

Methods of Responsibly Managing End-of-Life Foams and Plastics Containing Flame Retardants: Part I

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Received: April 14, 2017

Accepted in revised form: September 27, 2017

Abstract

Flame retardants (FRs) are added to foams and plastics to comply with flammability standards and test requirements in products for household and industrial uses. When these regulations were implemented, potential health and environmental impacts of FR use were not fully recognized or understood. Extensive research in the past decades reveal that exposure to halogenated FRs, such as those used widely in furniture foam, is associated with and/or causally related to numerous health effects in animals and humans. While many of the toxic FRs have been eliminated and replaced by other FRs, existing products containing toxic or potentially toxic chemical FRs will remain in use for decades, and new products containing these and similar chemicals will permeate the environment. When such products reach the end of their useful life, proper disposal methods are needed to avoid health and ecological risks. To minimize continued human and environmental exposures to hazardous FR chemicals from discarded products, waste management technologies and processes must be improved. This review discusses a wide range of issues associated with all aspects of the use and responsible disposal of wastes containing FRs, and identifies basic and applied research needs in the areas of responsible collection, pretreatment, processing, and management of these wastes.

Keywords: disposal, flame retardants, foams, plastics

Introduction

FLAME RETARDANTS (FRs) are added to foams and plastics in consumer products to comply with flammability standards for household and industrial items. Common items that have contained FRs include furniture foam and plastics used in electronic equipment such as computers, monitors, and TVs. The type of FRs used depends on material compatibility, costs, and the flammability standard. For example, from 1975 to 2014, halogenated FRs (HFRs) were widely

used in U.S. residential upholstered furniture and other foam-filled products to comply with California Technical Bulletin 117 (TB 117). While a California regulation, TB 117-compliant products were sold throughout the United States and Canada. When these flammability standards were implemented, potential health and environmental impacts of FRs were not fully recognized.

Numerous studies have revealed that HFR exposure is associated with and/or causally related to multiple health effects in animals and humans, including endocrine disruption, immunotoxicity, reproductive toxicity, impaired fetal/child development, and cancer (Herbstman *et al.*, 2010; Shaw *et al.*, 2010; Eskenazi *et al.*, 2013; Bellanger *et al.*, 2015; Lyche *et al.*, 2016). The annual cost to society from

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polybrominated diphenyl ether (PBDE) FRs in the United States has been estimated at 266 billion dollars (Attina *et al.*, 2016; see Bond and Dietrich, 2017 and references, therein), and the fire safety benefits relative to the environmental and health impacts of FRs have been questioned (Shaw *et al.*, 2010). The San Antonio Statement has drawn attention to these issues with recommendations for actions to minimize harm caused by HFRs (DiGangi *et al.*, 2010)

In response to health and environmental concerns, some HFRs have come under regulatory control. In 2009, certain PBDEs were listed under Annex A of the Stockholm Convention, which calls for elimination and new use of listed chemicals. Parties to the Convention (181 countries) have banned them and control products containing them.¹ Provisions were added to allow for the recycling of plastics containing c-OctaBDE and c-PentaBDE at levels less than 1,000 ppm (DiGangi and Strakova, 2015). In 2012, hexabromocyclododecane (HBCD) was also added to the Stockholm Convention, with limited exemptions granted for HBCD use in building insulation, and most recently, decabDE was added to the Stockholm Convention.

Existing products containing toxic or potentially toxic FRs such as PBDEs and HBCD will remain in use for decades, even after regulatory controls have been implemented (Abasi *et al.*, 2015). New products containing similar, potentially toxic chemicals will continue to be sold as information on toxicity is gathered and each HFR is adjudicated by national authorities for their “safety.” Until that adjudication, provisions under chemical management plans allow for the use of existing, but insufficiently characterized HFRs in a potentially large stock of in-use products. At end of life, these products move into the waste stream where HFRs may pose a risk to waste handlers and the surrounding environment. Thus there is a need at the product’s end of life for disposal methods that minimize health and ecological risks posed by HFRs already designated as toxic, and those that could receive such a designation in future.

This review resulted from a series of workshops hosted by UC Berkeley and the Green Science Policy Institute in 2016–2017 with an international group of experts in varied fields. It discusses issues associated with the responsible disposal of wastes containing FRs, and identifies basic and applied research needs in the areas of responsible collection, pretreatment, processing, and management of these wastes. Our intention is to bring to light best practices from countries and regions with well-developed protocols and regulatory frameworks.

Types of FRs in common use

FRs can be categorized according to their chemistry and the mode of addition to polymers. Additive FRs are mixed into, but not chemically bonded to, the polymer formulation. As such, they migrate from the polymer to surrounding media such as air and dust (Kemmlin *et al.*, 2003; Rauert and Harrad, 2015). Some of the most widely used additive HFRs included commercial formulations of pentabromodiphenyl ether (c-PentaBDE, or “Penta”), octabromodiphenyl ether (c-Octa-BDE), decabromodiphenyl ether (c-DecaBDE), and

hexabromocyclododecane (c-HBCD). Commercial formulations of PBDEs are a mixture of congeners. For example, c-PentaBDE is primarily a mixture of tetra-, penta-, and hexa-BDE congeners (U.S. Department of Health and Human Services, 2004).

Reactive FRs are chemically bonded to the polymer to which they are added or applied and, as such, should less readily migrate from the polymer (U.S. EPA, 2008). Tetrabromobisphenol A (TBBPA), used as a reactive FR in printed circuit boards, is the highest-volume FR in use (Shaw *et al.*, 2010). Despite TBBPAs presumed lower release rate, the volume used results in measurable environmental levels (Kajiwara *et al.*, 2011). Reactive brominated flame retardants can also degrade or be transformed to other toxic compounds (Koch *et al.*, 2016).

Table 1 lists several HFRs that have been designated by one or more national or international authorities as toxic.

Regrettable substitutions. Several HFRs have been regulated, followed by replacement by another HFR. In general, without disclosure from manufacturers, the quantities and types of FRs used as replacements are largely unknown. A pattern has emerged of replacing HFRs with known hazard with alternative HFRs with poorly known hazards—“risk migration” (Alcock and Busby, 2006). While there has been some voluntary phase-out of BFRs, the substitutes are in some cases also problematic.

An example of what has come to be known as “regrettable substitution” (Howard, 2014) can be found with replacements for c-PentaBDE. In the United States, manufacturers voluntarily phased-out production of c-PentaBDE by 2004. c-PentaBDE in flexible polyurethane foam (FPF) was largely replaced by Tris(1,3-dichloro-2-propyl) phosphate (TDCPP) and Firemaster 550 [mainly 2-ethylhexyl-2,3,4,5-tetrabromobenzoate (TBB) and bis(2-ethylhexyl)-2,3,4,5-tetrabromophthalate (TBPH)] in the United States. These alternatives to c-PentaBDE may pose similar health hazards to the compound they are replacing (Stapleton *et al.*, 2012; Cooper *et al.*, 2016). TDCPP has recently been listed under California Proposition 65 as a hazardous substance and several U.S. states have passed legislation restricting its use in children’s products. TBB and TBPH have also come under control in legislation passed by Massachusetts (Massachusetts, 2016). Tris(chloropropyl) phosphate (TCPP) has also been used in FPF (van der Veen and de Boer, 2012) and is now pending regulation in (Canada, 2016).

Figure 1 shows changes in the FRs used in sofas.

Health and environmental impacts. As noted above, HFRs migrate from the polymer to which they are added, particularly additive HFRs. Release occurs throughout the lifecycle of a product, starting at the point of chemical synthesis, addition to the product, the use phase of the product, and finally at end of life. Numerous HFRs are persistent and bioaccumulative, and can be transported long distances (Zhang *et al.* 2016). Due to the large usage volumes of some HFRs, their ability to migrate from products and for some, their persistence, they are found in elevated levels in indoor media, outdoor urban media, and in water bodies surrounding landfills and sewage treatment facilities (e.g., de Wit 2002; de Boer *et al.* 2003; Melymuk *et al.* 2012).

¹The Stockholm Convention list of POPs: <http://chm.pops.int/TheConvention/ThePOPs/ListingofPOPs>

TABLE 1. EXAMPLES OF ADDITIVE HALOGENATED FLAME RETARDANTS DESIGNATED AS TOXIC BY NATIONAL OR INTERNATIONAL AUTHORITIES AND A SUMMARY OF THEIR APPLICATIONS

Flame retardant	Products
c-Pentabromodiphenyl ether (Penta-BDE, PBDE, or Penta)	Flexible polyurethane foam: upholstered furniture, carpet padding, paints, sound insulation panels, small electronic parts, fabric coatings, epoxy resins
c-Octabromodiphenyl ether (Octa-BDE, OBDE, or Octa)	Acrylonitrile-butadiene-styrene (ABS) plastic: housings for computers, televisions, and other electronics; automobile trim, telephone handsets, kitchen appliance casings, small electronics parts, audio/video equipment, remote controls
c-Decabromodiphenyl ether (Deca-BDE, or Deca)	High impact polystyrene (HIPS): electronics, wire and cable, public buildings, construction materials, automotive, aviation, storage and distribution products, textiles, waterborne emulsions, paints and coatings
c-Chlorinated Tris [Tris(1,3-dichloro-2-propyl) phosphate, TDCPP, or Tris]	Polyurethane foams: baby mattresses, furniture cushions, automotive seating and trim; plastics, resins, textiles Polyisocyanurate and rigid polyurethane foam: insulation
c-Tetrabromobisphenol A (TBBPA)	Reactive and additive flame retardant: epoxy and polycarbonate resins; printed circuit boards in electronics, housings of computers, monitors, TV, office equipment; adhesive coatings in paper and textiles
c-Hexabromocyclodecane (HBCD or HBCDD)	Various plastics: Polystyrene (EPS, XPS, and HIPS); textiles and carpet backing, television and computer housings, textiles in automobiles, building materials (insulation), latex binders

PBDE, polybrominated diphenyl ether.

PBDE exposure is a major public health challenge (Eskenzi *et al.*, 2013). During the use phase of a product, indoor dust is the primary exposure pathway for PBDEs for humans, with electronics and FPF in upholstered furniture likely the main sources (Lorber 2008; Trudel *et al.* 2011; Zhang *et al.* 2011; Abbasi *et al.* 2015; Abbasi *et al.* 2016). PBDEs are known to influence thyroid, reproductive, and immune systems (Darnerud *et al.* 2001; Birnbaum and Staskal 2004; Legler 2008; Herbstman *et al.* 2010; Cowell *et al.* 2015), and

produce neurodevelopmental dysfunctions (Darnerud *et al.*, 2001; Jacobson *et al.*, 2016; Lipscomb *et al.*, 2017). Fetal and perhaps early life exposure to PBDEs have been associated with attention problems and decrements in processing speed, perceptual reasoning, verbal comprehension, and full-scale IQ (Erkin-Cakmak *et al.*, 2015; Lam *et al.*, 2017).

TBBPA, which is structurally related to bisphenol A (BPA), is associated with thyroid hormone activity and produces effects on neurotransmitter uptake at similar concentrations as for polychlorinated biphenyls (Legler, 2008). Higher HBCD concentrations in prenatal serum have been associated with negative effects on cognitive levels in children (Kiciński *et al.*, 2012).

Materials and waste flows with FRs

When used, FRs constitute 5–30% of flame-retarded plastics and foam by weight (Birnbaum and Staskal, 2004; D'Silva *et al.*, 2004). Estimating the waste stream flow of FR-treated foams and plastics is difficult, since there is limited data on use levels and product content. Abbasi *et al.* (2015) estimated a flow of ~10,000 tonnes/year of PBDEs in major products containing PBDEs to waste in North America (excluding vehicles) between 2005 and 2008. With no reuse, they predicted that PBDE-containing products entering the waste stream would decrease at 4–12% annually after 2013.

Flexible polyurethane foam. For decades, a large share of FPF in furniture in the United States was treated with c-Penta-BDE. The Polyurethane Foam Association reported that more than 500,000 tonnes of FPF was produced in the United States. FPF production for furniture in 2015 was estimated at ~10,000 tonnes (Luedeka, 2016). Furniture and

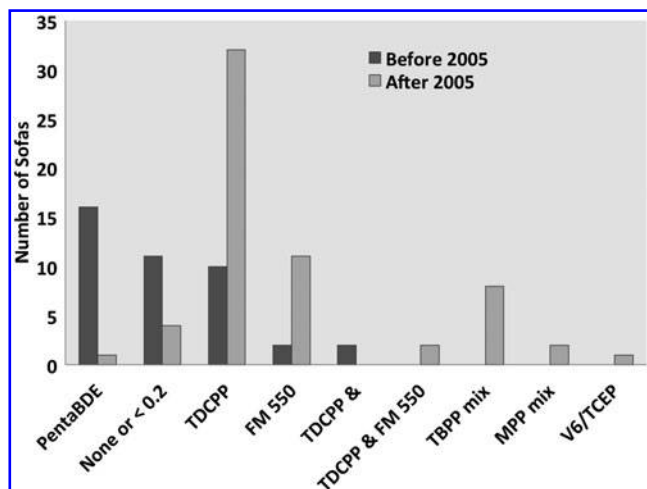


FIG. 1. Sofas purchased before and after 2005 containing specific types of FRs. After the phase-out in 2005, PentaBDE was replaced with TDCPP and other mixtures. Data are from Stapleton *et al.* (2012). FRs, Flame retardants; TDCPP, Tris(1,3-dichloro-2-propyl) phosphate.

furnishings in municipal solid waste (MSW) increased from 2 million tonnes in 1960 to 10.5 million tonnes in 2013, 4.6% of total MSW (U.S. EPA, 2015). In 2016, total U.S. FPF production was ~950,000 tonnes, with 610,000 tonnes in slabstock, and about 340,000 tonnes in molded foam. Abbasi *et al.* (2015), using data from the Polyurethane Foam Association, estimated that 5,000–50,000 tonnes of *c*-Penta-BDE was used in flexible foam in the United States and Canada between 1970 and 2004.

FRs in electrical and electronic equipment and related waste. From 1990 onward, household EEE products, such as TVs and computers, have estimated lifespans of ~5–10 years (U.S. EPA, 2010). In surface wipes of 65 products in Canadian homes that had X-ray fluorescence (XRF)-determined bromine of >0.1%, *c*-Penta-BDE was found in all personal computers and 70% of small household electronic products (Abbasi *et al.*, 2016). In the same study, *c*-decaBDE was found in all cathode ray tube televisions and most small household electronic products. The percentage of flat-screen TVs and monitors with PBDEs was much higher (54% and 40%, respectively). Abbasi *et al.* (2015) calculated that usage in North America of *c*-Penta, *c*-Octa, and *c*-DecaBDEs in electrical and electronic equipment from 1970 to 2013 will eventually add to the waste stream 500–1,500, 1,000–10,000, and 25,000–225,000 tonnes of these mixtures, respectively. TBBPA was detected in most household electronics at levels of 1,000–10,000 ppm (Wäger *et al.*, 2012). Kajiwara *et al.* (2011) found the highest concentrations of FRs in the power boards of LCD TVs sampled in Japan.

Vehicles. To meet U.S. Federal Motor Vehicle Safety Standard 302, HFRs were and continue to be added to polymers and textiles used in passenger cars and trucks. Commercial busses, airplanes, ships, and military vehicles have additional more-demanding flammability standards. Estimates show that the United States used 40% of pentaBDE for FPF slabstock foams in seats, head rests, ceiling, and acoustic systems (Alcock *et al.*, 2003; Stockholm Convention, 2015b, 2015c). In Japan, textile seat covering and floor mats contain BFRs (Kajiwara *et al.*, 2015). FPF represents ~13.6–18.1 kg/passenger vehicle.² *c*-Penta, *c*-Octa, and *c*-DecaBDE in North American lightweight vehicles from 1970 to 2013 will contribute 2,500–25,000, 3,000–35,000, and 25,000–225,000 tonnes, respectively, to the waste stream (Abbasi *et al.*, 2015). Abbasi *et al.* (2015) acknowledged that these values underestimate PBDEs entering the waste stream as they do not include heavy-duty vehicles, trains, buses, and aircraft, which were suspected to use these compounds to retard the flammability of FPF. There was limited use of PBDEs in molded foam vehicle components.

Materials with decaBDE and HBCDs are also in vehicles (Stockholm Convention, 2015a). The highest level of HBCD exposure in the United Kingdom was linked to automobile cabin dust (Harrad and Abdallah, 2011; Kajiwara *et al.*, 2015; Stockholm Convention, 2015a). The highest concentrations were in two end-of-life vehicle (ELV) floor coverings at 3,000 and 13,000 mg/kg (Kajiwara *et al.*, 2015).

²Estimate provided by The Woodbridge Group, a global manufacturer of foam and interior parts for vehicles.

Construction. BFRs are used in polymer building insulation such as EPS, XPS, or rigid polyurethane foam. HBCD is the main additive FR for polystyrene insulation foams, accounting for 80–90% of its global use (Babrauskas *et al.*, 2012). HBCD mass content in EPS ranges from 0.5% to 0.7%, and 0.8 to 2.5% for XPS. TCPP and, to a lesser extent, Tris (2-chloroethyl) phosphate (TCEP) are or were commonly used for spray and rigid polyurethane foam, added at mass levels of 2–25% (Babrauskas *et al.*, 2012). Truong (2016) confirmed this, finding TCPP concentrations of 2–26% by mass.

Textiles. Textiles accounted for 10% of global use of HBCD, but it has largely been phased out with listing of HBCD in the Stockholm Convention (Stockholm Convention, 2015a). HBCD was applied on cotton, cotton blends, or synthetics on a back coating at concentrations from 2.2% to 4.3%. The U.S. EPA (2014) determined that 20–25% of the total global decaBDE production (~300,000 tonnes) was applied to textiles from 1970 to 2013. Based on these values, Abbasi *et al.* (2015) estimated that 6,000–60,000 tonnes of decaBDE applied to textiles will have or enter the waste stream until 2020.

Recycled products. In the last 30 years, plastics and foams with HFRs have been recycled into new products. FR plastics are removed from other plastics to a limited extent and remain in the recycled product at detectable levels. Samsonek and Puype (2013) reported TBBPA and decaBDE in thermal cup lids as high as 1,294 mg/kg from recycled waste electrical & electronic equipment (WEEE). Chen *et al.* (2009) reported that PBDEs were detected in all of 69 toys purchased in Guangzhou City, China. DecaBDE had the highest concentrations of 4.3 mg/g and a median concentration of 0.03 mg/g. Ionas *et al.* (2014) found low levels of PBDEs (0.14 mg/g maximum) in 106 toys purchased in Belgium. However, 50% of the toys had organophosphate esters with concentrations of triphenyl phosphate (TPhP) as high as 1.3%. Both studies concluded that FRs in toys were contaminants from recycled materials (Chen *et al.*, 2009; Ionas *et al.*, 2014).

In the United States, bonded foam carpet cushion represents ~90% of all carpet cushion sold. Even with pentaBDE phased out in 2004, old foam scrap continued to be recycled into bonded carpet cushion, and pentaBDE may be detected at 0.1% by weight (Carpet Cushion Council, 2016).

Criteria for evaluating methods. We propose several criteria for evaluating the best available technologies and environmental practices for managing waste foams and plastics with toxic FRs (Table 2). Criteria for regulatory guidance on waste management are discussed in greater detail by guidance provided under the Stockholm Convention (Stockholm Convention, 2015c) and the International HCH and Pesticides Association (IHPA) report on the destruction of obsolete pesticides (IHPA *et al.*, 2008).

Collection and Pretreatments

In the United States, there has been a steady program development to collect items no longer needed by commercial and residential consumers. In general, default programs are provided at taxpayer expense by local governments. Specific programs, developed on a state-by-state basis, meet particular

TABLE 2. CRITERIA TO CONSIDER WHEN MANAGING WASTE FOAM AND PLASTIC MIXED WITH FLAME RETARDANTS

<i>Criteria</i>	<i>Description</i>	<i>Considerations</i>
Environmental and health impacts	Goal: minimize detrimental human and ecosystem impacts	Toxic byproducts—identify residue streams that may be toxic, including off-gas treatment systems (e.g., from mechanical processes, incineration, leachate from landfills, landfill off-gas, etc.). Uncontrolled releases—potential releases are identified with measures to ensure that such releases do not occur (e.g., open burning). Wastes/Residuals—includes secondary waste stream volumes/masses (e.g., dust, sludge, and residual ash) and greenhouse gases. Worker exposure—occupational exposure of workers involved with waste treatment and recycling. Environmental impact assessment—monitor discharges and residues appropriately, assess safe handling and disposal. Risks—include risks that are inherent to particular disposal methods.
Technological feasibility	Scalability of technology, and capacity and ability to treat toxics by reduction, removal, or destruction of the flame retardants.	As above: toxic byproducts, uncontrolled releases, wastes/residuals, worker exposure, environmental impact assessment, and risks.
Economic feasibility	Includes cost of avoiding negative outcomes.	Resource needs—measure annual outlay of financial resources for specific disposal methods. What advancements or changes are needed to bring costs down? What is the cost for proper monitoring? Funding sources—includes sources, as well as conditions for use of funds. Are there feasible funding mechanisms for the technology or practice? Is there any market for waste components, residuals, or energy recovery before/after treatment? Costs—estimation of costs, including waste collection, deconstruction, separation, processing, and transportation; site installation and commissioning, site preparation, monitoring, reporting, compliance, running with and without waste, and decommissioning. What is the cost of current technologies and practices?
Policy/regulatory considerations	What regulatory or legal framework is needed to ensure implementation with minimal health and environmental impact?	U.S. Regulations—What U.S. regulations currently apply to the technology or practice? Are these sufficient? Are there legal impediments to its effective use? Can international regulations produce better U.S. policies? State and local regulations—includes regulations imposed by state and local authorities. Transparency—pertains to peer sampling and external audit/review conducted by trained personnel and independent organizations (including other countries).
Societal acceptability	Can historical concerns be addressed? What is the societal context?	Previous instances of opposition—Is there a history of opposition to a disposal method, and what are the reasons? Can they be addressed? Demographic considerations—examines exposure to chemicals from the entire process and particular disposal methods.

needs where local governments have not developed programs for safety and/or financial reasons.

Collection methods

Pick-up programs. In the United States, pick-up programs are done by waste collection agencies or retailers. Bulky units too large to fit into curbside garbage or recycling bins/bags may be collected through programs like curbside collection. Collection occurs at varying frequencies, from

weekly to semiannually. Demand service includes calling a private waste hauler or municipal government office. Costs are typically borne by municipal or state governments, administered either by public employees or private vendors.

Drop-off locations. Drop-off locations are often municipally owned and operated transfer stations, where residents bring their recyclable material and sort it according to commodity type. Bulky items are collected and transported to

processing facilities for disposal. This reduces transportation costs for local governments because residents transport their own materials. Staffed transfer stations (most, but not all are) help ensure that materials are properly sorted and nonrecyclable items are not intermingled. The mattress recycling programs in California and Connecticut use this method.³ Financial incentives for drop-off can be built into the program to encourage participation.

Retail return locations. Retail returns encourage consumers to return items to a retailer at the end of its life. Participation is encouraged when financial benefits are available. Tires provide an example of a successful retail return program. Consumers are charged a recycling fee when purchasing new tires, and retailers take back old tires. However, unlike tires and mattresses, household furniture has irregular shapes that do not stack efficiently, presenting significant storage challenges for retailers.

Take-back programs are often regarded as the best solution for diverting expensive or bulky waste items from landfills. They have been successful with appropriately funded initiatives, such as recycling fees at purchase and manufacturer's product stewardship programs.

Funding mechanisms: governments

Waste management is considered a public interest; so taxpayers traditionally fund it. MSW disposal programs, initially designed to meet public health goals, have evolved to handle aluminum, glass, plastic, durable goods, and organics. Local governments, by necessity, responded to these changes. This created a mindset in the general public that local governments are best positioned to manage the ever-changing materials in waste. Local governments are often challenged to optimize collection systems for changing waste composition, creating inefficiencies. This is certainly the case with FR foams and plastics.

Funding mechanisms: extended producer responsibility. Placing the onus of waste disposal on state and local governments shifts the burden from manufacturers of products, goods, and packaging. Some manufacturers could avoid developing safer products or waste management methods for the health or environmental impacts of their products (Organization for Economic Cooperation and Development, 2007). To mitigate this, policy shifts are needed to require producers to create, fund, and implement collection programs for end-of-life products. Such programs may have an up-front fee paid by consumers at the time of product purchase (currently used in some states for paints and mattresses). Alternatively, an Extended Producer Responsibility (EPR)⁴ model (used in electronics and e-waste, but not for HFRs) could be used, whereby individual producers pay for the collection, transport, and processing associated with their products. This cost is usually borne by the consumers at the time of purchase. Solutions should consider the type and size of the retail business, since smaller retailers have more expense per item due to fewer resources for storage and transportation.

Collection and funding policies. Examples of collection programs currently in place include the California Used Mattress Recovery and Recycling Act (Cal. Public Resources Code §42985) and the California Tire Recycling Act (1989). Both require a recycling fee at purchase. The mattress program is an EPR program, mandating that manufacturers create a statewide recycling program for discarded mattresses. The tire recycling program is an example of a retail take-back model to divert waste from landfills. Neither of these programs involves waste material that is potentially hazardous for waste handlers.

Designating specific products as universal hazardous waste (u-waste) alters the requirements of hazardous waste collection and transportation. U-waste mandates could require separation of waste either through establishment of dedicated collection sites or by mandatory removal from MSW streams (as in California and certain other states). U-waste designation would impose additional expense to consumers or local governments since u-wastes are handled in specially permitted facilities. It is unlikely that foams and plastics containing FRs could easily be designated as u-waste: FR use is rarely disclosed for specific products, and a wide variety or mixture of FRs may be used and be challenging to easily identify.

Pretreatments

Electrical and electronic equipment, automobiles, and furniture contain multiple components and various materials, with a portion of the plastics and thermosets containing FRs or other hazardous materials. Recovery and separation options can include manual disassembly, shredding followed by automated separations, or a combination of these approaches.

For ELVs, the first step removes fluids, tires, batteries, airbags, and parts that can be reused or easily recycled, and valuable components such as catalytic converters. The ELVs are crushed for easy storage and transport, and finally shredded into pieces (<10 cm). Metals are automatically removed using magnets and eddy current separators (for non-ferrous metals). The remaining material, containing plastics, foam, rubber, wood, and so on, is called automotive shredder residue (ASR) (Vermeulen *et al.*, 2011). In the United States, other metal-containing products such as large appliances are often processed through the same shredder; so ASR often contains materials from these products.

For WEEE, the process is similar in that, certain components such as cords, batteries, and CRT glass are removed manually. WEEE is then sorted manually or processed using shredders and metal recovery equipment, not unlike the case for ELVs. Both approaches are used in the United States, but the latter is more common in Europe.

For furniture, the PPF can, in some cases, be readily separated. Sofa cushions may be opened manually using zippers or cutting tools, creating a highly concentrated stream of foam. Currently there is no equipment capable of shredding furniture on a cost-efficient basis.

Manual or automated separation steps for the streams above typically include steps to separate materials containing FR or hazardous materials from other materials. The Stockholm Convention implementation guidance addresses identification and separation of brominated and nonbrominated polymers for PBDE-containing material flows (Stockholm

³Mattress Recycling Council. "Bye Bye Mattress." <http://byebyemattress.com>

⁴CalRecycle EPR, Available here: www.calrecycle.ca.gov/EPR

Convention, 2015c). Proper engineering controls and personal protective equipment are necessary to protect workers.

Manual sorting of BFR materials from other materials is done visually or with rapid analytical methods. The plastic type is often stamped on parts; so technicians can isolate some materials by noting the type of plastic. XRF spectroscopy is commonly used. XRF allows detection of specific elements to ~ 10 – 100 ppm for Br and thus, it can be used to identify Br in BFR plastics, where Br levels are typically $>100,000$ ppm, or to quantify levels of Br contamination in products (Riise *et al.*, 2000). Sliding spark spectroscopy also enables the detection of Br, with a detection limit of $\sim 0.1\%$ (Seidel, 1993). XRF and sliding spark spectroscopy are not reliable means of detecting non-Br FR contaminants.

Automated sorting typically includes methods based on density or X-ray transmission. Heavy elements (including Br) strongly absorb X-rays, while lighter elements only weakly absorb (Hubbell and Seltzer, 1996). Equipment for automated sorting ejects BFR-containing pieces by blasting air at them (www.tomra.com/en/solutions-and-products/sorting-solutions/recycling/recycling-technology). Density sorting is useful because BFR plastics tend to have much higher densities than similar plastics without BFRs (Schlummer and Mäurer, 2006). Density sorting alone cannot create isolated streams of BFR or BFR-free plastics, as the separations are not perfect and some types of non-BFR plastics (e.g., polycarbonate/acrylonitrile butadiene styrene [PC/ABS]) have densities similar to BFR plastics. These methods can be used alone or in combination to produce the highest purity non-BFR streams and most concentrated streams of BFR plastics (Riise *et al.*, 2014).

Discussion and recommendations

Designing a system for safe collection, centralization, and transportation of household waste is difficult without the input of producers, retailers, and municipal officials in existing programs or others responsible for a new system. Recognizing the absence of such a dialogue, there are some core considerations for the standardization and implementation of aggressive collection systems with the primary goal of efficiently removing hazardous HFR foams and plastics from wastes.

Screening technologies need to be easy to use, reliable, and economical. XRF and sliding spark techniques appear appropriate for identification of BFRs in WEEE or foams (Freegard *et al.*, 2006).

Research needs the following:

- Explore new technologies that can reliably identify Br and non-Br FR contaminants in wastes in an effective manner.
- Research methods that can effectively separate toxic FRs in different materials.
- Monitor the air and dust at dismantling and shredding facilities, and at transfer stations for release of toxic FRs.
- Monitor collection programs for gaps in service or accessibility to marginalized communities.
- Monitor occupational exposure to FRs during waste collection, handling, and dismantling.
- Explore willingness of FR manufacturers to dialogue regarding product stewardship programs.

Landfilling

Landfills are currently the most economical method of disposal in most of the United States, and most FR-containing waste in the United States is disposed in landfills. The EPA estimates that 41.7% of selected electronics were recycled in 2015.⁵ Although landfills in developed countries typically have liners that contain waste and collect leachate, potential risks remain, including exposure of landfill workers and surrounding communities, migration of FRs into the environment by volatilization into landfill gas and leachate (both accidental seepage and incomplete attenuation), and FR migration into food.

Processes

Typically, furniture is delivered directly to a landfill, to a solid waste transfer station by self-haul, or by a collection crew as part of provided bulk collection. Furniture in the waste stream is not separated from other MSW for recycling or reuse.

MSW landfills in the United States are engineered to contain waste and separate it from the environment, capture leachate, and control gas migration (RCRA 42 U.S.C. §6901). The liner and leachate collection system described below is mandated by federal regulation (Code of Federal Regulations; 40 Parts 257 and 258; 1991).

A common system to restrict leachate migration consists of a 0.67- to 1-m-thick clay layer with a hydraulic conductivity of no more than 10^{-7} cm/s overlain with a geomembrane (GM). The GM is typically 1.5-mm-thick polyethylene with an equivalent hydraulic conductivity of about 10^{-12} cm/s. A drainage layer to collect leachate is placed above the liner, with a protective barrier above the drainage layer. Waste is placed above the protective barrier, compacted, and covered daily to minimize wind-blown refuse, odors, and the attraction of disease vectors. The daily cover is traditionally 15 cm of soil, but other materials, referred to as alternative daily cover (ADC), are also used.⁶ Examples of ADC include ash, cement kiln dust, ASR, shredded green waste, contaminated soil, textiles, and spray-on foams. The filling process continues until refuse has reached the landfill's design elevation, at which point a final cover is applied. The final cover includes, at a minimum, a layer of low permeability soil designed to minimize water infiltration, a GM, a drainage layer, and a soil layer to support vegetation to minimize soil erosion and promote evapotranspiration.

Owners manage gas and liquid leachate throughout the landfill's operation and for a postclosure period (Laner *et al.*, 2012). In the United States, this period is 30 years unless it is extended by the governing regulatory agency. Both landfill gas and leachate are migration routes for FRs into the environment. The gas has a methane content of 50% to 60%, and is typically burned in a flare or for energy generation. Gas

⁵U.S. EPA Advancing Sustainable Materials Management. Available here: www.epa.gov/smm/advancing-sustainable-materials-management-facts-and-figures

⁶Alternative daily cover (ADC) means cover material other than earthen material placed on the surface of the active face of a landfill at the end of each operating day. Alternative Daily Cover (ADC): Local Govt. Basics. Available at: www.calrecycle.ca.gov/lgcentral/basics/adcbasic.htm

collection efficiency varies from zero before gas collection system installation to 90% or more once a final cover is installed (Barlaz *et al.*, 2009).

There are three alternatives for leachate management: off-site treatment, on-site treatment, and evaporation. Most landfills transport leachate to a wastewater treatment plant (WWTP), although some pretreatment at the landfill may occur. The WWTP typically treats the leachate with its major waste stream, domestic wastewater, and discharges treated water to a surface water body in accordance with its National Pollution Discharge Elimination System (NPDES) permit. In some cases, leachate is treated at the landfill such that it can be released to surface water. At some landfills, leachate is recirculated into the waste to enhance waste decomposition and *in situ* leachate treatment (Bareither *et al.*, 2010). However, even at landfills that recirculate leachate, some leachate typically requires off-site disposal. Finally, in arid regions, leachate may be evaporated from storage ponds. Where natural evaporation is not sufficient, leachate may be evaporated using specialized equipment.

Health and environmental impacts

Some of the potential migration routes for PBDEs from landfills are particulate matter released during waste compaction, volatilization into landfill gas or dissolution into landfill leachate, and uptake into the food chain by organisms foraging in waste. Each migration route is discussed below.

FR release from landfills to air. FR release to air may result from particulate matter or landfill gas. Particulate matter is generated during waste compaction at both transfer stations and landfills. Particulates are also released when trucks are emptied. These particulates may contain FRs, and can be transported by air currents and deposited in surrounding areas. This is particularly relevant for foam products due to their low density. Danon-Schaffer (2010) found elevated concentrations of PBDEs in soil adjacent to landfills and dumpsites in various regions of Canada. Tang *et al.* (2015) found elevated PBDE concentrations in soils near landfills relative to other sites. Similar results were found in soils around open disposal sites in five developing Asian countries (Eguchi *et al.*, 2013).

Studies from open dumps show elevated levels of FRs and other POPs in humans living nearby (Kunisue *et al.*, 2006; Qu *et al.*, 2007; Athanasiadou *et al.*, 2008). These studies were conducted in developing countries, and limited studies on worked exposure exist in developed countries. One study evaluated the association between residential proximity to MSW landfills and transfer stations, and blood serum levels of PBDEs in 923 California adult women in the same occupation (Liu *et al.*, 2016). Those living <2 km from a landfill had significantly higher levels of BDE-47 and BDE-100 than those who lived >10 km from a landfill. The relationship between the time of exposure, facility operational practices, types of waste accepted, and cover materials has not been studied.

Volatilization is also a potential route of PBDE migration, especially as debromination to higher vapor pressure congeners is possible in the reducing environment of a landfill. Chamber studies with BFR-containing materials reported volatilization of BFRs (Kemmlin *et al.*, 2003; Kajiwara and

Tagigami, 2013). Furthermore, the high temperatures typically found in landfills (37°C to >40°C) further increase volatilization. For the landfill gas that is collected, PBDEs present in landfill gas will be at least partially destroyed by combustion. The extent of PBDE combustion in landfill gas has not been investigated. More significant is the fact that there are fugitive emissions associated with landfills, as not all generated landfill gas is collected and treated (Barlaz *et al.*, 2009).

Open burning of solid waste it is not allowed in engineered landfills. Nonetheless, there are occasionally fires at landfills. Such fires can release BFRs, and incomplete combustion releases dioxins and furans. In the United States, landfill fires do occur and represent an upset condition, but BFR and by-product emissions have not been quantified.

Another potential source of airborne BFRs is from the use of ASR as ADC. ASR consists of seats, floor mats, and other miscellaneous nonrecyclable material, and may also include nonmetals from the shredding of large appliances. Kajiwara *et al.* (2015) analyzed BFRs in ELVs in Japan and found that seat fabric and filler had the highest BFR content, up to 5% by weight. ASR contains many small particles capable of traveling long distances. Exposing ASR to sunlight can debrominate BFRs (Söderström *et al.*, 2004), making compounds more volatile and soluble.

Most air emission studies focused on landfills, with little attention devoted to transfer stations or processing facilities where some bulk items are broken down and compacted, potentially creating elevated concentrations of BFR-containing particulates. Multiple studies focused on worker exposure in the electronics dismantling industry in developing countries (Qu *et al.*, 2007; Athanasiadou *et al.*, 2008; Julander *et al.*, 2014). However, the potential exposure of solid waste workers has been largely overlooked.

FR release from landfills by water. Most FR-containing consumer products were introduced in the 1970s and liners in MSW landfills did not become a common practice until the 1980s–1990s. Thus, there is likely some release of FRs from landfills that were built without leachate collection systems, although the bulk of FR-containing wastes in the United States is likely disposed in lined landfills. Studies have reported BFRs and phosphorous FRs in landfill leachate and groundwater pollution plumes from unlined landfills (Barnes *et al.*, 2004; Eggen *et al.*, 2010). While there is some risk of accidental leachate release (i.e., liner failure and leachate seeps), this is likely to be relatively low.

Chemical and biological transformation of PBDEs (including debromination) in landfills affect their fate and transport. Hydroxylation and methylation are important degradation pathways for PBDEs (Ueno *et al.*, 2008). Hydroxylated PBDEs are more soluble and thus have a higher potential to be in leachate. Hydroxylated PBDEs can have higher toxicity compared to their parent PBDE (Su *et al.*, 2014), and some exhibit dioxin-like toxicity (Su *et al.*, 2012). Danon-Schaffer and Mahecha-Botero (2010) modeled e-waste debromination in landfills and concluded that centuries are sufficient to debrominate waste. While the presence of PBDE daughter products may increase with time, leachate generation decreases to de minimus quantities over time at landfills that have been closed in accordance with regulations that require the installation of a low permeability cover

system designed to eliminate infiltration (Bareither *et al.*, 2010; Laner *et al.*, 2012).

In the case of engineered landfills, leachate discharged by approved means likely poses a greater concern. The presence of BFRs in landfill leachates has been demonstrated in multiple studies. A study of seven landfills in Japan found PBDEs at relatively low concentrations, but showed high levels of TBBPA (Osako *et al.*, 2004). Landfills in Europe had BFRs in leachate, some in $\mu\text{g/L}$ levels (Öman and Junestedt, 2008). In a survey of Canadian landfills, all 27 leachates tested contained BFRs, mainly at ng/L levels (Li *et al.*, 2012). BFRs were also detected in leachates in the United States (Oliaei *et al.*, 2010). The range of values reported is broad (<1–133,000 ng/L), and the waste disposal practices of the country where the study was conducted should be taken into consideration (Stubbings and Harrad, 2014).

Collected leachate is often sent to a WWTP, but these plants are not equipped to remove contaminants such as PBDEs. At WWTPs, FRs can concentrate in the biosolids (Kim *et al.*, 2014), but discharge from final effluent is also evident (Melymuk *et al.* 2014). The most common disposal alternatives for biosolids are landfill disposal and land application. Land-applied biosolids can leach FRs with runoff and infiltration into surface water and groundwater. FRs have also been shown to bioaccumulate in organisms such as earthworms collected at biosolids application sites (Kinney *et al.*, 2008; Navarro *et al.*, 2016).

Some landfills have a NPDES permit to treat and release leachate to the environment. Common treatment processes include metals precipitation and biological treatment to decrease organics and oxidize ammonia. The extent of FR removal from landfill leachate in these processes is largely unstudied. Japanese leachate treatment systems demonstrated good removal of BFRs (Osako *et al.*, 2004). However, those systems included activated carbon adsorption, with one plant also using Fenton's advanced oxidation, a peroxidation process.

There are no data to compare FRs in WWTPs that do and do not receive landfill leachate. Some FRs detected at WWTPs could result from the disposal of water used to launder items that contain FRs, washing off household dust from FR-containing objects, and a variety of other pathways (Schreder and La Guardia, 2014; Saini *et al.*, 2016). Therefore, the relative importance of landfill disposal of FRs on ultimate release of these chemicals through wastewater effluent and biosolids is not yet established.

The presence of moisture and temperatures of $\sim 40^\circ\text{C}$ in a landfill may accelerate PBDE leaching. Landfill leachate contains dissolved humic matter, which was shown to increase PBDE leaching from plastics in a laboratory-scale study (Choi *et al.*, 2009). Osako *et al.* (2004) showed that landfills that receive higher organic content waste leached higher concentrations of BFRs, and suggested that dissolved organic matter enhanced mobilization. The ability of organic matter to enhance dissolution and mobility of hydrophobic organic chemicals is a recognized phenomenon.

FR uptake from landfills into foods. Several studies suggest that PBDEs can be taken up by organisms that forage for food at landfills, migrating them further into the food web. One study showed that PBDE concentrations in eggs of European starlings nesting near landfills contained orders of magnitude higher concentrations of PBDEs than eggs col-

lected elsewhere (Chen *et al.*, 2013). The authors hypothesize that omnivorous/insectivorous starlings may be exposed to PBDEs by either feeding on insects collected at the landfill or directly foraging for food from the refuse. Worms and other insects collected by starlings can have PBDEs on the surface from direct contact with PBDE-containing materials or ingesting food materials contaminated with PBDE.

Gentes *et al.* (2015) reported that individual ring-billed gulls nesting in the Montreal area routinely visited a landfill. Those birds had elevated BFR levels compared to birds that did not forage there. Finally, Tang *et al.* (2015) found that muscle tissue collected from 37 Eurasian tree sparrows had higher concentrations of PBDEs in samples collected near landfill sites relative to samples collected at urban and industrial sites. They showed a correlation between the concentration of PBDEs in sparrow tissue and the soil where the sparrow was collected.

Discussion

Disposal of BFR-containing consumer products with MSW has the potential for environmental release of BFRs into the air, water, and food chain. Leachate is treated by methods that typically do not remove BFRs, and treated leachate is ultimately released to surface water. Understanding the mechanisms by which BFRs are released from solid waste operations is important for finding ways to minimize environmental impacts.

Short-term management options

While it may be desirable to phase out the use of HFRs, for the short-term, it is important to manage the disposal of products currently in circulation in a manner that is protective of human health and the environment, as well as to minimize HFR release from unlined landfills. The recommendations presented in this study assume that landfilling of products containing HFRs will continue.

Designate as hazardous waste. Hazardous waste landfills have more rigorous requirements for leachate containment. However, the construction and operation of hazardous waste landfills are costlier than conventional landfills. Considering the large volumes and spatial distribution of consumer products that may eventually become HFR-containing waste, there would be considerable cost involved in treating this waste as hazardous. Proper disposal of household hazardous waste is at the discretion of the residential waste generator and is difficult to enforce. Finally, disposal as a hazardous waste would result in additional transport and energy consumption and associated emissions.

Separation at MSW landfill. HFR-containing materials could be disposed in landfills that do not contain organics that attract insects, rodents, and birds. Despite the ADC that minimizes nocturnal animal foraging, daytime animal exposure cannot be fully eliminated. Placing HFR-containing items in a separate landfill area would prevent exposure to leachate containing dissolved humic matter that helps mobilize HFR. A reduction in BFR leachate concentrations would reduce the release of BFRs from treated leachate. In addition, disposal of BFR-containing waste in monofills would reduce or eliminate gas production and subsequent

fugitive emissions. Separation of BFR-containing products may be achieved at transfer stations as most of the items are fairly bulky (furniture, appliances) and do not easily blend with other waste. However, smaller items (electronics and sofa cushions) could still be easily disposed by consumers with mixed MSW. The use of ASR as ADC should be evaluated for HFR content and leaching, and if present, its use discontinued or limited to portions of landfills that only receive HRF-containing waste.

Construction and demolition (C&D) debris could be separated into two streams depending on whether the item contains HFRs, but additional labor and worker training would be necessary. Identification and potential separation of MSW containing non-Br FR contaminants remain problematic.

Additional costs for dedicated landfill sections (or monofills) would lead to resistance from landfill operators unless additional funding was provided or costs transferred to consumers. If funding for disposal was provided by the manufacturers producing FR-containing products, these costs would likely be passed to the consumer.

Dispose with MSW, but improve landfilling practices. Operational changes could be made to reduce emissions, but the changes are complex, difficult to implement, and may result in other complications. For example, waste could be wetted before compaction to reduce dust, but this creates a large water (or other liquid) demand and the need for operators to wet the waste under all weather conditions. Excessive water could increase fugitive gas emissions and odors. Additional leachate treatment may be possible, but it is important to analyze whether this treatment should be implemented at a landfill or at the WWTP. More complete gas collection is possible, but analyses are required to quantify the potential benefit.

Any policy or requirement to reduce the release of BFRs from landfills would require new, evidence-based regulations for the disposal of materials containing FRs on a state or national level. While FRs in the environment are a global issue, initial regulatory actions may be more easily achieved at the state level. Some states have agencies that regulate generation and disposal of FRs (e.g., the California Department of Toxic Substances Control). However, consistent national regulations based on research on safe disposal of BFR-containing materials would be desirable.

The protective management of HFRs in solid waste will require cooperation from stakeholders, including the solid waste industry, HFR manufacturers, and federal and state agencies. Benefits and costs of short-term alternatives must be thoroughly evaluated and research is needed to inform better short-term and long-term policies.

Recommended research

- Quantify HFRs in landfill gas
- Determine destruction efficiencies in landfill gas treatment systems (e.g., flares and engines), and byproduct formation during combustion, landfill fires, and evaporation of landfill leachate.
- Quantify the presence and leaching of HFRs from shredder residue and C&D fines used as ADC.
- Quantify HFRs in leachate from newer engineered landfills and in leachate from older, unlined landfills.

- Determine if additional treatment processes are required for BFRs in wastewater treatment.
- Evaluate occupational exposure to FRs for solid waste workers.

Mechanical Recycling of FPF to Reduce FR Content

Mechanical recycling of FPF is an efficient and effective method of reducing FR content in FPF, although FRs cannot be eliminated through recycling. Mechanical recycling of FPF is accomplished by collecting and grading scrap from various sources, grinding the scrap into small pieces, and processing these into bonded carpet cushion. Bonded cushion is manufactured by blending postindustrial and postconsumer scrap with a polymeric binder under pressure and steam. In 2015, from U.S. sources, ~363,000 tonnes of FPF waste (combined postindustrial and end-of-life postconsumer scrap) was recovered and about 90,718 more tonnes of scrap was imported mainly from Europe.⁷

Methods

Since the introduction of polyurethane foam carpet cushion in the late 1960's, bonded carpet cushion has overtaken other materials, achieving a 90% market share.⁸ The supply chain for bonded cushion consists of postmanufacturing FPF trim scrap and end-of-life, postconsumer FPF scrap. Trim scrap ("take up") primarily comes from fabrication of upholstered furniture and automotive interior parts (domestic and imported). U.S. postconsumer scrap ("take up") mainly comes from recovered used carpet cushion and, more recently, the supply chain also includes end-of-life mattresses. ASR is not a significant source of postconsumer foam scrap. Postconsumer scrap fills two critical needs: an adequate supply of suitable postindustrial trim scrap to meet U.S. market demands and the production of higher density grades of bonded cushion.

The largest source of postconsumer scrap is "take up" carpet cushion collected during new carpet installation. "Take up" gathered from a number of different sources' jobs is consolidated by a recycler or scrap foam broker and taken to bonded foam manufacturers. There, preprocessing includes a visual inspection, grading, and/or magnetic screening to eliminate foreign objects. Bales typically contain foam scrap of differing color, scrap size, density, and hardness. Postindustrial trim scrap also is graded, including estimating density and hardness. There currently is no practical way to analyze scrap for FR content. Grading relies heavily on knowledge of source product use and typical foam specifications for such products. Postconsumer "take up" scrap and some trim scrap from certain automotive, medical, and upholstered furniture sources may contain FR additives, and, when suspected of having FR content, is often segregated for "blend down" formulation to reduce FR levels in new bonded carpet cushion. "Blend down" involves blending scrap having suspected FR contamination with non-FR foam scrap, typically obtained from postindustrial trim sources. Grading

⁷Estimates are based upon input from Carpet Cushion Council cushion manufacturers (March 2016) and carpet sales data from Floor Covering Weekly (July 25, 2016).

⁸Carpet Cushion Council. December 2016. Sales Statistics Report.

is followed by mechanical processing, including grinding the scrap into smaller pieces, but large enough for good binding and support. The pieces are blended based on a “recipe” for the type of cushion being manufactured. A polymer binder is added for adhesion and extra support. This mixture is bonded under high-pressure steam into blocks or “logs,” and cured before being sliced or peeled into a commercial product (ISOPA, 2001).

FRs in some trim scrap and postconsumer scrap create challenges for bonded foam manufacturers. In the absence of practical tools for quantifying FR levels in scrap foam, they must assume maximum FR content in scrap suspected of FR content. In “take up” scrap, pentaBDE could be as high as 5%, with organophosphate esters as high as 12% (by weight) (Luedeka, 2012). Foam with theoretical FR content is “blended down” with known “clean” scrap to achieve lower levels of FRs. In states that regulate content, the objective is no more than 0.1% by weight. The carpet cushion industry currently employs random inventory sampling and third-party GC/MS testing to demonstrate compliance.

Impact of FRs

Beginning in the 1970’s, FR requirements for upholstered furniture influenced FPF cushion content and, through subsequent use of industrial trim scrap, the content of bonded carpet cushion. In 1975, California’s TB-117 required foam and other resilient materials in upholstered furniture to pass a small open-flame test, effectively mandating the addition of FR chemicals for California’s furniture foam. Furniture sold outside of California also frequently contained FR foam because furniture companies sought to maintain uniform national manufacturing and distribution programs.

Eventually, health and environmental characteristics of FRs came under scrutiny and the overall benefits of TB-117 were questioned. Beginning in 2005, a dozen states and the District of Columbia banned or restricted FRs in various household products. States generally allow regulated products to contain a de minimis FR level of 0.1% to account for unintended impurities. In 2004, the U.S. foam industry voluntarily discontinued the use of pentaBDE, and in 2013, California revised TB-117 to allow non-FR upholstered furniture to be sold.

If there is not sufficient “clean” scrap for blending, suspected FR scrap may need to be disposed. Unfortunately, current alternative disposal methods are limited and land-filling is sometimes the only available option. The Healthy Building Network suggests testing for a wide range of FRs in recycled FPF feedstocks (Stamm, 2016). Currently, analytic testing of scrap feedstock for FRs requires off-site GC/MS analysis and is not logistically or economically feasible. The random nature of postconsumer scrap makes sampling additionally challenging.

Health and environmental impacts

There are few studies of potential FR exposure among carpet cushion installers and bonded cushion production workers. In 2004,⁹ ISO16000 (emission chamber) testing on a

selection of bonded carpet cushion was performed over 72 h at 23°C/50% relative humidity, with an air exchange rate of 0.5/h and a loading of 0.4 m²/m³. Air samples had no detectable pentaBDE.

Stapleton *et al.* (2008) measured serum PBDE concentrations for foam workers ($n=12$), carpet layers ($n=3$), and a control group ($n=5$). Total PBDE concentrations were 160, 178, and 19 ng/g lipid, respectively. While the sample size was small, foam workers and carpet layers had levels nearly an order of magnitude higher compared with the general U.S. population (measured by National Health and Nutrition Examination Survey [NHANES]). The change in the mix of FRs used commercially since 2008, along with the small study size, argues for further research.

California identified TDCPP as a substance known to cause cancer in humans. A Maximum Allowable Dose Level of 5.4 µg/day was established (California Office of Environmental Health Hazard Assessment [OEHHA]). Exposures associated with the production or removal of TDCPP contaminated carpet cushion are unknown. The volatility characteristics of possible TDCPP impurities in bonded carpet cushion have not been determined.

Recommended research

- Develop practical and cost-effective methods to identify and quantify FRs in contaminated foam waste and bonded cushion products.
- Find or develop new sources of “clean” scrap as a bonded foam blending feedstock as upholstered furniture manufacturing moves overseas. Increase amount “clean” foam from mattress recycling programs.
- Quantify pentaBDE substitutes in existing carpet cushion to support future blend down recipes.
- Develop technologies and logistical support to help manage future recovery and recycling of current upholstered furniture that may contain FRs.
- Monitor carpet cushion installers and bonded cushion production workers to assess their risk to FR exposure.

Concluding Remarks for Part I

Part II of this review is published separately and continues the discussion of other technologies, policy review, and a summary. In Part I, we focused on the challenges presented in handling waste products that contain HFRs. HFRs are present in many products that we are exposed to daily, including furniture, foam, vehicles, electronics, selected textiles, and building materials. At present, most HFR-containing wastes are disposed in landfills. While there is evidence to suggest the release of HFRs from the solid waste matrix to air and water, our understanding of HFR release from landfills and the severity of HFR releases is limited and uncertain. Future research is essential to evaluate the costs and benefits associated with either diverting HFR-containing materials from landfills or altering landfill disposal practices to reduce HFR releases, and the associated harm to human health and the environment. Concurrent with the study of landfill disposal, policies to improve the controlled collection of HFR-containing wastes should be developed. In Part 2 of this review, we explore alternative technologies for the management of HFR-containing wastes, including chemical, mechanical, and thermal processes for recycling, treatment, and disposal.

⁹“Volatility of Penta Bromodiphenylether (PentaBDE),” Carpet Cushion Council, the proceedings of the Polyurethane Foam Association Technical Conference, October, 2004.

Acknowledgments

This material is based upon work supported by the National Science Foundation (www.nsf.gov) under NSF award number 1500091. Research reported in this report was supported by *National Institute of Environmental Health Sciences* of the National Institutes of Health under award number R13ES025490.

Author Disclosure Statement

No competing financial interests exist.

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