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**Environmental Impacts and Use of Brominated Flame Retardants in Electrical  
and Electronic Equipment**

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## Abstract

Management of waste electrical and electronic equipment (WEEE) or e-waste is becoming a major issue as around 20 to 50 million tonnes such waste is generated worldwide and increasing at a higher rate other solid waste streams. Electrical and electronic equipment (EEE) contains over 1000 materials of which brominated flame retardants (BFRs) such as polybrominated biphenyls (PBBs) and polybrominated diphenyl ethers (PBDEs) have been the target of the regulators forcing manufacturers to adopt halogen free flame retardants. As far as these alternatives are concerned key consideration should be its performance during the whole life cycle through design, use and end-of-life management. Global halogen free flame retardant movement has reached a point of no return. The most important issue as far as the environment is concerned, for which the transformation to halogen free retardants was initially targeted, is to make sure that life span of the EEE using the alternatives to BFRs is not shortened thereby resulting in unforeseen increase in e-waste to deal with. The aim of this paper is to investigate the environmental issues and current developments related to the use of BFRs in EEE manufacture. It describes the sources, toxicity and human exposure of BFRs, EOL management such as recycling and thermal treatments, exposure of BFRs from e-waste processing facilities and the environment around them and examines the developments and feasibility of the alternatives to BFR in EEE manufacture.

**Keywords:** Brominated flame retardants; Electronic waste; E-waste; Halogen-free flame retardants; Recycling; PBDEs; WEEE

## **1 Introduction**

Rapid uptake of information and communication technology (ICT) and the increasing demand for consumer electrical and electronic equipment (EEE) have led to a significant expansion of the electrical and electronic manufacturing industry worldwide during the last two decades. Although the benefits of ICT and EEE to the society are well known and recognised, little is known about the end-of-life (EOL) management of such devices. Commonly known as waste electrical and electronic equipment (WEEE) or electronic waste (e-waste), every year around 20 to 50 million tonnes of these wastes are generated worldwide, bringing significant risks to human health and the environment (Schwarzer et al., 2005). E-waste mainly consists of used computers, monitors, mobile phones and televisions. It is estimated that in the United States over 100 million computers, monitors, and televisions become obsolete each year and this amount is growing (United States Government Accountability Office, 2005) and more than 500 million computers will become obsolete in the United States alone between 1997 and 2007 (Schwarzer et al., 2005). This is growing at a rapid rate as the global computer market is far from saturation and the average lifespan of a computer is decreasing rapidly from 4 to 6 years in 1997 to only 2 years in 2005 (Widmer et al., 2005). The problem is e-waste is global, for example, in China about 20 million consumer EEEs and 70 million mobile phones reach EOL each year (Yongguang et al., 2006) and in India computer ownership per capita grew 604% during the period 1993-2000 far exceeding the world average of 181% (Sinha-Khetriwal et al., 2005). Hence e-waste is one of the fastest growing waste streams in our society and is certainly a global problem.

Apart from the significant number of EEEs reaching EOL worldwide, e-waste is also of major concern due to the toxicity of some of the materials present in the waste stream. E-waste contains over 1000 different substances, many of which are toxic, such as mercury, lead, arsenic, cadmium, selenium, hexavalent chromium flame retardants (Widmer et al., 2005). Cathode ray tubes (CRTs) of used computer monitors and televisions are one of the major sources of lead in the municipal solid waste stream. Brominated-flame retardants (BFRs) such as polybrominated biphenyls (PBB) and polybrominated diphenylethers (PBDEs) used in computer equipment are both an occupational and environmental health hazard (Kang and Schoenung, 2005). One of the main reasons for the current concerns regarding the use of BFRs is that nearly all of the BFRs generate polybrominated dibenzo-dioxins and polybrominated dibenzo-furans (PBDDs/PDDFs) during the EOL processes such as heating of EEE.

The aim of this paper is to investigate the environmental issues and current developments related to the use of BFRs in EEE manufacture. It describes the sources, toxicity and human exposure of BFRs, EOL management such as recycling and thermal treatments, exposure of BFRs from e-waste processing facilities and the environment around them and examines the developments and feasibility of the alternatives to BFR in EEE manufacture.

## **2 About Brominated Flame Retardants (BFRs) and their Use**

BFR is the group of brominated organic substances that inhibits the ignition of combustible materials and is commonly used in the manufacture of EEE to reduce the

flammability of the product. BFRs are manufactured synthetically and following are the major classes of them:

1. TBBPA: Tetrabromobisphenol -A
2. HBCD: Hexabromocyclododecane
3. PBDEs: Polybrominated diphenyl ethers consisting of Deca-BDE (Decabromodiphenyl ether), Octa-BDE (Octabromodiphenyl ether) And Penta-BDE (Pentabromodiphenyl ether)
4. PBB: Polybrominated biphenyls

The manufacture of EEE consumes a major portion of the global BFR market by accounting for 56% of the product. Nearly two thirds (59%) of the EEE industry's BFR consumption is destined for housings followed by the printed wiring boards (30%), connectors & relays (9%) and wire & cabling (2%). The BFR group TBBPA represents half of BFR volumes and is contained in 96% of printed wiring boards (Bromine Science and Environmental Forum, 2000).

It is estimated that around 200,000 tonnes of BFRs are produced globally each year with Asia being the largest consumer. According to the studies undertaken in 2001, Asian region consumed 56% of the total market demand followed by Americas (29%) and Europe (15%) (Birnbaum and Staskal, 2004). The Table 1 shows the consumption of different types BFRs in various regions around the world.

(Insert Table 1 here)

## 2.1 *Tetrabromobisphenol A (TBBPA)*

As seen from Table 1, TBBPA is the most widely used BFR. 90% of the TBBPA produced is used as a reactive (chemically bound) flame retardant in epoxy and polycarbonate resins. These polymers are used in the manufacture of printed circuit boards and other electronic equipment (Alaee et al., 2003). Approximately 10% of total TBBPA produced globally is used as an additive (mixed with plastics) flame retardant in acrylonitrile-butadiene-styrene (ABS) resins and high impact polystyrenes (Sjodin et al., 2003). Studies conducted by the World Health Organisation and the European Union have concluded that TBBPA presents no risk to the human health and, therefore, no legislative restrictions on the use of TBBPA in Europe or other parts of the world (Bromine Science and Environmental Forum, 2006).

## 2.2 *Hexabromocyclododecane (HBCD)*

HBCD is an additive flame retardant used in various polymers such as polystyrene – EPS (Expandable Polystyrene Foam) and XPS (Excruded Polystyrene Foam- for thermal insulation in building and construction industry). It is also applied as backcoating of textiles in upholstery industry and in ‘High Impact Polystyrene (HIPS)’ used in EEEs.

Although recent studies have indicated that HBCD is not persistent, it has a relatively high bioaccumulation potential and could be found in high concentrations in environment (Bromine Science and Environmental Forum, 2003).

### 2.3 *Polybrominated diphenyl ethers (PBDEs)*

PBDEs are a class of additive BFRs used in a variety of polymers and foams. The PBDE class includes 209 different congeners of the PBDE molecule varying in both number and the position of bromine atom. PBDEs are manufactured by the chemical reaction of bromine with diphenyl ether. [The three main types of commercial PBDE mixtures used in consumer products are](#) Penta-BDE, Octa-BDE and Deca-BDE.

Penta-BDE is used in products such as mattresses, seat cushions and other upholstered furniture, and rigid insulation. The most common use of Penta-BDE is in polyurethane foam. Octa-BDE is used in ABS plastics which are used in the housings for fax machines and computers, telephone handsets and kitchen appliance casings. Deca-BDE is used in variety of plastics such as HIPS, polybutylene terephthalate (PBT), nylon, polypropylene and low-density polyethylene (LDPE). Typical products which use Deca-BDE include housings for televisions, computers, stereos, and other electronics, audiotape cassettes and upholstery textiles. It is also used in automotive industry and wires, cables and pipes used in the building and construction industry (Washington State Department of Ecology, 2006).

The estimated global demand for PBDEs in 1999 was 70,000 tonnes of which Penta-BDE, Octa-BDE and deca-BDE were 13%, 6% and 81% respectively. Furthermore it was estimated that North America consumed 98% of Penta-DBE, 36% of Octa-BDE and 44% of deca-BDE of the global production (Sjodin et al., 2003). The European Union (EU) has banned the use of Penta-BDE and Octa-BDE from EEE sold after 1 July 2006. [In The United States, the sole manufacturer of Penta-BDE and Octa-BDE ceased the production on 31 December, 2004. Also in the United States, Washington](#)



and Maine States have partially banned the use of Deca-BDE and many other States including California, Hawaii, Illinois, Michigan, Minnesota, Montana and New York are proposing laws to ban Deca-BDE. In European Union, Sweden has a partial ban on Deca-BDE since 1 January 2007.

#### 2.4 *Polybrominated biphenyls (PBBs)*

PBBs are identical to their chlorinated counterpart polychlorinated biphenyls (PCBs) except for the use of bromine atom instead of chlorine. As with PCBs, PBBs have been found to be persistent and bioaccumulative toxins and are classified as possible carcinogens. Due to a toxic accident involving hexabromobiphenyl flame retardant in the US in 1973, the product was banned in 1974. The production of octabromobiphenyl and decabromobiphenyl continued until 1979 in the USA. EU has banned the use of PBBs from EEEs sold after 1 July 2006. Due to this PBB is rarely found in EEE manufactured today.

### **3 End-of-life Management**

End-of-life management (EOL) of plastics used in EEE that contain BFRs can be achieved through recycling (mechanical and feedstock). Mechanical recycling involves processes such as sorting, dismantling, identification followed by shredding of large parts and results in materials (plastics) for use in original or other applications. Feedstock recycling involves recovery of bromine and energy contained in EEE. Due to growing emphasis on sustainable utilisation of resources recycling is becoming requirements of many regulations of EOL management of used EEE. For

example European Union's waste from electrical and electronic equipment (WEEE) directive (see section 5 below) mandates specific recovery and recycling targets to be met by the manufacturers. Recycling of BFRs is a major challenge as it contains different types of polymers and hazardous substances. Also [it is well documented](#) that some BFRs are reported to emit dioxins and furans when exposed to thermal stress.

Several studies have investigated the feasibility of recycling BFRs. Dawson and Laundry conducted several studies (Dawson and Landry, 2003; Dawson and Landry, 2005) on the mechanical recyclability of HIPS resin used in EEEs under various conditions such as standard recycling, extreme temperature and extreme humidity. They found that HIPS resin can be recycled even under extreme conditions into high value molded parts with unchanged mechanical properties and same original flame retardability. (Tange and Drohmann, 2004) reports of a similar study conducted in Sweden on HIPS containing BFR. Number of mechanical tests were carried out and the outcome was that flame retarded HIPS withstands thermo-oxidative aging much better than the non-flame retarded HIPS confirming earlier studies (Bromine Science and Environmental Forum, 2000) that HIPS containing Deca-BDE can be recycled five times in full compliance with the German legislation on dioxin/furan product content and emission limit values, one of the strictest such legislation in the world. The studies conducted on the recyclability of ten commercially available flame retarded plastic grades commonly used in EEEs (eight halogen-free grades and two grades containing BFRs) revealed that none of the eight tested halogen-free plastic grades could maintain the fire safety rating after five recycling groups while both BFR plastics continued to retain the ratings confirming the superior recycling properties of BFR containing polymers (Imai et al., 2003).

Feedstock recycling of BFR containing polymers has received much attention during the past few years. Bromine recovery is one such option where BFR containing polymers are thermally treated to recover bromine to be used as raw material bromine for producing further BFRs or to be re-used in the chemical industry. Feedstock recycling of BFR polymers could also be undertaken with thermal treatment options such as municipal solid waste incineration, co-combustion, pyrolysis and gasification. It is estimated that with an effective energy recovery, incineration of mixed WEEE plastics is feasible due to the high heating value of these materials which ranges from 18 to 46 MJ/kg (Schlummer et al., 2007). (Tange and Drohmann, 2005) reports several successful pilot/industry scale studies conducted on feedstock recycling of BFR plastics. These include the Haloclean pyrolysis procedure based in a plant situated in Forschungszentrum Karlsruhe Germany to separate brominated additives from WEEE, a pilot trial carried out for the bromine industry at an Energy Research Centre in Holland where it was shown the feasibility of recovering bromine via thermal processes and co-combustion of BFR containing polymers with municipal solid waste at a pilot plant (TAMARA) at the Forschungszentrum Karlsruhe in Germany. The feasibility of using BFR containing polymers as alternative fuel in the cement manufacturing process has also been investigated.

One of the main challenges or drawbacks of thermal processes is the potential formation of PBDD/PBDFs. Furthermore, high load of halogens in WEEE may result in corrosive effects and may affect the normal operations of incineration plants.

Several studies have investigated these issues. (Yamawaki, 2003) conducted a study on gasification of BFR containing plastics where treatment was done at a temperature of over 1200<sup>0</sup>C and then gases shock-cooled to a temperature of less than 200<sup>0</sup>C. The process of high temperature treatment and shock cooling is known to suppress the emission of chlorinated dioxins to a very low level and similar results were achieved for brominated dioxins emissions in this experiment. (Barontini and Cozzani, 2006; Barontini et al., 2005) studied formation of hydrogen bromide and organobrominated compounds in the thermal degradation of BFR containing electronic circuit boards under varying heating rates and found the formation of considerable amounts of these compounds as well as the potential to form limited quantities of PBDD and PBDF. These outcomes are similar to findings from a study on thermal degradation of a brominated bisphenol A derivative (Luda et al., 2003) where it was observed hydrogen bromide and brominated phenols and bisphenols as the main volatile products formed. PBDDs were only included in the charred residue but not in the volatile phases. (Weber and Kuch, 2003) conducted a very comprehensive study on the relevance of BFRs and thermal conditions on the formation pathways of PBDDs and PBDFs. They found that total amount of PBDDs/PBDFs formed during thermal processes depends largely on the quality of precursor compounds and the specific conditions of the thermal treatment. They also found that under thermal stress situations which may occur in normal production and recycling processes, precursors such as PBDE are potentially relevant in forming PBDDs/PBDFs while for other BFRs no significant PBDDs/PBDFs formation was reported. The above study also revealed that BFRs if destroyed with high efficiency under controlled combustion conditions may then not serve as precursors for PBDDs/PBDFs formation and

suggested pyrolysis/gasification processes in combination with a melting step as an appropriate method for the processing of flame-retarded plastics.

#### **4 Exposure**

Researchers around the world are investigating the bioaccumulation of BFRs in the biota to examine whether they pose a threat to the human health and the environment. Majority of these studies are concentrating on PBDEs as they persist in the environment for extended periods of time. As a result, PBDEs have been detected in various environmental media which includes water (Kierkegaard et al., 2004; Litten et al., 2003), air (Farrar et al., 2004; Gevao et al., 2006) and general biota (Litten et al., 2003; Michael et al., 2002; Rayne et al., 2003; Sellstrom et al., 1993; Sellstrom et al., 1998). The presence of PBDEs in the human body (Choi et al., 2003; She et al., 2002), serum (Adibi et al., 2003; Koizumi et al., 2005; Sjodin et al., 2004; Sjodin et al., 2001b), blood (Adibi et al., 2003; Guvenius et al., 2003; Kim et al., 2005; Mazdai et al., 2003; Schechter et al., 2007; Sjodin et al., 2003) and breast milk (Akutsu et al., 2003; Bi et al., 2006; Kalantzi et al., 2004; Sjodin et al., 2003) has also been reported in the literature. (Hites, 2004) conducted a meta-analysis of concentration of PBDEs in the environment and people and found that total PBDE levels in human blood, milk, and tissues have increased exponentially by a factor of around 100 during the last 30 years effectively doubling the levels every 5 years. This study also found that the increase in PBDE levels in the environment and human in North America to be much higher than in Europe. These findings were further confirmed by (Schechter et al., 2005) where they found that the U.S. human breast milk and blood levels of PBDEs are presently the highest in the world. Although PBDEs have been detected in

humans, there is lack of data on human health studies so far. However, studies on animals, reported in detail by (Darnerud, 2003), have shown effects on the nervous system, reproduction and development.

Environmental impacts of PBDEs of used EEE is emerging as a highly topical issue for many researchers around the world. Research is being undertaken to investigate the exposure of workers at end-of-life EEE processing facilities and the environment around it. One of the initial studies in this field was undertaken in Sweden in 1999 (Sjodin et al., 1999). This study compared the PBDE levels in blood serum of personnel working full time at computer screens, personnel at an electronics dismantling plant and hospital cleaners. The study found significantly high levels of PBDE in personnel working at the electronic dismantling plant compared to other two groups. This study was further extended to a group of technicians with intense computer work and resulted in a similar outcome (Jakobsson et al., 2002). A recent study conducted in China by (Qu et al., 2007) involving three groups of people (residents from an e-waste dismantling region, residents living within 50km of the dismantling region and a control group with no occupational PBDE exposure) found significantly high levels of BDE-209 congener of PBDE in serum of residents from e-waste dismantling region which was 11-20 higher than the control group. The BDE-209 concentration observed in this group was highest for humans worldwide.

Environmental impacts of PBDEs in e-waste processing facilities and the surrounding areas have been more studied than the exposure of personnel working in them. One of the initial studies in this regard was carried out by (Sjodin et al., 2001a) where air samples from a e-waste recycling plant, a factory assembling printed circuit boards, a

computer repair facility, offices equipped with computers, and outdoor air were analysed. PBDEs were detected in all air samples with significantly higher levels present in air in the vicinity of the shredder at the dismantling plant. These findings were further confirmed by (Pettersson-Julander et al., 2004) when they studied the exposure to BFRs within an e-waste recycling facility personal air monitoring during a two year period. The study which involved workers representing three different categories: dismantlers, other workers and unexposed found a significant difference among the dismantlers and the unexposed categories for PBDE congeners 47, 100, 99, 154; 153, 183, 209. These studies were further extended by (Julander et al., 2005) where air samples were collected from an e-waste recycling facility in Sweden representing three different dust fractions; respirable, total and inhalable dust. The study found high concentrations of PBDE congener 209 in the samples from the inhalable dust fraction, and was 10 times higher than for the “total dust” fraction.

China’s e-waste recycling facilities have attracted significant attention worldwide due to their perceived unsustainable e-waste recycling operations. One of the first studies to document these practices was conducted by the Silicon Valley Toxics Coalition (SVTC) and the Basel Action Network (BAN) published in 2002 through a now well known document ‘Exporting Harm: The High-Tech Trashing of Asia (Puckett et al., 2002). This report asserts that 50 to 80 % of e-waste collected for recycling in the developed countries end up in China and designated Guiyu village in Guangdong province in China as an ‘electronics junkyard’. As a result significant amount of scientific research is being currently undertaken to assess the environmental impacts e-waste recycling operations in Guiyu area. (Wang et al., 2005) collected soil and sediment samples from the vicinity of an open e-waste recycling facility located in

Guiyu and analyzed for the levels of PBDEs. They detected PBDEs levels in the soil and sediment samples at levels of 0.26–824 ng/g (nano gram per gram of dry weight). These findings were confirmed by (Cai and Jiang, 2006) when they detected PBDE concentrations up to 600 parts per billion in soil samples collected from a similar e-waste recycling plant.

(Leung et al., 2006) conducted a detailed study in Guiyu area to identify the sources and quantify the pollution levels generated from e-waste. They found PBDE levels up to 1169 ng/g (dry weight) in soils near the plant which was 10-60 times higher than PBDE contaminated locations in the world. (Leung et al., 2007) extended this study to surface soils and combusted residue in Guiyu area and found total PBDE concentrations were highest in combusted residue of plastic chips and cables collected from a residential area (33 000-97 400 ng/g, dry wt), in soils from an acid leaching site (2720-4250 ng/g, dry w), and a printer roller dump site (593-2890 ng/g, dry wt). They also found that BDE congener 209 was the most dominant congener (35-82%) among the study sites confirming the existence of commercial Deca-BDE. Studies conducted by (Luo et al., 2007a; Luo et al., 2007b) further confirms the levels of PBDE in Guiyu area in their experiments on sediment and fish samples collected from rivers in Guiyu. The total concentrations of PBDE in this study ranged from 4434 to 16088 ng/g (dry weight) in Nanyang River bank sediment, from 55 to 445 ng/g in Nanyang River bottom sediment and 51.3 to 365 ng/g in Lianjiang River bottom sediment in Guiyu. The authors compared these findings with those from 16.1 to 21.4 ng/g in wastewater discharged from a vehicle repairing workshop in Hong Kong. They also found PBDE concentrations in sediment and fish in this area were 10 and 1000 times higher than other studies. In other parts of the world, (Binelli et al., 2007)



conducted a study on a mangrove wetland in north-eastern part of Bay of Bengal in India and found concentrations of several PBDE congeners, however, they could not confirm the main source in this area which includes an e-waste dump site. [There are also reports of incredibly high air concentrations of PBDD/Fs in Guiyu area, most likely due to the high PBDE levels.](#)

## **5 Related Regulations**

Laws and regulations governing the EOL of EEEs and the hazardous materials utilised to manufacture EEEs are being implemented in various countries around the world and have direct impacts on the use of BFRs. Two of these regulations emerge from the European Union (EU) where Directives on Waste Electrical and Electronic Equipment (WEEE) and the Restriction of the Use of Certain Hazardous Substances in Electrical and Electronic Equipment (RoHS) have come into force (European Union, 2003a; European Union, 2003b). The WEEE Directive was agreed by European Parliament on 13 February 2003, transposed into Member State legislation by 13 August 2004 and came into force on 13 August 2005 and aims to increase the recycling and recovery of WEEE (including BFR containing plastics) through mechanical recycling, feedstock recycling and energy recovery. The directive also requires that by 31 December 2006, the member states must achieve a collection rate of at least 4 kg of WEEE per person per year. The WEEE Directive also requires certain recovery and recycling rates to be achieved by 31 December 2006 for different types of equipment. The RoHS Directive came into effect on 1 July 2006 and applies to new electrical and electronic equipment put on the European market on or after 1 July 2006. It names six substances of immediate concern: lead, mercury, cadmium,

hexavalent chromium, PBB and PBDE (Penta-BDE, and Octa-BDE). The maximum concentration values for RoHS substances were established in an amendment to the Directive on 18 August 2005. The maximum tolerated value for lead, mercury, hexavalent chromium, PBB and PBDE is 0.1% by weight in homogenous materials and 0.01% by weight in homogenous materials for cadmium (European Union, 2005b).

The RoHS has the same coverage of EEE as the WEEE Directive, with the exception of medical, monitoring and control equipment and number of specific exemptions declared as Annexes to the Directive. One of these relate to Deca-PBDE which was exempted from all polymeric applications (European Union, 2005a). This decision was taken by the European Commission (EC) based on 10 year environment and health risk assessment study conducted in the EU. However, in January 2006, the European Parliament and Denmark both filed suit against this decision with the European Court of Justice which is still progressing.

Following the EU's RoHS regulations, China, Korea and State of California in the US have implemented similar regulations. Referred to as China's equivalent to EU RoHS, *Measures for the Administration of the Control of Pollution by Electronic Information Products*, was promulgated by the Ministry of Information Industry and other participating agencies on 28 February 2006 and came into effect on 1 March 2007. The goal is to make companies disclose and control the use of hazardous/toxic materials through disclosure via labelling requirements and control via substance restrictions. The substances covered by this law are the same six substances covered by EU RoHS which include PBDE. California's Electronic Waste Recycling Act also

has a mandate to reduce the use of hazardous substances in electronic products sold within the State. Accordingly California RoHS (SB50) came into effect on 1 January 2007. On 2 April 2007, Korea's National Assembly passed the 'Act Concerning the Resource Recycling of Electrical/Electronic Products and Automobiles' which has similarities to EU's RoHS, WEE and ELV (End of Life Vehicles) Directives but also containing items specific to Korea. Also known as Korea's RoHS this regulation goes into force on January 1, 2008.

Also on 30 December 2006, the European Parliament issued the Directive on Registration, Evaluation and Authorisation of Chemicals (REACH) (European Union, 2006). Its purposes include establishing a comprehensive database of chemical substances in the European Community to motivate manufacturers to seek cost-effective and safer alternatives to hazardous substances and came into force on 1 June 2007.

## **6 Alternatives to BFRs**

Despite the uncertainties and knowledge gaps related to the environmental and health effects of BFRs used in the EEE manufacture there is significant effort within the electronics industry to find substitutes, particularly in Europe, [United States](#) and Japan. This is mainly driven by WEEE and RoHS regulations in the EU and the demand for green markets in Japan. Similar to the situation with lead-free soldering the challenge is to find a drop-in substitute for BFR which has the technical, environmental and economic edge to the material being substituted.

There is a large number of commercially available alternatives to BFRs mainly based upon inorganic compounds such as aluminium trihydrate, aluminium trihydroxide, magnesium hydroxide, ammonium polyphosphate, red phosphorous, zinc borate and antimony oxides, organophosphorous compounds such as resorcinol bis(diphenyl phosphate, nitrogen based melamine products such as melamine cyanurate, melamine polyphosphate and melamine pyrophosphate and nanoclays (Lincoln et al., 2005; Markarian, 2005). Many of the major EEE manufacturers are currently using alternatives to BFRs. The description of such alternatives will not be attempted in this research paper as all the details could be obtained from their respective websites.

## **7 Conclusions**

Flame retardants play an important role in protecting people and property by saving life and damage from the spread of flame and fire. Fire statistics confirm that risk of injury or damage from fire involving consumer EEE can be reduced significantly by using fire retardants in them. Their application is wide ranging and extends from EEE to fibres, textiles, plastics and furniture. Many of the present consumer products contain flame retardants and with the predicted continued growth of the EEE manufacturing industry their use is likely to continue to increase. Out of all fire retardants BFRs have major preference given its excellent fire retardant properties and the low cost. Mechanical recycling and feedstock recycling are considered to be only available EOL management options for the BFRs. This is aligned with the recent regulatory requirements mandating the recovery and recycling of BFR containing polymers. However, there is much debate about the environmental impacts of these

options, in particular feedstock recycling through thermal treatment, due to perceived emissions of PBDDs and PBDFs in the combustion process.

In the recent past there had been much controversy of using BFRs in EEE given its perceived impact on the environment and health. [Many](#) scientists have detected the occurrence of PBDE (used in BFRs) in soil, air, water and general biota. However there are very few scientific publications confirming exposure of BFRs among the workers in state-of-art WEEE recycling facilities and the environment surrounding them although much research has been published confirming the exposure of PBDE among people in [areas such as Guiyu in China and New Delhi in India where unsustainable recycling practices are adopted.](#)

Number of regulatory initiatives and decisions requiring the substitution of BFRs with halogen free flame retardants are being implemented or developed around the world today. As a result the manufacturers, particularly in Europe and Japan, are striving to develop alternatives to BFRs. As far as these alternatives are concerned key consideration should be the performance during the whole life cycle through design, use and EOL. The environmental, technical and functional aspects of the alternatives should be superior to the material being replaced. The world is moving rapidly towards lead-free soldering which requires much higher processing temperatures than leaded solders. Therefore, the alternatives to BFRs should withstand these new requirements and should not decrease the product safety and performance. The question is therefore whether we substitute the BFRs with a relatively unknown halogen free chemical or a material or do we re-design all our EEE so that there is no need for the fire retardants in the first place, for example, manufacturing of televisions

with greater spacing or metal barriers between components. Cleaner production and Design for Environment (DfE) practices in EEE manufacturing could be a key in finding answers this question.

BFRs have been used in the EEE manufacture for a long time. [As for the global halogen free or non-persistent and non-bioaccumulative brominated or chlorinated retardant transformation is concerned](#), it has reached a point of no return as the regulators and leading manufacturers have invested significant amount of resources to ensure alternatives to BFRs are developed without any delay. Therefore, for many electronic manufacturers it's not a matter of 'if' but it's a matter of 'when' they have to switch into halogen free retardants. However, the most important issue as far as the environment is concerned, for which the transformation to halogen free retardants was initially targeted, is to make sure that life span of the EEE using the alternatives to BFRs is not shortened thereby resulting in unforeseen increase in e-waste to deal with.

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