



Development of European Ecolabel and Green Public Procurement Criteria for Imaging Equipment

BACKGROUND REPORT
INCLUDING DRAFT CRITERIA PROPOSAL
Annex-Working Document

for

2nd AHWG MEETING FOR THE
REVISION OF GPP CRITERIA
FOR IMAGING EQUIPMENT

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4 ANNEXES

DRAFT

4.1 Checklist for recyclable design

A: Structure and Connection Technology

	Components made of materials incompatible with each other are connected separably or via separation aids	Case parts, chassis, electric modules, toner modules	M	<input type="checkbox"/>
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Important connections are those between case and chassis as well as those between chassis and electric modules. Their separability is a prerequisite for separate use/recycling of modules and materials as well as for a quick and save separation of pollutant-containing components. Adhesive labels (e.g. company logos and labels) are concerned as well.

The term "separation aids" stands, for example, for predetermined breaking points.

	Electric modules are easily traceable and removable	Entire unit, including lamps	M	<input type="checkbox"/>
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Minimum recycling strategy means: removal of the pollutant freight.

Electric modules and components according to Annex III, ElektroG (Electrical and Electronic Equipment Act), as, for example, batteries and condensers involving the risk of pollutant-containing ingredients as well as mercury-containing fluorescent lamps must be easily traceable and separable.



	Disassembly can be done with universal tools exclusively	Case, chassis, electric modules	M	<input type="checkbox"/>
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"The term „universal tool“ stands for general commercial tools.

	Necessary points of application and working space for disassembly tools have been taken into consideration	Case parts, chassis, electric modules	M	<input type="checkbox"/>
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A point of application is the point from where the impact is transferred from tool to connecting element. Sufficient working space is needed for the execution of the tool's separating movement.

This requirement particularly refers to snap connections which, unlike during assembly, often require tools to be disconnected.

	Screwed connections between modules can be separated with no more than three tools	Case parts, chassis, electric modules	M	<input type="checkbox"/>
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Standardized and uniform connection elements facilitate disassembly. The less tools must be changed the easier is assembly and disassembly.

A tool is characterized by the type of drive (e.g. cross recession) and the drive size (spanner size).

	Disassembly can be done by a single person	Entire unit	M	<input type="checkbox"/>
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An optional number of snap connections of the same joining direction may be assembled at a time but not always be disassembled if the re-entrant angle is $\geq 90^\circ$. This requirement shall be considered not fulfilled if more than two connections must be separated at a time.

	Case parts are free from electronic modules	Case parts	M	<input type="checkbox"/>
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With regard to a clean and quick pollutant removal and separation of the electronic parts all electric modules must be connected to the chassis. The case may not contain any electronic modules. Here, a control element attached to the case and case parts which simultaneously perform the functions of the chassis are not considered as case parts.

	The manufacturer did a trial disassembly (e.g. according to A.1 - A.11) and prepared a test report focussing on the weak-points	Entire unit	M	<input type="checkbox"/>
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B: Selection and Marking of Materials

	The variety of materials forming plastic components performing comparable functions are limited to one material	Case parts, chassis mechanical parts ($\geq 25g$)	M	<input type="checkbox"/>
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The smaller the number of materials the more efficient are separation and recycling processes. This requirement shall not apply to parts that have been reused as can be proved.

	The coating of plastic components has been limited to the minimum necessary	Case parts, toner and ink modules	M	<input type="checkbox"/>
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Large-area layers of lacquer, vapour depositions and printings on plastic components require additional removal processes if recycling by the material is to be done thereafter. Reasons shall be given if metallic coatings are used. Laser-produced labelings shall not be considered as printings. This requirement shall not apply to parts case that have been reused, as can be proved.

	The materials and material compounds used can be recycled by the material	Case parts, chassis, toner modules	M	<input type="checkbox"/>
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This means that recycle materials identical to the original material (original recycling) can be obtained.

	The proportional use of recycle material is permitted	Case parts, chassis, toner and ink modules	M	<input type="checkbox"/>
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A real "cycle" does not exist before the manufacturer actually uses recycle goods or promises to do so along with the product specification.

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	Components and materials under Annex III to ElektroG (Electrical and Electronic Equipment Act) can be easily exchanged	Entire unit	M	<input type="checkbox"/>
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	Material selection according to B.1-B.5 has been done and recorded in writing	Case parts, chassis, toner modules	M	<input type="checkbox"/>
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	Plastic parts > 25 g according to EN/ISO 11469: considering ISO 1043 are marked	Entire unit	M	<input type="checkbox"/>
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Plastics marking enables all recycling companies to do a type-specific sorting and separation of plastics

C: Longevity

	At least 50% of the components of the device, except for standard parts, are identical in design to those in other devices of the same manufacturer and the same performance category and generation	Entire unit	M	<input type="checkbox"/>
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	The use of reprocessed modules or components is possible and permissible	Entire unit	M	<input type="checkbox"/>
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The manufacturer shall be prepared to use modules and components as spare parts or ETN-parts in the product, provided that they have been reprocessed under his guidance - (ETN- equivalent to new)



	Toner or ink modules can be reprocessed	Toner and ink modules, except for containers	M	<input type="checkbox"/>
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Reuse should not be prevented by constructive measures

All Requirements have been met and answered „Yes“?		M	<input type="checkbox"/>
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Place:
Date:

Applicant:
(Signature of authorized representative and company stamp)

4.2 Reporting of printouts produced after cancelation-measurement method

The applicant shall report the number of paper sheets which are printed or copied after the user has cancelled the printing or copying process separately for one-side printing and for double-side printing based on the measurement method described below.

Measurement method for the assessment of the reporting of number of printouts produced after cancelation

The following measurement method is proposed:

The devices shall be tested in the following modes while operating in high performance (speed and print quality):

One side monochrome printing

Double side monochrome printing

One side colour printing

Double side colour printing

In all cases A-4 size paper having a weight per unit area of 70 to 80 g/m² shall be used for the printouts. The double side printing test is only applicable only for devices equipped with automated duplex unit.

The same monochrome and colour sample will be used as the test sample as was used in the measurement of indoor emissions in Blue Angel Ecolabel criteria RAL-UZ122:2006-04 (also available via <http://www.ps.bam.de/RALUZ122E/>) originating from JBMS-74-1.

The printing process shall start and shall be interrupted (cancelled) when the forth printout leaves the internal printing part and is in on the respective casing part available and ready for the user to take. The cancelation can be made either using the software cancelling option or if available via a button directly on the hardware.

The number of paper sheets printed after the printing cancelation shall be reported.

The final reported value shall be the average of three tests.

The following table shall be completed:

Table 1 Form for reporting the number of printouts produced after cancelation

Tested operation	Speed in ipm	Number of printouts printed after cancelation
One side monochrome printing		
Double side monochrome printing		
One side colour printing		
Double side colour printing		

In case of copiers the same test measurement procedure shall be used.

4.3 Investigation for brominated aromatic additives used in plastics

4.3.1 Background information of brominated aromatic flame retardants

In this section, an overview of the field of the most commonly found brominated aromatic flame retardants is presented. Information with regard the different types of FRs and in which plastic parts they are used is given. The relevant reference list for this section is given in Table 3:

Flame retardants are commercially used in PC, PC-blends, ABS, SAN and PET. The average loads are between 10 – 20% w/w and depend on the specific requirements and properties needed for the end product as presented in more detail in Table 2.

A selection of brominated flame retardants commercially used in PC, PC blends, ABS, SAN and PET is presented in Table 2.

Table 2 A selection of brominated flame retardants commercially used in PC, PC blends, ABS, SAN and PET²³

Additive used for the flame retardant properties	Chemical name	CAS RN	Material							
			Poly carbonate (PC)	Poly carbonate blends (PC/ABS)	Acrylonitrile butadiene	Styrene acrylonitrile	Poly ester (PET)			
BROMINATED FLAME RETARDANTS (BFR)	Brominated polystyrene/ATO ²⁴	88497-56-7							X	
	DecaBDE/ATO	1163-19-5	X	X			X			
	Decabromodiphenyl Ethane/ATO	84852-53-9	X	X			X			
	Ethylene bistetrabromo phthalimide	32588-76-4	X	X					X	
	Pentabromobenzyl acrylate	85-22-3							X	
	Poly(pentabromobenzyl acrylate) / as polymer Pentabromobenzyl acrylate / as reactive monomer	59447-55-1	As polymer	X		As monomer				As monomer

²³ Norwegian EPA (KLIF), "EMERGING "NEW" BROMINATED FLAME RETARDANTS IN FLAME RETARDED PRODUCTS AND THE ENVIRONMENT" (2009)

²⁴ ATO: Antimony trioxide, CAS Nr: 1309-64-4, a common synergist together with brominated flame retardants

Pentabromotoluene	87-83-2			X		X
TBBP-A	79-94-7	X	X			
TBBPA carbonate oligomer				X		
Tris(tribromoneopentyl) phosphate	19186-97-1				X	
1,2-Bis(2,4,6-tribromophenoxy)ethane	37853-59-1			X		

Polybrominated diphenylethers (PBDEs) are a group of additive flame retardants. The term PBDE includes commercial pentabromodiphenylether (C-PentaBDE), commercial octabromodiphenylether (C-OctaBDE) and commercial decabromodiphenylether (C-DecaBDE), which are commercial mixtures of homologues of brominated diphenylethers with various substitution levels of bromine substituted to the aromatic rings.

Since C-PentaBDE and C-OctaBDE, due to their hazardous properties, have been phased out in several regions and countries worldwide, C-DecaBDE is the PBDE still produced and used in large quantities worldwide.

DecaBDE is applied in the range between 10%-30% of the polymer weight to various polymers such as styrenics and polyolefins, but may also be applied in textile back coatings. The vast majority of the decaBDE currently produced is used in the outer casing of electric equipment.

In order to reduce the overall amount of brominated-FR use in a polymer, the synergist antimony²⁵ is often added in the form of antimony trioxide. DecaBDE has an optimum applied

²⁵ The term synergistic effect is used in the connection with the development of flame retardants. This term means that the desired effect of two or more components working together is greater than the effect of each component separately. Perhaps one of the most important effects historically in flame retardant chemistry is the one between halogen and

ratio with antimony trioxide of *1 part antimony: 3 parts bromine*, which has shown to be the most effective ratio concerning flame retardant synergetic effects in a wide range of polymers in order to achieve optimum fire retardant properties.

TBBPA is an aromatic brominated organic compound and, as previously described, is primarily used as a reactive intermediate in the manufacture of flame retarded epoxy and polycarbonate resins. Therefore, it is an integral part of the polymer.

Further, TBBPA may also be used as an additive flame retardant physically mixed into the polymer, for example in the manufacture of acrylonitrile-butadiene-styrene resins (ABS) and phenolic resins. When TBBPA is used as an additive flame retardant it is usually with antimony trioxide which is not the case when it is used as a reactive flame retardant.

TBBPA is also used for the manufacture of derivatives to TBBPA-dimethylether, TBBPA-dibromopropylether, TBBPA-bis(allylether), TBBPA-bis (2-hydroxyethyl ether), TBBPA-brominated epoxy oligomers and TBBPA-carbonate oligomers.

Besides the PBDEs and TBBPA and its oligomers, there are several other aromatic brominated flame retardants that may be applied in imaging equipment.

Table 3 Reference list 1 for brominated flame retardants

Alaee, M; Arias, P; Sjödin, A; Bergman, Å: An overview of commercially used brominated flame retardants, their applications, their pattern use in different countries/regions and possible modes of release”, Environment International 29 pp 683 – 689. (2003)

<u>Bromine Science Environmental Forum - Science & Knowledge dedicated to Bromine</u>

antimony, where antimony reacts in the form of antimony trioxide with the formation of radicals, finally forming antimony tribromide and antimony oxibromide and their chlorinated equivalents. These flame retardants react in gaseous phase and usually contain halogens of the bromine and chlorine type.

and BFRs, www.bsef.org
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S. Lundstedt, ”Emissions, transformation and formation of brominated substances during fires”, University of Umeå , 2009 (Written in Swedish)
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Posner, S. "Guidance on alternative flame retardants to the use of commercial pentabromodiphenylether (c-PentaBDE)", UNEP-POPS-POPRC.4-INF-13.English.pdf (2009)
Renner, R: "PBDE – Polybrominated diphenylethers – What fate for Brominated Fire retardants?", Environmental Science and technology – American Chemical Society (2000).
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R. Weber and B Kuch "Relevance of BFRs ang thermal conditions on the formation pathways of brominated and brominated-chlorinated dibenzdioxins and dibenzofurans", Environ Int. 2003 Sep;29(6):699-710.
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4.3.2 Life cycle consideration of brominated aromatic flame retardants

Environmental concerns about the use of Brominated flame retardants are raised due to their role in the formation of Br-dioxins (PBDD), Br-furans (PBDF) as well as Br/Cl-dioxins (PXDD) and Br/Cl-furans (PXDF) when chlorine is present.

In general the degree of Bromine containing dioxins and furans formation depends on the precursor quality and on the type of treatment. In Table 4 the potential of dioxin formation in relation to the thermal treatment undertaken is given. All brominated aromatic compounds can act as PBDD/PBDF precursors in thermal processes of:

- a) pyrolysis /gasification as in this case 350 – 800 °C is often reached with low oxygen content (category 2 of Table 4) and;
- b) in thermal processes with insufficient combustion conditions i.e. incinerators operating in non-BAT conditions, secondary metal plants, uncontrolled burning (3 of Table 4).

The relevance of incinerators operating in non-BAT conditions is higher as a large share of e-waste (including exported European electronics and e-waste) or plastic from e-waste is finally treated in developing countries and countries in transition economies under such conditions.

Table 4: Categories of thermal treatment, related actual processes, prevailing formation pathways ways and potential of PBDD/PBDF formation (Weber and Kuch 2003)

Category	Processes	Conditions	Prevailing formation pathways	PXDD/PXDF formation potential
1) Thermal stress	Production, recycling (include shredding, molding and extrusion)	100 – 300 °C, mechanical stress	Selected precursors (e.g. PBDEs)	Low -> moderate
2) Pyrolysis/gasification	Pyrolysis/gasification facilities (pyrolysis of plastics, shredder)	350 – 800 °C, low oxygen	Precursors + formation of aromatic	High

		fractions or sludges). Accidental fires; uncontrolled burning	content	compounds and halogenation.	
3)	Insufficient combustion conditions	Accidental fires, uncontrolled burning, non-BAT incinerators. Secondary metal plants	Uncontrolled parameters: temperature, residence time, oxygen content, turbulence.	Precursors	High
4)	Controlled combustion conditions	BAT incinerators, cement plants	Optimised combustion control	Formation of aromatic compounds and halogenation	Low

With regard to brominated dioxins and furans, the precursor pathway for their formation can be identified more easily than in the case of Cl-dioxins and Cl-furans (PCDD/PCDF) in which the precursor pathway for their formation cannot be identified precisely. Bromine is primarily present in specific plastics.

Bromine is found in plastics in the form of BFRs. Brominated aromatic compounds can act as precursors for PBDD/PBDF formation (e.g. brominated diphenylethers (PBDEs) or brominated phenols (PBP)). Hence, the precursor pathway is of higher relevance for the formation of polybrominated dibenzodioxins (PBDDs) and polybrominated dibenzofurans (PBDFs) during thermal processes compared to their chlorinated analogues.

PBDD/PBDF and brominated-chlorinated PXDD/PXDF have similar toxicity as PCDD/PCDF and sometimes even exceed the toxicity of their chlorinated counterparts (see Annex 6.11.6). PBDD/PBDF and PXDD/PXDF are relevant contributors to dioxin-like exposure to humans.

The total amounts of brominated dioxins/furans generated from the polybrominated diphenyl ethers alone are estimated in the tonnes scale and are comparable in magnitude to the total global release of chlorinated dioxins and furans from all thermal sources combined (Zennegg et al. 2009).

During the entire life cycle of BFR-containing materials, PBDD/PBDF, and in the presence of a chlorine source also brominated-chlorinated PXDD/PXDF, can be formed and released into the environment (Ebert and Bahadir 2003, Kajiwara et al.2008; Kajiwara & Takigami 2010; Weber and Kuch 2003; WHO 1998).

Hence, a key concern regarding brominated flame retardants (this is applicable also for halogenated FRs in general) is at their end-of-life management. The three main options for materials treated with aromatic brominated flame retardants and other halogenated flame retardants are:

- (1) recycling,
- (2) incineration/thermal,
- (3) land filling.

Management of end-of-life waste flows can vary widely based on the type of waste management schemes used in various countries and the different materials involved (e.g. flame-retarded plastic in e-waste). The relevant environmental concerns raised for each option of waste management will further be analysed in more detail.

As there are numerous studies in which the environmental considerations along the product life cycle of products containing brominated flame retardants, which are related to the formation of Br-dioxin (PBDD) and Br-Furans (PBDF), are highlighted. A list of references used is given in section 6.11.9 of Annex 6.11.

4.3.3 Recycling of Brominated Flame Retarded Materials

The advantage of mechanical recycling as a method for end-of-life treatment of flame retarded materials is that it contributes to the overall reduction of the energy-intensive production of new materials.

Brominated aromatic flame retardants are precursors of brominated dioxins and brominated dibenzofurans. Some brominated aromatic BFRs (in particular PBDE) can form brominated dibenzofurans, even during necessary recycling operations like extrusion and molding of new plastic products requiring elevated temperatures (Meyer et al 1993, Mc Alister 1990, Weber and Kuch 2003).

Further, significant material flows containing halogenated flame retardants and other toxic materials are often exported to developing/transition countries and recycled there with cheap labour and primitive methods²⁶ (see also Annex 6.12). As documented for global e-waste flow or shipment, these recycling practices have resulted in large-scale environmental and human contamination (Basel Action Network 2002, Greenpeace 2008, Wong et al 2007, Bi et al 2007, Leung et al 2007, Luo et al 2008).

Recycled PBDE-containing plastics are frequently contaminated with PBDDs/PBDFs, and it has been reported that it can exceed legislative limits (Mayer et al 1993, Schlummer et al 2007). Moreover, workers in industrial countries can be exposed to high levels of brominated flame retardants and other toxic chemicals during the recycling of e-waste (including the plastic fraction) or during production of i.e. polyurethane foam (Sjödin et al 2001, Stapelton et

²⁶ Siddharth Prakash, Andreas Manhart, Yaw Amoyaw-Osei, Obed Opoku Agyekum "Socio-economic assessment and feasibility study on sustainable e-waste management in Ghana for Inspectorate of the Ministry of Housing, Spatial Planning and the Environment of the Netherlands (VROM-Inspectorate) and the Dutch Association for the Disposal of Metal, and Electrical Products (NVMP), Oeko Institute 2010, <http://www.oeko.de/oekodoc/1057/2010-105-en.pdf>

al. 2008). In industry reports it has been also concluded that mechanical recycling of such plastic waste is not recommended (Mayer et al 1993, Mark 2006).

However, it is common practise that mixed plastic fractions from electronic waste are normally not reused in electronics but are mostly “downcycled” into less demanding applications. Recent studies showing that brominated flame retardants are present in plastic household goods (Chen et al 2010), video tape casings (Hirai and Sakai 2007) and plastic children's toys. Chen et al. 2009 have revealed that such downcycling takes place in an uncontrolled manner for BFR-containing plastics. These practices dilute BFRs and chlorinated flame retardants in plastic streams, leading to unnecessary human exposure to plastic products from recycled materials. This shifting of environmental burdens from one product life cycle to another highlights the importance of focusing on environmental strategies in which prevention of environmental impacts shall be the first priority.

4.3.4 Incineration/thermal treatment of materials containing halogenated flame retardants

It should be highlighted that BFRs can be destroyed with high efficiency if BFR/HFR-containing wastes are destroyed in incinerators constructed according to best available technology (BAT) and operated according to best environmental practice (BEP) (Sakai et al 2001, Vehlow et al 2000, Weber and Kuch 2003). However, for BAT incineration, costs per tonne of incinerated material are high (in the order of \$100/t) and such facilities are too costly for treating municipal waste in developing/transition countries (Brunner and Fellner 2007).

Nonetheless, even in BAT grate incinerators (the technology applied in most municipal waste incinerators), elevated PBDD/PBDF levels were found in the bottom ashes (Wang et al 2010b), most probably due to grate shifting (particle matter falling through the grate not subjected to a complete burnout).

A large proportion of brominated flame retarded materials are combusted. Depending on the quality of combustion, high levels of brominated dioxins and furans can be formed and released as a result of the dioxin precursor properties of aromatic brominated flame retardants (Weber and Kuch 2003). In particular, open burning of e-waste is estimated to

globally generate PBDDs/PBDFs and PXDDs/PXDFs on a scale of tonnes (Zennegg et al 2009) and for many geographical areas can be considered as common practice. Areas in which such open e-waste burning has been practised for years have been transformed into PCDD/PCDF, PBDD/PBDF and PXDD/PXDF contamination sites (Li et al 2007, Yu et al 2008, Zennegg et al 2009). With regard to chlorinated PCDD/PCDF its source for the release is the presence of PVC as reported by Zennegg et al 2009.

PBDD/PBDFs are also emitted from open waste burning in industrial countries (Gullett et al 2010), other open burning practices, and other sources (Ebert and Bahadir 2003). Large amounts of brominated flame retardants in e-waste (together with car interiors) are treated in the metal industry for recovery of precious metals from e-waste (or iron and aluminium from cars and other goods). These operations can be regarded as an incomplete combustion process resulting in the emission of PBDD/PBDFs and halogenated flame retardants (Wang et al 2010a, Odabasi et al 2009).

4.3.5 Deposition and release of halogenated flame retardants from landfills

A large portion of HFR-treated products end up in landfills. This is particularly true for end-of-life treatment in developing and transition countries having no thermal waste treatment options other than open burning or other limited incineration methods not meeting BAT requirements. Even in industrialized countries, a large share of flame retarded wastes is landfilled, as was recently documented in California (Petreas and Oros 2009). It is worth highlighting that in many countries there are very few or even no waste incineration capacities at all (i.e. Australia which is an industrialised country).

There is growing evidence and concern that brominated flame retardants including POPs/PBDEs are leaching from landfills and contaminating the environment (Danon-Schaffer 2010, Odusanya et al 2009, Oliaei et al 2002, Osako et al 2004, Weber et al 2010). Significant PBDE emissions in leachates are detected from landfills in industrial countries (Danon-Schaffer 2010, Oliaei et al 2002, Osako et al 2004). Substantial concentrations of PBDEs were present in the soil adjacent to all landfills and dumpsites in various regions of

Canada (Danon-Schaffer, 2010), revealing significant POP/PBDE releases from landfills in an industrial country.

Recently, PBDE-contaminated groundwater from South African landfills has been reported (Odusanya et al 2009), indicating that new POPs/ PBDEs are present in significant amounts in the end-of-life stage in developing/transition countries and are adding to the environmental contamination generated by primitive e-waste recycling.

However, with engineered landfills with bottom liners, leachates that escape to the environment can be collected and treated to reduce the flow of contaminants to ground and surface water for some time (Osako et al 2004). Nevertheless, such treatments are expensive, and the resulting solids from adsorption of pollutants need further treatment or deposition. Because of their persistence, POPs/PBDEs will remain in landfills for many decades – and probably centuries. Over these extended time frames, landfill engineering systems, including basal and capping liners, gas and leachate collection systems, will inevitably degrade and lose their ability to contain the contaminants (Buss, Butler et al. 1995; Allen 2001, Danon-Schaffer, 2010). Therefore, land filling does not appear to be a sustainable solution for long-term containment of brominated FR-treated materials and other persistent organic pollutant-containing waste.

4.3.6 Toxicity of brominated dioxins and brominated furans, and mixed halogenated dioxins and furans

Brominated dioxins and furans have been shown to have toxicities similar to, and in some cases greater than, their chlorinated counterparts in human cell lines and mammalian species (World Health Organisation 1998; Birnbaum et al. 2003; Weber & Greim 1997; Behnisch et al. 2003; Samara et al. 2010; Olsman et al. 2007; Matsuda et al. 2010; D'Silva et al. 2004). Thymic atrophy, wasting of body mass, lethality, teratogenesis, reproductive effects, chloracne, immunotoxicity, enzyme induction, decreases in T₄ and vitamin A, and increased hepatic porphyrins have been observed in animal studies of both brominated and chlorinated dioxins and furans (WHO, 1998, Birnbaum et al 2003, Weber and Greim 1997). *In vitro* responses of brominated PBDD/DF are similar to chlorinated PCDD/PCDF, including enzyme induction, anti-oestrogen activity in human breast cancer cells, and transformation of

mouse macrophages into tumour cells and standard bio assays for dioxin-like toxicity testing (World Health Organisation 1998; Behnisch et al. 2003; Samara et al. 2010; Samara et al. 2009; Olsman et al. 2007; Matsuda et al. 2010).

In particular it should be emphasised that 2,3,7,8-Tetrabromodibenzofuran (2,3,7,8-TBDF) has a dioxin-like toxicity close to 2,3,7,8-TCDD (Matsuda et al. 2010; Behnisch et al. 2003; Samara et al. 2009; Samara et al. 2010) and is therefore far more toxic than the 2,3,7,8-TCDF. This is of particular concern because 2,3,7,8-TBDF is one of the most prevalent PBDD/PBDF congeners in human tissue (Ericson Jogsten et al. 2010) and human milk samples (Kotz et al. 2005) but can also be rather high in flame retarded plastics from WEEE (Riess et al. 2000).

Recent studies indicate that some brominated-chlorinated (PXDD/DF) dioxins/furans also have similar, and for some mixed congeners possibly greater, toxicity compared to their chlorinated homologues (Behnisch et al.2003; Mennear and Lee 1994; Birnbaum et al. 2003; Olsman et al. 2007; Matsuda et al. 2010). It is particularly notable that 2,3-Dibromo-7,8-dichlorodibenzodioxin and 2,3,7,8-T4BDD are shown to elicit up to 2.5 times the toxic response of 2,3,7,8-T4CDD, often thought to be the most toxic anthropogenic chemical. With regard to the contemporary relevance of PBDD, PBDF and brominated-chlorinated PXDD/PXDF more information is given below.

In conclusion, PBDD/PBDF and brominated-chlorinated PXDD/PXDF are of high concern similar to PCDD/PCDF and the prevention of their formation is of particular importance.

4.3.7 Contemporary relevance of PBDD, PBDF and brominated-chlorinated PXDD/PXDF

Today brominated dioxins and furans are relevant dioxin contributors in daily life as can be derived from their high contribution (about 30%) of the dioxin-like toxicity in food in the UK²⁷

²⁷ Rose, M.D. Fernandes, A.R. (2010). Are BFRs responsible for brominated dioxins and furans (PBDD/Fs) in food? BFR2010. Kyoto

(Rose et al 2010) It is notable that the EU country with the most stringent flame retardant standards (UK) report on this consistently high PBDD/PBDF and PXDD/PXDF levels in food. A direct link between these PBDD/PBDF levels and the BFR use have not been established yet.

Furthermore PBDD/PBDF were found as the major dioxin-like compounds in the first comprehensive assessment of dioxin-like compounds in house dust. The dioxin-like PBDF levels considerable exceeded the dioxin-like contribution of polychlorinated dioxins, furans and dioxin-like PCBs²⁸ (Suzuki et al 2010). While for PCDD/PCDF food exposure is for the average population the most relevant exposure pathway for the PBDD/PBDF the indoor contamination of house dust from flame retarded material can have a significant contribution or is probably the most relevant exposure pathway.

PBDD/PBDF have globally been detected in house dust^{29 30 31 32 28}(Franzblau et al. 2009; Takigami et al. 2008; Suzuki et al. 2006; Ma et al. 2009; Suzuki et al. 2010). Japan is the first

²⁸ Suzuki, G.,Someya, M. Takahashi, S, Tanabe, S. Sakai, S., Takigami, H. (2010). Dioxin-like Activity in Japanese Indoor Dusts Evaluated by Means of in Vitro Bioassay and Instrumental Analysis: Brominated Dibenzofurans Are an Important Contributor Environ. Sci. Technol.44 (21), 8330–8336

²⁹ Franzblau, A.,Demond, A.,Towey, T.,Adriaens, P.,Chang, S.C.,Luksemburg, W.,Maier, M.,Garabrant, D.,Gillespie, B.,Lepkowski, J.,Chang, C.W.,Chen, Q. Hong, B., (2009). Residences with anomalous soil concentrations of dioxinlikecompounds in two communities in Michigan, USA: a case study. Chemosphere 74(3): 395-403

³⁰ Takigami, H.,Suzuki, G.,Hirai, Y. Sakai, S.-i., (2008). Transfer of brominated flame retardants from components into dust inside television cabinets. Chemosphere 73(2): 161-169

³¹ Suzuki, G.,Nose, K.,Takigami, H.,Takahashi, S. Sakai, S.-I., (2006). PBDE and PBDD/Fs in house and office dust from Japan. Organohalogen Compounds 68

³² Ma, J.,Addink, R.,Yun, S.,Cheng, J.,Wang, W. Kannan, K., (2009). Polybrominated Dibenzo-p-dioxins/Dibenzofurans and Polybrominated Diphenyl Ethers in Soil, Vegetation, Workshop-Floor Dust, and Electronic Shredder Residue from an Electronic Waste Recycling Facility and in Soils from a Chemical Industrial Complex in Eastern China. Environ Sci Technol 43(19): 7350-7356;

country to have performed a more systematic screening of total dioxin-like compounds in house and office dust²⁸ (Suzuki et al. 2010). It was found that PBDF are the major contributors to dioxin-like toxicity in this particularly relevant exposure matrix. Suzuki showed that dioxin-like PBDF toxicity considerably exceeded the combined amount from chlorinated PCDD, PCDF and dioxin-like PCBs in the samples from 19 households and 14 offices/laboratories²⁸ (Suzuki et al. 2010).

The bio-TEQ levels in these dusts were up to 1,400 ng (median 160 ng) CALUX-TEQ/kg. These levels are three to five orders of magnitude higher than those in food samples. As there was a difference of approximately three orders of magnitude for dioxin-like PCBs and PBDD/DF concentrations in these indoor dusts Suzuki concluded that this indicated the specific source of PBDD/PBDF as the indoor environment²⁸ (Suzuki et al. 2010).

An assessment of daily intake of dioxin-like compounds for children (1-5 years) revealed a significant contribution to daily intake from dust even for a moderate intake scenario of 50 mg dust/day. A simple calculation for the highest concentration shows that a child could be ingesting 280 pg TEQ/day. Child specific exposure factors from United States Environmental Protection Agency (US-EPA)³³ show that the mean bodyweight over this age range is between approximately 11 kg at 1 year to 19.5 kg at 5 years. Consequently the daily intake can range from about 14 to over 25 pg/kg bw/day from dust alone. These exposures far exceed of the WHO and European Tolerable Daily Intakes for chlorinated dioxins. For a high dust exposure scenario of 200 mg dust/day and median dust concentrations levels the daily intake of dioxin-like compounds of children via dust exceeded the daily intake of TEQ from PCDD/PCDF via food³⁴ (Suzuki et al. 2007):

³³ Environmental Protection Agency (US), Child-Specific Exposure Factors 2002,

<http://cfpub.epa.gov/ncea/cfm/recordisplay.cfm?deid=199243#Download>

³⁴ Suzuki, G., Takigami, H., Nose, K., Takahashi, S., Asari, M. Sakai, S.i., (2007). Dioxin-Like and Transthyretin-Binding Compounds in Indoor Dusts Collected from Japan: Average Daily Dose and Possible Implications for Children. *Environ. Sci. Technol.* 41(4): 1487-1493

4.3.8 Conclusion on assessment of end-of-life treatment of flame retardant containing electronics (and other flame retarded materials)

In conclusion, the end-of-life management of imaging equipment, in which brominated aromatic substances are used in plastics, entails health and environmental risks.

Plastic containing brominated aromatic substances has a negative influence on the recycling of imaging equipment as the plastic fraction containing BFR needs to be removed from any separately collected WEEE and disposed of or recovered with specific requirements based on the provisions of Directive 2002/96/EC on waste electrical and electronic equipment (WEEE). Difficulties on WEEE are presented in more detail in Annex 6.12.3.

A large proportion of brominated flame retarded materials are combusted. Depending on the quality of combustion, high levels of brominated dioxins and furans can be formed and released as a result of the dioxin precursor properties of aromatic brominated flame retardants. In particular, open burning of e-waste is estimated to globally generate PBDD/PBDFs and PXDD/PXDFs on a scale of tons and for many geographical areas can be considered as common practice (see also section 4.3.4). The toxicity and environmental concerns related to dioxins and furans are high (see section 4.3.6). Brominated flame retardants in plastics can be destroyed with high efficiency only if the plastics are treated in incinerators constructed and operating according to best available technology (BAT) and best environmental practices (BEP). However, in this case costs per ton of incinerated material are considered high (in the order of \$100/t).

Additionally, a large portion of Brominated FR-treated products end up in landfills and there is growing evidence and concern that brominated flame retardants including POPs/PBDEs are leaching from landfills and contaminating the environment in industrial countries as well as in developing/transition countries. Only in engineered landfills with bottom liners, leachates that escape to the environment can be collected and treated to reduce the flow of contaminants to ground and surface water for some time but such treatments are expensive and not state-of-the art. Because of their persistence, POPs/PBDEs will remain in landfills for many decades – and probably centuries and are expected to be eventually released to the

environment as the landfill engineering systems (including basal and capping liners, gas and leachate collection systems) will inevitably degrade and lose their ability to contain the contaminants. Therefore, land filling does not appear to be a sustainable solution for long-term containment of brominated FR-treated materials (as presented in section 4.3.5).

The praxis showed that these substances can be avoided and resulting to avoidance of their associated environmental impacts.

4.3.9 Reference list for brominated flame retardants

The references used in Annex 6.11 regarding brominated flame retardants are presented in Table 5.

Table 5 Reference list for brominated flame retardants

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4.3.10 Information on waste streams for imaging equipment³⁵

Information and data regarding the waste stream of imaging equipment follows.

Imaging equipment originating from households in EU-27 shall be collected following the provisions of WEEE Directive. In this frame imaging equipment waste streams are reported and statistics are kept in Eurostat. In these statistics imaging equipment are documented together with other IT equipment i.e. computers and laptops. A significant amount of electronic equipment which is primarily sold and used in EU-27 is exported as second hand products from EU-27 to third countries mainly from Africa and Asia. From a life cycle perspective the investigation of the environmental impacts of end-of-life phase of these products shall also be captured. In life cycle assessments the boundaries of the product life cycle system has no restrictions based on the geographical origin in which the environmental impacts may occur. Further, numerous environmental considerations are associated with the end-of-life management of the imaging equipment whose are shipped out from EU-27 as in the destination countries the recycling facilities, the thermal treatment (i.e. uncontrolled burning) or the land filing does not meet the European health and environmental standards.

4.3.11 Waste streams within the geographical area of EU-27

The WEEE Directive currently sets a minimum collection target of 4 kg per year per inhabitant for WEEE from private households. This target was originally based on estimates made by the EU Priority Waste Stream project group that future quantities of WEEE are expected to be over 20 kg per person per year, of which the consumer sector accounts for 12 kg, the industrial sector for 5 kg, and the cables sector for 3 kg. No collection target was set for non-household WEEE.

Based on the study on WEEE of 2008 for DG ENV³⁵, currently in Western Europe the amount of WEEE produced per person is estimated to be higher than in the new Member States in which the amounts are substantially lower, but expected to rise in the future. A general characteristic of current collection rates is that these are, in the majority of Member States, far below 100% of the goods sold many years ago. Increasing collection is therefore one of the key issues to enhance the effectiveness of WEEE and to achieve the original intent of the Directive.

The new estimate³⁵ of the expected waste of electric and electronic equipment (WEEE) arising across the EU27 is between 8.3 and 9.1 million tonnes per year for 2005. This increase is due to expansion of the EU, growth in the number of households and inclusion of items that may have been excluded previously (B2B). Forecasts models predict that by 2020, total WEEE arisings will grow annually between 2.5% and 2.7% reaching about 12.3 million tonnes. Out of them 8.0% is allocated to the category of IT and Telecom excl. CRT's, in which imaging equipment are classified together with computers which makes 664 to 828 thousand tonnes for 2005 and 984 thousand tonnes in 2020. The estimate for imaging equipment is 268 thousand tonnes for 2008 as given in Table 6. These values are higher than the collected volumes reported in Eurostat (given here in Table 7-Annex 6.10.) which reach just in 2008 the 539,833 tonnes.

Table 6 Imaging equipment placed in the EU25 expressed in weight in tonnes³⁵

Weight (tonnes)	2004	2005	2006	2007	2008
B&W laser printers	56,310	55,230	55,710	55,425	55,635
Colour laser printers	20,070	25,020	26,190	29,310	30,570
B&W copiers	50,950	54,000	52,000	50,500	47,500
Colour copiers	13,700	14,300	16,300	17,200	17,900
Inkjet printers & MFDs	109,010	112,185	113,575	115,435	116,780
Total	250,040	260,735	263,775	267,870	268,385

³⁵ Huisman, Jaco, Delgado Clara, Magalini Federico, Kuehr Ruediger, Maurer, Claudia Artim, Eniko Szlezak, Josef Ogilvie, Poll Jim, Steve Abs, final Report for DG ENV, 2008 Review of Directive 2002/96 on Waste Electrical and Electronic Equipment (WEEE), United Nations University, 2008

Table 7 Eurostat data for WEEE collected for IT and telecommunications equipment (including imaging equipment) in tonnes

IT and telecommunications equipment (including imaging equipment) waste collected (in tonnes per year

GEO/TIME	2005	2006	2007	2008
Belgium	10,673.94	12,155.19	14,372.85	16,284.71
Bulgaria	:	:	3,835.72	4,553.71
Czech Republic	:	:	7,270.9	9,784.1
Denmark	:	11,380	17,043	16,507
Germany (including former GDR from 1991)	:	102,336	117,749	155,007
Estonia	:	687.04	870.71	970.7
Ireland	:	:	11,163	9,599
Greece	449	1,001	2,981.53	5,941.94
Spain	:	:	14,406	17,019
France	:	8,540	28,574	47,766
Italy	:	:	:	:
Cyprus	263.6	856.9	127.62	290.11
Latvia	:	:	800.78	853.48
Lithuania	288	1,072	1,575	1,415
Luxembourg	:	570.18	852.94	827.51
Hungary	1,242	2,377.8	3,472.5	5,719.1
Netherlands	16,838	18,440	21,049	23,069
Austria	3,148.2	11,365.5	13,212	15,342.8
Poland	:	3,067.3	8,714.74	14,948.73
Portugal	:	1,175.65	3,238.15	11,937
Romania	:	274.24	1,164.32	6,252.69

Slovenia	:	:	:	:
Slovakia	285.65	857.9	2,101.01	2,532
Finland	2,841	7,012	10,375	11,647
Sweden	9,440	25,174	29,782	29,556
United Kingdom	:	:	55,831.34	132,009.89
Malta	:	:	:	:
EU-27 Total	45,469	208,343	370,563	539,833

: not available

The EU15 Member States' average collection performance is roughly half that of Switzerland and Norway. This is mainly due to lower performance in the collection of categories other than the category referring to large household appliances. Although the WEEE Directive collection target can be easily met by EU15 Member States, it remains a very challenging target for the New Member States³⁵.

The estimated amount of WEEE currently collected and treated as a percentage of the amounts of WEEE arising for the EU27 in 2005 for the category in which imaging equipment are covered is 27.8%. There is no information available on differences between imaging equipment and computers as these are not differentiated in Eurostat categories but it is suggested that there shall be no large differences.

This means that for computers and imaging equipment there is substantial room for improvement (up to 70 %) of the waste stream covered by WEEE Directive. Further, it is important to highlight that there were identified large differences in performance by different Member States per sub-category which indicates that there is much room for improvement in collection performance³⁵.

Regarding impacts of WEEE for metals there appear to be no major difficulties concerning the recovery and recycling for metals. On the contrary for plastics data from literature seems to confirm that at present plastic output streams from WEEE recycling operations are mostly

not recovered, but are landfilled together with other residue streams, as opposed to the apparent preference for the recycling option.

Annex II of WEEE Directive requires that plastics containing brominated flame retardants are removed from any separately collected WEEE and are disposed of or recovered in compliance with Article 4 of Council Directive 75/442/EEC on Waste.

It is suggested that the removal obligation reduces the amount of plastics available for recycling and hinders the meeting of recycling targets in some plastic dominated WEEE categories. Further, recycling of BFR plastics into non electronic and electric applications (houseware, automotive, building...) can cause dispersion of additives into other diverse streams, which could be interpreted as against the principle that recovery should ensure that pollutants are not transferred into products and minimises the formation, transfer and dispersion of hazardous substances in the process.

4.3.12 Waste streams outside the geographical area of EU-27

It is considered that a high number of EEE including imaging equipment is exported from EU27 as reused products. This way the waste of these products is not handled and treated in EU27 (in which WEEE Directive is applicable) but in the destination countries in which often lower health and environmental standards are applied along the end-of-life phase of the products. Trade statistics to non EU Members embrace a part of the actual exports. However, statistics do not differentiate between used and new goods. Thus, capturing the actual volume of these products is difficult. In Eurostat exports outside EU27 of WEEE are reported to be either zero or in marginal amounts compared with the overall arising of waste of EEE.

As this issue has gained awareness a study for the German Federal Environmental Agency regarding the trans-boundary shipment of waste electrical and electronic equipment /

electronic scrap was conducted in 2010 ³⁶. This report describes approaches, measures and regulation structures for the export of used electrical/electronic equipment and waste electrical/electronic equipment to non-EU countries with aim to optimise the protection of the environment and resource flows. Volumes on used imaging equipment shipped in 2008 from the port of Hamburg to Ghana, Nigeria, South Africa, Vietnam, Philippines and India.

The export of the equipment types investigated in non-EU countries took place exclusively as used equipment. However, the value of these products which are exported for reuse lies significantly below the value compared with the respective items exported to Member States. Notified waste exports of such equipment is not known. In general the equipment originates from a multiplicity of sources (in all >4,000 sources), in part from private end users, in part from commercial sources and partially from the waste regime.

Private collection and reloading points represent one of the most important pivotal points for the export of low-value equipment. In the countries of destination, the equipment encounters recovery and disposal structures, which are not suitable to ensure the protection of human health and the environment as well as the extensive recovery of resources. Through this, several hundred of kilos of precious metals and rare earths are lost from the economic cycle.

The total quantity of the exports extrapolated within the framework of the investigation from German Federal Agency in 2008 lay between 93,000 t and 216,000 t. In the countries of destination, the equipment encounters recovery and disposal structures, which are not suitable to ensure the protection of human health and the environment as well as the extensive recovery of resources.

As found in Table 8 for imaging equipment the largest amount 5.154 tonnes was shipped from Hamburg to South Africa while 2.875 tonnes to India, 754 tonnes to Vietnam 722 tonnes to Nigeria and lower amounts to Philippines and Ghana 178 tonnes and 106 tonnes respectively. In total the amount of imaging equipment shipped as reused items towards

³⁶ Knut Sander, Stephanie Schilling for the Federal Environment Agency (Germany), "Transboundary shipment of waste electrical and electronic equipment / electronic scrap – Optimization of material flows and control", Ökopol GmbH, 2010
http://www.umweltbundesamt.de/uba-info-medien/mysql_medien.php?anfrage=Kennummer&Suchwort=3933

countries in which the end-of-life facilities are not meeting the health and environmental standards of EU in one year and only from the port of Hamburg was 9,789 tonnes.

Table 8 Export from Germany to select countries of destination (reference year 2007)³⁶

Goods code	Plaintext	Ghana	Nigeria	South Africa	Vietnam	Philippines	India
		Weight in t					
WA8415	Air conditioning units	33	32	778	83	46	525
WA8418	Refrigerators, freezers, heat pumps	64	312	791	169	24	517
WA8443	Printing machines and accessories for printing machines	106	722	5.154	754	178	2.875
WA8450	Machines for washing or drying clothes	22	19	358	19	27	49
WA8469	Typewriters, word processing machines	1	-	-	0	1	1
WA8471	Automatic data processing machines	56	152	782	170	24	108
WA8510	Razors, shears with electric motor	0	-	25	-	0	1
WA8516	Electric hot water heaters and immersion heaters	20	56	1.106	3	85	68
WA8517	Telephone sets, telecommunication equipment	10	478	1.234	116	34	1.737
WA8521	Audio/video recording equipment	1	60	6	0	0	0
WA8525	Transmission equipment for broadcast etc., television cameras	1	56	64	24	3	25
WA8527	Receiving equipment for radiotelephone traffic or broadcast	1	4	149	0	0	10
WA8528	Television sets, video monitors	153	787	145	802	1	129
Total		468	2.678	10.592	2.140	423	6.045

[UESTATIS 2008]

4.3.13 Difficulties of plastics recycling from WEEE

Number of facilities which can separate plastics in the EU including separation of BFR plastic

Whilst there are several WEEE recycling plants in Europe using 'state-of-the-art' technologies for the separation of plastics, there are still only a handful of facilities for comprehensive treatment integrating plastic separation and recycling, and only a very few

plants separate BFR-containing plastic³⁷. Most WEEE recycling companies dismantle and sort equipment into various streams and then pass the plastic rich fraction to other specialised operators. Some aim for the recovery of residual metals in these polymer-rich fractions, others recycle parts of the plastic fraction, a third group recovers energy and a proportion of WEEE plastics are sold as mixed plastic for export - mainly to China³⁷. Only about 8 % of plastic from WEEE is recycled^{38,39}.

Reasons for the limited recycling of WEEE plastic

The four main reasons for the limited number of approaches for recycling plastics from WEEE are included below³⁷:

a) Industry using secondary plastic materials has tight specifications in relation to polymer quality, both chemically (RoHS compliance) and with respect to material properties. This is also used as an argument to depress the prices of recycle thus increasing the economic challenges.

b) WEEE plastics contain at least 15 different plastic types (Dimitrakakis et al. 2009, UNEP 2011). The efficient sorting of this mixture presents difficult technical challenges and a degree of cross-contamination is inevitable in practice. According to Dimitrakakis³⁹ three polymers (ABS, PS, PP) account for between 70 % and 85 % of total while other studies estimate that this fraction is a bit lower at 50-70%.

c) WEEE plastics contain RoHS listed BFRs (PentaBDE, OctaBDE, DecaBDE; but normally no PBB which is also listed). Potential customers are therefore risk adverse to contaminated recycle.

d) Production of larger volumes with identical properties and performance, as required by many major manufacturers, requires consistent quality and composition of inputs.

Japan has already established a Japanese Industrial Standard (JIS) for plastics for use in electric home appliances, "*marking for identification of plastic parts for electrical and*

³⁷ UNEP (2010) Technical review of the implications of recycling commercial penta and octabromodiphenyl ethers. Stockholm Convention document for 6th POP Reviewing Committee meeting (UNEP/POPS/POPRC.6/INF/6) Geneva 11-15. October 2010.

³⁸ UNEP (2011). Guidance on the Best Available Techniques (BAT) and Best Environmental Practice (BEP) for recycling and waste disposal of articles containing POP-PBDEs. Draft 10/2011.

electronic equipment (C9912).” This standard requires the marking of plastic parts such as flame retardants (FR), recycled plastics and dismantling assistance. In particular, the marking system includes plastics already recycled by ‘closed-loop recycling’ (i.e. recycling within the same product group). Target recycling rates for different electronic categories have also been set⁴⁰ (Aizawa et al. 2010).

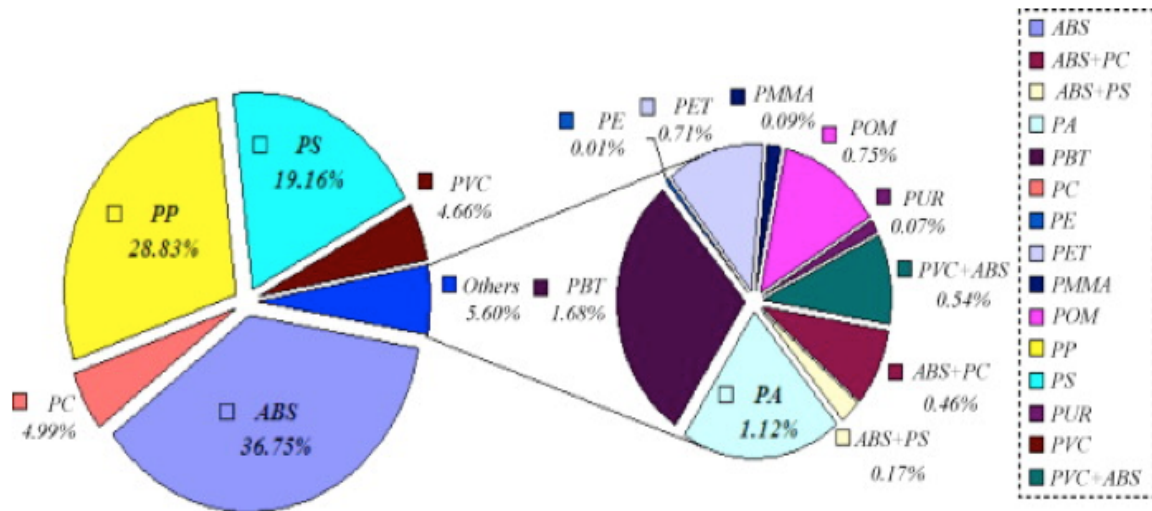


Figure 1: Polymer types identified in WEEE plastic samples (% w/w)³⁹

³⁹ Dimitrakakis E., Janz A., Bilitewski B. Gidaracos E. (2009) Small WEEE: Determining recyclables and hazardous substances in plastics. *Journal of Hazardous Materials* 161(2-3): 913-919

⁴⁰ Aizawa H., Hirai Y., Sakai S.-I. (2010) Development of Japanese Recycling Policy for Electric Home Appliances by the Addition of Plastics Recycling. *BFR2010: 5th International Symposium on Brominated Flame Retardants*. 7.-9. April 2010, Kyoto/Japan

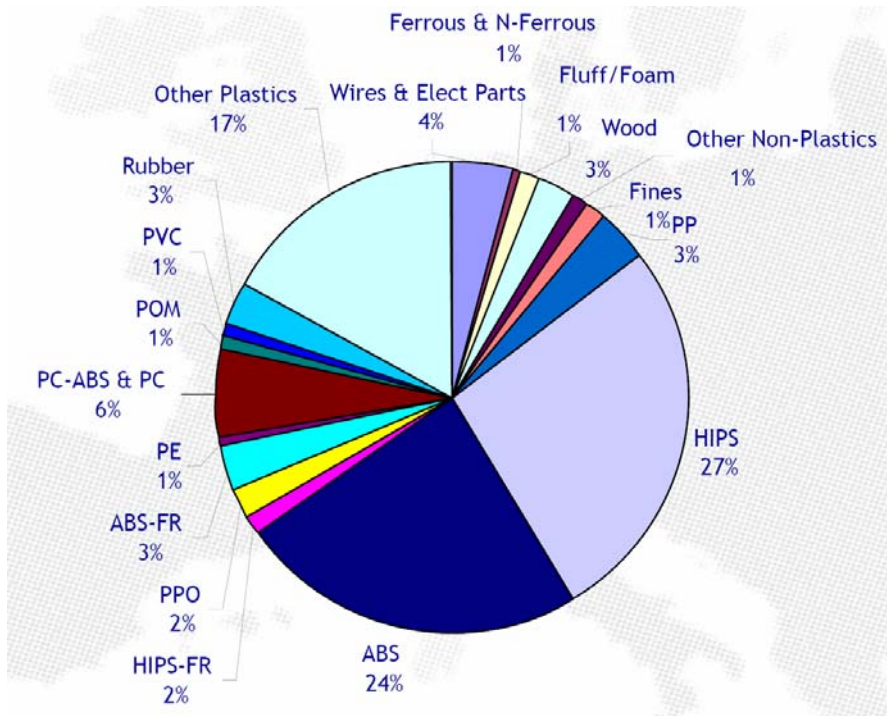


Figure 2: Polymer types and other materials identified in WEEE plastic samples (% w/w)³⁸

4.4 Information regarding the end-of-life considerations of PVC

A key concern regarding PVC is at end-of-life management. The management of PVC waste should be assessed in the context of the European waste management policy. The Communication from the Commission on the review of the Community strategy for waste management⁴⁵ has confirmed *“the hierarchy of principles that prevention of waste shall remain the first priority, followed by recovery and finally by the safe disposal of waste.”* It is further stated that *“preference should be given, where environmentally sound, to the recovery of material over energy recovery operations. This general rule is based on the fact that material recovery has a greater effect on waste prevention than energy recovery. It will nevertheless be necessary to take into account the environmental, economic, and scientific effects of either option. The evaluation of these effects could lead, in certain cases, to preference being given to the energy recovery option.”* In its Resolution⁴⁶ of 24 February 1997, the Council endorsed this hierarchy of principles.

The three main options for end of life management of materials containing PVC are: (1) reuse/recycling (2) incineration/thermal treatment (3) deposition. Management of end-of-life waste flows can vary widely based on the type of waste management schemes used in various countries and the different materials involved (PVC in mixed plastic fraction; PVC as; PVC in hospital wastes).

4.4.1 Mechanical recycling of PVC and PVC containing materials

Mechanical recycling refers to recycling processes where PVC waste is treated only mechanically, mainly through shredding, sieving, and grinding. From a life cycle perspective, the preferred method for end-of-life treatment of PVC and PVC containing wastes would be mechanical recycling because it reduces the energy-intensive production of new materials.

A number of life cycle assessments⁴⁹ on some specific PVC products have shown that mechanical recycling provides an environmental advantage for production waste, cut-offs and post-consumer PVC waste, which can be separated. The environmental advantages of

the down-cycling of mixed plastics for the production of products which substitute concrete, wood or other non-plastic applications are less certain.

However, the presence of additives classified as hazardous, such as lead, cadmium and PCB, in large PVC waste streams, raises specific issues during their potential recycling. The recycling of PVC waste containing heavy metals results in a dilution of these substances in a greater quantity of PVC, since it is necessary to add virgin material. The heavy metals are not directly released into the environment during the recycling process and the renewed service life. The recycling of PVC material containing these heavy metals postpones the final disposal to a later stage. Due to the product-specific additives formulations, recyclers would prefer recycling into similar applications.

A prohibition of the recycling of PVC waste containing heavy metals would eliminate the mechanical recycling of post-consumer PVC wastes from building applications - the waste stream with the highest potential for high-quality recycling - as they virtually all contain lead or cadmium. It should be noted that, except for Denmark, Member States, which have banned the use of cadmium as stabilisers, allow the recycling of PVC waste containing cadmium.

The problem of PCBs in PVC cable waste has been addressed in Directive EC/96/59 on the disposal of PCB and PCT, which states that cables containing more than 50 ppm of PCBs are considered PCBs and therefore have to be decontaminated or disposed of in accordance with the provisions established under this Directive.

The resulting recyclates (in powder form) can be processed into new products. Depending on the degree of contamination and the composition of the collected material, the quality of the PVC recyclates can vary to a large degree. The quality of the recyclates determines the degree to which virgin material can be substituted by recyclates: "high-quality" recyclates can be reused in the same types of PVC applications, whereas "low-quality" recyclates from mixed waste fractions can only be "down-cycled" into products usually made from other material (EC 2000).

In the baseline scenario presented in the PVC green paper about 9 % of the total PVC waste could be mechanically recycled in 2010 and 2020, representing about 400 000 tonnes of PVC waste in 2010 and 550 000 tonnes in 2020 (EC 2000, Prognos 2000). Compared to this baseline scenario, maximum recycling potentials, which represent the PVC quantities which can be recycled (taking into account the technical and economic limits of PVC) recycling have been estimated by Prognos in 2000 (EC 2000). According to this scenario, the potential for post-consumer waste was estimated to about 800 000 tonnes in 2010.

However the recycling quota of PVC today demonstrates that PVC recycling is still a major challenge. According to the estimates of the Vinyl Institute *"it is anticipated that the total recycled will reach up to 200 000 tons of PVC waste in 2010"*⁴¹ only 4 % of the PVC waste amount for 2010 will be recycled.

The largest part of the remaining 96 % of the PVC wastes goes to landfills and some of the waste into incineration. The suggested recycled amount would therefore be only 50 % of the baseline scenario and only 25 % of the estimated maximum recycling potential projection from the Commission in the 'Green Paper – Environmental Issues of PVC' (EC 2000). The Vinyl Institute states that this small recycling quota would only be reached with governmental support *'For this recycling volume to be reached there is a need for support from public authorities to create and organise appropriate waste collection schemes'* (Vinyl 2010). This means that governmental support in collection (which is the most expensive part of recycling) is a precondition even for a 4 % recycling quota.

According to the EC Green Paper, the recycling of PVC is also limited by the overall recycling costs (EU 2000) which is probably the main driving force for the low recycling quota. Economic profitability is reached when the net recycling costs (i.e. the overall costs for collection, separation and processing minus the revenues from sale of the recyclates) are lower than the prices for alternative waste management routes for related PVC wastes. The EC further states that *'If economic profitability cannot be reached, the recycling of PVC waste will not take place under free market conditions, unless there are legal obligations or*

⁴¹ Vinyl 2010, Reporting of the activities of the year 2010 and summarising the key milestones of the past 10 years, The European PVC Industry's Sustainable Development Programme, http://www.vinyl2010.org/images/progress_report/2011/vinyl2010_progress_report_2011_final.pdf

voluntary measures enforcing or promoting the recycling of PVC. Collection represents the major bottleneck regarding the availability of waste and costs.'

For PVC in electronics like imaging equipment, the sorting cost would add to these expenses. Cable insulation is the only post-consumer waste which can be recycled at competitive costs, due to the presence of valuable metals, such as copper (EC 2010). High-quality mechanical recycling of PVC for post-consumer wastes is considered to be in a preliminary stage and in 2000 existed only for few product groups and with low quantities (about 3.600 tonnes of rigid profiles, 5,500 tonnes of PVC pipes and 550 tonnes of flooring) (EC 2000).

In respect of the use of PVC in articles which finally generate a mixed plastic fraction (like electronics including imaging equipment), the EC paper concludes that PVC can have a negative influence on the recycling of other plastics in mixed plastic waste (EC 2010): *'When PVC is processed with other plastics, such as in the packaging waste stream, the processing temperature is limited to the range of PVC-processing, which is a relatively low range compared to other plastics. Due to similar densities, polyethylene terephthalate (PET) and PVC waste are difficult to separate and the presence of PVC puts additional costs on some PET recycling schemes such as the PET bottles.'* (EC 2010). This is an important consideration for restriction of PVC in electronics such as imaging equipment.

4.4.2 Chemical recycling of PVC and PVC containing materials

Chemical recycling denotes a number of processes, by which the polymer molecules that constitute plastic materials are broken up into smaller molecules. These can either be monomers that can be used directly to produce new polymers or other substances that can be used elsewhere as starting materials in processes of the basic chemical industry.

In the case of PVC, in addition to the breaking up of the backbone of the polymer molecules, the chlorine attached to the chains is set free in the form of hydrogen chloride (HCl). Depending on process technology, HCl can be reused after purification or has to be neutralised to form various products that can be used or have to be disposed of (EC 2000).

For 2010, the total PVC waste quantities which could be chemically recycled in the baseline scenario, were estimated to about 80 000 tonnes as a fraction in mixed plastics waste with low chlorine content (mostly from packaging) and about 160 000 tonnes in mixed plastic fractions with higher PVC content, mostly from automotive and electric and electronic waste (EC 2000).

4.4.3 Incineration/thermal treatment of PVC

4.4.3.1 Negative impact of PVC in cement kilns

PVC has a crucial negative impact on the incineration and thermal recovery of polymer waste in cement kilns. PVC (and other halogenated material) limits or even restricts the use of thermal recycling in cement plants which normally accept polymer waste as secondary fuel up to a chlorine content of 0.5 %. In the best available techniques reference documents on cement, lime and magnesium oxide manufacturing industries⁴² is reported in the BAT chapter that in order to reduce HCl emission the producer has to use a) raw materials and fuels containing a low chlorine content, b) limit the amount of chlorine content for any waste that is to be used as raw material and/or fuel in a cement kiln. Moreover, cement quality composition specification included limitation of chlorine content due to corrosion problems that may occur in the concrete reinforcement.

4.4.3.2 Negative impact of PVC in waste incinerators

An assessment of the quantities of flue gas cleaning residues resulting from the incineration of PVC waste concluded that the incineration of 1 kg of PVC generates on average⁵⁹ between 1 and 1.4 kg of residues for the dry process with lime, semi-dry and semi-wet wet processes (Bertin technology 2000). With the use of sodium hydrogen-carbonate as a

⁴² European Commission, Best available techniques reference documents on cement, lime and magnesium oxide manufacturing industries, Joint Research Centre, Institute for Prospective Technological Studies, IPPC bureau, 2010

neutralisation agent in semi-dry process, 1 kg of PVC generates about 0.8 kg of residue. In case of wet processes, between 0.4 and 0.9 kg of liquid effluent is generated⁴³.

These flue gas cleaning residues are classified as hazardous waste (EC 1994). The residues are generated separately (in particular in semi-wet and wet systems) or mixed with fly ash. The residues contain the neutralisation salts, the excess neutralisation agent as well as pollutants such as heavy metals and dioxins that were not destroyed. Landfilling of the residues is, with some exceptions, the only option used within the Member States. Several processes have been devised to recover calcium chloride and sodium chloride from the residues of the dry and semi-dry processes, but few of them are currently used commercially. The EC highlighted that these technologies are “end of the pipe” solutions, less preferable than a preventive measure aimed at reducing at source the quantity of residues generated (EC 2000).

Therefore - for the treatment in BAT incineration, the minimization of PVC input should be considered.

In summary, PVC at in the municipal solid waste stream has the following effect on the flue gas cleaning residues in comparison to incineration of municipal solid waste without PVC⁶³:

- PVC incineration contributes to an increase in the quantity of flue gas cleaning residues (about 37 % for the dry systems, 34 % for semi-dry systems and 42 % for semi-wet wet 64);

⁴³ . There is a difference between the amounts of neutralisation agent required and residues produced between soft and rigid PVC. Flexible PVC contains less chlorine than rigid PVC. The amounts of neutralisation agents required and of residues generated are therefore lower for flexible PVC than for rigid PVC (1 kg of soft PVC generates between 0.5 and 0.78 kg of residues) (Bertin Technology 2000).

- PVC incineration contributes to an increase in the content of leachable salts in the residues by a factor of two. These are primarily chlorides of calcium, sodium, and potassium;
- The incineration of PVC increases the amount of leachates from the residues put into landfill (about 19 % for dry systems, 18 % for semi-dry systems, 15 % for the semi-wet wet systems and 4 % for wet systems). The leachates need to be treated prior to any discharge.

Furthermore PVC waste incineration increases the operating costs of the incinerators due to the use of neutralisation agents to neutralise the acid flue gas, and adds additional costs for the waste management of the resulting residues. Up to EUR 300 per tonne for dry systems (Bertini et al 2000). These additional costs are not borne specifically by new PVC products or by PVC waste, but are included in the overall incineration cost of waste. A reduction of PVC waste would have a positive effect on this.

4.4.3.3 PVC and PCDD/F formation in thermal processes

The question on the role of PVC in the formation and release of PCDD/PCDF in combustion raised awareness since many years now (many references go back to 1999). A similar situation exists for combustion of PVC-containing materials as described for brominated flame retardants in Table 7.

State of the art municipal waste incinerators can treat PVC containing waste (up to 1 % chlorine) and hazardous waste incinerators (wastes above 1 % chlorine) without a significant increase in PCDD/PCDF formation, since the limiting parameter for PCDD/PCDF de novo synthesis in the cooling zone is not chlorine (which in fly ashes is about 10%), but carbon (which in BAT incineration ash is at levels well below 1 % or even 0.1 %).

However for small scale incinerators and non-BAT incinerators with lower combustion efficiency, carbon levels (products of incomplete combustion) are high, and the chlorine content (with PVC as main contributor) is the determining factor for PCDD/PCDF formation and release. PCDD/PCDF formation and release has a strong correlation to the PVC content, which is the major driver for high PCDD/PCDF emission levels (Ibashi 2011).

PVC products disposed of in landfills contribute to the formation of dioxins and furans during accidental landfill fires (EC 2000). The release of PCDD/PCDF from landfill fires and open burning is one of major PCDD/PCDF sources in the national inventories established under the Stockholm Convention. Of particular interest in respect to EEE (including imaging equipment) is the high release of PCDD/PCDF in primitive WEEE recycling in developing countries: it has been shown for e-waste recycling sites in China that PVC was the main source of PCDD/PCDF environmental contamination, while the brominated flame retardants were the main source for the brominated and chlorinated-brominated PXDD/PXDF (Zennegg et al 2009). Since the EU is party to the Stockholm Convention, unintentional POP releases need to be minimised, and the substitution approach applied. Similar considerations were made in the development of the criteria establishment by the Nordic Swan⁴⁴.

4.4.4 Deposition of PVC in landfills

Deposition in landfills is the most common waste management route for PVC waste. It can be estimated that several tens of million tonnes of PVC waste have already been sent to landfills during the past 30 years.

All materials in landfills including PVC are subject to different reactive conditions, which are determined by parameters such as temperature, moisture, presence of oxygen, activity of micro-organisms and the interactions between parameters at different stages of the ageing process of landfills⁴⁵.

⁴⁴ Nordic Swan (2007): "The Nordic countries are bound by the Stockholm Convention on Persistent Organic Pollutants (POPs) to omit waste containing POPs at source so that POPs are destroyed or pacified without impacting the environment (www.pops.int). The convention considers the incineration of PVC, for example, to be a source of POPs, in particular dioxins". (Nordic Swan 2007) "Furthermore, there is always a risk of POP emissions (dioxins) from uncontrolled fires that involve PVC and other materials containing chlorine, e.g. houses, hotels and hospitals". (Nordic Swan 2007).

⁴⁵ Four main phases can be distinguished: short initial aerobic phase, anaerobic acidogenic phase (variable duration, longer than aerobic phase), anaerobic methanogenic phase (up to several centuries), final aerobic phase.

Investigations have been carried out on both rigid and soft PVC samples mainly through laboratory equipment studies, examination of the effects of biological treatment, and of microbiological tests (Argus 1999).

The PVC polymer is generally regarded as being resistant under soil-buried and landfill conditions (Mersiovski et al 1999). However, an attack on the PVC polymer of a thin packaging foil has been detected (Argus 1999). This remains an isolated result and the attack was observed under aerobic conditions and at 80°C, conditions which, if they occur in landfills, are transient.

Losses of plasticisers, especially phthalates, from flexible PVC are widely recognised in the literature (EC 2000). Results from studies on the degradability of phthalates under landfill conditions show that degradation of phthalates occurs, but may not be complete depending on conditions and type of phthalate. Both phthalates and their degradation substances can be detected in landfill leachates (EC 2000). In addition, long-chain phthalates, such as DEHPs, are only partly degraded in usual leachate and sewage treatment plants and accumulate on suspended solids. Losses of phthalates could also contribute to gaseous emissions from landfills. As for other emissions from landfills, emissions resulting from the presence of PVC in landfills can last longer than the guarantee of the technical barrier, and there is no evidence that the release of phthalates will end after a given period of time.

A study into the long-term behaviour of PVC waste under landfill conditions showed a release of lead stabiliser from flexible PVC waste (Mersiowski et al 1999). Stabilisers in rigid PVC waste are more encapsulated in the matrix. Hence, migration is expected to be lower in rigid PVC and would mainly affect the surface of the PVC. The effect on the bulk of the material is uncertain.

As mentioned above, PVC products disposed of in landfills contribute to the formation of dioxins and furans during accidental landfill fires (EC 2000). The release of PCDD/PCDF from landfill fires and open burning is one of major PCDD/PCDF sources in the national inventories established under the Stockholm Convention.

4.4.5 Conclusion on the assessment of end-of-life treatment of PVC

The PVC recycling quota is very low. Based on the prediction of vinyl2010⁴⁶ the recycling quota of PVC would be only 4% of the total PVC waste amount for 2010. This recycled amount would be only 50 % of the baseline scenario estimated from the Commission in the “Green Paper – Environmental Issues of PVC”. Even for this small recycling quota, governmental support in collection (which is the most expensive part of recycling) is a precondition.

PVC has a negative influence on the recycling of other plastics such as the recycling of mixed plastic waste (i.e. imaging equipment). As stated in the Green Paper for PVC, the processing temperature is limited to the range of PVC processing, which is a relatively low range compared to other plastics.

PVC (together with other chlorine containing polymers) has crucial negative impact on the incineration and thermal recovery of polymer waste in cement kilns. The presence of PVCs in mixed plastic waste fractions restrict the use of PVC containing waste as fuel in cement plants which normally accept polymer waste up to a chlorine content of 0.5%.

PVC has a negative impact in incineration plants which operate under BAT conditions, due to the high costs of the treatment of flue gas cleaning residues which are produced in increased volumes when PVC is present. Flue gas cleaning residues are classified as hazardous waste and their treatment is associated with high costs which are directly linked to the operational costs of the incinerator. As highlighted in the EC Green Paper for PVC, a preventive

⁴⁶ Vinyl 2010 is the voluntary programme on Sustainable Development by the PVC industry. Vinyl 2010 represents the whole PVC industry chain. Its four founding members are: ECVM (the European Council of Vinyl Manufacturers), representing the 13 European PVC resin producing companies which account for almost 100% of the current total EU-27 PVC resin production, ESPA (the European Stabilisers Producers Association), representing 11 companies which produce more than 98% of the stabilisers sold in Europe, ECPI (the European Council for Plasticisers and Intermediates), representing the seven major European plasticiser and intermediate producers, EuPC (the European Plastics Converters) represents close to 50,000 companies in Europe, producing over 45 million tonnes of plastic products of various types every year.

measure aimed at reducing the quantity of residues generated at source is more preferable than the treatments of them afterwards.

PVC-containing waste is associated with the formation of dioxins and furans in thermal processes with insufficient combustion conditions i.e. incinerators operating on non-BAT conditions, uncontrolled burning, accidental fires etc.

Deposition in landfills is the most common waste management route for PVC. Degradation of plasticisers used in PVC occurs in landfills, resulting to emissions both to leachates and to air. Environmental impacts are related to the release of these substances. Emissions from PVC can last longer than the guarantee of the technical barrier used in landfills.

4.4.6 Reference list related to the end-of-life environmental consequences of PVC⁴⁷

AEA Technology, Economic evaluation of PVC waste management, a report produced for the European Commission Environment Directorate-General, June 2000.

Argus in association with University Rostock (1999) ,The Behaviour of PVC in Landfill, Study for DG ENV,

Bertin Technologies, The influence of PVC on quantity and hazardousness of flue gas residues from incineration, Study for DG XI, April 2000

European Commission (2000) GREEN PAPER Environmental issues of PVC; COM (2000) 469 final; Brussels, 26.7.2000.

Mersiowski et al. 1999, Behaviour of PVC in landfills, ECVM, Technical University Hamburg-Harburg

Prognos, Mechanical recycling of PVC wastes, Study for DG XI, January 2000

Vinyl (2010) REPORTING ON THE ACTIVITIES OF THE YEAR 2010 and summarising the key milestones of the past 10 years.

⁴⁷ With regard to the formation of dioxin and furans see also references presented in section 6.11.9

Zennegg, M., Yu, X., Wong, M.H. Weber, R., (2009). Fingerprints of chlorinated, brominated and mixed halogenated dioxins at two e-waste recycling sites in Guiyu/China. *Organohalogen Compounds* 71, 2263-2267

4.5 Information regarding the green paper on environmental issues of PVC

The following questions and answers as published in the official website of the EU⁴⁸ related to the publication of the Green Paper on environmental issues of PVC⁴⁹ are presented as follows:

Green Paper on environmental issues of PVC

Reference: MEMO/00/46 Date: 26/07/2000

Brussels, 26 July 2000

Green Paper on environmental issues of PVC

QUESTIONS & ANSWERS

1. Why is the Commission taking an initiative on PVC?

The Green Paper follows the commitment of the Commission, made in its Proposal for a Directive on end of life vehicles⁵⁰.

⁴⁸ Press release European Commission

<http://europa.eu/rapid/pressReleasesAction.do?reference=MEMO/00/46&format=HTML&aged=0&language=EN&guiLanguage=en>

⁴⁹ European Commission, COM(2000) 469 GREEN PAPER Environmental issues of PVC

<http://ec.europa.eu/environment/waste/pvc/pdf/en.pdf>

The whole life cycle of PVC raises a number of environmental issues, in particular due to the use of certain additives (lead, cadmium and phthalates) and during the management of PVC waste.

Some Member States have recommended or adopted measures related to specific aspects of the PVC life cycle. These measures are not identical and some may have consequences for the functioning of the internal market.

PVC is one of the most widespread plastics used today with a production of about 5.5 million tonnes in Europe in 1998 (21 million tonnes world-wide, about 20 % of all plastics production). Overall production of PVC compounds (PVC resin and all additives) was at about 7.2 million tonnes in 1998. The economic weight of the industrial sector is important: the total PVC producing and transforming industry in Western Europe represents about 21,000 companies, 530 000 jobs and a turnover of more than 72 billion €.

2. What is PVC used for?

The main applications of PVC in Europe today are in the building sector (windows, profiles, pipes, flooring), which accounts for 57% of all uses. The other largest uses are in the fields of packaging (9%), household (18%) and automotive appliances (7%).

3. The main distinction between the numerous applications is between « rigid PVC » (accounting for about two thirds of total use) and « flexible PVC » (accounting for about one third). The main uses of rigid PVC are pipes, window frames, other profiles, and parts of household appliances. Flexible PVC is mainly used in cables, flooring, as packaging material (flexible films), car under-floor protection and instrument panels for cars. What are the key figures for additive use and Waste quantities?

About 1 million tonnes of phthalates are used annually as plasticisers to manufacture flexible PVC products. About 112,000 tonnes of lead stabilisers (containing about 51,000 tonnes of lead metal) were used in 1998 (overall consumption of lead for use in all kinds of products was 1.6 mill. Tonnes in Europe in 1997), 50 t of cadmium metal were used in 1170 tonnes of stabiliser formulations.

About 3.6 million tonnes of post-consumer PVC waste are generated annually in Europe. An increase of PVC waste quantities of about 80% is expected in the coming 20 years. About

0.5 million pre-consumer waste (production and installation waste) is generated, 85% of which is recycled.

4. Why is the Commission only adopting a Green Paper and not a strategy with proposals for specific measures?

The Green Paper is the first step towards the adoption of a Community strategy on PVC to be adopted in early 2001.

The Commission wants to open a transparent consultation process to stimulate a debate on PVC. The European Parliament, the Member States, the NGOs, the Consumers, the industry have to be involved in this important issue.

It is essential to launch a Europe-wide public debate on the basis of the fair and balanced analysis made in the Green Paper. Similar debates have already taken place in some Member States, but not in all of them.

5. Is it better to recycle, incinerate or landfill PVC?

Currently about 3% of PVC waste is recycled (100,000 t), about 17% incinerated (about 600,000 t) and the rest landfilled (about 2.9 million t).

The Green Paper gives a detailed analysis on the advantages and disadvantages of each of these options. On the basis of this analysis and given the present low recycling rate, the Commission considers that recycling of PVC should be increased. However, it is expected that recycling of PVC waste could contribute only to the management of about one fifth of PVC post-consumer waste. Other waste management routes will therefore remain important.

The Green Paper presents the problems linked with the incineration of PVC and in particular the generation of residues from flue gas cleaning (classified as hazardous) due to presence of chlorine in PVC.

Concerning landfilling, the Green Paper discusses the stability of PVC under landfill conditions and the losses of phthalates from soft PVC, which are widely recognised in a number of published studies, but the quantities and the associated risks need to be assessed further.

The Green Paper presents a number of potential measures to improve, in accordance with the general Community Waste Management Strategy, the management of PVC waste present in various waste streams and analyses the economic consequences of deviating PVC from incineration to recycling or landfill.

It is too early to define a clear preference between incineration and landfilling for all types of

PVC under all conditions. It is one of the purposes of the Green Paper to collect further information and the various opinions about this question.

6. What does the Commission think about the voluntary commitment of the PVC industry?

The European PVC industry has signed a voluntary commitment on the sustainable development of PVC, which, *inter alia*, addresses the reduction of the use of certain heavy metal stabilisers, the mechanical recycling of certain post-consumer wastes, and the development of further recycling technologies.

The Green Paper describes the content of this voluntary commitment and presents this initiative of the industry as one of the potential future options for the Community strategy on PVC.

The signing and entry into force of this commitment (the implementation will start in 2001), which involves the complete industrial chain from producers to transformers, represents an important step which needs to be assessed in function of the effectiveness criteria mentioned in the Communication of the Commission concerning agreements in the area of the environment (COM(96)561 final).

The success of this approach will require a constant progression in the efforts realised in the specific areas covered by the agreement and, in particular, reduction in the production and use of certain additives, more ambitious target quantities for recycling, industry's contribution to added costs of incineration, and a fully operational funding mechanism.

While this can be seen as a first step there is still work to be done to ensure an effective participation by industry in achieving Community goals in this area. It should be underlined that the services of the Commission are currently preparing a Proposal for a framework Regulation concerning Community environmental agreements to be adopted by Council and Parliament.

7. Is the Commission going to propose legislative instruments on PVC?

There is a whole range of instruments available to address the environmental impacts of PVC, among them legislative measures, such as a Proposal for a Directive on PVC, or proposals for adaptation of various existing legal instruments. All possible options are mentioned in the Green Paper, together with questions regarding their effectiveness as well as their environmental and economic implications.

The Commission expects that the contributions provided by the stakeholders during the public consultation process will give further indications on which strategy is the most appropriate.

On the basis of the analyses developed in the Green Paper and the outcome of this consultation process, the Commission will present at the beginning of 2001 a Communication setting out a comprehensive Community strategy on the environmental issues of PVC. The strategy will set out all necessary measures, including, if appropriate, the development of proposals for legislative measures.

8. Is the Commission going to adopt measures concerning phthalates, lead and cadmium in PVC? Are there going to be risk assessments on these hazardous substances?

In line with its general policy, the Commission will, in the light of a scientific and economic evaluation, propose and adopt all appropriate measures to address the use of these substances in PVC.

Five phthalates have been included on the first three priority lists for risk assessment in accordance with Council Regulation 793/93 on the evaluation and control of existing substances. The risk assessments on these five substances are carried out by Member State rapporteurs (France, Sweden and The Netherlands respectively). The risk assessments on DEHP, DIDP, DINP, DBP have been or are expected to be completed in 2000 and in 2001 for BBP.

It should also be noted that the specific risks due to the use of phthalates in certain soft PVC toys and childcare articles have been assessed by the Scientific Committee on Toxicity, Ecotoxicity and the Environment (CSTEE). On the basis of these opinions, the Commission has adopted on 10 November 1999 a Proposal for a Directive and on 7 December 1999 a Decision under the emergency procedure of Directive 92/59/EEC in order to ban of the use of phthalates in certain toys and childcare articles intended to be put in the mouth.

Although at present no comprehensive risk assessments have been completed on the use of cadmium and lead compounds as stabilisers in PVC products, important work is already ongoing: a risk assessment is being finalised on cadmium and cadmium oxide under Regulation 793/93. For lead, the CSTEE has recently adopted an opinion regarding a draft ban on the use of lead in products in Denmark.

The CSTEE is currently working on the issue of risks from the use of lead in general and an opinion, building *inter alia* on a study to be commissioned by the services of the Commission, should be adopted by mid-2001 on both the environmental and human health risks of lead.

9. What will happen after the Green Paper?

In addition to the publication, the Green Paper will be transmitted to the European Parliament, the Council, the Economic and Social Committee and the Committee of the

Regions. It will be published in the Official Journal and on the internet.

A public hearing will be organised in October 2000. The targeted stakeholders are the general public as well as environmental and consumer NGOs, the producers and transformers of PVC, the users of PVC (automobile, electronic, construction, distribution and packaging industries), as well as the public administrations of the Member States. The consultation period is scheduled to continue until the end of November 2000.

On the basis of the analyses developed in the Green Paper and the outcome of this consultation process, the Commission will present, at the beginning of 2001, a Communication setting out a comprehensive Community strategy on the environmental issues of PVC to be implemented through various initiatives and measures.

10. Why is the Green Paper not focussing more explicitly on prevention and the substitution of PVC?

Prevention is explicitly addressed on several occasions, in particular concerning the use of certain hazardous additives and recycling.

The question of substitution is also explicitly addressed in the context of promoting more sustainable products as part of an Integrated Product Policy. Such a substitution policy could be considered for specific applications, which can not be easily separated from the general waste stream and therefore are difficult to recycle such as in packaging, motor vehicles, electric and electronic equipment.

At present, PVC is competing with alternative materials for a number of applications. Given the large range of applications, often requiring a very specific technical performance, PVC cannot be replaced by one single material in all its applications. For each product type, potential substitute materials are different. These can either be other plastics such as polypropylene (PP) or polyethylene terephthalate (PET) in packaging, polyethylene (PE) and PP for construction applications, or other types of materials such as wood (e.g. for window frames) or concrete, cast iron for pipes

It is stressed in the Green Paper that a potential substitution policy would need to be underpinned by a comprehensive and objective assessment of the main environmental impacts both of PVC and of potential substitutes during their whole life cycle. As the information on the environmental impacts of potential substitutes is generally scarce, the Green Paper at this stage cannot draw firm conclusions.

11. Why is the Green Paper only addressing environmental issues? Why not also human health concern?

Firstly, the Commitment accepted by the Commission in 1997 and restated in 1999 concerned exclusively environmental aspects of PVC.

Nevertheless, all related issues regarding human health that are known today have been addressed in the paper. In addition, as human health is mostly concerned indirectly through environmental impacts, the issue is (implicitly) taken into account.

12. What is the link between the PVC initiative and the integrated product policy approach which the Commission is developing?

The PVC Initiative 'integrates' the basic principles of this approach, i.e. the examination of all issues from cradle to grave, as for other waste initiatives on packaging, end-of-life vehicles, WEEE and others have also already anticipated.

The Commission has the intention to adopt a Green Paper on the Integrated Product Policy in 2000.

13. What About the Incineration of PVC and dioxin formation?

The potential influence of the incineration of PVC waste on the emissions of dioxins has been at the centre of a major scientific debate since PVC is currently the largest contributor of chlorine into incinerators. The contribution of incinerators to the total emissions of dioxins in the Community was about 40% between 1993 and 1995.

It has been suggested that the reduction of the chlorine content in the waste can contribute to the reduction of dioxin formation, even though the actual mechanism is not fully understood. It is most likely that the main incineration parameters, such as the temperature and the oxygen concentration, have a major influence on the dioxin formation and much more so than the content of chlorine.

Whilst at the current levels of chlorine in municipal waste there does not seem to be a direct quantitative relationship between chlorine content and dioxin formation, it is possible that an increase of chlorine content in the waste stream above a certain threshold could contribute to an increase of the dioxin formation in incinerators. The threshold of 1% of chlorine has been suggested but uncertainties remain on the level of this threshold.

The Proposal for a Directive on the incineration of waste⁽²⁾ foresees an emission limit value of 0,1 ng/m³. This shall decrease the emissions of dioxins from incinerators, esp. from those that at present are not yet operating with the state-of-the art technology.

⁽¹⁾COM(97) 358 final

⁽²⁾COM(1998) 558 final

4.6 Information for footprint life cycle assessment of cartridges

Below the relevant DE Europe feedback is presented:

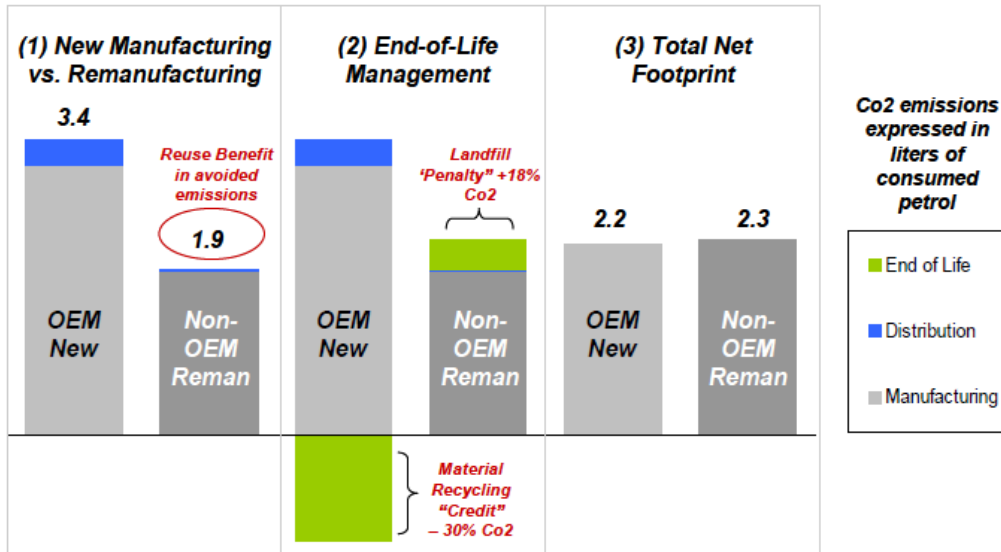
Industry data shows 80 % of aftermarket remanufactured toner cartridges are discarded after use due to non-OEM remanufacturers preference for virgin empties. [InfoTrends 2007 Supplies Recycling Report, pg 16].

Reuse and Remanufacturing can, under certain conditions offer the greatest carbon avoidance potential. In contrast to material recycling, however, the reuse of a toner cartridge does not end the product's life cycle. Quality and reliability during use of the remanufactured cartridge and its ultimate end-of-life management are crucial factors that shape the full life cycle footprint of the cartridge. Poor quality or irresponsible end-of-life handling can quickly offset the benefits of materials reuse. The following sections illustrate these factors using carbon footprint measurements of new and remanufactured cartridges.

To demonstrate the significant impact of proper end-of-life management, the chart below compares the carbon footprint of a new OEM cartridge (with material recycling after use) and a remanufactured cartridge without an end-of-life recycling program⁵¹ (i.e. assuming the cartridge ends up in a landfill⁵²):

⁵¹ Examples based on a toner cartridge Life Cycle Assessment by WSP Environment and Energy for Lexmark, 7/2009, represents a T64X 21,000 page toner cartridge. Conducted in accordance with ISO 14044 guidelines for Life Cycle Assessment. Expressed in liters of petrol consumed from EPA Greenhouse Gas Equivalencies Calculator <http://www.epa.gov/RDEE/energy-resources/calculator.html>. Converted from gallons.

⁵² "InfoTrends" research found that the chance of a remanufactured cartridge ending up in a landfill after the first remanufacturing cycle is high. In the U.S. and Europe, 80% of remanufactured toner cartridges and 86% of remanufactured inkjet cartridges are thrown away. This is because remanufacturers have such a strong preference for virgin empties" (InfoTrends 2007 Supplies Recycling Report, pg 16)



1. Here we see the initial benefit of remanufacturing vs. new manufacturing: Reuse of the cartridge materials through remanufacturing has a carbon footprint equivalent to burning 1.9 liters of petrol vs. 3.4 liters petrol equivalent for manufacturing a new cartridge.

2. Impact of End-of Life management: Assuming the empty new cartridge is collected and fully recycled (providing a 30% 'credit' to the footprint for returning those materials back to the materials stream), the remanufactured cartridge by weight (75 %) goes to landfill⁵³. This gives the remanufactured cartridge an incremental +15 % footprint 'penalty'.

3. The net footprints of both the new cartridge (with material recycling at end-of-life) and the remanufactured cartridge (in landfill) are about equal, with the remanufactured version having a slightly larger environmental impact overall.

Without end-of-life material recycling, the reuse on an empty toner cartridge does not deliver an overall environmental benefit versus a new OEM cartridge material recycled at end-of-life.

⁵³ Of the unusable cartridges collected by U.S. and European remanufacturers, we estimate that about 25% of the material is recycled." (2007 Supplies Recycling in US and Europe. InfoTrends. May, 2007. Page 10).

Ultimately, the vendor of a remanufactured cartridge must collect and material recycle the product at its ultimate end-of-life in order to avoid offsetting the benefits of reuse. This is also the case for cartridges the vendor collects but does not reuse.

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