Costs and Benefits of the Addition of Hexabromocyclododecane (HBCD) to the Stockholm Convention and the 1998 POPs Protocols
Executive summary

This project is aimed at providing information to Defra on the costs and benefits of the addition of hexabromocyclododecane (HBCD) to the Stockholm Convention on Persistent Organic Pollutants (POPs) and the Protocol on Persistent Organic Pollutants to the UNECE Convention on Long-Range Transboundary Air Pollution (CLRTAP). The project provides information on HBCD in the UK context to Defra for use in developing policy on this substance.

The project also provides information in relation to potential controls on HBCD under Regulation (EC) No 850/2004 on Persistent Organic Pollutants. To do this, the project has considered the cost implications of restrictions on emissions in waste, and the costs of monitoring and reporting under these regulations.

The outputs of the research are designed to inform Defra in its negotiations in relation to HBCD under the Stockholm Convention and the POPs protocol. This report provides information to enable Defra to consider the annexes under each agreement that HBCD should be listed under. The decision in this regard will determine whether HBCD needs to be eliminated entirely or whether it needs to be restricted to a limited range of critical applications. It will also determine whether the unintentional production of HBCD needs to be addressed. Defra will need to consider whether the UK should seek any derogation for the use of HBCD, and will need to understand the impact of any restrictions on HBCD on UK industry.

This project contains the following components:

- Evaluation of HBCD against Stockholm Convention criteria
- Evaluation of sources, uses and controls of HBCD in the UK
- Cost-benefit analysis and partial impact assessment of applying additional controls to HBCD
- Completion of UNECE questionnaire on production and use of HBCD

Hexabromocyclododecane (HBCD) is a brominated organic chemical with the chemical formula $\text{C}_{12}\text{H}_{18}\text{Br}_6$. The chemical structure of $1,2,5,6,9,10$ HBCD is shown below.
HBCD is a flame retardant used mainly in thermal insulation in the construction industry. It is used to a lesser extent in textile coatings and in high-impact plastics in the computer industry.

Evaluation of HBCD against Stockholm Convention criteria

Chemical substances can be listed under the Stockholm Convention on the basis of their persistence in the environment; their potential to accumulate in the food chain; their potential for long-range transport in the environment; and their potential for adverse effects on human health and the natural environment. The Convention sets out qualitative and quantitative criteria for these aspects. HBCD was assessed against these criteria to reach a view on whether listing under the Stockholm Criteria should be supported by Defra.

- Persistence: On balance, HBCD meets the criteria of the Stockholm Convention with regard to persistence
- Bio-accumulation: HBCD meets the criteria of the Stockholm Convention with regard to bio-accumulation
- Long-range transport: HBCD meets the criteria of the Stockholm Convention with regard to long-range transport
- Adverse effects: HBCD does not meet the criteria of the Stockholm Convention with regard to the potential for damage to human health, taking account of plausible exposure. It is uncertain whether HBCD meets the criteria of the Stockholm Convention with regard to the potential for damage to the environment

It is concluded that HBCD does not demonstrably meet the criteria for adverse effects on the environment set in the Convention. However, if a precautionary approach is adopted with regard to the potential for environmental harm, this would suggest that HBCD could provisionally be considered a Persistent Organic Pollutant under the terms of the Stockholm Convention. Further research should focus on confirming the range of exposure of sensitive species to HBCD in the environment, and should also consider the potential adverse effects of the main chemical sub-species of HBCD.

Evaluation of sources, uses and controls of HBCD in the UK

It is estimated that approximately 1,000 tonnes of HBCD are used per year in the UK. About 90% of HBCD is used in the construction industry, with the balance used in the textiles and electronics industries.

Releases of HBCD to the environment were estimated for 2010 and for 2030, assuming no further restrictions on the use of HBCD (i.e. “business as usual”). The most significant environmental releases of HBCD were estimated to be as follows:

- Disposal to land of polystyrene insulation board is forecast to become increasingly significant up to 2030. In 2030, disposal to land of polystyrene insulation board is estimated to account for 49% of all HBCD releases
- The next highest estimated release is emissions to air of HBCD from textiles, estimated to account for 18% of all HBCD releases in 2030.
- Disposal to landfill of waste polystyrene insulation board during installation is estimated to account for 13% of all HBCD releases in 2030.
- Disposal of end-of-life textiles containing HBCD to landfill is estimated to account for 12% of all HBCD releases in 2030.

At present, there are no specific controls on HBCD during the manufacture, use and disposal of goods containing HBCD in the UK. Controls may be forthcoming under the terms of the Long-Range Transboundary Air Pollution (LRTAP) Convention. A number of regimes could potentially be used to introduce controls on HBCD, if required.
• The Environmental Permitting regime, covering the disposal of waste materials which could contain HBCD
• The Waste Electrical and Electronic Equipment (WEEE) Regulations, covering the recovery and recycling of plastics from electrical and electronic goods containing HBCD
• The REACH regulations provide the framework for authorisation of chemicals throughout Europe. HBCD is under consideration for authorisation under the terms of REACH.
• The Restriction of Hazardous Substances directive prevents equipment containing hazardous chemicals from being marketed in the EU. HBCD could potentially be included in the scope of this directive.
• The brominated flame retardant industry operates a scheme for management of HBCD emissions from industrial processes which use HBCD.
• Other existing voluntary environmental management programmes in the key industries could potentially be applied to the control of HBCD.

Cost-benefit analysis of applying additional controls to HBCD

Five scenarios were identified for potential approaches to control of HBCD under the Stockholm Convention:

• Scenario 1: Complete ban on HBCD use in the UK, with the introduction of safe disposal requirements
• Scenario 2: Restriction on HBCD use in the UK, with an exemption for critical uses in the textiles industry
• Scenario 3: Restriction on HBCD use in the UK, with an exemption for the use of HBCD in building cavity insulation (this application is currently declining)
• Scenario 4: Restriction on HBCD use in the UK, with an exemption for the use of HBCD in internal and external cladding insulation (this application is currently growing)
• Scenario 5: No restriction on HBCD use in the UK, but an introduction of safe disposal requirements

The costs and benefits of each scenario were assessed for the key sectors in which HBCD is used at present.

• Scenario 1 (Complete ban and safe disposal requirements): this is estimated to result in a reduction in emissions of HBCD of 730 tonnes per year. This scenario would result in increased costs in separating materials containing HBCD from construction waste; reduction of employment in insulation manufacture; loss of employment in textiles industry; increased cost of separation of electrical/electronic waste; increased cost of incineration of textile products; reduction in consumer choice for textile products; negative impact on competitiveness of electronics industry; and reduced building energy efficiency

• Scenario 2 (Ban on most uses and safe disposal requirements; derogation for textiles): this is estimated to result in a reduction in emissions of HBCD of 495 tonnes per year. This scenario would result in increased costs in separating materials containing HBCD from construction waste; reduction of employment in insulation manufacture; increased cost of separation of electrical/electronic waste; negative impact on competitiveness of electronics industry; and reduced building energy efficiency
Scenario 3 (Ban on most uses and safe disposal requirements; derogation for cavity insulation): this is estimated to result in a reduction in emissions of HBCD of 480 tonnes per year. This scenario would result in increased costs in separating materials containing HBCD from construction waste; smaller reduction of employment in insulation manufacture; loss of employment in textiles industry; increased cost of separation of electrical/electronic waste; increased cost of incineration of textile products; reduction in consumer choice for textile products; negative impact on competitiveness of electronics industry; and smaller reduction in building energy efficiency.

Scenario 4 (Ban on most uses and safe disposal requirements; derogation for cladding insulation): this is estimated to result in a reduction in emissions of HBCD of 480 tonnes per year. This scenario would result in increased costs in separating materials containing HBCD from construction waste; smaller reduction of employment in insulation manufacture; loss of employment in textiles industry; increased cost of separation of electrical/electronic waste; increased cost of incineration of textile products; reduction in consumer choice for textile products; negative impact on competitiveness of electronics industry; and smaller reduction in building energy efficiency.

Scenario 5 (Safe disposal requirements only): this is estimated to result in a reduction in emissions of HBCD of 490 tonnes per year. This scenario would result in increased costs in separating materials containing HBCD from construction waste; increased cost of separation of electrical/electronic waste; increased cost of incineration of textile products; increase in employment in waste/recycling industry, and an increase in carbon footprint of plastics industry unless appropriate market conditions are facilitated.

Completion of UNECE questionnaire on production and use of HBCD

The questionnaire was duly completed, drawing on information gathered during the course of the project, and is provided in Appendix 1 of this report.
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Appendix 1  Annex I Questionnaire on production and use of HBCD
Appendix 2  Stockholm Convention Annex D
Appendix 3  Measured levels of HBCD isomers and degradation products in the UK
Appendix 4  Health effects of HBCD
1 Introduction

1.1 Background

This project is aimed at providing information to Defra on the costs and benefits of the addition of hexabromocyclododecane (HBCD) to the Stockholm Convention on Persistent Organic Pollutants (POPs) and the Protocol on Persistent Organic Pollutants to the UNECE Convention on Long-Range Transboundary Air Pollution (CLRTAP). The project provides information on HBCD in the UK context to Defra for use in developing policy on this substance.

The project also provides information in relation to potential controls on HBCD under Regulation (EC) No 850/2004 on Persistent Organic Pollutants. To do this, the project has considered the cost implications of restrictions on emissions in waste, and the costs of monitoring and reporting under these regulations.

The outputs of the research are designed to inform Defra in its negotiations in relation to HBCD under the Stockholm Convention and the POPs protocol. This report provides information to enable Defra to consider the annexes under each agreement that HBCD should be listed under. This will determine whether HBCD should be dealt with by elimination, restriction or minimisation of unintentional production. Defra will need to consider whether the UK should seek any derogation for the use of HBCD, and needs to understand the impact of any restrictions on HBCD on UK industry.

1.2 Hexabromocyclododecane

1.2.1 Chemical structure

Hexabromocyclododecane (HBCD) is a brominated organic chemical with the chemical formula $\text{C}_{12}\text{H}_{18}\text{Br}_6$. Commercially produced HBCD is $1,2,5,6,9,10$-hexabromocyclododecane. The chemical structure of $1,2,5,6,9,10$ HBCD is shown in Figure 1.

*Figure 1: Chemical structure of hexabromocyclododecane*
1.2.2 Stereoisomers of HBCD

1,2,5,6,9,10-HBCD can theoretically exist in 16 different forms with the atoms connected as shown in Figure 1, but oriented in different ways relative to each other. These forms are known as “stereoisomers.” These 16 stereoisomers comprise 8 pairs of mirror images. Mirror image pairs are known as “enantiomers” and are denoted as “+” or “−”. Each pair of enantiomers is denoted with a Greek letter. These stereoisomers have different environmental and toxicological properties.

The most common stereoisomers of HBCD are α, β and γ-HBCD. Commercial formulations of HBCD contain the following stereoisomers at the following approximate relative abundances: α-HBCD (8-9%), β-HBCD (6%), and γ-HBCD (80-85%) (Albemarle Corporation, 2005). Minor constituents also reported to be present are δ-HBCD (0.5%) and ε-HBCD (0.3%) (Heeb et al, 2005).

Transformation of the stereoisomer pattern is known to occur from that present in the commercial HBCD formulations (predominantly γ-HBCD) to those found in indoor air and dust (broadly equal contributions from α- and γ-HBCD, with typically 10% β-HBCD), and in biota (including humans), where α-HBCD is the predominant stereoisomer. HBCD in indoor dust is likely to originate from abrasion of treated textiles, or abrasion of plastics. Tracing the pattern from source to humans, it is known that isomerisation occurs at the temperatures (>160˚C) required to incorporate HBCD formulations within products, and that this results in a partial shift from γ- to α-HBCD (e.g. Peled et al, 1995). The relative abundance of β-HBCD is essentially unaffected during this process.

This shift is reflected in the stereoisomer pattern observed in indoor dust in which the average abundances are: 33% α-, 11% β-, and 56% γ-HBCD (Abdallah et al, 2008a). The pattern in matched indoor air samples contains significantly more γ-HBCD (65%) and less α-HBCD (22%). As discussed below, this is because of degradation occurring within indoor dust (Harrad et al, 2009a). Such variable contributions from α- and γ-HBCD are observed in UK outdoor air, soil, lake sediment, and freshwater (Harrad et al, 2010b). In UK fish and particularly marine mammals, there is a further shift towards the dominance of α-HBCD (Harrad et al, 2009b; Law et al, 2008). Finally, data has just emerged on concentrations and stereoisomer patterns of HBCD in human milk (Harrad and Abdallah, 2010). These data show that in UK human milk, α-HBCD comprised 62-95% of total HBCD isomers while β- and γ-HBCD ranged from 2-18% and 3-33% of total HBCD respectively. This is consistent with observations from other countries.

The predominance of the α-diastereomer in human tissues has been suggested to result from preferential metabolism of β- and γ-HBCDs and transformation to α-HBCD (Zegers et al, 2005). However, there are data to suggest that other factors may contribute to the prevalence of α-HBCD in humans. These include intake via dust (Roosens et al, 2009) which contains a substantial proportion of α-HBCD, as well as the physicochemical properties of α-HBCD such as water solubility that may render α-HBCD more bioavailable than the other isomers (Abdallah et al, 2009). Higher adipose tissue deposition and lower faecal elimination rate has been observed for α-HBCD compared to γ-HBCD in female mice (Brandsma et al, 2009). In around half of the fish analysed in a study of UK lakes, the δ-HBCD stereoisomer was detected at abundances of 1 – 11%, much higher than the levels in the commercial formulation (Harrad et al, 2009b). This may indicate bio-isomerisation within fish of δ-HBCD. The toxicological significance of this is unknown.

A summary of measured levels of HBCD stereoisomers is given in Appendix 3.

1.2.3 Uses of HBCD

HBCD is a flame retardant used mainly in thermal insulation foams in the construction industry. It is used to a lesser extent in textile coatings and in high-impact plastics in the computer industry.
As noted above, technical grade HBCD comprises mainly γ-HBCD with lower levels of α- and β-HBCD. δ- and ε-HBCD are present at levels below 1% in technical HBCD.

The main use of HBCD is in Expanded Polystyrene (EPS) and Extruded Polystyrene (XPS) insulation foam boards. These boards are used to improve thermal insulation of buildings. These foam boards are required to meet stringent fire safety regulations EU countries including the UK. HBCD provides the required level of performance, at a relatively low concentration. XPS is a more dense product than EPS, with better thermal insulation properties. EPS is about one-third the cost of XPS, and can be used where space permits. XPS is more appropriate in areas where space for insulation is limited.

HBCD is also applied in the back coating of textiles, mainly for upholstered furniture and in vehicle upholstery. The use of HBCD enables furniture manufacturers to meet fire safety requirements.

HBCD is used as an additive in the manufacture of high-impact polystyrene (HIPS). This is used in electrical and electronic equipment and appliances.

There are no significant unintentional production sources of HBCD.

### 1.3 Policy Context

#### 1.3.1 Stockholm Convention

The policy context for this work is set within the UNEP Stockholm Convention, which the UK has ratified. Amongst other commitments, the Convention requires the development of release inventories for a listed group of Persistent Organic Pollutants (POPs) to air, land and water. Under the original text of the convention, 12 substances were to be covered, the majority of which are dispersed within the environment as a consequence of their use or disposal following commercial manufacture. The manufacture or sale of these substances has not occurred in the UK for a considerable period of time; consequently, the only substances of present interest for release inventories are those POPs controlled under the Stockholm Convention and which are produced unintentionally (U-POPs); these are the polychlorinated-p-dioxins, polychlorinated dibenzofurans, polychlorinated biphenyls (PCBs) from combustion processes, and hexachlorobenzene (HCB). More recently, a further nine substances and substance groups have been added to the convention.

On 18 June 2008, Norway submitted a proposal to list the brominated flame retardant hexabromocyclododecane (HBCD) as a possible Persistent Organic Pollutant under Annex A of the Convention. This substance is the focus of the present study.

#### 1.3.2 UNECE Convention on the Long-Range Transport of Pollutants

The CLRTAP Executive Body adopted a Protocol on Persistent Organic Pollutants on 24 June 1998 in Aarhus (Denmark). The objective of the protocol is to eliminate any discharges, emissions and losses of POPs. The Aarhus Protocol includes a number of requirements which are additional to those of the Stockholm Convention. In particular, the Aarhus Protocol requires signatories to dispose of the materials covered by the Protocol “in an environmentally sound manner,” and to carry out disposal within national boundaries, taking environmental considerations into account. Signatories are also required to report “information on the levels of emissions of persistent organic pollutants using, as a minimum, the methodologies and the temporal and spatial resolution specified by the Steering Body of EMEP.” This information is required for a reference year specified in the Protocol, and as required during periodic updates.

#### 1.3.3 EU Regulation (EC) No. 850/2004 on Persistent Organic Pollutants

Regulation (EC) No. 850/2004 on Persistent Organic Pollutants entered into force on the 20th of May 2004. The Regulation was enforced in the UK via the Persistent Organic
Pollutants Regulations 2007 (Statutory Instrument 2007 No. 3106). The main purpose of this Regulation is to enable the European Community to ratify the Stockholm Convention and the Aarhus Protocol. The Regulation goes beyond the provisions of the Convention and Protocol by limiting exemptions to situations “where a substance fulfils an essential function in a specific application.” This will need to be considered for the purposes of the present project. The Regulation also deals with stockpiles of redundant substances, although as HBCD has not been manufactured in the UK since 2003, this is unlikely to be a significant issue for the present project.

1.4 Project scope

This project contains the following components:

- Chapter 2: Evaluation of HBCD against Stockholm Convention criteria
- Chapter 3: Evaluation of sources, uses and controls of HBCD in the UK
- Chapter 4: Alternatives to use of HBCD
- Chapter 5: Cost-benefit analysis and partial impact assessment of applying additional controls to HBCD
- Chapter 6: Completed questionnaire on HBCD (refers to Appendix 1)
- Chapter 7: Conclusions

A number of appendices are provided which provide additional study methodological details and information.

1.5 Project team

The project was led by AEA Technology. Expertise on the environmental chemistry of HBCD was provided by Dr Stuart Harrad of Birmingham University. Expertise on toxicological issues was provided by Dr Alison Searl of the Institute of Occupational Medicine (IOM).
2 Evaluation against Stockholm Convention criteria

2.1 Introduction
The evaluation criteria for the listing of substances under the Stockholm Convention are set out in Annex D to the Convention. The text of Annex D is reproduced in Appendix 2 of this report. This Annex sets the following criteria:

(a) Persistence:
   (i) A half-life in water greater than two months, or a half-life in soil or sediment greater than six months; or
   (ii) Sufficient persistence to justify its consideration within the Convention;

(b) Bio-accumulation:
   (i) A bio-concentration factor or bio-accumulation factor in aquatic species greater than 5,000 or, in the absence of such data, a log Kow value greater than 5; or
   (ii) Other reasons for concern, such as high bio-accumulation in other species, high toxicity or ecotoxicity; or
   (iii) Monitoring data in biota indicating sufficient bio-accumulation potential to justify its consideration within the Convention;

(c) Potential for long-range environmental transport:
   (i) Measured levels of the chemical in locations distant from the sources of its release that are of potential concern; or
   (ii) Monitoring data showing that long-range environmental transport of the chemical, with the potential for transfer to a receiving environment, may have occurred via air, water or migratory species; or
   (iii) Environmental fate properties and/or model results that demonstrate that the chemical has a potential for long-range environmental transport through air, water or migratory species, with the potential for transfer to a receiving environment; and

(d) Adverse effects:
   (i) Adverse effects to human health or to the environment that justifies consideration of the chemical within the Convention; or
   (ii) Toxic or ecotoxic properties that indicate the potential for damage to human health or to the environment.

This chapter provides a review of HBCD against the listing criteria set out above, to determine whether it should be treated as a persistent organic pollutant (POP).

2.2 Evaluations of HBCD
The main source of data for this evaluation comprised evaluations prepared in relation to the proposed listing of HBCD under the Stockholm Convention, together with documents
prepared by the European Chemicals Agency in relation to the proposed registration of HBCD under the REACH regime. The key reference documents are:

- UNEP, Stockholm Convention on POPs, “Additional information submitted on hexabromocyclododecane,” September 2009
- UNEP, Stockholm Convention on POPs, “Report of the Persistent Organic Pollutants Review Committee on the work of its fifth meeting,”
- Norwegian Climate and Pollution Agency, “Exploration of Management Options for Hexabromocyclododecane (HBCD),” August 2010

These documents were reviewed. A study has also been carried out on behalf of the bromine products industry which has also been considered in this study (Arnot et al., 2009). Additionally, a literature search was carried out to investigate whether any new information has been published which would be relevant to consideration of HBCD.

2.3 Persistence

The Stockholm Convention requires consideration of persistence in water, soil and sediment.

2.3.1 Half-lives

The convention requires consideration of the half-life in water, soil and sediment. Although not specified in the convention text, the focus is normally considered to be on the primary half-life of a substance, rather than complete degradation to inorganic substances such as carbon dioxide and hydrogen bromide. In the case of HBCD, the primary degradation step is likely to be debromination with formation of a carbon-carbon double bond. This would typically be followed by further stepwise debromination to cyclododecatriene.

Because HBCD is a highly non-polar organic compound (as indicated by the relatively high $K_{ow}$ values discussed in Section 2.4), it partitions preferentially to organic phases in soil and sediment. Consequently, chemical hydrolysis of HBCD in water is unlikely. This means that the main pathway to be considered is the degradation of HBCD by micro-organisms. No biodegradation of HBCD in water was observed over a 28 day period in a study carried out in accordance with a standard methodology (Schaefer and Haberlien, 1996.) However, the test used was not appropriate for HBCD because of its low solubility. Consequently, the
concentration of HBCD in the test solution was about a thousand times higher than its solubility. This makes interpretation of the study results difficult. A similar study was carried out by a Japanese laboratory in 1990 but not reported in a peer-reviewed paper (Kurume Laboratories, 1990). This study used still higher concentrations of HBCD, and found a degradation of 1% - 6% over a 28 day period.

Modelling studies can be used to provide estimates of half-lives in the aquatic environment, but the results of these studies are sensitive to the study inputs, and cannot be specifically set up for complex cyclic chemicals such as HBCD (TemaNord, 2008; Arnot et al., 2009). This limits their reliability for HBCD.

The key laboratory studies of the degradation half-life of HBCD in sediment were carried out by Davis (2005 and 2006). Davis (2006) indicated that the half-lives in aerobic sediment were 128, 72 and 92 days for the α-, β- and γ-isomers, respectively. Shorter half-lives of 11 to 32 days were reported in Davis (2005), possibly due to the lower concentrations used in the earlier study. The difference in degradation rates between isomers was not statistically significant. Studies using sediment cores indicate that HBCD congeners deposited in marine sediments in the 1970s and 1980s are still present at detectable levels (e.g. Christensen et al. (2004) quoted in European Commission (2008)). This indicates a higher persistence in sediments than indicated by experimental studies.

Davis (2005) measured the half-life of technical HBCD in soil to be 63 days under aerobic conditions, and 7 days under anaerobic conditions. In a subsequent study, no degradation was observed over a 112 day period. Arnot et al. (2009) provided indicative half-lives based on model outputs and a review of the limited available data. These were adjusted by Hirai et al. (2010) to give isomer-specific values, as set out in Table 1. These values are subject to an estimated uncertainty factor of 3 to 10, as set out in the table. The values for water and sediment are applicable to freshwater and marine waters. The values are subject to a wide uncertainty factor, reflecting experimental uncertainty and the variety of conditions that could be encountered in the environment.

Table 1: Estimated half lives for HBCD (Hirai et al. 2010)

<table>
<thead>
<tr>
<th>Isomer</th>
<th>Estimated half life (days)</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Air</td>
</tr>
<tr>
<td>α-HBCD</td>
<td>0.4 – 4</td>
</tr>
<tr>
<td>β-HBCD</td>
<td>0.4 – 4</td>
</tr>
<tr>
<td>γ-HBCD</td>
<td>0.4 – 4</td>
</tr>
<tr>
<td>Uncertainty factor</td>
<td>3</td>
</tr>
</tbody>
</table>

These are considered to be the best available estimated half-lives. The uncertainty factors quoted for soil and sediment are high, and it is considered likely that the actual half-lives in soil and sediment are closer to the values in Table 1 than indicated by the wide uncertainty factors proposed by Arnot et al. (2009) and Hirai et al (2010). The European Commission (2009) found that the half-lives in sediment are approximately 210 days (α-HBCD), 130 days (β-HBCD) and 190 days (γ-HBCD). These values are towards the middle of the ranges quoted by Hirai et al. (2010).

In indoor dust, photolysis appears to be an important degradation mechanism for HBCD. The half-life of total HBCD in the presence of light was 12 weeks, compared to 26 weeks in the absence of light. Biodegradation in dust is not likely to be enantioselective. This is in contrast to UK and Spanish human milk where enantioselective degradation of α-HBCD has been observed (Harrad and Abdallah, 2010; Eljarrat et al, 2009). Similar selective degradation of α-HBCD has been observed in fish from UK lakes (Harrad et al, 2009b). While indications exist of some slight enantioselective metabolism of γ–HBCD; β-HBCD does not appear to be enantioselectively processed by biota including humans.
In indoor dust, degradation occurs via sequential dehydrobromination (loss of HBr) to yield pentabromocyclododecenes (PBCDs) and tetrabromocyclododecadienes (TBCDs) (Abdallah et al, 2008b; Harrad et al, 2009a). PBCDs and TBCDs have both been reported in commercial HBCD formulations (Barontini et al, 2001). Preliminary data reveals the presence of both PBCDs and TBCDs in UK surface soils, UK lake sediments and UK human milk samples (Harrad et al, 2010b, Harrad et al, 2009b, Harrad and Abdallah, 2010). The presence of these degradation products in sediments suggests degradation via the dehydrobromination pathway, in contrast to an earlier laboratory study which reported HBCD degradation in sediment to occur via sequential debromination (loss of Br₂) to tetrabromocyclododecene (TBCDe), dibromocyclododecadiene, and cyclodecatriene (Davis et al, 2006).

The toxicological significance of PBCDs and TBCDs (and of the corresponding lower brominated products) is as yet unknown, although there is evidence of higher binding affinities of PBCDs to the human transthyretin receptor than the parent HBCDs. Also unknown at the current time, is the persistence of these degradation products.

The information in Table 1 and the above discussion indicates that there is a relatively low level of agreement between studies of the persistence of HBCD in the environment. The half-life is influenced by conditions such as temperature, the presence of oxygen, micro-organisms and sunlight, and the concentration of HBCD. The study results are also influenced by the specific isomer of HBCD which is under consideration. This means that the conclusions in relation to persistence must take account of the uncertainty in experimental results, and also consider the range of conditions likely to be encountered in the environment.

It is concluded that experimental half-lives in soil and sediment are likely to be close to or below 6 months (180 days). Field evidence indicates that half-lives in sediment are likely to be longer than those indicated by experimental evidence. Further information on these aspects would be useful to define whether HBCD is persistent in soil and sediment, particularly as HBCD will be almost exclusively present in organic media. However, further refinement of these values is unlikely to affect the findings of the present study.

It is concluded that the degradation half lives in water are on balance likely to exceed 2 months. α-HBCD is more prevalent in biota, and is also identified as having longer half-lives than the other stereoisomers. It is concluded that HBCD is on balance likely to meet the criterion of a 2 month half-life in water for classification as a POP. The aquatic compartment is likely to be less important for a relatively insoluble substance such as HBCD, which may reduce the weight placed on this conclusion in an evaluation against the Stockholm Convention criteria. Further work to identify the factors influencing degradation half-life would be helpful. The further development and improvement of test methodologies for low solubility substances would also be helpful in this regard.

2.3.2 Other considerations

Annex D of the Stockholm Convention confirms that persistence can also be considered in terms other than compliance with the laboratory test criteria discussed in Section 2.3.1 above. Substances which are judged to have "sufficient persistence" to justify consideration within the Convention can be included, whether or not they comply with the numerical persistence criteria.

In the case of HBCD, evidence indicates that HBCD can persist in aerobic sediments for long periods, suggesting that the laboratory measurements of half-life in aerobic sediment may under-estimate the environmental half-life.

Furthermore, Arnot et al. (2009) and TemaNord (2008) report measurements of HBCD levels in biota in remote locations such as the Arctic and Pacific Ocean. The levels observed in biota in the Arctic tend to be lower than the levels observed in biota closer to the sites of
manufacture, use and disposal of articles containing HBCD. Measured levels of HBCD in “source” and “remote” locations are summarised in Table 2 (adapted from Arnot et al., 2009).

**Table 2: Measured levels of HBCD in source and remote locations**

<table>
<thead>
<tr>
<th>Compartment</th>
<th>Units</th>
<th>Source location</th>
<th>Remote location</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td></td>
<td>Range</td>
<td>Median</td>
</tr>
<tr>
<td>Air</td>
<td>pg m⁻³</td>
<td>0.1 – 600</td>
<td>10</td>
</tr>
<tr>
<td>Sediment</td>
<td>µg·kg⁻¹·dw</td>
<td>&lt;0.1 – 30</td>
<td>0.5</td>
</tr>
<tr>
<td>Benthic invertebrates</td>
<td>µg·kg⁻¹·lw</td>
<td>0.5 – 100</td>
<td>30</td>
</tr>
<tr>
<td>Fish (upper trophic level)</td>
<td>µg·kg⁻¹·lw</td>
<td>10 – 300</td>
<td>30</td>
</tr>
<tr>
<td>Birds</td>
<td>µg·kg⁻¹·lw</td>
<td>25 – 160</td>
<td>65</td>
</tr>
<tr>
<td>Marine mammals</td>
<td>µg·kg⁻¹·lw</td>
<td>30 – 750</td>
<td>150</td>
</tr>
</tbody>
</table>

The bird samples are taken from peregrine falcon, guillemot, European shag, gulls, kittiwake, northern fulmar and Atlantic puffins. The exposure and metabolism of the sample populations are likely to affect the measured levels of HBCD. Nevertheless, measured levels in birds in remote and “source” locations cover similar ranges. This is likely to reflect the wide feeding and migratory range of birds. Median air and sediment concentrations in remote and “source” locations are also comparable, although higher in “source” locations.

Arnot et al. (2009) suggest that because monitoring systems are able to detect HBCD at very low levels in the environment, the presence of detectable levels of HBCD in remote locations should not in principle be taken to mean that HBCD exhibits persistent characteristics, although it does indicate the potential for long-range transport. However, the air and sediment concentrations shown in Table 2 are comparable, suggesting that the presence of HBCD in remote locations is not simply due to the use of extremely sensitive measurement techniques.

Furthermore, Arnot et al. (2009) report that measured levels of HBCD in polar bears were found to be higher in the Eastern Arctic (closer to more intense European sources of HBCD) than in the Western Arctic, where HBCD is less intensively used. This is consistent with the expected pattern based on sources of HBCD, if HBCD persists for a sufficient time in the environment to reach remote Arctic locations. This is consistent with persistent behaviour of HBCD in the environment.

This indicates that HBCD persists in the environment long enough to be transported over considerable distances, and that the substance therefore can be considered sufficiently persistent to be of concern under the terms of the Stockholm Protocol.

### 2.3.3 Conclusion with regard to persistence

On the basis of the observed levels of HBCD in remote locations, it is concluded that HBCD should be considered as persistent for the purposes of the Stockholm Convention. This conclusion has some support from laboratory measurements of the half-lives of HBCD isomers in sediment and water, together with sediment core data indicating a longer half-life than suggested by laboratory data. These measurements indicate that HBCD should on balance be considered as persistent, but the measured half-life is subject to considerable uncertainty.
2.4 Long-range transport

The Stockholm Convention requires consideration of measured levels of HBCD in remote locations; monitoring data showing the potential for transfer to a receiving environment; and chemical properties or model results which indicate the potential for transfer to a receiving environment. This is linked to the discussion of persistence, but focuses more on the evidence for transfer mechanisms.

2.4.1 Measurements in remote locations

Measurements in remote locations are summarised in Table 2 above. ECHA (2008) report that HBCD has been found in abiotic and biotic samples taken from remote areas such as Sweden, Finland, Norway and the north Pacific Ocean.

While concentrations of HBCD are generally lower in remote locations than in locations close to potential sources, HBCD is nevertheless present at detectable levels in remote environmental media. As there are no known natural sources of HBCD, the measurements in air and sediment provide evidence that there is potential for long-range transport of HBCD to remote locations.

2.4.2 Monitoring data showing potential for transfer

Measurements of HBCD in birds are summarised in Table 2 above. In view of the consistent results across species, these measurements indicate that there is significant potential for transfer of HBCD to remote environments following uptake by birds in areas close to sources of HBCD.

Ueno et al. (2006) measured levels of HBCD isomers in skipjack tuna in the Pacific Ocean. This study found that levels of HBCD were reduced to half the levels in fish close to the source of HBCD at a distance of 8500 km (α-HBCD) and 1600 km (γ-HBCD). These measurements also indicate significant potential for transfer of all isomers of HBCD, but α-HBCD in particular, to remote environments such as the north Pacific Ocean. The longer transportation distances for α-HBCD may be partly due to conversion of other isomers to α-HBCD within biota.

2.4.3 Chemical properties and model results

Modelling studies indicate an atmospheric half-life of 0.4 to 5 days (UNEP, 2010), depending mainly on the ambient levels of OH radicals. Annex D to the Stockholm Convention sets a benchmark of 2 days for the atmospheric half-life of a chemical to be considered subject to long-range transport. Under some circumstances (e.g. relatively clean air with low levels of OH radicals), the atmospheric half-life of HBCD would be sufficient for it to meet this criterion for classification as a POP. The estimated atmospheric half-life indicates that it would be possible for significant airborne transfer of HBCD to remote environments to take place. For example, with an average wind speed of 5 m/s, and assuming a half-life of 2 days, 50% of HBCD in the atmosphere would be present at a distance of 860 km from a source of HBCD to the atmosphere.

Arnot et al (2009) carried out a modelling study which included estimates of characteristic transport distances. Using median or worst-case estimated model parameters gave transport distances of approximately 1,000 km, comparable to other POPs. The estimated transportation pathways are strongly affected by the mode of release to the environment.

Arnot et al (2009) also reported a global scale modelling study. This study forecast minimal levels of HBCD at remote Arctic and Canadian locations where measurement data were available. The forecast levels were lower than measured concentrations at these locations. These discrepancies are likely to be due to the coarse resolution of the model, with a grid size of several thousand square kilometres. Nevertheless, the model predicted non-zero
levels of HBCD in air and sediment across the remote Arctic region, consistent with long-range transport of HBCD to these areas.

The OECD screening tool (OECD, 2008) was applied using the parameters summarised by Hirai et al. (2010). This tool suggested that estimated characteristic transportation distances were approximately 1450 km (α- and β-HBCD), and 2400 km (γ-HBCD).

2.4.4 Conclusion with regard to long-range transport

On the basis of measurements in remote locations, measurements in transfer media, and modelling studies, it is concluded that HBCD should be considered subject to long-range transport for the purposes of the Stockholm Convention. α-HBCD shows the greatest potential for long-range transport.

2.5 Bio-accumulation

The Stockholm Convention requires consideration of bio-concentration factors in aquatic species; the log $K_{ow}$ value; other reasons for concern such as toxicity or evidence for bio-accumulation in other species; and monitoring data in biota.

Studies of bio-accumulation are complicated by conversion between HBCD isomers in biota. This complicates the interpretation of laboratory studies of HBCD bio-accumulation.

2.5.1 Laboratory measurements

The key reference documents are consistent in reporting a bio-concentration factor (BCF) in freshwater fish of 18,100 L kg$^{-1}$ taken from Veith et al. (1979). Drottar and Krueger (2000) reported a BCF for technical γ-HBCD in freshwater fish of up to 13,100 L kg$^{-1}$. These studies both used a co-solvent technique, which introduces some additional uncertainty into the study results. Nevertheless, these values are above the criterion of 5,000 specified in Annex D to the Stockholm Protocol.

The European Chemicals Agency (2008) reviewed the available data and found that HBCD isomers have log $K_{ow}$ values of 5.07 – 5.62. This is indicative of a chemical likely to accumulate in animal fats. The log $K_{ow}$ value is above the criterion of 5 specified in Annex D to the Stockholm Protocol.

2.5.2 Other reasons for concern

Studies of bio-accumulation in mammals are reported in UNEP (2010). This work found evidence for accumulation of HBCD in fatty tissue following repeated exposure to HBCD. The toxicity and eco-toxicity of HBCD are discussed in Section 2.6 below.

2.5.3 Monitoring data in biota

Studies of bio-accumulation in species other than freshwater fish are reported in Arnot et al. (2009) and UNEP (2010). This work identifies bio-accumulation factors of greater than 1 in a range of species and ecosystems. For example, levels of HBCD were found to magnify up the aquatic food chain in the Arctic as far as the ringed seal (Sørmo et al., 2006). Evidence for bio-magnification from ringed seal to polar bear was equivocal, possibly reflecting enhanced metabolism of HBCD in polar bears. Further studies have considered bio-accumulation of HBCD in the North Sea, the North Atlantic, Canadian lakes, and freshwater ecosystems in China.

De Wit et al. (2010) found that γ-HBCD is the predominant isomer in Arctic air, whereas α-HBCD predominates in biota, and similar concentrations of α-, β- and γ-HBCD are found in sediments. De Wit et al. concluded that α-HBCD biomagnifies in the food chain, whereas γ-HBCD undergoes trophic dilution.
Evidence from an on-going study in UK lakes also indicates that HBCD bioaccumulation in freshwater fish relative to the water concentration is greater for $\alpha$-HBCD than for either $\beta$- or $\gamma$-HBCD (Harrad et al, 2009b and 2010), consistent with the findings of de Wit et al. (2010). The average lipid weight-based bio-accumulation factor for total HBCDs in the muscle tissue of all 30 fish samples analysed was 2,100,000 L kg$^{-1}$. The average fresh weight bio-accumulation factors for $\alpha$-, $\beta$- and $\gamma$-HBCD were found to be 295,000, 65,000, and 40,500 L kg$^{-1}$. These values are well above the bioaccumulation criterion of 5,000 L kg$^{-1}$ for listing under the Stockholm Convention.

It is concluded that there is ample evidence of bio-accumulation from monitoring of HBCD in biota.

2.5.4 Conclusion with regard to bio-accumulation

On the basis of the observed BCF and log $K_{ow}$ values, evidence for bio-accumulation in mammals, and measurements taken in biota, it is concluded that HBCD should be considered as bio-accumulative for the purposes of the Stockholm Convention. The available field evidence indicates that $\alpha$-HBCD exhibits greater bio-accumulation potential than the $\beta$- and $\gamma$- stereoisomers, although laboratory evidence suggests that $\gamma$-HBCD would be expected to have greater bio-accumulation potential. Nevertheless, laboratory and field data indicate that all three isomers can be considered bio-accumulative.

2.6 Adverse effects

The Stockholm Convention requires consideration of evidence for adverse effects to human health or to the environment; and consideration of toxic or ecotoxic properties that indicate the potential for damage to human health or to the environment.

HBCD has been identified as “toxic” under the terms of the Long-Range Transboundary Air Pollution Convention Protocol on Persistent Organic Pollutants. The toxicity criterion for listing under this Protocol is specified in UNECE Executive Body (EB) Decision 1998/2:

“Toxicity: potential to adversely affect human health and/or the environment”

This is similar to the criterion used in the Stockholm Convention.

In the present study, a risk-based approach was adopted to the question of toxicity, rather than a simple “hazard” based assessment. A chemical was considered as “toxic” if it is toxic at levels likely to be encountered in the environment, either at present or in the future. Hence, a chemical is described as toxic if there is a risk of toxic effects occurring. The alternative approach would be to adopt an absolute definition of toxicity based on the observation of toxic effects at any exposure. This is less discriminating, because essentially any substance is likely to demonstrate some toxic effects if exposure and uptake are at sufficiently high levels. Adopting an absolute definition of toxicity would be of limited practical value, and would result in substances being identified as “toxic” even if there is negligible risk of toxic effects occurring.

2.6.1 Evidence for adverse health or environmental effects

There does not appear to be direct evidence for adverse health effects due to exposure to HBCD. Hence, any discussion of the potential adverse health effects of HBCD must be focused on an evaluation of human exposure against benchmarks for adverse effects on health.

Similarly, there does not appear to be direct evidence for adverse environmental effects due to exposure of the natural environment to HBCD. Hence, any discussion of the potential adverse environmental effects of HBCD must be focused on an evaluation of exposure of components of natural ecosystems against benchmarks designed to protect natural features from harm due to HBCD.
2.6.2 Evidence for adverse health properties

A study of the health effects of HBCD was carried out by IOM on behalf of AEA Technology. The full study is provided in Appendix 4.

The study found that HBCD is extensively metabolised but has some potential to accumulate in fatty tissue and there is some evidence that it crosses the blood brain barrier. HBCD has a relatively low toxicity in short and long term experiments in animals. The main effects that are of potential concern are developmental neurotoxicity and effects on thyroid function. However, the lowest levels of exposure associated with adverse effects in animals are much greater than current levels of exposure in humans. Similarly the Derived no effects levels (DNELs) are much higher than current exposure levels – see Table 3.

Table 3: Estimated exposures and toxicological benchmarks for HBCD

<table>
<thead>
<tr>
<th>Effect</th>
<th>Intake (mg/kg-day)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Estimated reasonable worst case exposure to HBCD in UK adults (excluding workplace exposure)</td>
<td>0.000066</td>
</tr>
<tr>
<td>Estimated reasonable worst case exposure to HBCD in UK toddlers</td>
<td>0.000375</td>
</tr>
<tr>
<td>Proposed DNEL for liver effects in workers</td>
<td>1.1</td>
</tr>
<tr>
<td>Proposed DNEL for liver effects in the general population</td>
<td>0.5</td>
</tr>
<tr>
<td>Proposed DNEL for fertility in workers</td>
<td>0.2</td>
</tr>
<tr>
<td>Proposed DNEL for fertility in the general population</td>
<td>0.1</td>
</tr>
<tr>
<td>Benchmark Dose Limit for repeated dose toxicity</td>
<td>22.9</td>
</tr>
<tr>
<td>Benchmark Dose Limit in 2 generation study</td>
<td>10.2</td>
</tr>
<tr>
<td>Benchmark Dose Limit for developmental toxicity (effects of uncertain significance)</td>
<td></td>
</tr>
<tr>
<td>Effects on trabacular bone in females</td>
<td>0.056</td>
</tr>
<tr>
<td>Immune effects in males</td>
<td>0.46</td>
</tr>
<tr>
<td>Decreased weight testes</td>
<td>1.5</td>
</tr>
<tr>
<td>Effects on liver function</td>
<td>1.3</td>
</tr>
</tbody>
</table>

The major sources of adult human exposure to HBCD are via the diet and incidental ingestion of house dust. This has been found to be a more significant pathway than inhalation of HBCD, or ingestion of HBCD via the food chain (Abdallah et al, 2008a and b). Nursing infants are also exposed via consumption of human milk at levels that exceed those of most adults (Harrad, 2010).

There are limited data that suggest that there are gender-related differences in the metabolism of HBCD and its toxic effects but these are unlikely to be of importance in view of the level of human exposure to HBCD. There are also indications that substantial differences in the metabolism of different HBCD isomers are likely to give rise to some differences in toxic effect. However, there are insufficient data from animal experiments to determine the relative toxicity of different isomers. Limited data suggest that the cytotoxicity of γ-HBCD is greater than that of the other isomers, but the significance of this finding is uncertain in view of the more extensive metabolism of this isomer which would lead to lower levels of systematic exposure. Despite this uncertainty, it seems unlikely that the dose limits
and No Observed Adverse Effect levels reported in assays that have employed technical grade or γ-HBCD are substantially higher than would be found in similar assays with α- or β-HBCD, although some difference in effects is possible.

Workers handling products containing HBCD are likely to have slightly higher levels of exposure to HBCD than other members of the population but exposure levels would remain well below those associated with adverse effects in animals. Higher levels of exposure are possible where powdered HBCD is handled in the workplace, but exposures can be readily controlled through the design and implementation of appropriate workplace control measures. Workplace exposure is most unlikely to exceed the Derived No-Effect Level for fertility. Workplace exposures can be readily controlled to well below 1 mg/m³ as an 8 hour mean, which would result in an average daily intake of well below 0.4 mg/kg body weight. Furthermore, very few people in the UK are likely to be directly exposed to HBCD powder at work.

There is no evidence to suggest that reducing exposure to HBCD by phasing it out of consumer products and construction materials would have any significant direct benefit to human health for the general population. It is concluded that the available toxicity data does not indicate the potential for damage to human health.

**2.6.3 Evidence for adverse environmental properties**

UNEP (2010) reports a range of adverse environmental properties, including:

- Toxicity to earthworms was assessed by Aufderheide et al. (2003) who derived No Observed Effect Concentrations (NOEC) of 4190 and 59 mg HBCD per kg dry soil for survival and reproduction respectively.

- Toxicity to plants was found to be relatively low, with NOEC of over 5000 mg HBCD per kg dry soil for a range of plant species.

- Egg and chick survival was decreased in quails exposed to 2.1 mg/kg/day HBCD with a NOAEL of 0.7 mg/kg/day (Ministry of the Environment, Japan, 2009).

- Minor differences in courtship behaviour, earlier egg-laying, and a slower growth rate were observed in American kestrels exposed daily to technical HBCD dissolved in safflower oil at a concentration of 0.8 mg/kg (Marteinson et al 2009).

Research carried out by Saegusa et al (2009) and Eriksson et al (2006) was criticised for failing to adjust for maternal ingestion rates, litter effects and development (CEFIC, 2010).

More recently, differences in reproductive physiology and behaviour were observed in American kestrels exposed daily to technical HBCD dissolved in safflower oil at a concentration of 0.054 mg/kg (Marteinson et al 2010). This is substantially lower than the criterion of 30 mg/kg used to assess the avian toxicity of biocides and pesticides prior to the implementation of REACH (European Chemicals Bureau, 2003). However, the potential for exposure to a chemical additive such as HBCD is substantially lower than the potential for avian exposure to a biocide or pesticide, and the ECB guideline is not directly applicable to consideration of REACH. It remains unclear whether avian predators such as predators could be exposed to HBCD above the NOAEL in the environment.

Studies in aquatic media are problematic because of the low solubility of HBCD. This requires the use of solvents to enable dissolution of HBCD. This introduces uncertainty into these studies, in terms of their representativeness for environmental exposures. There is a further difficulty in determining the solubility of HBCD in the natural environment. For example, the presence of organic matter in water bodies could increase the solubility of HBCD. There are no established methods for measuring the solubility of highly insoluble substances such as HBCD under environmental or laboratory conditions. The available evidence suggests that the water solubility of HBCD is approximately as follows (ECHA, 2008):
• α-HBCD: 34-49 µg/L
• β-HBCD: 10 – 15 µg/L
• γ-HBCD: 2.1 – 3.4 µg/L

These values are subject to some uncertainty and likely to be variable in the environment.

No adverse effects have been observed at a concentration of 2 µg/L. However, adverse effects have been observed at slightly higher concentrations. Effects on Daphnia magna fly have been reported at levels of 5.6 µg/L technical HBCD, with no effect at concentrations of 3.1 µg/L technical HBCD (Graves and Swigert, 1997). No effect was observed on rainbow trout at levels up to 3.7 µg/L (Drottar et al., 2001). Palace et al. (2008) reported altered thyroid status in juvenile rainbow trout at levels in food of 29 ng/g α-HBCD, 12 ng/g β-HBCD and 22 ng/g γ-HBCD.

A single study of measured levels of HBCD in freshwater invertebrates, which form much of the diet of rainbow trout, was reported in ECHA (2008). The findings of this limited study are reproduced in Figure 2 below.

**Figure 2: Measured levels of HBCD in freshwater invertebrates (from ECHA, 2008)**

The data in Figure 2 indicate that measured levels of HBCD in a single study were generally below 10 – 30 ng/g, the levels tested by Palace et al (2008).

The UNEP (2010) report considered potential exposures of predators to contaminated food stocks. A predicted no-effect concentration (PNEC) of 5 mg technical HBCD per kg food was derived from the study of Ema et al. (2008). This level has been criticised as being over-protective, and a value of 15 mg per kg suggested instead (CEFIC, 2010). Levels of 5 mg HBCD per kg have been observed exceptionally in some locations such as eels and brown trout in polluted river estuaries and in harbour porpoises. In view of the infrequency of measurement of levels above 5 mg/kg in fish, it would be unlikely for a predator to consume prey with an HBCD level above 5 mg/kg for an extended period. Levels in prey in remote locations such as Arctic ringed seal are well below the levels of potential concern for predators such as the polar bear.
The very low and uncertain solubility of HBCD together with the marginal nature of toxicological findings at low concentrations mean that it is not possible to state definitively whether HBCD has adverse eco-toxicological effects at the levels that could be experienced in the environment.

Overall, it appears possible that levels of HBCD may be close to or marginally above a level which could potentially give rise to adverse developmental effects in some species such as American kestrel and rainbow trout. \( \gamma \)-HBCD is relatively insoluble, and has not been found to have toxic properties at levels which could occur in aquatic media. At present, there is insufficient data to judge whether \( \alpha \)-HBCD or \( \beta \)-HBCD could potentially exhibit toxic effects at higher exposures. The evidence indicates that there is no widespread environmental harm arising from current levels of HBCD. If any effects are occurring, they are likely to be localised to sources of environmental pollution, and occasional in nature. It would be beneficial to obtain more detailed information on the relative toxicity of HBCD stereoisomers at the levels which could potentially be encountered in the environment.

It is concluded that the available eco-toxicity data indicates the potential for environmental damage at relatively high exposures compared to those currently encountered. At the levels encountered in the environment, the available data indicate a marginal possibility for damage to the environment.

There is currently a proposal before the European Chemicals Agency for classification of HBCD as a reprotoxicant. HBCD has recently been evaluated under the terms of the CLRTAP. The toxicity criteria for identifying a chemical as a POP are similar under the terms of the CLRTAP to the criteria under the Stockholm Convention. In both cases, the criteria are set in qualitative terms. The CLRTAP considers chemicals to be toxic if they have the “potential to adversely affect human health and/or the environment.” The UNECE task force on POPs has recommended that HBCD fulfills the CLRTAP criterion for toxicity, and should be considered as a POP. The evaluation set out in the Lead Reviewer’s Summary (UNECE, 2009) refers to evidence for toxic effects as discussed above. However, it does not consider the risks in terms of potential environmental exposure.

There is no doubt that HBCD exhibits a range of toxic effects for the natural environment. If an absolute definition of toxicity were to be adopted, then it would be concluded that HBCD has the potential for toxic effects, in common with many other substances. In the present study, a practical definition of toxicity was adopted, such that a chemical was considered as “toxic” if it is toxic at levels likely to be encountered in the environment, either at present or in the future. Adopting an absolute definition of toxicity based on hazard alone was considered to be of less practical value, because essentially any substance is likely to demonstrate a toxic effect if exposure and uptake are at sufficiently high levels.

On this basis, it is concluded that the available data indicates that there is no more than a marginal possibility for harm to the environment. Adopting a precautionary approach, HBCD could be considered as exhibiting the potential for harm to the environment. In order to verify this conclusion, the focus should be on identifying the potential exposures of sensitive components of natural ecosystems – in particular, avian predators. More detailed information on the relative toxicity of HBCD stereoisomers at the levels which could potentially be encountered in the environment would also be valuable.

2.6.4 Conclusion with regard to adverse effects

On the basis that human exposure to HBCD in the UK is significantly below levels of potential concern, it is concluded that HBCD should not be considered as having toxic properties for humans for the purposes of the Stockholm Convention.

Environmental levels of HBCD are close to the levels of potential concern for some species. The uncertainty in environmental exposures and the levels giving rise to potentially significant adverse effects means that it is not possible to state definitively whether HBCD exhibits toxic environmental properties, for the purposes of the Stockholm Convention. It is
recommended that more detailed information on the exposure to HBCD of sensitive species such as avian and freshwater predators would be helpful in reducing uncertainty in relation to the potential toxicity of HBCD. More detailed evaluation of the relative toxicity of HBCD stereoisomers at the levels which could potentially be encountered in the environment should also be obtained.

2.7 Overall conclusion

The evaluation set out above is summarised as follows.

- Persistence: On balance, meets the criteria of the Stockholm Convention
- Bio-accumulation: Meets the criteria of the Stockholm Convention
- Long-range transport: Meets the criteria of the Stockholm Convention
- Adverse effects: Does not meet the criteria of the Stockholm Convention with regard to the potential for damage to human health. Uncertain whether HBCD meets the criteria of the Stockholm Convention with regard to the potential for damage to the environment

It is concluded that HBCD does not demonstrably meet the criteria set in the Convention. However, if a precautionary approach is adopted with regard to the potential for environmental harm, this would suggest that HBCD could provisionally be considered a Persistent Organic Pollutant under the terms of the Stockholm Convention, pending further research to confirm the range of exposure of sensitive species and the potential adverse effects of HBCD stereoisomers.
3 Evaluation of sources, uses and controls of HBCD in the UK

3.1 Introduction
This chapter provides an assessment of the uses of HBCD in the UK. This is based on a range of information from published data sets, through discussion with industry bodies and experts, and estimates developed for the purposes of this project. This leads to the preparation of an estimated release inventory for HBCD. The release inventory identifies the key sources of emissions of HBCD to air, water, land/landfill, and to recycling processes. Following the estimation of emissions, the currently available measures which are or could be used to control releases of HBCD to the environment are discussed.

3.2 Uses of HBCD
The main uses of HBCD are as an additive flame retardant used to treat expanded polystyrene (EPS) or extruded polystyrene (XPS) insulation material used in cavity walls of buildings. The purpose of such material is to improve the energy efficiency of the building. Around 90% of the market share of HBCD is used in polystyrene applications.

XPS and EPS are manufactured as sheets which can be inserted into building wall cavities, and are widely used in new buildings in this way. Advice from the insulation industry is that the installation of insulation in wall cavities in existing buildings has now largely been completed. In view of this, there is likely to be a shift in use of XPS and EPS in existing buildings. One option is to apply insulation sheets to the inside of a building, and cover the wall with a plaster finish. Alternatively, insulation sheets can be fixed to the outside of a building, and covered with a weatherproof rendering. These methods can be used to improve the thermal insulation of buildings which do not have cavity walls, and may become more widespread in the future.

HBCD is also used as a backing for textiles, for example in soft furnishings such as domestic sofa covers, transportation seating, and bed mattress ticking. While the thermal instability of HBCD has previously limited its use in high impact polystyrene (HIPS) deployed in electronic goods, recent development of a more thermally stable HBCD formulation means that the UL94-V2 flame retardancy standard can be met with the use of HBCD. This renders the use of HBCD in audiovisual equipment feasible. HBCD can now be used in televisions, for which only the lower UL94-HB standard is required in Europe (Weil and Levchik, 2007).

This is compatible with recent evidence that indicates the use of HBCD in a TV set in the UK (Harrad et al, 2009a). This is supported by further research presented at the Dioxin 2010 symposium, which shows concentrations of HBCD in air in the vicinity of a UK e-waste recycling plant to be 35 ng total HBCD m$^{-3}$ (average of two samples inside the plant) and 22 ng total HBCD m$^{-3}$ in a single sample taken outdoors but within the plant boundaries (Harrad et al, 2010b). These concentrations exceed by three orders of magnitude those reported on the University of Birmingham campus (Abdallah et al, 2008a; Harrad et al, 2010b).

3.2.1 Sales of HBCD in the UK
HBCD was produced in the UK until 2003. Law et al. (2008) reported information on sales of HBCD in the UK from European suppliers for the period 1997 – 2006. In 2006, the European manufacturing companies (Albermarle, Chemtura and ICL) sold 294 tonnes of HBCD into the UK. Information collated by the European Chemicals Agency indicates that
an increasing proportion of HBCD used in Europe was imported from other suppliers (ECHA, 2008). The estimated proportion of HBCD used in Europe which was imported from outside the EU was 38% in 2003, rising to 48% in 2006. It is likely that a similar proportion of HBCD was imported into the UK over this period. The total sales of HBCD in the UK were estimated from the data in Law et al. (2008), increased to account for the estimated proportion of sales from non-European suppliers. The total estimated sales are set out in Figure 2.

*Figure 3: Estimated total sales of HBCD in the UK 1997 - 2006*

The reduction in sales between 2001 and 2003 might reflect the closure of the UK manufacturing facility in 2001. This could have resulted in a delay in identifying alternative suppliers of HBCD, and/or could have affected the quality, price and delivery of HBCD to UK users. The estimated sales of HBCD in the UK in 2006 were 570 tonnes per year.

The European manufacturing companies represented by EBFRIP estimate that approximately 90% of HBCD sold by them into the UK was used in the manufacture of EPS/XPS boards. Approximately 9% was used for textile manufacture, and a small fraction (approximately 1%) was used for the manufacture of HIPS. This pattern may be different for manufacturers other than the European manufacturers.

For comparison, information on HBCD usage in Japan was reviewed by Watanabe (2010). Sales of HBCD increased steadily from 1986 to 2008, with sales in the year 2000 of approximately 2,000 tonnes. The Gross Domestic Product of the UK is approximately 54% of the GDP of Japan. Other factors being equal, sales of HBCD are likely to be roughly proportional to GDP. This indicates that the expected rate of sales in the UK in 2000 might be estimated from the value for Japan to be approximately 1100 tonnes per year. This estimate is similar to the estimate of 860 tonnes in 2000 shown in Figure 2. Since the year 2000, sales in the UK have generally declined, whereas sales in Japan have generally increased.

The European Chemicals Agency estimated that approximately 9,600 tonnes of HBCD was used in Europe in 2004 (ECHA, 2008). The Gross Domestic Product of the UK is approximately 16% of the GDP of the European Union. This indicates that the expected rate of sales in the UK in 2004 might be approximately 1600 tonnes per year, rather higher than
the figure of 570 tonnes per year shown in Figure 3. This discrepancy may be due to differences between the UK market for HBCD and the market in the rest of Europe.

HBCD can be purchased in small quantities for approximately £30 per kilogram (2010 prices).

3.2.2 HBCD required for building construction

These estimates can be independently checked by consideration of the generation of construction and demolition waste, and the amount of HBCD that would be required to replace material removed from buildings. As the majority of HBCD is used in the manufacture of insulation materials for construction, an estimate of HBCD usage could be derived from the rate of construction waste. As a first estimate, it was assumed that construction materials are used at the same rate that demolition waste is generated.

Figures provided by Defra indicate that approximately 90 million tonnes of construction and demolition waste are produced per year (Defra, 2010). This material includes a relatively small proportion of insulation material, which varies between 0.6% and 8% depending on the type of building (Defra 2008). It was estimated that 25% of building waste is from residential buildings, 12% from each of office, retail, industrial, leisure and public buildings, 6% from each of civil engineering and education buildings, and 3% from healthcare buildings. This indicates that approximately 2.8% of demolition waste is likely to comprise insulation material. Evidence from the construction industry indicates that EPS and XPS accounts for approximately 5% of insulation material in buildings, with the majority accounted for with other materials such as mineral fibre. This is broadly consistent with the findings of research carried out by Environment Agency Wales, which indicated that approximately 1% of building waste was plastic materials (Environment Agency Wales, 2006).

To estimate the amount of HBCD present in materials used to replace insulation removed in demolition waste, the level of HBCD in XPS and EPS was assumed to be approximately 2% and 0.7% by mass, respectively (UNEP, 2010). It was assumed that XPS accounted for 52% of the use of HBCD in the manufacture of insulation materials, and EPS accounted for 48% of the use of HBCD in insulation (ECHA, 2008).

Bringing these estimates together provides a preliminary estimate of HBCD generated from building demolition in the UK of 1300 tonnes per year. This is approximately twice the estimated figure of 570 tonnes per year in Figure 2. Agreement to a factor of 2-3 is reasonable in view of the uncertainties in the estimated value. The uncertainty in the estimated amount of HBCD used in the replacement of building waste materials is subject to greater uncertainty than the estimate based on UK and European sales shown in Figure 2.

3.3 Sources and Pathways of HBCD

HBCD is not produced in the UK. However, as discussed above, it is widely used in goods in the UK. Emissions of HBCD into the environment could occur from the manufacture of goods containing HBCD, the use of these goods, or at the end of life of goods containing HBCD.

3.3.1 Manufacturing goods containing HBCD

Manufacture of EPS and XPS
Industry data (VECAP, 2009) provides emission factors for the manufacture of plastic goods containing HBCD. The emission factors are derived from sites registered under the VECAP/SECURE programme. While not all processing facilities in the UK are covered under this programme, emissions to air and water are likely to be limited by other environmental regulations/controls such as Environmental Permitting. In view of this, the VECAP/SECURE report values were considered to provide a reasonable representation of emissions from UK processes.

The factor for emissions to land is appropriate for users with the opportunity to participate in the industry-sponsored VECAP/SECURE product stewardship programmes. Only about half the HBCD sold in the UK is from suppliers who are involved in these programmes. The emissions to land could therefore be higher than the quoted emission factor for 2008 of 22 grams per tonne HBCD used. A figure of 170 grams per tonne was identified for 2007, but this is likely to be an over-estimate of HBCD emissions because it assumes that the majority of HBCD is placed in uncontrolled landfill, whereas manufacturing waste materials would normally be placed in controlled landfill, where this is appropriate in view of the nature of the materials. An intermediate emission factor was estimated based on the geometric mean of these two values.

- Emissions to air: 6 grams HBCD emitted to air per tonne HBCD used
- Emissions to water: 7 grams HBCD emitted to water per tonne HBCD used
- Emissions to land: 60 grams HBCD emitted to land per tonne HBCD used

The main use of HBCD in the UK is the manufacture of Expanded Polystyrene (EPS) or Extruded Polystyrene (XPS). These materials are used for building insulation.

The manufacture of EPS typically starts with steam heating of polystyrene resin granules with additives such as HBCD in a pre-expander at a temperature of approximately 100°C. During this pre-expansion stage, the raw material is changed from a compact bead form into lower density beads with a cellular structure. The beads are then cooled and matured in aerated silos. The stabilised pre-expanded beads are then transported to a mould and steam heated to bind the beads together in the desired form.

The manufacture of XPS similarly uses solid polystyrene resin granules. The granules are fed into an extruder where they are melted to form a viscous fluid. Additives such as HBCD are mixed with the fluid, and a blowing agent is added. The fluid is forced under increased temperature and pressure through a die, which causes the fluid to foam and form the desired shape. After cooling, the rigid foam can be trimmed as required. XPS has a smoother and more uniform structure than EPS.

Data from EBFRIP suggests that 90% of HBCD sold into the UK is used in the manufacture of EPS and XPS. However, it appears that a higher proportion of HBCD is used in the textile industry than envisaged by EBFRIP, implying that a lower proportion of HBCD is used in the production of EPS and XPS. On this basis, it is estimated that approximately 700 tonnes per year of HBCD is used in the manufacture of EPS and XPS in the UK, split equally between EPS and XPS.

The British Plastics Federation estimates that in 2008, 20,600 tonnes of EPS was sold in the UK, of which 40% contains HBCD. Assuming a HBCD content of 0.7% by mass suggests approximately 60 tonnes of HBCD was used in the manufacture of EPS. This is lower than would be expected on the basis of sales of HBCD, and possibly suggests that the HBCD content of EPS is greater than 0.7% by mass. Alternatively, it may be that some of the HBCD sold in the UK is stored for use in future years. The lifespan of EPS and XPS in buildings is 30-50 years. The BPF indicates that some EPS is recycled, but none of this material contains HBCD.

Using the emissions factors set out above, estimated emissions from the manufacture of EPS and XPS are as follows:
• Emissions to air: 4.2 kilograms HBCD per year
• Emissions to water: 4.9 kilograms HBCD per year
• Emissions to landfill: 42 kilograms HBCD per year

These preliminary estimates could potentially be developed using sector-specific emission factors, if it becomes necessary to consider control strategies in more detail.

**Manufacture of High-Impact Polystyrene**

A description of this process is provided by the European Chemicals Agency (ECHA, 2008). High Impact Polystyrene is produced in batch and continuous processes. The raw material is homogenised and extruded into HIPS pellets. Flame retardants such as HBCD or other BFRs are mixed and homogenised with the HIPS pellets. Alternatively, a mix of HIPS pellets containing a high concentration of HBCD can be prepared and blended with pellets as an intermediate step to give the required blend. The pellets are heated and pressed through a die to give a range of product types.

The HIPS pellets are dried and packed for use in the manufacture of HIPS products. This process using extrusion techniques and injection moulding. Most HIPS products containing flame retardants are used in electrical and electronic appliances. The HBCD content of flame-retarded HIPS is typically 1% - 7% (ECHA, 2008).

It is estimated that approximately 50 tonnes per year of HBCD is used in the manufacture of HIPS in the UK. This estimate is based on the assumption that HIPS is a relatively small component of total use of HBCD. It is consistent with data provided for the manufacture of HIPS containing HBCD in Germany in 1990 provided by ECHA (2008). Using the emissions factors set out above, estimated emissions from the manufacture of HIPS are as follows:

• Emissions to air: 0.30 kilograms HBCD per year
• Emissions to water: 0.35 kilograms HBCD per year
• Emissions to landfill: 3.0 kilograms HBCD per year

These preliminary estimates could potentially be developed using sector-specific emission factors, if it becomes necessary to consider control strategies in more detail.

**Manufacture of textiles containing HBCD**

A description of this process is provided by the European Chemicals Agency (ECHA, 2008). The first step in this process is the preparation of water-based dispersions containing flame retardants and other additives. The dispersion typically contains water, a polymer such as synthetic latex, acrylates or PVC, thickener and dispersion agent. The dispersion may contain a blend of flame retardants. The concentration of HBCD in the dispersion may be up to 48%.

The dispersion is applied as a back-coating to a textile. The material can be applied as a paste or as a foam via a rotating screen. The foam cells break up, resulting in a thin coating film on the fabric. The coating is dried and fixed in an oven at a temperature of 140 to 180°C. HBCD is usually applied with antimony trioxide as a back-coating. Textiles with flame retardant back-coating are used for residential and commercial furniture, upholstery seatings in transportation, curtains and wall coverings, mattresses, interior textiles, and car interior textiles.

Information from the Flame Retardant Textiles Network ("Fretwork", 2010) indicates that 250 tonnes per year of HBCD is used in the manufacture of flame-retardant textiles in the UK. This differs substantially from the estimate made by the European manufacturers of HBCD, which indicated that only 26 tonnes per year are sold by these manufacturers to the UK textile industry. It is concluded that a significant proportion of HBCD used by the UK textile industry is sourced from non-European manufacturers.
The European Commission Risk Assessment Report (European Commission 2008) provides a worst-case estimate of 724 kg HBCD per year released to surface water, and 2.6 kg HBCD per year released to air on a regional basis. Releases to landfill were estimated using the VECAP dataset for the plastics industry as set out above, in the absence of other data.

Estimated emissions from the manufacture of flame-retardant textiles are as follows:

- Emissions to air: 2.6 kilograms HBCD per year
- Emissions to water: 724 kilograms HBCD per year
- Emissions to landfill: 15 kilograms HBCD per year

These preliminary estimates could potentially be developed using sector-specific emission factors, if it becomes necessary to consider control strategies in more detail.

### 3.3.2 Use of goods containing HBCD

#### Use of EPS and XPS

As described above, HBCD is used in the manufacture of flame resistant insulation materials for use in the construction industry. Once installed in a building, losses of HBCD from these materials are likely to be minimal. The key loss process is therefore likely to be during the process of construction.

Evidence provided to the House of Lords indicates that 13% of materials delivered to construction sites are landfilled (House of Lords, 2008). This provides a reasonable basis for estimating the proportion of EPS and XPS insulation material which is wasted during the course of construction – e.g. as offcuts, or as a result of errors or material damage. This material is likely to be disposed to landfill, with no particular controls in relation to HBCD.

A small proportion of material will be released to air as dust during cutting and fitting of EPS and XPS insulation material. No figures could be obtained for the proportion of material released in this way. A preliminary estimate was made that 1% of material could be released to air during cutting and installation of EPS and XPS boards in the construction industry.

This gives the following estimated emissions of HBCD during the use of EPS and XPS insulation materials in the construction of buildings:

- Emissions to air: 7.0 tonnes HBCD per year
- Emissions to water: Negligible HBCD per year
- Emissions to landfill: 91 tonnes HBCD per year

#### Use of High-Impact Polystyrene

Recent research has begun to provide evidence that dusts from high-impact polystyrene can provide a potential human exposure route (Harrad et al, 2009a). Dusts containing elevated levels of HBCD have been identified in the air at a UK electronic waste recycling facility (Harrad et al, 2010b), and in offices in the UK (Harrad et al., ref). The most likely source of HBCD is thermal degradation or abrasion of products containing HBCD. Either HIPS or fabrics could potentially be the source of HBCD identified in indoor environments, but the presence of levels of HBCD three orders of magnitude above ambient levels at the e-waste recycling facility indicates that emissions of HBCD from high-impact polystyrene are at least a contributory source.

HIPS is designed to be resistant to abrasion. Consequently, only a small fraction of HIPS will be released as dust during the use of electrical and electronic goods. However, this is potentially of concern, because any HBCD released in this way is likely to be readily available to individuals (including children) present in the same building as the source of the dust. No estimates for the proportion of HIPS released as dust by abrasion could be identified. For the purposes of this study, it was estimated that 0.1% of HIPS could
potentially be released as dust during the product lifetime. This gave the following estimate for emissions to indoor air of HBCD:

- Emissions to air: 50 kilograms HBCD per year
- Emissions to water: Negligible HBCD per year
- Emissions to landfill: Negligible HBCD per year

**Use of textiles containing HBCD**

As discussed above, high-impact polystyrene or fabrics containing HBCD could potentially comprise a source of HBCD in indoor environments. Because fabrics are less resistant to abrasion, a higher proportion of fabrics may be degraded into dust, with resultant release of HBCD into the indoor atmosphere, or into private or public vehicles. This material may be removed from the home environment by cleaning with fluids or vacuum cleaner.

HBCD may also be released from fabrics by volatilisation, or by contact with fluids such as sweat or saliva. HBCD could also potentially be released to water as a result of washing curtains or furniture/upholstery impregnated with HBCD. Ghanem (2009) found that 1% - 4% of the HBCD in fabrics in contact with fluids could be released from the fabric. It was also found that HBCD was thermally removed from fabrics with a half-life of approximately 5 years (incorrectly reported as a factor of 10 shorter half-life in Ghanem (2009)). The half-life in the field is likely to be typically longer than this. In contrast with dust particles, vapour phase HBCD could not be removed by cleaning.

In view of these considerations, it was estimated for the purposes of this study that approximately 1% of the HBCD in fabrics is likely to be released as dust. Approximately 5% of the HBCD in fabrics may be released in washing fluids. Approximately 50% of the HBCD in fabrics may be thermally removed over the lifetime of goods containing HBCD-impregnated fabrics.

This gave the following estimate for emissions of HBCD from fabrics:

- Emissions to air: 128 tonnes HBCD per year
- Emissions to water: 12.5 tonnes HBCD per year
- Emissions to landfill: Negligible HBCD per year

**3.3.3 End of life of goods containing HBCD**

*Disposal/recycling of demolition waste*

As discussed above, almost all the waste polystyrene insulation material HBCD produced from building demolition is currently landfilled (NFDC, 2010). Although a scheme for recycling of EPS from cold stores has been introduced by the insulation manufacture industry, this has not been widely taken up by the demolition industry.

It is likely that a small proportion of EPS and XPS with any associated HBCD content is released to the atmosphere as dust during the demolition process. In the absence of specific data, the proportion of the polystyrene insulation material released as dust was estimated to be 2%. The remaining HBCD will be sent to landfill as an inert waste. Because no special precautions will be applied to the waste during landfill, this was treated as equivalent to an emission to land, following the approach adopted by VECAP (2009).

At present, the majority of buildings being demolished or refurbished are likely to have been constructed prior to the widespread use of HBCD in the construction industry. Hence, releases from building demolition are likely to be at a relatively low level at present, with forecast emissions likely to rise slowly over a period of decades. Ultimately, emissions of HBCD from the disposal of EPS and XPS insulation materials used in building construction are estimated to be as follows:

- Emissions to air: 12 tonnes HBCD per year
Cost Benefits study of Hexabromocyclododecane

- Emissions to water: Negligible HBCD per year
- Emissions to landfill: 590 tonnes HBCD per year

The rate of removal of materials containing HBCD was estimated by assuming that HBCD was used as an additive in building insulation from approximately 1975 (based on information in ECHA, 2008). It was estimated that building insulation materials remain in place for 50 years on average, with normal distribution of lifetimes characterised by a standard deviation of 20 years. This indicated that emissions were 23% of the ultimate rate in 2010, rising to 60% in 2030.

Disposal/recycling of waste electrical and electronic equipment

The WEEE directive was adopted in 2003. It sets targets for collection of 70%-75% of goods likely to contain high impact polystyrene. The target for recycling these goods is 50% - 65% for different types of equipment.

No figures were available for the proportion of WEEE which is collected and treated under the requirements of the WEEE directive in the UK. For the purposes of this study, it was assumed that 50% of WEEE waste is collected and treated separately. The majority of plastics in separately collected WEEE waste materials are separated and recycled either in the UK or, increasingly, overseas. For the purposes of this inventory, it was assumed that HIPS in the waste stream which is not collected is disposed of via landfill or via incineration, in the same proportions as municipal solid waste, based on statistics published by National Statistics for October 2008 – September 2009: that is, 80% landfill and 20% incineration. It was assumed that incineration removes 99% of the HBCD present in the waste (European Environment Agency, 2009), with the remainder emitted to air.

It was assumed that the HIPS collected under the terms of the WEEE directive is exported with no resultant UK emissions of HBCD, or reprocessed in the UK. Industry advice is that re-use in the UK is relatively low, and consequently, it was assumed that 20% of HIPS is reprocessed in the UK with the balance exported. This gave an estimated quantity of HBCD in material sent for recycling overseas of 24000 kg HBCD per year. It was assumed that reprocessing in the UK results in emissions of HIPS comparable to the emissions reported by VECAP (2009) set out in Section 3.3.1 above.

This gave the following estimated emissions of HBCD from the disposal/recycling of WEEE:
- Emissions to air: 53 kilograms HBCD per year
- Emissions to water: 0.042 kilograms HBCD per year
- Emissions to landfill: 14690 kilograms HBCD per year

Measurements of HBCD in goods containing recycled plastics provide evidence that HBCD is being recycled into new products. For example, Hirai and Sakai (2007) found HBCD at levels of up to 30 mg/kg in video cassettes. Chen et al (2009) found levels of BFRs in plastic toys at levels of up to 0.4% by mass (Chen et al did not investigate levels of HBCD). These levels are lower than the levels of approximately 10% by mass that would be used intentionally to act as a flame retardant. It is concluded that the lower levels of BFRs found in these products are due to the use of recycled plastics containing BFRs in the manufacture of these goods.

Disposal/recycling of textiles containing HBCD

The textile industry was consulted during the course of the project. It was found that there are no specific arrangements for dealing with fabrics impregnated with brominated flame retardants such as HBCD. At the end of their lives, some fabrics are recycled, but the majority of fabrics are disposed of via the normal waste disposal routes. Hence, fabrics are likely to be landfilled, or in some cases, disposed of via incineration.

It was assumed that fabrics impregnated with HBCD are disposed of via landfill or incineration, in the same proportion as the use of these disposal routes for municipal solid waste.
waste in 2009. It was estimated that HBCD present in fabrics disposed of via incineration will be destroyed with a 99% efficiency (European Environment Agency, 2009), with the remainder being emitted to air.

This gave the following estimated emissions of HBCD from the disposal of textiles:

- Emissions to air: 0.29 tonnes HBCD per year
- Emissions to water: Negligible HBCD per year
- Emissions to landfill: 88 tonnes HBCD per year

### 3.3.4 Release inventory summary

The estimated UK annual release inventory of HBCD is set out in Table 4 below.

**Table 4: Estimated UK annual release inventory of HBCD**

<table>
<thead>
<tr>
<th>Source</th>
<th>Air (vapour)</th>
<th>Air (dust)</th>
<th>Water</th>
<th>Uncontrolled landfill</th>
<th>Recycling</th>
<th>Total</th>
</tr>
</thead>
<tbody>
<tr>
<td>Manufacture of goods containing HBCD</td>
<td>7</td>
<td>730</td>
<td>60</td>
<td>-</td>
<td>-</td>
<td>800</td>
</tr>
<tr>
<td>Manufacture of EPS/XPS</td>
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<td>4.9</td>
<td>42</td>
<td>-</td>
<td>-</td>
<td>51</td>
</tr>
<tr>
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<td>0.4</td>
<td>3</td>
<td>-</td>
<td>-</td>
<td>3.7</td>
</tr>
<tr>
<td>Manufacture of Textiles</td>
<td>2.6</td>
<td>720</td>
<td>15</td>
<td>-</td>
<td>-</td>
<td>740</td>
</tr>
<tr>
<td>Use of goods containing HBCD</td>
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<td>13000</td>
<td>91000</td>
<td>-</td>
<td>240000</td>
</tr>
<tr>
<td>Use of EPS/XPS</td>
<td>7000</td>
<td>-</td>
<td>91000</td>
<td>-</td>
<td>-</td>
<td>98000</td>
</tr>
<tr>
<td>Use of HIPS</td>
<td>50</td>
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<td>-</td>
<td>-</td>
<td>-</td>
<td>50</td>
</tr>
<tr>
<td>Use of Textiles</td>
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<td>13000</td>
<td>-</td>
<td>-</td>
<td>140000</td>
</tr>
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<td>2800</td>
<td>0</td>
<td>240000</td>
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<td>240000</td>
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<td>Disposal/recycling of goods containing HBCD (2030)</td>
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<td>490000</td>
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<td>Disposal/recycling of EPS/XPS (2010)</td>
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<td>140000</td>
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<td>140000</td>
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<tr>
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<td>350000</td>
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<td></td>
</tr>
<tr>
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<td>13000</td>
<td>330000</td>
<td>0</td>
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<td>170000</td>
<td>13000</td>
<td>550000</td>
<td>20000</td>
<td>730000</td>
</tr>
</tbody>
</table>

**Note:** Values considered to be order of magnitude estimates

The Risk Assessment Report (European Commission, 2008) estimated “regional” emissions of HBCD from a population of approximately 20 million people as follows:

- Industrial formulation/use: Air: 41 kg/year Water: 850 kg/year
- Installation/Service use: Air: 23 kg/year Water: 25 kg/year
The equivalent figures from Table 4 are:

- **Industrial formulation/use:** Air: 7 kg/year Water: 730 kg/year
- **Installation/Service use:** Air: 130,000 kg/year Water: 13,000 kg/year

The figures for industrial discharge to water set out in Table 4 were partly based on the EC (2008) figures. Differences between these datasets reflect the more detailed focus on industrial manufacturing processes in the EC study, as well as the controls on process emissions via the Environmental Permitting system in the UK. Conversely, the EC (2008) figures for installation and service use are much lower than the values in Table 4. This may reflect the more detailed focus on these sources in the present study. As these sources are likely to dominate emissions and public/environmental exposure, it is considered important for the purposes of this study that these sources are represented as well as the available data permit.

The estimates in Table 4 should be considered as order of magnitude estimates only, and can be used to focus attention on key sectors and sources.

The estimated release inventory is shown graphically in Figure 4. The data in Table 4 and Figure 4 indicate that the most significant environmental releases of HBCD are:

- Disposal to land of EPS/XPS board is forecast to become increasingly significant up to 2030. In 2030, disposal to land of EPS/XPS is estimated to account for 49% of all HBCD releases
- The next highest estimated release is emissions to air of HBCD from textiles, estimated to account for 18% of all HBCD releases in 2030.
- Disposal to landfill of waste EPS/XPS board during installation is estimated to account for 13% of all HBCD releases in 2030.
- Disposal of end-of-life textiles containing HBCD to landfill is estimated to account for 12% of all HBCD releases in 2030.

![Figure 4: Estimated UK annual release inventory of HBCD](image)
3.4 Controls on HBCD

At present, there are no specific controls on HBCD during the manufacture, use and disposal of goods containing HBCD in the UK. HBCD has recently been confirmed as a POP under the terms of the Long-Range Transboundary Air Pollution (LRTAP) Convention. Controls aimed at minimising emissions of HBCD under the terms of this convention may therefore be expected in the future.

Although there are no specific controls on HBCD at present, there are a number of regimes which currently affect HBCD, and which could potentially be used for improved control of HBCD. The control regimes identified during the course of this study are:

- The Environmental Permitting (England and Wales) Regulations 2007 (Statutory Instrument 2007 No. 3538) as amended
  This lays down environmental controls on (inter alia) the landfill of waste materials. Could be used to introduce concentration limits and monitoring regimes for HBCD, possibly via updated guidance to waste facility operators under Environmental Permitting.

- The Waste Electrical and Electronic Equipment Regulations 2006 (Statutory Instrument 2006 No. 3289) as amended
  This lays down targets for recovery and recycling of waste electrical and electronic equipment. Includes record-keeping requirements. Could be adapted to provide for separate processing of waste containing high levels of HBCD

- HBCD was identified as a “Substance of Very High Concern” (SVHC) under the REACH Regulations in October 2008, and added to the Candidate List for inclusion in Annex XIV of the REACH Regulations. This is the first step of the authorisation procedure for substances with adverse toxicological and environmental properties. Once included in Annex XIV, a SVHC cannot be placed on the market or used after a specified date, unless the company is granted an authorisation. At present, no substances have been included in Annex XIV. However, if HBCD were to be added to Annex XIV of the REACH regulations, the procedures laid down for control of chemicals under REACH, including enforcement provisions, would be available for application to HBCD. Further information is available from the Health and Safety Executive, the competent authority for implementation of the REACH regulations (www.hse.gov.uk/reach).

- HBCD is under consideration for inclusion in revisions to the Restriction of Hazardous Substances (RoHS) directive (2002/95). The Restriction of the Use of Certain Hazardous Substances in Electrical and Electronic Equipment Regulations 2008 (the “RoHS Regulations”) implement EU Directive 2002/95 which bans the placing on the EU market of new electrical and electronic equipment containing more than agreed levels of chemicals including polybrominated biphenyl and polybrominated diphenyl ether flame retardants. Revisions to the directive and regulations could potentially be used as the means for implementing controls on HBCD in the electronics industry.

- The European brominated flame retardant industry operates a programme called the “Voluntary Emissions Control Action Programme.” This aims to reduce emissions of a range of brominated chemicals, and covers producers and industrial users of these chemicals. This is now linked with a programme focused on management of HBCD in the polystyrene foams supply chain, known as “SECURE.” This programme has resulted in improvements in the control of HBCD emissions at source. It covers 60 of the 73 European user sites for HBCD, accounting for approximately 93% of HBCD consumption in Europe.

- Other existing voluntary programmes could potentially be applied to the control of HBCD. These include “Responsible Care”, the chemicals industry initiative for
improving environmental, health and safety performance. The construction industry research and information association (CIRIA) provides advice and information for those engaged with the delivery and operation of the built environment. There may be opportunities for developing awareness programmes alongside CIRIA’s current training resources (see www.ciria.org). A further option could be extending the scope of the BRE Environmental Assessment Method (BREEAM) for assessment of buildings to include consideration of brominated flame retardant use (see www.breeam.org for further information).
4 Alternatives to HBCD

4.1 Introduction

HBCD is currently used as a flame retardant substance in EPS, XPS, HIPS and textiles. Alternative products are commercially available, although technical performance and economic viability might be an issue for some applications. The options for alternatives to current uses of HBCD may be classified as follows (Norwegian Climate and Pollution Agency, 2010):

- Drop-in chemical substitute
- Resin and material substitution
- Other insulation materials
- Product redesign

These options are discussed in this chapter. The cost-benefit assessment is set out in Chapter 5.

4.2 Building sector

Three options for alternatives to HBCD in EPS and XPS insulation can be identified:

- Alternative brominated flame retardants;
- Resin/material substitution: polyurethane and polyisocyanurate products, or phenolic foam;
- Other insulation materials.

4.2.1 Alternative flame retardants

Examples of commercially available flame retardants which could potentially be suitable for use as a replacement for HBCD in EPS are as follows:

- Saytex BC-48 (contains tetrabromo-cyclooctane);
- Saytex BCL- 462 (contains dibromoethyl dibromo-cyclohexane);
- BE- 51 (contains TBBPA).

These materials are also based on polybrominated chemicals, which may potentially have associated environmental issues. A preliminary search revealed no readily available environmental studies of tetrabromo-cyclo octane or dibromoethyl dibromo-cyclohexane. In view of the absence of detailed environmental information, it was concluded that these materials should not be considered as readily available for use in the construction sector due to limitations imposed as a result of the REACH regulations for new chemical applications. The EPS sector group of Plastics Europe indicated in 2008 that there were no technically and economically viable HBCD substitutes for the use of HBCD in EPS and XPS (as reported in ECHA, 2008). This is because any currently available flame retardant would need to be added to EPS/XPS at higher concentration than HBCD. This would adversely affect the physical and thermal properties of the material.

It is concluded that there are no available alternative flame retardants for use in building insulation.
4.2.2 Resin/material substitution

Two groups of alternative substitute materials have been identified.

Polyurethane and Polyisocyanurates. Polyisocyanurate modified urethane foams are used in a variety of construction applications and are commonly referred to as “polyiso” products. Possible applications are: single ply membrane systems and cold-applied modified bitumen applications, insulation and interior finish system for walls and ceilings, commercial interior wall construction behind gypsum board, thermal insulation of large panels, water heaters, and storage tanks. Polyiso insulation products use the following flame retardant chemicals: tris monochloropropyl phosphate (TMCPP), trichloroethyl phosphate (TCEP) and RB-79 (diester or ether diol of tetrabromophthalic anhydride). TCEP is manufactured from ethylene oxide and phosphorus oxychloride. Polyisocyanurate is not recyclable at present and, depending on the flame retardant used (for instance, TCEP), there may also be concerns with regard to potential effects on human health.

Polyurethane is also technically viable. However, it is more expensive than EPS: the ECHA (2009) estimated that raw material costs are about three times those of EPS, and similar to XPS.

Phenolic foam. Phenolic resins are used to bind glass fibre, mineral wool or shredded waste to make insulation products. Glass fibre is the most commonly used material, accounting for 88% of all phenolic insulation products. The raw materials used to produce the phenolic resin monomer are phenol and formaldehyde.

The European Phenolic Foam Association (2010) indicates that phenolic foam has good performance with regard to fire resistance, meeting the UK Class 0 fire certification. Its thermal conductivity performance is also more efficient than comparable products. Although the currently small market share suggests a relatively high cost, an increase in the market share has the potential to lead to economies of scale, resulting in reduced manufacturing costs. However, there are concerns over the potential for adverse effects on human health and the environment. Phenolic foam is manufactured from potentially toxic and carcinogenic materials and is not recyclable at present.

4.2.3 Alternative insulation materials

This sector accounts for the majority of HBCD used in the UK.

There are several insulating materials that can be used as an alternative to EPS and XPS in the building sector. Flame retardancy is generally provided by the use of fire resistant coverings or coatings that separate the insulation material from the building interior, acting as thermal barriers. The thermal barriers commonly used include: gypsum board, gypsum or cement plasters, perlite board, spray-applied cellulose, mineral fibre or gypsum coatings and select plywood.

Insulating products alternative to EPS and XPS in the building sector are:

- Blanket insulation. Usually made of fibreglass or rockwool, it can be fitted between studs, joists, and beams. Fibreglass and rockwool are synthetic vitreous and mineral fibres;
- Loose-fill insulation. Typically blown into place or spray-applied, it can be used to fill existing wall cavities and for irregularly shaped areas. Materials used for blown-in or spray-applied insulation include rockwool, fibre glass, cellulose or polyurethane foam. Loose-fill cellulose insulation can be made of waste paper (recycled newsprint, cardboard, etc) and needs to be treated with flame retardants and pesticides such as boron salts.
- Vermiculite or perlite (produced from the expansion of minerals in a furnace) can also be used as loose-fill insulation;
• Rigid insulation. The materials typically used include polyisocyanurate. This may be faced with reflective foil to reduce heat flow;

• Reflective insulation systems (thermal barriers). Common reflective system materials include foil-faced paper, foil-faced polyethylene bubbles, foil-faced plastic film and foil-faced cardboard. The thermal barriers used are gypsum board, gypsum or cement plasters, perlite board, spray-applied cellulose, mineral fibre or gypsum coatings and select plywoods. The Norwegian Climate and Pollution Agency suggests that EPS or XPS can be used without a flame retardant, using a construction method including a glass fibre layer to provide flame retardancy. This approach is unlikely to be an option for existing buildings.

Fibre glass blankets or batts, loose-fill fibre glass, perlite and loose-fill rock wool are innately fire and heat resistant, and pose no more than a minor risk to health. The materials can be recycled after use. The industry (Eurisol, 2010) indicates that for each tonne of carbon dioxide generated during the manufacture of mineral wool, around 200 tonnes of carbon dioxide are saved over a 50 years period from the thermal benefits of the insulation. The raw material cost of mineral wool is reported to be slightly higher than the EPS raw material cost and lower than the raw material cost of XPS. However, as they are less effective insulators than XPS and polyisocyanurate boards (as measured by the “R-value”), they are not the ideal options for those applications where a high R-value is required (Lowell Centre, 2006).

With regard to cellulose insulation, as the material used has a relatively low moisture resistance and degrades in damp conditions, its application in building insulation is not the best technical option in northern European countries.

Table 5: Alternative products for use in construction

<table>
<thead>
<tr>
<th>Alternative to HBCD</th>
<th>Potential human health issues</th>
<th>Potential environmental issues</th>
</tr>
</thead>
<tbody>
<tr>
<td>Polyurethane and polyisocyanurate products</td>
<td>May emit toxic fumes if burnt, otherwise low toxicity in use, but manufacture involves the use of isocyanates (potent respiratory sensitisers)</td>
<td>No recycling at present. Long term disposal to landfill with potential for dust emissions to air and surface water</td>
</tr>
<tr>
<td>Phenolic foam</td>
<td>Low toxicity in use but manufactured from potentially toxic and carcinogenic materials</td>
<td>No recycling at present. Long term disposal to landfill with potential for dust emissions to air and surface water</td>
</tr>
<tr>
<td>Alternative insulation - Thermal barriers - Loose-fill insulation - Blanket insulation</td>
<td>Relatively minor health issues - Inhalation of low toxicity dust generated during installation and removal; no significant emissions while in use in buildings. Manufacture of mineral wool requires control of combustion process emissions</td>
<td>Materials can be recycled post consumer use</td>
</tr>
</tbody>
</table>

Based on the above summary, mineral wool-based insulation materials seem to be the most appropriate alternative to HBCD use in EPS for building insulation purposes, in terms of technical, environmental and economic performance. Furthermore, mineral wool insulation does not require flame retardant substances, as it is innately fire resistant.

However, mineral wool-based insulation materials exhibit a lower R-value than XPS, hence making it a sub-optimal alternative to XPS in any applications where high R-values are required.
In an assessment for the European Chemical Agency, when addressing the alternative to HBCD in EPS and XPS, IOM stated: “There are however a number of alternative forms of insulation that can be used in place of XPS or EPS. These alternative insulation systems have different characteristics to XPS and EPS and may be less appropriate for some specific use scenarios or may incorporate different environmental issues such as increased energy costs during transportation” (ECHA, 2008).

It is possible that there would be no alternative insulating materials available, in the event of a ban on the use of HBCD for thermal insulation. If this were to occur, this could result in reduced thermal efficiency and consequently increased emissions of carbon dioxide. An industry body (EUMEPS 2010) estimates that 1 m3 of EPS would save carbon dioxide emissions equivalent to those that would result from approximately 22 kg fuel oil over a 6 month period. This is approximately equivalent to a saving of 6 tonnes CO2 per year per tonne of EPS, or 300 tonnes CO2 per tonne of EPS over a typical 50 year product lifetime. The equivalent figure for XPS is estimated to be about 240 tonnes CO2 per tonne of XPS over a typical 50 year lifetime. Applying these factors to the estimated quantities of EPS and XPS produced per year indicates that a ban on the use of HBCD for thermal insulation could result in an increased emission of 3 to 17 million tonnes CO2 per year. This increased CO2 would be released over the 50 year product lifetime. If alternative insulation products were available, the amount of CO2 released would be less.

### 4.3 High Impact Polystyrene

Apart from its application in the building sector, HBCD is also used in HIPS and the textiles sector. Table 6 below sets out alternative products to HBCD suitable for HIPS applications, together with the associated impacts on human health and the environment (table adapted from ECHA, 2008):

**Table 6: Alternate products for use in HIPS applications**

<table>
<thead>
<tr>
<th>Alternative to HBCD</th>
<th>Potential human health issues</th>
<th>Potential environmental issues</th>
</tr>
</thead>
<tbody>
<tr>
<td>Decabromodiphenylether/ATO</td>
<td>Potentially harmful degradation products</td>
<td>Potentially harmful degradation products</td>
</tr>
<tr>
<td>Decabromodiphenylethane/ATO</td>
<td>Limited data, but likely to be of low toxicity</td>
<td>Not readily biodegradable, may be persistent</td>
</tr>
<tr>
<td>Ethylenebis-(tetrabromophthalimide)/ATO</td>
<td>Low toxicity</td>
<td>Not biodegradable, persistent. Non-toxic.</td>
</tr>
<tr>
<td>Triphenyl phosphate</td>
<td>Chronic toxicant with effects on liver</td>
<td>Readily biodegradable, toxic to aquatic organisms</td>
</tr>
<tr>
<td>Resorcinol bis (biphenyl phosphate)</td>
<td>Chronic toxicant with effects on liver</td>
<td>Inherently biodegradable, may be persistent and bio-accumulative</td>
</tr>
<tr>
<td>Bis phenol A bis (biphenyl phosphate)</td>
<td>Limited data, likely to be of low toxicity</td>
<td>Poorly biodegradable. Non-toxic and not bio-accumulative</td>
</tr>
<tr>
<td>Diphenyl cresyl phosphate</td>
<td>Chronic toxicant with effects on liver, kidney and blood. Effects on fertility.</td>
<td>Readily biodegradable</td>
</tr>
<tr>
<td>Polyethylene with Magnesium Hydroxide</td>
<td>Insufficient data but likely to be of low toxicity</td>
<td>Polythene particles are highly persistent in the aquatic environment and may contribute to reduced nutritional intake by organisms.</td>
</tr>
<tr>
<td>Antimony trioxide (ATO), used with halogenated flame retardants</td>
<td>Not an alternative to HBCD, as it is used as a synergistic additive to other flame retardants</td>
<td></td>
</tr>
</tbody>
</table>
As a brominated flame retardant, deca-BDE has associated health and environmental issues, which have been the subject of extensive investigation and evaluation. Defra does not view deca-BDE as an associated substitution product.

There is insufficient available information to make distinctions between the costs associated with HBCD and other brominated flame retardants.

The organic Aryl Phosphorous Compounds listed in Table 6 are commercially available. They constitute viable alternatives to HBCD in HIPS applications from a technical point of view. However, because larger quantities of non halogen flame retardants are required to achieve the same level of protection as HBCD results in comparatively higher manufacturing costs.

The use of polyethylene with magnesium hydroxide is a technically viable alternative and is expected to be less expensive in terms of raw material costs than HIPS containing HBCD. However, additional one-off costs of substitution for product testing and changes to processing equipment have to be factored in.

In an assessment for the European Chemical Agency, when addressing the alternative to HBCD in HIPS applications, IOM stated: “Given that HBCDD is not widely used in HIPS, it is perhaps reasonable to assume that some technically and economically feasible alternatives are already on the market, although it is uncertain whether the human health and environmental impacts of these alternatives are any less than those associated with HBCDD products” (ECHA, 2008).

The Norwegian Climate and Pollution Agency (2010) indicates that a composite of HIPS and polyphenylene ether could potentially offer satisfactory flame retardancy performance for some applications.

### 4.4 Textile sector

The UK has set particularly demanding standards for flame retardancy in soft furnishings. The flame retardants act by protecting the covering against flammability, which in turn protects the contents. However, some flame retardants melt when hot, and expose the foam interior to the fire. In contrast, HBCD and DecaBDE provide highly effective protection and act as a barrier to the rest of the article. These demanding fire resistance standards result in reduced harm from domestic fires, but do result in the requirement for use of brominated flame retardants. Hence, a ban or restriction on the use of HBCD in furnishings could have a significant effect on the sale of interior furnishings in the UK.

One manufacturer has manufactured a set of polymeric bromide flame retardants which reportedly offer satisfactory flame retardance performance. As a polymeric substance, the potential for adverse environmental and health effects may be lower than other brominated chemicals. However, these substances are currently much more expensive than HBCD, and the toxicological and environmental issues are yet to be evaluated. As these have not yet been investigated, the polymeric products are not considered to constitute viable alternatives to HBCD.

The phosphorylated flame retardants are highly fibre specific and this can cause substantial process problems in situations where fabric blends are used (Horrocks, 2010). Horrocks indicated that there are no flame retardants which are ready to use at present. It would take a minimum of 5 years for any new substances to pass through the REACH registration process, and this would be at substantial cost.

Ethane 1,2 (bis) Pentabromophenyl (EBP) has been studied as part of a UK initiative, but has not yet been widely tested for applicability in the textiles sector. It was found to pose generally lower environmental and health risks than substances such as Deca-BDE or
HBCD. EBP represents a potentially viable alternative to HBCD in textiles, but is as yet unproven, and cannot be viewed as a currently available alternative.

The Norwegian Climate and Pollution Agency (2010) indicates that intumescent systems (i.e. a surface layer which would char and swell with heat to provide a barrier to fire spreading) are under consideration. However, these systems are not well proven, particularly with regard to their performance in relation to the demanding UK fire regulations, and are therefore not considered to constitute viable alternatives to HBCD.

The remaining candidate alternative products to HBCD suitable for textiles uses are set out in the table below (adapted from ECHA, 2008):

**Table 7: Candidate alternate products for use in textile applications**

<table>
<thead>
<tr>
<th>Alternative to HBCD</th>
<th>Potential human health issues</th>
<th>Potential environmental issues</th>
</tr>
</thead>
<tbody>
<tr>
<td>Decabromodiphenylether</td>
<td>Neurotoxicant</td>
<td>Not readily biodegradable, low to moderate bioaccumulation potential</td>
</tr>
<tr>
<td>Short and medium chain chlorinated paraffins</td>
<td>Substances exhibit reproductive toxicity, chronic toxicity with effects on liver and kidneys; some are potential carcinogens</td>
<td>SCCPs have been confirmed as POPs. MCCPs have concerns in relation to persistence, bioaccumulation and toxic properties. LCCPs potentially meet the persistent or very persistent criterion. They do not meet the toxic or bio-accumulative criterion</td>
</tr>
<tr>
<td>Phosphorylated substances (e.g. ammonium polyphosphates)</td>
<td>Low toxicity</td>
<td>Not an ecotoxicant</td>
</tr>
</tbody>
</table>

DecaBDE and HBCD are the two key flame retardants for textiles as they can be used on any textile, whereas the phosphorylated flame retardants are much more fibre specific and this can cause problems in situations where fabric blends are used. Solubility of the phosphorylated flame retardants can cause longer-term problems, as the flame retardants can be washed out of the fabric. The ECHA (2008) indicates that the alternative flame retardant products listed in Table 7 are available at a cost which is similar to or lower than HBCD (ECHA, 2008). However, these substances are yet to be fully tested for their performance in relation to the UK market, and in relation to their environmental and toxicological properties. The textiles industry considers that there are no viable alternatives which are ready to use, and the cost of REACH registration means that new chemicals cannot be readily developed (Fretwork, 2010).

It is concluded that alternatives to HBCD and Deca-BDE are not available at present. Deca-BDE is not viewed by Defra as an acceptable alternative to HBCD.

**4.5 Summary**

The candidate options for the different applications of HBCD are summarised as follows:

- **Polystyrene insulation material (EPS and XPS):**
  - No drop-in chemical substitutes are available to our knowledge.
  - Alternative insulating foams are available for some applications.
o Rock wool is already used in preference to EPS or XPS where possible, because of reduced transportation costs.

o Blanket insulation products, possibly using EPS/XPS without the use of flame retardant chemicals may be used in some circumstances.

o It is likely that no alternative products will be suitable for some or all of the current uses of HBCD in EPS and XPS. This means that a ban on the use of HBCD in insulation materials would result in reduced insulation in some buildings. This would carry a penalty in reduced energy efficiency for buildings where alternative materials cannot be used (e.g. existing buildings without a wall cavity), and consequent increases in fuel usage for heating/cooling with associated carbon dioxide emissions.

• High Impact Polystyrene
  o Aryl phosphates could potentially be used as alternative chemical additives in HIPS in the electrical and electronic goods industry, but do have potentially adverse environmental and health effects.
  o A composite of HIPS and polyphenylene ether could potentially offer satisfactory flame retardancy performance
  o Other insulation materials: not applicable
  o Product redesign: no product redesign to enable the use of BFRs to be reduced or eliminated has been identified

• Textile industry
  o No drop-in chemical substitutes are available to our knowledge.
  o Resin and material substitution: not applicable
  o Other insulation materials: not applicable
  o Product redesign: intumescent systems are under consideration, but are not well proven.
  o It is likely that no alternative products will be suitable for the current uses of HBCD in textiles. This means that a ban on the use of HBCD in textiles would result in some textile products becoming unavailable in the UK (e.g. furniture and curtains). This would affect consumer choice, and would also have an effect on the furniture manufacturing industry in the UK and elsewhere.

In summary, alternative options to the use of HBCD are available for some insulation applications. Alternatives to the use of HBCD in HIPS are available for most applications. Alternatives to the use of HBCD in textiles in a wide range of applications are not available. There is no foreseeable prospect of relaxing flame retardancy requirements for HIPS and textiles.
5 Costs, benefits and partial impact assessment of additional HBCD regulation

5.1 Introduction

This chapter collates and discusses the data gathered from research into the potential costs, benefits and other impacts of a range of additional HBCD regulatory options. The assessment was carried out to identify changes from the “base case” or “business as usual” scenario.

If HBCD were listed under Annex A of the Stockholm Convention, there would be a complete ban on manufacture or use in the UK and other signatory countries. In addition it would no longer be permitted to import or to export HBCD on its own, in preparations or as a constituent of articles. This last condition would mean that in practice EPS/XPS, HIPS and textile products that contain HBCD could no longer be traded in the UK.

Furthermore, there would be a requirement for safe disposal of all wastes with a HBCD level above a specified level. A level of 0.2% is suggested as an appropriate threshold. Materials with HBCD above this threshold would include primary materials such as EPS/XPS, HIPS and textiles. However, materials with elevated “secondary” HBCD levels would fall below this threshold. These materials contain HBCD due to the use of recycled HBCD-containing materials in their manufacture. The safe disposal of the primary materials globally would over time tend to reduce the quantity of secondary materials being manufactured. However these secondary materials are not considered in the analysis that follows.

Safe disposal is defined as the destruction or irreversible transformation rather than recycling of any waste materials. For this study we have assumed that the requirement for safe disposal would mean that in practice, primary HBCD materials would have to undergo sorting and separation (where feasible) and the HBCD separated would then go for incineration as a hazardous waste. We have further assumed that under business as usual conditions the textile and EPS/XPS wastes would have gone to landfill as standard wastes.

The remainder of this chapter describes the impacts that would thus arise from the requirement to ban and safely dispose of primary HBCD materials.

5.2 Assessment methods

5.2.1 Key assumptions

We have assumed steady-state market conditions with no growth or shrinkage in the markets for primary HBCD materials from 2010 onwards. This means that the quantities of HBCD materials and products produced would not vary year to year.

A working assumption was made in relation to the timescale for cessation of use of HBCD, in the event of a ban being applied. It was assumed that a ban could be implemented from 2011, with implementation over a 5 year timetable up to 2015, consistent with the 54 month timetable applied to the restrictions on use of HBCD under the REACH regulations. A trajectory was applied in which HBCD is gradually removed from use over the period up to 2015: that is the impacts were assumed to be ‘back-loaded’ with most enterprises waiting until later to action due to regulation. The assumed trajectory is presented below and means
that the impacts, costs and benefits for the scenario would build from 2011 and be fully experienced from 2015 onwards.

**Table 8: Assumed phase-out of HBCD in the event of a ban**

<table>
<thead>
<tr>
<th>Year</th>
<th>2010</th>
<th>2011</th>
<th>2012</th>
<th>2013</th>
<th>2014</th>
<th>2015</th>
</tr>
</thead>
<tbody>
<tr>
<td>Reduction from baseline levels</td>
<td>0%</td>
<td>10%</td>
<td>25%</td>
<td>45%</td>
<td>70%</td>
<td>100%</td>
</tr>
</tbody>
</table>

We have assumed that the lifetime of HIPS and textile products is 10 years on average. Therefore there would be no HIPS or textile products being produced from 2015 onwards, no wastes arising from 2025 onwards and the quantities of wastes would reduce from approximately 2020 onwards.

The fate of EPS/XPS products is more complex since the lifetime of insulation is longer (we have assumed an average of 50 years) and wastes of these products are currently increasing as a legacy of their gradual introduction to building stock in past years (see table 4 for data and discussion of wastes arising from these products). Under business as usual conditions primary HBCD wastes arising may peak in 2030 after which a steady state would occur (assuming no change in demand for these products). This means that, in the event of a restriction being applied, although no EPS/XPS products would be used from 2015 onwards, the quantities of primary HBCD wastes would not begin to decrease until 2045, with progressive reductions thereafter.

Our approach has been to estimate the quantities (tonnes) of primary HBCD products and wastes that are produced annually and the quantities alternative products that would be required to replace the HBCD products (where these are feasible). As HBCD is a constituent of the primary products the quantities of wastes that would require sorting and separation of HBCD (where this is possible) are much larger than the tonnage of HBCD actually used.

**EPS/XPS**

Other government policies such as the UK Climate Change Programme will ensure that there will continue to be strong drivers for insulation products to be used in buildings, hence EPS/XPS materials would be replaced as far as possible. The following data were used to quantify amounts of HBCD and alternative products in the building insulation market.

**Table 9: Data relating to amounts of EPS, XPS and alternative products**

<table>
<thead>
<tr>
<th>Insulation product</th>
<th>Density (kg/m³)</th>
<th>Thickness (mm)</th>
<th>Market share (%) (BPF, 2005)</th>
</tr>
</thead>
<tbody>
<tr>
<td>EPS</td>
<td>24</td>
<td>75</td>
<td>13</td>
</tr>
<tr>
<td>XPS</td>
<td>40</td>
<td>75</td>
<td>8</td>
</tr>
<tr>
<td>Mineral wool</td>
<td>20</td>
<td>100</td>
<td>44</td>
</tr>
<tr>
<td>Polyurethane</td>
<td>40</td>
<td>40</td>
<td>23</td>
</tr>
</tbody>
</table>

Since replacements for EPS/XPS are not available for all applications, we assumed that 75% of EPS/XPS could be replaced by either mineral wool or polyurethane and that the share of the replacement market would be split between these two alternatives according to their relative market share (i.e. approximately twice as much mineral wool products would be used as polyurethane products).

In the event of restriction on the use of HCB in the construction industry, it was assumed that EPS/XPS wastes would be separated from demolition wastes, diverted from landfill and sent for incineration.
HIPS

It was assumed that HBCD could be replaced by alternative flame retardant products with no change in quantities arising.

It was assumed that EEE wastes can be sorted and HIPS separated from the remaining wastes. HIPS wastes containing HBCD would be diverted from WEEE process and sent for incineration, or potentially for recovery of bromine and hydrocarbons, subject to appropriate market conditions for the use of this technology.

Textiles

The evaluation set out in Section 4 indicates that there are no appropriate replacements for HBCD within this sector. The use of non-flame retardared products, or inadequately protected products, would not be tenable.

There are no readily available methods for sorting and removing the flame-retarded materials from the remainder of the textile products. It was therefore assumed that the entire waste stream would need to be diverted from landfill and sent for incineration.

5.2.2 Impacts on UK industry

We consulted with the UK trade associations related to the sectors most likely to be affected by regulations; i.e. the construction industry (and the provision and use of insulation materials in particular), the textiles industry and the plastics industry. The study focused on those alternative options identified as technically viable, based on the evidence from published literature and consultation with the relevant industries and trade associations (see chapter 4 for detailed discussion). The published literature provided largely qualitative information on the alternatives to HBCD. Consequently, in order to obtain quantitative data, a questionnaire was developed and discussed with relevant industry stakeholders, with a view to both improving the understanding of the market structure of the affected sectors and gathering all the information required to perform a more detailed analysis of the impacts which would potentially be associated with a change from HBCD to its alternatives. Although stakeholders could not provide all information that would be required to carry out a full quantitative analysis, it was possible to reliably assess the direction of changes envisaged under each scenario.

With regard to the broader impacts we surveyed impacts on:

- Overall level of employment and distribution within Small and Medium size Enterprises;
- Market competitiveness;
- Investments such as R&D and
- Consumer choice.

The following industry stakeholders were contacted:

- Fretwork (Flame retardants in textile organisation) Textiles
- British Interior Textiles Association Textiles
- The Carpet Foundation Textiles
- British Plastics Federation Construction, HIPS
- Springvale - XPS/EPS manufacturer Construction
- Kay-Metzeler - XPS/EPS manufacturer Construction
- Sundolitt - XPS/EPS manufacturer Construction
- Richard Horrocks - Bolton Uni Textiles, HIPS
The following sections present the data that was received, supplemented where possible by a wider survey of available data. Assumptions and uncertainties are made explicit throughout.

5.2.3 Costs

We have estimated the annual costs for the lifetime impact of a ban on and destruction or irreversible transformation of HBCD products. Values presented are annualised (2010 prices) and future annual costs have been discounted at a rate of 3.5%.

The analysis of costs covered the variation in costs envisaged with the application of additional controls to HBCD in the building insulation, textiles and electronics industries. The costs analysis focused on two issues: 1) the differential cost of replacement products or materials and 2) the differential cost of the destruction or irreversible transformation of the remaining HBCD materials.

5.2.3.1 Replacement product costs

The differential replacement product costs (where such replacements are feasible) potentially consist of:

1. Price of alternative flame retardants or other process inputs relative to the existing HBCD process;
2. Price of alternative products relative to the existing HBCD products;
3. Additional one-off R&D costs or other capital costs for changing/adapting production processes;

Any additional costs are assumed to be entirely passed on in the market price for the products under consideration. Net product replacement costs account for the reduction in HBCD product markets but the increase in markets for alternative products.

EPS/XPS

Product prices used in the analysis are presented below.

Table 10: Pricing for EPS, XPS and alternative products

<table>
<thead>
<tr>
<th>Insulation product</th>
<th>Product price (£/m$^2$)</th>
<th>Product price (£/kg)</th>
</tr>
</thead>
<tbody>
<tr>
<td>EPS</td>
<td>22</td>
<td>12</td>
</tr>
<tr>
<td>XPS</td>
<td>22</td>
<td>36</td>
</tr>
<tr>
<td>Mineral wool</td>
<td>38</td>
<td>19</td>
</tr>
<tr>
<td>Polyurethane</td>
<td>24</td>
<td>15</td>
</tr>
</tbody>
</table>

HIPS

The replacement of HBCD with alternative flame retardants has been estimated to involve additional annual costs of production for the UK of £1.24 million and £3.39 million, depending on whether HBCD is replaced by other BFR or non-halogenated flame retardants. These costs are the averages from the low-case and high-case values extrapolated from the figures reported by the Danish Environment Ministry with regards to the costs impacts in the EU-27 market (Danish Environment Ministry, 2010).

Textiles

Since there are no feasible replacement materials we have simply assessed the potential UK market size for furniture to indicate the scale of impact which could theoretically result in the event of a ban on the use of HBCD in textiles.
5.2.3.2 Costs of destruction or irreversible transformation

Safe disposal costs involve the following elements:

- Administrative or regulatory costs for new waste management regimes
- Sorting and separation costs for HBCD materials
- Incineration costs for the HBCD materials
- Reduced landfill tax costs for HBCD materials diverted

Administrative costs

EPS/XPS

At present, the building and demolition industry is not subject to specific controls on disposal of waste materials, other than controls on specific hazardous materials such as asbestos, contaminated soil or other hazardous wastes. The construction and demolition industry is also required to comply with general waste management legislation and regulation, such as the provisions of the Environmental Protection Act in relation to fly tipping, and restrictions on open waste burning (see www.netregs.gov.uk/netregs/businesses/construction/62345.aspx).

The EU Waste Framework Directive (2008/312) Article 11b envisages an increase in re-use and recycling of construction and demolition waste to 70% of the waste materials generated by 2020. However, Defra and the Welsh Assembly Government do not propose to introduce any additional policy measures to meet this target at present because England and Wales are already close to or achieving the target (Defra, 2010).

Therefore, the costs that a builder would incur for registering to a scheme of building materials recycling that include HBCD separation and destruction or irreversible transformation is assumed to be an entirely new cost. The registration cost was assumed to be similar to the cost incurred for registering under WEEE. These costs are estimated to be £5,703 in the first year and £2,103 in subsequent years per building firm (Environ, 2010).

HIPS

HIPS containing HBCD is a component of Electric and Electronic Equipments, which are subject to the WEEE Directive. Registering and licensing costs are already being incurred in this sector by manufacturers, retailers and distributors of EEE in order to comply with the WEEE Directive. It was assumed that no additional administrative costs would accrue to the EEE sector in the event of a ban on the use of HBCD in this sector.

Textiles

Currently, about 500,000 tonnes of bulky and textile waste is reused (Defra, 2009). This is estimated to represent about 15% of bulky/textile waste arisings (Environment Council, undated), suggesting that approximately 2,800,000 tonnes of bulky/textile waste is disposed of per year in the UK.

Although voluntary re-use or disposal programmes are well established (e.g. Textile Recycling Association, Salvation Army, Oxfam and local schemes), there are no specific regulatory requirements applying to textile products. Hence a new requirement for destruction or irreversible transformation of the HBCD contained in textiles is expected to generate additional registering, regulatory, compliance and safe disposal costs.

The costs that such enterprises would incur for registering to a scheme that includes HBCD separation and destruction or irreversible transformation is assumed to be similar to the cost incurred for registering under the WEEE scheme. These costs are estimated to be £5,703 in the first year and £2,103 in subsequent years per enterprise.

Summary

If a separate system is introduced for control of HBCD, this would result in costs incurred by enterprises and the regulatory authorities. In view of the limited range of materials and single
chemical focus, this regulation is expected to be less complex than the regulation of WEEE. Consequently, these costs have been assumed to be about 25% of those incurred by the Environment Agency in regulating compliance with the WEEE Directive. The Environment Agency spent £58 million on waste regulation in 2009/10 (Environment Agency, 2010). This covered 9 separate waste regulation schemes, of which WEEE was one. There is no information available on the breakdown of costs between the nine schemes, and so as a first estimate, the regulatory cost for managing HBCD in England and Wales has been estimated as being 25% of one-ninth of the annual spend on waste regulation – that is, £1.6 million annually.

This cost would cover the management of a scheme to cover all HBCD-containing products including EPS/XPS, HIPS and textiles.

It is assumed that all administrative costs would be transferred firstly to the regulated waste facility operators (via a permit/licence charging scheme), onward to the enterprises transferring the waste (for construction and demolition waste in particular), add to their costs for waste management registration costs thence finally to the consumers and building developers who give rise to the demand for HBCD-containing products or their disposal.

Sorting/separation costs

EPS/XPS in construction and demolition wastes

As EPS and XPS containing HBCD are polymers, it has been assumed that similar processes to those used to separate HBCD from HIPS would be suitable for EPS and XPS waste. Therefore, the cost of compliance incurred by treatment facilities is likely to be similar (see later discussion of these costs). This is an estimated cost of £600/tonne of waste (excluding energy costs, recycling costs and other running costs) to separate HBCD from the waste.

HIPS

One option under this scenario would be to separate HBCD from the polymer matrix and then dispose of the HBCD in an appropriate facility. At least one process is available for this. The process comprises a separation technology which could become commercially competitive under specific market conditions, such as those resulting from a requirement for separation and treatment of HIPS containing HBCD. Granulated feed plastic is dissolved in a high boiling point solvent. A series of filtration, precipitation and particle formation extracts BFR from the polymers into the solvents which are recovered by crystallisation. The polymer is extruded to form a recycled product with a bromine content of less than 0.1% (WRAP, 2006).

The capital cost of this separation process is estimated at £12 million. The incremental capital expenditure is then combined with additional operating costs for the use of solvents (£12.5/t feed) (WRAP, 2006) and energy (to treat the feed and regenerate the solvent). The separation costs involved with the use of this process are approximately £600/tonne of HBCD containing material.

Textiles

There is no information available on existing techniques to separate HBCD from textiles. Therefore the most appropriate approach to destruction or irreversible transformation of HBCD is likely to be the disposal of textiles themselves by incineration or safe landfill, without separating out the HBCD from the fabric.

Incineration costs

The cost of disposing of separated HBCD in a safe way has been assumed to be equal to the cost of incinerating or land-filling hazardous waste: £700/tonne of HBCD assumed to be ultimately borne by the consumer either including such materials in buildings or to the body contracting demolition and waste management services.
Landfill costs

Landfill tax costs for 2010/11 are £48/tonne with regulation suggesting a minimum £8/tonne increase annually until 2014/15 at least. We have assumed that the tax would not rise after 2015. Net safe disposal costs account for the reduction in landfill tax costs.

5.2.4 Benefits

A full benefit analysis would proceed through to a monetisation of any benefits; thus allowing like-for-like costs and benefits to be compared. Such an approach is possible for a limited range of pollutants such as particulate matter where research and government guidance has defined the damage costs (i.e. social costs of human health impacts) associated with emissions to air. However, no such values currently exist for HBCD and the work involved to derive such is beyond the scope of this assessment.

Therefore, at this stage our approach to identifying the potential annual benefits of additional regulations has been to quantify the reduction in annual emissions of HBCD associated with the regulatory options. These values are derived from Table 4 which presents the baseline annual emissions of HBCD. Note that this analysis includes two separate release pathways:

1. Via manufacture and use
2. Via wastes disposal and recycling

5.3 Potential regulatory scenarios

The following scenarios have been defined and researched for this chapter:

5.3.1 Base case: no additional regulation

This is the current situation with no additional restriction on the use of HBCD. It is included for comparison with the following scenarios.

5.3.2 Scenario 1: complete ban on UK manufacture or use; safe disposal

HBCD would be listed under Annex A of the Stockholm Convention – i.e. a complete ban on manufacture or use. It would no longer be permitted to import or to export HBCD on its own, in preparations or as a constituent of articles. It would not be permitted to manufacture or use HBCD in the UK.

Where possible, alternative flame retardants to HBCD would be used or alternative products would replace the primary HBCD products. HBCD products would have to be destroyed or irreversibly transformed, and there would therefore be a reduction in the quantity of wastes going to landfill.

5.3.3 Scenario 2: exemption for the textile industry; safe disposal

HBCD would be listed under Annex B of the Stockholm Convention but with derogation granted to textile products. The effects would be as set out for Scenario 1, but without the potential effect on the UK furniture and textile industry.

5.3.4 Scenario 3: exemption for use of EPS/XPS in cavity insulation; safe disposal

HBCD would be listed under Annex B of the Stockholm Convention but with derogation granted to EPS/XPS in cavity insulation where mineral materials cannot be used. The
effects would be as set out for Scenario 1, but without the potential effect on the UK construction industry associated with the use of these products.

5.3.5 Scenario 4: exemption for use of EPS/XPS in cladded insulation; safe disposal

HBCD would be listed under Annex B of the Stockholm Convention but with derogation granted to EPS/XPS as an internal/external cladding material for properties with no wall cavity. The effects would be as set out for Scenario 1, but without the potential effect on the UK construction industry associated with the use of these products.

5.3.6 Scenario 5: safe disposal of materials containing HBCD

Under all circumstances where HBCD is listed under Annex B of the Stockholm Convention there would be a requirement to destroy or irreversibly transform wastes containing HBCD. This scenario simply isolates these safe disposal costs and benefits from others.

Each of these scenarios is now examined in turn.

5.4 Base case: no additional regulation

5.4.1 Consequence on UK industry

This scenario would have no impacts on UK industry which would continue under business as usual conditions. That is, approximately 500-1000 tonnes of HBCD would be imported and used in the UK annually.

5.4.2 Benefits

Following the method described previously the annual average UK HBCD emissions have been calculated.

**Table 11: Annual average HBCD emissions (2010-2030) (Base Case)**

<table>
<thead>
<tr>
<th>Source</th>
<th>Estimated annual release (kg)</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Air (vapour)</td>
</tr>
<tr>
<td>Manufacture of goods containing HBCD</td>
<td>7</td>
</tr>
<tr>
<td>Manufacture of EPS/XPS</td>
<td>4.2</td>
</tr>
<tr>
<td>Manufacture of HIPS</td>
<td>0.3</td>
</tr>
<tr>
<td>Manufacture of Textiles</td>
<td>2.6</td>
</tr>
<tr>
<td>Use of goods containing HBCD</td>
<td>130000</td>
</tr>
<tr>
<td>Use of EPS/XPS</td>
<td>7000</td>
</tr>
<tr>
<td>Use of HIPS</td>
<td>50</td>
</tr>
<tr>
<td>Use of Textiles</td>
<td>130000</td>
</tr>
<tr>
<td>Disposal/recycling of goods containing HBCD (2010)</td>
<td>3000</td>
</tr>
<tr>
<td>Disposal/recycling of goods containing HBCD (2030)</td>
<td>270</td>
</tr>
</tbody>
</table>
As this is the base case (i.e. no additional regulation) no benefits accrue and it is assumed that this level of annual average emission would continue.

5.4.3 Costs

As this is the base case (i.e. no additional regulations) no additional costs accrue.

5.5 Scenario 1: complete ban on UK manufacture or use

5.5.1 Impact on UK industry

Construction materials Sector

A complete ban on the use of HBCD could potentially force twelve manufacturers of EPS and XPS out of business. The two XPS manufacturers have been reported to be companies with less than 250 employees. Four of the ten EPS manufacturers are companies with less than 250 and more than 50 employees. The remainder is made up of companies with less than 50 employees. Therefore, 600 – 1800 jobs are at risk from this scenario. 30% of the impact in terms of jobs would fall on small producers.

However, the income generated and distributed by the EPS and XPS industry is expected to be at least partly absorbed by the manufacturers of alternative insulating materials, mineral wool and polyurethane in particular (see below). That is, competition and opportunity within the insulation material industry would be skewed away from EPS/XPS manufacture towards these alternatives.

The net effect on jobs would rely on the capacity of the alternative insulating materials manufacturers to increase production without increasing the number of employees.

Box 1 Industrial capacity and employment rate

The ability of alternative product manufacturers to increase employment will mostly depend on the marginal rate of technical substitution between the factors of production (capital and labour) required to achieve a predetermined level of output. As the combination of the inputs of production reflects the technology of each plant, an increase in production needed to meet the higher demand for, say, mineral wool is likely to require more inputs. If the technology is kept unchanged (e.g. the optimal combination of the inputs of production is constant) more labour and more capital will be needed, resulting in additional investment. However, if a company does not have access to credit or ready funds, it might be not able to finance the level of investment required, and would then be unable to increase the level of employment.
During a period of more difficult access to credit and uncertainty in the buildings sector, it is anticipated that small and medium-sized companies may find it difficult to increase their capacity and employment rates. Overall, the net impact on employment is assessed as being negative in the short term (less than one year), due to the lag time needed to restructure production. The net impact on employment is highly uncertain in the medium (from two to ten years) to long term (more than 10 years), due to the uncertainty over the investment decisions needed to increase capacity.

Currently available and viable alternatives to the use of HBCD in insulating boards would be to replace EPS/XPS panels with insulating polyurethane or mineral wool-based insulating materials.

The alternatives identified are not like-for-like substitutes for EPS/XPS in every application. They vary in cost, flame retardant properties and insulation properties (e.g. leading to higher energy consumption over the lifetime of the building). In some applications, there are no useable alternatives. We have assumed that this affects 25% of the EPS/XPS market.

Textiles Sector

No available flame-retardant alternatives were identified for applications where HBCD is currently used which comply with current UK fire safety regulation (Furniture and Furnishings (Fire) (Safety) Regulations, SI 1988/ No.1324, as amended by SI 1989/ No.2358 and SI 1993/No. 207). The lack of feasible alternatives that comply with fire safety regulations means that the indirect consequences might potentially be high if the use of HBCD in the manufacture and use of textiles were banned.

The sector affected is mostly made of micro, small and medium-sized manufacturers.

The high cost involved with compliance with the REACH Directive is expected to be a significant disincentive for chemicals companies to invest in the development of new products, especially during a period of protracted economic downturn. Even if companies were willing to undertake an investment to develop new flame retardants, this process would take a long period of time. During this period, if no change is made to the fire safety regulations, most currently available HBCD-treated textiles would be banned in the UK, which would result in a reduction in consumer choice for products such as soft furnishings and curtains.

Hence this option is not considered to be feasible until alternative flame retardants with the same or improved performance are available.

Electronics Sector

A complete ban on HBCD would affect HIPS manufacture and the EEE industry, although brominated and other non-halogenated flame retardant alternatives to HBCD are currently available for use in HIPS. These materials are copolymers either without flame retardants, or using other brominated or non-halogenated flame retardants. Such copolymers have a higher inherent resistance to burning and spreading a fire, because they form an insulating char foam surface when heated. They may also have higher impact strength.

HIPS manufacturers still using HBCD would be required under this scenario to change their manufacturing investment decisions, by using flame retardant additives that they would not otherwise have used. The use of new flame retardants would require a restructuring in the process of production, including replacement of existing moulds with new ones, and incurring additional costs for testing the new material. This could be a significant cost for all those manufacturers that invested in equipment not yet fully recouped and, in particular, for those smaller HIPS manufacturers.

The additional cost of production estimated for UK HIPS manufacturers would fall unevenly over UK manufacturers, according to their size. In 2002, the number of small and medium-sized plastics manufacturers in the UK was 5,260 including 3,365 micro enterprises (from 0 to 9 employees) (Corden and Postle, 2002). Incremental costs are expected to affect micro, small and medium-sized manufacturers more heavily than the larger-sized ones, mostly due
to the proportionally higher burden of identifying suitable alternatives and greater investment in R&D.

5.5.2 Costs

Construction materials Sector

The use of alternative insulating materials in the building sector would result in significantly higher consumer costs for all those applications where the use of EPS is replaced by use of polyurethane, as the raw material used in the latter is more expensive than the raw material used in the former. However, polyurethane raw material costs could decrease over time as a consequence of the economies of scale generated by a widening of its market share.

For all those applications where EPS and XPS are replaced by mineral wool, no significant differences in the price paid by end-user consumers are envisaged.

Textiles Sector

Although alternatives are not considered a viable option in the short term, if new flame retardants were successfully identified, the additional costs to develop, register and produce the new chemical are likely to be passed on to end-user consumers, including households. However, disposable income is unlikely to be significantly affected as the purchase of textiles and upholstery furniture is not a frequently recurring expenditure.

Electronics Sector

The replacement of HBCD with alternative flame retardants has been estimated to involve additional annual costs of production for the UK of £1.243 million and £3.391 million, depending on whether HBCD is replaced by other BFR or non-halogenated flame retardants. These costs are the averages from the low-case and high-case values extrapolated from the figures reported by the Danish Minister of the Environment with regards to the costs impacts in the EU-27 market.

Although the electronics industry could face expenditure, the impact on the final price paid by end-user consumers is likely to be small to negligible as the increased costs of production would be spread over a large volume of products over many years.

To indicate of the size of the impact on consumer prices, it has been estimated (Lassen et al, 2006) that the extra raw material costs of replacing HIPS/Deca-BDE compounds with the alternative polymers PPE/HIPS or PC/ABS with halogen-free flame retardants would be about 5-6 € for the full enclosure of an average 27.5-inch TV-set (front and rear enclosure), of which the average cost of production is around €300. The impact generated by the replacing of HBCD is expected to be of similar magnitude (i.e. approximately ~2%).

Safe Disposal

An additional, unquantifiable number of direct and indirect jobs are likely to be created by the mandatory additional management and destruction or irreversible transformation of textiles, construction and demolition wastes. Regulatory, waste separation and incineration costs would arise and there would be a reduction of material going to landfill.

As noted in Section 4, a ban on the use of HBCD for thermal insulation could result in an increased emission of 3 to 17 million tonnes CO$_2$ per year. This increased CO$_2$ would be released over the 50 year product lifetime.

Table 12 provides a summary of the estimated cost impact of Scenario 1.
The principal influence on costs arises from the requirement to process a large quantity of demolition material in order to separate out materials containing HBCD.

### 5.5.3 Benefits

The removal of HBCD from manufacturing and use activities in the UK would immediately eliminate emissions from these sources. Over time, products not containing HBCD would progressively replace and eventually dominate the material going to waste or recycling.

This scenario would eventually reduce annual emissions by 730t.

The following table details these emissions benefits.

<table>
<thead>
<tr>
<th>Aspect</th>
<th>Cost element</th>
<th>Annual tonnage of material (kT)</th>
<th>Annualised cost (£M)</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td></td>
<td>Low</td>
<td>Mid</td>
</tr>
<tr>
<td></td>
<td>Loss of EPS market</td>
<td>8.2</td>
<td>25</td>
</tr>
<tr>
<td></td>
<td>Loss of XPS market</td>
<td>3.1</td>
<td>11</td>
</tr>
<tr>
<td>Product replacement costs</td>
<td>Increased Mineral wool market</td>
<td>6.2</td>
<td>17</td>
</tr>
<tr>
<td></td>
<td>Increased polyurethane market</td>
<td>2.6</td>
<td>7.2</td>
</tr>
<tr>
<td></td>
<td>Alternative HIPS market (other BFRs)</td>
<td>1.7</td>
<td>3.6</td>
</tr>
<tr>
<td></td>
<td>Alternative HIPS market (non-halogenated)</td>
<td>1.7</td>
<td>3.6</td>
</tr>
<tr>
<td></td>
<td>Loss of textile market (furniture)</td>
<td>140</td>
<td>280</td>
</tr>
<tr>
<td></td>
<td>Total (assuming non-halogenated HIPS)</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Safe disposal costs</td>
<td>Construction and demolition material requiring separation</td>
<td>4,203</td>
<td>13,259</td>
</tr>
<tr>
<td></td>
<td>EPS going to incineration</td>
<td>4.3</td>
<td>13.1</td>
</tr>
<tr>
<td></td>
<td>XPS going to incineration</td>
<td>1.6</td>
<td>5.5</td>
</tr>
<tr>
<td></td>
<td>EEE waste requiring separation</td>
<td>387</td>
<td>387</td>
</tr>
<tr>
<td></td>
<td>HIPS going to incineration</td>
<td>1.4</td>
<td>2.8</td>
</tr>
<tr>
<td></td>
<td>Textile products going to incineration</td>
<td>140</td>
<td>280</td>
</tr>
<tr>
<td></td>
<td>Material diverted from landfill</td>
<td>68</td>
<td>81</td>
</tr>
<tr>
<td></td>
<td>Regulatory costs</td>
<td>1.6</td>
<td>1.6</td>
</tr>
<tr>
<td></td>
<td>Total</td>
<td>4,805</td>
<td>14,028</td>
</tr>
<tr>
<td></td>
<td>Total</td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

The principal influence on costs arises from the requirement to process a large quantity of demolition material in order to separate out materials containing HBCD.
Table 13: Annual average HBCD emission reductions (2010-2030) (Scenario 1)

<table>
<thead>
<tr>
<th>Source</th>
<th>Air (vapour)</th>
<th>Air (dust)</th>
<th>Water</th>
<th>Uncontrolled landfill</th>
<th>Recycling</th>
<th>Total</th>
</tr>
</thead>
<tbody>
<tr>
<td>Manufacture of goods containing HBCD</td>
<td>7</td>
<td>730</td>
<td>60</td>
<td>-</td>
<td>800</td>
<td></td>
</tr>
<tr>
<td>Manufacture of EPS/XPS</td>
<td>4.2</td>
<td>4.9</td>
<td>42</td>
<td>-</td>
<td>51</td>
<td></td>
</tr>
<tr>
<td>Manufacture of HIPS</td>
<td>0.3</td>
<td>0.4</td>
<td>3</td>
<td>-</td>
<td>3.7</td>
<td></td>
</tr>
<tr>
<td>Manufacture of Textiles</td>
<td>2.6</td>
<td>720</td>
<td>15</td>
<td>-</td>
<td>740</td>
<td></td>
</tr>
<tr>
<td>Use of goods containing HBCD</td>
<td>130000</td>
<td>9600</td>
<td>13000</td>
<td>91000</td>
<td>- 240000</td>
<td></td>
</tr>
<tr>
<td>Use of EPS/XPS</td>
<td>7000</td>
<td>-</td>
<td>91000</td>
<td>-</td>
<td>98000</td>
<td></td>
</tr>
<tr>
<td>Use of HIPS</td>
<td>50</td>
<td>0</td>
<td>0</td>
<td>-</td>
<td>50</td>
<td></td>
</tr>
<tr>
<td>Use of Textiles</td>
<td>130000</td>
<td>2500</td>
<td>13000</td>
<td>-</td>
<td>140000</td>
<td></td>
</tr>
<tr>
<td>Disposal/recycling of goods containing HBCD (2010)</td>
<td>3000</td>
<td>2800</td>
<td>0</td>
<td>240000</td>
<td>- 240000</td>
<td></td>
</tr>
<tr>
<td>Disposal/recycling of goods containing HBCD (2030)</td>
<td>270</td>
<td>7200</td>
<td>0</td>
<td>460000</td>
<td>20000</td>
<td>490000</td>
</tr>
<tr>
<td>Disposal/recycling of EPS/XPS (2010)</td>
<td>2800</td>
<td>-</td>
<td>140000</td>
<td>-</td>
<td>140000</td>
<td></td>
</tr>
<tr>
<td>Disposal/recycling of EPS/XPS (2030)</td>
<td>7200</td>
<td>-</td>
<td>350000</td>
<td>-</td>
<td>360000</td>
<td></td>
</tr>
<tr>
<td>Disposal/recycling of HIPS</td>
<td>49</td>
<td>0</td>
<td>20000</td>
<td>20000</td>
<td>40000</td>
<td></td>
</tr>
<tr>
<td>Disposal/recycling of Textiles</td>
<td>220</td>
<td>-</td>
<td>88000</td>
<td>-</td>
<td>89000</td>
<td></td>
</tr>
<tr>
<td>Total (2010)</td>
<td>130000</td>
<td>120000</td>
<td>13000</td>
<td>33000</td>
<td>0</td>
<td>490000</td>
</tr>
<tr>
<td>Total (2030)</td>
<td>130000</td>
<td>170000</td>
<td>13000</td>
<td>55000</td>
<td>20000</td>
<td>730000</td>
</tr>
</tbody>
</table>

5.6 Scenario 2: exemption for critical applications in the textile industry

Under this scenario, there would be no impact on the textiles industry. The impacts on the EPS, XPS and HIPS sectors would be as described under Scenario 1.

5.6.1 UK industry impacts and costs

Construction materials Sector

600 – 1800 jobs could be at risk within the EPS/XPS manufacturing sector although it is uncertain how many jobs could be created by additional demand for the alternative products.

Costs would be the same as in Scenario 1.

Textiles Sector

There would be no impact on the textiles sector.
Electronics Sector

The additional cost of production estimated for UK HIPS manufacturers would fall unevenly over UK manufacturers, according to their size. Costs would be the same as in Scenario 1.

Safe Disposal

An additional, unquantifiable number of direct and indirect jobs are likely to be created by the mandatory additional management and destruction or irreversible transformation of textiles, construction and demolition wastes. Regulatory, waste separation and incineration costs would arise and there would be a reduction of material going to landfill. These impacts would be the same as scenario 1 with the removal of textiles from the analysis.

The following table provides a summary of the estimated cost impact of Scenario 2.

### Table 14: Summary of estimated cost impact (Scenario 2)

<table>
<thead>
<tr>
<th>Aspect</th>
<th>Cost element</th>
<th>Annual tonnage of material (kT)</th>
<th>Annualised cost (£M)</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td></td>
<td>Low</td>
<td>Mid</td>
</tr>
<tr>
<td>Product replacement costs</td>
<td>Loss of EPS market</td>
<td>8.2</td>
<td>25</td>
</tr>
<tr>
<td></td>
<td>Loss of XPS market</td>
<td>3.1</td>
<td>11</td>
</tr>
<tr>
<td></td>
<td>Increased Mineral wool market</td>
<td>6.2</td>
<td>17</td>
</tr>
<tr>
<td></td>
<td>Increased polyurethane market</td>
<td>2.6</td>
<td>7.2</td>
</tr>
<tr>
<td></td>
<td>Alternative HIPS market (other BFRs)</td>
<td>1.7</td>
<td>3.6</td>
</tr>
<tr>
<td></td>
<td>Alternative HIPS market (non-halogenated)</td>
<td>1.7</td>
<td>3.6</td>
</tr>
<tr>
<td></td>
<td>Loss of textile market (furniture)</td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>Total (assuming non-halogenated HIPS)</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Safe disposal costs</td>
<td>Construction and demolition material requiring separation</td>
<td>4203</td>
<td>13259</td>
</tr>
<tr>
<td></td>
<td>EPS going to incineration</td>
<td>4.3</td>
<td>13</td>
</tr>
<tr>
<td></td>
<td>XPS going to incineration</td>
<td>1.6</td>
<td>5.5</td>
</tr>
<tr>
<td></td>
<td>EEE waste requiring separation</td>
<td>387</td>
<td>387</td>
</tr>
<tr>
<td></td>
<td>HIPS going to incineration</td>
<td>1.4</td>
<td>2.8</td>
</tr>
<tr>
<td></td>
<td>Textile products going to incineration</td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>Material diverted from landfill</td>
<td>68</td>
<td>81</td>
</tr>
<tr>
<td></td>
<td>Regulatory costs</td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>Total</td>
<td>4665</td>
<td>13748</td>
</tr>
<tr>
<td></td>
<td>Total</td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

Again, the principal cost is associated with the requirement to process and separate demolition materials containing HBCD.
5.6.2 Benefits

The potential benefits from Scenario 2 are the same as those for Scenario 1 but with any benefits associated with textiles manufacture, use or disposal removed. This scenario would eventually reduce annual emissions by 495 t.

The following table details these emissions benefits.

**Table 15: Annual average HBCD emission reductions (2010-2030) (Scenario 2)**

<table>
<thead>
<tr>
<th>Source</th>
<th>Air (vapour)</th>
<th>Air (dust)</th>
<th>Water</th>
<th>Uncontrolled landfill</th>
<th>Recycling</th>
<th>Total</th>
</tr>
</thead>
<tbody>
<tr>
<td>Manufacture of goods containing HBCD</td>
<td>4.5</td>
<td>5.3</td>
<td>45</td>
<td>-</td>
<td>-</td>
<td>55</td>
</tr>
<tr>
<td>Manufacture of EPS/XPS</td>
<td>4.2</td>
<td>4.9</td>
<td>42</td>
<td>-</td>
<td>-</td>
<td>51</td>
</tr>
<tr>
<td>Manufacture of HIPS</td>
<td>0.3</td>
<td>0.4</td>
<td>3</td>
<td>-</td>
<td>-</td>
<td>3.7</td>
</tr>
<tr>
<td>Manufacture of Textiles</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Use of goods containing HBCD</td>
<td>0</td>
<td>7050</td>
<td>0</td>
<td>91000</td>
<td>-</td>
<td>98050</td>
</tr>
<tr>
<td>Use of EPS/XPS</td>
<td>7000</td>
<td>-</td>
<td>91000</td>
<td>-</td>
<td>98000</td>
<td></td>
</tr>
<tr>
<td>Use of HIPS</td>
<td>50</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>50</td>
<td></td>
</tr>
<tr>
<td>Use of Textiles</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Disposal/recycling of goods containing HBCD (2010)</td>
<td>49</td>
<td>2800</td>
<td>0</td>
<td>160000</td>
<td>-</td>
<td>163000</td>
</tr>
<tr>
<td>Disposal/recycling of goods containing HBCD (2030)</td>
<td>49</td>
<td>7200</td>
<td>0</td>
<td>370000</td>
<td>20000</td>
<td>397000</td>
</tr>
<tr>
<td>Disposal/recycling of EPS/XPS (2010)</td>
<td>2800</td>
<td>-</td>
<td>140000</td>
<td>-</td>
<td>140000</td>
<td></td>
</tr>
<tr>
<td>Disposal/recycling of EPS/XPS (2030)</td>
<td>7200</td>
<td>-</td>
<td>350000</td>
<td>-</td>
<td>360000</td>
<td></td>
</tr>
<tr>
<td>Disposal/recycling of HIPS</td>
<td>49</td>
<td>0</td>
<td>20000</td>
<td>20000</td>
<td>40000</td>
<td></td>
</tr>
<tr>
<td>Disposal/recycling of Textiles</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Total (2010)</td>
<td>49</td>
<td>9800</td>
<td>0</td>
<td>161000</td>
<td>0</td>
<td>261000</td>
</tr>
<tr>
<td>Total (2030)</td>
<td>49</td>
<td>14200</td>
<td>0</td>
<td>461000</td>
<td>20000</td>
<td>495000</td>
</tr>
</tbody>
</table>

5.7 Scenario 3: exemption for use of EPS/XPS in cavity insulation

The potential impacts and benefits associated with this Scenario are the same as those described under Scenario 1 with the exception that use of EPS/XPS products containing HBCD for use in cavity wall insulation would be permitted. These products are used almost exclusively in retrofit situations where mineral materials (the product of choice normally) cannot be used. Industry estimates are that this includes only 5% of total retrofit applications. However, there are no values at this time to indicate the proportion of treated EPS/XPS products that are used in this application. For the purpose of illustrating potential
impacts and benefits we have assumed that 50% of treated EPS/XPS manufacture and use is for this application.

5.7.1 UK industry impacts and costs

Construction materials Sector
If the impacts of this Scenario are spread evenly across the sector then 300 – 900 jobs could remain at risk within the EPS/XPS manufacturing sector although it is uncertain how many jobs could be created by additional demand for the alternative products (i.e. for non-cavity wall retrofit applications). The demand for cavity wall insulation is declining, whereas the demand for non-cavity wall retrofit applications is increasing. If some manufacturers are more specialised and not involved in the non-cavity wall insulation market then the impacts will fall disproportionately on them. There are no data to suggest whether this is the case.

Textiles Sector
The impact on this sector would be the same as under Scenario 1. That is, neither alternative products nor alternative retardants are considered to be viable options in the short term. Consequently this regulation would end the manufacture or use of HBCD containing textiles in the UK. To maintain textile markets and fire regulations, such textiles would be imported from outside the EU with loss of consumer choice and loss of this sector from the UK economy.

Electronics Sector
The additional cost of production estimated for UK HIPS manufacturers would fall unevenly over UK manufacturers, according to their size. Costs would be the same as in Scenario 1.

Safe Disposal
An additional, unquantifiable number of direct and indirect jobs are likely to be created by the mandatory additional management and destruction or irreversible transformation of textiles, construction and demolition wastes. Regulatory, waste separation and incineration costs would arise and there would be a reduction of material going to landfill. These impacts would be the same as scenario 1 with the removal of 50% of EPS/XPS from the analysis.

The following table provides a summary of the estimated cost impact of Scenario 3.
### Table 16: Summary of estimated cost impact (Scenario 3)

<table>
<thead>
<tr>
<th>Aspect</th>
<th>Cost element</th>
<th>Annual tonnage of material (kT)</th>
<th>Annualised cost (£M)</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td></td>
<td>Low</td>
<td>Mid</td>
</tr>
<tr>
<td>Product replacement costs</td>
<td>Loss of EPS market</td>
<td>4</td>
<td>13</td>
</tr>
<tr>
<td></td>
<td>Loss of XPS market</td>
<td>1.6</td>
<td>5.3</td>
</tr>
<tr>
<td></td>
<td>Increased Mineral wool market</td>
<td>3.1</td>
<td>8.6</td>
</tr>
<tr>
<td></td>
<td>Increased polyurethane market</td>
<td>1.3</td>
<td>3.6</td>
</tr>
<tr>
<td></td>
<td>Alternative HIPS market (other BFRs)</td>
<td>1.7</td>
<td>3.6</td>
</tr>
<tr>
<td></td>
<td>Alternative HIPS market (non-halogenated)</td>
<td>1.7</td>
<td>3.6</td>
</tr>
<tr>
<td></td>
<td>Loss of textile market (furniture)</td>
<td>140</td>
<td>280</td>
</tr>
<tr>
<td></td>
<td>Total (assuming non-halogenated HIPS)</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Safe disposal costs</td>
<td>Construction and demolition material requiring separation</td>
<td>4,203</td>
<td>13,259</td>
</tr>
<tr>
<td></td>
<td>EPS going to incineration</td>
<td>2.1</td>
<td>6.5</td>
</tr>
<tr>
<td></td>
<td>XPS going to incineration</td>
<td>0.8</td>
<td>2.7</td>
</tr>
<tr>
<td></td>
<td>EEE waste requiring separation</td>
<td>387</td>
<td>387</td>
</tr>
<tr>
<td></td>
<td>HIPS going to incineration</td>
<td>1.4</td>
<td>2.8</td>
</tr>
<tr>
<td></td>
<td>Textile products going to incineration</td>
<td>140</td>
<td>280</td>
</tr>
<tr>
<td></td>
<td>Material diverted from landfill</td>
<td>68</td>
<td>81</td>
</tr>
<tr>
<td></td>
<td>Regulatory costs</td>
<td>1.6</td>
<td>1.6</td>
</tr>
<tr>
<td></td>
<td>Total</td>
<td>4942</td>
<td>14019</td>
</tr>
<tr>
<td>Total</td>
<td>Total</td>
<td>1,391</td>
<td>3,880</td>
</tr>
</tbody>
</table>

Processing and separation of demolition materials containing HBCD remains the principal cost for Scenario 3.

#### 5.7.2 Benefits

The potential benefits from Scenario 3 are the same as those for Scenario 1 but with benefits associated with EPS/XPS manufacture, use or disposal reduced. This assessment has assumed that 50% of EPS/XPS use is in cavity insulation applications. Therefore this scenario would eventually reduce emissions by 480t. The following table details these emissions benefits.
### Table 17: Annual average HBCD emission reductions (2010-2030) (Scenario 3)

<table>
<thead>
<tr>
<th>Source</th>
<th>Estimated annual emissions reduction (kg)</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Air (vapour)</td>
</tr>
<tr>
<td>Manufacture of goods containing HBCD</td>
<td>5</td>
</tr>
<tr>
<td>Manufacture of EPS/XPS</td>
<td>2</td>
</tr>
<tr>
<td>Manufacture of HIPS</td>
<td>0.3</td>
</tr>
<tr>
<td>Manufacture of Textiles</td>
<td>2.6</td>
</tr>
<tr>
<td>Use of goods containing HBCD</td>
<td>130000</td>
</tr>
<tr>
<td>Use of EPS/XPS</td>
<td>3500</td>
</tr>
<tr>
<td>Use of HIPS</td>
<td>50</td>
</tr>
<tr>
<td>Use of Textiles</td>
<td>130000</td>
</tr>
<tr>
<td>Disposal/recycling of goods containing HBCD (2010)</td>
<td>270</td>
</tr>
<tr>
<td>Disposal/recycling of goods containing HBCD (2030)</td>
<td>270</td>
</tr>
<tr>
<td>Disposal/recycling of EPS/XPS (2010)</td>
<td>1400</td>
</tr>
<tr>
<td>Disposal/recycling of EPS/XPS (2030)</td>
<td>3600</td>
</tr>
<tr>
<td>Disposal/recycling of HIPS</td>
<td>49</td>
</tr>
<tr>
<td>Disposal/recycling of Textiles</td>
<td>220</td>
</tr>
<tr>
<td>Total (2010)</td>
<td>130000</td>
</tr>
<tr>
<td>Total (2030)</td>
<td>130000</td>
</tr>
</tbody>
</table>

### 5.8 Scenario 4: exemption for use of EPS/XPS in cladded insulation

The potential impacts and benefits associated with this Scenario are the same as those described under Scenario 1 with the exception that use of EPS/XPS products containing HBCD for use in internal or external wall insulation would be permitted (i.e., non-cavity wall applications). As there are no values to indicate the proportion of treated EPS/XPS products which is used for this application, in order to illustrate potential impacts and benefits we have assumed that 50% of treated EPS/XPS manufacture and use is for this application.

#### 5.8.1 UK industry impacts and costs

**Construction materials Sector**

If the impacts of this Scenario are spread evenly across the sector then 300-900 jobs could remain at risk within the EPS/XPS manufacturing sector although it is uncertain how many jobs could be created by additional demand for the alternative products (i.e., for cavity wall retrofit applications). In fact the issue that EPS/XPS is only used in cavity wall applications when other materials cannot be used suggests that additional demand (i.e., 5% of the UK cavity wall retrofit market) could not be served by alternative producers. If manufacturers are...
more specialised in the internal/external cladding wall insulation market then the impacts will fall disproportionately on them. There are no data to suggest whether this is the case.

Textiles Sector

The impact on this sector would be the same as under Scenario 1. That is, neither alternative products nor alternative retardants are considered to be viable options in the short term. Consequently this regulation would end the manufacture or use of HBCD containing textiles in the UK. To maintain textile markets and fire regulations, such textiles would be imported from outside the EU with loss of consumer choice and loss of this sector from the UK economy.

Electronics Sector

The additional cost of production estimated for UK HIPS manufacturers would fall unevenly over UK manufacturers, according to their size. Costs would be the same as in Scenario 1.

Safe Disposal

An additional, unquantifiable number of direct and indirect jobs are likely to be created by the mandatory additional management and destruction or irreversible transformation of textiles, construction and demolition wastes. Regulatory, waste separation and incineration costs would arise and there would be a reduction of material going to landfill. These impacts would be the same as scenario 1 with the removal of 50% of EPS/XPS from the analysis.

The following table provides a summary of the estimated cost impact of Scenario 4.

**Table 18: Summary of estimated cost impact (Scenario 4)**

<table>
<thead>
<tr>
<th>Aspect</th>
<th>Cost element</th>
<th>Annual tonnage of material (kT)</th>
<th>Annualised cost (£M)</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td></td>
<td>Low</td>
<td>Mid</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Low</td>
<td>Mid</td>
</tr>
<tr>
<td></td>
<td>Loss of EPS market</td>
<td>4</td>
<td>13</td>
</tr>
<tr>
<td></td>
<td>Loss of XPS market</td>
<td>1.6</td>
<td>5.3</td>
</tr>
<tr>
<td></td>
<td>Increased Mineral wool market</td>
<td>3.1</td>
<td>8.6</td>
</tr>
<tr>
<td></td>
<td>Increased polyurethane market</td>
<td>1.3</td>
<td>3.6</td>
</tr>
<tr>
<td></td>
<td>Alternative HIPS market (other BFRs)</td>
<td>1.7</td>
<td>3.6</td>
</tr>
<tr>
<td></td>
<td>Alternative HIPS market (non-halogenated)</td>
<td>1.7</td>
<td>3.6</td>
</tr>
<tr>
<td></td>
<td>Loss of textile market (furniture)</td>
<td>140</td>
<td>280</td>
</tr>
<tr>
<td></td>
<td>Total (assuming non-halogenated HIPS)</td>
<td>90</td>
<td>246</td>
</tr>
<tr>
<td></td>
<td>Construction and demolition material</td>
<td>4,203</td>
<td>13,259</td>
</tr>
<tr>
<td></td>
<td>requiring separation</td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>EPS going to incineration</td>
<td>2.1</td>
<td>6.5</td>
</tr>
<tr>
<td></td>
<td>XPS going to incineration</td>
<td>0.8</td>
<td>2.7</td>
</tr>
<tr>
<td></td>
<td>EEE waste requiring separation</td>
<td>387</td>
<td>387</td>
</tr>
<tr>
<td></td>
<td>HIPS going to incineration</td>
<td>1.4</td>
<td>2.8</td>
</tr>
<tr>
<td></td>
<td>Textile products going to incineration</td>
<td>140</td>
<td>280</td>
</tr>
</tbody>
</table>
5.8.2 Benefits

The potential benefits from Scenario 4 are the same as those for Scenario 1 but with benefits associated with EPS/XPS manufacture, use or disposal reduced. This assessment has assumed that 50% of EPS/XPS use is in cavity insulation applications. Therefore this scenario would eventually reduce emissions by 480t annually. The following table details these emissions benefits.

Table 19: Annual average HBCD emission reductions (2010-2030) (Scenario 4)

<table>
<thead>
<tr>
<th>Source</th>
<th>Estimated annual emissions reduction (kg)</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Air (vapour)</td>
</tr>
<tr>
<td>Manufacture of goods containing HBCD</td>
<td>5</td>
</tr>
<tr>
<td>Manufacture of EPS/XPS</td>
<td>2</td>
</tr>
<tr>
<td>Manufacture of HIPS</td>
<td>0.3</td>
</tr>
<tr>
<td>Manufacture of Textiles</td>
<td>2.6</td>
</tr>
<tr>
<td>Use of goods containing HBCD</td>
<td>130000</td>
</tr>
<tr>
<td>Use of EPS/XPS</td>
<td>3500</td>
</tr>
<tr>
<td>Use of HIPS</td>
<td>50</td>
</tr>
<tr>
<td>Use of Textiles</td>
<td>130000</td>
</tr>
<tr>
<td>Disposal/recycling of goods containing HBCD (2010)</td>
<td>270</td>
</tr>
<tr>
<td>Disposal/recycling of goods containing HBCD (2030)</td>
<td>270</td>
</tr>
<tr>
<td>Disposal/recycling of EPS/XPS (2010)</td>
<td>1400</td>
</tr>
<tr>
<td>Disposal/recycling of EPS/XPS (2030)</td>
<td>3600</td>
</tr>
<tr>
<td>Disposal/recycling of HIPS</td>
<td>49</td>
</tr>
<tr>
<td>Disposal/recycling of Textiles</td>
<td>220</td>
</tr>
<tr>
<td>Total (2010)</td>
<td>130000</td>
</tr>
<tr>
<td>Total (2030)</td>
<td>130000</td>
</tr>
</tbody>
</table>
5.9 Scenario 5: safe disposal of materials containing HBCD

This Scenario differs from those examined thus far in that it is focused solely on the destruction or irreversible transformation of HBCD containing materials. That is, it aims to prevent HBCD from entering the environment via waste streams rather than address HBCD emitted during manufacture and use of HBCD containing products.

5.9.1 Costs

Safe Disposal

An additional, unquantifiable number of direct and indirect jobs are likely to be created by the mandatory additional management and destruction or irreversible transformation of textiles, construction and demolition wastes. Regulatory, waste separation and incineration costs would arise and there would be a reduction of material going to landfill.

The following table provides a summary of the estimated cost impact of Scenario 5.

(Table 20: Summary of estimated cost impact (Scenario 5))

<table>
<thead>
<tr>
<th>Aspect</th>
<th>Cost element</th>
<th>Annual tonnage of material (kT)</th>
<th>Annualised cost (£M)</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td></td>
<td>Low</td>
<td>Mid</td>
</tr>
<tr>
<td>Safe disposal costs</td>
<td>Construction and demolition material requiring separation</td>
<td>4,203</td>
<td>13,259</td>
</tr>
<tr>
<td></td>
<td>EPS going to incineration</td>
<td>4.3</td>
<td>13</td>
</tr>
<tr>
<td></td>
<td>XPS going to incineration</td>
<td>1.6</td>
<td>5.5</td>
</tr>
<tr>
<td></td>
<td>EEE waste requiring separation</td>
<td>387</td>
<td>387</td>
</tr>
<tr>
<td></td>
<td>HIPS going to incineration</td>
<td>1.4</td>
<td>2.8</td>
</tr>
<tr>
<td></td>
<td>Textile products going to incineration</td>
<td>140</td>
<td>280</td>
</tr>
<tr>
<td></td>
<td>Material diverted from landfill</td>
<td>68</td>
<td>81</td>
</tr>
<tr>
<td></td>
<td>Regulatory costs</td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>Total</td>
<td>4,805</td>
<td>14,028</td>
</tr>
</tbody>
</table>

As would be expected, processing and separation of demolition materials containing HBCD is the principal cost element of Scenario 5.

5.9.2 Benefits

The potential benefits from Scenario 5 are different from all scenarios previously discussed. The scenario would have no impact on emissions associated with manufacture or use of materials containing HBCD. However the regulations could potentially reduce emissions associated with disposal and recycling (we have assumed that they would be eliminated) from 2010 onwards. This scenario is therefore the most effective one at managing HBCD emissions associated with existing materials yet to enter the waste stream. This scenario would eventually reduce emissions by 490t annually. The following table details these emissions benefits.
Table 21: Annual average HBCD emission reductions (2010-2030) (Scenario 5)

<table>
<thead>
<tr>
<th>Source</th>
<th>Estimated annual emissions reduction (kg)</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Air (vapour)</td>
</tr>
<tr>
<td>Manufacture of goods containing HBCD</td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
</tr>
<tr>
<td>Manufacture of EPS/XPS</td>
<td></td>
</tr>
<tr>
<td>Manufacture of HIPS</td>
<td></td>
</tr>
<tr>
<td>Manufacture of Textiles</td>
<td></td>
</tr>
<tr>
<td>Use of goods containing HBCD</td>
<td></td>
</tr>
<tr>
<td>Use of EPS/XPS</td>
<td></td>
</tr>
<tr>
<td>Use of HIPS</td>
<td></td>
</tr>
<tr>
<td>Use of Textiles</td>
<td></td>
</tr>
<tr>
<td>Disposal/recycling of goods containing HBCD (2010)</td>
<td>270</td>
</tr>
<tr>
<td>Disposal/recycling of goods containing HBCD (2030)</td>
<td>270</td>
</tr>
<tr>
<td>Disposal/recycling of EPS/XPS (2010)</td>
<td>2800</td>
</tr>
<tr>
<td>Disposal/recycling of EPS/XPS (2030)</td>
<td>7200</td>
</tr>
<tr>
<td>Disposal/recycling of HIPS</td>
<td>49</td>
</tr>
<tr>
<td>Disposal/recycling of Textiles</td>
<td>220</td>
</tr>
<tr>
<td>Total (2010)</td>
<td>270</td>
</tr>
<tr>
<td>Total (2030)</td>
<td>2700</td>
</tr>
</tbody>
</table>

5.10 Summary

The key issues identified in the cost-benefit assessment are as follows:
### Table 22: Summary of findings of cost-benefit assessment

<table>
<thead>
<tr>
<th>Scenario</th>
<th>Environmental benefits</th>
<th>Cost and economic issues</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Scenario 1: Complete ban</strong></td>
<td>Reduction in HBCD (2030): 730 tonnes/yr</td>
<td>Half from disposal/recycling of EPS/XPS, balance mainly from use of EPS/XPS and from use of textiles (emission to air) and disposal/recycling of textiles</td>
</tr>
<tr>
<td><strong>Scenario 2: Derogation for textiles</strong></td>
<td>Reduction in HBCD (2030): 495 tonnes/yr</td>
<td>70% from disposal/recycling of EPS/XPS; balance mainly from use of EPS/XPS</td>
</tr>
<tr>
<td><strong>Scenario 3: Derogation for cavity insulation</strong></td>
<td>Reduction in HBCD (2030): 480 tonnes/yr</td>
<td>40% from disposal/recycling of EPS/XPS, balance mainly from use of textiles (emission to air) and disposal/recycling of textiles</td>
</tr>
<tr>
<td><strong>Scenario 4: Derogation for cladding insulation</strong></td>
<td>Reduction in HBCD (2030): 480 tonnes/yr</td>
<td>40% from disposal/recycling of EPS/XPS, balance mainly from use of textiles (emission to air) and disposal/recycling of textiles</td>
</tr>
<tr>
<td><strong>Scenario 5: Safe disposal</strong></td>
<td>Reduction in HBCD (2030): 490 tonnes/yr</td>
<td>70% from disposal/recycling of EPS/XPS, balance mainly from disposal/recycling of textiles</td>
</tr>
</tbody>
</table>
6 Annex I Questionnaire on HBCD

The completed questionnaire is provided in Appendix 1.

The questionnaire refers to both monetarised and non-monetarised benefits of withdrawal of HBCD. As envisaged at the outset of the project, the benefits of withdrawal of HBCD cannot be quantified at present, and these issues are summarised as non-monetarised benefits.
7 Conclusions

7.1 Evaluation of HBCD against Stockholm Convention criteria

HBCD was assessed against the Stockholm Convention criteria for persistence, bioaccumulation, long-range transport and health/environmental effects, in order to reach a view on whether listing under the Stockholm Criteria should be supported by Defra.

- Persistence: On balance, HBCD meets the criteria of the Stockholm Convention with regard to persistence
- Bio-accumulation: HBCD meets the criteria of the Stockholm Convention with regard to bio-accumulation
- Long-range transport: HBCD meets the criteria of the Stockholm Convention with regard to long-range transport
- Adverse effects: HBCD does not meet the criteria of the Stockholm Convention with regard to the potential for damage to human health. It is uncertain whether HBCD meets the criteria of the Stockholm Convention with regard to the potential for damage to the environment

It is concluded that HBCD does not demonstrably meet the criteria set in the Convention. However, if a precautionary approach is adopted with regard to the potential for environmental harm, this would suggest that HBCD could provisionally be considered a Persistent Organic Pollutant under the terms of the Stockholm Convention, pending further research to confirm the range of exposure of sensitive species and the potential adverse effects of HBCD stereo-isomers.

7.2 Sources, uses and controls of HBCD in the UK

It is estimated that approximately 1,000 tonnes of HBCD are used per year in the UK. About 90% of HBCD is used in the construction industry, with the balance used in the textiles and electronics industries.

Releases of HBCD to the environment were estimated for 2010 and for 2030. The most significant environmental releases of HBCD were estimated to be as follows:

- Disposal to land of polystyrene insulation board is forecast to become increasingly significant up to 2030. In 2030, disposal to land of polystyrene insulation board is estimated to account for 49% of all HBCD releases
- The next highest estimated release is emissions to air of HBCD from textiles, estimated to account for 18% of all HBCD releases in 2030.
- Disposal to landfill of waste polystyrene insulation board during installation is estimated to account for 13% of all HBCD releases in 2030.
- Disposal of end-of-life textiles containing HBCD to landfill is estimated to account for 12% of all HBCD releases in 2030.

At present, there are no specific controls on HBCD during the manufacture, use and disposal of goods containing HBCD in the UK. Controls may be forthcoming under the terms of the Long-Range Transboundary Air Pollution (LRTAP) Convention. A number of regimes could potentially be used to introduce controls on HBCD, if required:
• The Environmental Permitting regime, covering the disposal of waste materials which could contain HBCD

• The Waste Electrical and Electronic Equipment (WEEE) Regulations, covering the recovery and recycling of plastics from electrical and electronic goods containing HBCD

• The REACH regulations provide the framework for authorisation of chemicals throughout Europe. HBCD is under consideration for restriction under the terms of REACH.

• The Restriction of Hazardous Substances directive prevents equipment containing hazardous chemicals from being marketed in the EU. HBCD could potentially be included in the scope of this directive.

• Existing voluntary environmental management programmes in the key industries could potentially be applied to the control of HBCD.

7.3 Cost-benefit analysis

Five scenarios were identified for potential approaches to control of HBCD under the Stockholm Convention:

• Scenario 1: Complete ban on HBCD use in the UK, with safe disposal requirements

• Scenario 2: Restriction on HBCD use in the UK, with safe disposal requirements and an exemption for critical uses in the textiles industry

• Scenario 3: Restriction on HBCD use in the UK, with safe disposal requirements and an exemption for the use of HBCD in building cavity insulation (this application is currently declining)

• Scenario 4: Restriction on HBCD use in the UK, with safe disposal requirements and an exemption for the use of HBCD in internal and external cladding insulation (this application is currently growing)

• Scenario 5: No restriction on HBCD use in the UK, but an introduction of safe disposal requirements

The costs and benefits of each scenario were assessed for the key sectors in which HBCD is used at present. All five scenarios would result in reduced emissions of HBCD, with consequential environmental benefits. Scenario 1 would give rise to the greatest benefit in terms of reduced emissions of HBCD, while Scenarios 3 and 4 would give rise to the least benefit. Scenarios 1 to 5 would all result in potentially significant effects on employment, competitiveness and consumer choice.

• Scenario 1 (Complete ban and safe disposal requirements): this is estimated to result in a reduction in emissions of HBCD of 730 tonnes per year. This scenario would result in increased costs in separating materials containing HBCD from construction waste; reduction of employment in insulation manufacture; loss of employment in textiles industry; increased cost of incineration of textile products; negative impact on competitiveness of electronics industry; and reduced building energy efficiency

• Scenario 2 (Ban on most uses and safe disposal requirements; derogation for textiles): this is estimated to result in a reduction in emissions of HBCD of 495 tonnes per year. This scenario would result in increased costs in separating materials containing HBCD from construction waste; reduction of employment in insulation manufacture; loss of employment in textiles industry; increased cost of incineration of textile products; negative impact on competitiveness of electronics industry; and reduced building energy efficiency
manufacture; increased cost of separation of electrical/electronic waste; negative impact on competitiveness of electronics industry; and reduced building energy efficiency

- Scenario 3 (Ban on most uses and safe disposal requirements; derogation for cavity insulation): this is estimated to result in a reduction in emissions of HBCD of 480 tonnes per year. This scenario would result in increased costs in separating materials containing HBCD from construction waste; smaller reduction of employment in insulation manufacture; loss of employment in textiles industry; increased cost of separation of electrical/electronic waste; increased cost of incineration of textile products; reduction in consumer choice for textile products; negative impact on competitiveness of electronics industry; and smaller reduction in building energy efficiency

- Scenario 4 (Ban on most uses and safe disposal requirements; derogation for cladding insulation): this is estimated to result in a reduction in emissions of HBCD of 480 tonnes per year. This scenario would result in increased costs in separating materials containing HBCD from construction waste; smaller reduction of employment in insulation manufacture; loss of employment in textiles industry; increased cost of separation of electrical/electronic waste; increased cost of incineration of textile products; reduction in consumer choice for textile products; negative impact on competitiveness of electronics industry; and smaller reduction in building energy efficiency

- Scenario 5 (Safe disposal requirements only): this is estimated to result in a reduction in emissions of HBCD of 490 tonnes per year. This scenario would result in increased costs in separating materials containing HBCD from construction waste; increased cost of separation of electrical/electronic waste; increased cost of incineration of textile products; increase in employment in waste/recycling industry, and an increase in carbon footprint of plastics industry unless appropriate market conditions are facilitated
8 References


British Plastics Federation (2005), market analysis research carried out by AMA Research, reported to AEA Technology in discussions with BPF, August 2010


Carpet Federation (2010), Communication to AEA Technology

CEFIC, Industry Comments on HBCD nomination dossier, 2010


Danish Environment Ministry (2010), “Inclusion of HBCDD, DEHP, BBP, DBP and additive use of TBBPA in annex IV of the Commission’s recast proposal of the RoHS Directive,” Environmental Project No. 1317


Environ (2010). B2B WEEE Scheme membership fees (see www.b2bweeessceme.com/services/fees)


European Phenolic Foam association (2010), Communication to AEA Technology


Graves WC and Swigert JP 1997, “Hexabromocyclododecane (HBCD): A 96-hour flowthrough acute toxicity test with the rainbow trout (Oncorhynchus mykiss),” Final report. 439a-101. Wildlife International Ltd., Easton, Maryland, USA. p. 70

Harrad S, Abdallah MA, Covaci A, 2009a “Causes of variability in concentrations and
diastereomer patterns of Hexabromocyclododecanes in indoor dust”, Environment
International, 35, 573–579

Harrad S, Abdallah MA, Rose NL, Turner SD, Davidson TA “Correction/Addition to “Current-
Use Brominated Flame Retardants in Water, Sediment, and Fish from English Lakes””,
Environmental Science and Technology, 44, 5318 (2010a).

Harrad S, Abdallah MA, Rose NL, Turner SD, Davidson TA, 2009b “Current-Use Brominated
Flame Retardants in Water, Sediment, and Fish from English Lakes”, Environmental Science
and Technology, 43, 9077–9083

environment with HBCD and its degradation products”, presented at Dioxin 2010 conference.

Heeb NV, Schweizer WB, Kohler M and Gerecke AC, 2005 “Structure elucidation of
hexabromocyclododecanes - a class of compounds with a complex stereochemistry,”

Hinkson NC, Whalen MM 2010, “Hexabromocyclododecane decreases tumor-cell-binding
capacity and cell-surface protein expression of human natural killer cells.” J Appl Toxicol.

paper presented at Brominated Flame Retardants conference 2007

Hirai Y, Eguchi T and Sakai S (2010), “Development of Environmental Fate Model for
Hexabromocyclododecanes (HBCDs) with Isomerisation Process”, paper presented at
Brominated Flame Retardants conference 2010

Horrocks, R, University of Central Lancashire, (2010), Communication to AEA Technology

House of Lords (2008), Science and Technology Committee, 6th Report of Session 2007–08,
“Waste Reduction”

Johnson-Restrepoa B, Adams, DH, Kannana K. Tetrabromobisphenol A (TBBPA) and
hexabromocyclododecanes (HBCDs) in tissues of humans, dolphins, and sharks from the
United States Chemosphere 2007

KEMI (Swedish Chemicals Inspectorate) (2006a), “Survey and technical assessment of
alternatives to TBBPA and HBCDD,” Ref. PM1/06

KEMI (Swedish Chemicals Inspectorate) (2006b), “Hexabromocyklododekan (HBCDD) och
tetrabrombifsenol - A (TBBPA)” Report Ref. 03/2006 (Swedish language only)

Electrical and Electronic Equipment”, report to Danish Ministry of the Environment.
Environmental Project No. 1141 2006

Knutsen HK, Kvalem HE, Thomsen C, Frøshaug M, Haugen M, Becher G, Alexander J,
Meltzer HM. Dietary exposure to brominated flame retardants correlates with male blood
levels in a selected group of Norwegians with a wide range of seafood consumption. Mol

Köppen, R., Becker, R., Jung, C., Nehls, I., 2008. On the thermally induced isomerisation of

Kurume Laboratory (1990), “Biodegradation Study of Hexabromocyclododecane,” study
sponsored by Japan Ministry of International Trade and Industry,

Downturn in Levels of Hexabromocyclododecane in the Blubber of Harbor Porpoises
(Phocoena phocoena) Stranded or Bycaught in the UK: An Update to 2006,” Environ. Sci.
Cost Benefits study of Hexabromocyclododecane


Ministry of the Environment, Japan (2009), “6-Week Administration Study of 1,2,5,6,9,10-Hexabromocyclododecane for Avian Reproduction Toxicity under Long-day Conditions using Japanese Quail,” report produced by Research Institute for Animal Science in Biochemistry & Toxicology

National Federation of Demolition Contractors (2010), Communication to AEA Technology

Norwegian Climate and Pollution Agency 2010, “Hexabromocyclododecane in the UNECE region,” Powerpoint presentation to the UNECE CLRTAP Task Force POPs Expert Group

Norwegian Climate and Pollution Agency, “Exploration of Management Options for Hexabromocyclododecane (HBCD),” (updated version) 18 August 2010


UNEP, Stockholm Convention on POPs, “Additional information submitted on hexabromocyclododecane,” September 2009
Cost Benefits study of Hexabromocyclododecane

UNEP, Stockholm Convention on POPs, “Hexabromocyclododecane: draft risk profile,” April 2010


Appendices

Appendix 1  Annex I Questionnaire on production and use of HBCD
Appendix 2  Stockholm Convention Annex D
Appendix 3  Measured levels of HBCD isomers and degradation products in the UK
Appendix 4  Health effects of HBCD
Appendix 1 – Annex I Questionnaire on production and use of HBCD

[As this is a protected document, page breaks could not be corrected]
Appendix 2 – Stockholm Convention Annex D

1. A Party submitting a proposal to list a chemical in Annexes A, B and/or C shall identify the chemical in the manner described in subparagraph (a) and provide the information on the chemical, and its transformation products where relevant, relating to the screening criteria set out in subparagraphs (b) to (e):

   (e) Chemical identity:
   (i) Names, including trade name or names, commercial name or names and synonyms, Chemical Abstracts Service (CAS) Registry number, International Union of Pure and Applied Chemistry (IUPAC) name; and
   (ii) Structure, including specification of isomers, where applicable, and the structure of the chemical class;

   (f) Persistence:
   (i) Evidence that the half-life of the chemical in water is greater than two months, or that its half-life in soil is greater than six months, or that its halflife in sediment is greater than six months; or
   (ii) Evidence that the chemical is otherwise sufficiently persistent to justify its consideration within the scope of this Convention;

   (g) Bio-accumulation:
   (i) Evidence that the bio-concentration factor or bio-accumulation factor in aquatic species for the chemical is greater than 5,000 or, in the absence of such data, that the log Kow is greater than 5;
   (ii) Evidence that a chemical presents other reasons for concern, such as high bio-accumulation in other species, high toxicity or ecotoxicity; or
   (iii) Monitoring data in biota indicating that the bio-accumulation potential of the chemical is sufficient to justify its consideration within the scope of this Convention;

   (h) Potential for long-range environmental transport:
   (i) Measured levels of the chemical in locations distant from the sources of its release that are of potential concern;
   (ii) Monitoring data showing that long-range environmental transport of the chemical, with the potential for transfer to a receiving environment, may have occurred via air, water or migratory species; or
   (iii) Environmental fate properties and/or model results that demonstrate that the chemical has a potential for long-range environmental transport through air, water or migratory species, with the potential for transfer to a receiving environment in locations distant from the sources of its release. For a chemical that migrates significantly through the air, its half-life in air should be greater than two days; and

   (i) Adverse effects:
   (i) Evidence of adverse effects to human health or to the environment that justifies consideration of the chemical within the scope of this Convention; or
   (ii) Toxicity or ecotoxicity data that indicate the potential for damage to human health or to the environment.
2. The proposing Party shall provide a statement of the reasons for concern including, where possible, a comparison of toxicity or ecotoxicity data with detected or predicted levels of a chemical resulting or anticipated from its long-range environmental transport, and a short statement indicating the need for global control.

3. The proposing Party shall, to the extent possible and taking into account its capabilities, provide additional information to support the review of the proposal referred to in paragraph 6 of Article 8. In developing such a proposal, a Party may draw on technical expertise from any source.
# Appendix 3 – Measured levels of HBCD isomers and degradation products in the UK

Table A3.1: Summary of Concentrations, Chiral Signatures (expressed as enantiomer fractions (EFs) - given in parentheses) of HBCDs, PBCDs, and TBCDs in the UK Environment

<table>
<thead>
<tr>
<th>Matrix</th>
<th>Parameter</th>
<th>α-HBCD</th>
<th>β-HBCD</th>
<th>γ-HBCD</th>
<th>ΣHBCDs</th>
<th>ΣPBCDs</th>
<th>ΣTBCDs</th>
</tr>
</thead>
<tbody>
<tr>
<td>Indoor Air (Homes) (n=45) pg m⁻³</td>
<td>Median</td>
<td>37</td>
<td>22</td>
<td>120</td>
<td>180</td>
<td>nm</td>
<td>nm</td>
</tr>
<tr>
<td></td>
<td>Range</td>
<td>14-430</td>
<td>5-54</td>
<td>39-710</td>
<td>67-1300</td>
<td>nm</td>
<td>nm</td>
</tr>
<tr>
<td>Outdoor Air (n=12) pg m⁻³</td>
<td>Median</td>
<td>12</td>
<td>2.5</td>
<td>33</td>
<td>47</td>
<td>nm</td>
<td>nm</td>
</tr>
<tr>
<td></td>
<td>Range</td>
<td>2.3-48</td>
<td>0.9-22</td>
<td>9-59</td>
<td>34-130</td>
<td>nm</td>
<td>nm</td>
</tr>
<tr>
<td>E-waste facility indoor air (n=2) ng m⁻³</td>
<td>Average</td>
<td>11</td>
<td>3.0</td>
<td>21</td>
<td>35</td>
<td>nm</td>
<td>nm</td>
</tr>
<tr>
<td>E-waste facility outdoor air (n=1) ng m⁻³</td>
<td>-</td>
<td>6</td>
<td>2</td>
<td>14</td>
<td>22</td>
<td>nm</td>
<td>nm</td>
</tr>
<tr>
<td>Indoor Dust (Homes) (n=45, n=21 for PBCDs and TBCDs)</td>
<td>Median</td>
<td>380</td>
<td>93</td>
<td>670</td>
<td>1300</td>
<td>81</td>
<td>12</td>
</tr>
<tr>
<td></td>
<td>Range</td>
<td>22-66000</td>
<td>9-26000</td>
<td>70-75000</td>
<td>140-140000</td>
<td>11-6400</td>
<td>1.0-51</td>
</tr>
<tr>
<td>E-waste facility indoor dust (n=2) ng g⁻¹</td>
<td>Average</td>
<td>16000</td>
<td>2200</td>
<td>14000</td>
<td>33000</td>
<td>nm</td>
<td>nm</td>
</tr>
<tr>
<td>Lake water (n=27) pg L⁻¹</td>
<td>Median</td>
<td>35</td>
<td>21</td>
<td>77</td>
<td>120</td>
<td>nm</td>
<td>Nm</td>
</tr>
<tr>
<td></td>
<td>Range</td>
<td>19-63</td>
<td>10-31</td>
<td>44-210</td>
<td>80-270</td>
<td>nm</td>
<td>Nm</td>
</tr>
<tr>
<td>Topsoil (n=24) ng g⁻¹ dw</td>
<td>Median</td>
<td>0.25 (0.50)</td>
<td>0.10 (0.50)</td>
<td>0.52 (0.50)</td>
<td>0.77</td>
<td>&lt;0.01</td>
<td>&lt;0.01</td>
</tr>
<tr>
<td></td>
<td>Range</td>
<td>0.07-59 (0.47-0.54)</td>
<td>0.02-28 (0.46-0.56)</td>
<td>0.07-337 (0.47-0.53)</td>
<td>0.07-424</td>
<td>&lt;0.01-7.3</td>
<td>&lt;0.01-1.3</td>
</tr>
<tr>
<td>Lacustrine Sediment (n=9) ng g⁻¹ dw</td>
<td>Median</td>
<td>0.23 (0.49)</td>
<td>0.19 (0.51)</td>
<td>1.4 (0.48)</td>
<td>1.7</td>
<td>0.037</td>
<td>0.27</td>
</tr>
<tr>
<td></td>
<td>Range</td>
<td>0.11-0.62 (0.47-0.53)</td>
<td>0.06-0.50 (0.47-0.52)</td>
<td>0.66-3.8 (0.47-0.53)</td>
<td>0.88-4.8</td>
<td>&lt;0.02-0.22</td>
<td>0.07-0.81</td>
</tr>
<tr>
<td>Freshwater fish (n=30) ng g⁻¹ lw</td>
<td>Median</td>
<td>57 (0.45)</td>
<td>7.4 (0.51)</td>
<td>28 (0.53)</td>
<td>92</td>
<td>nm</td>
<td>Nm</td>
</tr>
<tr>
<td></td>
<td>Range</td>
<td>7.8-250 (0.42-0.47)</td>
<td>1.8-30 (0.45-0.52)</td>
<td>3.7-94 (0.49-0.56)</td>
<td>14-290</td>
<td>nm</td>
<td>Nm</td>
</tr>
<tr>
<td>Human milk (n=28) ng g⁻¹ lw</td>
<td>Median</td>
<td>3.2 (0.28)</td>
<td>0.30 (0.50)</td>
<td>0.56 (0.52)</td>
<td>3.8</td>
<td>&lt;0.03</td>
<td>0.14</td>
</tr>
<tr>
<td></td>
<td>Range</td>
<td>0.75-20 (0.15-0.49)</td>
<td>0.08-0.75 (0.46-0.54)</td>
<td>0.13-2.3 (0.45-0.61)</td>
<td>1.0-22</td>
<td>0.03-0.20</td>
<td>&lt;0.03-0.36</td>
</tr>
</tbody>
</table>

nm = not measured
Appendix 4 – Health effects of HBCD

A4.1 INTRODUCTION

This report reviews the potential health impacts that may be associated with exposure to HBCD arising from its use in consumer products and construction materials and its presence in the wider environment. Consideration is also given to the relative harmfulness of different HBCD isomers.

The three main sections of the report summarise information about the toxicokinetics of HBCD (i.e., uptake and absorption following exposure by different routes, metabolism, elimination and potential for long term accumulation in the body), the toxic effects of HBCD reported in animal experiments and levels of human exposure to HBCD. The final section of the report compares current levels of exposure to HBCD with those associated with adverse effects in animals.

A4.2 TOXICOKINETICS

Uptake and absorption

HBCD appears to be relatively readily absorbed following ingestion. The results of a study in adult female C57BL/6 mice suggest that 85% of the administered dose (3mg/kg) was absorbed after oral exposure (Szabo et al, 2010). Disposition was dose-independent and didn't significantly change after 10 days of exposure. There is no information about absorption of inhaled HBCD but it is likely to be as least as efficient as following ingestion. It is possible that the uptake of HBCD from respirable dust in the lungs would even greater than from ingested HBCD. There is also no information about the uptake of HBCD through the skin which would be expected to be very much less than that following ingestion. The EU Risk Assessment assumed 100% absorption following exposure by ingestion or inhalation and 4% absorption following dermal contact. In animal experiments, brominated fire retardants are normally administered in corn oil in order to maximise absorption. It is likely that adsorption of HBCD from other matrices is less efficient. Based on the results from an \textit{in vitro} test system, Abdallah et al (2009) reported that the oral bioaccessibility of gamma HBCD in house dust was 72% whereas that of alpha and beta HBCD was 92% and 81% respectively.

Distribution

Szabo et al (2010) reported that the highest burdens of HBCD in mice, 4 days after oral exposure to HBCD were found in the liver (<0.3% of dose) treatment followed by blood, fat and then brain. After intraperitoneal injection in rats, significant quantities of HBCD were found in brain and liver (Reistad et al, 2006). Tissue concentrations of HBCD were about 10 times greater in the liver than in the brain.

Metabolism and elimination

1,2,5,6,9,10-Hexabromocyclododecane-gamma (gamma-HBCD) is the dominant isomer in commercial HBCD and environmental samples, whereas alpha-HBCD predominates human tissue and wildlife. The reported proportion of alpha-HBCD in the serum of a group of Belgian adults was much greater than in dust or food, suggesting selective metabolism and excretion of gamma-HBCD (Roosens et al, 2009).

In animals, Szabo et al (2010) reported that gamma-HBCD was rapidly metabolized following oral administration to mice and the metabolites eliminated in the urine and faeces. The gamma-diastereoisomer was transformed to the beta-diastereoisomer in liver and brain tissues, and to the alpha- and beta-diastereoisomer in fat and faeces. Polar metabolites in the blood and urine were a major factor in determining the initial whole-body half-life after a single oral exposure. Initial half-lives for both the whole body and for individual tissues were approximately 1 day, whereas terminal half-lives were up to 4 days.
In a study of the hydroxylated metabolites of HBCD, Brandsma et al (2009) exposed rats to 30 and 100 mg HBCD/kg/day for 28 days. Five different groups of hydroxylated HBCD metabolites were identified in rat adipose, liver, lung, and muscle: monohydroxy metabolites of HBCD, pentabromocyclododecene (PBCDe), tetrabromocyclododecene (TBCDe), dihydroxy-HBCD and dihydroxy-PBCDe. Debromination of HBCD to PBCDe was also reported.

HBCD is reported to induce cytochrome P450 (CYP) 3 enzymes in the liver. In rat liver cells, HBCD induces CYP3A1 but to a lesser extent than the prototype inducer dexamethasone whereas in human liver cells, its effects were similar to the prototype inducer rifampicin (Fery et al, 2009). In contrast, HBCD failed to induce luciferin-PFBE dealkylase in human liver cells, a common catalytic activity of a number of CYP3A enzymes, possibly reflecting enzyme inhibition. Gerner et al (2006) investigated the effects of HBCD on rat liver cytochrome P450 (CYP) levels following exposure of juvenile/young rats to HBCD by gavage over a 28 day period. HBCD treatment led to a significant induction of CYP2B1 mRNA, CYP2B1/2B2 protein and 7-pentoxyresorufin O-depentylylase (PROD) activity suggesting a phenobarbital-type of induction. A significant increase in CYP3A1/3A3 mRNA, CYP3A1 protein, and luciferin benzylether debenzyllase (LBD) activity was also found, particularly in females. The effect on CYP3A1/3A3 mRNA was significant in female rats at a daily dose of 3.0mg/kg body weight and above.

### A4.3 TOXIC EFFECTS

#### Acute toxicity

The BFRIP (2001) data summary indicates that HBCD has a low acute toxicity in rats and in rabbits following exposure by ingestion, skin contact or inhalation. The rat oral median lethal dose (LD\(_{50}\)) was >10 g/kg, the rabbit dermal LD\(_{50}\) was >8g/kg and the median lethal concentration (LC\(_{50}\)) in rats was >200 000 mg/m\(^3\).

#### Repeated dose toxicity

The BFRIP (2001) data summary indicates that the no observed adverse effects level (NOAEL) was about 1000 mg/kg/day in rats following oral administration in two 28 and two 90 day studies. In one of the 90 day studies, some mild reversible effects on clinical chemistry were observed at 100 and 300 mg/kg/day that were believed to be secondary to the induction of liver enzymes (considered an adaptive rather than toxic change). There was no associated impact on the health of the animals. Additional effects observed at 1000 mg/kg/day included increased size of liver and prostate. These effects were reversible on cessation of exposure and not associated with any organ damage or changes in organ function. These changes were therefore considered of limited, if any toxicological significance.

In one of the 28 day studies in which HBCD was administered in the diet at doses equivalent to 0, 940, 2410 and 48 mg/kg/day, liver weights were increased and thyroid hyperplasia was observed in some animals at all doses. There was also a “a very slight numerical development of the follicles and ripening follicles in the ovaries of females” at the high dose. The increase in liver weight was attributed to increased thyroid activity.

The SHVC documentation reviewed the studies presented by BFRIP and a more recent study that had investigated lower levels of exposure to HBCD and potentially more sensitive markers of effect (measures of effects on thyroid activity and immune parameters; Van der Ven, 2006). The LOEL for liver weight increase in rats which was seen consistently across several studies was assessed to be ≤100 mg/kg/day. The SHVC documentation also indicates that HCB appears to interfere with thyroid function in rats, although noted substantial inconsistencies between the findings of different studies. Two studies reported no histological effects, one study reported thyroid hyperplasia at 940 mg/kg/day in both sexes, another study reported increased thyroid weight in females only at 300 mg/kg/day but decreased serum T4 and increased TSI in both sexes at all doses (≥100 mg/kg/day) and a fifth study reported increased thyroid weight and decreased serum T4 in females only with
the lower confidence bounds on the estimated doses associated with a 10% incidence (BMDL) being 2 mg/kg/day and 55 mg/kg/day respectively (Van der Ven et al, 2006).

In the Van der Ven et al (2006) study, rats were exposed by daily gavage to HBCD dissolved in corn oil in 8 doses of 0 to 200 mg/kg/day. A variety of effects were seen in females but not males at relatively low levels of exposure (see Table below) and the authors determined the overall BMDL as 1.6 mg/kg/day, based on a 10% increase of the thyroid weight in females, which was the most sensitive parameter in the sequence of events. The only effect reported in males was a reduction in the number of splenocytes that might reflect potential effects on immune function. The lower bound on the estimated dose associated with an incidence of 20% was 0.3-6.3 mg/kg/day (effect only evaluated in males).

<table>
<thead>
<tr>
<th>Effects (females only)</th>
<th>Lower bound on estimated dose (mg/kg/day) with incidence of 10%*</th>
<th>Lower bound on estimated dose (mg/kg/day) with incidence of 20%</th>
</tr>
</thead>
<tbody>
<tr>
<td>Decreased total thyroxin</td>
<td>55.5</td>
<td></td>
</tr>
<tr>
<td>Increased pituitary weight</td>
<td>29</td>
<td></td>
</tr>
<tr>
<td>Increased immunostaining of thyroid stimulated hormone in the pituitary, increased thyroid weight</td>
<td>1.6</td>
<td></td>
</tr>
<tr>
<td>Increase in absolute liver weights</td>
<td></td>
<td>22.9</td>
</tr>
<tr>
<td>Induction of T4-glucuronyl</td>
<td>4.1</td>
<td></td>
</tr>
<tr>
<td>Increased cholesterol</td>
<td>7.4</td>
<td></td>
</tr>
<tr>
<td>Increased tibial bone mineral density</td>
<td>&gt;49</td>
<td></td>
</tr>
</tbody>
</table>

*Effectively BMDL

SCHER (2008) noted that as HBCD has the potential to bioaccumulate, the true threshold of effect for long term exposure may be lower than that observed in 28 or 90 day experiments.

**Mutagenicity**

HBCD gave negative results in the Ames, in vivo mouse micronucleus and in vitro chromosome aberration tests (EPA 2010 data summary).

**Carcinogenicity**

The BFRIP (2001) data summary indicates that no evidence of carcinogenicity was found in an 18 month study in which rats were fed diets containing HBCD at concentrations up to 10,000 ppm. Liver lesions were found at all dose levels but were not correlated with dose. The study was not conducted to current guidelines.

**Developmental toxicity**

The BFRIP (2001) data summary reviews two developmental studies. No adverse effects on foetal development or maternal toxicity was observed when pregnant rats were exposed to doses of HBCD of up to 1000 mg/kg/day from gestation day 6 to 19. In a separate study in which pregnant rats were exposed to HBCD in the diet at doses of 500 mg/day from days 1 to 20 of pregnancy, no adverse effects were seen on foetal development, maternal health or neonatal development (following up to 7 weeks following delivery).

The SVHC documentation indicates that a more two recent generation study found a dose-dependent decrease 8-14% in fertility index in both generations. There was also a significantly reduced number of primordial follicles in the mid and high dose groups and a high, dose-dependent pup mortality in the F2 generation. A NOAEL of 2 mg/kg/day was deduced. The Annex 15 dossier also considered a study by Eriksson et al (2006) that showed that neonatal HBCD may cause developmental neurotoxicity characterised by
changes in spontaneous behaviour, learning and memory deficits. A LOAEL of 0.9 mg/kg/day was reported but the Annex 15 dossier considered that the results needed confirmation by findings in other laboratories.

Saegusa et al (2009) exposed pregnant rats to gamma HBCD, at doses of 100, 1000 or 10,000 ppm in a soy-free diet from gestation day 10 until the day 20 after delivery. The offspring of the high dose group exhibited mild hypothyroidism with increases in thyroid weight, thyroid follicular cell hypertrophy and serum concentrations of thyroid-stimulating hormone as well as decreases of serum T(3) at weaning. Increased thyroid weights and decreased serum T(3) concentrations were also observed in the offspring of the mid dose group. The offspring of the high dose group also showed evidence of effects on development of the brain in the form of a reduced density of CNPase-positive oligodendrocytes suggesting impaired oligodendroglial development. The no effects level for changes in thyroid parameters at the adult stage following maternal exposure was determined as 100 ppm in the diet equating to a dose of 8.1-21.3 mg/kg/day.

In a 2 generation study, Ema et al (2009) exposed rats to HBCD at concentrations of 0, 150, 1500 or 15,000 ppm in the diet from the onset of a 10-week pre-mating period and continuing through the mating, gestation and lactation periods for two generations. The mean daily intakes of HBCD during the whole period of administration were 10.2, 101 and 1008 mg/kg in F0 males, 14.0, 141 and 1363 mg/kg in F0 females, 11.4, 115 and 1142 mg/kg in F1 males, and 14.3, 138 and 1363 mg/kg in F1 females for 150, 1500 and 15,000 ppm, respectively. The main effects observed were on thyroid function:

<table>
<thead>
<tr>
<th>Concentration HBCD in diet</th>
<th>Effects with increased incidence in HBCD-exposed animals</th>
</tr>
</thead>
<tbody>
<tr>
<td>≥ 1500 ppm</td>
<td>Decreased thyroid follicles size in F0 and F1 males and females</td>
</tr>
<tr>
<td></td>
<td>Increased serum TSH levels in F0 and F1 females</td>
</tr>
<tr>
<td></td>
<td>Reduced number of the primordial follicles in the ovary of F1 females</td>
</tr>
<tr>
<td></td>
<td>Increases in the absolute and relative liver weights of male adults and male and female weanlings</td>
</tr>
<tr>
<td></td>
<td>Low incidences of the completion of eye opening in female F2</td>
</tr>
<tr>
<td>15000 ppm</td>
<td>Decreased serum T4 levels in F0 males and females</td>
</tr>
<tr>
<td></td>
<td>Increases in the absolute and relative liver weights of adult females</td>
</tr>
<tr>
<td></td>
<td>Increases in the absolute and relative thyroid weights of adult males and females</td>
</tr>
<tr>
<td></td>
<td>Decreased body weight and body weight gain associated with reduced food consumption in F1 males and females</td>
</tr>
<tr>
<td></td>
<td>Reduced viability index of F2 pups and reduced body weight of male F1 and F2 pups and female F2 pups</td>
</tr>
<tr>
<td></td>
<td>Low incidences of the completion of eye opening in male F2 pups and of completed mid-air righting in female F2 pups</td>
</tr>
</tbody>
</table>

The NOAEL of HBCD in this study was 150 ppm (10.2mg/kg /day), which is consistent with the findings of Saegusa et al (2009).

In a separate study Van der Ven et al (2009) exposed rats to HBCD in the diet at target doses of 0, 0.1, 0.3, 1, 3, 10, 30 and 100 mg/kg/day in parental rats during 10 (males) or 2 (females) weeks pre-mating, gestation and lactation, and in their F1 offspring from weaning. A variety of effects were reported at levels of exposure much lower than those previously
reported to be associated with developmental toxicity. The livers of the FI offspring showed a dose-dependent increase in HBCD concentrations and a dose dependent decrease in trabecular bone (females, BMDL=0.056 mg/kg/day). The IgG response after immunization with sheep red blood cells (SRBC) was increased in males (BMDL=0.46 mg/kg/day). Other observed effects were decreased weight of the testis (BMDL=1.5 mg/kg/day), increased fraction of neutrophilic granulocytes (BMDL=7.7 mg/kg/day), decreased concentration of a polar retinoids in female livers (BMDL=1.3 mg/kg/day), and decreased plasma alkaline phosphatase in females (BMDL=8.6 mg/kg/day). CYP19/aromatase activity in the ovary was correlated to the concentration of gamma-HBCD in the liver. The significance of the various reported effects is uncertain as some are related to how HBCD is metabolised by the liver and might be considered as adaptations to exposure rather than necessarily an adverse effect. Most of the effects would not have been investigated as part of a standard assay of reproductive toxicity.

In conclusion the NOAEL for clear developmental toxicity as evidenced by effects on thyroid function appears to be of the order of 10 mg/kg/day. A range of effects of less certain significance have however been reported that have BMDLs that are between 0.05 and 8 mg/kg/day.

Developmental neurotoxicity

The adverse effect that has been most extensively discussed in relation to HBCD and other brominated compounds is developmental neurotoxicity. In a recent review, Williams and DeSesso (2010) found that although the results of the majority of studies suggest that perinatal exposure to PBDE congeners, HBCD, or TBBPA affects motor activity, there were inconsistencies in reported effects. The inconsistencies included the type of motor activity (locomotion, rearing, or total activity) affected, whether it increased or decreased, the pattern of change associated with exposure, the existence of a dose response, the permanency of findings, and the possibility of gender differences in response. Other studies that have complied Good Laboratory Practices (GLP) and followed US Environmental Protection Agency (EPA)/Organization for Economic Cooperation and Development (OECD)guidelines for developmental neurotoxicity testing have found no adverse effects associated with exposure to PBDE209, HBCD, or TBBPA at doses that were orders of magnitude higher and administered over longer durations than those used other studies that had reported effects. These authors conclude that the lack of consistency across studies precludes establishment of a causal relationship between perinatal exposure to these substances and alterations in motor activity.

Eriksson et al (2006) reported developmental neurotoxicity in mice following exposure to HBCD by intraperitoneal injection, but these findings were not endorsed during the EU risk assessment process. More recently, as part of the Van der Ven et al (2009) study described above, Lilenthal et al (2008) undertook neurological testing. Haloperidol-induced catalepsy and brainstem auditory evoked potentials (BAEPs) were used to assess dopamine-dependent behaviour and hearing function in adult offspring. On the catalepsy test, reduced latencies to movement onset were observed mainly in female offspring, indicating influences on dopamine-dependent behavior. The overall pattern of BAEP alterations, with increased thresholds and prolonged latencies of early waves, suggested a predominant cochlear effect. Effects were dose-dependent with lower bounds of benchmark doses (BMDL) between 1 and 10 mg/kg body weight for both catalepsy and BAEP thresholds.

The results of a number of mechanistic investigations involving in vitro cell systems have been claimed to support the hypothesis that HBCD can exert neurotoxic effects including Dingemans et al (2009), Reistad et al (2006) and Mariussen and Fonnum (2003).

In conclusion, it seems likely that high levels of neonatal exposure to HBCD can lead to a range of adverse effects on development that may include small effects on neurobehavioural development. The exposure levels reported to be associated with developmental neurotoxicity are within the same range as those associated with small effects on immune function and sexual development and smaller than those associated with effects on thyroid
function. They are 2 to 3 orders of magnitude lower than exposures associated with any adverse impact on offspring viability. The significance of the neurotoxic effects is highly uncertain.

Skin sensitisation
The BFRIP (2001) data summary reviewed four studies that gave inconsistent results. In two Japanese studies that used HBCD manufactured in Japan, positive results were reported in the guinea pig maximisation test, although patch testing in human volunteers gave negative results. A more recent US test in guinea pigs using US product, gave negative results even at the highest dose levels. The BFRIP (2001) concluded that the potential for HBCD to produce even a mild sensitisation reaction in humans was questionable.

Immunotoxicity
There are limited experimental data that suggest that it is possible that HBCD is capable of impairing immune function including findings in repeated dose experiments and developmental studies of impacts on spleenocyte counts (Van der Ven, 2006) and on immunoglobulin responses (Van der Ven, 2009).

Experimental data suggest that HBCD may reduce the ability of human natural killer (NK) cells to bind to and destroy tumour cells and virally infected cells implying that HBCD may potentially lead to an increased susceptibility to cancers and viral infections (Hinkson and Whalen, 2010).

Fery et al (2009) report that HBCD is an agonist of the human and rat pregnane-X-receptor, a nuclear receptor whose primary function is to sense the presence of foreign toxic substances and in response up regulate the expression of proteins involved in the detoxification and clearance of these substances from the body.

Gender differences in susceptibility to HBCD
The results of some repeated dose and developmental studies suggest that female rats may be more sensitive to HBCD than male rats (above). Gene expression analysis of samples of liver tissues from a 28-day HBCD repeated dose study (OECD407) Wistar rats, showed that females had a higher number of regulated genes and were thus more sensitive to HBCD than males (Canton et al, 2008). Pathways affected by HBCD exposure, included PPAR-mediated regulation of lipid metabolism, triacylglycerol metabolism, cholesterol biosynthesis, and phase I and II pathways. Cholesterol biosynthesis and lipid metabolism were especially down-regulated in females. Genes involved in phase I and II metabolism were up-regulated predominantly in males, which could explain the observed lower levels of HBCD found in the livers of male rats in the 28-day study.

Comparative toxicity of HBCD isomers
There has been no investigation of the comparative toxicity of HBCD isomers in animals. Most investigations have employed technical grade HBCD which is dominated by gamma HBCD and a small number have employed gamma HBCD alone. Given that the metabolic fate of the different isomers is different (see toxicokinetics), it seems likely that some differences in toxicity do exist. There is insufficient information about the mechanisms by which HBCD and its metabolites cause adverse effects to enable prediction of the likely differences in effect associated with different isomers. Although the preferential accumulation of the alpha isomer in tissue might be of concern, its presence does not necessarily imply any associated impact on health and it may be that the harm associated with HBCD in animal experiments is mediated or largely mediated by HBCD metabolites rather than the parent compound.

There are limited data that suggest that different HBCD isomers show differing levels of cytotoxicity in cellular assays. Zhang et al (2008) reported that the toxicity ranking of the 3 isomers in the 3-(4,5-dimethylthioazol-2-yl)-2,5-diphenyl tetrazolium bromide (MTT), resazurin reduction and lactate dehydrogenase (LDH) release assays was gamma-HBCD>beta-HBCD>alpha-HBCD. Significantly lower cell viability and higher LDH release
were observed in all (+)-enantiomers ((+) alpha-, (+) beta- and (+) gamma-HBCD) when compared to the corresponding (-)-forms ((-) alpha-, (-) beta- and (-) gamma-HBCD).

In an investigation into the mechanisms by which HBCD may exert neurotoxic effects, Dingemans et al (2009) examined its impacts on intracellular Ca\textsuperscript{2+} homeostasis in a neuroendocrine in vitro model (PC12 cells). Exposure of PC12 cells to the technical HBCD mixture or individual HBCD stereoisomers did not affect basal intracellular Ca\textsuperscript{2+} or the frequency of basal neurotransmitter release but did cause a dose-dependent reduction of a subsequent depolarization-evoked increase in intracellular Ca\textsuperscript{2+}. The effects of alpha- and beta-HBCD were comparable to that of the technical mixture, whereas the inhibitory effect of gamma-HBCD was larger.

In conclusion, the relative toxicity of different HBCD isomers is uncertain although there are limited data that suggest that the gamma form and (+) enantiomers are more biologically active than other isomers. There are no data that suggest that the alpha and beta isomers are likely to be substantially more harmful than the gamma isomer. It seems unlikely that the risks to human health associated with the alpha and beta isomers would be substantially underestimated as a result of assessment based on animal assays that have employed gamma or technical grade HBCD.

**Summary of toxicity data**

HBCD is readily absorbed following inhalation or ingestion and most is eliminated in faeces and urine within a few days, but a small proportion may accumulate in lipid rich tissue. There are metabolic differences between males and females that may increase the susceptibility of females to HBCD toxicity. HBCD has a low acute toxicity but in long term experiments in animals effects have been reported on the liver and on thyroid function and it is a developmental toxin. There is no evidence that HBCD is genotoxic or is likely to be associated with an increased cancer risk.

The Annex XV dossier indicates that the NOAEL/BMDLs for HBCD in repeated dose and developmental toxicity studies were 22.9 and 10 mg/kg/day based on effects on the liver and on fertility respectively. The DNELs for liver effects were calculated as 0.5 and 1.1 mg/kg/day for the general population and workers respectively. The DNELs for fertility were 0.1 and 0.2 mg/kg/day for the general population and workers respectively.

More recent assays based on more sensitive endpoints have reported lower BMDLs for developmental toxicity:

- Reduced trabecular bone in female offspring – 0.056 mg/kg/day;
- Effects on immune function in male offspring – 0.46 mg/kg/day;
- Decreased weight to testes – 1.5 mg/kg/day;
- Increased neutrophilic granulocytes – 7.7 mg/kg/day;
- Effects on liver function – 1.3 mg/kg/day.

The significance of these low dose effects is uncertain.

There are limited data derived from cellular assays that suggest that the gamma isomers and (+) enantiomers are more biologically active than other isomers but no comparison has been undertaken in animal experiments. The gamma isomer is more readily metabolised than the alpha and beta isomers but in the absence of a detailed understanding of the mechanisms underlying HBCD toxicity, it is not clear whether that would confer a greater or lesser toxicity. It seems unlikely that the toxicity of alpha and beta HBCD in animals would be substantially greater than that observed with the technical mixture or with gamma HBCD.
A4.4 EXPOSURE TO HBCD

Occupational exposure to HBCD

There are few published data describing occupational exposure to HBCD. The highest levels of exposure are likely to occur where workers are directly handling HBCD in operations such as adding HBCD to polystyrene or the use of HBCD in fabric coatings. Workers handling products containing HBCD such as construction products are likely to have higher levels of exposure to HBCD than the general population but exposure levels are likely to be a small fraction of those associated with handling HBCD itself.

Thomsen et al (2007) reported that personal exposure concentrations of HBCD for workers producing expanded polystyrene varied from 0.002 to 0.15 mgm$^{-3}$ (mean 0.0122 and median 0.0021 mgm$^{-3}$). The mean serum concentration for the 10 investigated workers was 190 ng/g lipids; the median was 101 ng/g lipids (range 6 to 856 ng/g lipids). HBCD was not detected above 1 ng/g lipids (LOD) in any samples from a reference group with no occupational exposure to HBCD. The contribution of gamma-HBCD to the total HBCD serum concentrations in workers was higher (39%) than that typically observed in biological samples. Serum levels were not clearly correlated with measured HBCD exposure concentrations in air.

In a study to inform the EU risk assessment for HBCD, IOM (Searl, 2007) undertook measurement surveys at 12 plants across the EU representing six different industrial settings where HBCD exposure is possible:

- Primary production (1 plant);
- Micronising (1 plant);
- Preparation of flame-retarded coatings for textiles (1 plant);
- Preparation of Masterbatch (1 plant);
- Production of flame-retarded expanded polystyrene (EPS- 4 plants); and
- Production of flame-retarded extruded polystyrene (XPS - 4 plants).

Measured exposure concentrations were generally below UK standards for respirable and inhalable dust (4 and 10 mgm$^{-3}$ respectively; Figure A4.1). Micronising was associated with the highest exposures and the plant has subsequently taken measures to reduce operator exposure. Other tasks associated with relatively high exposures included weighing out of HBCD, addition of HBCD to the reaction vessel during production of EPS and handling of HBCD powder during the preparation of textile coatings. These tasks were of short duration such that shift mean exposure concentrations were generally well below 10 mgm$^{-3}$. 
The reported personal exposure concentrations shown in Figure 1 are much higher than the long term mean exposure concentrations that would be typical for individual workers as workers rotated between tasks and/or only handled HBCD intermittently. In none of the workplaces visited, were workers likely to be repeatedly exposed on a large number of successive shifts to high concentrations of HBCD (Searl, 2007).

**General population exposure to HBCD**

The main sources of population exposure to HBCD are ingestion in food and in house dust. The relative importance of diet versus exposure in house dust depends on the types and sources of food in the diet which varies in different parts of the world. Airborne concentrations of HBCD are negligible and there is no appreciable absorption of HBCD through the skin following dermal contact with materials containing HBCD.

In the UK, the main sources of exposure to HBCD for most of the population are likely to be house dust and diet (Abdallah et al, 2008a and b), with breast milk dominant for nursing infants at levels that exceed those received by adults and toddlers (Harrad and Abdallah, 2010). The median concentration of HBCD (total of all isomers) in samples of house dust from 31 UK homes was 730 ng/g and not dissimilar to that reported in dust from UK offices (n=6, 650 ng/g). The highest recorded concentration was 110,000 ng/g. Concentrations in UK dust samples were statistically indistinguishable from those in samples of house dust from the US and Canada. The upper bound intake of HBCDs in ingested dust for UK adults and toddlers was estimated to be 1100 and 4400 ng/day respectively. This was based on an assumed level of dust ingestion of 50 and 200 mg/day for adults and toddlers respectively with levels of HBCD contamination equivalent to the 95th percentile of measured levels. The estimated upper bound average UK dietary exposures to total HBCD for adults and toddlers, respectively, were 413 and 240 ng/day (Food Standards Agency, 2006). Dust ingestion was estimated to represent 24% of adults and 63% of toddlers exposure to HBCDs (Abdallah et al, 2008a). The average composition of HBCD in dust comprised 33% gamma HBCD (range...
14-67%), alpha-, 11% beta-, and 56% gamma-HBCD whereas that in air was 22% alpha-, 11% beta-, and 65% gamma-HBCD and the technical product contains more than 80% gamma HBCD (Abdallah et al, 2008a and b).

In a more detailed study of 21 UK adults, Abdallah and Harrad (2009) estimated that for an average dust ingestion scenario, personal exposures to HBCD ranged from 4.5 to 1851 ng/day whereas if a high dust ingestion scenario was assumed, exposures ranged from 11 to 4630 ng/day. Exposure to HBCD was dominated by ingestion of house dust due to the large time fraction spent in houses with dust in cars contributing (17%) to total HBCD exposure (reflecting the relatively high levels found in car dust) and that in office dust only contributing 13%.

Roosens et al (2009) has similarly demonstrated that HBCD in dust, together with dietary exposures, is a significant source of exposure of the Flemish population to HBCD than diet. Measured duplicate dietary intakes of total HBCDs were 1.2-20 ng/day (average, 7.2 ng/day), whereas those estimated under average (20 mg dust/day) and high (50 mg dust/day) dust ingestion scenarios were 1.1-15 ng/day (average intake, 3.2 ng/day) and 2.8-38 ng/day (average intake, 8.0 ng/day), respectively. These intakes were reported to be at the low end of those reported elsewhere in Europe. Of particular note was the correlation between HBCD concentrations in serum with those in dust but not diet. In a later separate study, HBCD intake was found to vary with age. The total median intake of HBCD by newborns was 3.1 ng/kg/day and the 95th percentile was 15.2 ng/kg/day for higher exposed newborns (Roosens et al, 2010). For median exposed individuals, the HBCD intake peaked at the age 3 to 6 years with an intake of 6.59 ng/kg/day and declines to approximately 1 ng/kg/day at later age.

Levels of HBCD in the diet are variable. HBCD has been detected in eggs, farmed fish, wild freshwater and marine fish and farmed sea food. In a study of fish regularly consumed by Dutch citizens, HBCD was detected in 22 out of the 44 samples in concentrations of between 0.20 ng/g in marine fish and 230 ng/g ww in eel. The medium bound intake of HBCD in fish for the Dutch population was estimated at 8.3 ng/day for a 70-kg person (0.12 ng/kg/day; Van Leeuwen et al, 2008). In a subsequent study of farmed fish from southeast Asia, Europe, and South America, Van Leeuwen et al (2009) reported that contaminant concentrations generally decreased per species in the following order of salmon > trout >> tilapia approximately equal to pangasius approximately equal to shrimp. Contaminant concentrations were very low (mostly <1 ng/g wet weight) and far below the European and Dutch legislative limits. Absolute levels of HBCD were lower than those of most of the other contaminants investigated. Based on a study of HBCD concentrations in Scottish seafood, Fernandes et al (2008) estimated that adult dietary intakes of HBCD arising from the consumption of a typical portion of seafood in combination with an otherwise average UK diet were in the range 5.9-7.9 ng/kg/day. In a study of home produced free range eggs in Belgium, HBCD was only detected in some samples (<0.4 and 2.9 ng/g lipid for the autumn 2006 and spring 2007 campaigns, respectively and the highest HBCD value was 62 ng/g lipid (Covaci et al, 2009). The main source of HBCD appeared to be from soil.

Knutsen et al (2008) reported a correlation between the consumption of freshwater fish and levels of HBCD and other brominated fire retardants found in the serum of Norwegian men although there was no evidence that the fish contained unusually high levels of HBCD (Knutsen et al, 2009). Levels of fish and seafood consumption in the UK are lower than in Norway.

Concentrations of HBCD in human tissues and breast milk

The presence of HBCD in human tissues confirms that exposure to HBCD has occurred but the relationship between intake and tissue burden has not been established, so it is not possible to infer levels of population exposure to HBCD on the basis of reported concentrations in tissue.
In a Flemish study, Roosens et al (2009) reported that concentrations of HBCDs in blood serum were < 0.5 to 11 ng/g lipid (average, 2.9 ng/g lipid). Concentrations of HBCD in serum were significantly correlated with those in house dust but not with dietary intake. Pulkrabová et al (2009) reported concentrations of HBCD in adipose tissue samples (n=98) obtained by liposuction of Czech subjects of <0.5 to 7.5 ug/kg lipid weight. In a Norwegian population with relatively high levels of exposure to HBCD in fish, median serum HBCD concentrations were 4.1 and 2.6 ng/g lipids for men and women, respectively (Knutsen, 2008). Concentrations of HBCDs in human adipose tissue obtained in New York City were 0.333 ± 0.571 ng/g lipid (Johnson-Restrepo et al, 2007). The authors indicate that these levels were 3–4 orders of magnitude lower than the concentrations of polybrominated diphenyl ethers previously reported for the same set of tissue samples and 1–5-fold lower than the concentrations reported in human tissues from several European countries.

Thomsen et al (2009) measured levels of HBCD in 310 of 393 breast milk samples from mothers living in various regions throughout Norway. The median concentration of HBCD was 0.32 ng/g lipids, comparable with those reported elsewhere in Europe. The frequency distributions were quite skewed with long tails towards higher concentrations. The mothers' diet was not found to influence the breast milk HBCD levels. In a study of milk samples from 335 Swedish women who delivered between 1996 and 2006, concentrations of HBCD were below the quantification limit (<0.20 ng/g lipid) in most samples (Lignell et al, 2009). Harrad and Abdallah (2010) reported median levels of HBCD in breast milk in the UK of 4 ng/g lipid, primarily α-HBCD. The levels were comparable to levels reported from Norway and Canada, but higher than levels reported from Sweden, Russia, France, Japan and the USA.

A4.5 CONCLUSIONS

HBCD is extensively metabolised but also accumulates in fatty tissue and there is some evidence that it crosses the blood brain barrier. HBCD has a relatively low toxicity in short and long term experiments in animals. The main effects that are of potential concern are developmental neurotoxicity and effects on thyroid function. The lowest levels of exposure associated with adverse effects in animals are vastly greater than current levels of exposure in humans (Table). Similarly the Derived no effects levels (DNELs) outlined in the Annex XV dossier prepared by Sweden are much higher than current exposure levels.

<table>
<thead>
<tr>
<th>Effect</th>
<th>Intake mg/kg/day</th>
</tr>
</thead>
<tbody>
<tr>
<td>Estimated reasonable worst case exposure to HBCD in UK adults excluding workplace exposure</td>
<td>0.000066</td>
</tr>
<tr>
<td>Estimated reasonable worst case exposure to HBCD in UK toddlers</td>
<td>0.00375</td>
</tr>
<tr>
<td>Proposed DNEL liver effects - workers</td>
<td>1.1</td>
</tr>
<tr>
<td>Proposed DNEL liver effects – general population</td>
<td>0.5</td>
</tr>
<tr>
<td>Proposed DNEL fertility - workers</td>
<td>0.2</td>
</tr>
<tr>
<td>Proposed DNEL fertility – general population</td>
<td>0.1</td>
</tr>
<tr>
<td>BMDL repeated dose toxicity (28 day study)</td>
<td>22.9</td>
</tr>
<tr>
<td>BMDL developmental toxicity in 2 generation study</td>
<td>10.2</td>
</tr>
<tr>
<td>BMDL developmental toxicity – Effects of uncertain significance</td>
<td></td>
</tr>
<tr>
<td>Effects on trabacular bone in females</td>
<td>0.056</td>
</tr>
<tr>
<td>Immune effects in males</td>
<td>0.46</td>
</tr>
<tr>
<td>Decreased weight testes</td>
<td>1.5</td>
</tr>
<tr>
<td>Effects on liver function</td>
<td>1.3</td>
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The major sources of adult human exposure to HBCD are via the diet and incidental ingestion of house dust. This has been found to be a more significant pathway than inhalation of HBCD, or ingestion of HBCD via the food chain (Abdallah et al, 2008a and b). Nursing infants are also exposed via consumption of human milk at levels that exceed those of most adults (Harrad, 2010).

There are limited data that suggest that there are gender-related differences in the metabolism of HBCD and its toxic effects but these are unlikely to be of importance given the low level of human exposure to HBCD. There are also data that indicate that there are substantial differences in the metabolism of different HBCD isomers that are likely to give rise to some differences in toxic effect. There are insufficient data from animal experiments to determine the relative toxicity of different isomers. Limited data from cellular assays suggest that the cytotoxicity of gamma HBCD is greater than that of the other isomers but the significance of this finding is uncertain, given the more extensive metabolism of the gamma isomer which would lead to lower levels of systematic exposure. Despite the uncertainty as to the relative toxicity of different HBCD isomers, it seems unlikely that the BMDLs and NOAELs reported in assays that have employed technical grade or gamma HBCD are substantially higher than would be found in similar assays with alpha or beta HBCD, although some difference in effects is possible.

Workers handling products containing HBCD are likely to have slightly higher levels of exposure to HBCD than other members of the population but exposure levels would remain well below those associated with adverse effects in animals. Higher levels of exposure are possible where powdered HBCD is handled in the workplace, but exposures can be readily controlled through appropriate workplace design. Workplace exposure to a long term mean concentration of 1 mg/m$^3$ as an 8 hour mean would equate to an average daily intake of 0.4 mg/kg which would exceed the DNEL for fertility, but it should be possible to control exposure to considerably lower levels. Very few people in the UK are likely to be directly exposed to HBCD powder at work.

There is no evidence to suggest that reducing exposure to HBCD by phasing it out of consumer products and construction materials would have any significant direct benefit to human health for the general population.