.5.

There is not information available in EUROSTAT for D9 and R4, what however, is not deemed important due to the fact that these are not expected to be important treatment options.

Table 9-19: Capacity of treatment facilities in 2004 and 2006 in EU 27 (source: EUROSTAT)

	R1(kt/y)	D10 (kt/y)	Total (kt/y)
2006 EUROSTAT (capacity)	71,521	45,062	116,583

Table 9-20: Correlation of capacity needs and available capacity

Waste requiring incineration at set LPCL (kt/y)

⁵⁸ Data available at:
http://epp.eurostat.ec.europa.eu/portal/page/portal/waste/data/wastemanagement/waste_treatment, Table "Waste treatment facilities on country level"

	0.1 ppm	Critical LPCL	Non-critical LPCL
Capacity 2006	116,583	116,583	116,583
D10/R1 need	~27,273	49,083	5,023
Ratio	23.4 %	~42%	~4.3%

Table 9-20 illustrates the additional D10/R1 capacity needed in the next roughly 10 years (up to 2018) In the first column the consequences of a general LPCL value of 0.1 ppm is represented. In column 2 and 3 the additional waste amounts which need incineration are represented. The non-critical and critical LPCL values are derived from Table 9-18. The border between non-critical and critical LPCL are considered to be reached at a corresponding waste amount of 2,000 kt/y which represents less than 2 % of the total capacity. It can be seen that at an critical LPCL values the additional waste amounts would represent about 42 % of the D10/R1 capacity whereas at a non-critical LPCL value these waste would represent less than 5 %.

Capacity shortcoming however can be expected to occur in individual Member States, which currently do not dispose of sufficient capacities for thermal treatment/hazardous waste landfill.

It is expected that capacities for waste treatment increases especially in the new EU Member States in the following years as concerning the installation of specific recycling facilities and also concerning incineration capacities⁵⁹. Especially PL, CZ, SK and BG are planning the installation of new incinerators. In addition, in several countries there is an overcapacity of incineration plants, declining amounts of waste currently incinerated.⁶⁰

As a result of the table and of the "available capacity approach" no problems are expected at any of the potential LPCLs.

The resulting LPCL values for the criterion C are therefore:

TetraBDE:	10 ppm
PentaBDE:	10 ppm
HexaBDE:	10 ppm
HeptaBDE:	1 ppm
PFOS:	10 ppm
PeCB:	3 ppb
SCCP:	10 ppm

⁵⁹ Information from awareness raising events 2007-2010, see: www.bipro.de/waste-events

⁶⁰ Information "The future from waste to energy in Europe", News from 5.11.2009 and from <http://www.ask-eu.com/>, Article "Major decline of Europe's orders for new incineration plants"

HBCD: 5,100 ppm

PCP: 1 ppm

HCBD: 5 ppb

PCN: 100 ppb

Results for Criterion D: Economic feasibility

Based on the results of the mass flow analysis (see chapters 5 and 6) it is possible to assess economic impacts of potential low POP content limits, in terms of arising restrictions for recycling and landfilling operations. Such restrictions will create both additional costs of disposal and analysis or loss of return for the waste owner.

This chapter compiles the results of waste flow scenarios in relation to potential low POP limits. It was not possible in the scope of the project to perform an economic impact analysis in all details and for every single case where consequences arise from low POP content limit values. It has to be taken into consideration that costs strongly depend on market conditions and can differ significantly between countries.

It is not possible to assess economic feasibility with a consensus of all concerned stakeholders as interests are obviously different. For economic feasibility also costs of a no action option (including possible environmental damage) have to be considered what defines problems due to a lack of data for many cases.

For the treatment options for a waste exceeding the low POP concentration it has to be clarified if such waste is a hazardous waste or not. With respect to this, the following can be stated:

- A waste is considered a POP waste if its POP content exceeds a certain concentration limit (the LPCL according to Article 7.4. (a) specified in Annex IV of the POP Regulation). According to the Stockholm Convention (Art 6d) and the corresponding articles of the EU POP Regulation, the fact that a waste is a POP waste and listed in the annexes is to be disposed only and not recycled except with listed methods which can destroy the POP content (currently R1 and R4) ;
- Whether a waste is considered a hazardous waste is defined in the Waste Framework Directive 2008/98/EC in Article 3.2. and Annex III. Exhibition of H-criteria and listing with asterisk in the EWL are the decisive criteria from the legal point of view. It however, should be noted that for PCB the classification as hazardous in the EWL is linked to the LPCL, and for dioxins or organic solvents even the mere existence is enough for classification as hazardous. In addition, according to the Basel Convention all organohalogen compounds (Y45) need special control. The Stockholm Convention is explicitly linked to the Basel Convention.
- Whether or not a POP waste according to LPCL is automatically a hazardous waste is currently not fully clear and discussed. This aspect is reflected in this chapter as concerns incineration (no clear differentiation between MSWI and HWI) but does not influence the cost calculation for landfilling.
- If a waste is not hazardous according to Directive 2008/98/EC it does not become a hazardous waste if it exceeds the LPCL established in the POP regulation. Whether a waste is considered a POP waste is therefore independent from its hazardousness. The exceedance of

POP concentration limits will not change the quantities of hazardous waste.

- According to Directive 2000/76/EC on the incineration of waste or to the new Directive 2010/75/EU on industrial emissions, there is no legal obligation to incinerate hazardous waste exclusively in a hazardous waste incinerator. Directive 2000/76/EC does not differentiate between HWI and MSWI plants. Each incineration plant has to have a permit for each waste incinerated. However there are some specific provisions for hazardous waste. These are related to acceptance, reporting, documentation and emission limits which in general is different for MSWI and HWI plants. In addition there is an obligation to meet specific operating conditions (see Article 6) if hazardous waste with a halogen content above 1% is incinerated, which in practice are generally only met at hazardous waste incineration facilities.. The classification of a waste as a POP waste does neither impact on the hazardousness of the waste nor on the permit of the incineration plant and thus not on the distribution of wastes between incineration plants.
- Controlled incineration of POP waste in EU incineration plants under conditions as defined in Directive 2000/76 is appropriate to destroy all new and all candidate POPs, as it is for the existing POPs (see chapter 9.7 on disposal and recovery operations).

As a consequence, a clear distinction between the incineration capacities needed in order to destroy the POPs in MSW incinerators and hazardous waste incinerators respectively in D10 or R1 is not possible and is not taken into account for the purposes of the present project.

Waste which cannot be incinerated (e.g. ashes from incineration processes) will require other treatment e.g. underground disposal according to Annex V part II; for the cost impact assessment it is assumed that such waste is currently reused or landfilled and will be disposed of in underground storage sites or stabilised and disposed of at hazardous landfill sites if it exceeds the LPCL.

According to the Basel Convention technical guidelines, POP wastes have to be managed as hazardous wastes: "Wastes consisting of, containing or contaminated with POPs above the low POP contents referred to in section III.A should be managed as hazardous waste, to prevent spills and leaks leading to worker exposure, releases to the environment and exposure to the community" [BC TG 2007]. The management of POP waste as hazardous waste will have several consequences including the export ban of hazardous wastes according to Article 36 of Regulation 1013/2006.

Against this background it can be considered that the following changes in waste management are relevant in waste management practice:

- (1) Change from current landfill operation to future waste incineration,
- (2) Change from current recycling/reuse to future incineration,
- (3) Change from current landfill to underground storage or landfilling with stabilisation (e.g. waste that cannot be incinerated as ashes) and

(4) Change from current recycling/reuse to underground storage or landfilling with stabilisation.

Cost for the relevant waste management options vary considerable between Member States and even within one country. Costs can e.g. differ depending on spare capacity and local market conditions and depend on specific factors ranging from the size of a facility, the nature and duration of contracts to technology, the age of facility and possible revenues from the sale of recovered materials.

However, usually costs for waste management at recycling facilities and landfill sites are lower than those at incineration facilities. However, in many Member States there is an ongoing trend make landfill relatively more expensive over time compared with other waste management options (see e.g. [WRAP 2010]).

There is only limited information available on specific cost of different waste management options.

In the UK for example, gate fees for waste management options were investigated in the beginning of 2010. For the relevant options mentioned above the cost are indicated in the following table:

Table 9-21: Costs for relevant waste management option in the UK⁶¹

Waste management option	Remark	Costs per tonne		
		Median (€)	Min (€)	Max (€)
Recycling/Reuse	paper, card, plastic bottles, cans	40	-6	50
Landfill	including tax	84	71	110
Incineration	existing facilities	59	39	95
Incineration	planned facilities	110	94	131
Incineration	current market estimate	n.a.	102	144

According to a German underground disposal site operator, costs for underground disposal can be assumed around 250€ per tonne.

These figures from the UK and Germany are not representative for the EU. However, due to the lack of more specific information the figures are used for a rough cost impact assessment for EU27.

Current costs for recycling/reuse are therefore considered to amount to around 40€ per tonne in the cost assessment.

Landfill costs vary considerably and are subject to short time changes with an ongoing trend to increase. For the cost assessment, the costs for landfill are considered to amount to approximately 80€ per tonne.

Costs for the stabilisation of waste is estimated at approximately 100 € per tonne and a weight increase of 100%. Accordingly the amount to be disposed of would double (→ disposal cost 160€) and the costs would amount to 160€ plus 100€. For the cost impact estimates the costs are calculated in the same range as for underground disposal, at 250 € per tonne.

⁶¹ Based on WRAP 2010, Gate fees at waste management facilities in the UK; considering that 1 GBP is 1.2 €

Incineration costs vary significantly between older (less costly) and new (more costly) and planned (most costly) facilities [WRAP 2010]. This corresponds to information on costs for new waste incineration facilities between 100 and 150€ per tonne for Europe⁶². Taking this into account for the cost assessment, the costs for incineration are considered to amount to approximately 100€ per tonne (old and new facilities).

Against this background, the shift from the relevant options is estimated in Table 9-22.

Table 9-22: Costs for relevant waste management options

Current option	Future option if LPCL is exceeded	Related cost impact per tonne
Landfill (80 €)	Incineration (100 €)	20 €
Recycling/reuse (40€)	Incineration (100 €)	60 €
Landfill (80 €)	Underground disposal (250 €)	170 €
Landfill without stabilisation (80 €)	Landfill with stabilisation (250 €)	170 €
Recycling/reuse (40€)	Underground disposal (250 €)	210 €
Recycling/reuse (40€)	Landfill with stabilisation (250 €)	210 €

The four different waste treatment changes and corresponding costs from Table 9-22 have been used to estimate the theoretical cost impacts for all analysed waste types from section 6 and summarised in Table 9-23.

Table 9-23: Waste requiring additional incineration or other treatment at set LPCL (t/y) and related cost estimates

Tetra BDE flows	0.1 ppm	1 ppm	10 ppm	50 ppm	100 ppm
Landfill to incineration [t/y]	3,765,256	3,765,256	265,629	265,629	265,629
Landfill to incineration [€/y]	75,305,120	75,305,120	5,312,580	5,312,580	5,312,580
Recovery to incineration [t/y]	78,049	78,049	78,049	78,049	78,049
Recovery to incineration [€/y]	4,682,940	4,682,940	4,682,940	4,682,940	4,682,940
Total costs [Million €/y]	80	80	10	10	10
Penta BDE flows	0.1 ppm	1 ppm	10 ppm	50 ppm	100 ppm
Landfill to incineration [t/y]	7,797,498	7,797,498	297,871	297,871	265,629
Landfill to incineration [€/y]	155,949,960	155,949,960	5,957,420	5,957,420	5,312,580
Recovery to incineration [t/y]	96,923	96,923	96,923	96,923	78,049
Recovery to incineration [€/y]	5,815,380	5,815,380	5,815,380	5,815,380	4,682,940
Total costs [Million €/y]	162	162	11.8	11.8	10
Hexa BDE flows	0.1 ppm	1 ppm	10 ppm	50 ppm	100 ppm
Landfill to incineration [t/y]	7,797,498	7,797,498	297,871	32,242	0
Landfill to incineration [€/y]	155,949,960	155,949,960	5,957,420	644,840	0
Recovery to incineration [t/y]	96,923	96,923	96,923	18,874	0
Recovery to incineration [€/y]	5,815,380	5,815,380	5,815,380	1,132,440	0
Total costs [Million €/y]	162	162	11.8	1.8	0
Hepta BDE flows	0.1 ppm	1 ppm	10 ppm	50 ppm	100 ppm
Landfill to incineration [t/y]	7,797,498	297,871	32,242	32,242	32,242
Landfill to incineration [€/y]	155,949,960	5,957,420	644,840	644,840	644,840
Recovery to incineration [t/y]	96,923	96,923	18,874	18,874	18,874
Recovery to incineration [€/y]	5,815,380	5,815,380	1,132,440	1,132,440	1,132,440

⁶² See <http://www.ask-eu.de/Artikel/3395/Risiken-der-mechanisch-biologischen-Abfallbehandlung.htm>, accessed on 17.2.2011

Total costs [Million €/y]	162	11.8	1.8	1.8	1.8
PFOS	0.05 ppm	1 ppm	10 ppm	50 ppm	100 ppm
Landfill to incineration [t/y]	4,840,373	4,840,373	1,315,980	1,300,410	0
Landfill to incineration [€/y]	96,807,460	96,807,460	26,319,600	26,008,200	0
Recovery to incineration [t/y]	7,766,051	388,303	0	0	0
Recovery to incineration [€/y]	465,963,060	23,298,180	0	0	0
Total costs [Million €/y]	563	120	26.3	26.0	0
PeCB	1 ppb	5 ppb	50 ppb	1 ppm	10 ppm
Landfill to incineration [t/y]	1,980,113	0	0	0	0
Landfill to incineration [€/y]	39,602,251	0	0	0	0
Recovery to incineration [t/y]	5,604,884	0	0	0	0
Recovery to incineration [€/y]	336,293,046	0	0	0	0
Landfill to underground disposal [t/y]	1,118,912	8,765	8,765	0	0
Landfill to underground disposal [€/y]	190,215,063	1,490,119	1,490,119	0	0
Recovery to underground disposal [t/y]	58,018,044	40,001	26,956	0	0
Recovery to underground disposal [€/y]	12,183,789,181	8,400,109	5,660,696	0	0
Total costs [Million €/y]	12,750	9.9	7.2	0	0
SCCP	1 ppm	10 ppm	1.000 ppm	10,000 ppm	100,000 ppm
Landfill to incineration [t/y]	1,834,782	119,814	29,553	26,654	3,590
Landfill to incineration [€/y]	36,695,640	2,396,280	591,060	533,080	71,800
Recovery to incineration [t/y]	7,766,051	388,303	0	0	0
Recovery to incineration [€/y]	465,963,060	23,298,180	0	0	0
Total costs [Million €/y]	502	25.7	0.59	0.53	0.07
HBCD	0.015 ppm	14 ppm	100 ppm	40,000 ppm	80,000 ppm
Landfill to incineration [t/y]	124,000,000	3,244,825	18,509	11,492	9,289
Landfill to incineration [€/y]	2,480,000,000	64,896,500	370,180	229,840	185,780
Recovery to incineration [t/y]	70,000,000	3,234,577	2,337	1,244	0
Recovery to incineration [€/y]	4,200,000,000	194,074,620	140,220	74,640	0
Total costs [Million €/y]	6,680	259	0.51	0.30	0.19
PCP	1 ppm	10 ppm	100 ppm	1000 ppm	25,000 ppm
Landfill to incineration [t/y]	6,420	6,420	6,420	6,420	6,420
Landfill to incineration [€/y]	128,400	128,400	128,400	128,400	128,400
Total costs [Million €/y]	0.13	0.13	0.13	0.13	0.13
HCBD	0.6 ppb	5 ppb	10 ppb	0.1 ppm	1 ppm
Landfill to incineration [t/y]	1,430,000	n.a.	n.a.	n.a.	n.a.
Landfill to incineration [€/y]	28,600,000	n.a.	n.a.	n.a.	n.a.
Recovery to incineration [t/y]	6,495,000	n.a.	n.a.	n.a.	n.a.
Recovery to incineration [€/y]	389,700,000	n.a.	n.a.	n.a.	n.a.
Total costs [Million €/y]	418	n.a.	n.a.	n.a.	n.a.
PCN	1 ppb	10 ppb	50 ppb	100 ppb	5,000 ppb
Landfill to incineration [t/y]	1,956,113	1,956,113	96,000	0	0
Landfill to incineration [€/y]	39,122,251	39,122,251	1,920,000	0	0
Recovery to incineration [t/y]	6,218,684	6,114,684	1,848,800	613,800	0
Recovery to incineration [€/y]	373,121,046	366,881,046	110,928,000	36,828,000	0
Landfill to underground disposal [t/y]	708,486	0	0	0	0
Landfill to underground disposal [€/y]	120,442,672	0	0	0	0
Recovery to underground disposal [t/y]	4,243,477	70,328	19	19	19
Recovery to underground disposal [€/y]	891,130,134	14,768,784	3,990	3,990	3,990
Total costs [Million €/y]	1,424	421	113	36.8	0.004

PetraBDE: As shown in Table 9-23, at a LPCL level of 10 ppm approximately 266 kt/y of landfilled and 78 kt/y waste from recycling would require incineration. When lowering the LPCL the waste amounts continuously increase and a significant increase can be recognised at a LPCL of 1 ppm with around 3,765 kt/y landfilled and 78 kt/y recycled waste. This corresponds to an increase in total costs for additional incineration from ~10 million €/y to ~80 million €/y, as shown in Table 9-23.

Penta/HexaBDE: At a LPCL level of 10 ppm approximately 298 kt/y of landfilled and around 97 kt/y of recycled waste would require incineration. When lowering the LPCL the waste amounts continuously increase and a significant increase can be observed at a LPCL of 1 ppm with around 7,797 kt/y landfilled and 97 kt/y recycled waste. This corresponds to a significant increase in costs for additional incineration from ~11.8 million €/y to ~162 million €/y, as shown in Table 9-23.

HeptaBDE: At a LPCL level of 10 ppm approximately 32 kt/y of landfilled and around 19 kt/y of recycled waste would require incineration. When lowering the LPCL the waste amounts continuously increase and a significant increase can be recognised at a LPCL of 1 ppm with around 298 kt/y landfilled and 97 kt/y recycled waste. This corresponds to an increase in costs for additional incineration from approximately 1.8 million €/y to 11.8 million €/y, as shown in Table 9-23. By lowering LPCL further to 0.1 ppm approximately 7,797 kt/y of landfilled and around 97 kt/y of recycled waste would require incineration, which will result in an increase of costs from 11.8 million €/y to 162 million €/y.

PFOS: Already at a set LPCL value of 50 ppm significant costs of about 26 million € arise due to disposal of carpet waste. However, it is assumed that these amounts will dramatically drop in the following years. Broken down to the economic impact of the Member states it can be seen that this will be very different depending on the current treatment of bulky waste. Another strong economic impact can be seen at a set LPCL of 1 ppm, which affects large amounts of sewage sludge.

PeCB: At a LPCL level above 1 ppm no wastes are affected. At a level of 50 ppb about 1 million tonnes of ash are affected. With lowering LPCL the waste amounts continuously increase and a significant increase is recognised at a LPCL of 1 ppm with 72 million tonnes of waste exceeding this limit value.

SCCP: The relatively little amounts of SCCP containing wastes result in a relatively irrelevant economic impact on an EU wide scale, down to a set LPCL of about 10 ppm. At the set LPCL of 10 ppm contaminated sewage sludge cover a significant share of the costs of about 94% (~14.2 million €), either from the landfill or from the recycling stream. At a set LPCL of 1 ppm this share increases to 99.7 % (501.1 million €).

HBCD: In case of HBCD several changes in waste amounts and corresponding costs can be observed when lowering the LPCL (see Table 9-23). The most significant changes can be observed when lowering the LPCL from 5,100 ppm to 7 ppm and further to 0.015 ppm. As shown in Table 9-23, at a LPCL level of 5,100 ppm approximately 18.5 kt/y of landfilled and 2.3 kt/y of recycled waste would require incineration. When lowering the LPCL to 7 ppm around 3,245 kt/y of landfilled waste and around 2,235 kt/y of recycled waste would require incineration. This corresponds to a significant increase in costs for additional incineration from approximately 0.51 million €/y to 259 million €/y. By lowering the LPCL further to 0.015 ppm the waste amounts increase to 124,000 kt/y of landfilled and around 70,000 kt/y of recycled waste which would need to be sent to incineration. The total costs increase from 259 million €/y to 6,680 million €/y.

PCP: From the two existing relevant industries from which PCP containing wastes arises, only the

textile industry seems to be relevant regarding an economic impact from the LPCL as impregnated wood is already incinerated in hazardous waste incineration plants nowadays. As the PCP content of textile is above 10,000 ppm different settings of LPCLs below 10,000 ppm do not have an economic impact. Investigations with LPCLs above 10,000 ppm are not reasonable as other criteria (e.g. crit. Y) would contradict with such high LPCLs. However it can be seen that the economic consequences of LPCL regarding PCP containing waste is relatively little.

HCBD: As only little information is available for different HCBD concentrations in waste no reasonable estimation of set LPCL on the economic impact can be given. At a set LPCL of 0.6 ppb sewage sludges could be identified to require special treatment and would therefore cover a huge amount of waste with corresponding financial consequences.

PCN: The majority of the PCN containing waste has a concentration below 1 ppm. However, a LPCL of 1 ppm already includes wastes from hospital waste incineration plants which have concentration levels of 4 to 5 ppm. The amount of such waste types (e.g. ashes), is however enough that it would have a strong economic impact if affected. Another waste type with a PCP concentration over 1 ppm is slag from the copper production with about 600 kt/y.

For a better overview only the arising total costs are provided in Table 9-26. Only those LPLC values are presented, which are below and above the additional costs of 50 million €.

Table 9-24: Significant LPCL values for criterion D

Tetra BDE	1 ppm	10 ppm
Costs [Million €/y]	80	10
Penta BDE flows	1 ppm	10 ppm
Costs [Million €/y]	162	11.8
Hexa BDE flows	1 ppm	10 ppm
Costs [Million €/y]	162	11.8
Hepta BDE flows	0.1 ppm	1 ppm
Costs [Million €/y]	162	11.8
PFOS	1 ppm	10 ppm
Costs [Million €/y]	120	26.3
PeCB	1 ppb	5 ppb
Costs [Million €/y]	12,7550	9.9
SCCP	1 ppm	10 ppm
Costs [Million €/y]	502	25.7
HBCD	14 ppm	100 ppm
Costs [Million €/y]	259	0.51
PCP	> 1 ppm	
Costs [Million €/y]	0.13	
HCBD	0.6 ppb	5 ppb
Costs [Million €/y]	418	n.a.
PCN	50 ppb	100 ppb
Costs [Million €/y]	113	37

Table 9-24 gives a brief overview of Table 9-23. For Criterion D the following values would result:

TetraBDE: 10ppm

Penta BDE:	10 ppm
Hexa BDE:	10 ppm
Hepta:	1 ppm
HBCD:	100 ppm
PFOS:	10 ppm
PeCB:	5 ppb
SCCP:	10 ppm
PCP:	1 ppm
HCBD:	5 ppb
PCN:	100 ppb

9.2.2 Results for upper limitation Criteria

Results for criterion Z: Existing limit values

There are generally not many limit values for new POPs in existing legislation at international, European or national scale. The limited information is compiled in this chapter by substance/substance group including conclusions as regards contamination levels the LPCL may not go below.

PBDEs

Table 9-25: Relevant European legislation (PBDEs)

Legislation	Content	Existing Limit Values (LVs)
Proposal amended POPs Reg 850/2004	Concentration of PBDE homologues (tetra-, penta-) below which they are considered as "unintentional trace contaminant" in substances/ preparations	0.001 % by weight (10 ppm)
	Exemption for ban of production, placing on the market and use for articles and preparations produced partially or fully from recycled materials or materials from waste prepared for re-use	0.1% by weight (1,000 ppm) for relevant PBDE congener groups
RoHS Dir 2002/95	Ban of placing on the EU market of new EEE if containing PBDE (except DecaBDE) in homogeneous material	0.1% by weight (1,000 ppm)
REACH Reg 1907/2006	Ban of placing on the EU market / using as a substance or a constituent of substances or of preparations Finished articles may not be placed on the EU market if they, or flame-retardant parts thereof, contain pentaBDE in higher concentrations	0.1% by weight (1,000 ppm)
Water FrameDir	Environmental Quality Standards (EQS), annual average	Inland AA-EQS: 0,0005 [µg/l]

2000/60 and Dir 2008/105	(AA) for tetra-, penta and hexaBDEs	Other surface waters AA-EQS: 0,0002 [µg/l]
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- National legislation

Table 9-26: Relevant national legislation (PBDEs)

MS	National Legislation	Existing Limit Values (LVs)
AT	Restrictions on transboundary transport of waste (notification)	0.1% (1,000 ppm) for the sum of penta-, octa- and decabromodiphenyl ether in wastes (homogenous material)
AT	Emission limit waste water	0.5 µg/litre (ppb) for technical pentaBDE from chemicals industry
NL	Occupational exposure limit for C-OctaBDE; CAS 32536-52-0)	0.1 mg/m ³ [SER 2010]
NO	Definition as hazardous waste	Waste containing 0.25% of pentaBDE or octaBDE (2,500 ppm) is defined as hazardous waste
Korea	Recycling of electronic and electric devices	Only allowed if the concentration of penta- and octaBDE in EEE product is below the threshold of 0.1 % (=1,000 ppm)

PFOS

Table 9-27: Relevant European legislation (PFOS)

Legislation	Content	Existing Limit Values (LVs)
POPs Reg 850/2004 as amended	Concentration of PFOS below which it is considered as “unintentional trace contaminant” in substances/ preparations	Equal to or below 10 mg/kg (0,001 % by weight)
	Concentration of PFOS below which it is considered as “unintentional trace contaminant” in semi-finished products or articles	Lower than 0,1 % by mass (i.e. 1000 mg/kg) calculated with reference to the mass of structurally or micro-structurally distinct parts that contain PFOS or, for textiles or other coated materials, if the amount of PFOS is lower than 1 µg/ m ² of the coated material; exempted uses listed
REACH Reg 1907/2006	Placing on the market of PFOS in concentrations as substance or in mixtures	Equal to or greater than 50 mg/kg (0,005 % by weight)
	Placing on the market of PFOS in concentrations in semi-finished products or articles, or parts thereof	Equal to or higher than 0,1 % by mass (i.e. 1000 mg/kg) calculated with reference to the mass of structurally or microstructurally distinct parts that contain PFOS or, for textiles or other coated materials, if the amount of PFOS is equal to or higher than 1 µg/m ² of the coated material; exempted uses listed.

- National legislation

Table 9-28: Relevant national legislation (PFOS)

MS	National Legislation	Existing Limit Values (LVs)
AT	Sewage sludge	Limit values of 100 µg PFOS+PFOA /kg (0.1 ppm)
DE	Occupational exposure limit	0.01 mg/m ³ inhalable aerosol (8h; DFG) 0.08 mg/m ³ inhalable aerosol (short term; DFG)
Korea	defined as hazardous waste	Fire fighting foam waste containing 0,005 % (50 ppm) by weight or more of PFOS and PFOS related compounds
Korea	Ban of production, import, export and marketing of impregnating articles	If containing more than 0.005% (50 ppm) of PFOS and PFOS related compounds (see EU, REACH)

PeCB

Table 9-29: Relevant European legislation (PeCB)

Legislation	Content	Existing Limit Values (LVs)
POPs Reg 850/2004	Proposed limit values for POPs content in wastes	LPCL: 50 mg/kg; MPCL: 5000 mg/kg.
WFD 2000/60 (water) and Dir 2008/105	Environmental Quality Standards (EQS), annual average (AA) Maximum Allowable Concentration (MAC)	Inland AA-EQS: 0,007 [µg/l] Other surface waters AA-EQS: 0,0007 [µg/l].

- National legislation

Table 9-30: Relevant national legislation (PeCB)

MS	National Legislation	Existing Limit Values (LVs)
CN	interim chronic exposure water quality guideline for the protection of freshwater aquatic life	0.006 mg/L for PeCB

SCCP

More details regarding the legal background can be found in section 6.5.4.

Table 9-31: Relevant European legislation (SCCP)

Legislation	Content	Existing Limit Values (LVs)
Directive 2002/45/EC amending for the twentieth time Council Directive 76/769/EEC	The placing 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 was banned	1 % (10,000 ppm)

- National legislation

Table 9-32: Relevant national legislation (SCCP)

MS	National Legislation	Existing Limit Values (LVs)
BE; DK; FR; FI; DE; IS; IE; LU; NL; NO; PT; ES; SE; CH; UK	PARCOM decision 95/1 is more stringent than current EU legislation and aims to phase out the use of SCCPs by restricting also the other uses of SCCPs	

HB CD

More details regarding the legal background can be found in section 6.6.4.

Table 9-33: Relevant European legislation (HB CD)

Legislation	Content	Existing Limit Values (LVs)
REACH Regulation, Art 7.2 (Notification to ECHA), Art 7.3 (Supply of appropriate instructions), Art 31.1 (Provision of Safety Data Sheet) and Art 33 (Duty to communicate safe use information or responding to customer request)	-HB CD underlies specific obligations resulting from the inclusion in the Candidate List of Substances of Very High Concern. From the date of inclusion EU or EEE suppliers of articles which contain substances on the Candidate List in a concentration above 0.1 % (w/w) have to provide sufficient information to allow safe use of the articles -From 2011 EU and EEA producers or importers of articles have to notify ECHA if their article contains a substance on the Candidate List. This obligation applies if the substance is present	0.1 % (1,000 ppm)

	in quantities totalling over a concentration of 0.1 % (w/w)	
Commission Regulation (EU) No 143/2011 and Reach Regulation Article 56(6) lit. (a)	HBCD is listed in Annex XIV Reach regulation and is therefore subject to authorisation. Article 56(6) lit. (a) REACH Regulation foresees that the authorisation regime does not apply for PBT substances such as HBCD, if they are present in a mixture below a concentration limit of 0.1 % w/w.	0.1% (1,000 ppm)

No national limit values regarding HBCD could be found.

PCP

Table 9-34: Relevant European legislation (PCP)

Legislation	Content	Existing Limit Values (LVs)
Directive 1999/51/EC	Pentachlorophenol and its salts and esters shall not be used in a concentration equal to or greater than 0.1 % by mass in substances or preparations placed on the market.	1,000 ppm (0.1%)

- National legislation

Table 9-35: Relevant national legislation (PCP)

MS	National Legislation	Existing Limit Values (LVs)
AT	Fed. Law Gaz. No. 1996/668 limitation in the ordinance on waste water emissions for the production of plant protecting agents and crop spraying.	1 mg/l; 23 g/t installed capacity per plant
BE	Never been authorized as pesticide and authorization as biozide was withdrawn in 1987	
DE	BBodSchV soils	50 mg/kg playgrounds; 100 mg/kg residential area; 250 mg/kg parks and industrial areas
NL	Prohibition for trade and import of articles	5 mg/kg
PL	Dz. U. Nr. 27, poz. 169 for sewage sluges and Dz. U. Nr 162, poz. 1008 for water	Sewage sludge: 2.0 mg/kg (daily average) 1.0 mg/kg (monthly average) Water: 1.0 µg/l

HCBd

See chapter 6.8.4 for more detail. HCBd is a POP candidate under the Stockholm convention and is listed in the Annex I to the POP Protocol. However, there are no limit values set.

- National legislation

Table 9-36: Relevant national legislation (HCBd)

MS	National Legislation	Existing Limit Values (LVs)
PL	Dz. U. Nr. 27, poz. 169 for sewage sluges and Dz. U. Nr 162, poz. 1008 for water	Sewage sludge: 3.0 mg/kg (daily average) 1.0 mg/kg (monthly average) Water: 0.6 µg/l

PCN

Details can be found in chapter 6.9.4. PCN is listed in Annex 1 to the POP Protocol, but not in the Stockholm Convention. There are no limit values set.

Based on the existing legal background the following conclusions can be taken as regards the limitation criterion Z for the new POPs.

Substance	Existing limit value	Result for limitation criterion Z
Limit PBDE	10 ppm (unintentional trace contamination in substances, preparations, articles) 1,000 ppm (ban of placing on the market in new products; for technical substance) 0.5 ppb (C-pentaPBDE waste water) 0.5 ppt (annual average inland waters)	1,000 ppm
PFOS	10 ppm (unintentional trace contamination in substances) 50 ppm (ban for placing on the market as substance or mixture REACH) 1,000 ppm (unintentional trace contamination in semi-finished products and articles) 1,000 ppm (ban for placing on the market in semi-finished products and articles REACH) 0.1 ppm (sewage sludges for agricultural soils) 0.01 ppm (OEL workplace air)	1,000 ppm
PeCB	50 ppm (proposed LPCL) 5,000 ppm (proposed MPCL) 6 ppb (water quality guidance level Canada) 7 ppt (quality standard inland waters)	50 ppm
SCCP	10,000 ppm (ban for placing on the market as substance or mixture)	10,000 ppm
HBCD	1,000 ppm (authorisation needed REACH)	1,000 ppm
PCP	1,000 ppm (ban for placing on the market as substance or mixture)	1,000 ppm
HCBD	----	----
PCN	----	----

Limit values for contamination levels in sectors such as sewage sludge, and quality criteria for water cannot be used as upper limitation criterion but give an additional indication for levels which may not be passed below. Another limit that should not be passed below by the LPCL is the 10 ppm limit for PBDE congener groups and PFOS in products. If there are major reasons for an LPCL below that level these limits would need to be changed as well.

Results for Criterion Y: Worst case scenario for human health risks

In the course of a risk assessment regarding the exposure to new POPs and candidate POPs in the scope of the POP regulation (1) waste generation, (2) storage and (3) transport is not relevant because legislation does not require differing management relating to whether or not the waste is classified as POP waste. Thus only the sector waste disposal/recovery is investigated in the scope of this project.

POP waste requires specific treatment. The POP content has to be destroyed or irreversibly transformed by means of D9, D10, R1 or R4 operations in accordance with Annex V of the POP Regulation. In exceptional cases, authorities may permit permanent storage in salt mines, hard rock formations or at hazardous waste landfill sites, if the environmental preferability has been demonstrated. Thus, the risk assessment can be restricted to all disposal/recovery pathways that may be used for "non-POP" waste

(below the low POP content limit), but which will not be permitted for POP wastes (i.e. R2, R3, R5 to R13 and D1 to D8 and D11 to D15). In this respect, both occupational and general exposure may be relevant.

Risk of occupational and general exposure is related to major release pathways for the different substances. Table 9-37 provides an overview of the most relevant exposure pathways of the new and candidate POP substances. This includes – where available - information concerning transfer rates from waste to environment and environmental compartments to food.

It has to be admitted that knowledge and understanding of the related processes currently is not sufficient developed. However there is a basic consensus about certain aspects which helps to further reduce the number of pathways to investigate for a first worst case assessment.

Table 9-37: Overview of most relevant exposure pathways of new and candidate POP substances

Substance	Most relevant exposure pathways for general population	Source
C-PentaBDE	<ul style="list-style-type: none"> Oral (Food: particularly fish, and agricultural products) Inhalative (indoor air at home and workplaces due to levels in products like furniture and electronic devices) 	<ul style="list-style-type: none"> The application of sewage sludge to agricultural land is one of the reasons for detected levels of PentaBDE in food products and can explain the detected levels in vegetables and root crops in experimental studies. Levels in fish and root crops can be the source of exposure to domestic animals like chickens and pigs, and the source of PBDEs in meat products for human nourishment. Available hazard assessment (VCCEP 2003, COT 2004, VKM 2005) conclude that exposure information is inadequate to fully characterize the risks. PBDE concentrations in Europe from the early 2000 show low median values on a lipid basis of 0.2-1.0 ng/g LW of PentaBDE (BDE-99) for all countries and relatively small variations. [UNEP/POPS/POPRC.3/20/Add.7] The ingestion of soil/dust is an important route of exposure, in particular for toddlers and children. But the median lifelong exposure to PBDEs via ingestion of soil/dust seems only a minor route of exposure when compared with the exposure via food. Harrod et al. 2004 estimate diet at 93% and inhalative share at 7%. The inhalative share however can be up to 35%. Absorption of PBDEs originating from house dust is expected to significant less than from food. In this context the (widely made) assumption that the absorption of PBDEs from soil/house dust lies between 50 and 100 % (Otazo et al., 2005) may easily overestimate the accumulation of PBDEs originating from housedust in the body. [RIVM 2006] The maximum daily human exposure via the inhalation pathway represented 4.1-4.4% of overall daily intake. [Wilford et al. 2004] The geometric mean concentrations of total PBDEs (sum of 19 congeners) in offices (205 pg m⁻³), internet cafes/computer rooms (127 pg m⁻³) and computers/electronic shops (85 pg m⁻³) were significantly higher than those in furniture stores (12 pgm⁻³), homes (8pg m⁻³) and outdoor air (18 pg m⁻³). The daily inhalation intake of PBDEs estimated for the employees of the four occupational settings ranged from 0.2 to 1.4 ng/day and it was significantly lower than the expected dietary intake (77 ng/day). Inhalation generally represented a small fraction of the overall daily exposure to PBDEs (1%); in heavily contaminated office (10 848 pg m⁻³ of total PBDEs) intake from this route (65 ng day⁻¹) may, in some extreme cases, be as important as diet. [Mandalakisa et al. 2008]
C-OctaBDE	<ul style="list-style-type: none"> Oral exposure is expected to be the most relevant exposure 	Transposure rate >90%. Biota-sediment accumulation factors between 1 and 3 for Hexa and HeptaBDE on freshwater fish.

Substance	Most relevant exposure pathways for general population	Source
	pathway (fish, meat)	Highest potential for bioaccumulation for HexaBDE; but significant differences between the control population and occupationally exposed groups were also reported. Fish major contributor (around 40% DDI) due to the high PBDE levels. Meat account for around 30%, dairy products and eggs contribute to a lesser degree (< 30%). [UNEP/POPS/POPRC.3/20/Add.6]
PFOS	<ul style="list-style-type: none"> Major route into local food chains through fish (release from WWTP) 	PFOS does not follow the “classical” pattern of partitioning into fatty tissues followed by accumulation, because PFOS is both hydrophobic and lipophobic. PFOS binds preferentially to proteins in the plasma; bioconcentration factors (BCF) 2800-3100. These values are below the numeric BCF criteria in Stockholm Convention Annex D (the reported BCF values are below 5000; high level of bioaccumulation observed in mammals. BMF of 10 to 20 in bald eagles. Calculated hypothetical BMFs = 22 – 160; PFOS has shown a high oral uptake (95%) [UNEP/POPS/POPRC.2/1/17/Add.5, 2006]
PeCB	Inhalation, food and drinking water.	PeCB is persistent in soils and sediments under aerobic conditions (>6 years); considered to have a high bioaccumulation potential [UNEP/POPS/POPRC.3/20/Add.7, 2007]
SCCP	<ul style="list-style-type: none"> Oral exposure is expected to be the most relevant exposure pathway (food e.g. fish). 	Generally, private individuals are not expected to handle materials and mixtures that contain SCCPs apart from a few cases (possibly some sealants or paints, commercial tents, etc.) [RPA 2010]. Iino et al. (2005) concluded that food was the main exposure pathway of SCCPs to humans, but did not pose any health risk in Japan in 2003 [POPRC.5/2/Rev.1]. The available BCF data indicate that short-chain chlorinated paraffins do meet the very bioaccumulative (vB) criterion [ERA 2008].
HBCD	<ul style="list-style-type: none"> Inhalation, oral and dermal exposure 	The human population can be exposed to HBCD by inhalation of vapour and airborne dust, ingestion and by dermal contact. In addition there is a risk that babies can be exposed during pregnancy and due to breast-feeding [ECB 2008a].
PCP	<ul style="list-style-type: none"> Inhalation, oral and dermal exposure 	General population exposure may occur through contact with contaminated environmental media, particularly in the vicinity of hazardous waste sites. Important routes of exposure appear to be inhalation of contaminated air, ingestion of contaminated groundwater used as a source of drinking water, ingestion of contaminated food and soils, and dermal contact with contaminated soils or products treated with the compound [IEP 2008].
HCBD	<ul style="list-style-type: none"> Inhalation, oral and dermal exposure (oral exposure is expected to be the most relevant exposure pathway, i.e. food and drinking-water) 	Exposure of the general public mainly occurs indirectly via drinking-water and food with high lipid content [IPCS 1993].
PCN	<ul style="list-style-type: none"> Oral exposure is expected to be the most relevant exposure pathway (i.e. ingestion of fish) 	The main route of exposure of humans to PCNs is probably by ingestion of fish [Haskoning 2002].

The overview demonstrates that exposure of the general population to the new POPs and the candidate POPs is in general dominated by food intake. Inhalative and dermal exposure is predominant in professional settings as indicated in the table below.

Table 9-38: Overview of most relevant information on occupational exposure

Substance	Information on occupational exposure	Source
C-PentaBDE	In Sweden, occupational exposure to PBDE has been identified among electronics recycling personnel (Sjødin <i>et al.</i> , 1999) and in technicians responsible for repair and maintenance of computers (Jacobsson <i>et al.</i> , 2002) as well as in nearby soil and sediment (Wang <i>et al.</i> 2005). Also workers in industry manufacturing C-PentaBDE, or polyurethane foam and electronic equipment containing it can be exposed to PentaBDE. There is an extensive literature on such exposures.	[UNEP-POPS-POPRC.2-17-Add.1.English[2]]
C-OctaBDE	Significant workplace exposure expected similar to C-Penta according to EU Risk Assessment Report 2003	[UNEP/POPS/POPRC.3/14]
PFOS	Highest PFOS concentrations of in workers at production plant	
PeCB	Inhalation and dermal contact e.g. at wood treatment plants, dielectric fluid spill and cleanup, MSWI, HWI, magnesium production plants, quitozene production and use.	
SCCP	The short chain length chlorinated paraffins are viscous non-volatile liquids and therefore skin contact is the predominant occupational route of exposure. However, there is a potential for significant inhalation exposure in two use areas. Although there is no information available on the extent of absorption of short chain length chlorinated paraffins following their inhalation, toxicokinetic data indicate that they are likely to be poorly absorbed via the dermal route. According to EC (2000) the number of people occupationally exposed to SCCPs in the EU is not known. However, EC (2000) estimated that the numbers occupationally exposed during all formulation processes (rubber, textiles, paints and coatings and adhesives and sealants) in the EU at that time could be of the order of several thousands. Similarly, it was estimated that the number of people occupationally exposed during the industrial use of paints, adhesives and sealants would be of the order of thousands. Information on releases at the workplace is not available. However, EC (2000) estimated the exposure of workers using the EASE Model (i.e.for inhalation and dermal exposure).	[ERA 2000]; [ECHA 2009]
HBCD	During the production of HBCD, there is a potential for exposure to HBCD during packing, compaction (to form granules), when process operators have to enter the centrifuge or dryers, and, to a lesser extent, during handling of packaged HBCD in the warehouse. Besides, HBCD flame retarded products may cause serious environmental pollution and put workers at risk of exposure when they are produced or disposed of. The occupational inhalation exposure to HBCD is dominated by exposure to airborne dust because of the low vapour pressure, the high melting temperature, the process temperature at the manufacture industrial uses and the particle size of the product. Workers in the industry who may potentially be exposed are primarily those workers who come into direct contact with the pure substance. This includes workers working close to processes emitting HBCD, workers packing the substance and workers transferring the substance to other systems in the chemical industries. For all activities the exposure is strongly influenced by plant conditions and worker habits. Bad hygiene in a plant could lead to high background concentrations. Examples of this are broken bags, dusty pallets and dusty rooms. The presence of effective control measures can also have a great influence on the exposure. Dermal exposure to HBCD may occur in direct handling of HBCD, either indirectly by contamination of surfaces or directly by dermal deposition of airborne dust. Assuming that oral exposure is prevented by personal hygienic measures, ingestion of HBCD does not seem to be a relevant route of occupational exposure, except to the extent that this follows initial inhalation exposure.	[Swerea 2010]; [ECB2008a]
PCP	Occupational exposure to technical PCP mainly occurs through inhalation and dermal contact. Workers handling treated lumber or maintaining PCP-contaminated equipment would be exposed dermally to PCP in solution, and may take up from one-half to two-thirds of their total PCP burden through the skin. The actual concentrations to which workers have been exposed are seldom measured but, where they have been monitored, they have been predictably high. Airborne levels at PCP-production and wood-preservation facilities have	[IEP 2008]

	ranged from several mg/m ³ to more than 500 mg/m ³ in some work areas. The outer layer of treated wood can contain up to several hundred mg/kg, though levels are usually less than 100 mg/kg. These exposures result in concentrations of PCP in the serum and urine that are 1-2 orders of magnitude higher than those found in the general population without known exposure.	
HCBD	Occupational exposure to HCBD may occur during its production and during its use as a solvent, heattransfer liquid, transformer liquid, hydraulic fluid, washing liquor, pesticide or chemical intermediate.	[IARC 1999]
PCN	Severe skin reactions (chloracne) and liver disease have both been reported after occupational exposure to chlorinated naphthalenes. Chloracne was common among workers handling chlorinated naphthalenes in the 1930s and 1940s. The clinical and toxicological symptoms of PCNs are very similar to those caused by PCBs, PCDDs, and PCDFs. Systemic effects resulting in liver disease have been reported only from the inhalation of PCNs, as fumes arising from hot PCN wax processing operations.	[IPCS 2001]

For all new POPs professional exposure is an important aspect as long as production has not been banned, or in case contaminated material is recycled in dust generating processes. This aspect however, is covered by measures and regulations for workplace safety.

Table 9-39: Overview of occupational exposure and/or emission limit values (OEL)

Substance	CAS	OEL	Emission limit TA Luft (Air emission manual Germany)	Reference
tetraBDE	40088-47-9	-		
C-PentaBDE	32534-81-9	-	Exhaust air included emissions in dust form should not exceed values of 0.20 kg/h or 20 mg/m ³	[Gestis 2010]
hexaBDE	36483-60-0	-		
heptaBDE	68928-80-3	-		
C-OctaBDE	32536-52-0	0.1 mg/m ³		[SER 2010]
PFOS	1763-23-1	0.01 mg/m ³ inhalable aerosol (8h; DFG) 0.08 mg/m ³ inhalable aerosol (short term; DFG)		[Gestis 2010]
PFOSF	307-35-7	-		
PeCB	608-93-5	-	Exhaust air included emissions in dust form should not exceed values of 0.20 kg/h or 20 mg/m ³	[Gestis 2010]
SCCPs	85535-84-8	-	Exhaust air included emissions in dust form should not exceed values of: 0.10 kg/h or 20 mg/m ³	[Gestis 2010]
HBCD	25637-99-4 (mixture of 3 diastereomers)	OEL values for dust and mist: Sweden: 5 mg/m ³ USA: 10 mg/m ³		[ECB 2008a]
HBCDD	3194-55-6 (1,2,5,6,9,10-Hexabromcyclododecan)	-	Exhaust air included emissions in dust form should not exceed values of 0.20 kg/h or 20 mg/m ³	[Gestis 2010]
PCP	87-86-5	0.06 mg/m ³ (8h; based on data from DFG, Germany) 0.5 mg/m ³ (8h; BE, FR, PL, ES)		[SER 2010]

Substance	CAS	OEL	Emission limit TA Luft (Air emission manual Germany)	Reference
		(skin), SE; USA, NIOSH and OSHA, UK); 0.05 mg/m ³ (8h; Canada, DK, CH); 0.001 mg/m ³ (8h; HU) 0.1 mg/m ³ (short term; DK); 1.5 mg/m ³ (short term; PL, SE, UK)		[Gestis 2010]
HCBD	87-68-3	0.24 mg/m ³ (8h; DK, CH, USA NIOSH) 0.21 mg/m ³ (8h; BE, Canada); 0.20 mg/m ³ (8h; ES (skin)) 0.48 mg/m ³ (short term; DK)	Organic substance; in exhaust air included emissions should not exceed values of in massflow: 0.10 kg/h or mass concentration: 20 mg/m ³	[Gestis 2010] [SER 2010]
PCN	70776-03-3	substance for which no OEL-value could be established yet		[Gestis 2010]

Sources:

- Gestis (http://bgia-online.hvbg.de/LIMITVALUE/WebForm_gw.aspx)
- ESIS (<http://ecb.jrc.ec.europa.eu/esis/>)
- SER (http://www.ser.nl/en/oel_database.aspx)

The best indicator for human risks can be derived from tolerable daily intake and related levels for the different substances. Such limits and related background information are compiled in Table 9-40 below.

Table 9-40: Overview of tolerable daily intake (TDI), estimated daily intakes and minimal risk levels (MRL) for new POPs and candidate POPs

Substance	TDI	MRL/RfD	Source	Note
TetraBDE (47), PentaBDE (99)	0.26 ng/kg b.w. per day	0.03 mg/kg/day (oral acute)	[ATSDR 2004]	MRL based on the lowest NOAEL of 1 mg/kg/ b.w. per day for rats. UF = 30 (10 for animal to human extrapolation; 3 for human variability). TDI for PentaBDE based on LOAEL of 60 µg/kg b.w. per day and risk assessment procedures for 2,3,7,8-TCDD; [RIVM 2006]
PentaBDE (99)		0.007 mg/kg/day (oral int.)	[UBA 2009a]	
TetraBDE (47) PentaBDE (99)		RfD 0.1 µg/kg bw/day	U.S. EPA	based on a benchmark dose lower confidence limit (BMDL) for neurobehavioural effects
HexaBDE (153, 154)		RfD 0.2 µg/kg bw/day	U.S. EPA	based on a no-observed-adverse-effect-level (NOAEL) for neurobehavioral effects.
DecaBDE (209)		RfD 7 µg/kg bw/day		
C-PentaBDE			[EU Risk Assessment Report, 2000]	Chronical exposure; most sensitive NOAEL of 0.45 mg/kg b.w./d
C-OctaBDE			[UBA 2005]	Most sensitive NOAEL of 2 mg/kg b.w./d
C-PentaBDE		RfD 2 µg/kg bw/day (chronic)	U.S. EPA	based on a (NOAEL) of 1.77 mg/kg bw/day for induction of hepatic enzymes in experimental animals and an uncertainty factor of 1,000.
C-OctaBDE		RfD 3 µg/kg bw/day	US EPA	based on a NOAEL of 2.51 mg/kg bw/day for induction of hepatic enzymes in

Substance	TDI	MRL/RfD	Source	Note
		(chronic)		experimental animals and an uncertainty factor of 1,000.
Total PBDE			[JECFA ⁶³ 2002/2005]	LOAEL 100 µg/kg bw per day. Adverse effects unlikely to occur at lower doses. Current estimates of dietary intake approximately 0.004 µg/kg bw per day, for the sum of all measured PBDE congeners, including the toxic ones.
C-PentaBDE			[EU Risk Assessment Report, 2000]	Chronical exposure; most sensitive NOAEL of 0.45 mg/kg b.w./d
C-OctaBDE			[UBA 2005]	Most sensitive NOAEL of 2 mg/kg b.w./d
PFOS	0.15 µg/kg b.w. per day (PFOS)		[EFSA 2008]	Based on the lowest NOEL of 0.03 mg/kg/ b.w. per day for Cynomolgus monkeys for PFOS and 0.3 mg/kg/ b.w. per day for rats and mice for PFOA. UF = 200 (100 for inter and intra-species differences; 2 for uncertainties in connection to the short duration of the key study and the internal dose kinetics).
	1,5 µg/kg b.w. per day (PFOA)		[UNEP/POPS/ POPRC.3/20/A dd.7]	0.5 - 2 mg/kg bw/day, LOAEL (Goldenthal et al., 1978a); NOAEL rats 0.1 mg/kg bw/day. (Christian et al., 1999); Critical toxicity value of 40.8 mg/kg ww liver
PeCB		0.005 mg/kg/ b.w. per day	[ATSDR 2010]	
SCCPs	100 µg/kg b.w. per day		[WHO 1996]	Based on the lowest 10 mg/kg b.w. per day for rats. UF = 100 (10 for interspecies variation; 10 for intraspecies variation). The value is repeated in the Revised draft risk profile: short-chained chlorinated paraffins [Stockholm Convention 2009].
HBCD (HBCDD)	1 µg/kg b.w. per day		[Haagensen 2007]	
PCP		0.005 mg/kg/day (acute oral) 0.001 mg/kg/day (Int. oral)	[ATSDR 2010]	UF 1000
HCBD	0.2 µg/kg/ b.w. per day	0.2 µg/kg/day (int. oral)	[WHO 2008] [ATSDR 2010]	Based on a NOAEL of 0.2 mg/kg/ b.w. per day for rats. UF 1000 (100 for inter- and intraspecies variation; 10 for limited evidence of carcinogenicity and genotoxicity of some metabolites).
PCN	---	---	---	---

Notes:

TDI = Limit used by WHO

MRL = Estimate of the daily human exposure to a hazardous substance that is likely to be without appreciable risk of adverse noncancer health effects over a specified duration of exposure. These substance specific estimates, intended to serve as screening levels, are used by ATSDR to identify contaminants and potential health effects that may be of concern at hazardous waste sites. [ATSDR website: <http://www.atsdr.cdc.gov/mrls/index.html>]

⁶³ Joint FAO/WHO Expert Committee on Food Additives

RfD= reference dose for chronic exposure; the United States Environmental Protection Agency's maximum acceptable oral dose of a toxic substance. The EPA defines an oral reference dose as: [A]n estimate, with uncertainty spanning perhaps an order of magnitude, of a daily oral exposure to the human population (including sensitive subgroups) that is likely to be without an appreciable risk of deleterious effects during a lifetime

NOAEL = no-observed-adverse-effect level

UF = Uncertainty factor

Acute oral = acute oral exposure (14 days or less)

Int. oral = intermediate-duration oral exposure (15 - 364 days)

In order to assess tolerable levels of POP in waste in the light of potential risks to human health, the potential of new POP in waste to enter the food chain (or drinking water) would be the crucial parameter to know. However this causal chain, so far cannot be reliably quantified.

There a number of studies investigating into transfer rates from environmental compartments into the food chain, but there is generally only general information about the transfer rates from waste into the environment.

In general it is concluded for PBDEs:

- The application of sewage sludge to agricultural land is one of the reasons for detected levels of PentaBDE in food products and can explain the detected levels in vegetables and root crops in experimental studies. Levels in fish and root crops can be the source of exposure to domestic animals like chickens and pigs, and the source of PBDEs in meat products for human nourishment. Available hazard assessment (VCCEP 2003, COT 2004, VKM 2005) conclude that exposure information is inadequate to fully characterize the risks. PBDE concentrations in human tissue in Europe from the early 2000 show low median values on a lipid basis of 0.2-1.0 ng/g LW of PentaBDE (BDE-99) for all countries and relatively small variations. [UNEP/POPS/POPRC.3/20/Add.7]
- The ingestion of soil/dust is an important route of exposure, in particular for toddlers and children. But the median lifelong exposure to PBDEs via ingestion of soil/dust seems only a minor route of exposure when compared with the exposure via food. Harrod et al. 2004 estimate diet at 93% and inhalative share at 7% with a maximum inhalative share however can be up to 35%. Absorption of PBDEs originating from house dust is expected to significant less than from food. In this context the (widely made) assumption that the absorption of PBDEs from soil/house dust lies between 50 and 100 % (Otazo et al., 2005) may easily overestimate the accumulation of PBDEs originating from housedust in the body. [RIVM 2006]. [Wilford et al. 2004] observed a maximum daily human exposure via the inhalation pathway represented 4.1- 4.4% of overall daily intake, and [Mandalakisa et al. 2008] concluded that inhalation generally represented a small fraction of the overall daily exposure to PBDEs (1%); in heavily contaminated office (10 848 pg m⁻³ of total PBDEs) intake from this route (65 ng day⁻¹) may, in some extreme cases, be as important as diet.
- Transposure rate >90% between environment and biota seems to be very high for PBDEs. Biota-sediment accumulation factors between 1 and 3 for Hexa and HeptaBDE on freshwater fish. Highest potential for bioaccumulation for HexaBDE; but significant differences between the control

population and occupationally exposed groups were also reported. In general, fish is a major contributor to dietary intake (around 40% DDI) due to the high PBDE levels. Meat account for around 30%, dairy products and eggs contribute to a lesser degree (< 30%). [UNEP/POPS/POPRC.3/20/Add.6]

- According to a Chinese study [J.-P. Wu et al. 2008] on contamination of aquatic biota at contaminated e-waste disassembly sites net uptake rates of PCB and PBDE are comparable. Bio accumulation factors (BAF) were found to be slightly lower (2.9-5.3) than for PCB (1.2 to 8.4) reflecting reduced bioavailability or enhanced depuration. Also the relationship between log K_{ow} show similarities between PBDE and PCB.
- According to a recent Flemish study [Roosens et al. 2010] PBDE intake via food and dust is dominated by PBDE 209 (DecaBDE) followed by PBDE 47 (tetra) and PBDE 99 (penta).

For PFOS the following information is provided as regards pathways and environmental fate:

- PFOS does not follow the “classical” pattern of partitioning into fatty tissues followed by accumulation, because PFOS is both hydrophobic and lipophobic. PFOS binds preferentially to proteins in the plasma; nevertheless a high level of bioaccumulation can be observed in mammals. Calculated hypothetical BMFs are reported to be 22 – 160; PFOS has shown a high oral uptake (95%) [UNEP/POPS/POPRC.2/1/17/Add.5, 2006].
- A German study [PFOS-Duesseldorf 2010] showed clear correlations between use of contaminated ground water for irrigation and levels in plants and fruits, even if not constituting a direct risk compared to current TDIs.

The information on PeCB in the risk profile is scarce and limited to the following:

- PeCB is persistent in soils and sediments under aerobic conditions (>6 years); considered to have a high bioaccumulation potential [UNEP/POPS/POPRC.3/20/Add.7, 2007].

For SCCP the following information is provided:

- SCCP concentration in sewage sludges have been reported to be as high as 200 µg/g dry wt and was also found in water downstream of sewage treatment plants. [UNEP /POPS/POPRC.5/2/Rev.1]
- In the UK SCCP was also detected in Human breast milk up to 820 µg/kg lipid. In the same study SCCP was also detected in cow’s milk, butter and fish oil. [UNEP /POPS/POPRC.5/2/Rev.1]
- Health Canada states that virtually the entire intake of SCCP for each age group in Canada is from food. [UNEP /POPS/POPRC.5/2/Rev.1]

The exposure route for HBCD includes the following:

- Data on concentrations of HBCD in surface water are available mainly from recipients of major point sources. Due to the very high adsorption and low water solubility, concentrations in the dissolved phase are not expected to be very high. ($< 0.1 \mu\text{g/l}$) [UNEP/POPS/POPRC.5/1]
- HBCD has been analysed and detected in breast milk in Sweden, Norway and Mexico with up to 5.4 ng HBCD/g lw . [UNEP/POPS/POPRC.5/1]
- According to the European Commission a daily intake of HBCD is approximately 33 ng/kg due to frequent consumption of fish [UNEP/POPS/POPRC.5/1]

For PCP the following exposure information are provided:

- Humans may be exposed to pentachlorophenol in occupational settings through inhalation of contaminated workplace air and dermal contact with the compound or with wood products treated with the compound. General population exposure may occur through contact with contaminated environmental media, particularly in the vicinity of hazardous waste sites. Important routes of exposure appear to be inhalation of contaminated air, ingestion of contaminated groundwater used as a source of drinking water, ingestion of contaminated food and soils, and dermal contact with contaminated soils or products treated with the compound. [IEP 2002]
- Biotransformation of some chlorinated hydrocarbons (e.g., lindane, hexachlorobenzene) to PCP also contributes to the human body burden. The ambient air in urban areas typically contains several ng/m^3 , while concentrations in less developed areas are roughly an order of magnitude lower. [IEP 2002]
- Drinking-water concentrations of PCP rarely exceed several $\mu\text{g/litre}$, even in highly industrialized regions, and most are less than $1 \mu\text{g/litre}$. [IEP 2002]
- Fruits, vegetables, and other produce usually contain much less than $10 \mu\text{g/kg}$, but may on occasion exceed this level. Most meat contain similar concentrations of PCP ($10 \mu\text{g/kg}$) but, a few samples, particularly liver, can contain over $100 \mu\text{g/kg}$. Fish skeletal muscle typically contains PCP levels of $4 \mu\text{g/kg}$ or less. Overall estimates of PCP intake from all foods, based on total diet samples in the USA and the Federal Republic of Germany, are remarkably similar, i.e., up to $6 \mu\text{g/person per day}$. [IEP 2002]
- PCP is also present in a wide variety of consumer products, including veterinary supplies, disinfectants, photographic solutions, fabrics, home-care products, and pharmaceutical products. No calculated estimates of the contribution made by consumer products to overall exposure to PCP are available. [IEP 2002]

For HCBd the following exposure routes could be identified

- HCBd has been detected in Canadian surface water, sediments and aquatic organisms but rarely in ambient air. In general, HCBd stays in the environmental compartment into which it is released. [Environment Canada 2004]
- It was considered that, as an approximation, a dose of $0.2 \text{ mg/kg b.w. per day}$ would correspond to the continuous inhalation of air containing 0.6 mg/m^3 of HCBd, equivalent to an air concentration of about 60 ppb . [COT 2000]

For PCN the following information is provided as regards pathways and environmental fate:

- US exposure standards for single PCN congeners vary between 0.1 to 5 mg/m³. [NICNAS 2002]

Based on this information the following aspects are considered most relevant for new POPs flows in the environment:

1. Due to the lipophilic properties all new POPs (except PFOS) strongly adsorb to particles; diffusion and elution is existing but a significant dilution effect can be expected. Disposal at a dedicated landfill strongly reduces this pathway due to the protective function of required sealing layers.
2. Volatilization is not negligible as all new POPs are classified semi-volatile. It leads to long-range transboundary transport and atmospheric deposition. A dilution can be expected and calculated.
3. Transport from ground to surface water exists. However due to the lipophilic properties most of the new POPs strongly adsorb to particles and sediments which largely decreases the mobility so that levels in water are far below levels in soil or sediment and a strong dilution effect can be expected.
4. Transport between different soil layers is not very well investigated, however transport can only be by diffusion or elution by infiltrating rain water. As an up-ward transport from deeper to superficial soil layers would have to work against the effect of infiltrating precipitation the resulting effect can be assumed to be at least extremely small.
5. Irrigation and atmospheric precipitation have a significant effect on top soil concentrations and plant surfaces.
6. Contaminated material (sewage sludge) is considered a major source for potential contamination.
7. Plants react to soil contamination, irrigation and precipitation.
8. Numerous studies have shown a bioaccumulation of POPs in foodstuff from animal origin. A particularly strong relation has shown to exist between elevated contamination in sediments/water and levels in fish and shell fish.

Worst case scenarios for new POPs

Due to lack of information on the causal chain between POP contamination in waste, environment, food and humans the assessment for new POPs within this study is based on the results derived for PCDD/PCDF and PCB and hence on available TDIs making use of similarities in toxicology and physico-chemical characteristics of the substances.

Based on a tolerable daily intake for Σ PCB of 0.01- 0.03 $\mu\text{g}/\text{kg}$ b.w. d and a Dioxin TDI of 1-4pg/kg b.w. d (0.000001-0.000004 $\mu\text{g}/\text{kg}$ b.w./d; difference 4 orders of magnitudes) and differences in relevant food limits (e.g. egg) of 5 orders of magnitude (3 pg/g fat versus 0.3 $\mu\text{g}/\text{g}$ fat), the project team considered specific provisions on management for all wastes necessary when they exceed the following levels:

- 1 ppb for PCDD/PCDF
- 50 ppm (0.05 $\mu\text{g}/\text{kg}$) for PCB and other POPs

Taking into consideration the TDIs/RfD for new POPs of:

- RfD 0.1 $\mu\text{g}/\text{kg}$ bw/day for TetraBDE/PentaBDE (US EPA)
- RfD 0.2 $\mu\text{g}/\text{kg}$ bw/day for HexaBDE (US EPA)
- RfD 2-3 $\mu\text{g}/\text{kg}$ bw/day for Commercial OctaBDE and PentaBDE (US EPA)
- 0.0003 $\mu\text{g}/\text{kg}$ b.w.d for PentaBDE (RIVM)
- 5 $\mu\text{g}/\text{kg}$ b.w.d for PeCB (MRL Canada)
- 0.15 $\mu\text{g}/\text{kg}$ b.w.d for PFOS
- 1.5 $\mu\text{g}/\text{kg}$ b.w.d for PFOA
- 100 $\mu\text{g}/\text{kg}$ b.w.d for SCCP
- 1 $\mu\text{g}/\text{kg}$ for HBCD
- 5 $\mu\text{g}/\text{kg}$ b.w.d for PCP (MRL)
- 0.2 $\mu\text{g}/\text{kg}$ HCBd
- No value for PCN

and assuming comparable health risks and bioavailability as for PCB and PCDD/PCDF the critical levels in waste prone to direct application to soil (e.g. sewage sludge), to unregulated evaporation/precipitation or release to ground and/or surface water could be defined as:

- 50 ppm for PBDE congener groups (if based on US EPA)
- 500 ppm for commercial products (if based on US EPA)
- 0.5 ppm for PentaBDE (if based on Dutch TDI)
- 500 ppm for PeCB (if based on MRL)
- 5 ppm for PFOS (could be considered due to the higher water solubility)
- 100,000 ppm for SCCP
- 1,000 for HBCD
- 5,000 ppm for PCP (if based on MRL)
- 100 ppm for HCBd
- No value for PCN

In this context it is important to note that Austria has established a limit of 100 μg PFOS/kg (0.1 ppm) for sewage sludge used on agriculture agricultural soils.

The project team is well aware of the high level of uncertainty in this assumption and would urgently recommend to thrive for more data and information on the waste/food chain for a better justified risk assessment. For more details see Annex 4 Table 12-7 to this report.

9.2.3 Options for low POP concentration limits

The determination of LPCLs for the different new POPs is based on the methodology developed for the determination of limit values for the initial 12 POPs, and is the result of the limitations from the criteria A-Z discussed in the chapters above. In this context criterion A-D generally represent levels that should be lower than the proposed LPCL, whereas criterion Y and Z represent levels which should not be exceeded by the limit value.

Lower limitation criteria:

- A: Analytical potential
- B: Environmental background contamination
- C: Disposal/recovery capacities
- D: Economic feasibility
- Z: Existing limit values already agreed by the European Union (PBDE, PFOS)

Upper limitation criteria:

- D: Economic feasibility (PBDE, PFOS)
- X: Precautionary principle
- Y: Worst case scenario for human health risks
- Z: Existing limit values already agreed by the European Union

Criterion A (analytical potential) corresponding to the Limit of Quantification (LoQ) that can be reached in laboratories. (Details see chapter 7)

Criterion B (environmental levels) shall take into account the fact that any limit value for waste should be significantly higher than average or background levels observed in the environment (namely soil and sediments). “Hot spot” data indicate the factor by which common background values are likely to be exceeded. The limitation values hence are based on highest background data and an “uncertainty factor” (related to the availability of representative data).

Criterion C (disposal/recovery capacity) considers the D10/R1 capacity. It is taken care that the additional waste amounts which need incineration, deriving from one new or candidate POP does not exceed 2,000 kt/y.

As regards **Criterion Y (potential health impacts)** there are alternative options for limitations depending on the calculation basis and the level of precaution used for PBDE and PFOS. The higher limitation value for PBDE congener groups and for commercial products is based on RfDs applied by US EPA. The lower is a result from a recent risk assessment conducted by the Netherlands and is restricted to PentaBDE (applying corresponding standards as for PCDD/PCDF). The value for PFOS, SCCP, HBCD and HCB are based on the TDI. For PFOS a safety margin one order of magnitude higher than for the other new POPs due to the higher water solubility. For PFOS also a lower limitation value

was established, which is oriented at the limit values set for agricultural application of sewage sludge in Austria, and hence is restricted to sewage sludge intended for agricultural application and not to be understood as general limitation. No values could be found for PCN.

Criterion Z (existing legislation) has the function of lower and of upper limitation criterion for PBDE and PFOS due to existing EU legislation. In consequence the LPCL may generally – unless there are severe arguments in favour to it - not be lower than the value considered as unintentional trace contamination in EU law (10 ppm), and should not be higher than the 1,000 ppm considered as limit for placing on the market of products and substances according to ROHS Directive and REACH.

These limitation criteria are to be understood as borderline and not as recommendation for limit values. The precautionary principle is to be applied in addition requesting to choose the lowest feasible level.

The evaluation of lower and upper limitation criteria for the new POPs leads to the results as compiled in Table 9-41 and Table 9-42.

Table 9-41: Overview of lower limitations for LPCLs for new POPs

	Crit. A	Crit. B	Crit C	Crit. D	Crit. Z
TetraBDE	0.1 ppm	1 ppm	10 ppm	10 ppm	10 ppm
PentaBDE	0.1 ppm	1 ppm	10 ppm	10 ppm	10 ppm
HexaBDE	0.1 ppm	1 ppm	10 ppm	10 ppm	10 ppm
HeptaBDE	0.1 ppm	1 ppm	1 ppm	1 ppm	10 ppm
PFOS	0.1 ppm	1 ppm	10 ppm	10 ppm	10 ppm
PeCB	0.001 ppm	1 ppm	0.003 ppm	0.005 ppm	-----
SCCP	10 ppm	10 ppm	10 ppm	10 ppm	-----
HBCD	10 ppm	100 ppb	5,100 ppm	5,100 ppm	-----
PCP	0.1 ppm	1 ppm	1 ppm	1 ppm	-----
HCBD	0.05 ppm	1 ppm	5 ppb	5 ppb	-----
PCN	0.1 ppm	10 ppm	0.1 ppm	0.1 ppm	-----

Table 9-42: Overview of upper limitations for LPCLs for new POPs

	Crit. Y	Crit. Y (alternativ)*	Crit. Z
TetraBDE	50 ppm	-----	1,000 ppm
PentaBDE	50 ppm	0.5 ppm*	1,000 ppm
HexaBDE	50 ppm	-----	1,000 ppm
HeptaBDE	50 ppm	-----	1,000 ppm
PFOS	5 ppm	0.5 ppm*	1,000 ppm
PeCB	500 ppm	-----	50 ppm
SCCP	100,000 ppm	-----	10,000 ppm
HBCD	1,000 ppm	-----	1,000 ppm
PCP	5,000 ppm	-----	1,000 ppm
HCBD	100 ppm	-----	-----
PCN	-----	-----	-----

* limited to direct application of waste (sewage sludge) to soil

Evaluating the lower and upper limitation criteria for each substance or substance groups leads to the following range of possible LPCLs for the individual substances:

Table 9-43: Conclusions for LPCLs for new POPs based on results from upper and lower limitation criteria

	Lower limitation	Upper limitation	Upper limitation II	Upper limitation III	Conclusion
TetraBDE	10 ppm (crit. Z)	50 ppm (crit Y)	1,000 (crit. Z)		10-50 ppm
PentaBDE	10 ppm (crit. Z)	50 ppm (crit Y)	0.5 ppm (crit Y*)	1,000 (crit. Z)	conflict
HexaBDE	10 ppm (crit. Z)	50 ppm (crit Y)	1,000 (crit. Z)		10-50 ppm
HeptaBDE	10 ppm (crit. Z)	50 ppm (crit Y)	1,000 (crit. Z)		10-50 ppm
PFOS	1 ppm (crit D)	0.5 ppm (crit Y*)	1,000 (crit. Z)		conflict
PeCB	1 ppm (crit D)	50 ppm (crit Z)	500 ppm (crit Y)		1-50 ppm
SCCP	10 ppm (crit. D)	10,000 ppm (crit. Z)	100,000 ppm (crit Y)		10-10,000 ppm
HBCD	5,100 ppm (crit. D)	1,000 ppm (crit. Y)	1,000 ppm (crit Z)		conflict
PCP	1 ppm (crit. D)	1,000 ppm (crit. Z)	5,000 ppm (crit. Y)		1-1,000 ppm
HCBD	1 ppm (crit. B)	1000 ppm (crit. Y)			1-1,000 ppm
PCN	10 ppm (crit. B)				> 10 ppm

* limited to direct application of waste (sewage sludge) to soil;

As illustrated in Table 9-43, the range for potential limits seems to be quite clear except for PeCB, PFOS and HBCD.

For PentaBDE there is a potential conflict between the proposed limits and upper limitation criterion Y(alternative)* (health risks; national TDI) for PentaBDE. As PBDE containing waste in general is not prone to agricultural application and the TDI for PentaBDE is opposed by contrary assessments, this conflict does not seem to be inevitable.

For PFOS there is a conflict between lower limitation Criterion Z and upper limitation criterion Y (as based on a national limit for sewage sludge). This conflict could be solved by requesting a specific limit for sewage sludge.


For HBCD the plastic fraction from construction and demolition (~ 9,700 kt) is most relevant. Only the waste quantities which can be separated from this fraction as pure HBCD containing product contains HBCD at concentrations above 5,100 ppm. The remaining HBCD quantities will be diluted in the plastic fraction from construction and demolition at concentration levels far below 14 ppm. Therefore no capacity shortcomings are expected above an LPCL of 14 ppm.

In the assessment of conflicts it has also to be taken into account that some of the limitations relate namely to specific treatments (crit Y) or shall be evaluated and assessed (crit. Z). Furthermore time trends for waste streams need to be taken into account.

In the following subchapters possible options to set LPCLs are briefly discussed for the individual substances.

9.2.4 Impact on PBDE contaminated wastes


In the following specific symbols are used for the illustration of the upper and lower limitation criteria:

Criterion A: analytical potential  ;

Criterion B: Environmental level:  ;

Criterion C: disposal/recovery potential  ;

Criterion D: Economic impact €

Criterion Y: Potential health impact  ;

Criterion Z: existing legislation §

Figure 9-15 gives an overview of the evaluated criteria of Tetra-, to HeptaBDE and visualises possible conflicts.

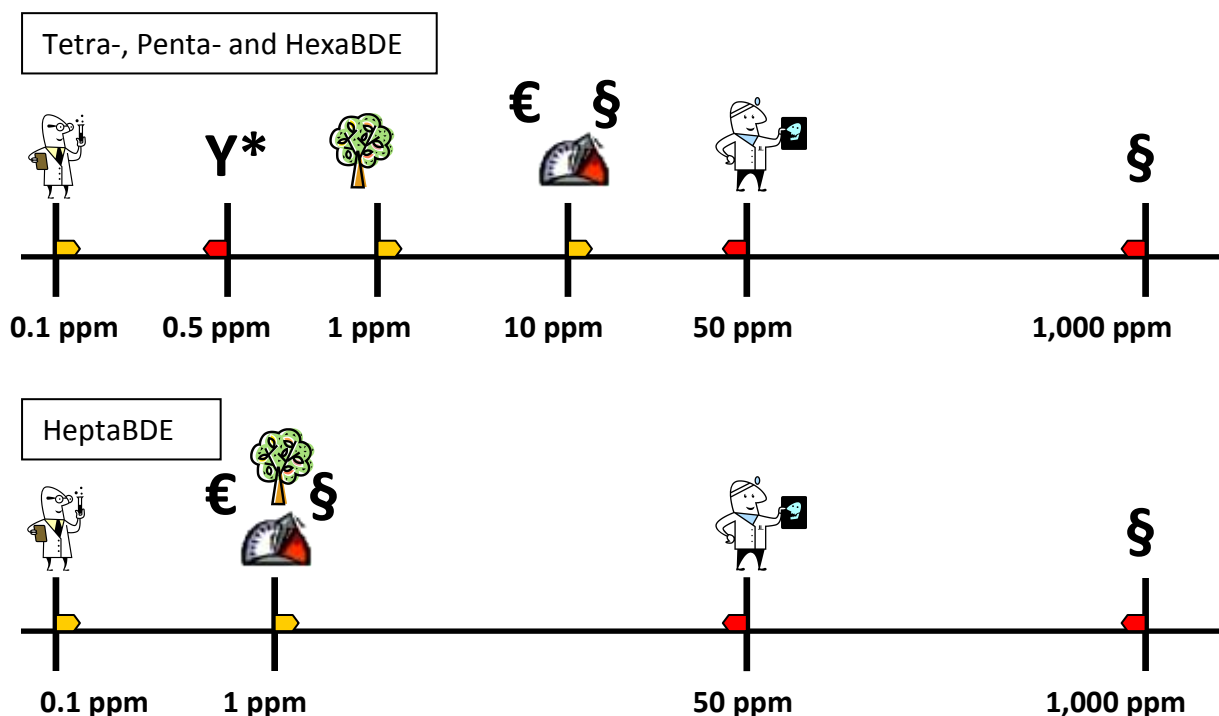


Figure 9-15: Overview of criteria for Tetra-, Penta- and Hexa BDE (Y* only valid for PentaBDE)

Figure 9-15 illustrates that there are no conflicts except for PentaBDE for the upper limitation criterion Y* (health risks; national TDI). However, LPCLs can be set in a range between 10 and 50ppm

for tetra to hexaBDE and between 1 and 50 ppm for HeptaBDE. In the following three levels of LPCL limits are considered for each congener group and 3 options for setting LPCLs are derived. The following tables and figures represent the consequences of different LPCL options.

Table 9-44: Quantity of waste considered as POP waste at different LPCL (TetraBDE)

TetraBDE – Figures indicate the quantity of waste considered as POP waste at different LPCL [kt]					
LPCL	automotive	upholstery	WEEE (Scenario 1)	*WEEE (Scenario 2)	Total waste considered as POP waste
Option 1 (5ppm)	655.9	5,215	-	-	5,870.9
Option 2 (50ppm)	655.9	-	-	-	655.9
Option 3 (150ppm)	-	-	-	-	-

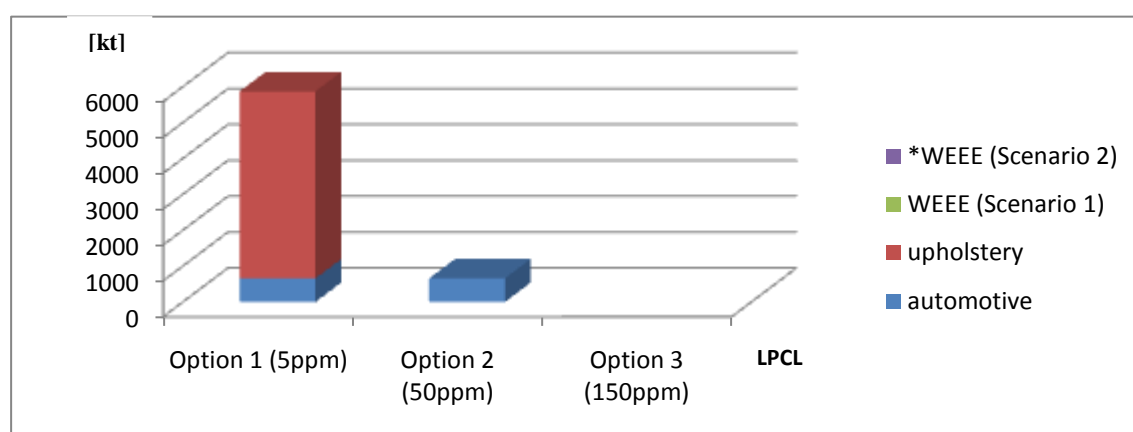


Figure 9-16: Quantity of waste considered as POP waste at different LPCL (TetraBDE)

Table 9-45: Quantity of waste considered as POP waste at different LPCL (PentaBDE)

PentaBDE – Figures indicate the quantity of waste considered as POP waste at different LPCL [kt]					
LPCL	automotive	upholstery	WEEE (Scenario 1)	*WEEE (Scenario 2)	Total waste considered as POP waste
Option 1 (5ppm)	655.9	5,215	105.0	*0.732	5,975.9 (*5,871.6)
Option 2 (50ppm)	655.9	-	105.0	*0.732	760.9 (*656.6)
Option 3 (300ppm)	-	-	-	*0.732	*0.732

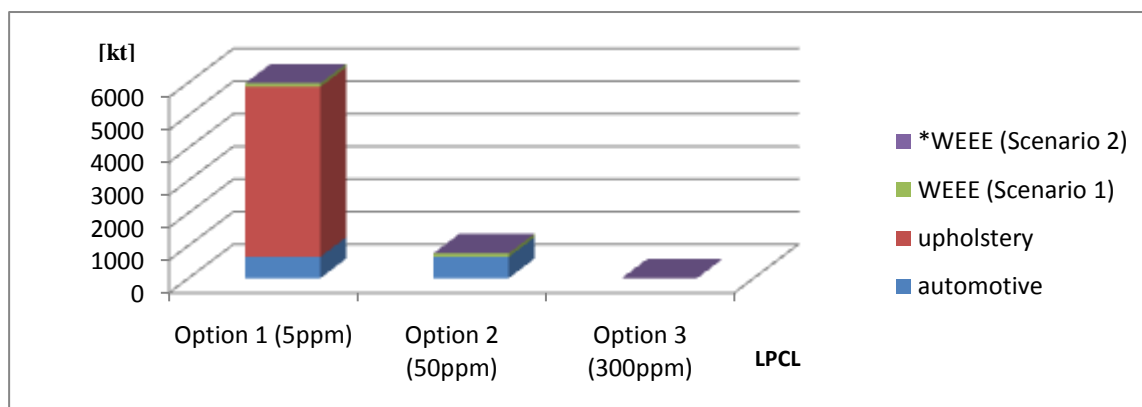


Figure 9-17: Quantity of waste considered as POP waste at different LPCL (PentaBDE)

Table 9-46: Quantity of waste considered as POP waste at different LPCL (HexaBDE)

HexaBDE – Figures indicate the quantity of waste considered as POP waste at different LPCL [kt]					
LPCL	automotive	upholstery	WEEE (Scenario 1)	*WEEE (Scenario 2)	Total waste considered as POP waste
Option 1 (1ppm)	655.9	5,215	105.0	*0.732	5,975.9 (*5,871.6)
Option 2 (10ppm)	655.9	-	105.0	*0.732	760.9 (*656.6)
Option 3 (100ppm)	-	-	-	*0.732	*0.732

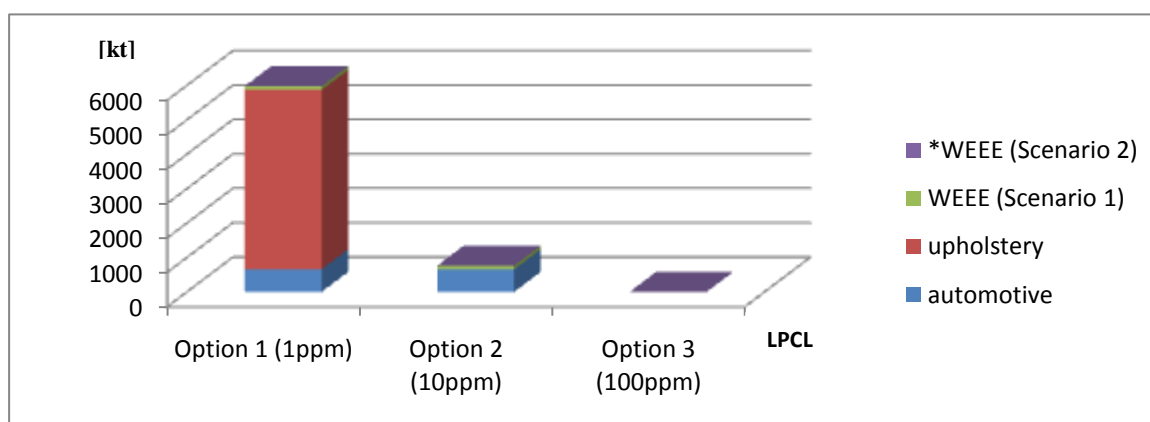


Figure 9-18: Quantity of waste considered as POP waste at different LPCL (HexaBDE)

Table 9-47: Quantity of waste considered as POP waste at different LPCL (HeptaBDE)

HeptaBDE – Figures indicate the quantity of waste considered as POP waste at different LPCL [kt]					
LPCL	automotive	upholstery	WEEE (Scenario 1)	*WEEE (Scenario 2)	Total waste considered as POP waste
Option 1 (0.1ppm)	655.9	5,215	105.0	*0.732	5,975.9 (*5,871.6)
Option 2 (1ppm)	655.9	-	105.0	*0.732	760.9 (*656.6)
Option 3 (500ppm)	-	-	-	*0.732	*0.732

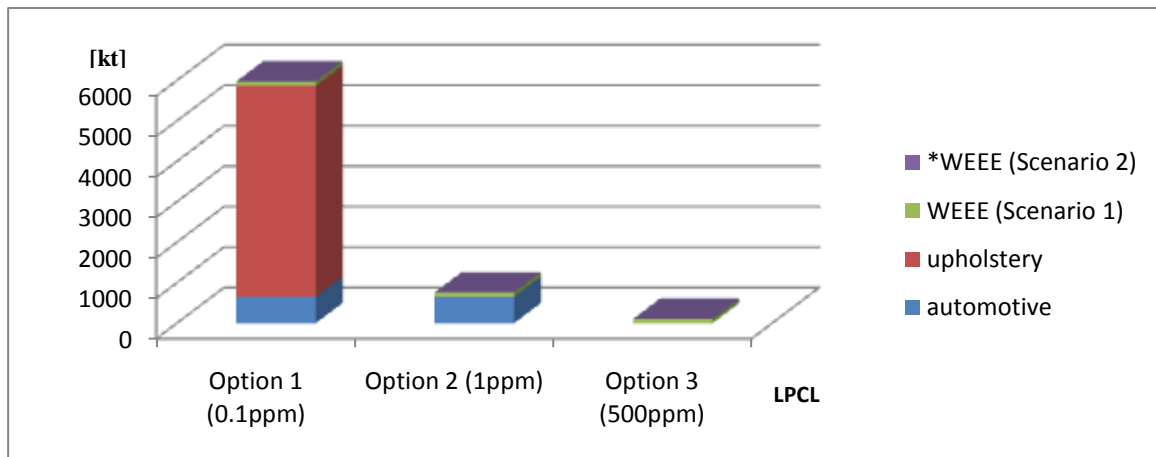


Figure 9-19: Quantity of waste considered as POP waste at different LPCL (HeptaBDE)

Possible options to set LPCLs for the individual congener groups are the following:

Option 1

At LPCL limits for PBDE of

TetraBDE	5 ppm
PentaBDE	5 ppm
HexaBDE	1 ppm
HeptaBDE	0.1 ppm

With Option 1 limits (LPCLs: TetraBDE = 5 ppm, PentaBDE = 5 ppm, HexaBDE = 1 ppm and HeptaBDE = 0.1 ppm) all relevant wastes streams are considered as POP waste, including PUR foams from upholstery (~5,200 kt/y). Under such conditions up to 3,800 kt/y of waste currently landfilled would need to be incinerated or used for energy recovery for the next 5-10 year.

Under Option 1 the limits for Tetra-, Hexa- and HeptaBDE would conflict with lower limitation Criterion Z (existing limit values) and the limit for HeptaBDE in addition would conflict with the lower limitation criterion B (analytical potential). On the other hand there is a potential conflict between the proposed limits and upper limitation criterion Y(alternative)* (health risks; national TDI) for PentaBDE. As PBDE containing waste in general is not prone to agricultural application and the TDI for PentaBDE is opposed by contrary assessments, this conflict does not seem to be inevitable.

Option 2

At LPCL limits for PBDE of

TetraBDE	50 ppm
PentaBDE	50 ppm
HexaBDE	10 ppm
HeptaBDE	1 ppm

At Option 2 limits (LPCLs: TetraBDE = 50 ppm, PentaBDE = 50 ppm, HexaBDE = 10 ppm and HeptaBDE = 1 ppm) WEEE, and ELV is covered, whereas upholstery is not.

Option 2 would request alternative treatment for about 344 kt/y of waste currently expected to be landfilled or recycled due to the limit values of Tetra- to HeptaBDE. Another 51 kt/y are affected due to Penta to HeptaBDE limit values if contaminated WEEE is not separately treated. For more information see chapter 8.3.1, 8.3.2 and 8.3.3.

Under Option 2 only for HeptaBDE conflicts arise regarding criterion Z (existing limit values: unintentional trace contamination in POP regulation).

Option 3

In order to generally not affect shredder light fractions from ELV and WEEE but to concentrate on the separated contaminated PUR foams and ABS plastics, it would be necessary to set the limits at:

TetraBDE	150 ppm
PentaBDE	300 ppm
HexaBDE	100 ppm
HeptaBDE	500 ppm

Option 3 limits (LPCLs: TetraBDE = 150 ppm, PentaBDE = 200 ppm, HexaBDE = 100 ppm and HeptaBDE = 500 ppm) are designed in order to generally not affect shredder light fractions from ELV and WEEE. Such limits reduce the cycling of PBDE only via thermal treatment of shredder residues, or if WEEE and ELV dismantlers manage to effectively separate the contaminated PUR foams and ABS plastics, before the shredding process. Due to the fact that the majority of WEEE shredder light fraction is expected to be already currently incinerated or used for energy recovery. Due to currently implemented post shredder treatment it is expected that the contamination levels in the fibre fraction will increase by a factor of 4 compared to the contamination levels in the shredder light fractions from ELV. As a consequence, also the LPCL could be raised correspondingly. The fibre fraction is destined for material recycling. As a consequence it seems more appropriate to attempt to separate the relevant waste fractions prior to the shredding process and to set comparatively low LPCLs.

Due to the fact, that existing stocks of contaminated material will continuously diminish, option 2 and 3 limits should be reviewed by 2016 at the latest. The expected impacts do not take into account changes in shares of treatment methods towards landfilling or recycling e.g. due to the fact that the 1,000 ppm threshold is applied for shredder light fraction as a basis for recycled products.

Under Option 3 all congener groups would conflict in addition with criterion Y. As PBDE containing waste in general is not prone to agricultural application the conflict with criterion Y can be solved by banning application on agricultural soils.

Against this background, the project team proposes to set LPCLs at a comparatively low level of 10 ppm per PBDE congener group. Alternatively higher LPCLs at 200 ppm (for TetraBDE, PentaBDE and HexaBDE) and 1,000 ppm for HeptaBDE could be set for a restricted time frame in order to facilitate the enforcement of the LPCLs once the LPCLs will be put into force. These levels may cause possible health risks and should therefore be reviewed by 2016 in the light of new relevant information.

9.2.5 Impact on PFOS contaminated wastes

Figure 9-20 gives an overview of the evaluated criteria of PFOS and visualises possible conflicts.

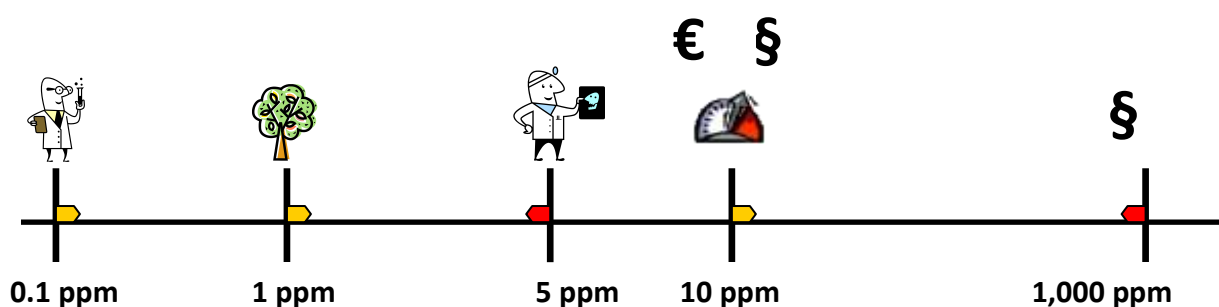


Figure 9-20: Overview of criteria for PFOS

Figure 9-20 illustrates a conflict due to potential health effects at 5 ppm and several lower limitation criteria at 10 ppm. In the following three options for LPCLs and a possible solution of the conflict is discussed. The health risks are related to direct application of sewage sludge to soil. This risk could be avoided by limiting the application of PFOS containing sewage sludge to soil to sludges with a PFOS content below 0.5 ppm.

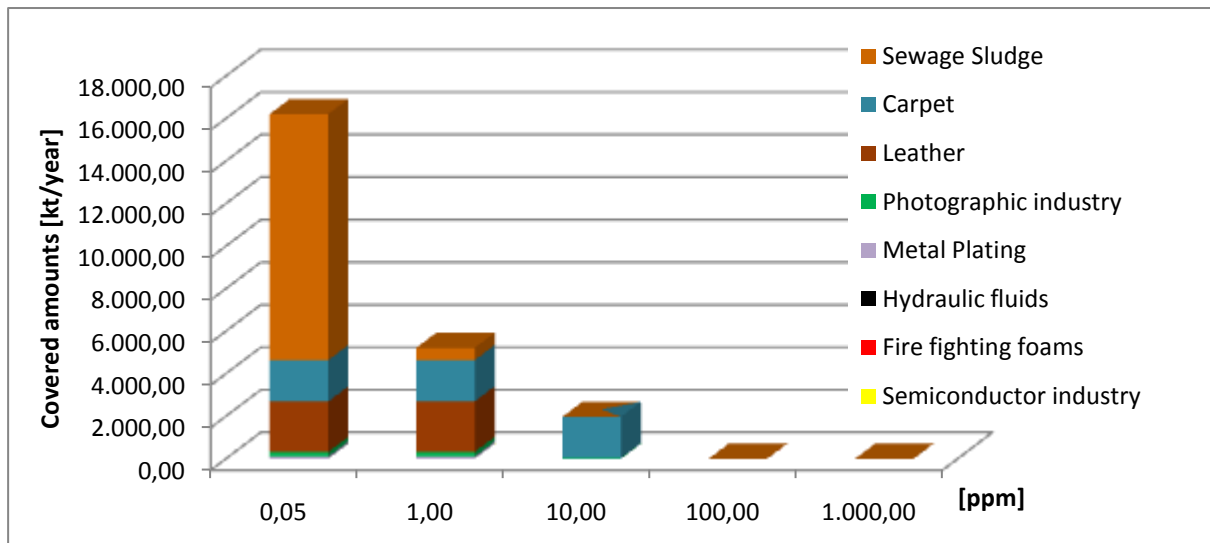


Figure 9-21: Waste amounts considered as POP waste at different threshold limits (PFOS)

Option 1 (1 ppm)

With Option 1 (LPCL of 1 ppm) close to 100 % of the PFOS mass flow, comprising wastes from all sectors related to PFOS would be considered as POP waste, with leather upholstery and carpets in bulky waste constituting the major fractions. Due to the huge quantity of sewage sludge with a low contamination level, these figures however, correspond to only 32% of PFOS waste flows. Based on current treatment shares roughly 5,200 kt/y of the wastes considered as POP waste would need additional thermal treatment.

Under Option 1 the LPCL would conflict with lower limitation Criterion Z (existing limit values), Criterion C (disposal/recovery potential), Criterion D (environmental impact) and at the same with the stricter alternative of upper limitation criterion Y (health risks). The conflict could be solved by limiting the limit for agricultural applications to < 5 ppm. However this limit value would have to be set in the sewage sludge Directive (86/278/EEC) and is therefore out of scope from this report.

Option 2 (10 ppm)

With Option 2 (LPCL of 10 ppm) the ratio of coverage would drop to 12 % in terms of the waste flow but still >90% of the PFOS mass flow would be considered as POP waste. In this case, leather upholstery, wastes from photographic industry (except x-ray picture waste and coating solutions) and chromium sludges would not be considered as POP waste. From the waste considered as POP waste roughly 1,300 kt (carpets and x-ray pictures) would need additional thermal treatment (R1 or D10).

Under Option 2 the LPCL would conflict with the upper limitation criterion Y (health risks; both alternatives). The conflict could be solved by fixing the limit for agricultural applications to < 5 ppm and by requesting a review of the limit by the year 2016 with the objective to tighten it further.

Alternative LPCL 50 ppm

At a LPCL of 50 ppm the situation would remain almost the same with only X-ray waste as additional waste not being classified as POP waste. However, still 1,940 kt/y of waste (thereof 1,938 kt) of contaminated carpets) would need thermal treatment.

Option 3 (100 ppm)

Option 3 (100 ppm) is designed in a way to not cover carpets (almost 50% of PFOS related wastes) the LPCL would need to be lifted at 100 ppm or higher. Less than 1% of all wastes and only about 5 % of the PFOS substance flow would be considered as POP waste in that case. From the 3.1 kt/y of waste considered as POP waste by the limit only 1.24 kt/y (waste from the vacuum distillation from the chromium industry) would need additional thermal treatment, whereas the remaining 1.8 kt/y are already treated by incineration.

Under Option 3 the same conflicts occur and the same solutions can be chosen as for option 2.

Against this background, the project team proposes to set LPCLs at a comparatively low level of 10 ppm. Alternatively higher LPCLs at 50 ppm could be set for a restricted time frame in order to facilitate the enforcement of the LPCLs once the LPCLs will be put into force. These levels are causing possible health risks and should therefore be reviewed by 2016 in the light of new relevant information.

9.2.6 Impact on PeCB contaminated wastes

Figure 9-22 gives an overview of the evaluated criteria of PeCB and visualises possible conflicts.

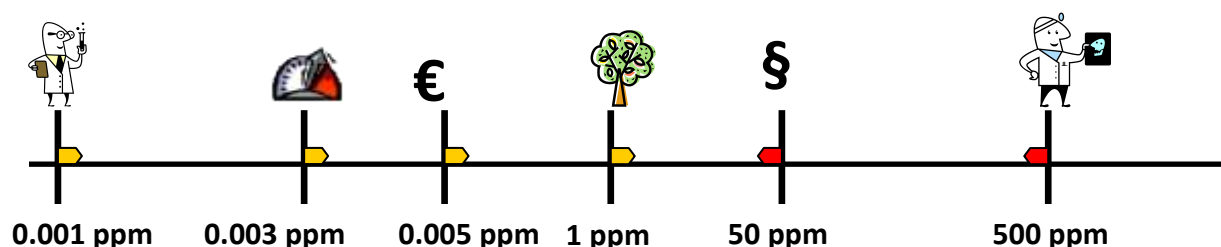


Figure 9-22: Overview of criteria for PeCB

Figure 9-22 illustrates that there are no conflicts between lower and upper limitation criteria. LPCLs can be set in a range between 1 and 50 ppm. In the following two options of LPCL limits are discussed.

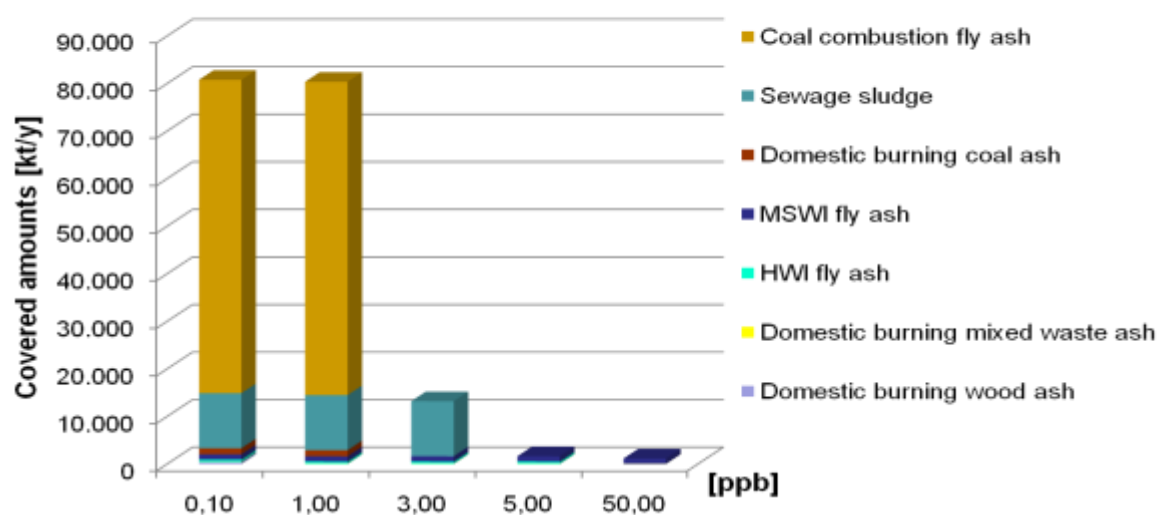


Figure 9-23: Waste amounts considered as POP waste at different threshold limits (PeCB)

Option 1 (1 ppm)

At Option 1 limits (1 ppm) none of the identified waste would be concerned. The limit would correspond to a strict application of the precautionary principle. A lower limit does not appear reasonable due to the limitation criterion related to environmental contamination.

Option 2 (50 ppm)

At Option 2 (LPCL of 50 ppm as set on the amended POP regulation) all of the identified waste would be far from being concerned. A LPCL of 50 ppm does not seem to cause severe health concerns based on the few existing information (Criterion Y). Due to the precautionary principle a lower limit however would be recommended in this case for sewage sludge to be applied on agricultural soils even more as common contamination levels are as low as 5 ppb in average.

Alternative low limit

If a LPCL with impact on waste management was to be chosen it would need to be set at 3 or 5 ppb. Due to the fact that the large part of environmental contamination is via air emissions and deposition such a limit is not recommended.

Against this background, the project team proposes to set LPCLs at a level of 50 ppm, with exception for sewage sludges for which a limit value of 1 ppm should be set. For values higher than 50 ppm no waste types have been identified and therefore no alternatively limit value is provided.

9.2.7 Impact on SCCP contaminated wastes

Figure 9-24 gives an overview of the evaluated criteria of SCCP and visualises possible conflicts.

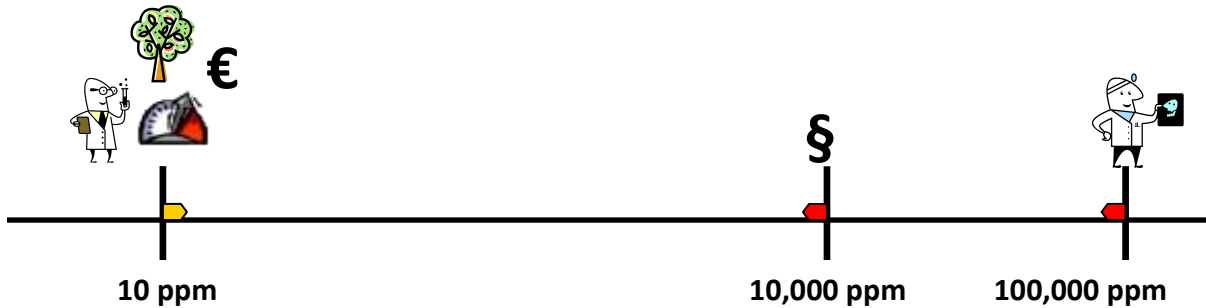


Figure 9-24: Overview of criteria for SCCP

Figure 9-24 illustrates that there are no conflicts between lower and upper limitation criteria. LPCLs can be set in a wide range between 10 and 10,000 ppm. In the following three options of LPCL limits are discussed.

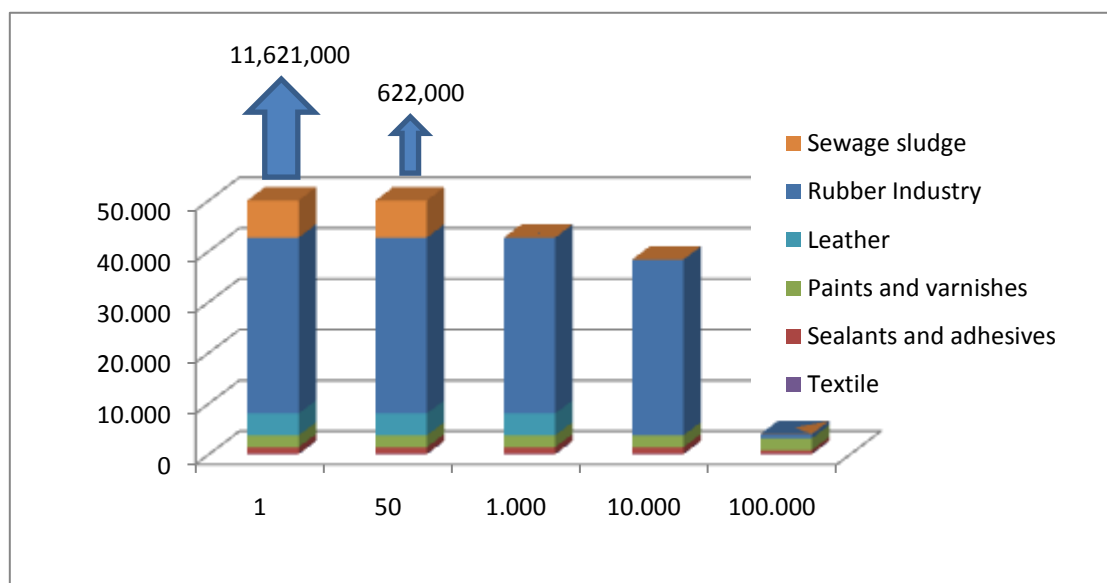


Figure 9-25: Waste amounts considered as POP waste at different threshold limits (SCCP)

Option 1 (50 ppm)

With option 1 all SCCP containing waste are covered except low contaminated sewage sludge which accounts to about 0.5 % of the total SCCP amount in waste.

The affected wastes at 50 ppm sum up to about 621,665 t/y. This amount also includes the high contaminated sewage sludge which has a share of about 93 % of the waste amount, but only represents 1.4 % of the total SCCP amount in the waste.

Option 1 is not in conflict with any of the evaluated criteria.

Option 2 (1,000 ppm)

Option 2 covers all waste types except sewage sludge. The total waste amount is about 42,765 t/y and covers on the other hand about 98.1 % of the investigated SCCP amount

Option 2 is not in conflict with any of the evaluated criteria.

Option 3 (10,000 ppm)

At a set limit value of option 3 wastes from the rubber industry, sealants and adhesives, paint and varnishes and textiles are covered. The waste amount is about 38,359 t/y and covers about 97.2 % of the total investigated SCCP amount in waste. In comparison with Option 2 waste from the leather industry are not covered anymore.

Option 3 is not in conflict with any of the evaluated criterions. Increasing the limit value over 10,000 ppm would result in conflicts with criterion Y (health impact) as well as with criterion Z (existing legislation).

Against this background, the project team proposes to set LPCLs at a level of 1,000 ppm, which covers most of the waste. It is not considered appropriate to propose an alternative higher LPCL.

9.2.8 Impact on HBCD contaminated wastes

Figure 9-26 gives an overview of the evaluated criteria of HBCD and visualises possible conflicts.

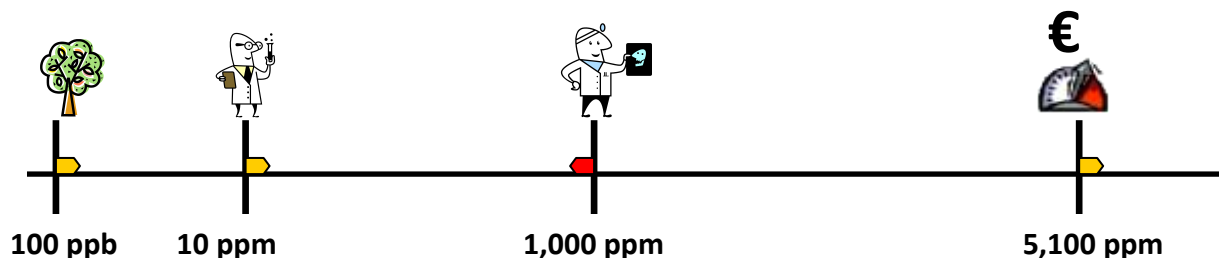


Figure 9-26: Overview of criteria for HBCD

Figure 9-26 illustrates a conflict due to potential health effects at 1,000 ppm and economic and capacity limitation criteria at 5,100 ppm. In the following two options for LPCLs and a possible solution of the conflict is discussed.

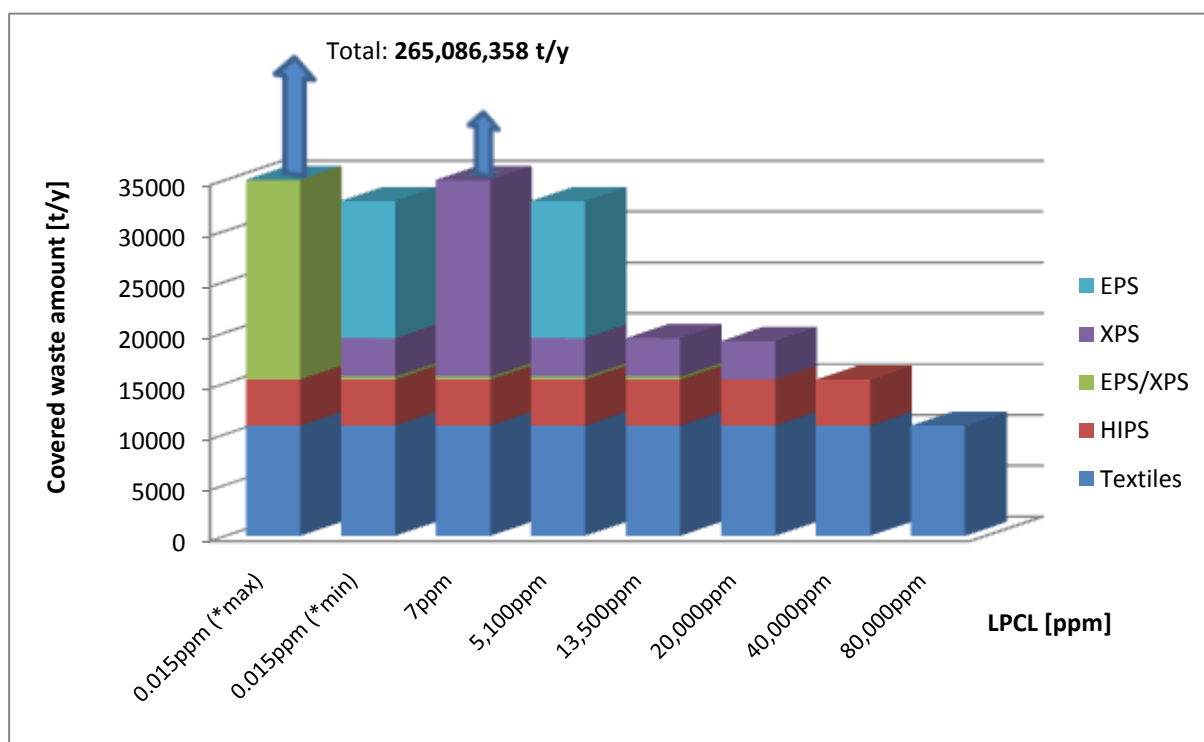


Figure 9-27: Waste amounts considered as POP waste at different threshold limits (HBCD)

Option 1 (100 ppm)

At a limit value of 100 ppm all HBCD waste containing pure HBCD containing products are covered, with a total amount of 32,931 t/y of which 20,846 t/y need alternative treatment. Wastes with lower

HBCD values are mixtures of HBCD containing wastes and HBCD free wastes. For more details see chapter 8.3.7.

Option 1 seems to be in conflict with capacity and economic criteria. In practice this is not the case. Most relevant is the plastic fraction from construction and demolition (~ 9,700 kt). Only the waste quantities which can be separated from this fraction as pure HBCD containing product contains HBCD at concentrations above 5,100 ppm. The remaining HBCD quantities will be diluted in the plastic fraction from construction and demolition at concentration levels far below 14 ppm. Therefore no capacity shortcomings are expected above an LPCL of 14 ppm.

Option 2 (20,000 ppm)

Option 2 covers waste from textiles, HIPS and XPS (C&D) with 9,289 t/y, 3,447 t/y and 1,585 t/y respectively, which need alternative treatment (incineration). EXP/XPS (other than C&D) and EPS (C&D) are not covered anymore.

Option 2 is in conflict with Criterion Y (health impact) and Z (legal).

Against this background, the project team proposes to set LPCLs at a comparatively low level of 100 ppm, which covers all pure waste fractions but also relatively strong diluted waste mixtures. Alternatively a higher LPCL at 1,000 ppm could be set to focus on separated waste fractions and to not exceed the limit value of 1,000 ppm from the REACH Regulation.

9.2.9 Impact on PCP contaminated wastes

Figure 9-28 gives an overview of the evaluated criteria of PCP and visualises possible conflicts.

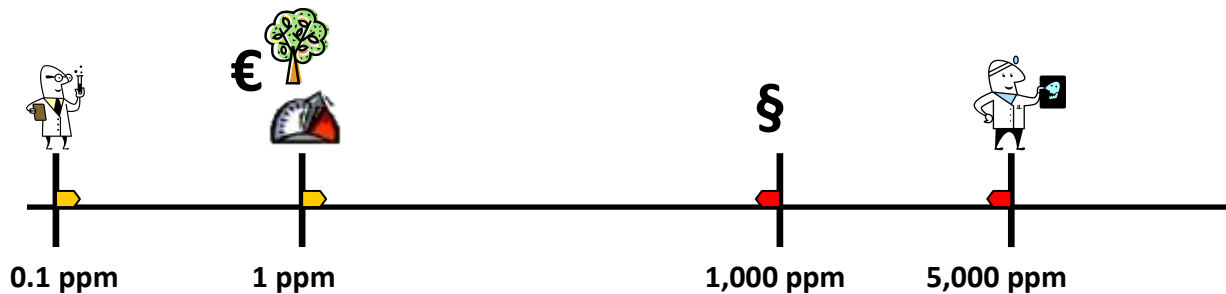


Figure 9-28: Overview of criteria for PCP

Figure 9-28 illustrates that there are no conflicts between lower and upper limitation criteria. LPCLs can be set in a wide range between 1 and 1,000 ppm. In the following three options of LPCL limits are discussed.

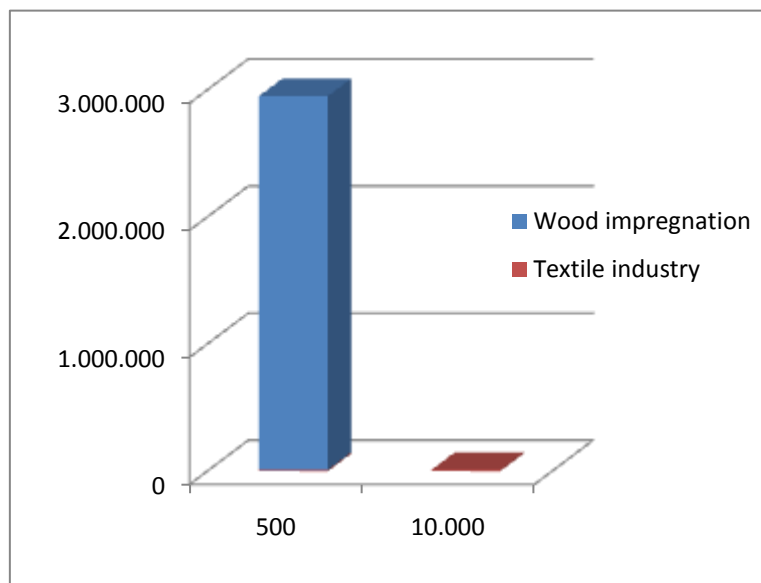


Figure 9-29: Waste amounts considered as POP waste at different threshold limits (PCP)

Option 1 (500 ppm)

With option 1 all PCP containing waste amounting to 2,950 kt/y would be affected. However, the majority of this (~2,940 kt/y) is impregnated wood and already treated in hazardous waste incineration plants. The waste which would need additional incineration arises from textiles. From the total textile amount of 9.6 kt/y about 3.2kt/y are already incinerated in the moment, which leaves about 6.4 kt/y which are currently landfilled and have to be incinerated at that limit value.

For Option 1 there is no conflict with the evaluated criteria.

Option 2 (1.000 ppm)

For a set LPCL level of 1.000 ppm the affected waste amount is reduced to the waste from textiles. As the set LPCL level has no effect regarding a change of treatment for impregnated woods Option 2 has the same consequences as Option 1.

For Option 2 there is no conflict with the evaluated criteria.

Option 3 (30.000 ppm)

At the limit value of Option 3 also textiles are not covered anymore and therefore no wastes are left. Option 3 is in conflict with criterion Y (health impact at 5,000 ppm) and with criterion Z (limit value at 1,000 ppm).

Against this background, the project team proposes to set LPCLs at a level of 1,000 ppm, which covers all waste types which are currently not incinerated. Based on the available data an alternative proposal is not made.

9.2.10 Impact on HCBd contaminated wastes

For HCBd only from one waste stream (sewage sludge) information has been available to evaluate this waste stream. This waste type is sewage sludge with a HCBd concentration of 0.6 ppb. However, this level is too little to be set as it is on the one hand in conflict with criterion B (environmental level), criterion C (disposal/recovery potential) and criterion D (economic impact). Furthermore the effect on waste contaminated with unknown HCBd concentrations cannot be estimated.

Figure 9-30 gives an overview of the evaluated criteria of HCBd and visualises possible conflicts.

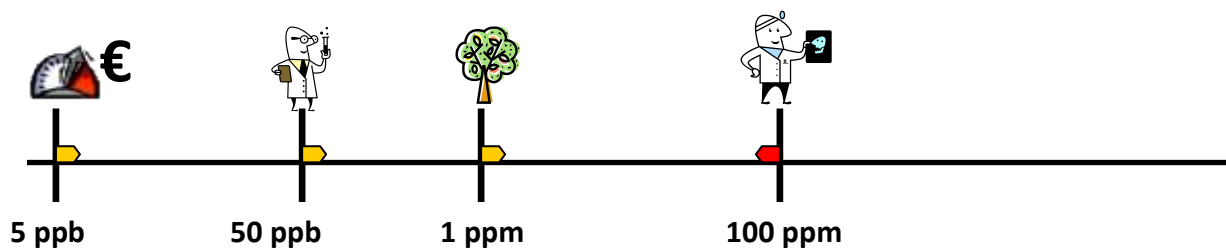


Figure 9-30: Overview of criteria for HCBd

Against this background, the project team proposes to set LPCLs at a comparatively high level of 100 ppm in order to avoid a conflict with criterion Y (health impact). Due to the limited amount of data the possible risks are not possible to be evaluated and the LPCL should therefore be reviewed by 2016 in the light of new relevant information.

9.2.11 Impact on PCN contaminated wastes

Figure 9-31 gives an overview of the evaluated criteria of HCBD and visualises possible conflicts.

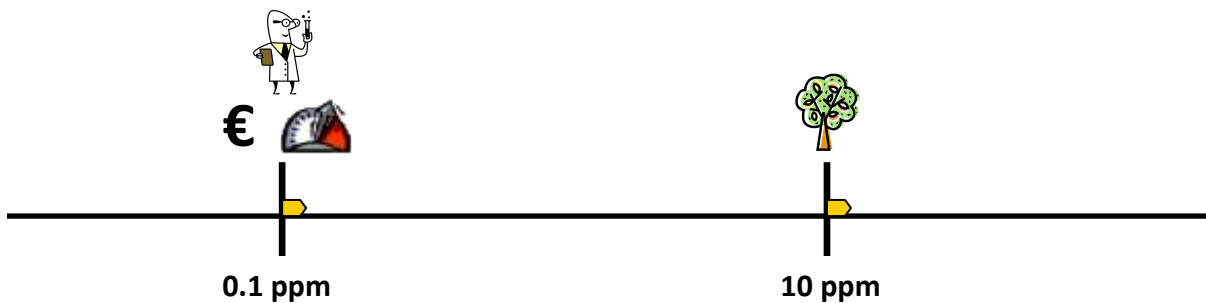


Figure 9-31: Overview of criteria for HCBD

Figure 9-31 illustrates that there are no upper limitation criteria available. LPCLs can be set above 10 ppm. In the following two options of LPCLs are discussed.

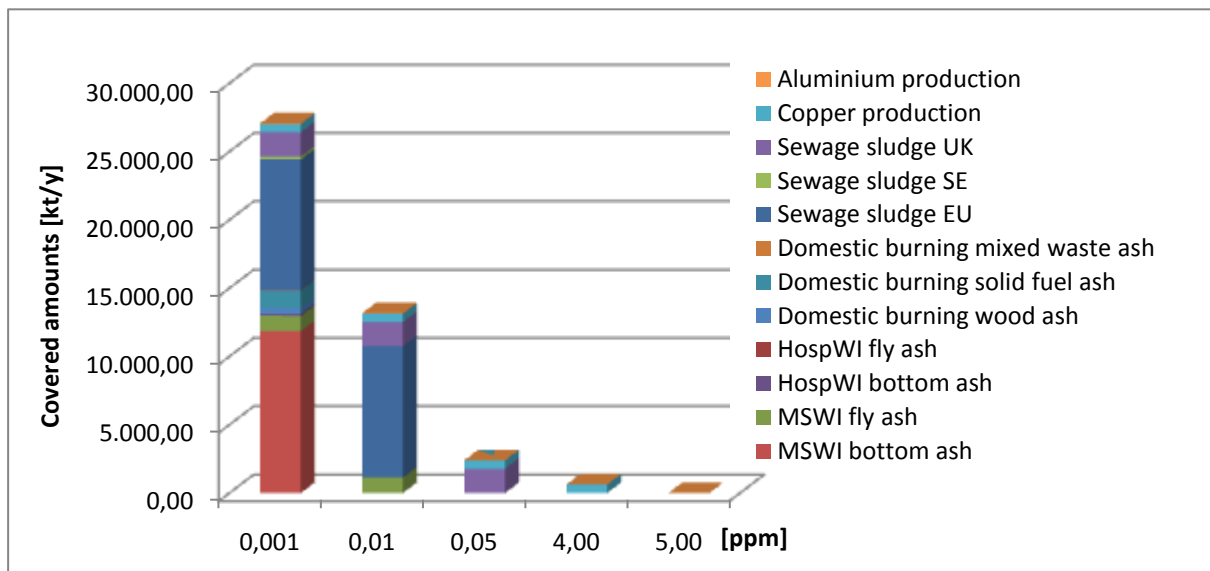


Figure 9-32: Waste amounts considered as POP waste at different threshold limits (PCN)

Option 1 (1 ppm)

At Option 1 limits (1 ppm) two identified waste would be concerned. Option 1 would be in conflict with criterion B (environmental background level).

Option 2 (10 ppm)

At Option 2 all of the identified waste would not cover any investigated waste types containing PCN.

Option 2 is not in conflict with any evaluated criterion.

Alternative low limit

If a LPCL with impact on waste management was to be chosen it would need to be set at 10 or 50 ppb. Due to the fact that the large part of environmental contamination is via air emissions and deposition such a limit however is not recommended.

Against this background, the project team proposes to set LPCLs at a level of 10 ppm in order not to exceed background contamination. No identified waste is covered at that LPCL value. An alternative proposal is not considered appropriate.

Table 9-48: Overview of preferred proposals (LPCL 1) and alternative transitional or restricted proposals (LPCL 2) for LPCLs

Substance	LPCL 1 (ppm)	LPCL 2 (ppm)	Remark
TetraBDE	10	200	LPCL 2 to be reviewed by 2016
PentaBDE	10	200	LPCL 2 to be reviewed by 2016
HexaBDE	10	200	LPCL 2 to be reviewed by 2016
HeptaBDE	10	1000	LPCL 2 to be reviewed by 2016
PFOS	10	50	LPCL 1 and 2 Review by 2016; With stricter limitation for sewage sludge at 0.5 or 5 ppm
PeCB	50	50	With potential limitation for sewage sludge at 1 ppm
SCCP	1,000	1,000	
HBCD	100	1,000	LPCL 2 co cover separated waste types
PCP	1,000		Only including Textile waste
HCBd	100	100	to be reviewed by 2016
PCN	10	10	No identified waste type is covered

9.3 Proposals for maximum POP content levels

The proposals for maximum content limits for new POP, that are presented in this chapter, reflect current knowledge on leaching properties of new POPs from waste and long-term safety of landfills engineered according to the requirements in the landfill directive (1999/31/EC). As current data are scarce the results might change with up-coming knowledge and technical changes, so that a review process and a flexible adaptation of the limits in the Annexes to the POP regulation should be foreseen.

Risks of potential catastrophes have been taken into account in Directive 1999/31 EC and Decision 2003/33/EC. This concern the general requirements for the location of landfills, as well as specific safety assessments required for underground disposal.

The methodology is based on the results from the previous study on POPs and restricts to effects related to new POPs and candidate POPs. Restrictions or limitations in management due to other pollutants already subject to existing legislation are not in the scope of this project but will also influence the management of a number of EWC.

Results for criterion Y (worst case scenario for human health risks)

Following the methodology for the upper limitation criteria, criterion Y and the precautionary principle can support the derivation of a maximum POP content limit.

The disposal routes foreseen as alternatives to destruction or irreversible transformation include landfill for hazardous wastes and underground storage in salt mines or hard rock formations.

Thus it has to be investigated above which POP concentration risks to human health and the environment might occur. This investigation in principle includes the same procedure as for the low POP content limits (chapter 9.2).

As a first step some basic issues have to be clarified in order to be able to identify the most relevant potential pathway to the environment. The possibility to authorise an alternative treatment pursuant to

Annex V is limited to EWC 17: bulky residues from construction and demolition

EWC 10, 16, 19: solid residues from thermal processes in power production, waste incineration and metallurgical industry.

All other wastes have to be treated with D10, D9 or R1 operations if exceeding the low POP content limit.

2. Only disposal of solidified waste at hazardous waste landfills and underground storage is foreseen as potential alternative management operation.
3. The landfill directive (1999/31/EC) applies; (existing sites have to comply with the standards by 2011)

4. Specific technical requirements as set in Annex I to the directive (see description below) have to be adhered to and thus can be assumed as prerequisite for the intended disposal.

Annex I to directive 1999/31/EC contains the following general requirements related to potential risks and protection of the surrounding nature:

1. The location of a landfill must take into consideration requirements relating to:
 - (a) the distances from the boundary of the site to residential and recreation areas, waterways, water bodies and other agricultural or urban sites;
 - (b) the existence of groundwater, coastal water or nature protection zones in the area;
 - (c) the geological and hydrogeological conditions in the area;
 - (d) the risk of flooding, subsidence, landslides or avalanches on the site;
 - (e) the protection of the nature or cultural patrimony in the area.
 (acc. to 1.2 The landfill can be authorised only if the characteristics of the site with respect to the abovementioned requirements, or the corrective measures to be taken, indicate that the landfill does not pose a serious environmental risk.)
2. **Water control and leachate management:** Appropriate measures shall be taken, with respect to the characteristics of the landfill and the meteorological conditions, in order to:
 - control water from precipitations entering into the landfill body,
 - prevent surface water and/or groundwater from entering into the landfilled waste,
 - collect contaminated water and leachate.
 - treat contaminated water and leachate collected from the landfill to the appropriate standard required for their discharge.
3. **Protection of soil and water:** A landfill must be situated and designed so as to meet the necessary conditions for preventing pollution of the soil, groundwater or surface water and ensuring efficient collection of leachate as and when required according to Section 2. Protection of soil, groundwater and surface water is to be achieved by the combination of a geological barrier and a bottom liner during the operational/active phase and by the combination of a geological barrier and a top liner during the passive phase/post closure.

The geological barrier for hazardous waste landfills has to fulfil the following requirements: $K \leq 1,0 \times 10^{-9}$ m/s; thickness ≥ 5 m. In addition to the geological barrier a leachate collection and sealing system consisting of artificial sealing liner and drainage layer ≥ 0.5 m is required so as to ensure that leachate accumulation at the base of the landfill is kept to a minimum.

In addition a surface sealing may be prescribed if the competent authority after a consideration of the potential hazards to the environment finds that the prevention of leachate formation is necessary.

Table 9-49: Recommendations for the surface sealing of landfills according to 1999/31/EC

Landfill category	Hazardous
Artificial sealing lines	required
Impermeable mineral layer	required
Drainage layer > 0.5 m	required
Top soil cover > 1 m	required

According to Annex III of directive 1999/31/EC meteorological conditions and emissions to leachate, surface and groundwater have to be sampled and monitored. Ground water has to be monitored for level (every six month) and for composition (according to expected composition of the leachate and groundwater quality in the area).

According to expert information hazardous waste landfills are generally covered with an surface sealing, so that leaching from the stored to the leachate and the penetration of the basement sealing layers (artificial sealing liner, drainage layer and geological barrier) have to be considered as the critical pathways for discharge to the environment.

Concerning underground disposal Decision 2003/33/EC contains the following additional specifications.

For the acceptance of waste a site-specific safety assessment as defined in Annex A must be carried out. This has to comprise operational and post-operational phase and must include geochemical, geomechanical and geohydrological aspects and has to demonstrate the long-term safety as concerns the isolation to the biosphere.

Assessment of leaching risk via leachate water and penetration through the baseline sealing and geological barrier

Based on the above mentioned provisions and requirements in landfill legislation potential leaching and leachate have been identified as the major parameter for the assessment of long-term safety of landfills and the derivation of maximum POP concentration limits (MPCL).

In expert discussions also some other risks besides leachate were mentioned (e.g. fire inside underground disposal sites; flooding, earthquakes). With respect to underground fires it can be stated that waste acceptance criteria for underground disposal exclude all explosive and flammable wastes. Floods, earthquakes and other catastrophes should be taken into account as far as possible already during the planning of a disposal site (see above general requirements of 1999/31/EC) however cannot be completely excluded.

Leachate in hazardous waste landfill is highly reduced in relation to other types of landfill due to the surface linings minimising infiltration of precipitation and due to the low organic content in most wastes. In addition leachate water is completely collected, treated in a three-step WWTP including activated coal, biological treatment and reverse osmosis, and is largely recycled in order to prevent any pollutants to reach the environment.

For the assessment of potential risks the project team evaluates the potential risk to the environment due to leachate passing the WWTP of the landfill and due to diffusion of dissolved pollutants through the geological barrier. These depend on long-term leaching properties of the waste on technical standards of the landfill and the retention capacity of the WWTP.

1. Approach: The acceptable concentration in waste is calculated on the basis of the leaching rate from waste, the retention capacity of the WWTP and target concentration in surface water or effluent limits—where available—, or on the basis of lowest environmental levels observed.

Such values are as follows are indicated in Table 9-50.

Table 9-50: Overview of target concentrations in surface water, effluent limit concentration, lowest observed environmental concentration

Substance	Concentration	Description	Source
PBDE	0.5 ng/l (0.0005 ppb)	Environmental Quality Standard (EQS) for annual average (AA-EQS) for brominated diphenyl ethers	WFD 2000/60
PBDE	0.5 µg/l (0.5 ppb)	Emission limit for waste water from chemical industry in Austria for technical pentaBDE.	AT
PFOS	1 ng/L	General level all types of water in China	[Risk Profile 2006]
PFOS	up to 0.020 µg/L	Effluents from WWTPs	[Risk Profile 2006]
PFOS	0.038 – 0.152 µg/L	Leachate levels from landfills	[Risk Profile 2006]
PFOS	<1.5-3.8 (1.1) ng/L	Leachate concentrations in landfills	[Bossi et al. 2008]
PFOS	0.01-235 (30.9) ng/L	Leachate concentrations in landfills (treated and untreated)	[J.Busch et al. 2010]
PeCB	No data for water		[Risk Profile 2007]
PeCB	Inland AA-EQS: 0,007 [µg/l]	Environmental Quality Standard (EQS) for annual average (AA-EQS) for inland water for PeCB	WFD 2000/60 (water) and Dir 2008/105
PeCB	13 ng/L (0.013 µg/l)	Concentration in surface waters in Canada 1982	Health Canada www.hc-sc.gc.ca
SCCP	0.168 to 1.75 ng/L in 1999 0.074 to 0.77 ng/L in 2000 (→ average level ~ 1 ng/l)	Aquatic background inland in Canada: Low levels of dissolved total (C ₁₀₋₁₃) SCCPs measured in western Lake Ontario	UNEP/POPS/POPRC.6/11
SCCP	< 100 to 1700 ng/L (→ lower levels ~ 100 ng/l)	Aquatic background inland in Europe: Levels in rivers in England and Wales	UNEP/POPS/POPRC.6/11
HBCD	No data for water	No data on levels in water reported	[UNEP/POPs/POPRC.6/10], [Nordic Com 2007]
HBCD	0.0031 mg/l (3.1 ppb)	PNEC for surface water	Quoted from [Haagensen 2007]
HBCD	0.006 µg/L	Water at point source in Europe; No data for background contamination available	[Arnot et al 2009]
PCP	Inland AA-EQS: 0.4 [µg/l] Other surface waters AA-	Environmental Quality Standards (EQS) for annual	WFD 2000/60 (water) and Directive 2008/105/EC

Substance	Concentration	Description	Source
	EQS: 0.4 [µg/l].	average (AA-EQS) for inland surface water for PCP	
PCP	Average/median PCP concentrations in water are below 1 µg/l (range: non detected to 0.79 µg/l. Typical concentrations for coastal and marine waters are estimated to be 0.07 µg/l	Marine waters, EU	[PCP IEP 2008]; [OSPAR 2001]
HCBD	Inland AA-EQS: 0.1 [µg/l] Other surface waters AA-EQS: 0.1 [µg/l].	Environmental Quality Standards (EQS) for annual average (AA-EQS) for inland surface water for HCBD	WFD 2000/60 (water) and Directive 2008/105/EC
HCBD	0.28 ng/l	Average HCBD concentration in North Sea water (collected 1983/1984)	[Eurochlor 2004]
HCBD	6 ng/l	Mean value of the concentration distribution from statistical analysis of the monitoring data of the EU COMMPS database (1998), which contains more than 10,000 measured HCBD concentrations from rivers of six European countries (Be, De, ES, GR, UK, NL) over the period 1994-1997	[Eurochlor 2004]
PCN	0.89 ng/g	Surface fresh water, River Eman, PCB polluted (there are no other data on environmental levels in water available)	[Järnberg et al. 1997]

The retention capacity of the WWTP depends on the molecular structure of the substances and the combined treatment steps. There are a number of recent studies assessing the leaching rates for the new POPs from non- solidified waste in leaching tests. [Kim et al. 2006], [Busch et al 2010], [Danon-Schaffer, 2010] and one study assessing leaching from solidified waste [Riaz et al. 1998].

In comparison between concentrations in waste and in leachate using most unfavourable conditions, rate in the dimension of 10^{-5} and 10^{-6} have been detected for all substances and homologues. For PBDEs this pertains to granular waste; for PeCB the dilution factor 10^{-6} relates to solidified waste. For PFOS there are no data for direct comparison, but only measured values in real leachate (treated and untreated) from landfills where PFOS containing waste has been deposited in the past.

As regards the retention potential of WWTP there is specific information for PFOS [Busch et al 2010]. In the study it is stated that the major parameter for the concentration in the final leachate is a treatment by reverse osmosis followed by a treatment with activated coal, whereas biological treatment and wet air oxidation do not influence the concentration in the leachate water. As a rough estimate activated

coal, and reverse osmosis reach a combined retention efficiency of >99.5 %. For PFOS a regular exchange and reactivation of the activated coal seems to be an important parameter for efficiency. Hazardous waste landfills generally are equipped with activated coal and reverse osmosis.

In the worst case estimate the project team is using a dilution factor of 10,000 and a retention efficiency of the WWTP of 99.5% (corresponding to a factor for losses into the environment of 0.005)

Based on the available data maximum permissible level for hazardous landfills can be derived by multiplying the relevant concentration level with the dilution factor divided by the WWTP retention efficiency (concentration*100,000/0.005). The results of the calculation for the new and candidate POPs on the basis of relevant concentration levels (see table Table 9-50) are given in the following table.

Substance	Concentration (ppb=µg/l)	Based on	Derived maximum permissible level (ppm)
PBDE	0.0005	Environmental Quality Standard	10
PBDE	0.5	Emission limit	10,000
PFOS	0.001	Environmental level	20
PeCB	0.01	Environmental Quality Standard	200
PeCB	0.007	Environmental level	140
SCCP	0.001	Environmental level Canada	20
SCCP	1	Environmental Level Europe	20,000
HBBD	0.006	Environmental level	120
HBBD	3.1	Environmental target value (PNEC surface water)	62,000
PCP	0.4	Environmental Quality Standard	8,000
PCP	0.8	Environmental level	16,000
HCBD	0.1	Environmental Quality Standard	2,000
HCBD	0.006	Environmental level	120
PCN	0.009	Environmental level	180

2. Approach: Percolation through the divers layers of the basement sealing system is a slow process which depends on various factors such as the content in dissolved organic matter (DOC), physical factors (solubility, porosity, permeability, temperature, etc.) and solid-liquid phase chemistry (pH, redox-potential). However no final modelling for leaching of POPs from disposed waste is available which could be used for the definition of maximum limits. Thus the project team decided to use a worst case approach as a first step to identify the dimension of a potential maximum limit. Due to the precautionary principle the assessment of maximum limits for hazardous waste landfills does not take into account the safety barriers of the landfill itself. This does not apply for underground storage in salt mines, where this aspect is already included in the safety assessment (see above). The acceptable concentration in waste is calculated on the basis of results from corresponding leaching test, resulting in a so-called “dilution factor”, which is compared, where available -to target values in agricultural soils or to lowest background levels observed.

Based on the observed experimental leaching rates for the new POPs from non- solidified waste in leaching tests of 10^{-5} - 10^{-6} of the original contamination in the waste [Kim et al. 2006], [Busch et al 2010], [Danon-Schaffer, 2010], [Riaz et al. 1998]; the project team uses the dilution factor of 100,000/year derived from PCDD/PCDF leaching results, as developed in the previous study. The protective effect of the baseline sealing system is almost not taken into account in this approach.

As target levels for agricultural soil have not been set for new POPs and candidate POPs so far, an extrapolation from recommended values for PCB and PCDD/PCDF based on data for low environmental levels in soil is performed in a first approach.

Environmental background levels have been identified as follows:

PBDE:	0.06-12 ppb
PentaBDE:	<0.2-7 ppb
OctaBDE:	0.065 – 22 ppb
PFOS:	0.03-1 ppb
PeCB:	0.4-1.3 ppb
SCCP:	< 1,000 ppb (occasionally < 10,000 ppb)
HBCD:	< 10 ppb (occasionally < 100 ppb)
PCP:	5 to 30 ppb
HCBD:	<0.2 to 3 ppb
PCN:	0.1 - 15 ppb

These values can be compared to target values of 0.005 ppb for PCDD/PCDF, 0.002 ppm for PCB, and 0.005 ppm for POP pesticides and other POPs. Corresponding environmental background levels are 0.001 ppb for PCDD/PCDF, 0.02 ppm for PCB, 0.001–5 ppm for POP pesticides and 0.0006-0.005 mg/kg (ppm) for other POPs.

Thus the following calculation can be made for acceptable concentrations in waste:

PBDE:	$0.06-12 \text{ ppb} \times 1,000,000 = 60-12,000 \text{ ppm}$
PentaBDE:	$0.2-7 \text{ ppb} \times 1,000,000 = 200 - 7000 \text{ ppm}$
PFOS:	$0.03-1 \text{ ppb} \times 1,000,000 = 30-1000 \text{ ppm}$

PeCB:	$0.4\text{-}1.3 \text{ ppb} \times 1,000,000 = 400 - 1,300 \text{ ppm}$
SCCP:	$1\text{-}1,000 \text{ ppb} \times 1,000,000 = 1,000 - 1,000,000 \text{ ppm}$
HBCD:	$1\text{-}10 \text{ ppb} \times 1,000,000 = 1,000 - 10,000 \text{ ppm}$
PCP:	$5\text{-}30 \text{ ppb} \times 1,000,000 = 5,000 - 30,000 \text{ ppm}$
HCBD:	$0.2\text{-}3 \text{ ppb} \times 1,000,000 = 200 - 3,000 \text{ ppm}$
PCN:	$0.1\text{-}15 \text{ ppb} \times 1,000,000 = 100 - 15,000 \text{ ppm}$

Based on the twofold approach the following alternatives for maximum concentration limits for new POPs are possible:

PBDE:	60-12,000 ppm (Approach 2, background soil)
C-PentaBDE:	200 – 7000 ppm (Approach 2, background soil)
C-OctaBDE:	65 – 22,000 ppm (Approach 2, background soil)
PBDE:	10 ppm (Approach 1, EQS water)
PBDE:	10,000 ppm (Approach 1, based on emission limit for effluents)

PFOS:	30-1000 ppm (Approach 2, background soil)
PFOS:	20 ppm (Approach 1, background water)

PeCB:	400 - 1,300 ppm (Approach 2, background soil)
PeCB:	200 ppm (Approach 1, based on environmental levels in water)
PeCB:	140 ppm (Approach 1, EQS water)

SCCP:	1,000 to 1,000,000 ppm (Approach 2, background soil)
SCCP:	20,000 ppm (Approach 1, background water Europe)
SCCP:	20 ppm (Approach 1, background water Canada)

HBCD:	1,000 to 10,000 ppm (Approach 2, background soil)
HBCD:	120 ppm (Approach 1, background water)
HBCD:	62,000 ppm (Approach 1, PNEC surface water)

PCP:	5,000 to 30,000 ppm (Approach 2, background soil)
PCP:	8,000 ppm (Approach 1, EQS water)
PCP:	16,000 ppm (Approach 1, background water)

HCBd:	200 to 3,000 ppm (Approach 2, background soil)
HCBd:	2,000 ppm (Approach 1, EQS water)
HCBd:	120 ppm (Approach 1, background water)

PCN:	100 to 15,000 ppm (Approach 2, background soil)
PCN:	180 ppm (Approach 1, background water)

These values are to be compared to set limits for PCDD/PCDF of 5 ppm, for PCB of 50 ppm and for POP pesticides and other POPs of 5,000 ppm.

Taking into consideration corresponding TDIs, toxicological properties and applying the precautionary principle the following maximum concentration limits are proposed for exceptional wastes:

- PBDE: 5,000 ppm
- TetraBDE congener group: 2,500 ppm
- PentaBDE congener group: 2,500 ppm
- HexaBDE congener group: 2,500 ppm
- HeptaBDE congener group: 2,500 ppm

- PFOS: 50 ppm
- PeCB: 500 ppm
- SCCP: 5,000 ppm
- HBCD: 5,000 ppm
- PCP: 5,000 ppm
- HCBd: 1,000 ppm
- PCN: 1,000 ppm

This proposal represents the lower edge of the range of possible worst case estimates and is based on a twofold worst case approach to determine the risks from solidified wastes in hazardous landfills to leak into the environment. These values according to legal requirements may only be applied for solidified waste. The calculation is rough and may only give a first indication on the dimension of maximum POP content limit values based on leaching risks. Further studies or modelling work on leaching behaviour (from waste into leachate water) would be needed in order to enable more comprehensive estimates.

As the approach represents a worst case estimation, alternatively MPCLs, which are a factor 10 higher, could be defined without irresponsibly endangering the environment.

According to current knowledge, Annex V wastes⁶⁴ suitable for potential exemptions and relevant for maximum concentration limits occur in relation to PeCB in terms of ashes and slags. PBDE and PFOS contaminated wastes are generally suitable for thermal treatment except of contaminated soils. Information on remediation methods for contaminated soil is given in chapter 6.10.

Table 9-51 compiles the wastes suspected to potentially exceed the maximum limit (MPCL) proposed under Annex V.

Table 9-51: Examples for wastes that may exhibit concentrations above the proposed MPCL limit

Substance	MPCL	Example for possibly relevant waste	Remark
PBDE congener groups (tetra to hepta)	2,500 ppm	Selected PBDE treated PUR foam in upholstery and automotive applications; Selected PBDE treated ABS plastics	Preferred waste management option: destruction by controlled incineration (R1/D10)
PFOS	50 ppm	filter residues (activated carbon, ion exchange resins) desilvered solution x-ray liquid photographic fluids carpets waste aviation fluids PFOS containing FFF	Preferred waste management option: destruction by controlled incineration (R1/D10)

⁶⁴ EWC 17: bulky residues from construction and demolition; EWC 10, 16, 19: solid residues from thermal processes in power production, waste incineration and metallurgical industry.

PeCB	500 ppm	None	
SCCP	5,000 ppm	Conveyor belts Gaskets and hoses Sealants and adhesives Textiles	Preferred waste management option: destruction by controlled incineration (R1/D10)
HBCD	5,000 ppm	EPS/XPS waste HIPS Textiles	Preferred waste management option: destruction by controlled incineration (R1/D10)
PCP	5,000 ppm	Textiles	Preferred waste management option: destruction by controlled incineration (R1/D10)
HCBd	1,000 ppm	n.a.	
PCN	1,000 ppm	None	

9.4 Impacts and Implications on Product and Waste Legislation

LPCLs for new POPs under the POP Regulation may have impacts namely on the WEEE, ELV, REACH and ROHS legislation.

Impacts on the WEEE Directive 2002/96/EC

The WEEE Directive aims at preventing WEEE by promoting reuse, recycling and recovery and is of major interest for the project. For the ten WEEE categories defined in Annex IA, the Directive sets targets for recovery and for component, material and substance reuse and recycling. As regards new POPs (PBDEs contained in C-octaBDE used for ABS plastics in the past) the targets set for categories 3 and 4 are of relevance, requesting:

- A recovery rate of a minimum of 75 % by an average weight per appliance, and
- A reuse and recycling rate for component, material and substance of a minimum of 65 % by average weight per appliance

In this context it has to be noted that, the plastic fraction in WEEE range from 10-40%, and that recycling will not be allowed anymore for a WEEE material classified as POP waste.

According to Annex II of the WEEE Directive, “Selective treatment for materials and components of waste electrical and electronic equipment in accordance with Article 6(1)” as a minimum [...] [e.g. plastic containing brominated flame retardants] has to be removed from any separately collected WEEE.

A concentration limit above which the removal has to be carried out is not specified. The obligation could therefore be considered relevant for any detectable PBDE concentration.

The FAQ Document on the RoHS and WEEE Directive states that Article 6 and Annex II to the WEEE Directive lay down the requirements for the treatment of WEEE. To ensure compliance with the waste framework Directive (75/442/EEC), the treatment shall, as a minimum, include the removal of all fluids and a selective treatment for materials and components in accordance with Annex II. Member States authorities that are responsible for the enforcement of the national legislation implementing these provisions have produced a guidance document on their national provisions. This document covers (i) an interpretation of the term “has to be removed”, (ii) technical guidance regarding treatment of the substances, preparations and components that have to be removed from any separately collected WEEE and (iii) an interpretation of the further provisions of the Annex II. More information can be obtained directly from the Member States applying this guidance document. [EC 2006]

A corresponding guidance document has not been established by now⁶⁵ and it is in the responsibility of the Member States to apply the detection limit or other limit values above which plastic containing brominated flame retardants has to be removed from any separately collected WEEE.

⁶⁵ Personal communication from DG Environment on 3.12.2010

According to the POP regulation, plastic waste containing PBDEs above the proposed LPCLs has to be treated as POPs waste. The POP content has to be destroyed or irreversibly transformed.

Specific exemption from the ban on marketing and use on intermediate use and other specifications, unintentional trace contamination levels and provisions have been set in the POP Regulation as regards Annexes I and III for tetra-, penta-, hexa-, and hepta- PBDE congener groups.

For the relevant PBDE congener groups the exemption allows production marketing and use of EEE within the scope of Directive 2002/95/EC and articles in use, as well as of articles and preparations produced partly or fully from recycled materials or materials prepared for reuse with a contamination of the relevant PBDE congener groups up to 1,000 ppm (0.1%).

A conflict within the POP Regulation would arise, if the POP Regulation would permit the recycling of POP waste above a concentration limit of 1,000 ppm of the relevant congener groups⁶⁶, because otherwise the EEE produced from recycled material could contain the relevant congeners above that limit. As a consequence, the LPCL for the PBDE congener groups has to be set below 1,000 ppm.

A conflict between the POP regulation and the WEEE directive could arise if the Member States would interpret the obligation to remove plastic containing brominated flame retardants from any separately collected WEEE above a congener group specific limit of 1,000 ppm because otherwise the EEE produced from recycled material could contain the relevant congeners above that limit.

The LPCL which will be laid down in the POP regulation could serve as a guidance to Member States as a concentration limit above which plastic containing relevant PBDE congeners should be removed from any separately collected WEEE.

If plastics containing PBDE congener groups above the LPCL will not be removed from separately collected WEEE, a significant share of the current quantities of the relevant PBDEs in products in use will continue to circulate after these products will have become waste and will contribute to continuous releases and exports of these substances as non-waste outside the EU.

The requirement of Annex II of the WEEE Directive to remove BFR containing plastics from any separately collected WEEEs already avoids such a contamination and allows setting the LPCL below 1,000 ppm without an impact on the WEEE Directive.

The relevant congeners are particularly contained in old EEE (mainly produced in the 1990ies until 2003). Most relevant are ABS plastics from Information and Communication Technology and consumer equipment, Cathode Ray Tube (CRT) monitors and CRT TVs. Therefore, the study concludes that

⁶⁶ By adding "recycling" of the relevant materials to the permitted disposal and recovery operations specified in Annex V, Part 1 of the POP Regulation

OctaBDE concentrations have to be expected in plastics from ICT (with the highest levels in monitors), as well as plastics from CRT TVs. This is seen as a consequence from the application of OctaBDE in ABS plastics in the past. Manual separation of the plastics from old computer and TV monitors seems to be one feasible option by which the bulk of the relevant PBDE congeners could be separated from the WEEE waste stream. The targets for reuse and recycling according to the WEEE Directive would be achievable.

If in practice a separation is not feasible, energy recovery of the mixed shredder fraction is considered the preferred solution for the next 5-10 years in order to destroy the relevant PBDE congeners until the vast majority of WEEE plastics containing the relevant PBDE congeners will be eliminated. If the whole mixed shredder fraction would have to be incinerated it will be difficult to achieve the targets for reuse and recycling of the WEEE Directive.

Impacts on the ELV Directive 2000/53/EC

The ELV Directive sets out provisions with respect to end-of-life vehicles aiming at the prevention of waste from vehicles and, in addition, at the reuse, recycling and other forms of recovery of end-of life vehicles and their components.

As regards recycling and recovery the following targets are set pursuant to article 7:

Since 1 January 2006:

- Reuse and recovery of a minimum of 85 % by average weight per vehicle and year.
- Reuse and recycling of a minimum of 80 % by average weight per vehicle and year;
- Not lower than 75 % for reuse and recovery and not lower than 70 % for reuse and recycling for vehicles produced before 1 January 1980

From 1 January 2015:

- Reuse and recovery of a minimum of 95 % by an average weight per vehicle and year
- Re-use and recycling of a minimum of 85 % by an average weight per vehicle and year

In this context it is important to note that vehicles in average consist of 75% metal and 25% plastic compounds including approximately 0.5% PUR foams by weight, but that only part of this PUR foam is prone for C-PentaBDE contamination (mainly PUR foam in cars produced until the year 2000). In order to not interfere with recycling targets it thus is recommended to separate PUR foam from other plastic parts.

Otherwise thermal treatment (energy recovery) of the whole fibre fraction (approximately 5% of the vehicle weight) of the shredder light fraction could be the preferred treatment option. Given the further decline of PBDE contamination in PUR foam, the recycling/reuse of plastic fractions would not cause any problems anymore in 2015. If so, the recycling targets of the ELV Directive would be achievable.

Impacts on the REACH Regulation (EC) No 1907/2006

The REACH regulation contains several concentration limits regarding the placing on the market or other obligations for specific new and candidate POPs (PentaBDE, PFOS, HBCD, PCP, SCCPs). The relevant concentration limits are 0.1% (1,000 ppm) or 1% (for SCCPs). LPCLs in the POP Regulation should be below these concentration limits because otherwise it would be possible that materials resulting from recycled waste streams would contain the relevant substances above the concentration limits established in REACH. As all LPCLs proposed are below the concentration limits established in the REACH Regulation, there are no impacts on REACH expected.

As regards potential impacts and implication on REACH it seems to be important to clarify that the 0.1% threshold by PBDE congener group for articles from recycled material (from Annex I of the POP Regulation) is not to be understood as concentration in waste used as basis material for recycling. Otherwise most of the relevant PBDE congeners currently circulating in products will continue to circulate and operators will not have any incentive to separate or to eliminate PBDE containing material.

Impacts on the ROHS Directive 2002/95

The ROHS Directive contains a ban of placing on the EU market of new EEE if containing PBDE (except DecaBDE) in homogeneous materials at a concentration limit of 0.1% by weight (1,000 ppm). The corresponding LPCLs in the POP Regulation should be below this concentration limit, because otherwise it would be possible that materials resulting from recycled waste streams would contain the relevant PBDE congeners above the concentration limit established in REACH. As all LPCLs proposed are below the concentration limits established in the ROHS Directive, there are no impacts expected.

Impacts on the Waste Shipment Regulation

LPCLs do not have a direct impact on the WSR (1013/2006) as shipping conditions are exclusively related to the Annexes of that Regulation and the EWL. A majority of the waste contaminated with new POPs and candidate POPs are already classified as hazardous in the current annexes. In addition also green listed wastes (e.g. PUR foams, upholstery) intended for recovery in third countries could be regulated via article 36 provisions if the third country authorities become active and ban the import. On the other hand, it might need to be discussed and decided upon whether or not it would be necessary from the point of view of knowledge development and the minimisation objective of the Stockholm Convention, to review the annexes to the Regulation and to reclassify, add or better specify limits in wastes for one or the other of the new and candidate POPs. Furthermore additional guidance on distinction between waste and products and on classification issues might be needed in the light of new POPs in the annexes, and taking into consideration the LPCLs in order to prevent large scale exports from the EU, as soon as obligations for POP waste enter into force.

Impacts on the Sewage Sludge Directive

The proposed LPCLs do not affect sewage sludge as the huge amount of annually produced sewage sludge in Europe would have a very strong economic impact, which would not be in reasonable

relation to the additional amount of destroyed PFOS or PeCB. Even so that high contaminated sewage sludges should not be used for agricultural application a destruction of this waste stream via incineration does not seem necessary and landfilling could be considered to be an adequate treatment. Under the POP Regulation a limit value in the range of 10-50 ppm would be sufficient. However, a more stringent limit value in the sewage sludge Directive is recommended to avoid that PFOS or PeCB is entering the food chain.

9.5 Impacts of proposed limit values on current recycling activities

The proposed limit values could particularly have impacts on the following current recycling activities

Table 9-52: Overview of relevant recycling activities, proposed measures and assessment of corresponding impacts

Relevant recycling activity	Substance of concern	Proposed measure	Econ. Impact	E&H impact	WM impact
Recycling of PUR foams from ELVs possibly containing C-PentaBDE	TetraBDE PentaBDE HexaBDE	LPCL 10 ppm each congener group Separation and destruction of PUR foams from cars built in 2000 and before	Low	Positive	Low
Recycling of WEEE plastics possibly containing C-OctaBDE	PentaBDE HexaBDE HeptaBDE	LPCL 10 ppm each congener group Separation and destruction of ABS plastics from cathode ray computer and TV monitors produced in 2003 and before	Low	Positive	Low
Recycling of EPS/XPS foams possibly containing HBCD	HBCD	LPCL 100 ppm Separation and destruction of EPS/XPS insulation boards as far as reasonably possible	Moderate	Positive	Moderate
Recycling of carpets possibly containing PFOS	PFOS	LPCL 10 ppm Separation from bulky waste and destruction	Low	Positive	Low

The possible impacts of proposed limit values on these recycling activities are discussed in the following subchapters.

9.5.1 Recycling of ELVs

The relevant issue of concern in the recycling of end of life vehicles (ELVs) in the context of the present study is the recycling of PUR foams from ELVs containing C-PentaBDE. The relevant congener groups contained in C-PentaBDE and listed in the POP Regulation are Tetra-, Penta- Hexa- and, to a lesser extent, HeptaBDE (representing about 31%, 56%, 8% and <1% respectively of the commercial mixture; see Table 6-1).

According to a car recycler from the NL (ARN), the PUR foam is purified by removing metals, fabric, and plastic parts and then shredded. The shredded foam is mixed with other types of fabrics to be reprocessed into synthetic fleece and lagging for use in new cars. The PU-foam flakes can also be mixed with glue to be pressed into large blocks that can be cut into shape for a variety of applications, including car seat padding, mattresses, sports mats, cow mats and furniture⁶⁷. ARN has recycled automotive PUR foam for over a decade. According to ARN, also for post shredder applications (which include PUR foam), material recycling applications are currently under development. ARN considers these crucial for achieving the 2015 target for re-use and recycling of a minimum of 85 % by an average weight per vehicle and year according to the ELV directive.

According to ACEA, at European level, recycling of PUR foam from cars as carried out in the Netherlands is not representative; the PUR foam is usually not separated prior to the shredding process. In general, the PUR foam becomes part of the shredder light fraction and is usually subject to R1 (incineration with energy recovery) or D10 (in Member States where the corresponding capacities are available). In order to achieve in 2015 the 85% reuse and recycling target of the ELV Directive it is intended to increase the material recycling of post shredder fractions such as the shredder light fraction. Corresponding developments and investments are currently under way.⁶⁸

According to recent ESTAT data for the year 2008 on materials from shredding of EOL vehicles⁶⁹ the shredder light fraction is recycled (11.9%) used for energy recovery (7.0%) and disposed (81%) in EU 27 with significant differences between the single Member States. Disposal comprises the two waste management options “landfilling” and “incineration without energy recovery” (D10).

The currently implemented post shredder treatment technologies allow the separation of the fiber fraction from the shredder light fraction (SLF = 20% of the ELV weight; fibre fraction of the post shredder treated SLF = 5% of the ELV weight). This fibre fraction contains among other the PU foams. The input of C-PentaBDE containing PU foam in the shredder process will significantly decrease in the coming years due to the continuous phase out of vehicles built in 2000 and before.

The European Automobile producers association (ACEA) informed that C-PentaBDE has not been used in automotive applications since 2000. For the use prior to the year 2000 there is no specific information available. The use may already have stopped before the year 2000.

A measurement campaign could clarify the question which manufactures have used C-PentaBDE and until which year and would enable to separate the PUR foams specifically from the relevant cars and to appropriately manage them. According to ACEA this would not be a feasible approach.

One other possible option would be to manually separate the PUR foams from all cars which were built in 2000 or before (except if it is proven that the PUR foam does not contain the relevant PBDE congener groups above the LPCLs) and to treat them by D10 or R1 in order to destroy the PBDEs.

⁶⁷ http://www.arn.nl/noezp/video_en/start.html

⁶⁸ Personal communication ACEA, 14.12.2010 and e-mail ACEA, 20.12.2010

⁶⁹ Arising in the country and treated within the country; published on 4.10.2010 for the year 2008

The identification of the cars built in 2000 or before and the possible option of manual separation of the PUR foams is not a technical problem. The related costs are considered to amount to approximately 13 € per car⁷⁰ and would only arise for cars built in 2000 and before. As a consequence, C-PentaBDE could be pragmatically separated from the waste stream and it is expected that the contamination level of the remaining shredder light fraction or the fibre fraction from ELVs would decrease to a level of zero or close to zero. Analytical measurements would usually not be required (except if it is intended to proof that specific PUR foam does not contain PBDEs above LPCLs).

Statistical data from Germany show that about 70% of ELVs are 15 years and younger, that the share of cars which are 16 years and older is approximately 30% of ELVs, that approximately 5% of ELVs are older than 20 years and that only an insignificant share of ELVs is older than 25 years. Therefore in 2016 it would be necessary to remove the PUR foams from about 30% of ELVs. This share would decrease to about 5% by 2020 and would become insignificant by 2023 (<1%).

Considering a number of 6.21 million ELV in EU 27⁷¹ and average costs of 13 Euros per car for the separation of the PUR foams, corresponding costs at EU 27 level for the separation of the PUR foams in 2016 would amount up to approximately 23 million Euros in 2016 (29% of ELVs affected) and 4 million Euros in 2020 (4.9% of ELVs affected). After 2020 the costs will further decrease and become insignificant. If the measure would be implemented by 2012, the expected cost impact would be up to 63 million € (78% of ELVs affected). It has to be noted that these cost estimates are a worst case estimate because (1) PUR foams only have to be separated if the SLF or fibre fraction is not intended for incineration (which is currently usually the case) and (2) the number of affected cars could be significantly reduced by identifying the relevant ELVs which are equipped with C-PentaBDE containing PUR foams more specifically than by the building year 2000 and before (see following paragraph).

Costs of 13 Euros per car for the separation of the PUR foams can be considered very low compared to other car related costs such as investment costs (e.g. on average ~ 15,000€) and operational costs (expected in the same dimension as investment costs or higher). The corresponding cost ratio would therefore be below 0.05%.

According to a telephone conversation with ACEA on 22.03.2011, ACEA will try to obtain more specific data on the use of the relevant BDE congeners in automobile manufacturing. If it will be possible to specify by when specific car manufacturers have stopped the use of C-PentaBDE, it may become possible to significantly reduce the number of affected cars. This would easily be feasible e.g. by establishing a list of manufacturers and/or cars which specifies for the individual manufacturer and/or car the year until which the PUR foams would have to be removed.

The removal of PUR foams from cars built in 2000 and before would only marginally affect the achievement of the 2015 reuse and recycling target according to the ELV directive due to the following reasons:

⁷⁰ ARN paid a fee of €12.70 per ELV for the manual dismantling of PUR foams until the end of 2010. This fee was based on modeling of manual labour, real estate, machinery and miscellaneous costs.

⁷¹ According to ESTAT data the number of ELVs in EU 27 in 2008 amounted to 6,211,520 ELVs

- The separation would only be necessary in those cases where the PUR foam is not finally thermally destroyed, i.e. only if it is intended to landfill the PUR foam or to re-use/recycle it. This is currently in the majority not the case as the ELV shredder light fraction is usually subject to R1 or D10 (according to ACEA)
- The separation would only be necessary for cars built in 2000 or before (or even fewer cars if it will be possible to reduce the number of affected cars as described above). Considering an average age of ELV of 15 years, the number of relevant ELVs is steadily decreasing. By the year 2015 the separation of PUR foams would only be necessary for 29% of the ELVs for about 5% in 2020 that the number of relevant cars would then decrease to insignificant amounts. The PUR foams from the majority of cars could then be recycled.
- The mass of the PUR foam of one car (6 to 7 kg) contributes only about 0.5% to the mass of a car. Considering that only the cars built in 2000 and before are concerned, the impact on the reuse and recycling quotient of ELVs would be negligible ($0.5 \times 29\% = \sim 0.15\%$) and would even decrease and phase out over time.

Against this background it is proposed to set an LPCL for each of the relevant PBDE congener groups at 10 ppm and to separate and destroy PUR foams from cars built in the year 2000 or before if the PUR foam is intended to be recycled or to be landfilled in the corresponding process. This would be related to the following impacts:

Proposal →	LPCL for each of the relevant PBDE congener groups at 10 ppm and separation and destruction of PUR foams from cars built in the year 2000 or before if the PUR foam is intended to be recycled or to be landfilled in the corresponding process.	
Substance →	TetraBDE, PentaBDE, HexaBDE, (HeptaBDE)	
Impact	Assessment	Justification
Economic impact	Low	Only a limited number of cars would be concerned PUR foams from the majority of cars could be recycled Costs for the possible option of manual separation could be supported The average costs per car about 13€ can be considered very low compared to other car related costs such as investment costs and operational costs The annual cost impact could be considered relevant in the beginning (up to ~63 million € in 2012) but will become low in 2016 (up to ~23 million €) and insignificant after 2020.
Environmental/Health impact	Positive	The POPs would be destroyed and thus withdrawn from the eco-cycle. Hence it can be assured that they will not become subject to LRT and bioaccumulate and cause adverse environmental and health effects.
Impact on waste management	Low	Easy identification and separation of the relevant PUR foams. Usually no analytical measurement required.
Compatibility with REACH	✓	LPCL is below the 0.1% concentration limit
Compatibility with WEEE	n.r.	n.r.
Compatibility with ELV:	✓	Negligible impact on the reuse and recycling quotient of ELVs; The impact would decrease and phase out over time.
Compatibility with ROHS	✓	LPCL is below the 0.1% concentration limit

9.5.2 Recycling of WEEE

The relevant issue of concern in electronics recycling in the context of the present study is the recycling of plastics from WEEE containing C-OctaBDE. The relevant congener groups contained in C-OctaBDE and listed in the POP Regulation are Penta- Hexa- and HeptaBDE (representing about 6.25%, 6.15% and 35.40% respectively of the commercial mixture; total: 47.8%; see Table 6-9).

According to the EERA⁷² the interim report of the present study ... proposes threshold values of certain legacy substances – particularly PBDE's - that will lead to a highly cost-ineffective recycling of WEEE, jeopardizing the technological developments as well as investments in this field ...".

According to EERA, "lowering the threshold limits for many substances beyond the ROHS and REACH thresholds will make it uneconomically if not impossible to recycle E-Waste plastics for re-use in new electronic equipment in an economically viable way."

It has to be noted that the limit values under REACH cannot be compared to the proposed limit values for the POP regulation due the following reasons:

- the PentaBDE congener group or "Diphenylether, pentabromo derivative (C₁₂H₅Br₅O)" which represents about 6% of C-OctaBDE which is listed under REACH is also listed in the POPs Regulation as PentaBDE meets the POPs criteria on its own. Therefore PentaBDE will be removed from REACH Annex XVII.
- the OctaBDE congener group or "Diphenylether, octabromo derivative (C₁₂H₅Br₈O)" as listed under REACH is not listed in the POPs Regulation.

The limit value under RoHS (current recast proposal of the ROHS Directive; status 30.11.2010) restricts the use for PBDEs above a maximum concentration value of 1,000 ppm (0.1%). The limit value is related to the sum concentration of PBDE congeners. A direct comparison with the limit values that are to be established for the POPs Regulation is therefore difficult. However, considering the typical composition of C-OctaBDE a limit value of 1000 ppm corresponds to approximately 60 ppm PentaBDE congeners (~6%), 60 ppm HexaBDE congeners (~6%) and 350 ppm HeptaBDE congeners (~35%). If a WEEE contains end of life products which where flame retarded with C-OctaBDE and the contamination of the WEEE is at a level of 1000 ppm this means that the WEEE contains 1000 ppm of PBDEs (sum all congeners) including ~470 ppm of the relevant congener groups.

Two alternative sets of limit values have been proposed for these congener groups. One set of limit values ranges slightly below (precautionary approach; 10 ppm per congener group) and one ranges slightly above (transitional approach; vs 200 ppm per congener group and 1000 ppm for the HeptaBDE congener group) these values derived from the limit value in the ROHS Directive.

Table 9-53: Compilation of limit values proposed in the interim report for PDBE congener groups relevant for C-OctaBDE and of limit values theoretically derived from limit values established under RoHS.

⁷² European Electronics Recyclers Association, Comments on interim report received on 26.11.2010

Congener group	Limit values proposal 1 (precautionary approach)	Limit values proposal 2 (transitional approach; review 2016)	Limit values for C-OctaBDE congener groups; theoretically derived from RoHS
PentaBDE	10	200	60
HexaBDE	10	200	60
HeptaBDE	10	1000	350

These explanations demonstrate that the proposed limit values are in the range of the existing limit values and do not propose a substantial lowering of the threshold limits established in the RoHS Directive.

According to EERA “The current threshold values of REACH and RoHS are perfectly possible targeted threshold values.” If this is true for the limit values under RoHS it can be stated that the proposed limit values reflect the state of the art of current recycling techniques and it is not expected that the proposed limit values will create unacceptable burden to the electronic recycling industry.

EERA states that technology to separate brominated flame retardants from plastics - so that plastics recycling from complex mixtures of WEEE becomes possible - does exist and is available in Europe. It can be assumed that such technology is already in place in order to comply with the legal requirement laid down in the WEEE directive according to its Annex II which requires that, as a minimum, e.g. plastic containing brominated flame retardants has to be removed from any separately collected WEEE.

The successful separation of brominated flame retardants enables the destruction of the waste fraction contaminated with PBDEs by means of incineration with energy recovery and the recycling of the remaining plastics from WEEE including economic and environmental benefits.

According to information from a WEEE recycler, the separation of plastics containing BFRs does not cause technical problems and can for example be carried out by using the sliding spark spectral analyses (see Table 7-1). This screening technology enables a fast and easy detection of BFRs at concentration levels down to 1% bromine content. Considering that typical contamination levels of relevant EEE with C-OctaBDE ranges between 12 and 22% this is sufficient to determine intentionally used C-OctaBDE. On the basis of this technology out of 1000 t of WEEE approximately 70 tonnes of BFR containing plastics are separated in practice. The investment cost of around 4,000 € per detector is acceptable. The hand held detector can be operated by a basically trained factory worker. According to a recycler with a throughput of approximately 1000 t WEEE per year 1 to 2 workers need to be employed in order to carry out the separation. Therefore significant operational costs may arise for the separation of plastics containing BFRs according to the WEEE directive. This may cause significant problems for small recycling facilities.

It has to be kept in mind, that the separation of BFR containing plastics from separately collected WEEE is an obligation arising from the WEEE Directive and not an impact of a limit value to be established under the POP regulation for relevant congener groups of C-OctaBDE.

The relevant PBDE congeners from the use of C-OctaBDE are particularly contained in old EEE (mainly produced in the 1990ies until 2003). Most relevant are ABS plastics from Information and Communication Technology and consumer equipment, Cathode Ray Tube (CRT) monitors and CRT TVs. Manual separation of the plastics from old computer and TV monitors seems to be one feasible option by which the bulk of the relevant PBDE congeners could be separated from the WEEE waste stream. The targets for reuse and recycling according to the WEEE Directive would be achievable. In conclusion, impacts on current WEEE recycling activities due to the proposed LPCLs are not expected.

Against this background it is proposed to set an LPCL for each of the relevant PBDE congener groups at 10 ppm and to separate and destroy ABS plastics from cathode ray computer and TV monitors produced in 2003 and before if the ABS plastic is intended to be recycled or to be landfilled in the corresponding process. This would be related to the following impacts:

Proposal →	LPCL for each of the relevant PBDE congener groups at 10 ppm and separation and destruction of ABS plastics from cathode ray computer and TV monitors produced in 2003 and before if the ABS plastic is intended to be recycled or to be landfilled in the corresponding process	
Substance →	PentaBDE, HexaBDE, HeptaBDE	
Impact	Assessment	Justification
Economic impact	No impact	<ul style="list-style-type: none"> Only a limited number of EEE products to be separated Moderately increasing operational costs are an impact of the WEEE directive obligation to separate BFR containing plastics and not of the LPCL set in the POP Regulation
Environmental/Health impact	Positive	<ul style="list-style-type: none"> The POPs would be destroyed and thus withdrawn from the eco cycle. Hence it can be assured that they will not become subject to LRT and bioaccumulate and cause adverse environmental and health effects.
Impact on waste management	Low	<ul style="list-style-type: none"> Easy identification and separation of the relevant EEE products No analytical measurement required in addition to obligations from WEEE
Compatibility with REACH	✓	<ul style="list-style-type: none"> LPCL is below the 0.1% concentration limit
Compatibility with WEEE	✓	<ul style="list-style-type: none"> Plastic containing BFRs have to be separated anyway due to the WEEE directive
Compatibility with ELV	n.r.	<ul style="list-style-type: none"> n.r.
Compatibility with ROHS	✓	<ul style="list-style-type: none"> LPCL is below the 0.1% concentration limit

9.5.3 Recycling of EPS/XPS insulation boards for construction

The relevant issue of concern in the in the context of the present study is the recycling of EPS/XPS insulation boards containing HBCD. In the relevant products, the substance is present in average concentrations around 0.5% for EPS and 2% for XPS. Current recycling rates are about 7.5% for EPS products expected to increase up to 10%. Concerning XPS, recycling is currently not a relevant end of life option but may become relevant in the future [PS foam 2011].

An LPCL below 5000 ppm (0.5%) will have as an effect, that the recycling of waste EPS and XPS boards containing average levels of HBCD will not be available for recycling as they will have to be treated

according to waste management options as laid down in Annex V, part 1 of the POP Regulation.

The potential contamination of end-of-life materials could be a barrier to the recycling of products. Recycling of products containing POPs will impact on the content of POPs in new products recycled from the POP containing products. Depending on the recycling process, the POP content will impact on the quality (here content of POPs) of the new products. Recycling should be performed in a way that minimises the carry over of POPs into new products or in a way that the POPs content in the new products will be manageable at all life stages and disposal of the recycled product. The minimisation of the carry over of POPs lies usually also in the interest of manufacturers. Manufacturing processes require high degree of process control and it is understandable that companies would not want to sacrifice the quality of their products by introducing unknown compounds [BNIW02 2008].

According to the PS foam industry, it will be difficult to distinguish and separate HBCD containing EPS and XPS insulation boards from those which do not contain HBCD including a possible alternative (to be introduced in future). To separate EPS or XPS waste not containing HBCD from HBCD-containing waste would not be possible due to technical, logistical and cost considerations (see [PS foam 2011]).

Currently arising waste quantities of EPS and XPS from construction amount to approximately 17,000 t/a (2010) but will significantly increase in the coming decades (approximately 150,000 tonnes in 2030 and approximately 400.000 tonnes in 2040).

Current waste management options for EPS and XPS waste from construction are incineration (D10 and/or R1), disposal and recycling.

In incineration the high energy content of the PS foams can be used for energy recovery and the HBCD content is irreversibly destroyed.

Specific information on the long term behaviour of HBCD containing PS foams under landfill conditions is not available. If landfilled, the HBCD content of waste is not destroyed. Landfilling is related to the uncertainty that in the long term HBCD may be released to the environment and to the drawback that the high energy content of the HBCD foams cannot be used for energy recovery. Therefore for PS foam waste, incineration with energy recovery is environmentally preferable to landfilling.

Concerning EPS, the current state of the art technology limits adding recyclate to virgin material to around 15% maximum. Due to the possibility and necessity to recycle internal production waste (cut offs at the conversion site) at a level of about 7.5 % the percentage to be mixed in as external recycled content would be 7.5 % maximum. Since the amount of HBCD in flame retarded EPS waste is on average 0.51%, the content of HBCD in a new article containing 7,5% of HBCD-containing recyclate would be on average $0,51 \times 0,075 = 0,038\%$ (380 ppm). Considering a maximum concentration of 0.7%, the maximum content of HBCD in a new article containing 7,5% of HBCD-containing recyclate would be at a maximum of 0,052% (525 ppm) (calculation based on [PS foam 2011]).

The emission of HBCD from the use phase of EPS and XPS insulation boards containing current levels of HBCD is negligible (see [SWEREA 2010]) and does not cause relevant environmental or health risks [ECB

2008]. The use of EPS and XPS insulation boards with a significantly reduced percentage of HBCD, coming from future recycling, would therefore not cause an immediate risk to environment or health.

It could therefore be reasonable to recycle a certain share of HBCD containing EPS/XPS insulation boards into new HBCD free EPS/XPS insulation boards under the following conditions: (1) there is no risk related to the deconstruction and demolition phase of EPS/XPS insulation boards. (2) The HBCD contained in recycled products will finally not be released to the environment but will be destroyed or irreversibly transformed at the end of life of the recycled products (or after further recycling).

According to the opinion of the PS foam industry, releases to the environment from the demolition phase will be lower in the case of demolition for recycling than in the case of demolition for incineration or landfilling due to “more attentive dismantling when recycling PS boards”. Based on this consideration the PS foam industry argues that there will be lower releases to the environment if recycling of PS foams will be possible. In the opinion of the project team, there should be no differentiation in the demolition practices whether intended for recycling or not.

The advantages of recycling and reuse (no costs for separation, resource efficiency, control over the substance by keeping it under control in the product) would have to be considered against the negative aspect of the unwanted carry over of HBCD to recycled products and the risk that the HBCD will not be destroyed or irreversibly transformed as required by the POP Regulation.

One general objective of the POP Regulation is to protect human health and the environment from POPs by e.g. phasing out as soon as possible the use of POPs (see article 1 of the POP Regulation). Article 7.2 requires that the POP content of waste is destroyed or irreversibly transformed without undue delay. According to this principle recycling of the POP content of waste is not allowed. Exemptions from the principle would be possible according to recital 15 of the POP regulation: “...the persistent organic pollutant content in waste is to be destroyed or irreversibly transformed into substances that do not exhibit similar characteristics, unless other operations are environmentally preferable.” If recycling is environmentally preferable it could therefore be listed as permitted disposal and recovery operation for recycling of waste from PS foam insulation boards into new PS foam insulation boards in Annex V part II of the POP regulation.

Taking this into account, generally recycling should not be allowed and HBCD containing waste should be incinerated at its end of life in order to destroy its HBCD content (the HBCD content in PS foams is destroyed in controlled incineration). A derogation could be reasonable, if recycling of the HBCD content is environmentally preferable.

Currently an important share of PS foam waste is landfilled (about 40%) and its HBCD content is consequently not destroyed. Landfill will not be allowed any more once a LPCL will be set in the POP regulation for waste exceeding the LPCL. As a consequence corresponding waste will have to be treated according to Annex V, Part 1 (practically relevant: incineration, i.e. D10 and R1) and Annex V, part 2 if there will be environmentally preferable operations listed.

In conclusion, the increasing quantities of EPS and XPS waste from construction will require appropriate waste management options for EPS and XPS waste in the future.

The most preferred waste management option is incineration with energy recovery because in this operation the POP content is destroyed and the high energy content of the waste can be used.

In landfilling the HBCD content is not destroyed and landfilling of HBCD is related to uncertainties in the long term because information on the long term behavior of PS foam waste and its HBCD content is lacking. HBCD may leach from products and from landfills. In an investigation by Nylund *et al.* (Nylund *et al.*, 2002) HBCD was found in sludge from all 50 investigated STP in Sweden. The mean value for all the STPs was 45 ± 94 µg HBCD/kg dwt. The levels found in sludge from STPs which were not related to point sources are in the range 4-78 µg HBCD/kg dwt regardless of the size of the STP, thus indicating a leakage from products and compounds in the society (see [ECB 2008]). No HBCD was detected in leachate water (dissolved or particulate phase) from landfills in Ireland or UK (half detection limit of 23 µg HBCD/l and 5.85 µg HBCDD/kg dwt, is used). The concentrations of HBCD in the particulate phase of leachate water from landfills in the Netherlands ranged from below the detection limit (15 µg HBCDD/kg dwt, half detection limit used) to 22000 µg HBCDD/kg dwt [ECB 2008]. Schlabach *et al.* (2002) measured HBCD in sedimentation basins for leachate waters from six landfills in southern Norway. The concentrations ranged from below the detection limit to 84 ng HBCD/kg ww (see [ECB 2008]). Sternbeck *et al.* (2001) reported measurements of HBCD in drain water from a construction waste deposit and in sediment from a sedimentation basin to the construction waste deposit. The concentrations were 0.003-0.009 µg HBCDD/l, and 0.05 µg HBCDD/kg dwt (half detection limit used), respectively (see [ECB 2008]). Usually, landfilling is therefore considered a less preferable environmental option compared to incineration.

It could also be reasonable to allow recycling of HBCD containing EPS and XPS waste for the use in EPS and XPS products for construction up to a certain HBCD content. Recycling of HBCD into new EPS or XPS insulation boards is related to specific advantages and drawbacks. (1) HBCD might be under good control if used in PS insulation boards and destroyed at the end of its service life. (2) Recycling of PS insulation boards could enable economic profit and natural resource optimisation. (3) Short transport distances for waste from PS insulation boards for recycling would be possible. (4) There is uncertainty concerning releases of HBCD during the deconstruction phase of PS insulation boards.⁷³ (5) If recycled, the HBCD content would be diluted in new products by a certain factor (up to 10%). As a consequence the incineration capacities required for an appropriate end-of-life management would correspondingly increase by a factor of 10 or more. In conclusion, it is therefore related to uncertainty, whether recycling

⁷³ When released to the environment during deconstruction and demolition, HBCD is released not as a pure substance but integrated in PS foam particles (e.g. as dust or bigger particles). According to information provided by the PS foam industry, a PS foam particle containing HBCD is not much worse than just a PS foam particle because: (1) HBCD is soluble in PS and is molecularly dispersed in the matrix of PS. (2) PS particles remain extremely large compared to diffusion of HBCD hence it is not much different from a board in term of emission potential from its surface. PS is a glassy polymer in which the molecular mobility is extremely slow. (3) In the long term the degradation of HBCD in the ground is relatively short and most often this is less than one year (see [Arnot *et al.* 2009]) while the PS in the ground would degrade much more slowly i.e. 10 to 100 times more slowly. (4) Experimentally HBCD could not be detected (below detection limit) in the ground below XPS boards loaded with HBCD [Haagensen 2007].

of HBCD into new EPS or XPS insulation boards could be considered an environmentally preferable option.

In order to assure that the majority of the HBCD content of PS foams from construction will be destroyed, it should be attempted to separate HBCD contaminated PS insulation boards as far as reasonably possible. This will enable to destroy the HBCD content of the contaminated fraction by incineration. A high degree of separation could be achieved e.g. by implementing deconstruction/renovation management plans for buildings which take into account whether it is known/probable or not if PS foams used in a specific building contain HBCD or not. The average probability that an EPS foam contains HBCD is currently about 77% within the EU. The probability further depends on regional differences (it is e.g. usually high in PS foams for construction in Austria, Czech Republic, Germany, Hungary, the Netherlands, Slovakia and Slovenia, Italy, Portugal and the UK but lower in France, Denmark and Finland and particularly low in Sweden). Depending on the expected time frame for the phase in of alternatives to HBCD the probability of HBCD contamination can therefore be easily assessed. In buildings of a certain dimension it could furthermore be required to perform a chemical analyses of the PS foam used for insulation in order to obtain information on the HBCD contamination. On the basis of such an approach for each individual building, only PS foams with a low probability of HBCD contamination (based on probability assessment) or without contamination (based on chemical analyses where appropriate or on information from marking/labelling) should be used for recycling. PS foams with a high probability of HBCD contamination or with proven contamination should be destroyed or irreversibly transformed preferably by incineration with energy recovery. The related costs seem to be economically acceptable (particularly if it will be possible to mark/label HBCD free products).

The approach should also enable to identify sufficient PS foam waste which is probably not contaminated with HBCD and which can be supplied for recycling for the manufacturing of new PS foam insulation boards (i.e. 7.5% of the consumed virgin material) or for other recycling purposes. The non-contaminated share of EPS insulation boards is currently about 23% but will increase in the future. The possibility to recycle PS foams will thus be maintained. Transport distances for the recycled fractions would increase.

Such an approach could be significantly facilitated if it would be possible to mark/label PS foam insulation products that do not contain HBCD as soon as possible. The PS foam industry is prepared to discuss any potential marking/labeling of the products to differentiate between HBCD-containing and non-HBCD-containing PS foams in order to facilitate the separation of HBCD containing waste from HBCD free waste.

According to the results of the present study it is proposed to set an LPCL for HBCD at 100 ppm and to attempt the separation and destruction of PS insulation boards as far as reasonably possible from waste not containing HBCD. This would be related to the following impacts:

Proposal →	LPCL for HBCD at 100 ppm and attempt to separate and destroy HBCD containing PS insulation boards as far as reasonable possible from non contaminated products; no
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	contaminated products would be available for recycling.	
Substance →	HBCD	
Impact	Assessment	Justification
Economic impact	Moderate	<ul style="list-style-type: none"> • non contaminated products remain available for recycling; • transport distances for the recycled fraction will increase; • certain effort for separation required; • the share of non-contaminated products will increase in the future and will facilitate future recycling
Environmental impact	Positive	<ul style="list-style-type: none"> • HBCD would be destroyed and thus withdrawn from the eco-cycle. Hence it can be guaranteed that they will not become subject to LRT and bioaccumulate and cause adverse environmental and health effects.
Impact on waste management	Low	<ul style="list-style-type: none"> • The increasing quantities of EPS and XPS waste from construction will require appropriate waste management options anyway. • The identification and separation of the relevant PS insulation boards appears to be feasible. Analytical measurement only required in specific cases.
Compatibility with REACH	✓	<ul style="list-style-type: none"> • LPCL is below the 0.1% concentration limit
Compatibility with WEEE	n.r.	<ul style="list-style-type: none"> • n.r.
Compatibility with ELV	n.r.	<ul style="list-style-type: none"> • n.r.
Compatibility with ROHS	n.r.	<ul style="list-style-type: none"> • n.r.

9.5.4 Recycling of carpets

From the existing synthetic fibre carpets only those can be recycled which consists of Polyamid 6, whereas Polyamid 6,6 cannot be recycled. In Europe there have been several attempts for a carpet recycling systems, however, most of them have not been successful. On national level recycling organization have been identified (e.g. Carpet recycling UK [Carpet UK 2010]) who are continuously working on new recycling technologies. In 2009 16,000 tonnes of carpets have been recycled by energy recovery out of 26,000 tonnes of total recycled carpets. The carpet recycling rate in the UK is ~ 2% [Carpet Gardner 2010] and is expected to be negligible all over Europe.

There are also some carpet companies in Europe, (including members of the before mentioned national recycling organization), redeem carpets for recycling. There are no data available regarding the share of recycled carpets or the involved company. It is assumed that only a small fraction is gathered and that the majority is landfilled. [Carpet Gardner 2010]

There are some recyclers in other countries (e.g. U.S.) but their system cannot be easily adopted for Europe as the market structure as well as the carpet compositions are different. In the U.S. carpet leasing is a typical system and the Polyamid content of the carpet is higher. Both are beneficial circumstance for a successful recycling, which are not given in Europe.

According to the results of the present study it is proposed to set an LPCL for PFOS at 10 ppm. This would be related to the following impacts:

Proposal →	LPCL for PFOS at 10 ppm	
Substance →	PFOS	
Impact	Assessment	Justification
Economic impact	Low	<ul style="list-style-type: none"> • There is no structural carpet recycling system in Europe, recycling is done punctually and the majority of the carpets end up as bulky waste; • The large quantity would result in high incineration costs which cannot be landfilled anymore; • the share of non-contaminated products will increase in the future and will facilitate future recycling
Environmental impact	Positive	<ul style="list-style-type: none"> • Carpets have a share of 89% of all PFOS in waste. • PFOS would be destroyed and thus withdrawn from the eco-cycle. Hence it can be guaranteed that they will not become subject to LRT and bioaccumulate and cause adverse environmental and health effects • Energy recovery
Impact on waste management	Low	<ul style="list-style-type: none"> • In some member states incineration is already typical to some extent.
Compatibility with REACH	✓	<ul style="list-style-type: none"> • LPCL is below the 0.1% concentration limit
Compatibility with ROHS	n.r.	<ul style="list-style-type: none"> • n.r.

9.6 Possible implications of concentration limits for PBDEs in new articles produced from recycled materials above which the articles cannot be placed on the market

According to Annex I of the POP regulation PBDE containing materials can be used for articles and preparations containing concentrations below 0.1% of the relevant congener groups when produced partly or fully from recycled materials or materials from waste prepared for re-use.

The POP content of POP waste has to be destroyed or irreversibly transformed in accordance with article 7.2. of the POP regulation.

If a POP containing material is intended for recycling, there are generally two options to handle the POP content in the corresponding process:

- (1) Separation of the POP containing articles or parts from non-contaminated components
- (2) Separation of the POP content from the contaminated material

Both options enable the reduction of the POP content in a specific fraction of the POP containing material. Corresponding processes can be applied in recycling activities in order to reduce the POP content of specific material fractions.

(1) Separation of the POP containing articles or parts from non-contaminated components

The general technical guidelines for the environmentally sound management of POP waste (see [BC TG 2007]) state that, where only part of a product or waste, such as waste equipment, contains or is contaminated with POPs, it should be separated and then disposed of as specified in subsections 1–4 of

these guidelines, as appropriate.

As explained in the previous chapter it is expected that the PBDE content in relevant waste streams can be reduced by possibility (1)⁷⁴ to levels below the LPCL proposed for the relevant PBDE congener groups (i.e. to below 10 ppm per congener group). As a consequence the remaining waste can be treated according to the European waste legislation which enables among other material recycling.

Considering option (1), the concentration limits for PBDEs in new articles produced from recycled materials in Annex I could be lowered correspondingly. The real contamination levels in the waste stream should be evaluated prior to lowering the concentration limits for PBDEs in new articles produced from recycled materials in Annex I.

(2) Separation of the POP content from the contaminated material

A potential solution to increase the material recycling quatum could be the separation of PBDE/BFRs from plastics and subsequent recycling of the plastic fraction and destruction or irreversible transformation of the PBDE/BFR fraction. Possibly also the bromine content could be recovered by the bromine industry.

Technologies to separate BFRs (including PBDEs) from materials have only been established for PBDE/BFR-containing plastics and printed circuit boards. The main driving force has been the commercial market value of these two material categories. However, even though the technology for PBDE/BFR separation seems ready for industrial application, none of the separation technologies is currently operating at industrial scale [UNEP 2010].

The CreaSolv® process is a physico-chemical treatment, developed by CreaCycle GmbH in cooperation with the Fraunhofer Institute IVV. It is a batch process which uses a high boiling point solvent to dissolve selectively the feed plastic, followed by a series of precipitation steps using a combination of CreaSolv® solvents. The process allows the removal of undissolved (e.g. non-target polymers and other interfering materials) and dissolved contamination (e.g. PBDEs or other BFRs). The BFRs and their toxic reaction products are extracted into the solvents which can be afterwards recovered. The volume of solvents used is very small in relation to the treated plastic (<1%), as the solvents circulate in closed-cycles and are routinely recycled. The by-product containing high levels of BFR can be for instance recovered by the bromine industry, chemically treated or incinerated. Considering the high market price for bromine (~2,500\$/t), bromine recovery represents an attractive option ([WRAP 2006], [CreaCycle 2010], [UNEP 2010]).

The CreaSolv® process produces recycled polymers of high purity. The process allows a reduction of the bromine content in the polymer to less than 0.1%. It is therefore possible to produce RoHS compliant polymers. The trial studies conducted by the United Kingdom's Waste & Resources Action Programme (WRAP) concluded that CreaSolv® can be successfully applied in order to remove BFRs from WEEE polymers. This has also been confirmed in a project by the Austrian Center of Excellence Electronics & Environment KERP and the Fraunhofer IVV. It was proven that heavily polluted shredder light fractions

⁷⁴ i.e. by separating PUR foams from cars built in 2000 and before and ABS plastics from cathode ray computer and TV monitors produced in 2003 and before.

can be successfully recycled. The resulting high quality recycled materials have similar mechanical properties to the virgin material and can therefore again be used for EEE. Moreover, according to the developer, the process could also be applied for the recycling of polymers from the automotive sector and the recycling of expanded plastics such as EPS from the construction and packaging industry ([WRAP 2006], [CreaCycle 2010], [UNEP 2010]).

Besides, it has been indicated that the CreaSolv® process can be a financially viable alternative to land filling and incineration. For instance, the UK WRAP study has shown that the CreaSolv® process has the potential to be commercially viable in the UK at a throughput of 10,000 t/y. Due to the fact that most of PBDE/BFR-containing plastics can be converted to RoHS compatible and marketable polymer products the process could be able to compete with BFR removal processes using spectroscopic sorting techniques, since it achieves higher yields. The study also concluded that the process could compete with export of mixed WEEE polymer outside the EU in case the compounded recyclate can be sold at about 80% of the virgin compound price ([WRAP 2006], [UNEP 2010]).

The WEEE Directive specifies minimum recovery and recycling limits of 50–70% for appliances depending on their category (Art. 7, Para. 2). Energy recovery through incineration has been, however, specifically excluded (Art. 3e). Therefore, one of the major questions will be how these ambitious goals for WEEE plastics can be achieved in the future. Achieving high recycling quotas could especially be difficult in case of appliances with high plastic contents (CreaCycle 2010).

Assessment of implications of the 0.1% limit related to recycling of PBDEs

Considering these potentials, processes such as the CreaSolv® process may be or become a competitive management option for PBDE/BFR contaminated wastes. The process may enable to achieve high levels of material recovery and can therefore facilitate to achieve recycling targets as laid down in the ELV or WEEE directive. Nevertheless, the process will only be competitive to current and future waste management options if operated at a moderate scale, which means that it needs to be scaled up from pilot- to full/industrial scale.

As mentioned above, the process allows a reduction of the bromine content in the polymer to less than 0.1%. The weight of bromine contributes around 60 to 75% to the weight of the Tetra-, Penta-, Hexa- and HeptaBDE congeners. This allows the conclusion that the process also allows a reduction of the content of each of the relevant BDE congener groups below 0.1%. In conclusion, physico-chemical treatment such as the CreaSolv® process are generally appropriate to reduce the POP content for each listed PBDE congener groups below the 0.1% limit for PBDEs in new articles produced from recycled materials. The separated POP content could be destroyed or irreversibly transformed according to the requirements of the POP regulation.

Considering option (2) it could therefore be reasonable to maintain the 0.1% limit for PBDEs in new articles produced from recycled materials. However, it is not expected that in the European Situation corresponding physico-chemical treatment processes will play a role in the case of the relevant PBDEs. The European situation is characterised by limited quantities of PBDEs in waste streams with a

decreasing trend. It is expected that by 2020 no relevant quantities of the listed PBDEs will occur in waste streams. Against this background it is not expected that corresponding physico-chemical treatment processes will become competitive at industrial scale.

In the opinion of the project team it is therefore not necessary to maintain the 0.1% concentration limit for PBDEs in new articles produced from recycled materials in Annex I of the POP regulation. It could be lowered to the level of the LPCL. Real contamination levels in the relevant waste streams should be evaluated prior to lowering the concentration limits for PBDEs in new articles produced from recycled materials in Annex I.

Considering the recycling targets of the ELV and the WEEE Directive and the continuous use of BFRs including not (yet) listed PBDEs, the separation of these substances from waste materials may become the only way to recycle BFR containing materials.

9.7 Disposal and recovery operations for new and candidate POPs

According to Article 7.2 and Annex V part 1 of the POP regulation, waste containing POPs, have to be treated by specific disposal and recovery operations in such a way as to ensure that the POP content is destroyed or irreversibly transformed. According to Annex V part 1, the following waste management options are possible:

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

A comprehensive compilation of potential destruction methods for POPs is given in the “Updated general technical guidelines for the environmentally sound management of wastes consisting of, containing or contaminated with POPs” prepared under the Basel Convention⁷⁵.

In the scope of the present project it is relevant to consider whether the disposal and recovery operations as specified in Annex V part 1 of the POP regulation can also be considered appropriate potential destruction methods for new and candidate POPs.

From a general point of view it can be expected that the new and candidate POPs as halogenated organic substances with generally similar physico-chemical characteristics to the already listed POPs will behave similar and it is therefore expected that the same disposal and recovery operations could be applicable also to the new and candidate POPs. It should be taken into account, that in addition to the

⁷⁵ <http://www.basel.int/meetings/sbc/workdoc/techdocs.html>

destruction of the relevant POPs, the unintentional formation of hazardous substances such as the unintentional formation of halogenated dioxins and furans is an aspect to be taken into account. In the following information is compiled related to the individual disposal and recovery operations (D9, D10, R1, R4).

Table 9-54: Evaluation of the appropriateness of disposal and recovery operation according to Annex V, part 1 of the POP regulation for the new and candidate POPs

Substance (group)	Disposal/recovery operation (specification)	Appropriateness
All new and candidate POPs	D9 (not specified)	Depending on the individual operation (appropriate if it is ensured that the POP content is destroyed or irreversibly transformed)
All new and candidate POPs	D10 or R1 (BAT/BEP incineration)	✓
PBDEs, HBCD	R1 (Cement Kiln co-incineration)	Depending on the individual operation Appropriate if it is ensured that the POP content is destroyed or irreversibly transformed; information on destruction efficiencies would be useful Some indications for the unintentional formation of PBDD/Fs (no conclusive information available)
All relevant POPs	R4 as described in BC TG 2007	✓ Appropriate if it is ensured that the POP content is destroyed or irreversibly transformed; information on destruction efficiencies would be useful Unintentional formation of PBDD/Fs should be considered for brominated POPs
PBDEs	R4 (material and energy recovery in metal smelters, electric arc furnaces, feedstock recycling in primary steel industry, recovery in secondary aluminum industry, recovery in antimony smelters)	Appropriate if it is ensured that the POP content is destroyed or irreversibly transformed; more information on destruction efficiencies would be useful Some indications for the unintentional formation of PBDD/Fs (no conclusive information available)

D9 – Physico-chemical treatment

According to the BREF on waste treatment industries more than 14,000 waste treatment installations exist in the EU. The majority thereof (approximately 9,900) are physico-chemical waste treatment installations [BREF WTI 2006]. Physico chemical treatment is applied to waste waters, waste solids and waste sludges with a wide range of techniques applied.

An example of a physico-chemical treatment facility of waste waters typically contains the following unit processes: cyanide destruction, chromium reduction, two-stage metal precipitation, pH adjustment (e.g. neutralisation), solid filtration, biological treatment, carbon adsorption, sludge dewatering, coagulation/flocculation and some others. Typical physico chemical treatments of waste solids and

waste sludges are extraction and separation, thermal treatment, mechanical separation, conditioning, immobilisation (this treatment covers solidification and stabilisation), dewatering, drying, thermal desorption, vapour extraction from excavated soil, solvent extraction from solid waste (e.g. excavated soil), excavation and removal of excavated soil and soil washing.

Physico chemical treatment is configured on a case-by-case basis depending on requirements and/or application which is geared to the waste to be treated. For this reason, there is no 'standard' physico-chemical treatment plant. Best available technology (BAT) for physico-chemical treatment is described in the BREFs on waste treatment industries and on waste incineration ([BREF WTI 2006] and [BREF WI 2006]). Under the Basel Convention a technical guideline on physico-chemical treatment has been elaborated [BC TG D9 2000].

Annex V Part 1 of the POP regulation does not specify which physico-chemical treatment is appropriate for disposal and recovery of POPs but qualifies that the operation has to "ensure that the POP content is destroyed or irreversibly transformed". This qualification applies also to any newly listed substance. As a consequence only physico-chemical treatment operations are permitted that fulfil the qualification of destruction or irreversible transformation of all POPs (existing, new and candidate POPs).

It is not possible (and not necessary) to provide specific information due to the wide range of existing physico-chemical treatment operations.

D10 (incineration on land) and R1 (use as a fuel or other means to generate energy, excluding waste containing PCBs)

Typical D10 and R1 operations are municipal solid waste incineration (MSWI) or hazardous waste incineration (HWI) or co-combustion in cement kilns.

Incineration conditions in European incineration plants:

To achieve good burn out of the combustion gases, a minimum gas phase combustion temperature of 850°C (1100°C for some hazardous wastes) and a minimum residence time of the flue-gases, above this temperature, of two seconds after the last incineration air supply have been established in legislation:

"Incineration plants shall be designed, equipped, built and operated in such a way that the gas resulting from the process is raised, after the last injection of combustion air, in a controlled and homogeneous fashion and even under the most unfavourable conditions, to a temperature of 850 °C, as measured near the inner wall or at another representative point of the combustion chamber as authorised by the competent authority, for two seconds. If hazardous wastes with a content of more than 1 % of halogenated organic substances, expressed as chlorine, are incinerated, the temperature has to be raised to 1 100 °C for at least two seconds." (Directive 2000/76/EC, Article 6.1)

Rotary kilns are, in particular, very widely applied for the incineration of hazardous wastes. Operating temperatures of rotary kilns used for wastes range from around 500 °C (as a gasifier) to 1450 °C (as a high temperature ash melting kiln). Higher temperatures are sometimes encountered, but usually in

non-waste applications. When used for conventional oxidative combustion, the temperature is generally above 850 °C. Temperatures in the range 900 - 1200 °C are typical when incinerating hazardous wastes. [BREF WI 2006]

The temperatures in the post combustion chamber (PCC) typically vary between 900 - 1200 °C depending on the installation and the waste feed. Most installations have the ability to inject secondary air into the post combustion chamber. Due to the high temperatures and the secondary air introduction, the combustion of the exhaust gases is completed and organic compounds (e.g. PAHs, PCBs and dioxins) including low molecular weight hydrocarbons, are destroyed. [BREF WI 2006]

As mentioned above, if hazardous wastes with a content of more than 1 % of halogenated organic substances, expressed as chlorine, are incinerated, the temperature has to be raised to 1 100 °C for at least two seconds. (Directive 2000/76/EC, Article 6.1)

Information on destruction of new and candidate POPs in incineration processes (D10, R1):

PBDEs

“... PBDE/BFR-containing waste can be destroyed in BAT waste incinerators operated according to Best Environmental Practice (BEP) ...” ([Watson et al. 2010], page 122). Within the EU MSWI and HWI is operated according to BAT/BEP and it can therefore be concluded, that PBDEs can be destroyed in MSWI and HWI.

Watson et al. 2010 question the appropriateness of the co-combustion of PBDEs waste in cement kilns mainly due to the risk of the possible formation of brominated dioxins and furans. No study has yet been published with monitoring of releases of PBDE and PBDD/PBDF. It is therefore not possible to be confident about the effectiveness of cement kilns for wastes containing PBDE. According to Watson et al. 2010, the destruction efficiency of PBDE in the waste will depend to a significant extent on the feeding point in the cement kiln. Watson et al. 2010 therefore recommend carrying out a properly configured test-burn, together with the establishment of the destruction efficiency which incorporates analysis of all emissions from the process including from products and the bypass stack, which should always be carried out before any POPs waste is considered for routine disposal. In addition Watson et al. recommend performing a general evaluation of the options and limitations for the destruction of wastes containing PBDE in cement kilns. (see [Watson et al. 2010] pages 120/121)

In conclusion, PBDEs are largely destroyed at conditions in accordance with Directive 2000/76/EC. However, the destruction efficiency and the possible formation of brominated dioxins and furans during co-combustion in cement kilns should be further assessed.

PFOS

It has been proven that the PFOS content of contaminated sludge which is incinerated at conditions in accordance with Directive 2000/76/EC is largely destroyed (below the detection limit of 15 ng/m³ in the exhaust air; below detection limits of all other output fractions) [NRW_2007a].

PeCB

Incinerators complying with the demand of the low PCDD/F emissions will undoubtedly minimize the emissions of PeCB. Efficiencies similar to that of dioxins (> 99.9%) can be obtained, e.g. in case of catalytic destructions above 300° C or the use of carbonaceous adsorbents as cleanup of the flue gas (see [UNEP POPS POPRC.4 15 Add.2]).

However, different emissions of PCDD/Fs and PeCB formed in *de novo* synthesis in the flue gas might still be possible and will depend on the type of the abatement technology, applied for the specific emission reduction of PCDD/Fs. A variation between the correlation of emissions of PeCB and PCDD/Fs from various incinerators has been observed (Lavric et al., 2005) and there is conflicting information about effectiveness for various abatement techniques (Liljelind et al., 2001). In addition, due to the relatively high volatility of PeCB in comparison to PCDD/Fs the adsorption to particles will be distinctly less, and, therefore, compounds like PeCB will be more present in gas phase in comparison to PCDD/Fs (Chen et al., 2007). Hence, abatement techniques focused at the elimination of dust might have a somewhat lower efficiency for the removal of PeCB formed *de novo* in the flue gas. [UNEP POPS POPRC.4 15 Add.2]

In conclusion, PeCB is largely destroyed during incineration at conditions in accordance with Directive 2000/76/EC.

SCCPs

A large proportion of SCCP is destroyed in the process of waste management, especially since these substances decompose at temperatures above 200 °C ([OSPAR 2006], [BUWAL 2003]). Accordingly ECHA states that incineration is likely to completely destroy the SCCP [ECHA 2008].

SCCPs are expected to be degraded by incineration at low temperatures because of their low thermal instability, however, data supporting this statement is not provided. In Europe, a large portion of products containing SCCPs (painted material, plastics, leather) will be incinerated at the end of their product life. For this reason, it is important to further study the efficiency of SCCP destruction in waste incinerators. Emission data for chlorinated hydrocarbons can be measured at many waste incinerators in the north western part of Europe. This data could provide more insight as to SCCP emission levels resulting from the incineration of products after use. [UNECE 2006]

In conclusion, SCCPs are largely destroyed during incineration at conditions in accordance with Directive 2000/76/EC. Specific information on destruction efficiencies could be useful.

HBCD

In well-functioning incinerators the risk of formation of brominated and mixed halogenated dioxins and furans due to HBCD input is negligible. In case of uncontrolled fires (accidental fire) and at co-combustion at lower temperatures or not well functioning incinerators, however, there is a risk of formation of PBDDs and PBDFs. [Öko Institut 2008]

A study has shown that HBCD containing waste can be effectively destroyed in co-combustion in a MSWI. The brominated dioxins and furans were generated at very low levels of pg/m³ in the case of PBDD and typically below 1 ng/m³ for PBDF. The generation of mono- and di-brominated PBDD and PBDF could also be detected at very low levels (see [Vehlow and Mark 1995] and [Vehlow and Mark 1996]). It is assumed that HBCD is largely destroyed during incineration at conditions in accordance with Directive 2000/76/EC.

However, based on the evidence provided for PBDEs (see above), the destruction efficiency for HBCD and the possible formation of brominated dioxins and furans during co-combustion in cement kilns should be further assessed.

PCP

Controlled combustion of PCP treated achieves destruction efficiencies greater than 99.9 % (see [USEPA 1984] and [LFU 1996]).

PCP is destroyed by incineration [WFD 2004 PCP screening]. The disposal of PCP-containing waste should preferably involve controlled high-temperature combustion. PCP treated wood combusted at 910 °C – 1025 °C resulted in no detectable chemicals in the off gas. However, the incineration temperature needs to be high enough and the resident time long enough to fully degrade the PCP, as it combusts incompletely at temperatures under 800 °C [UNEP 2010]. The controlled combustion conditions under which most wood and wood based materials are incinerated today in Sweden quickly destroy dioxins and other chlorinated aromatic substances in the treated timber [SEPA 2009].

It can therefore be considered, that PCP is largely destroyed during incineration at conditions in accordance with Directive 2000/76/EC.

HCBD

Controlled combustion of HCBD containing waste achieves destruction efficiencies greater than 99.9 % (see [Ineris 2005], [EuroChlor 2004], [ATSDR 1994]). It can therefore be considered, that HCBD is largely destroyed during incineration at conditions in accordance with Directive 2000/76/EC.

PCN

Controlled combustion of PCN containing waste (MSW and contaminated rubber belts) achieves destruction efficiencies between 99.32 and 99.96 % [Noma et al. 2004]. It can therefore be considered, that PCNs are largely destroyed during incineration at conditions in accordance with Directive 2000/76/EC.

R4 – Recycling reclamation of metals and metal compounds.

According to Annex V part 1 of the POP regulation R4 includes *“Recycling/reclamation of metals and metal compounds, under the following conditions: The operations are restricted to residues from iron- and steel-making processes such as dusts or sludges from gas treatment or mill scale or zinc-containing filter dusts from steelworks, dusts from gas cleaning systems of copper smelters and similar wastes and*

lead-containing leaching residues of the non-ferrous metal production. Waste containing PCBs is excluded. The operations are restricted to processes for the recovery of iron and iron alloys (blast furnace, shaft furnace and hearth furnace) and non-ferrous metals (Waelz rotary kiln process, bath melting processes using vertical or horizontal furnaces), provided the facilities meet as minimum requirements the emission limit values for PCDDs and PCDFs laid down in Directive 2000/76/EC of the European Parliament and of the Council of 4 December 2000 on the incineration of waste (1), whether or not the processes are subject to that Directive, and without prejudice to the other provisions of Directive 2000/76/EC where it applies and to the provisions of Directive 96/61/EC.”

Processes in thermal and metallurgical production of metals for the destruction and irreversible transformation of POPs processes are described in [BC TG 2007] as follows:

Processes which are relevant for the destruction of the POP content in iron-containing wastes use certain types of blast furnace, shaft furnace or hearth furnace. All these processes operate under reducing atmospheres at high temperatures (1,200°C–1,450°C). The high temperature and the reducing atmosphere destroy PCDDs and PCDFs contained in the wastes and avoid de novo synthesis. The blast furnace and the shaft furnace processes use coke and small amounts of other reducing agents to reduce the iron-containing input to cast iron. There are no direct emissions of process gas as it is used as a secondary fuel. In the hearth furnace process, the iron-containing material is charged to a multihearth furnace together with coal. The iron oxide is directly reduced to solid direct reduced iron (DRI). In a second step the reduced iron is melted in an electric arc furnace to produce cast iron; [BC TG 2007]

Processes which are relevant for the destruction of the POP content in wastes containing NFMs are the Waelz rotary kiln process and bath melting processes using vertical or horizontal furnaces. These processes are reductive, reach temperatures of 1,200°C and use rapid quenching thus PCDDs and PCDFs are destroyed and de novo synthesis is avoided. In the Waelz process zinc-containing steel mill dusts, sludges, filter cakes, etc. are pelletized and smelted together with a reductant. At temperatures of 1,200°C, the zinc volatilizes and is oxidized to “Waelz Oxide”, which is collected in a filter unit. In the vertical bath furnace process, copper-containing residues are smelted at temperatures of at least 1,200°C. The filter dust is used for the production of zinc and zinc compounds. In the horizontal bath furnace process, lead-containing residues and ore concentrates are charged continuously into a smelting bath which has an oxidizing and a reducing zone with temperatures between 1,000°C and 1,200°C. The process gas (sulphur dioxide concentration above 10 per cent) is used for sulphuric acid production after heat recovery and de-dusting. The dust from the process is recycled after cadmium leaching. [BC TG 2007]

For the relevant processes there are no data on the POPs destruction efficiency or the destruction removal efficiency are not available [BC TG 2007].

Data on the destruction efficiency are neither available for the old nor for the new or candidate POPs. However R4 processes are considered appropriate for the recovery and disposal of POPs currently listed under the Basel Convention.

This consideration is confirmed by information recently provided by industry on the destruction efficiency for PBDEs in a relevant thermal metallurgical process: According to information provided by industry, results from spot tests from 2010 in a thermal metallurgical process (bath melting) using printed circuit board as input material indicate a destruction efficiency for the PBDEs above 99.99 %. In the spot measurements also the content of brominated dioxins (PBDDs) and furans (PBDFs) was analysed in three measurements. The concentrations in the off gas were below the detection limits for PBDD congeners and below 0.1 ng/m³ for PBDF congeners.

In conclusion it can be considered that – analogue to old POPs – new and candidate POPs are largely destroyed during the described processes for Recycling/reclamation of metals and metal compounds. Specific information on destruction efficiencies is scarce (for old, for new and for candidate POPs) but would be helpful for a conclusive evaluation. When brominated POPs⁷⁶ are treated in R4 processes, the possible formation of brominated dioxins and furans should be taken into account. There are several indications on possible releases of unintentionally generated brominated dioxins and furans in R4 processes (see particularly the following paragraphs).

Watson et al. 2010 (pages 124/125) question the appropriateness of treating PBDE containing materials in smelters and other metal industries e.g. for PBDE/BFR-containing materials treated in copper smelters and other metal industries such as electric arc furnaces or secondary alumina smelters.

Copper smelters are used for recovering metals from Printed Circuit Boards, cables and other plastic materials which are firmly combined with the metals to be recovered. Recent studies have reported on releases of PBDE and polybrominated dioxins and furans (PBDD/DF) from electric arc furnaces, sinter plants and aluminium smelters revealing that also PBDE and BFR-containing materials are processed in these facilities. The latest review by Du also reports releases of PXDD/DF. Whilst the type of feeding materials was not specifically measured or documented in the underlying literature these emissions indicate that PBDE-containing waste have been processed in these facilities. (see [Watson et al. 2010], pages 124/125)

Since thermal processes can lead to debromination of DecaBDE to lower-brominated PBDE the emission pattern of PBDE in these studies only allow limited conclusions on the actual input of c-PentaBDE and c-OctaBDE into these processes. Nor, without specific details of the concentration levels of brominated compounds in the inputs can the destruction efficiency or appropriateness of treatment for PBDE-containing waste be assessed. ([Watson et al. 2010], page125)

Therefore only limited conclusions can be made of the effectiveness and environmental impacts of these processes for recovering energy and materials from articles containing c-PentaBDE and c-OctaBDE. ([Watson et al. 2010], page 125)

Watson et al. 2010 (pages 125 to 127) are furthermore questioning the use of PBDE containing residues for material recovery and energy recovery in smelters.

⁷⁶ Relevant in the context of the present study: Tetra-, Penta-, Hexa- and Hepta BDE; HBCD

Smelters treat a wide range of mixed waste streams such as shredder residues (which can contain high concentrations of PBDEs), PVC, and catalytic metals such as copper. Other flame retarded materials, notably printed circuit/wiring boards, are often processed in secondary copper smelters for recovery of the copper and other precious metals. The polymers containing the PBDE/BFRs are firmly combined with precious and other metals to be recovered. The plastics/resin content serves a dual function as a reducing agent and as a source of energy for the smelting process. There have been full-scale tests to additionally utilise PBDE/BFR containing plastic from WEEE as substitute for coke or oil as reduction agent for the metals and as fuel for smelting. (see [Watson et al. 2010], page 125)

State-of-the-art integrated smelters require investments of well above US\$ 1 billion. Currently, only a few plants within the EU feature the technological performance necessary for the described operations. These include e.g. the plants run by UMICORE (Belgium), Norddeutsche Affinerie AG (Germany) and Boliden (Sweden/Finland). Since the scale of these feedstock recycling operations as practiced for printed circuit boards involves tens of thousands of tonnes/year and is currently recommended for PBDE/BFR-containing plastic from electronics by industry. The releases of PBDE and PBDD/DF could be substantial depending on destruction efficiency of these facilities. (see [Watson et al. 2010], page 125)

Watson et al. 2010 expect that high levels of brominated-chlorinated dioxins and furans will be generated and released from smelter furnaces treating PBDE and other BFR plastics. Accordingly, in the draft BREF on BAT for non ferrous metal industries it is stated “If major amounts of electronic scrap with brominated flame-retardants are used as feedstocks, this may result in the formation of brominated and mixed halogenated PXDD/F” ([BREF NFM 2009]; working draft in progress).

The use of an afterburner could possibly contribute to largely destroy unintentionally produced POPs. It is not clear whether an afterburner is fitted to the European smelters (see [Watson et al 2010], page 127) or whether the (potentially) generated mixed dioxins and furans are emitted or are removed in flue gas treatment.

According to information provided by two European copper smelters using WEEE fractions as input material for a bath melting process, the process is designed in a way that also gases occurring during charging of the material are subject to incineration conditions (temperature and residual times) that are appropriate for the destruction of POPs and for to avoid the generation of PBDD/F. Spot test from one of the corresponding plants indicate a destruction efficiency for PBDEs above 99.99 %. In the spot measurements also the content of brominated dioxins (PBDDs) and furans (PBDFs) was analysed in three measurements. The concentrations in the off gas were below the detection limits for PBDD congeners and below 0.1 ng/m³ for PBDF congeners. This information is related to this specific process.

This is generally consistent with information from [UBA DE 2000] according to which PBDD/F under the process conditions in a Germany copper smelter PBDD/Fs would be destroyed and their generation would be hindered due to the reducing atmosphere (high concentration of sulphuric acid) [UBA DE 2000].

Apart from this specific data, there is no other study available demonstrating the destruction efficiency of secondary copper smelters for PBDEs and the level of generation and release of brominated and mixed halogenated dioxins and furans. There is no conclusive information available whether secondary copper smelters are generally appropriate for the use of PBDE containing waste.

Watson et al. 2010 bring several arguments forward that put the appropriateness of the following R4 operations for the treatment of PBDEs in question:

- Electric arc furnace processes (see [Watson et al. 2010], p. 127)
- Feedstock recycling of PBDE containing plastic in primary steel industry (see [Watson et al. 2010], p. 127/128)
- Secondary aluminum plant processes (see [Watson et al. 2010], p. 128/129)
- Antimony smelter processes (see Watson et al. 2010], p. 129)

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10 Conclusions and recommendations

10.1 Low and Maximum POP Concentration Limits

Based on the evaluation of major substance and waste flows, given the expected trend for remaining stocks of PBDEs and PFOS in used products, and taking into consideration the impacts of potential limit values on the substance flows and potential conflicts with limitation criteria, the project team suggests a preferred option for LPCLs for new POPs and also makes a preferred proposal for candidate POPs ("LPCL 1" in Table 10-1). This option would correspond to a strict application of the precautionary principle in the light of the objective of a fast elimination of the new and candidate POPs. This option aims to eliminate POPs directly upon becoming waste and would prevent that the POPs would be diluted due to recycling in new products. It will be much more difficult to identify and eliminate the concerned substances at later stages when recycled products will become waste. This option in addition takes into consideration the high calorific value of the majority of concerned waste streams and their suitability for thermal treatment – including energy recovery – and adds an additional argument to prefer R1 operations instead of landfilling as predominant treatment option.

Furthermore Table 10-1 lists alternative proposals for a transitional period and/or related to specific restrictions ("LPCL 2" in Table 10-1). The concentration levels proposed as LPCL 2 could be set for a restricted time frame in order to facilitate the enforcement of the LPCLs once the LPCLs will be put into force. Moreover, stricter limits for the agricultural use of sewage sludge containing relevant POPs could be laid down. This approach takes into account that to date the data background is sometimes limited and the contamination levels in several waste flows are based on modeling and justified assumptions. The corresponding concentration levels could probably be lowered and should be reviewed within an appropriate time frame of e.g. five years. The review should be done in the light of new relevant information. In order to improve the data background, analytical results on concentration levels in relevant waste streams should be generated in due time.

Table 10-1: Overview of preferred proposals (LPCL 1) and alternative transitional or restricted proposals (LPCL 2) for Low POP Concentration Limits (LPCLs) referred to in Article 7(4)(a) and to be specified in Annex IV of the POP Regulation.

Substance	LPCL 1 (ppm)	LPCL 2 (ppm)	Remark
TetraBDE	10	200	LPCL 2 to be reviewed by 2016
PentaBDE	10	200	LPCL 2 to be reviewed by 2016
HexaBDE	10	200	LPCL 2 to be reviewed by 2016
HeptaBDE	10	1000	LPCL 2 to be reviewed by 2016
PFOS	10	50	LPCL 2 Review by 2016; With stricter limitation for sewage sludge at 0.5 or 5 ppm
PeCB	50	50	With potential limitation for sewage sludge at 1 ppm
SCCP	1,000	1,000	
HBCD	100	1,000	LPCL 2 covers separated waste types
PCP	1,000	1,000	Only including Textile waste

Substance	LPCL 1 (ppm)	LPCL 2 (ppm)	Remark
HCBD	100	100	To be reviewed by 2016
PCN	10	10	No identified waste type is covered

As regards MPCLs the project team would suggest the lower set of MPCL options as preferred solution, if the preferred proposal for LPCLs is selected by the European Commission and the Member States.

Table 10-2: Overview of preferred proposals (MPCL 1) and alternative transitional proposals (MPCL 2) for Maximum POP Concentration Limits (MPCLs) to be specified in Annex V, Part 2 of the POP Regulation

Substance	MPCL 1 (ppm)	MPCL 2* (ppm)
TetraBDE	2500	25,000
PentaBDE	2500	25,000
HexaBDE	2500	25,000
HeptaBDE	2500	25,000
PFOS	50	500
PeCB	500	5,000
SCCP	5,000	50,000
HBCD	5,000	50,000
PCP	5,000	50,000
HCBD	1,000	10,000
PCN	1,000	10,000

* MPCL 2 to be reviewed by 2016

The preferred proposal (MPCL 1) would represent a maximally precautionary approach, taking into consideration concerns that have been raised by certain stakeholders against the MPCL limits for the first 12 POPs. In case the transitional solution is selected for the LPCLs (LPCL 2), the higher set of transitional MPCLs (MPCL 2) should be chosen in order to avoid conflicts between LPCLs and MPCLs.

10.2 Relevance of Low and Maximum POP content levels for waste flows

The following table shows the waste amounts which are expected to be concerned due to the both proposed LPCLs:

Table 10-3: Waste quantities concerned by the proposed LPCLs

Waste quantities considered as POP waste [kt]

Preferred proposal			Alternative transitional or restricted proposal		
Substance and LPCL 1	[kt]	Quantity [kt] and waste characterisation	Substance and LPCL 2	[kt]	Quantity [kt] and waste characterization
TetraBDE (10 ppm)	656	656 automotive	TetraBDE (200 ppm)	0	0
PentaBDE (10 ppm)	761 (657*)	656 automotive; 105 WEEE	PentaBDE (200 ppm)	657*	656 automotive; 0.73* WEEE
		656 automotive; 0.73* WEEE			
HexaBDE (10 ppm)	761 (657*)	656 automotive; 105 WEEE	HexaBDE (200 ppm)	0.73*	0.73* WEEE
		656 automotive; 0.73* WEEE			
HeptaBDE (10 ppm)	105 (0.73*)	105 WEEE	HeptaBDE (1000 ppm)	0.73*	0.73* WEEE
		0.73* WEEE			
PeCB (50 ppm)	0	0	PeCB (50 ppm)	0	0
PFOS (10 ppm)	1,972	1.62 Metal Plating	PFOS (100 ppm)	1.4	0.02 Metal Plating
		30.96 Photographic Industry		1.4	0.02 Photographic Industry
		0.72 Hydraulic fluids			0.72 Hydraulic fluids
		0.6 Fire fighting foams			0.6 Fire fighting foams
		1,938 Carpets			0.0 Carpets
SCCP (1,000 ppm)	42.3	34.63 Rubber Industry	SCCP (1,000 ppm)	42.3	34.63 Rubber Industry
		1.27 Sealants and Adhesives			1.27 Sealants and Adhesives
		2.33 Paints and varnishes			2.33 Paints and varnishes
		0.134 Textile			0.134 Textile
		4.406 Leather			4.406 Leather
HBCD (100 ppm)	>32.9	10.8 Textile	HBCD (1,000 ppm)	32.9	10.8 Textile
		4.52 HIPS			4.52 HIPS
		0.29 EPS/XPS (other than C&D)			0.29 EPS/XPS (other than C&D)
		3.8 XPS C&D			3.8 XPS C&D
		13.5 EPS C&D			13.5 EPS C&D
PCP (1,000 ppm)	9.6	9.6 Textile	PCP (1,000 ppm)	9.6	9.6 Textile
HCBd (100 ppm)	n.a.	n.a.	HCBd (100 ppm)	n.a.	n.a.
PCN (10 ppm)	0	0	PCN (10 ppm)	0	0

* In case C-OctaBDE containing plastics can be separated from the C-OctaBDE free plastics from WEEE category 3 and 4)

The LPCL 1 level is always lower or as high as LCPL 2. As a consequence the corresponding waste

amount of the LPCL 1 is at least as high as the waste amount related to LPCL 2. This is particularly relevant in the case of PFOS where the preferred LPCL also contains important quantities of carpet waste. However, in most other cases the higher alternative LPCL has no relevant consequence on the waste quantities to be considered as POP waste. E.g. in the case of HBCD, there was no waste identified which has a substance concentration between the preferred and the alternative LPCL. However In case of waste mixtures the resulting waste can have a substance concentration within this interval and the waste amount might increase. For some other new or candidate POP the preferred LPCL 1 and LPCL 2 are identical as no reasonable alternative can be established (e.g. SCCP, HCBd). This was the case if the preferred LPCL was set already relatively high (e.g. close to existing legal settings for SCCP and PCP) or it is so high that already for this level no waste could be identified (PCN, PeCB).

10.3 Proposed LPCLs and related impacts

The following tables (Table 10-4 to Table 10-11) list for all relevant source sectors and each POP substance the impacts of the proposed limit values in terms of POP substance quantities, related waste quantities, costs for changes in waste management, possible effectiveness to eliminate the POP substance and impacts on the elimination capacity.

The withdrawal of POPs from the eco-cycle and their elimination is considered to have positive environmental impacts as the eliminated POPs will not become subject to long range transport and will not bio-accumulate and will not cause adverse environmental and health effects. A monetarisation of the environmental impacts is not reasonably possible within the scope of the present project⁷⁷. However the environmental impact can be quantified in terms of tonnes of eliminated POP substances. The environmental impact is considered to be a function of the quantity of eliminated POP substances. A high quantity of eliminated POP substances is considered to be related to high positive environmental impacts.

In the following, for each POP substance the following is briefly discussed

- The additional costs for changes in waste management and recommended measures (including an appraisal of the proportionality of the burden)
- Possible impacts on WEEE and ELV recycling targets (only relevant for PBDEs)
- The effectiveness of the limit values concerning the elimination of the POP substance

⁷⁷ In a recent study on "Assessing the Health and Environmental Impacts in the Context of Socio-economic Analysis under REACH" methods to evaluate and compare the impacts on health and the environment with other socio-economic impacts, such as the costs to businesses and consumers have been analysed. Corresponding methods could be used to quantify impacts on health and environment. As part of the study two case studies were carried out. One of them was on the candidate POP substance HBCD. A monetarisation of the environmental impacts could not be achieved. (source: Contract ENV.D.1/SER/2009/0085r, unpublished final report version from March 2011. Publication of the final report version expected in April 2011)

- The impact in terms of additional elimination capacity
- The impacts on environment and health in terms of a quantification of the POP elimination

Table 10-4: Impacts of the proposed limit values for PBDEs in terms of POP substance quantities, related waste quantities, costs for changes in waste management, possible effectiveness to eliminate the POP substance and impacts on the elimination capacity (Reference year: 2010)

Source sector	PBDEs (tonnes)					Waste (tonnes)					Costs	Effectiveness		Elimination capacity
LPCL1: 10 ppm	Total in waste	Covered by LV	Not covered by LV	Currently incinerated	Destroyed at LV (total)	Total waste 1) 2) 3)	Covered by LV	Not covered by LV	Currently incinerated	Destroyed at LV (total)	Additional cost (million €)	Current POP elim. rate (%)	Possible POP elim. rate (%)	Share of existing capacity (change)
ELVs	243.3	243.3	0.0	115.6	243.3	655,873	655,873	0	311,540	655,873	6.9	47.5	100.0	0.30
Upholstery	91.3	0.0	91.3	30.0	30.0	5,215,540	0	5,215,540	1,715,913	1,715,913	0.0	32.9	32.9	0.00
WEEE	128.0	128.0	0.0	83.2	128.0	104,121	104,121	0	67,679	104,121	0.7	65.0	100.0	0.03
Total	462.6	371.3	91.3	228.8	401.3	5,975,534	759,994	5,215,540	2,095,131	2,475,907	7.6	49.5	86.8	0.3
LPCL2: 200 ppm and 1,000 ppm for HeptaBDE														
ELVs	243.3	243.3	0.0	115.6	243.3	655,873	655,873	0	311,540	655,873	6.9	47.5	100.0	0.30
Upholstery	91.3	0.0	91.3	30.0	30.0	5,215,540	0	5,215,540	1,715,913	1,715,913	0.0	32.9	32.9	0.00
WEEE	128.0	0.0	128.0	83.2	83.2	104,121	0	104,121	67,679	67,679	0.0	65.0	65.0	0.00
Total	462.6	243.3	219.3	228.8	356.5	5,975,534	655,873	5,319,661	2,095,131	2,439,464	6.9	49.5	77.1	0.30

- 1) Quantity of SLF (highest contamination: PentaBDE = 208 ppm); Quantity of contaminated PUR foam = 6,1 kt (highest contamination: PentaBDE = 22,400 ppm)
- 2) Total quantity of upholstery waste (highest contamination: PentaBDE = 9.8 ppm); Quantity of contaminated PUR foam = 2.4 kt (highest contamination: PentaBDE = 21,280 ppm)
- 3) Quantity of the plastic fraction of WEEE categories 3 and 4 (highest contaminations: PentaBDE 77 ppm and HeptaBDE 435 ppm); quantity of C-OctaBDE containing plastics from WEEE categories 3 and 4 = 732 tonnes (highest contaminations: PentaBDE = 10,938 ppm and HeptaBDE = 61,950 ppm)

Table 10-4 shows the impacts for PBDEs related to the relevant source sectors.

PUR foams from the ELV sector need specific consideration because they seem to be covered identically by both limit values. In fact, the figures in Table 10-4 reflect the situation in 2010 but do not reflect the situation for the years to come. The expected concentration levels of all relevant congener groups in the SLF are already in 2010 significantly below LPCL 2 and would therefore not be covered by this limit value. However, the expected concentration level of the PentaBDE congener is calculated at 208 ppm in the SLF for the year 2010 and is thus just above the proposed LPCL 2. The concentration levels are phasing out with a continuously decreasing trend. After 2010 it is therefore expected that the contamination levels will also for the PentaBDE congener group below the LPCL 2 level of 200 ppm. As a consequence, the corresponding POP substance amount (243 tonnes in 2010) and waste amount (656 tonnes of SLF) will not be covered at LPCL 2 in the years after 2010. The elimination of POPs from the ELV sector would therefore only be increased at the LPCL 1 together with accompanying measures (i.e. separation of contaminated PUR foams from ELVs).

PUR foams from upholstery contribute about 20% (91.3 tonnes) to the input of the relevant PBDE congener groups. Upholstery waste is not covered by both proposed LPCLs.

The plastic fraction from WEEE is not covered by LPCL 2 but by LPCL 1. The elimination of the relevant PBDEs would therefore only be increased at the LPCL 1 together with accompanying measures (i.e. separation of contaminated plastics from WEEE).

Cost considerations

Costs for changes in waste management and recommended measures arise on the one hand due to necessity to incinerate waste quantities which are currently landfilled. As indicated in Table 10-4 these direct costs due to corresponding shifts sum up to 7.6 million € at LPCL 1 and 6.9 million € at LPCL 2. The amount of up to 6.9 million € would be required at both limit values for the additional incineration of up to 345,000 tonnes of SLF (if the limit values will be exceeded). At LPCL 1 the plastic fraction from WEEE categories 3 and 4 would have to be incinerated at additional costs up to 0.7 million €. This is not necessary at LPCL 2.

On the other hand additional costs which are not considered in Table 10-4 will arise due to the proposed measures (1) “separation and destruction of PUR foams from cars built in the year 2000 or before if the PUR foam is intended to be recycled or to be landfilled in the corresponding process” and (2) “separation and destruction of ABS plastics from cathode ray computer and TV monitors produced in 2003 and before if the ABS plastic is intended to be recycled or to be landfilled in the corresponding process”.

Concerning the cost impacts of the first proposed measure the following has to be considered: Only a limited number of cars would be concerned and therefore PUR foams from the majority of cars could be recycled or otherwise treated. The average costs per car about 13€ can be considered very low compared to other car related costs such as the price of a car or costs for the operation of a car.

According to a worst case estimate, the annual cost impact could be considered relevant in the beginning (up to ~63 million € in 2012) but will become low in 2016 (up to ~23 million €) and insignificant after 2020. These figures are a worst case estimate because (1) PUR foams only have to be separated if the SLF or fibre fraction is not intended for incineration (which is currently usually the case) and (2) the number of affected cars could be significantly reduced by identifying the relevant ELVs which are equipped with C-PentaBDE containing PUR foams more specifically than by the building year 2000 and before. ACEA will try to obtain more specific data on the use of the relevant BDE congeners in automobile manufacturing. If it will be possible to specify by when specific car manufacturers have stopped the use, it may become possible to significantly reduce the number of affected cars. This would be feasible e.g. by establishing a list of manufacturers and/or cars which specifies for the individual manufacturer and/or car the year until which the PUR foams would have to be removed (for details see chapter 9.5.1).

Concerning the cost impacts of the second proposed measure it has to be considered that the separation and destruction of ABS plastics containing the relevant PBDEs does not go beyond the already existing requirement from the WEEE Directive to separate BFR containing plastics from separately collected WEEE. Related costs are not an impact of a limit value to be established under the POP regulation. Hence, there is no cost impact from the proposed LPCLs (for details see chapter 9.5.2).

Impacts on ELV and WEEE recycling targets

Vehicles in average consist of 75% metal and 25% plastic compounds including approximately 0.5% PUR foams by weight. Only a share of this PUR foam is prone for C-PentaBDE contamination (mainly PUR foam in cars produced until the year 2000). In order to not interfere with recycling targets it thus is recommended to separate PUR foam from other plastic parts.

Otherwise thermal treatment (energy recovery) of the whole fibre fraction (approximately 5% of the vehicle weight) of the shredder light fraction could be the preferred treatment option. Given the further decline of PBDE contamination in PUR foam, the recycling/reuse of plastic fractions would not cause any problems anymore in 2015. The recycling targets of the ELV Directive would be achievable.

The relevant PBDEs are particularly contained in old EEE (mainly produced in the 1990ies until 2003). Most relevant are ABS plastics from Information and Communication Technology and consumer equipment, Cathode Ray Tube (CRT) monitors and CRT TVs. Therefore, the study concludes that OctaBDE concentrations have to be expected in plastics from ICT (with the highest levels in monitors), as well as plastics from CRT TVs. This is seen as a consequence from the application of OctaBDE in ABS plastics in the past. Manual separation of the plastics from old computer and TV monitors seems to be one feasible option by which the bulk of the relevant PBDE congeners could be separated from the WEEE waste stream. The targets for reuse and recycling according to the WEEE Directive would be achievable.

It should be noted, however, that due to the WEEE requirement about 7% of BFR containing plastics are separated from WEEE in practice (see chapter 9.5.2). These separated BFR containing plastics include the relevant PBDE containing plastics. The proposed LPCLs do therefore not impact on the achievability of the WEEE recycling targets.

Effectiveness of POP elimination

The columns showing the current and possible POP elimination rates in Table 10-4 allow an evaluation of the effectiveness to eliminate the POP substance. The “current POP elimination rate” shows how much of the POP is destroyed under current waste management options by waste incineration. The “possible POP elimination rate” shows how much of the POP could be destroyed.

In the case of ELVs (52.6% of the relevant PBDEs in waste) it is possible to increase the POP elimination rate from currently 47.5% to 100% if all contaminated PUR foams from ELVs will be destroyed. According to the figures in Table 10-4, the possible elimination rates would be identical for LPCL 1 and LPCL 2. As explained above, this is due to the fact that the figures represent the situation in 2010. It is therefore expected that the POP elimination rates in the case of ELV would remain unchanged if LPCL 2 will be established but that they will increase close to 100% if LPCL 1 will be chosen.

Concerning upholstery (19.7% of the relevant PBDEs in waste) the elimination rate will remain at a low level of about 33% at both LPCLs.

With respect to WEEE (27.7% of the relevant PBDEs in waste) the current elimination rate could be increased from currently 65% close to 100% under LPCL 1. At LPCL 2 the elimination rate would remain at 65%. However, if the WEEE requirement to separate BFR containing plastics will be efficiently implemented, the POP elimination rate from WEEE will be close to 100% anyway.

Impact on waste management capacities

If all relevant separated waste streams containing HBCD would have to be incinerated this would require an annual capacity of about 381,000 tonnes (at LPCL 1) or 344,000 tonnes (at LPCL 2). This corresponds to 0.33% or 0.30% respectively of the current incineration capacity. Capacity problems do therefore not arise at EU level. The high energy content of PUR foams and WEEE plastics could be used for energy recovery.

Environmental impact

The total quantity of the relevant PBDEs in waste to be disposed of after 2010 amounts to about 780 tonnes⁷⁸. The ELV sector contributes about 432 tonnes to this amount.

As explained above, the separation and destruction of the relevant PBDEs from WEEE is already

⁷⁸ This figure is possibly slightly underestimated as it is based on average lifetimes of 12 years for cars, 10 years for upholstery and 7 to 9 years for EEE. The lifetimes for cars and EEE, however, may be some years longer and the quantities of relevant PBDEs to be disposed of after 2010 may be slightly higher.

covered by the WEEE directive. Both LPCLs do not cover upholstery waste. Therefore the direct environmental impact from the LPCLs under the POP regulation on the PBDEs will be related to the source sector of ELVs. The elimination rate for ELVs can be increased from 47.5% close to 100%. It is expected that at LPCL 1 an additional quantity of approximately 227 tonnes of PBDEs will be eliminated whereas at LPCL 2 no additional PBDEs would be eliminated.

Table 10-5: Impacts of the proposed limit values for PFOS in terms of POP substance quantities, related waste quantities, costs for changes in waste management, possible effectiveness to eliminate the POP substance and impacts on the elimination capacity (Reference year: 2010)

Source sector	PFOS (tonnes)					Waste (tonnes)					Costs	Effectiveness		Elimination capacity
LPCL1: 10 ppm	Total in waste	Covered by LV	Not covered by LV	Currently incinerated	Destroyed at LV (total)	Total waste 1) to 7)	Covered by LV	Not covered by LV	Currently incinerated	Destroyed at LV (total)	Additional cost (million €)	Current POP elim. rate (%)	Possible POP elim. rate (%)	Share of existing capacity (change)
Metal plating	1.68	1.31	0.38	1.5	1.5	102,251	2,746	99,506	52,499	52,499	0.0	88.8	88.8	0.00
Photographic industry	1.43	0.63	0.80	0.7	0.9	214,975	30,960	184,015	75,931	91,501	0.3	47.9	62.6	0.08
Hydraulic fluids	0.72	0.72	0.00	0.7	0.7	715	715	0	715	715	0.0	100.0	100.0	0.00
Fire fighting foams	90.28	90.28	0.00	90.3	90.3	9,028	9,028	0	9,028	9,028	0.0	100.0	100.0	0.01
Leather upholstery	5.71	0	5.71	1.9	1.9	2,378,076	0	2,378,076	782,387	782,387	0.0	32.9	32.9	0.67
Carpet industry	146	146	0	48.0	146.0	1,938,018	1,938,018	0	637,608	1,938,018	26.0	32.9	100.0	1.67
Sewage sludge	1.68	0	1.679	0.3	0.3	11,578,000	0	11,578,000	2,006,719	2,006,719	0.0	17.3	17.3	1.73
Total	247.50	238.94	8.56	142.56	241.7	16,221,063	1,981,468	14,239,596	3,512,759	4,830,742	26.4	57.6	97.7	0.08
LPCL2 = 50 ppm														
Metal plating	1.68	1.31	0.38	0.67	1.7	102251	2746	99506	372	2374	0.0	39.9	100.0	0.00
Photographic industry	1.43	0.16	1.26	0.69	0.7	214,975	20.5	214,954.6	75,931	75,931	0.0	47.9	47.9	0.00
Hydraulic fluids	0.72	0.72	0.00	0.72	0.7	715	715	0	715	715	0.0	100.0	100.0	0.00
Fire fighting foams	90.28	90.28	0.00	90.28	90.3	9,028	9,028	0	9,028	9,028	0.0	100.0	100.0	0.01
Leather upholstery	5.71	0.00	5.71	1.88	1.9	2,378,076	0	2,378,076	782,387	782,387	0.0	32.9	32.9	0.67
Carpet industry	146	146	0.00	48.03	146.0	1,938,018	1,938,018	0	637,608	1,938,018	26.0	32.9	100.0	1.67
Sewage sludge	1.68	0.00	1.68	0.29	0.3	11,578,000	0	11,578,000	2,006,719	2,006,719	0.0	17.3	17.3	1.73
Total	247.50	238.47	9.03	142.56	241.5	16221063	1950528	14270536	3564886	4865296	26.0	57.6	97.6	0.00

- 1) Chromium sludge with 3.8 ppm; Vacuum distillation concentrate with 240 ppm; Activated carbon from treatment with 297 ppm; Activated carbon filter with 950 ppm and Ion exchange resin with 19,000 ppm
- 2) Solid residue recycling from x-ray pictures with 127,898 ppm; Activated carbon from x-ray pictures 1,000 ppm, Desilvered solution from x-ray pictures with 20 ppm; X-ray picture waste with 13.45 ppm; Other film wastes with 4.33 ppm; Picture waste with 4.33 ppm; Liquid photographic fluids 410,000 ppm and recycled photo plates with 1 ppm

- 3) Hydraulic fluids with 1,000 ppm
- 4) Fire fighting foams with 10,000 ppm
- 5) Upholstery with 2.4 ppm
- 6) Carpets with 75 ppm
- 7) Sewage sludge low contaminated with 0.1 ppm and sewage sludge high contaminated with 1 ppm

Table 10-5 shows the impacts of both proposed limit values. The set LPCL of 50 ppm differs only slightly compared to a set LPCL of 10 ppm. Only in case of the photographic industry some waste types are not covered at the higher LPCL of 50 ppm. The major PFOS source which would result in additional incineration amounts is related to carpet, which is covered by both proposed LPCLs.

The elimination of PFOS would be increased at both LPCLs if appropriate accompanying measures are implemented (i.e. separation and destruction of contaminated carpets).

Cost considerations

The majority of the additional costs for waste management arise from the carpet industry with about 26 million € and would cover 100 % of the involved PFOS amount from this industry. PFOS from the other industries only have a limited effect due to the following reasons:

- They are already incinerated (e.g. hydraulic fluids),
- have to be incinerated due to already existing legislation (e.g. fire fighting foams)
- are not affected by the limit values as the PFOS contamination is below proposed LPCLs (e.g. upholstery, sewage sludge)
- the additional waste amounts are much less compared to the carpet industry (e.g. Metal industry, Photographic industry)

Effectiveness of POP elimination

There are three categories of PFOS containing waste types. The first category are waste types in which case all the included PFOS is already destroyed independent from the set LPCL. These waste streams are fire fighting foams and hydraulic fluids. The second waste type involves such wastes, which would lead to an additional incineration due to a set LPCL. This is the case for the carpet industry and the photographic industry in which cases wastes which are currently landfilled would have to be incinerated. In case of the carpet industry all the included PFOS would be destroyed, whereas in case of the photographic industry the PFOS destruction would rise from about 48 to 63 %. The third category of waste type are not affected by any of the set LPCL. In the metal industry a large fraction of the PFOS amount (89 %) is already incinerated. Several of the waste types from the metal industry as chromium sludge or vacuum distillate are recycled. However the relevant process steps are often thermal processing steps which also lead to PFOS destruction. In case of Leather upholstery waste streams and sewages sludge only a relative small fraction is currently incinerated. This will not change with the setting of one of the proposed limit values. However, due to the low PFOS content, the total amount of PFOS which is not destroyed is relatively low.

Environmental impact

Overall the PFOS destruction would rise from about 58% to about 98% for both LPCL settings. This strong increase is mainly due to the PFOS destruction from the carpet industry (75 ppm PFOS

content) which is covered at the set LPCL levels of 10 ppm and 50 ppm.

It is expected that at LPCL 1 an additional quantity of approximately 15,881 tonnes of PFOS will be eliminated whereas at LPCL 2 15,865 tonnes additional PFOS would be eliminated.

It should be highlighted that the waste types with the highest total PFOS amounts are phasing out. The corresponding waste types are carpets, fire fighting foams and leather upholstery. Their PFOS content should become negligible within the next 1 to 10 years. These waste types cover currently about 98 % of the annual PFOS amount. The result of too late legislative action would therefore be that relevant amounts of PFOS would not be destroyed and that the remaining amounts which are properly treated in the future are relatively small compared to the currently circulated and landfilled amounts. This also means that too late actions will have no relevant environmental impact, regardless of the set LPCL.

The remaining ongoing sources of PFOS are very low. However special care should be taken regarding the sewage sludge. As this material is currently also used for agricultural purposes this bears additional risk of PFOS entering the food chain. To cover this particular risk, the elaborated LPCL values from this report may not be sufficiently qualified and could be supplemented by other more stringent limit values as e.g. in the Regulation on fertilizers 2003/2003/EC or the Directive on sewage sludge 86/278/EEC.

Table 10-6: Impacts of the proposed limit values for PeCB in terms of POP substance quantities, related waste quantities, costs for changes in waste management, possible effectiveness to eliminate the POP substance and impacts on the elimination capacity (Reference year: 2010)

Source sector	PeCB (tonnes)					Waste (tonnes)					Costs	Effectiveness		Elimination capacity
LPCL1: 50 ppm	Total in waste	Covered by LV	Not covered by LV	Currently incinerated	Destroyed at LV (total)	Total waste 1) to 5)	Covered by LV	Not covered by LV	Currently incinerated	Destroyed at LV (total)	Additional cost (million €)	Current POP elim. rate (%)	Possible POP elim. rate (%)	Share of existing capacity (change)
MSWI	0.08	0.0	0.08	0.0	0.00	1,131,105	0	1,131,105	0	0	0.0	0.0	0.0	0.00
HWI	0.005	0.0	0.005	0.0	0.0	472,854	0	472,854	0	0	0.0	0.0	0.0	0.00
Coal combustion	0.16	0.0	0.16	0.0	0.0	65,884,140	0	65,884,140	0	0	0.0	0.0	0.0	0.00
Domestic burning	0.0048	0.0	0.0048	0.0010	0.0010	1,711,216	0	1,711,216	342,243	342,243	0.0	20.0	20.0	0.30
Waste water treatment	0.05615	0.0	0.05615	0.0099	0.0099	11,578,000	0	11,578,000	2,031,641	2,031,641	0.0	17.5	17.5	1.75
Total	0.31	0.0	0.31	0.01	0.0	80,777,314	0	80,777,314	2,373,884	2,373,884	64.3	3.5	3.5	0.00

- 1) Fly ash contaminated to 0.07442 ppm
- 2) Fly ash contaminated to 0.00957ppm
- 3) Fly ashes contaminated to 0.0024 ppm
- 4) Fly ashes contaminated to 0.07442 ppm, 0.0024 ppm and 0.0002 ppm
- 5) Sewage sludge contaminated to 0.00485 ppm

Table 10-6 shows the impacts of both proposed limit values. It can be seen that none of the identified waste streams are covered by the set LPCL as the PeCB concentration is usually in the low ppb range. Therefore an evaluation regarding costs, effectiveness of POP elimination or environmental impact is not given. However, due to the nature and waste management of the PeCB containing waste types (mainly different types of ashes as Fly ash and boiler ash), a good fraction of this waste is expected to be already covered by other limit values such as for PCDD/F or PCB.

Table 10-7: Impacts of the proposed limit values for SCCP in terms of POP substance quantities, related waste quantities, costs for changes in waste management, possible effectiveness to eliminate the POP substance and impacts on the elimination capacity (Reference year: 2010)

Source sector	SCCP (tonnes)					Waste (tonnes)					Costs	Effectiveness		Elimination capacity
LPCL1: 1,000 ppm	Total in waste	Covered by LV	Not covered by LV	Currently incinerated	Destroyed at LV (total)	Total waste 1) to 6)	Covered by LV	Not covered by LV	Currently incinerated	Destroyed at LV (total)	Additional cost (million €)	Current POP elim. rate (%)	Possible POP elim. rate (%)	Share of existing capacity (change)
Rubber industry	1,254	1,254	0	413	1,254	34,913	34,913	0	11,486	34,913	0.5	32.9	100.0	0.02
Sealant and Adhesives	424	424	0	139	424	2,687	2,687	0	872	2,687	0.0	32.9	100.0	0.002
Paint and Varnishes	320	320	0	105	320	2,330	2,330	0	767	2,330	0.0	32.9	100.0	0.001
Textile Industry	27	27	0	11	27	134	134	0	56	134	0.0	41.7	100.0	0.000
Leather Industry	17	17	0	6	17	4,320	4,320	0	1,508	4,320	0.1	34.7	100.0	0.002
Sewage sludge	40	0	40	7	7	11,578,000	0	11,578,000	2,006,720	2,006,720	0.0	17.3	17.3	0.000
Total	2,082	2,042	40	682	2,049	11,622,384	44,384	11,578,000	2,021,409	2,051,104	0.6	32.7	98.4	0.02

- 1) Gaskets and hoses with 135,000 ppm; conveyor belts with 33,333 ppm
- 2) Sealant and adhesives contaminated packaging with 20,000ppm, sealant and adhesives with 200,000 ppm; Production losses with 1,000,000 ppm
- 3) Paints and varnishes with 130,000 ppm or 300,000 ppm
- 4) Textiles with 200,000 ppm; Production losses with 1,000,000 ppm
- 5) Leather with 4,000 ppm
- 6) Sewage sludge low contaminated with 1 ppm; sewage sludge high contaminated 50 ppm

Table 10-7 shows the impacts of the proposed limit values. For SCCP only one SCCP has been proposed because already at 1,000 ppm nearly all the SCCP containing waste types are covered. The only waste type which is not covered is sewage sludge with a SCCP concentration of 1 or 50 ppm.

The elimination of SCCP would be increased at both LPCLs if appropriate accompanying measures are implemented (i.e. separation and destruction of contaminated conveyor belts for underground mining).

Cost considerations

The additional cost for the destruction of SCCP in different waste types is relatively low due to the low waste amounts. All together the expected additional costs are about 590,000 €/y. The most important sector is the rubber industry which covers about 60 % of the annual SCCP in waste and with about 470,000 €/y of additional costs it is with 80 % the main reason for the additional costs. Considering that this waste type can easily be separated (mainly conveyor belts for underground mining) only with this measure the destruction of SCCP could be raised from about 33 % to about 73 %.

Effectiveness of POP elimination

With the proposed limit value of 1,000 ppm about 98.4 % of annual SCCP in 2011 would be covered. The only waste type which is not covered is sewage sludge with a SCCP concentration of 1 or 50 ppm. However, this waste type represent about 99.6 % (11.58 million tonnes) of the waste amount but only 1.59 % of the SCCP amount if it would be completely covered by the LPCL.

At the LPCL of 50 ppm also about 5 % of the sewage sludge fraction would be covered, which would increase the cost by about 10 million €/y but would only cover about 1 % of the annual SCCP amount in waste.

Environmental impact

Overall the SCCP destruction would rise from about 32.7% to about 98% for the proposed LPCL setting. This strong increase is mainly due to the SCCP destruction from the rubber industry.

It is expected that at a proposed LPCLs an additional quantity of approximately 16,200 tonnes of SCCP would be eliminated of which about 6,700 t are from the rubber industry, 5,650 t are from the sealant and adhesives and 3,500 t from the paint and varnishes industry.

Table 10-8: Impacts of the proposed limit values for HBCD in terms of POP substance quantities, related waste quantities, costs for changes in waste management, possible effectiveness to eliminate the POP substance and impacts on the elimination capacity (Reference year: 2010)

Source sector	HBCD (tonnes)					Waste (tonnes)					Costs	Effectiveness		Elimination capacity
	Total in waste	Covered by LV	Not covered by LV	Currently incinerated	Destroyed at LV (total)	Total waste 1) to 5)	Covered by LV	Not covered by LV	Currently incinerated	Destroyed at LV (total)		Current POP elim. rate (%)	Possible POP elim. rate (%)	
LPCL1: 100 ppm														
EPS construction	68.8	68.8	0	36.3	68.8	13,495	13,495	0	7,166	13,495	0.17	52.7	< 100.0	0.005
XPS construction	75.8	75.8	0	44.0	75.8	3,790	3,790	0	2,202	3,790	0.03	58.1	< 100.0	0.001
EPS/XPS other	3.9	3.9	0	1.2	3.9	288	288	0	92	288	0.01	31.9	< 100.0	0.000
HIPS	181	181	0	43.1	181.0	4,524	4,524	0	1,077	4,524	0.07	23.8	< 100.0	0.003
PD textiles	867	867	0	123.1	866.7	10,834	10,834	0	1,538	10,834	0.19	14.2	< 100.0	0.008
Total	1196.2	1196.2	0.0	247.7	1196.2	32930.7	32930.7	0.0	12074.8	32930.7	0.46	20.7	< 100.0	0.018
LPCL2: 1,000 ppm														
EPS construction	68.8	68.8	0	36.3	68.8	13495	13495	0	7,166	13,495	0.17	52.7	< 100.0	0.005
XPS construction	75.8	75.8	0	44.0	75.8	3,790	3,790	0	2,202	3,790	0.03	58.1	< 100.0	0.001
EPS/XPS other	3.9	3.9	0	1.2	3.9	288	288	0	92	288	0.01	31.9	< 100.0	0.000
HIPS	181	181	0	43.1	181.0	4,524	4,524	0	1,077	4,524	0.07	23.8	< 100.0	0.003
PD textiles	867	867	0	123.1	866.7	10,834	10,834	0	1,538	10,834	0.19	14.2	< 100.0	0.008
Total	1196.2	1196.2	0.0	247.7	1196.2	32930.7	32930.7	0.0	12074.8	32930.7	0.46	20.7	< 100.0	0.018

- 1) Quantity of separated waste of EPS for construction (average contamination = 5,100 ppm); If not separated, dilution in the plastic fraction of C&D waste (quantity = 9,700,000 tonnes) to concentrations up to 7.1 ppm
- 2) Quantity of separated waste of XPS for construction (average contamination = 5,100 ppm); If not separated, dilution in the plastic fraction of C&D waste (quantity = 9,700,000 tonnes) to concentrations up to 7.1 ppm
- 3) Quantity of separated waste of EPS and XPS for non-construction uses (average contamination = 13500 ppm); If not separated, dilution in the MSW stream (quantity = 255,371,000 tonnes) to concentrations up to 0.015 ppm
- 4) Quantity of separated waste of HBCD containing HIPS products (average contamination = 40.000 ppm); If not separated, dilution in MSW (quantity = 255,371,000 tonnes) and other waste streams
- 5) Quantity of separated waste of HBCD containing Textile products (average contamination = 80.000 ppm); If not separated, dilution in MSW

Table 10-8 shows that the impacts of both limit values are identical. This is due to the fact that all relevant HBCD containing products contain HBCD levels above the proposed limit values. As a consequence all waste streams consisting of separated products are covered by the limit values. The figures in the table designate separated waste streams. Waste streams in which the substance is diluted if not separated are indicated in the corresponding notes to the table. The LPCL 1 value corresponds to a precautionary approach. Contaminated waste which is not separated is expected to be diluted in concentration levels below the LPCL 1.

The elimination of HBCD would be increased at both LPCLs if appropriate accompanying measures are implemented (i.e. separation of contaminated PS insulation boards from construction and demolition waste).

Cost considerations

Costs for changes in waste management and recommended measures arise on the one hand due to necessity to incinerate waste quantities which are currently landfilled (in most cases) and/or recycled (this is the case for some of the EPS wastes). These direct costs due to corresponding shifts sum up to 0.46 million € in 2010 as indicated in Table 10-8.

On the other hand additional costs will arise due to the proposed measure to separate HBCD containing waste from not contaminated waste. In order to assure that the majority of the HBCD content of PS foams from construction will be destroyed, it should be attempted to separate HBCD contaminated PS insulation boards as far as reasonably possible. This will enable to destroy the HBCD content of the contaminated fraction by incineration. A high degree of separation could be achieved e.g. by implementing deconstruction/renovation management plans for buildings which take into account whether it is known/probable or not if PS foams used in a specific building contain HBCD or not (for details see chapter 9.5.3). Marking or labelling of HBCD free products could facilitate the separation and the implementation of deconstruction/renovation management plans could be supported in the currently established guidelines for construction and demolition waste. Related cost impacts are difficult to quantify and cannot be evaluated without considering developments in the construction and deconstruction sector. Possible costs can therefore not be simply allocated to the impacts of measures proposed in the context of the POP regulation.

It has to be emphasised once again, that the relevance of HBCD waste from construction and demolition waste is still comparatively low (approximately 17,000 tonnes of waste containing about 145 tonnes of HBCD in 2010) but will become increasingly important within the next decades (approximately 150,000 tonnes of waste in 2030 and 400,000 tonnes of waste in 2040) with an expected peak after 2050. Other source sectors are continuously phasing out.

Effectiveness of POP elimination

The columns showing the current and possible POP elimination rates in Table 10-8 allow an evaluation of the effectiveness to eliminate the POP substance. The “current POP elimination rate” shows how much of the POP is destroyed under current waste management options by waste

incineration. The “possible POP elimination rate” shows how much of the POP could be destroyed.

In the case of HBCD the “possible POP elimination rate” amounts in all cases up to 100%. The 100% share is a theoretical value and depends on the efficiency of separation of the contaminated waste from the waste stream. This means e.g. for EPS waste from construction that the HBCD elimination rate will be between 52.7% (current rate) and 100% (possible rate). A 100% elimination would be achieved if all EPS insulation boards could be separated during deconstruction/demolition for incineration. If appropriate deconstruction/renovation management plans for buildings will be implemented it is expected that a high degree of separation will be possible. As a consequence it is expected that the efficiency of the POP elimination will be close to 100% for EPS from construction. The same can be stated for XPS waste from construction. Concerning other EPS/XPS waste it would be possible to increase the share of elimination by ~25% (approximately the share which is currently recycled) and to increase the POP elimination rate from currently ~32% to above 55%. Concerning HIPS and PD textiles it is considered difficult to separate these wastes from the corresponding waste stream and it is expected that the POP elimination rates will remain close to about 24% and 14% respectively. It has to be kept in mind that the relevance of the latter three source sectors is continuously phasing out.

Impact on waste management capacities

If all relevant separated waste streams containing HBCD would have to be incinerated this would require an annual capacity of about 20,000 tonnes or 0.018% of the current incineration capacity. Capacity problems do therefore not arise at EU level. The high energy content of most HBCD containing products could be used for energy recovery. Within the coming decades the annual quantities of HBCD containing waste from construction will immensely increase but will remain below 0.2% of the current incineration capacity.

Environmental impact

The environmental impact of the limit values and the related measures depends to a large degree from the management of the wastes from the EPS construction and XPS construction sector. The total quantity of HBCD in waste to be disposed of after 2010 amounts to almost 198,000 tonnes HBCD. At current POP elimination rates about 52% of this amount would be destroyed. If it will be possible to separate and destroy 100% of the waste only from the two mentioned sectors, about 95% of this amount would be destroyed. As a consequence an additional amount of about 85,000 tonnes of HBCD would be destroyed (~188,000 tonnes instead of ~ 103,000 tonnes). If it will be possible to separate and destroy 90% of the waste only from the two mentioned sectors, about 86% of this amount would be destroyed. As a consequence an additional amount of about 66,000 tonnes of HBCD would be destroyed (~169,000 tonnes instead of ~ 103,000 tonnes).

Table 10-9: Impacts of the proposed limit values for PCP in terms of POP substance quantities, related waste quantities, costs for changes in waste management, possible effectiveness to eliminate the POP substance and impacts on the elimination capacity (Reference year: 2010)

Source sector	PCP (tonnes)					Waste (tonnes)					Costs	Effectiveness		Elimination capacity
	Total in waste	Covered by LV	Not covered by LV	Currently incinerated	Destroyed at LV (total)	Total waste 1) and 2)	Covered by LV	Not covered by LV	Currently incinerated	Destroyed at LV (total)		Current POP elim. rate (%)	Possible POP elim. rate (%)	
LPCL1: 1,000 ppm														
Wood impregnation	1838	0.0	1838	1838	1838	2,941,066	0	2,941,066	2,941,066	2,941,066	0.0	100.0	100.0	0.00
Textile	239	239	0	79	239	9,568	9,568	0	3,148	9,568	0.128	32.9	100.0	0.01
Total	2077	239	1838	1917	2077	2,950,634	9,568	2,941,066	2,944,214	2,950,634	0.128	92.3	100.0	0.01

1) Impregnated wood with 625 ppm

2) Textile with 25,000 ppm

Table 10-9 shows the impacts of the proposed limit values. It can be seen that with the set LPCL all the PCP could be incinerated.

Cost considerations

As the amount of additional material which has to be incinerated due to the set limit value is quite small, the corresponding costs are also low. The reason is that currently the major amount of PCP is already incinerated. With 128,000 € within the EU 27 this impact is considered negligible.

Effectiveness of POP elimination

With the proposed limit value all the PCP containing waste would be covered. Typical PCP contaminated textiles are backcoated and should be easy to identify compared to non-backcoated textiles. However not all backcoated textile contain PCP and the practical awareness regarding identification and separation might be difficult.

Environmental impact

The majority of the PCP is already incinerated and the relatively small remainder is landfilled. The environmental impact of the set LPCL is considered to be relatively low, due to the small amount which is landfilled and the tight bonding of PCP on the textile and the corresponding low leachability of this waste type. It is expected that at a proposed LPCLs an additional quantity of approximately 11,155 tonnes of PCP would be eliminated.

Table 10-10: Impacts of the proposed limit values for HCBd in terms of POP substance quantities, related waste quantities, costs for changes in waste management, possible effectiveness to eliminate the POP substance and impacts on the elimination capacity (Reference year: 2010)

Source sector	HCBd (tonnes)					Waste (tonnes)					Costs	Effectiveness		Elimination capacity
	Total in waste	Covered by LV	Not covered by LV	Currently incinerated	Destroyed at LV (total)	Total waste 1)	Covered by LV	Not covered by LV	Currently incinerated	Destroyed at LV (total)	Additional cost (million €)	Current POP elim. rate (%)	Possible POP elim. rate (%)	Share of existing capacity (change)
LPCL1: 100 ppm														
Sewage sludge	5.89	0	5.89	1.13	1.13	9,816,000	0	9,816,000	1,891,000	1,891,000	0.0	19.2	19.2	0.00
Total	5.89	0.00	5.89	1.13	1.13	9,816,000	0	9,816,000	1,891,000	1,891,000	0.0	19.2	19.2	0.00

1) Sewage sludge with 0,6 ppb

Table 10-4 shows the impacts of proposed limit values.

The information regarding HCBd in waste streams are very scarce. It is known that HCBd should be in several wastes streams from the chemical industry however, no detailed data are available to assess the consequences of set limit values regarding **Cost considerations, Effectiveness of POP elimination or Environmental impact**. A relatively high LPCL has been proposed to avoid unexpected and unacceptable costs. However, this value should be reviewed in the following year with elaborated measurement data and possible HCBd containing waste stream identification.

Table 10-11: Impacts of the proposed limit values for PCN in terms of POP substance quantities, related waste quantities, costs for changes in waste management, possible effectiveness to eliminate the POP substance and impacts on the elimination capacity (Reference year: 2010)

Source sector	PCN (tonnes)					Waste (tonnes)					Costs	Effectiveness		Elimination capacity
	Total in waste	Covered by LV	Not covered by LV	Currently incinerated	Destroyed at LV (total)	Total waste 1) to 6)	Covered by LV	Not covered by LV	Currently incinerated	Destroyed at LV (total)		Current POP elim. rate (%)	Possible POP elim. rate (%)	
LPCL1: 1,000 ppm														
MSWI	0.04	0.00	0.04	0.00	0.00	12,975,932	0	12,975,932	0	0	0.0	0.0	0.0	0.00
HospWI ¹⁾	0.07	0.00	0.07	0.00	0.00	140,204	0	140,204	0	0	0.0	0.0	0.0	0.000
Domestic burning	0.01	0.00	0.01	0.00	0.00	1,711,216	0	1,711,216	342,243	342,243	0.0	20.0	20.0	0.000
WWT	0.52	0.00	0.52	0.09	0.09	11,578,000	0	11,578,000	2,031,641	2,031,641	0.0	17.6	17.6	0.000
secondary Cu	2.59	0.00	2.59	0.00	0.00	613,800	0	613,800	0	0	0.0	0.0	0.0	0.000
secondary Al ¹⁾	0.002	0.000	0.002	0.000	0.000	43,353	0	43,353	0	0	0.0	0.0	0.0	0.000
Total	3.23	0.00	3.23	0.09	0.09	27,062,504	0	27,062,504	2,373,884	2,373,884	0.0	2.9	2.9	0.00

1) Fly ash with 19.95 ppb, bottom ash with 1.28 ppb

2) Fly ash with 5,439 ppb, bottom ash with 1.28 ppb

3) Fly ash wood with 7.6 ppb, fly ash solid fuel with 5.4 ppb and fly ash mixed waste with 6 ppb

4) Sewage sludge with 40.25, 76 and 4.5 ppb

5) Cu slag with 4,224 ppb

6) Al filter dust with 38.43 ppb

Table 10-11 shows the impacts of both proposed limit values. It can be seen that none of the identified waste streams are covered by the set LPCL as the PCN concentration are below 5 ppm and in many cases in the range of ppb. Therefore an evaluation regarding costs, effectiveness of POP elimination or environmental impact is not given. However, due to the nature and current waste management of the PCN containing waste types (mainly different types of ashes as Fly ash and boiler ash), a good fraction of this waste is expected to be already covered by other limit values such as for PCDD/F or PCB.

10.4 Proposed measures and related impacts

The alternative transitional proposals for LPCLs (LPCL 2) take particularly account of the position of some Member States, expressed at or after the TAC Meeting in September 2010, of setting comparatively high (i.e. less strict) concentration limit values initially until more data becomes available.

However, also when the alternative transitional proposals will be set, all appropriate measures to destroy or irreversibly transform the POP substances should be taken as soon as possible in order to reduce the occurrence and the cycling of these substances and to avoid their carry over and dilution in new diverse products and waste.

Corresponding measures for the individual substances which are particularly considered appropriate are related to specific waste management activities. The relevant impacts related to these measures have been evaluated. The waste management activities, substances, proposed measures and the evaluation of relevant impacts are listed in Table 10-12. If the proposed measure will be efficiently implemented, it is expected that the concentration levels in the relevant waste streams will be reduced below proposed LPCLs (LPCL 1).

Table 10-12: Overview of relevant waste management activities, proposed measures and assessment of corresponding impacts

Relevant waste management activity	Substance of concern	Proposed measure	Econ. Impact	E&H impact	WM impact
Recycling of PUR foams from ELVs possibly containing C-PentaBDE	TetraBDE PentaBDE HexaBDE	<ul style="list-style-type: none"> LPCL 10 ppm each congener group Separation and destruction of PUR foams from cars built in 2000 and before 	Low	Positive	Low
Waste management of upholstery possibly containing C-PentaBDE	TetraBDE PentaBDE HexaBDE	<ul style="list-style-type: none"> LPCL 10 ppm each congener group It is not considered feasible to separate upholstery containing C-PentaBDE from bulky waste (mixed waste contamination level below 10 ppm) No action required. Incineration with energy recovery is regarded the preferred waste management option 	No impact	No impact	No impact
Recycling of WEEE plastics possibly containing C-OctaBDE	PentaBDE HexaBDE HeptaBDE	<ul style="list-style-type: none"> LPCL 10 ppm each congener group Separation and destruction of ABS plastics from cathode ray computer and TV monitors (mainly WEEE cat. 3 and 4) produced in 2003 and before 	No impact	No impact	No impact

		(required anyway by the WEEE Directive)			
Recycling of carpets possibly containing PFOS	PFOS	<ul style="list-style-type: none"> • LPCL 10 ppm • Separation from bulky waste and destruction by incineration 	Moderate	Positive	Low
Waste management of textiles possibly containing PFOS	PFOS	<ul style="list-style-type: none"> • LPCL 10 ppm • It is not considered feasible to separate PFOS containing textiles from the MSW stream (mixed waste contamination level below 10 ppm) • No action required. Incineration with energy recovery is regarded the preferred waste management option 	No impact	No impact	No impact
Treatment of PeCB containing waste	PeCB	<ul style="list-style-type: none"> • LPCL 50 ppm • No waste with concentration level above LPCL 	No impact	No impact	No impact
Management of conveyor belts possibly containing SCCP	SCCPs	<ul style="list-style-type: none"> • LPCL 1,000 ppm • Separation of conveyor belts for underground mining and destruction by incineration 	Moderate	Positive	Low
Management of adhesives and sealants possibly containing SCCP	SCCPs	<ul style="list-style-type: none"> • LPCL 1,000 ppm • Adhesives and sealants • An improved degree of separation could be achieved by implementing deconstruction/renovation management plans 	Moderate to high	Positive	Moderate to high
Recycling of EPS/XPS foams possibly containing HBCD	HBCD	<ul style="list-style-type: none"> • LPCL 100 ppm • Separation and destruction of EPS/XPS insulation boards as far as reasonably possible • An improved degree of separation could be achieved by implementing deconstruction/renovation management plans 	Moderate	Positive	Moderate
Waste treatment of textiles	PCP	<ul style="list-style-type: none"> • LPCL 1,000 • It is not considered feasible to separate PCP containing textiles from the MSW stream (mixed waste contamination level below 10 ppm) 	No impact	No impact	No impact
Treatment of HCBd containing waste	HCBd	<ul style="list-style-type: none"> • LPCL 100 ppm • No data available • No action proposed 	No impact	No impact	No impact
Treatment of PCN containing waste	PCN	<ul style="list-style-type: none"> • LPCL 10 ppm • No waste with concentration 	No impact	No impact	No impact

		level above LPCL			
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10.5 Impacts of proposed limit values on current recycling activities

Recycling activities which could be concerned by the proposed limit values are particularly the following:

- ELVs recycling (PBDEs)
- WEEE plastics recycling (PBDEs)
- Recycling of carpets (PFOS)
- Recycling of EPS/XPS foams (HBCD)

The impacts of the proposed LPCLs (LPCL 1) on these activities have been evaluated with the conclusion that they are related to low to moderate economic and waste management impacts. The implementation of the LPCLs is related to specific measures which are generally related to positive environmental and health effects due to the fact that increasing shares of new or candidate POPs will be destroyed or irreversibly transformed. An overview assessment on the impacts and the compatibility with other product and waste legislation is given in the following table.

Table 10-13: Impact of proposed LPCLs and related measures on current recycling activity

Proposal →	Activity: ELV recycling LPCL for each of the relevant PBDE congener groups at 10 ppm and separation and destruction of PUR foams from cars built in the year 2000 or before if the PUR foam is intended to be recycled or to be landfilled in the corresponding process.	
Substance →	TetraBDE, PentaBDE, HexaBDE, (HeptaBDE)	
Impact	Assessment	Justification
Economic impact	Low	<ul style="list-style-type: none"> • Only a limited number of cars would be concerned • PUR foams from the majority of cars could be recycled • Costs for the option of manual separation could be supported. • The average costs per car about 13€ can be considered very low compared to other car related costs • The annual cost impact could be considered relevant in the beginning (up to 63 million € in 2012) but will become low in 2016 (up to 23 million €) and insignificant after 2020.
Environmental/Health impact	Positive	<ul style="list-style-type: none"> • The POPs would be destroyed and thus withdrawn from the eco-cycle. Hence it can be assured that they will not become subject to LRT and bioaccumulate and cause adverse environmental and health effects.
Impact on waste management	Low	<ul style="list-style-type: none"> • Easy identification and separation of the relevant PUR foams. • Usually no analytical measurement required.
Compatibility with REACH	✓	<ul style="list-style-type: none"> • LPCL is below the 0.1% concentration limit
Compatibility with WEEE	n.r.	<ul style="list-style-type: none"> • n.r.
Compatibility with ELV:	✓	<ul style="list-style-type: none"> • Negligible impact on the reuse and recycling quotient of ELVs;

		The impact would decrease and phase out over time.
Compatibility with ROHS	n.r.	<ul style="list-style-type: none"> LPCL is below the 0.1% concentration limit
Proposal →	Activity: WEEE plastics recycling LPCL for each of the relevant PBDE congener groups at 10 ppm and separation and destruction of ABS plastics from cathode ray computer and TV monitors produced in 2003 and before if the ABS plastic is intended to be recycled or to be landfilled in the corresponding process	
Substance →	PentaBDE, HexaBDE, HeptaBDE	
Impact	Assessment	Justification
Economic impact	Low	<ul style="list-style-type: none"> Only a limited number of EEE products to be separated Moderately increasing operational costs are an impact of the WEEE directive obligation to separate BFR containing plastics and not of the LPCL set in the POP Regulation
Environmental/Health impact	Positive	<ul style="list-style-type: none"> The POPs would be destroyed and thus withdrawn from the eco-cycle. Hence it can be assured that they will not become subject to LRT and bioaccumulate and cause adverse environmental and health effects.
Impact on waste management	Low	<ul style="list-style-type: none"> Easy identification and separation of the relevant EEE products No analytical measurement required in addition to obligations from WEEE
Compatibility with REACH	✓	<ul style="list-style-type: none"> LPCL is below the 0.1% concentration limit
Compatibility with WEEE	✓	<ul style="list-style-type: none"> Plastic containing BFRs have to be separated anyway due to the WEEE directive
Compatibility with ELV:	n.r.	<ul style="list-style-type: none"> n.r.
Compatibility with ROHS	✓	<ul style="list-style-type: none"> LPCL is below the 0.1% concentration limit
Proposal →	Activity: Recycling of EPS/XPS foams LPCL for HBCD at 100 ppm and attempt to separate and destroy HBCD containing PS insulation boards as far as reasonable possible from non contaminated products; non contaminated products would be available for recycling.	
Substance →	HBCD	
Impact	Assessment	Justification
Economic impact	Moderate	<ul style="list-style-type: none"> non contaminated products remain available for recycling; transport distances for the recycled fraction will increase; certain effort for separation required; the share of non-contaminated products will increase in the future and will facilitate future recycling
Environmental impact	Positive	<ul style="list-style-type: none"> HBCD would be destroyed and thus withdrawn from the eco-cycle. Hence it can be guaranteed that they will not become subject to LRT and bioaccumulate and cause adverse environmental and health effects.
Impact on waste management	Low	<ul style="list-style-type: none"> The increasing quantities of EPS and XPS waste from construction will require appropriate waste management options anyway. The identification and separation of the relevant PS insulation boards appears to be feasible. Analytical measurement only required in specific cases.
Compatibility with REACH	✓	<ul style="list-style-type: none"> LPCL is below the 0.1% concentration limit

Compatibility with WEEE	n.r.	• n.r.
Compatibility with ELV:	n.r.	• n.r.
Compatibility with ROHS	n.r.	• n.r.
Proposal →	Activity : recycling of carpets LPCL for PFOS at 10 ppm, separation from bulky waste and destruction	
Substance →	PFOS	
Impact	Assessment	Justification
Economic impact	Low	<ul style="list-style-type: none"> • There is no structural carpet recycling system in Europe, recycling is done punctually and the majority of the carpets end up as bulky waste; • The large quantity would result in high incineration costs which cannot be landfilled anymore; • the share of non-contaminated products will increase in the future and will facilitate future recycling
Environmental impact	Positive	<ul style="list-style-type: none"> • Carpets have a share of 89% of all PFOS in waste. • PFOS would be destroyed and thus withdrawn from the eco-cycle. Hence it can be guaranteed that they will not become subject to LRT and bioaccumulate and cause adverse environmental and health effects • Energy recovery
Impact on waste management	Low	<ul style="list-style-type: none"> • In some member states incineration is already typical to some extent.
Compatibility with REACH	✓	<ul style="list-style-type: none"> • LPCL is below the 0.1% concentration limit
Compatibility with ROHS	n.r.	• n.r.

10.6 Possible impacts on product and waste regulation

The proposed LPCLs are compatible with existing EU product and waste legislation. There may be a need to adjust the legislation on shipments of waste.

The **REACH regulation** contains several concentration limits regarding the placing on the market or other obligations for specific new and candidate POPs (PentaBDE, PFOS, HBCD, PCP, SCCPs). The relevant concentration limits are 0.1% (1,000 ppm) or 1% (for SCCPs). LPCLs in the POP Regulation should be below these concentration limits because otherwise it would be possible that materials resulting from recycled waste streams would contain the relevant substances above the concentration limits established in REACH. As all LPCLs proposed are below the concentration limits established in the REACH Regulation, there are no impacts on REACH expected.

The **ROHS Directive** contains a ban of placing on the EU market of new EEE if containing PBDE in homogeneous materials at a concentration limit of 0.1% by weight (1,000 ppm). The corresponding LPCLs in the POP Regulation should be below this concentration limit, because otherwise it would be possible that materials resulting from recycled waste streams would contain the relevant PBDEs above the concentration limit established in REACH. As all LPCLs proposed are below the concentration limits established in the ROHS Directive, there are no impacts expected.

The **WEEE Directive** aims at preventing WEEE by promoting reuse, recycling and recovery. For the ten WEEE categories defined in Annex IA, the Directive sets targets for recovery and for component, material and substance reuse and recycling. As regards new POPs (PBDEs contained in C-octaBDE used for ABS plastics in the past) the targets set for categories 3 and 4 are of relevance, requesting (1) a recovery rate of a minimum of 75 % by an average weight per appliance, and (2) a reuse and recycling rate for component, material and substance of a minimum of 65 % by average weight per appliance. The plastic fraction in WEEE ranges from 10-40%, and its recycling will not be allowed anymore for a WEEE material classified as POP waste.

According to Annex II of the WEEE Directive, "Selective treatment for materials and components of waste electrical and electronic equipment in accordance with Article 6(1)" as a minimum [...] [e.g. plastic containing brominated flame retardants] has to be removed from any separately collected WEEE.

According to the POP regulation, plastic waste containing PBDEs above the proposed LPCLs has to be treated as POPs waste. The POP content has to be destroyed or irreversibly transformed.

A conflict between the POP regulation and the WEEE directive could arise if the Member States would interpret the obligation to remove plastic containing brominated flame retardants from any separately collected WEEE above a congener group specific limit of 1,000 ppm because otherwise the EEE produced from recycled material could contain the relevant congeners above that limit.

The requirement of Annex II of the WEEE Directive to remove BFR containing plastics from any separately collected WEEEs already avoids such a contamination and allows setting the LPCL below 1,000 ppm without an impact on the WEEE Directive.

The relevant congeners are particularly contained in old EEE (mainly produced in the 1990ies until 2003). Manual separation of the plastics from old cathode ray computer and TV monitors seems to be one feasible option by which the bulk of the relevant PBDEs could be separated from the WEEE waste stream. The targets for reuse and recycling according to the WEEE Directive would be achievable.

The **ELV Directive** sets out provisions with respect to end-of-life vehicles aiming at the prevention of waste from vehicles and, in addition, at the reuse, recycling and other forms of recovery of end-of life vehicles and their components. As regards recycling and recovery specific and ambitious targets are set pursuant to article 7 since 2006 which will be further increased from 2015 onwards. The relevant material in the context of the present project (PUR foam possibly containing C-PentaBDE) contributes approximately 0.5% to the weight of a car. Only a certain share thereof PUR foam is prone for C-PentaBDE contamination (mainly PUR foam in cars produced until the year 2000). In order to not interfere with recycling targets it thus is recommended to separate PUR foam from other plastic parts. Otherwise thermal treatment (energy recovery) of the whole fibre fraction (approximately 5% of the vehicle weight) of the shredder light fraction could be the preferred treatment option. Given the further decline of PBDE contamination in PUR foam, the recycling/reuse of plastic fractions would

not cause a relevant problem in 2015. Correspondingly it is expected that the recycling targets of the ELV Directive will continue to be achievable.

LPCLs do not have a direct impact on the Regulation (EC) No 1013/2006 on **shipments of waste** as shipping conditions are exclusively related to the Annexes of that Regulation and the European Waste List. A majority of the waste contaminated with new POPs and candidate POPs are already classified as hazardous in the current annexes. However, according to the Basel Convention technical guidelines, POP wastes have to be managed as hazardous wastes. This will have several consequences including the export ban of hazardous wastes according to Article 36 of Regulation 1013/2006. Also green listed wastes (e.g. PUR foams, upholstery) intended for recovery in third countries could be regulated via article 36 provisions if the third country authorities become active and ban the import. On the other hand, it might need to be discussed and decided upon whether or not it would be necessary from the point of view of knowledge development and the minimisation objective of the Stockholm Convention, to review the annexes to the Regulation and to reclassify, add or better specify limits in wastes for one or the other of the new and candidate POPs. Furthermore additional guidance on distinction between waste and products and on classification issues might be needed in the light of new POPs in the annexes, and taking into consideration the LPCLs in order to prevent large scale exports from the EU, as soon as obligations for POP waste enter into force.

The LPCL values for PFOS and PeCB in the POP Regulation may not be not sufficiently qualified concerning sewage sludge and it could therefore be discussed to supplemented them by more stringent and specific limit values e.g. in the Regulation on fertilizers 2003/2003/EC or the Directive on sewage sludge 86/278/EEC.

10.7 Possible implications of concentration limits for PBDEs in new articles produced from recycled materials above which the articles cannot be placed on the market

According to Annex I of the POP regulation PBDE containing materials can be used for articles and preparations containing concentrations below 0.1% of the relevant congener groups when produced partly or fully from recycled materials or materials from waste prepared for re-use.

If a POP containing material is intended for recycling, there are generally two options to handle the POP content in the corresponding process:

- (1) Separation of the POP containing articles or parts from non-contaminated components
- (2) Separation of the POP content from the contaminated material

Both options enable the reduction of the POP content in a specific fraction of the POP containing material. Corresponding processes can be applied in recycling activities in order to reduce the POP content of specific material fractions.

In the European situation it is expected that the PBDE content in relevant waste streams can be reduced by option (1) to levels below the LPCL proposed for the relevant individual PBDE congeners

(i.e. to below 10 ppm per congener group). As a consequence the remaining waste can be treated according to the European waste legislation which enables among other material recycling.

Considering option (1), the concentration limits for PBDEs in new articles produced from recycled materials in Annex I could be lowered correspondingly. The real contamination levels in the waste stream should be evaluated prior to lowering the concentration limits for PBDEs in new articles produced from recycled materials in Annex I.

As regards option (2), physico-chemical treatment processes are currently being developed and are generally appropriate to reduce the POP content for each listed PBDE congener group below the 0.1% limit for PBDEs in new articles produced from recycled materials. The separated POP content could be destroyed or irreversibly transformed according to the requirements of the POP regulation.

Considering option (2) it could therefore be rational to maintain the 0.1% limit for PBDEs in new articles produced from recycled materials. However, it is not expected that in the European Situation corresponding physico-chemical treatment processes will play a role in the case of the relevant PBDEs. The European situation is characterised by limited quantities of PBDEs in waste streams with a decreasing trend. It is expected that by 2020 no relevant quantities of the listed PBDEs will occur in waste streams. Against this background it is not expected that corresponding physico-chemical treatment processes will become competitive at industrial scale.

In the opinion of the project team it is therefore not necessary to maintain the 0.1% concentration limit for PBDEs in new articles produced from recycled materials in Annex I of the POP regulation. It could be lowered to the level of the LPCL. Real contamination levels in the relevant waste streams should be evaluated prior to lowering the concentration limits for PBDEs in new articles produced from recycled materials in Annex I.

10.8 Measurement Methods

The report contains a compilation of analytical measurement methods for the detection of new and candidate POP concentrations in different relevant matrices. The methods are compiled and discussed separately for each substance. Measurement methods are available for all relevant substances that enable the detection of the new and candidate POPs at levels usually well below the proposed limit values. The compilation comprises an evaluation of available screening methods. Details on available measurement methods can be found in the corresponding chapter of the report.

11 General References

Please note that specific reference lists are available in the following subchapters:

PentaBDE, see subchapter 6.1.11

OctaBDE, see subchapter 6.2.10

PFOS, see subchapter 6.3.14

PeCB, see subchapter 6.4.14

SCCP, see subchapter 6.5.14

HBCD, see subchapter 6.6.13

PCP, see subchapter 6.7.6

HCBD, see subchapter 6.8.11

PCN, see subchapter 6.9.16

- **[EC 2000]**
European Commission, Communication from the Commission on the precautionary principle Brussels, 02.02.2000, COM(2000) 1
- **[EC 2006]**
European Commission, DG Environment, *Frequently Asked Questions on Directive 2002/95/EC on the Restriction of the Use of certain Hazardous Substances in Electrical and Electronic Equipment (RoHS) and Directive 2002/96/EC on Waste Electrical and Electronic Equipment (WEEE)*, EC, DG Environment, May 2005, update August 2006 (http://ec.europa.eu/environment/waste/weee/pdf/faq_weee.pdf)
- **[EPA IRE 2011a]**
Personal communication Irish EPA e-mail from 22.02.2011 related to sampling and analysis of identified wastes for newly listed POPs
- **[EPA IRE 2011b]**
Personal communication Irish EPA e-mail from 23.03.2011 related to sampling and analysis of identified wastes for newly listed POPs
- **[UNEP 2002]**
UNITED NATIONS ENVIRONMENT PROGRAMME, CHEMICALS, *Regionally Based Assessment of Persistent Toxic Substances*, December 2002
- **[UNEP 2010]**
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pollutants as adopted by the Conference of the Parties, October 2010

- **[WFD 2004 PCP screening]**

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- **[US EPA 1984]**

L.A. Stretz, J.s. Vavruska, Controlled Air Incineration of PCP Treated Wood, Project Summary, US EPA, July 1984

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- **[BC TG 2007]**

Basel Convention, Updated general technical guidelines for the environmentally sound management of wastes consisting of, containing or contaminated with persistent organic pollutants (POPs), (K0760113 180607), 2007

- **[UBA DE 2000]**

Umweltbundesamt Deutschland: Erarbeitung von Bewertungsgrundlagen zur Substitution umweltrelevanter Flammenschutzmittel, Band II: Flammhemmende Ausrüstung ausgewählter Produkte - anwendungsbezogene Betrachtung: Stand der Technik, Trend, Alternativen von André Leisewitz und Winfried Schwarz, Im Auftrag des Umweltbundesamtes, Dezember 2000

- **[ITVA 2010]**

ITVA – Ingeniertechnischer Verband für Altlastenmanagement und Flächenrecycling e.V., Innovative In-situ-Sanierungsverfahren, Arbeitshilfe – H1 – 13, Juni 2010

12 Annex

List of Annexes:

- Annex 1. Questionnaire**
- Annex 2. Questionnaire evaluation**
- Annex 3. Evaluation Questionnaire for submission of information on New POPs in accordance with SC-4/19**
- Annex 4. Physico-chemical and toxicological properties of POPs**
- Annex 5. Overview of number and capacity of treatment facilities in 2004 and 2006 at country level**

12.1 Annex 1: Questionnaire

QUESTIONNAIRE

“Study on waste related issues of newly listed POPs and candidate POPs”

1) Aim of the questionnaire

The questionnaire is aimed at gathering information on waste related issues of the new POPs. This information will be very valuable in order to assess the current situation and will constitute the basis for policy measures. To this end a questionnaire has been prepared by BiPRO⁷⁹ in close coordination with the Commission Services.

2) Background information

The European Community has ratified both the Protocol to the regional UNECE Convention on Long Range Transboundary Air Pollution (CLRTAP) and the Stockholm Convention on persistent organic pollutants (POPs) aimed at reducing and eliminating production, use and releases of these substances.

The obligations deriving from the Protocol and the Convention have been implemented in Union law by Regulation (EC) No 850/2004 (POP Regulation) of the European Parliament and of the Council. The POP Regulation includes concentration limits for 14 POPs substances and substance classes, above which the POPs content in waste shall be subject to destruction or irreversible transformation (low POP content), as well as threshold limits (upper POP content) above which no derogation from the destruction obligation can be granted.

In 2009 new substances have been listed by decision of the fourth conference of the Parties to the Stockholm Convention (COP-4). Therefore, the POP Regulation has to be amended accordingly, adding the new POP substances to the annexes and setting concentration limits for these substances (so called “new POPs”).

In addition, SCCPs were listed in the UNECE POP Protocol in December 2009 and other substances are in consideration for listing in the Stockholm Convention and the UNECE POP Protocol (so called “candidate POPs”).

In order to set limit values for newly listed POPs and prepare limit values also for the candidate POPs, the Commission has launched a study to:

- propose limit values for tetrabromodiphenyl ether, pentabromodiphenyl ether, hexabromodiphenyl ether and heptabromodiphenyl ether (PBDE), perfluorooctane sulfonic acid and its derivatives (PFOS)
- verify whether the proposed limit value of 50 mg/kg for pentachlorobenzene (PeCB) is appropriate

⁷⁹ BiPRO GmbH, Munich, Germany (www.bipro.de), has been contracted by the European Commission to carry out the „Study on waste related issues of newly listed POPs and candidate POPs”

- propose limit values for short chain chlorinated paraffin (SCCP), hexabromocyclododecane (HBCD), pentachlorophenol (PCP), hexachlorobutadien, and polychlorinated naphthalene (PCN)

4) Instructions on using the questionnaire

Please do note that most of the questions raised in the questionnaire have already been addressed in **a questionnaire of the Secretariat to the Stockholm Convention** (“Questionnaire for submission of information on New POPs in accordance with SC-4/19”, in **the following designated as “Questionnaire SC-4/19”**), which was sent to all relevant parties and observers on 20 November 2009. Several respondents have already submitted their interim answers. Please find a copy of the Questionnaire SC-4/19 attached.

The questionnaire below will therefore exclusively focus on questions/topics which have **NOT** already been (fully) covered within the scope of the “Questionnaire SC-4/19”.

In order to minimise the reporting burden, please fill in:

- 1) The “Questionnaire SC-4/19” (see attachment). In case you have already filled in this questionnaire you can simply provide us the completed questionnaire SC-4/19
- 2) The questions and topics addressed in the questionnaire below

Any additional information that may be useful for the above mentioned study would be appreciated.

The 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**. Most of these are text fields where written input is required.

The questionnaire is structured in two main sections. Please only fill in the sections that are relevant for you.

- Section I – “new POPs”
- Section II – “candidate POPs”

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:

Contact person	E-mail address	Telephone
Mr Alexander Potrykus	alexander.potrykus@bipro.de	+49 89 18979050
Ms Anke Joas	anke.joas@bipro.de	
Mr Milos Milunov	milos.milunov@bipro.de	

Mr Peter Hofbauer	peter.hofbauer@bipro.de	
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We intend to contact you in order to discuss open questions or specific topics after having received your completed questionnaire. You may also indicate in the questionnaire if you would like to discuss a certain point with us.

Should you have additional background information in form of documents, reports, data sets or as links to websites, please do not hesitate to send us this information, preferably electronically.

Information provided will be used in a transparent and traceable way for the “Study on waste related issues of POPs”. If you would like to submit information on a confidential basis, please indicate this in your reply to the questionnaire. Confidential information will only be used in anonymous or aggregated form. The information provided will also be transmitted to the secretariat of the Stockholm Convention. If you do not agree, please indicate this in your reply to the questionnaire.

Please return the completed questionnaire SC-4/19, the completed questions below and relevant document files

to alexander.potrykus@bipro.de before 20 July 2010.

If you are interested to discuss with us by telephone, please let us know when we could possibly reach you by telephone.

Please provide your name and contact data:

Name of Institution:

Country:

City/ CIP Code:

Street:

Competent contact person:

E-mail:

Phone:

Remark:

Section I – “new POPs”

Tetra- and pentabromodiphenyl ether (commercial PentaBDE); Hepta- and hexabromodiphenyl ether (commercial OctaBDE); Perfluorooctane sulfonic acid and its derivatives (PFOS); Pentachlorobenzene (PeCB)

I. Recycling Operations/Waste Management Options PFOS and PBDE

- 1. Is (A) a transitional period needed, during which the recycling of PFOS and/or PBDE containing waste shall exceptionally be allowed or should (B) an effort be made to separate waste contaminated with PFOS and/or PBDE from non-contaminated waste as soon as possible to avoid dilution of POPs into all materials?**

If “(A)”, recycling should be exceptionally allowed, please continue with question 2
If “(B)”, PFOS and/or PBDE containing waste should be immediately separated, please continue with question 3

If possible, please suggest any alternative measures which might also be considered:

Suggestion for alternative measures:

- 2. If recycling should be exceptionally allowed (A)**
 a) *Please propose and justify the length of the exemption period*
 b) **Which recycling operations could be allowed, which maximum POP concentration would you recommend for each operation, what would be the related costs and why should the recycling operation be allowed?**

a)

b)

Recycling Operation/Waste Management Option	Relevant waste/new POP	Proposed maximum POP concentration	Treatment costs per ton of waste	Explanation
1.				
2.				
3.				

- 3. If PFOS and/or PBDE containing waste should be immediately separated (B)**
 a) *How could it be made possible to **distinguish contaminated from non contaminated waste**?*
 b) **Which operations should be used in practice to separate contaminated from non-contaminated waste,**
 c) *What should be the **preferred option for handling the contaminated waste fraction**, what would be the related **costs** and **why should the option be used** (justification)? Please add additional rows if necessary.*

a)

b)

Separation Operation	Relevant waste/new POP	Health risks for workers recycling facilities	Separation Costs /ton of waste	Explanation
1.				
2.				

3.				
----	--	--	--	--

c)			
Waste Management Operation	Relevant waste/new PO	Management Costs /ton of was	Explanation
1.			
2.			
3.			

II. Possible implications of concentration limits for PFOS and PBDE in new articles produced from recycled materials

4. What would in your opinion be the **major impacts from a reduction of existing limit values in new articles from recycled materials?**

III. Concentration Limits for new PFOS and PBDE

5. a) Are you aware of **any existing concentration limits for the PFOS and PBDE in waste?** (please list limits individually)

b) Are you aware of **any other existing concentration limits** applied in your country (e.g. in articles, soil, water, sediment)

a)

b)

6. Which lower **concentration limits for the PBDE and PFOS in waste** would you recommend (above which the POPs content shall be destroyed or irreversibly transformed)?

7. At which lower concentration limits for PBDEs and PFOS in waste would you expect relevant impacts (for recycling industry)? Please justify your answer.

8. If appropriate, please **propose a staggered approach** with decreasing concentration limits over time as the amount and concentrations of POPs in wastes decrease, and justify.

IV. Occurrence of pentachlorobenzene (PeCB)

9. a) Please indicate **whether your country has stockpiles of PeCB** (e.g. as impurity in relevant pesticides, PCP treated wood, PCB containing liquids, PER, etc.).
b) **If so**, please provide information on condition, quantity, concentrations, etc.

a)

b)

10. a) Please indicate **whether your country has sites contaminated with PeCB**.

b) **If so**, please provide information on the methods used for the remediation of these sites.

a)

b)

11. Please indicate the **occurrence and concentration of PeCB**, in different waste categories, recycled articles and articles in use. Please add additional rows if necessary.

Pentachlorobenzene (PeCB)	Specification of waste/article	Concentration [mg/kg]
Wastes (e.g. treated wood, quitozene stocks)		
Recycled articles		
Articles in use (e.g. PER)		
other		

V. Concentration limits for pentachlorobenzene (PeCB)

12. a) Are you aware of any **existing or proposed concentration limits for PeCB**?
b) **If so**, please provide specific information on the concentration limits.

a)

b)	Specification of waste/article	Existing concentration limit	Proposed concentration limit
Wastes			
Recycled articles			
Articles in use			
other			

13. Which lower concentration limits for PeCB in waste would you recommend (above which the POPs content shall be destroyed or irreversibly transformed)?

14. At which lower concentration limits for PeCB in waste would you expect relevant economic impacts? Please justify your answer.

15. If appropriate, please propose a staggered approach with decreasing concentration limits over time as the amount and concentrations of PeCB in wastes decrease, and justify.

Section II – “candidate POPs”

Short chain chlorinated paraffins (SCCP), Hexabromocyclododecane (HBCD), Pentachlorophenol (PCP), Hexachlorbutadien (HCBD) and Polychlorinated naphthalenes (PCN)

I. Occurrence and levels of “candidate POPs”

16. a) Please indicate whether your country has stockpiles of “candidate POPs” listed?

b) If yes, please provide information on condition, size, etc.

a)

b)

17. a) Please indicate *whether your country has sites contaminated* with any of the chemicals listed?

b) If *yes*, please provide information on the methods used for the remediation of these sites.

a)

b)

18. Please indicate the *occurrence and concentration of the “candidate POPs”*, in different waste categories, recycled articles and articles in use. Please add additional rows if necessary.

a) Short chain chlorinated paraffins (SCCP)	Specification of waste/article	Concentration [mg/kg]
Wastes		
Recycled articles		
Articles in use		

b) Hexabromocyclododecane (HBCD)	Specification of waste/article	Concentration [mg/kg]
Wastes		
Recycled articles		
Articles in use		

c) Pentachlorophenol (PCP)	Specification of waste/article	Concentration [mg/kg]
Wastes		
Recycled articles		
Articles in use		

d) Hexachlorbutadien (HCBD)	Specification of waste/article	Concentration [mg/kg]
Wastes		
Recycled articles		
Articles in use		

e) Polychlorinated naphthalenes (PCN)	Specification of waste/article	Concentration [mg/kg]
Wastes		
Recycled articles		
Articles in use		

19. If possible, please provide information on wastes containing “candidate POPs” that are currently recycled in your country (corresponding to the question above). Please specify the types of new articles produced from the recycled material?

“candidate POPs”	Types of waste currently recycled (please list wastes individually with recycling rates)	Recycling rate [%]	New articles produced from recycled material
Short chain chlorinated paraffins (SCCP)			
Hexabromocyclododecane (HBCD)			
Pentachlorophenol (PCP)			
Hexachlorbutadien (HCBD)			
Polychlorinated naphthalenes (PCN)			

20. Please list measurement methods for identifying the presence and levels of the “candidate POPs” in waste (if not existing, others that you use or about which you know).

II. Recycling Operations/Waste Management Options

21. Is (A) a transitional period needed, during which the recycling of waste containing candidate POPs shall exceptionally be allowed or should (B) an effort be made to separate waste contaminated with candidate POPs from non-contaminated waste as soon as possible to avoid dilution of POPs into all materials?

If “(A)”, recycling should be exceptionally allowed, please propose and justify the length of the exemption period. Which recycling operations could be allowed, which maximum POP concentration would you recommend for each operation?

If “(B)”, waste containing candidate POPs should be immediately separated,

If possible, please suggest any alternative measures which might also be considered:

Suggestion for alternative measures

22. If recycling should be exceptionally allowed (A)

a) What would be an appropriate exemption period?

b) What would be the related **costs** and **why should the recycling operation be allowed?** Please justify your answer and add additional rows if necessary.

a)

b)

Recycling Operation/Waste Management Option	Relevant waste/candidate POP	Proposed maximum POP concentration	Treatment costs per ton of waste	Justification
1.				
2.				
3.				

23. If waste containing candidate POPs should be immediately separated (B)

a) How would it be possible to distinguish contaminated from non-contaminated waste

b) Which operations should be used in practice to separate contaminated from non-contaminated waste (justification)?

c) What would be the preferred option of handling the contaminated waste fraction, what would be the related costs and why should the option be used (justification)? Please add additional rows if necessary.

a)

b)

Separation Operation	Relevant waste/candidate POP	Health risks for workers recycling facilities	Separation Costs /ton of waste	Justification
1.				
2.				
3.				

c)			
Waste Management Operation	Relevant waste/candidate POP	Management Costs /ton of waste	Justification
1.			
2.			
3.			

III. Concentration Limits

24. a) Are you aware of any **existing concentration limits for the “candidate POPs” in waste?** (please list limits individually)

b) Please indicate **other existing limits applied in your country** (e.g. articles, soil, water, sediment)

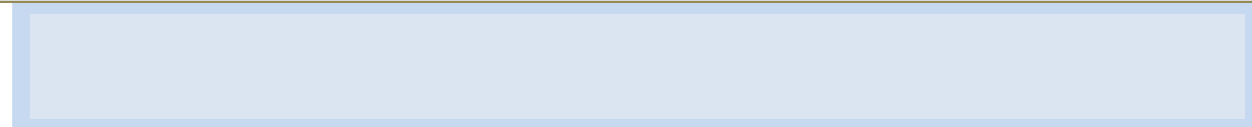
a)

b)

25. What is in your opinion an **appropriate concentration limits for “candidate POPs” in waste**, above which the POPs content shall be destroyed or irreversibly transformed? Please justify your opinion?

26. At **which lower concentration limits for candidate POPs in waste** would you expect **significant economic impacts**? Please justify your answer.

27. Which *operations* (apart of the ones listed in Annex V, part 1) could be accepted *for destruction and irreversible transformation of the “candidate POP” content*?



12.2 Annex 2: Evaluation Questionnaire

Table 12-1: Overview of information submitted by stakeholders on commercial PentaBDE, OctaBDE, PFOS, its salts and PFOSF

Country/ Respondent	Date of Submission	Institution/ Contact Person	Information related to PentaBDE, OctaBDE, PFOS, its salts and PFOSF
Austria	August 2010	Federal Ministry of Agriculture, Forestry, Environment and Water Management, Mrs. LOEW, Mrs. BUSIC, Mr. MOSER – Waste Management Mrs. SCHROTT; Mrs. Eberhartinger- Tafill - Chemicals	<p>Recycling Operations/Waste Management Options PFOS and PBDE</p> <p>Austria in general support the separation of PBDE or PFOS containing waste, however, problems with the fulfilment of the quotas laid down in the WEEE Directive are faced due to the fact that plastic wastes with PBDEs >10ppm are not allowed to be recycled any more in the future (see Amendment of the POP Regulation), a transitional period should be envisaged for the electronics sector only, allowing the limit of the ROHS-Directive (1000 ppm for the sum of Penta-, Octa and DecaBDE).</p> <p>Separation of wastes containing PFOS or PBDE should take place in any case. The separation of PBDE containing plastics has been laid down in Annex II of the WEEE Directive No. 2002/96 as amended, the national WEEE-Regulation and the Austrian Waste Treatment Ordinance. However it has not been laid down that the separation must take place at the step of the WEEE dismantler, so it may take place later on e.g. by means of NIR or density separation of the plastic shredder fraction, when the de-contaminated WEEE has been subject to a shredder process.</p> <p>The Austrian Waste Treatment Ordinance (Section 13 Para 2) allows recycling of plastics and wood containing all types of halogenated impregnation and flame retardants only in those cases when these substances have to be added to the new product due to technical requirements (more stringent approach than in the EU). However by August 26th 2010, when the Amendment of the POP Regulation will enter into force, the use of recycled materials containing more than 10ppm PBDEs for other applications than the electronics sector will be prohibited.</p> <p>Austrian Federal Ministry supports the effort to prohibit recycling operations for wastes exceeding 10ppm PBDE and PFOS. With regard to PBDE, only on a transitional basis an exemption for recycling of polymers (plastics) containing PBDEs in the electric/electronic sector should be maintained.</p> <p>When fixing a more stringent limit value (10 ppm) for admissible recycling of polymers containing PBDE this fact would have huge implications on the compliance with the present recycling quota laid down in the WEEE Directive and the EOL-Vehicles Directive. Therefore further considerations at EU-level seem to be necessary. The recycling quotas for the different WEEE-categories in the WEEE-Directive and the recycling quotas in the EOL-Vehicles Directive should be revised.</p> <p>An Amendment of the Waste Shipment Regulation would also be required, clearly prohibiting the shipment/export of POP-wastes exceeding the limit of 10 ppm for recycling in third states (transitional period for 1000ppm of PBDEs for recycling in the electronic sector ought to be fixed).</p> <p>In an amendment of the European Waste List it must be regulated whether POP-wastes are automatically hazardous wastes.</p> <p>A transition period (3 years) seems to be necessary for amendments of the above mentioned EC-Directives and for adjustment of technological processes for the implementation of any more stringent limit for recycling of polymers from WEEE (PBDEs) and all other polymers e.g. plastics from EOL Vehicles (PBDEs and PFOS).</p> <p>Furthermore, comprehensive answers were provided with regard to separation techniques and options for handling the contaminated waste streams.</p> <p>Possible implications of concentration limits for PFOS and PBDE in new articles produced from recycled material</p> <p>According to Austria implications on the quotas laid down at EU level in the WEEE Directive and the EOL-Directive are expected; in any case</p>

Country/ Respondent	Date of Submission	Institution/ Contact Person	Information related to PentaBDE, OctaBDE, PFOS, its salts and PFOSF
			<p>at EU-level a revision/amendment of these quotas has to be considered when reducing existing limit values for POPs in new articles from recycled materials. In parallel exports of wastes containing higher amounts of POPs from the EU to third states for recycling should be prohibited in any case, even if the recycling were in compliance with national legislation in the importing country.</p> <p>Concentration Limits for PFOS and PBDEs</p> <p>a) For the determination of the hazardous criterion H13 in wastes an AOX-limit of 100mg/kg d.s. in the LEACHATE (S:L= 1:10, therefore 10mg/l for liquids) has been laid down in Austria.</p> <p>b) 1) ROHS: PBDEs: Penta-, Octa and DecaBDE in sum in total 1000 ppm (=0,1%); the limit is referred to the pure substance (information of the COM).</p> <p>2) REACH: (0,1%) 1000 ppm PentaBDE and (0,1%) 1000 ppm OctaBDE (referred to the technical mixture) – overruled by the Amendment of the POP Regulation. In any case a harmonization of the calculation of the PBDE-limits shall be envisaged in the Amendments of the ROHS and the POP Regulation, either referring to the technical mixture of the relevant PBDE or the pure substance. Furthermore it shall be specified whether the limit refers to the sum of the PBDEs or the relevant prohibited congeners. In the future, a limit for DecaBDE (see limitation in the ROHS Regulation) should also be laid down in an Amendment of the POP Regulation. Specific calculation methods and analytical standard methods shall be adopted at EU level for legal certainty.</p> <p>3) Limit for fertilizers (Austrian Fertilizers Act 2010): 100 microgram PFOS+PFOA /kg = 0,1 mg/kg = 0,1 ppm</p> <p>4) WATER: DIRECTIVE 2008/105/EC on environmental quality standards in the field of water policy -limit for PBDE in surface water and other water.</p> <p>Concentration limit proposal</p> <p>Considering the hitherto existing („political“) limits for other POPs: a) 50 ppm for Perfluorooctane sulfonic acid (PFOS), its salts and perfluorooctane sulfonyl fluoride (PFOSF) b) 10 ppm for each prohibited PBDE congener + Deca BDE (see answer ref. to. question 2, b), which results in 50 ppm in total for Tetra-, Penta-, Hexa-, Hepta and Deca BDE.</p> <p>Staggered Approach</p> <p>10 ppm (per prohibited PBDE plus DecaBDE pursuant to ROHS Directive); 50 ppm PFOS should be laid down on the condition that the following EC-Directives will be amended simultaneously: ROHS (new limit for PBDE in the electronics sector), WEEE (revision of recycling quotas), EOL-Directive (revision of recycling quotas considering PBDEs and PFOS application in EOL- vehicles); consideration of reuse of used EEE and vehicles containing higher amounts of PFOS/PBDEs – e.g. if an appliance consists of 10% plastics with 1% PBDE the whole appliance contains 0,1% (1.000 ppm) PBDE and therefore is a POPs-waste requiring a treatment where the POPs-content is destroyed; This excludes re-use; as a consequence any appliance produced before the enforcement of the ROHS Directive and in the future all electronic equipment containing more than 10 ppm per prohibited PBDE (including also DecaBDE) needs to be excluded from re-use to be in compliance with the POP Regulation; a subsequent amendment of the Waste Shipment Regulation No. 1013/2006 and the WEEE-Directive should be envisaged.</p>
Ireland	24 th June 2010	Department of Environment, Heritage and	Ireland is not aware of any existing concentration limits of PFOS and PBDEs in waste. For concentration limits in articles, soil, water, sediments comprehensive information was provided, e.g. RoHS Directive (0.1% PBDEs); REACH (0.1% PentaBDE and OctaBDE; 50mg/kg PEFOS); Limits in Surface Waters (Brominated diphenyl ethers: 0.0005 µg/l Annual average Environmental Quality Standard Inland surface

Country/ Respondent	Date of Submission	Institution/ Contact Person	Information related to PentaBDE, OctaBDE, PFOS, its salts and PFOSF
		Local Government, Darren Byrne	<p>waters and 0.0002 µg/l for Annual Average Environmental Quality Standard Other surface waters).</p> <p>Besides, relevant information on the following topics was included in the cover e-mail:</p> <p>1. <i>WEEE-RoHS Directives + Recasting</i> In relation to WEEE recycling and implications for brominated flame retardants (BFRs) containing the BDEs, currently Annex II of the WEEE Directive sets out the criteria for the 'Selective treatment for materials and components of WEEE. Section 1 states that as a 'minimum the following substances, preparations and components have to be removed from any separately collected WEEE and this includes 'plastic containing brominated flame retardants'. Such substances, preparations and components are permitted to be disposed of or recovered in compliance with Article 4 of Council Directive 75/442/EEC (now repealed). If POPs containing these BFRs are above any agreed threshold/concentration they may have to be treated as POPs waste, therefore Annex II of WEEE may have to be amended so as to ensure compliance with any future POPs Regulation requirements. Any such amendments may need also to feed into the negotiations on the current Commission recasting of the WEEE Directive.</p> <p>2. <i>WEEE Reuse & Refurbishment</i> We feel that this issue is of pertinence as some articles containing new POPs such as BDEs and PFOS in EEE & and furniture & textiles may continue to be circulated and possibly exported as non-waste but could end up as waste and specifically POPs waste outside the EU.</p> <p>3. <i>WEEE Processing in Ireland</i> With regards to WEEE recycling in Ireland, a substantive quantity of our collected WEEE undergoes pre-treatment in Ireland whereby hazardous substances & fluids etc. are removed prior to been shipped for onward treatment to other MS - primarily the UK & Germany. This is one of the primary reasons why the WEEE Collective Compliance Schemes & certain WEEE Producers do not have detailed analytical data vis-à-vis concentration levels of certain PDEs in BFR containing WEEE. Given the above, it is therefore important to appreciate that whatever waste management provisions that are established in other MS with regards to new POPs waste concentration limits will have a bearing on Ireland.</p> <p>4. <i>TFS</i> There is currently no threshold limit although based on REACH and RoHS a threshold level of 0.1% by mass of penta and octaBDEs is suggested. It would be our initial understanding that any agreed waste concentration limits for new POPs will potentially have an impact on the TFS Regulation and any TFS Correspondents Guidance. Information may also need to be sought on the possible implications for the current guidance on transfrontier shipment of 'spare parts' and refurbished equipment.</p> <p>5. <i>Standards for BFRs in Ireland</i> In early 2010 a questionnaire concerning a European Commission, DG Health and Consumers commissioned study (Identification and evaluation of data on flame retardants in consumer products), was completed by Ireland's standards authority (National Standards Authority of Ireland (NSAI)). Please see attached the NSAI's response to this questionnaire which includes details on fire safety requirements in consumer products and regulations concerning the use of flame retardants in consumer products in Ireland.</p> <p>Recycling Operations/Waste Management Options PFOS and PBDE In case recycling should be exceptionally allowed, the length of the exemption period should be at least 20 years, according to the text "Part IV" in the decision SC-4/14.</p> <p>Poland has the opinion that there is no strong, convincing evidence that there are significant releases of hazardous substances during recycling processes, but the combustion of such waste should not be allowed.</p> <p>The costs of separating PFOS and PBDE containing waste would be disproportionally higher than the ecological effect and is in practice</p>
Poland	24 th June 2010	Ministry of Environment, Piotr Karczewski	

Country/ Respondent	Date of Submission	Institution/ Contact Person	Information related to PentaBDE, OctaBDE, PFOS, its salts and PFOSF
			impossible.
			Possible implications of concentration limits for PFOS and PBDE in new articles produced from recycled material Proposed amendments –the inclusion of flame retardants in the Stockholm Convention – will cause significant problems for existing recycling and reuse processes for plastic waste (the process costs will increase).

Table 12-2: Overview of information submitted by stakeholders on PeCB

Country	Date of Submission	Institution/Contact Person	Information related to PeCB
Austria	August 2010	Federal Ministry of Agriculture, Forestry, Environment and Water Management, Mrs. Loew; Mrs. Busic, Mr. Moser – Waste Management Mrs. Schrott; Mrs. Eberhartinger- Taffill - Chemicals	No stockpiles in Austria, No information on sites contaminated; PeCB can be found in sewage sludge wastes for composting (general concentration limit: AOX 500 mg/kg d.s.); Austria
Czech Republic	12 th April and 11 th May 2010	Mr. Karel Bláha, Mr. Prof. Dr. Ivan Holoubek	(SC-4/19 info) Completed, indicating that PeCb is not contained in articles within the country, but that possibly stockpiles exist and that there are possibly contaminated sites. According to the national POPs Inventory Project on contaminated sites Few thousand tonnes of HCHs isomers exist in the area of former producer Spolana Neratovice (estimation recognized ca 10,000 t of HCHs isomers) and in the dumping sites (Hajek) connected with the former production (in the past more than 5,000 t of ballast of HCH isomers deposited). Conditions in Neratovice and Hajek represent potential human and ecological risks. Other chemicals potentially present might be HCHs and PeCB with contaminated sites. Decontamination of these sites is currently under discussion. More detailed information is available in the submission.
Poland	24 th June 2010	Ministry of Environment, Piotr Karczewski; Beata Kowalczyk	<p>No stockpile sin PL and no contaminated sites; regarding available data there is no PeCB in articles, recycled articles and waste in Poland, but the existence in imported goods is not known. PL indicated existing regulation on concentration limits:</p> <p>b) The European Commision agreed (April 2010) for concentration limit for PeCB in waste equal to 50 mg/kg, as an amendment to the Regulation (EC) No 850/2004.</p> <p>In the Regulation (EC) No 166/2006 of the European Parliament and of the Council threshold for PeCB releases to air, water and ground is equal to 1 kg/year (Annex II to the Regulation). PeCB is a priority hazardous substance in annex X to Directive 2000/60/EC of the European Parliament and of the Council as well. With regard to Poland: Limits for PeCB in water in annex 8 (limits of the chemical water quality indicators) to the polish Regulation:</p> <ul style="list-style-type: none"> - in streams, creeks, rivers, sewers, lakes, reservoirs – 0,007 µg/l - in marine internal waters, coastal waters - 0,0007 µg/l <p>In Poland PeCB is listed as a substance which is not allowed to use in pesticides [3].</p> <p>Proposal on lower concentration limit for PeCB in waste: Concentration limits for PeCB that already exist seems to be strict enough, as there is scarce data on production, trade and users of this substance worldwide. Regarding small data about production, trade, users and the equipment, which may contain PeCB, we should consider if reduction of imposed concentration limits for PeCB is required at first place.</p> <p>(SC-4/19 info) Completed, indicating that there exists a ban on use in articles and that no stockpiles exist. Further, it is generally specified that a program of reclamation of sites contaminated with hazardous pesticides will be ended at the end of 2010, but PeCB contamination has not been indicated.</p>

Romania	19 th May 2010	Mihaela CIOBANU, Ministry of Environment and Forests	(SC-4/19 info) Completed, confirming that there is no PeCB contained in articles within the country, there exist no PeCB stockpiles and there are no sites contaminated with PeCB.
Sweden	15 th April 2010	Maria Nyholm, SWEDISH ENVIRONMENTAL PROTECTION AGENCY	(SC-4/19 info) No data on PeCB indicated.
Ireland	24 th June 2010	Department of Environment, Heritage and Local Government, Darren Byrne	PeCB was never used as a pesticide in Ireland. The marketing and use of quintozone as a pesticide in Ireland has been banned since 2001. For further information please see cover mail.

Table 12-3: Overview of information submitted by stakeholders on candidate POPs

Country	Date of Submission	Institution/Contact Person	Information related to candidate POPs
Austria	August 2010	Federal Ministry of Agriculture, Forestry, Environment and Water Management, Mrs. Loew; Mrs. Busic, Mr. Moser – Waste Management Mrs. Schrott; Mrs. Eberhartinger- Tafill - Chemicals	<p>Austria indicates for SCCPs that the substance can be found in wastes as sewage sludge for composting with AOX limit for composting of 500 mg/kg d.s., as well as in rubber (with flame retardants) with concentrations of 1-10% SCCPs added to rubber (http://www.unon.org/confss/doc/unep/pops/POPRC_05/UNEP_POPS_POPRC_5_02/K0952246.doc).</p> <p>HBCD occurs in wastes as EPS wastes with concentrations up to 0.7%, i.e. 7000 mg/kg, XPS wastes with concentrations up to 2.5% and textile wastes with concentrations of 6-15%; there is no recycling of EPS and XPS from demolition waste in Austria, small amounts of construction wastes of EPS and EXPs are recycled.</p> <p>PCP can be found in wood waste with allowed concentration limits in AT for wood waste destined for material recycling of 1.5 mg PCP/kg d.s. (concentrations in different wastes can be found at http://www.lfu.bayern.de/umweltwissen/doc/uw_50_pentachlorphenol_pcp.pdf), in sewage sludge wastes (concentration limit for composting is 500mg AOX/kg d.s.) and in recycled articles as chip boards with a maximum PCP content of 5 ppm (Standards: EN 13986 Wood based panels for use in construction -Characteristics, evaluation of conformity and marking in connection with EN 312, EN 300, EN 313 and 314 and EN 326, Tl.2 etc.).</p> <p>HCBD can be found in sewage sludge wastes (concentration limit for composting: 500 mg AOX/kg d.s).</p> <p>PCNs can be found in sewage sludge wastes (concentration limit for composting: 500 mg AOX/kg d.s) and in PCB oils (PCNs were found in all the PCB mixtures at concentrations including Aroclors, aneclors, Clophens, Phenoclor, Sovol, and Chlorofen ranging from 5.2 to 730 mg/kg Literature: http://pubs.acs.org/doi/abs/10.1021/es001122u).</p> <p>Types of waste currently recycled including HBCD: small amounts of insulation foams (PS), mainly from construction operations are recycled; there is no recycling from demolition waste at the moment with new articles produced thereof as PS-containing concrete for flooring.</p> <p>Types of waste currently recycled including PCP: wood wastes for chipboard production</p> <p>Measurement methods for SCCP, PCP (up to 5 ppb in environmental samples) and HCBD: GC/MS, for HBCD LC/MS and PCNs GC/HRMS.</p> <p>AT indicates that it is possible to distinguish contaminated from non contaminated waste by separation at source of potentially contaminated wastes (e.g. treated wood from demolition – outdoor use of wood; old carpets with synthetic fibers; etc.) further, as preferred treatment option – R1 (D10) (an EU-wide restriction of TOC for landfilling should be visaged)</p> <p>AT indicates a concentration limit of maybe 50 ppm (further studies are necessary).</p>
Poland	24 th June 2010	Ministry of Environment, Piotr Karczewski; Beata Kowalczyk	<p>No stockpiles; concerning contaminated sites PL is conducting the “Program of remediation on sites contaminated with hazardous pesticides”, which will end at the end of 2010.</p> <p>Most often methods involves extraction of pesticides from landfill facilities, transportation to utilization plant (with taking rules for transportation of hazardous waste into consideration). Contaminated soil is transported to utilization plant as well. Usually site is secured with impermeable, mineral screens made with clay and affore stated in the end.</p>

Romania	14 th July 2010	<p>National Environmental Protection Agency and Ministry of Environment and Forests</p> <p>Mariana Mihalcea Udrea, Head of Chemicals Department and Mihaela Claudia Ciobanu, National Focal Point for Stockholm Convention</p>	<p>No data on the occurrence of SCCPs in PL; HBCD: in polystyrene for thermal insulation in construction and refrigeration industry (no concentration limits specified), in recycled articles in polystyrene granulate, milling, packs, foil, EPS (c≈0.8 by weight), XPS (c≈2.5 by weight); no data on PCP, PCN and HCBd;</p> <p>No data on wastes containing the candidate POPs that are currently recycled in PL, except HBCD: EPS and XPS with recycling rate of 80-85% of polystyrene waste being recycled and new articles being produced from PS are granulates, milling, packs, foils, insulation material, sound-absorbing screens, drainage material, binder in foundry and waterproof material.</p> <p>No methods available in PL for identifying the presence and levels of the candidate in waste yet; no data on development of such methods.</p> <p>If recycling should be exceptionally allowed, PL indicates an exemption period for HBCD of at least 20 years (derived from data from industry and fact that alternatives research is implemented more or less since 2000);</p> <p>Already existing concentration limits: in PL hazardous substances are listed in the regulation on waste catalogue;</p> <p>PL also indicates other existing concentration limits:</p> <ul style="list-style-type: none"> -PCP in treated industrial sewage is equal to 2,0 mg/l (daily average) and equal to 1,0 mg/l (monthly average) - Hexachlorobutadien (HCBd) in treated industrial sewage is equal to 3,0 mg/l (daily average) and equal to 1,0 mg/l (monthly average) - PCP in streams, creeks, rivers, canals, lakes, natural and artificial lakes, marine internal, transitional and coastal waters equal to 1,0 µg/l - HCBd in streams, creeks, rivers, canals, lakes, natural and artificial lakes, marine internal, transitional and coastal waters equal to 0,6 µg/l <p>SCCPs: occur in articles as plastics containing PL56L-80 as plasticizer (c=56*10⁻⁵ mg/kg minimum)</p> <p>HBCD: occur in polystyrene as flame retardant</p> <p>EPS – expanded (1*10⁻⁵ maximum)</p> <p>XPS – extruded (40*10⁻⁵ maximum)</p> <p>Measurement methods known according to EPA 8121 – Identifying Chlorinated Paraffines by Gas Chromatography – from waste water, soil; EPA 8270D – Identifying Semivolatile Organic Compounds – from waste, sediments</p> <p>Suggestion on immediate separation and option to distinguish contaminated from not contaminated waste is to sample waste and analyse these by gas chromatography</p> <p>Indication on existing concentration limits :</p> <p>Romania applies the EU 2000/532/EC Commission Decision of 3 May 2000 replacing Decision 94/3/EC establishing a list of wastes pursuant to Article 1(a) of Council Directive 75/442/EEC on waste and Council Decision 94/904/EC establishing a list of hazardous waste pursuant to Article 1(4) of Council Directive 91/689/EEC on hazardous waste</p> <p>Indication on other existing limits in RO:</p> <ul style="list-style-type: none"> - in articles – the limits established by chemical legislation
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Ireland	24 th June 2010	Department of Environment, Heritage and Local Government, Darren Byrne	<p>- in soil – the limits established by the Ministerial Order no 756/1997 for the approval of the regulatory system on the evaluation of environmental pollution</p> <p>- in water - Governmental Decision no 188/2002 for the approval of certain norms on the conditions of waste water discharging into aquatic environmental</p> <p>Opinion on appropriate concentration limits: Currently due to the lack of information at national level we could not appreciate which are the appropriate concentration limit for “candidate POPs” in waste, above which the POPs content shall be destroyed or irreversibly transformed.</p> <p>Opinion on lower concentration limit on which significant economic impacts can be expected: In order to consider the lower concentration limits for candidate POPs in waste at which significant economic impacts occurs it is necessary to perform studies at national level.</p> <p>IE indicates that in terms of pesticide use, Pentachlorophenol (PCP) or its derivatives were never produced in Ireland and its use as a wood preservative stopped in 2002. IE indicates that they are not aware of any concentration limits in waste; there are limits in articles for PCP, HBCD, SCCPs according to REACH as well as limits in surface waters for naphthalene, PCP and HCBd. For further details please see submission.</p>
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12.3 Annex 3: Evaluation Questionnaire for submission of information on New POPs in accordance with SC-4/19

Table 12-4: Overview of information submitted from the European Parties to the Stockholm Convention (Questionnaire SC 4/19, Part II, Section A and B) on commercial PentaBDE and commercial OctaBDE

Country	Date of Submission	Institution/Contact Person	Information Section A	Detailed Information Section B
Austria	19 th May 2010	Federal Ministry of Agriculture, Forestry, Environment and Water Management	Completed a helpful and detailed response with data relating to all BDEs including DecaBDE. Confirm that since 2000 no production of PBDE containing EEE. No information on articles containing PBDE, but information on concentrations in recycled waste streams. Detailed information on types of articles recycled and the contamination figures taken from the MBA Polymers recycling plant. Plastic granulates are produced from recycled materials for the housing of EEE in compliance with the RoHS Directive. Extensive information provided to legal frameworks (also national legislation). Various environmentally sound recycling and disposal operations introduced. No sites in AT are contaminated.	No data in terms of quantities but commented that the existence of BDE-contaminated equipment is unlikely due to the limited lifetime; According to a survey carried out in 1994, flame retardants containing organic bromine compounds were used in the Austrian textile industry.
Bulgaria	7 th April 2010	National Focal Point, Bulgaria Ministry of Environment and Water; Ms. Dimcheva	Completed confirming that there was no production in Bulgaria and no data on recycling but that the EU regulatory framework applies and that the process of collecting information on commercial PentaBDE and OctaBDE is underway. Comprehensive information to EU legal and other frameworks.	No data
Czech Republic	11 th May 2010	Ministry of the Environment, Dr. Karel Blaha; National POPs Centre, Prof. Dr. Ivan Holoubek	Confirms that it is not known whether there was any manufacturing of articles, but that there is no data on stocks or recycling. Legal frameworks: EU legislation. Brief description of recycling and disposal operations in Czech Republic. No contaminated sites have been identified, however some contaminated sediments. Inventory of contaminated sites by new POPs is currently being developed (2011)	No data
Estonia	6 th April 2010	Ministry of the Environment, Pruul	Completed confirming that Penta and OctaBDE were never produced and there is no relevant data on stocks or recycling	No data
Germany	7 th April 2010	Federal Environmental Agency (UBA), Dr. Steffi Richter	Completed confirming that goods containing Penta and OctaBDE were manufactured in Germany, but that there is no information on articles in use. EEE devices are only allowed to be recycled after separation of BDE containing parts (there is no recycling of BDE containing polymers) This is subject to the 0,1% RoHS threshold. Briefly introduced recycling and disposal operations.	No data
Lithuania	9th April 2010	Ministry of Environment, Aurelija Bajoraitiene	Completed confirming that Penta and OctaBDE were not produced and that there is no relevant data on stocks or recycling. The EEE waste which is defined as dangerous is transferred to the company UAB 'Toksika' conducting the long storage and treatment of dangerous waste.	No data
The Netherlands	1 st July 2010	Mr. J.K. Kwisthout	Confirms articles containing POP-BDE were manufactured in the Netherlands. No information is available on the recycling of articles which may contain BDEs. Brief description of relevant legal frameworks and identification methods for identifying	No data, but temporal range from 1970-2001; OctaBDE was mainly used for ABS at around 18%. Most

Country	Date of Submission	Institution/Contact Person	Information Section A	Detailed Information Section B
Norway	1 st July 2010	Climate and Pollution Agency; Eli Vike	the presence and levels of Penta and OctaBDE. Suggests that the Netherlands consumption of OctaBDE was 600-800t/y in 1990. In addition, short summary of relevant information from the EU Risk Assessment Report is given. Confirms that Norway had manufactured articles containing POP-BDEs but that there is no information on articles containing BDEs. An enforcement strategy is in place to identify articles. Waste containing more than 0.25% of PentaBDE and OctaBDE is defined as hazardous waste and has to be treated according to the regulation on hazardous waste. The export of hazardous waste for recycling is not allowed. Remediation method described: Site contaminated: canal leading to lake Mjøsa. The excavated contaminated mud was disposed at a safe landfill.	likely applications of ABS plastics are monitors, TV's and business machines. Between 1997 and 2003 75kg/y was used for textiles. Production, import, exports and use of PentaBDE and OctaBDE in articles is prohibited from 01.07.2004. No further details given
Poland	13 th July 2010	Ministry of Environment, Piotr Karczewski	Confirmed that Poland never manufactured articles containing Penta and OctaBDE. No information available on articles in use or on articles produced from recycled material. Description of national legal framework, identification methods, relevant recycling and disposal operations. No contaminated sites.	No data
Romania	19 th May 2010	Ministry of Environment and Forests; Mr. Ciobanu	Completed confirming that it is not known whether there was any manufacturing but that there is no data on stocks or recycling. Briefly introduced REACH Regulation in connection to Penta and OctaBDE.	No data
Slovenia	30 th June 2010	Chemicals Office of	No information on production, use or levels in articles. Confirms that Slovenia uses the EU regulatory framework and POP Regulations	No data
Sweden	15 th April	Swedish environmental Protection Agency; Maria	Questionnaire has been developed in collaboration with producers/producers associations, Recycling industry representatives, Government Authorities, University Researchers and Consultants. Confirming that goods containing Penta and Octa were manufactured in SE but that there is not information on articles in use. Comprehensive answers provided for identification methods, recycling and disposal operations, including Boliden smelter. Besides, confirmed that contaminated sites are not known. Additional information has been provided to the SAICM-project (information needs about chemicals through the supply chain called Chemicals in Product, which is implemented by UNEP and additional information to methods for identifying presence and levels of PBDEs	No data on quantities, time period for use indicated as 1970-2001
Switzerland	1 st July 2010	Federal Office for the Environment; Bettina Hitzfeld	Confirming that articles containing Penta or OctaBDE that have been put on the market in CH before the ban in 2005 were imported. A market survey carried out in 2008 indicated that no articles containing C-PentaBDE and only few that contain C-OctaBDE are on the market. All material from recycling processes containing PentaBDE and OctaBDE is either disposed of by thermal destruction in waste incinerators with state of art emission control technology in CH or exported to specialised enterprise who recovers bromine-free polymers by a physical separation process. Comprehensive description of legal frameworks, identification methods of PBDEs in articles, recycling and disposal operations, measures for the environmental	End of the 90's : <500t PentaBDE <680t OctaBDE For more information: Morf L. et al.: Selected polybrominated flame retardants PBDEs and TBBPA: Substance flow analysis.

Country	Date of Submission	Institution/Contact Person	Information Section A	Detailed Information Section B
			management of recycling operations have been included. No contaminated sites have been identified.	

Table 12-5: Overview of information submitted from the European Parties to the Stockholm Convention (Questionnaire SC 4/19, Part III, Section A and B) on Perfluorooctane sulfonic acid (PFOS), its salts and perfluorooctane sulfonyl fluoride (PFOSF)

Country	Date of Submission	Institution/Contact Person	Information Section A	Detailed Information Section B
Austria	19 th May 2010	Federal Ministry of Agriculture, Forestry, Environment and Water Management	Confirming that AT is still manufacturing articles containing PFOS, its salts and perfluorooctane sulfonyl fluoride. Information on articles in use is limited. Comprehensive description of legal frameworks (e.g. REACH, Trade and Industry Act, The Austrian Waste Management Act). Confirms that PFOS is used in the electronic industry and electroplating sector. It is unknown whether articles containing PFOS are recycled. Methods for identifying articles described for coated articles and liquids and waste water. AT has not identified any contaminated sites.	Relevant data provided for Photolithography in semiconductor manufacturing (0.2-0.3kg PFOS used in the process), Emissions not know, but considered very low, small amount of PFOS also stays in the product; Electroplating (alternatives tested but assumed that PFOS compounds are still in use); Consumer products (survey carried out in 1994 indicates that PFOS were used for textile impregnation); Fire Fighting Foams (contaminated with PFOS based chemicals 1-2%, ~800kg) Description of processes using PFOS provided for electroplating and Photolithography in semiconductor manufacturing. No specific material recycling of PFOS containing articles.
Bulgaria	7 th April 2010	National Focal Point, Bulgaria Ministry of Environment and Water; Ms. Dimcheva	Confirming that BG has not produced articles containing PFOS, its salts or PFOSF. Extensive information to legal frameworks provided. It is not known whether BG uses PFOS in processes or recycles articles than contain PFOS. No information on identification methods and remediation methods. No further answers provided.	PFOS is not produced in BG; Articles containing PFOS: Fire fighting foams (~3%, stock 11,000kg, annual amount ~330kg) Fire fighting foams available on stock were imported before 2007. No further data available. Information on types and quantities of PFOS, its salts of PFOSF in articles that may exist in BG including concentrations of those substances in articles is currently being collected.
Czech Republic	11 th May 2010	Ministry of the Environment, Dr. Karel Blaha; National POPs Centre, Prof. Dr. Ivan Holoubek	It is unknown whether CZ has produced articles containing PFOS, its salts or PFOSF. There are currently no articles in use containing PFOS, its salts or PFOSF. Few identification methods for introduced. No further data included.	No data
Estonia	6 th April 2010	Ministry of the Environment, Pruul	EE has never manufactured articles containing PFOS, its salts or PFOSF. No further information included.	No data
Germany	7 th April 2010	Federal Environmental Agency (UBA), Dr. Steffi Richter	It is not known whether Germany manufactured or is still manufacturing articles contaminated with PFOS, its salts or PFOSF. There is no information regarding articles in use. Relevant EU and national legal frameworks are presented. It is confirmed that Germany uses PFOS, its salts or PFOSF in processes. Identification methods for PFOS in water have been introduced. It has been confirmed that no PFOS is used in production of vulcanised rubber industry.	Industrial processes: Non decorative hard chromium plating and wetting agents for use in controlled electroplating systems (PFOS compound, released amount 126kg/y=~6.7%of compound, ca. 1883kg/y used in the plating industry); Fire fighting foams (ca. 3-10% of PFOS in fire fighting foams, currently existing amount of PFOS containing fire fighting foams ca. 1,050,000kg). No further details included.
Lithuania	9 th April 2010	Ministry of Environment, Aurelija	Unknown whether Lithuania has ever manufactured or still manufactures articles containing PFOS, its salts or PFOSF. No	No data.

Country	Date of Submission	Institution/Contact Person	Information Section A	Detailed Information Section B
The Netherlands	1 st July 2010	Bajoraitiene Mr. J.K. Kwisthout	information is available whether any articles are currently used. Also not known whether the country uses PFOS, its salts or PFOSF in processes. Confirms that no articles containing PFOS, its salts or PFOSF were manufactured in the Netherlands. However, NL uses PFOS in the processes. All industrial processes need a permit in conformity with Environmental Management Act. It is not known whether articles containing PFOS are recycled in NL.	Industrial processes: Non decorative hard chromium plating (PFOS compound, ~2.5-10% of the marketed solution, ca. 390kg/y used in the plating industry); Fire fighting foams (not being placed on the market since 2006, between 1997 and 2001: 200-300m ³ /y). In 2009: 16,222 m3 PFOS containing fire fighting foams in use.
Norway	1 st July 2010	Climate and Pollution Agency; Eli Vike	Confirms that Norway has manufactured articles containing PFOS, its salts or PFOSF and also uses PFOS in processes. No articles containing PFOS are recycled. Contaminated site has been for instance identified at Rygge airport Norway. Excavated contaminated soil will be disposed at safe landfills according to the regulation for safe deposit of hazardous waste.	PFOS in fire fighting foams (In 2005, 22t of PFOS in fire fighting foam containing PFOS and PFOS related compounds. In 2007, 0t of PFOS in fire fighting foams containing PFOS and PFOS related compounds); Certain metal plating processes and for photographic purposes.
Poland	13 th July 2010	Ministry of Environment, Piotr Karczewski	Confirms that Poland has not manufactured articles containing PFOS, its salts or PFOSF. PFOS, its salts or PFOSF is also not used in processes. No sites contaminated with PFOS were identified.	No data
Romania	19 th May 2010	Ministry of Environment and Forests; Mr. Ciobanu	Confirms that Romania has not manufactured articles containing PFOS, its salts or PFOSF. PFOS, its salts or PFOSF is also not used in processes.	No data
Sweden	15 th April	Swedish environmental Protection Agency; Maria	Confirms that SE has not manufactured articles containing PFOS, its salts or PFOSF. However, PFOS, its salts or PFOSF is used in processes. It is not known whether articles containing PFOS are recycled. Contaminated site has been identified and treatment methods described (i.e. active carbon).	Industrial process: PFOS salt, 350kg/y; fire extinguisher (content by weight <1%, 250kg of articles stored).
Switzerland	1 st July 2010	Federal Office for the Environment; Bettina Hitzfeld	Confirming that articles containing PFOS, its salts or PFOSF were manufactured in CH. No information on articles in use in CH. PFOS, its salts or PFOSF are used in processes. No sites known to be contaminated with PFOS	Aviation hydraulic fluids (500 ppm, no information whether produced in CH, Estimated PFOS consumption 10kg/y); Electroplating, hard metal plating (decorative and non decorative, Tetraethyl ammonium perfluorooctanesulfonate CAS: 56773-42-3, produced articles do not contain PFOS or only traces, consumption of 56773-42-3 approx. 200kg/y (2010); Coating of components for use in in-vitro diagnostic medical devices (consumption of PFOS precursor, 340kg/y in 2009, the coated components are entirely exported).

Table 12-6: Overview of information submitted from the European Parties to the Stockholm Convention (Questionnaire SC 4/19, Part IV) on PeCB and other chemicals listed in Annex A at the fourth Meeting of the Conference of the Parties

Country	Date of Submission	Institution/Contact Person	Part IV
Austria	9 th May 2010	Federal Ministry of Agriculture, Forestry, Environment and Water Management, Sonja Loew, Susanna Eberhartinger, Helga Schrott	Completed confirming that PeCB is not contained in articles within the country, that no stockpiles exist, that no registered contaminated sites exist on PeCb except for one site, for which however, contamination with chlorinated benzenes has been reported; providing general information (Project MONARPOP – background monitoring of POPs in the Alps; available at www.monarpop.at - www.monarpop.at/publications/Technical_Report.php)
Bulgaria	7 th April 2010	MINISTRY OF ENVIRONMENT AND WATER, Ms.TSVETANKA DIMCHEVA	Completed, indicating that PeCB is not contained in articles within the country due to no reliable information available and that preliminary data is in the process of gathering. No information is available on whether stockpiles exist.
Czech Republic	12 th April and 11 th May 2010	Mr. Karel Bláha, Mr. Prof. Dr. Ivan Holoubek	Completed, indicating that PeCb is not contained in articles within the country, but that possibly stockpiles exist and that there are possibly contaminated sites. According to the national POPs Inventory Project on contaminated sites Few thousand tones of HCHs isomers exist in the area of former producer Spolana Neratovice (estimation recognized cca 10,000 t of HCHs isomers) and in the dumping sites (Hajek) connected with the former production (in the past more than 5,000 t of ballast of HCH isomers deposited). Conditions in Neratovice and Hajek represent potential human and ecological risks. Other chemicals potentially present might be HCHs and PeCBz with contaminated sites. Decontamination of these sites is currently under discussion. More detailed information is available in the submission.
Estonia	6 th April 2010	Ministry of the Environment, Reet Pruul	No relevant data.
Germany	7 th April 2010	Federal Environment Agency (UBA), Dr. Steffi Richter	No data on PeCB indicated.
Lithuania	9 th April 2010	MINISTRY OF ENVIRONMENT, Aurelija Bajoraitiene	Completed, indicating that there is no data on presence of PeCB in articles in the country. There is no reported data on stockpiles of PeCB and contaminated sites with those chemicals. More general information is available in the submission.
Netherlands	25 th June and 1 st July 2010	Mr. J.K. Kwisthout	No relevant data on PeCB.
Poland	13 th July 2010	Piotr Karczewski, Ministry of Environment	Completed, indicating that there exists a ban on use in articles and that no stockpiles exist. Further, it is generally specified that a program of reclamation of sites contaminated with hazardous pesticides will be ended at the end of 2010, but PeCB contamination has not been indicated.
Romania	19 th May 2010	Mihaela CIOBANU, Ministry of Environment and Forests	Completed, confirming that there is no PeCB contained in articles within the country, there exist no PeCB stockpiles and there are no sites contaminated with PeCB.
Sweden	15 th April 2010	Maria Nyholm, SWEDISH ENVIRONMENTAL PROTECTION AGENCY	No data on PeCB indicated.
Ukraine	9 th April 2010	Ministry of Environmental Protection, Department of	No relevant data on PeCB, additional information needed.

Country	Date of Submission	Institution/Contact Person	Part IV
		Environmental Safety	

12.4 Annex IV: Physico-chemical properties of POPs

Table 12-7: Physico-chemical and toxicological properties of POPs

Substance	CAS Nr.	Degradation time (DT50)	BAF	Carcinogenic group (IARC)	TDI (FAO)	log Kow	OEL	NOAEL	Minimal risk levels (ATSDR, Dec 2004)		Criterion Y
		soil/water			µg/kg b.w. d.	mg/kg	mg/m ³ *		mg/kg/day		ppm
									oral		
high risk substances		> 25 days	10 ⁵					NOEC < 0,5 µg/l	acute	chronical	
Aldrin	309-00-2	20 d-1,6 years		3		0.006-0.2	0.25E		0.002	0.00003	50
Dieldrin	60-57-1	3-4 years		3	0.1		0.25E	rats: 0.025 mg/kg b.w. 2 mg/kg diet		0.00005	50
Chlordane	57-74-9	~ 4 years	10 ⁵	2B		0.002-0.5	0.5E	rats: 0.1 mg/m ³ monkey: 10	0.001	0.0006	50
DDT	50-29-3	<15 years	10 ⁵ -10 ⁶	2B	20 (JMPR)	0.02-5	1E		0.0005		50
Heptachlor	76-44-8	0.75-2 years		2B		0.006-0.2	0.5E				50
Endrin	72-20-8	<12 years		3	0.2 (JMPR)		0.1E	rats: 0,05 mg/kg		0.0003	50
Mirex	2385-85-5	<10 years		2B						0.0008	50
Toxaphene	8001-35-2	100 d-12 years	10 ⁴ -10 ⁵	2B			0.5E	dog: 0,2 mg/kg	0.005		50
HCH (mixed isomeres)	608-73-1	>1-2 years		2B			0.5E		0.2 (beta)	0.008 (alpha)	50
Lindan	58-89-9								0.003		50
Chlordecone	143-50-0			2B			n.d.		0.01	0.0005	50
HCB	118-74-1	2.7-23 (estimated)		2B			n.d.	fish: <3.7 µg/l	0.008	0.00005	50
HxBB	36355-01-8			2B							50
PCB	1336-36-3	> 6 years		2A	0.03		0.7-1.1	trout: 0.01 µg/l		0.02 µg/kg/day	50
PCDD/F (TEQ)		TCDD: 10-12 years		3; TCDD cat.1	1-4 pg		50 pg				1 ppb

Substance	CAS Nr.	Degradation time (DT50)	BAF	Carcinogenic group (IARC)	TDI (FAO)	log Kow	OEL	NOAEL	Minimal risk levels (ATSDR, Dec 2004)		Criterion Y
TetraBDE (47)	40088-47-9	-	2.9-5.3 (PBDEs)	-	-	-	-	lowest NOAEL of 1 mg/kg/ b.w. per day for rats. UF = 30	0.03 (oral acute)	0.007 (oral int.)	50
PentaBDE (99)	32534-81-9	-	2.9-5.3 (PBDEs)	-	0.0003	-	-	LOAEL of 0.06 mg/kg b.w. per day	0.03 (oral acute)	0.007 (oral int.)	50
C-PentaBDE		-	2.9-5.3 (PBDEs)	-	-	-	-	Chronical exposure; most sensitive NOAEL of 0.45 mg/kg b.w./d	0.03 (oral acute)	0.007 (oral int.)	500
C-OctaBDE		-	2.9-5.3 (PBDEs)	-	-	-	0.1	Most sensitive NOAEL of 2 mg/kg b.w./d	0.03 (oral acute)	0.007 (oral int.)	500
PFOS	1763-23-1 (PFOS) 307-35-7 (PFOSF)	half life: >41y	BMF: 22-160	-	0.15 (PFOS) 1.5 (PFOA)	-	0.01 (inhalable aerosol, 8h) 0.08 (inhalable aerosol, short term)	lowest NOEL 0.03 mg/kg/ b.w. per day Cynomolgus monkeys for PFOS 0.3 mg/kg/ b.w. per day for rats and mice for PFOA. UF = 200 (100 for inter and intra-species differences; 2 for uncertainties .	-	-	5
PeCB	608-93-5	>6 years	high	-	0.5 µg/kg bw/day (Canadian value)	5.17-5.18	-	-	0.005 mg/kg/ b.w. per day	-	500
SCCPs	85535-84-8	>1year (sediment) 0.81 to 10.5 days (half lives in air)	-	2B (chlorinated paraffins of average carbon chain length C12 and average degree of	100	4.48-8.69	-	lowest 10 mg/kg b.w. per day for rats. UF = 100 (10 for interspecies variation)	-	-	10000

Substance	CAS Nr.	Degradation time (DT50)	BAF	Carcinogenic group (IARC)	TDI (FAO)	log Kow	OEL	NOAEL	Minimal risk levels (ATSDR, Dec 2004)		Criterion Y
				chlorination - approx. 60%)							
HBCD	25637-99-4	half-lives (aerobic and anaerobic conditions, respectively): 63 and 6.9 days	-	-	1	-	5 (SE) 10 (USA)	-	-	-	1000
PCP	87-86-5	half life: 2-4 weeks	-	2B (polychlorop henols and their sodium salts; mixed exposures)	-	5.12-5.18	-	-	0.005	0.001	5000
HCBD	87-68-3	half life: 4-26 weeks	-	3	0.2	4.78	0.24 (8h; DK, CH, USA NIOSH) 0.21 (8h; BE, Canada); 0.20 (8h; ES (skin)) 0.48 (short term; DK)	0.2 mg/kg/ b.w. per day for rats. UF 1000 (100 for inter- and intraspecies variation; 10 for limited evidence of carcinogenicity and genotoxicity of some metabolites)	0.0002	-	100
PCNs	75 numbers for individual congeners	-	-	-	-	-	-	-	-	-	-

12.5 Annex V: Number and capacity of treatment facilities in 2004 and 2006 at country level

Table 12-8: Overview of number and capacity of treatment facilities in 2004 and 2006 at country level (source: EUROSTAT)

Country	Year	Incineration with energy recovery (R1)			Other incineration (D10)			Recycling (R2 - 11)		Landfilling (D1, D3 - 5, D12)	
		Facilities	Capacity		Facilities	Capacity		Facilities	Capacity	Facilities	Capacity
		number	tJoules/year	tonnes/year	number	tJoules/year	tonnes/year	number	tonnes/year	number	m3/year
Belgium	2004	213	134,487	7,444,947	219	47,650	3,191,404	776	12,043,619	184	71,325,625
	2006	262	30,223	4,643,719	166	34,171	3,499,010	534	10,065,707	63	50,962,612
Bulgaria	2004	34		160,765	28		45,846	116	1,879,742	633	25,233,656
	2006	55		136,604	33		52,519	216	2,608,629	495	65,850,171
Czech Republic	2004	12		701,320	29		774,364	327	7,401,279	210	83,095,550
	2006	34		772,141	24		775,108	636	33,480,462	210	83,095,550
Denmark	2004	29	36,225	3,450,000	0	0	0	215	9,000,000	150	
	2006	1			29	36,225	3,450,000	215	10,768,000	150	
Germany	2004	618		20,623,263	120		18,420,002	10,069	252,872,178	2,019	686,728,241
	2006	682		27,012,007	111		18,142,217	10,860	315,818,795	1,747	499,657,445
Estonia	2004	86		267,874	8		2,457	251	3,603,989	66	110,613,288
	2006	83	4,326	256,567	2	0	41	139	7,805,760	39	269,272,405
Ireland	2004	3 ⁸⁰			6			73		67	
	2006	9			5			1,166		69	
Greece	2004	2	331		1		11,000	69	563,442	1,492	17,705,800
	2006	7	330	6,139	4		13,067	60	990,550	1,440	22,056,786
Spain	2004	64	2,048	1,672,250	16	83	769,517	1,574	58,729,276	462	73,233,509
	2006	73	468	2,645,304	64	375	1,348,121	1,643	81,172,185	482	104,293,584
France	2004	112			22			414		317	
	2006	182			186			1,040		769	

⁸⁰ These figures refer to facilities that are attached to industrial facilities (pharmaceutical) for the purpose of site specific waste management. These are not commercial incinerators and are therefore not relevant in the context of POPs waste management (Personal communication EPA Ireland; e-mail from 18.03.2011)

Country	Year	Incineration with energy recovery (R1)			Other incineration (D10)			Recycling (R2 - 11)		Landfilling (D1, D3 - 5, D12)	
		Facilities	Capacity		Facilities	Capacity		Facilities	Capacity	Facilities	Capacity
		number	tJoules/year	tonnes/year	number	tJoules/year	tonnes/year	number	tonnes/year	number	m3/year
Italy	2004	374			134			6,427		1,145	
	2006	687			63			2,960		557	48,256,915
Cyprus	2004	0			0	0	0	74	730,471	6	6,460
	2006	1		12,800	0	0	0	44	1,305,692	54	
Latvia	2004	5		31	9		0	22		151	
	2006	3		52,100	2		2,400	100		103	0
Lithuania	2004	184			15			221		291	
	2006	5	4,115	139,919	3	248	22,136	225	985,082	206	5,644,242
Luxembourg	2004	2			1		140	81	14,354	12	15,457,979
	2006	2		53	1		140	89	13,878	11	13,878,713
Hungary	2004	30		564,104	14		55,499	182	2,067,762	321	117,973,514
	2006	28		994,046	15	0	104,414	368	16,540,103	204	54,540,344
Malta	2004	0			0	0	0	1	80,000	5	2,094,884
	2006	0			0	0	0	4	150,421	12	4,181,982
Netherlands	2004	330		2,005,575	15		5,673,750	202	24,855,700	42	138,100,000
	2006	1,259		3,379,900	13		5,983,000	200	28,173,559	40	133,120,000
Austria	2004	120		2,028,300	65		2,509,116	1,233	8,973,790	666	76,109,513
	2006	177		2,925,570	12		1,666,130	1,233	8,652,180	542	89,097,700
Poland	2004	354		5,562,061	20		421,619	630	21,033,333	1,445	
	2006	678		3,886,712	14	0	444,142	754	23,186,417	1,157	
Portugal	2004	3		1,168,000	0	0	0	9	0	35	
	2006										
Romania	2004	37		5,352,009				2	2,870	449	4,213,313
	2006	172		2,741,417	11		56,042	96	1,345,180	410	1,383,485,892
Slovenia	2004	37		345,833	3		23,460	106	4,203,669	69	1,719,762

Country	Year	Incineration with energy recovery (R1)			Other incineration (D10)			Recycling (R2 - 11)		Landfilling (D1, D3 - 5, D12)	
		Facilities	Capacity		Facilities	Capacity		Facilities	Capacity	Facilities	Capacity
		number	tJoules/year	tonnes/year	number	tJoules/year	tonnes/year	number	tonnes/year	number	m3/year
	2006	13		121,978				157	7,731,764	53	8,836,103
Slovakia	2004	57		476,000	55			244	3,180,000	165	31,123,000
	2006	12		209,500	18		153,600	350	3,316,082	160	40,200,000
Finland	2004	26		422,390	4		355,000	336	3,128,335	269	
	2006	23		2,278,800	2		67,000	354	6,018,364	300	22,222
Sweden	2004	261		17,261	16		105	12,924	22,439	271	385,746
	2006	360		19,112,085	15		134,249	502	29,246,015	292	341,776,645
United Kingdom	2004	615		12,809,716	1,279		1,806,246	15,230	16,690,641	907	783,964,653
	2006	362		194,363	3,104		9,149,345	26,737	140,921,445	721	830,234,974

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