Margaret Simonson, Petra Andersson and
Martin van den Berg

Cost Benefit Analysis Model for Fire Safety

Methodology and TV (DecaBDE) Case Study

SP Swedish National Testing and Research Institute

SP Fire Technology
SP REPORT 2006:28

Abstract

A fire cost-benefit model (Fire-CBA) has been developed to evaluate the financial impact of regulations and voluntary industry initiatives, aimed at the removal of flame retardants. This model has been constructed to include such costs as: incremental increases in cost to flame retard a product relative to a non-flame retarded product; additional costs for disposal of the product at the end of the product life cycle. Similarly, the model includes provisions for benefits such as: lives saved, injuries avoided, capital costs avoided through fires averted.

In all, a total of 8 scenarios were tested for the TV set application of the Fire-CBA model developed in this report. In all cases the benefits of a high level of fire performance in a TV set far outweigh the costs associated with obtaining that high level of fire safety. The net benefit is a function of the choices made in the various scenarios but ranges between 657 to 1 380 million US$ (or approximately 520 – 1100 million €) per year.

The various scenarios were chosen to illustrate the significance of the various parameters included in the study as the specific value chosen for each parameter can vary depending on the assumptions made in the model.

Key words: fire, cost benefit, LCA, flame retardants

SP Sveriges Provnings- och Forskningsinstitut
SP Rapport 2006:
ISBN 91-85533-13-0
ISSN 0284-5172
Borås 2006

SP Swedish National Testing and Research Institute
SP Report 2006:28

Postal address:
Box 857,
SE-501 15 BORÅS, Sweden
Telephone: +46 33 16 50 00
Telex: 36252 Testing S
Telefax: +46 33 13 55 02
E-mail: info@sp.se

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Preface

This project has been conducted under the auspices of the International Consortium for Fire Safety, Health and the Environment. Funding has been provided by the Bromine Science and Environmental Forum (BSEF).

The project has been lead by Dr Margaret Simonson of the Department of Fire Technology at SP Swedish National Testing and Research Institute in collaboration with Dr Petra Anderson of the same department and Professor Martin van den Berg, Deputy Director and Head Toxicology Division, Institute for Risk Assessment Sciences (IRAS) and World Health Organization Collaborating Centre for Research on Environmental Health Risk Assessment, Utrecht University. Dr Simonson and Dr Andersson have been responsible for all fire safety, LCA and CBA aspects of the report while Professor van den Berg has been responsible for all toxicity aspects of the study.

Executive summary

In recent years there has been an increased focus on sustainable development. For development to be sustainable it must integrate environmental stewardship, economic development and the well-being of people, not just for today but for generations to come. For the past several decades, regulation of the environment has been covered by Environmental Protection Agencies worldwide.

Perhaps one of the most important lessons that have been learned from our experience of environmental regulation is that regulations have significant costs, not just benefits and that analysis of both the cost and benefit of proposed legislation is imperative. Despite our recognition of the importance of cost benefit analysis prior to legislation, this is still a controversial issue, especially in light of moral issues such as determination of the Value of a Statistical Life (VSL)[3].

In all balanced evaluations of the risks posed by a product or activity one must take a holistic approach. The most common method to assess the environmental impact of a product or activity is through the use of life-cycle assessment methodology. SP Fire Technology has, together with IVL Swedish Environmental Research Institute, developed a life-cycle assessment model (Fire-LCA) in which the effect of the chosen level of fire safety in the functional unit is included in the overall impact assessment[9]. The Fire-LCA tool is well equipped to take into account the fact that a product with a high level of fire safety is involved in fewer and smaller fires than a product with a lower level of fire safety. This model is, however, not able to take into account the cost associated with a loss of life or the societal and individual costs and benefits associated with exposure to fires and/or chemicals.

In this project, real and perceived risks associated with exposure to flame retardants and fires will be discussed and a monetary value placed on the costs and benefits associated with these chemicals. To this end a specialised cost benefit analysis model, the Fire-CBA, has been developed. The model is developed in a generic sense and then applied to a specific case study, i.e., a TV set. The case study compares a TV set with low fire performance with another of high fire performance. The high fire performance TV contains DecaBDE in the outer enclosure to protect the TV from small open flame ignition. More detailed information concerning a Fire-LCA analysis of this case study is provided elsewhere[10]. This case study represents the first attempt to establish the monetary cost and benefit of the use of flame retardants in TVs.

In order to evaluate the cost and benefit of a product or additive it is necessary to consider both chemical exposure and health risk and the fire exposure.

Based on available data there is a measurable human exposure to DecaBDE. However, various risk assessments for background occupation and infant exposure via breast milk or household dust indicate that no adverse health effects are to be expected due to the large margins of safety that exist. The only exception can be found for occupational settings when using the neurobehavioural effects found in neonatal mice[20]. However, it is extremely doubtful if such an experimental design using neonates is applicable to healthy adult workers. Consequently, the cost of exposure and lack of expected associated adverse health effects to DecaBDE have been considered zero in this risk-benefit analysis.

Exposure to a fire can lead to dire consequences. Numerous costs associated with fires include: fire fighting, post-fire clean up, replacement of destroyed or damaged equipment, treatment of fire victims that do not die, societal costs due to fire fatalities. This study

does not attempt to include the cost of fire fighting or post-fire clean up. An attempt has been made to include the cost of replacing destroyed or damaged equipment and the societal cost for treatment of injuries or untimely death. Other costs that are discussed and included in the model are: production costs, end of life costs, life-cycle costs etc.

A life-cycle approach must take into account the fact that averting behaviour can be used now to change mortality risks in the future. This is generally done through discounting of future lives relative to present lives. In 1980, the US EPA Office of Management and Budget (OMB) strongly urged discounting the value of human lives over the period of latency of the harm[32]. At that time they recommended a discount rate of 10% but more recent CBA by the US EPA use a more moderate discount rate of 3%[46]. The discounting rate has a significant effect on the results. To determine the sensitivity of the calculations to the discounting rate both rates have been included in this study.

A number of different input parameters are important to the outcome of the Fire-CBA calculation. To investigate the robustness of the model important parameters were varied in eight different scenarios. The parameters that were varied and the values used are: discounting value (3% or 10%); cost of disposal ($1 or $13,3); inclusion of the cost of the fire (fully, indirectly or not at all); inclusion of insurance costs (yes or no).

The results of the CBA calculation indicate clearly that in all cases investigated the benefits of a high level of fire performance in a TV set far outweigh the costs associated with obtaining that high level of fire safety. The net benefit is a function of the choices made in the various scenarios but ranges from $657 million per year to $1380 million per year.

Nomenclature

CBA	Cost Benefit Analysis
CPSC	Consumer Product Safety Commission
DE	Diphenyl ether
EC50	The concentration of a compound where 50% of its effect is observed.
EPA	Environmental Protection Agency
FR	Flame retardant
In vivo	occurring or carried out in the living organism
In vitro	made to occur outside the body of the organism, in an artificial environment
LCA	Life-cycle Assessment
LC50	The concentration of a compound where 50% of the exposed population will die under well defined conditions.
LOAEL	Lowest Observed Adverse Effect Level
LOEL	Lowest Observed Effect Level
mg/kg bw	milligram/kilogram body weight
mg/kg d	milligram/kilogram and day
Neonate	Newborn
NOAEL	No Observed Adverse Effect Level
NOEL	No Observed Effect Level
NTP study	National Toxicology Program study
PBDE	Polybrominated diphenyl ether
PCB	Polychlorinated biphenyl
Postnatal	After birth
Prenatal	Before birth
VSL	Value of a Statistical Life
WHO/FAO	World Health Organisation / Food and Agriculture Organization of the United Nations

1 Introduction

In recent years there has been an increased focus on sustainable development. For development to be sustainable it must integrate environmental stewardship, economic development and the well-being of all people, not just for today but for generations to come. This is the challenge facing governments, non-governmental organizations, private enterprises, communities and individuals.

For the past 30 to 40 years, well before the concept of sustainable development became politically correct, regulation of the environment has been covered by Environmental Protection Agencies worldwide. The Swedish EPA was established in 1967, the US EPA in 1970, the Australian EPA in 1971 and the list goes on. Since this time regulations to protect the environment have been made with some unquestionable benefits but since the initial efforts to revitalise water supplies and reduce industry emissions, regulation has become increasingly complex and a more holistic approach needs to be taken when developing regulations.

A great deal of experience has been gained since the inception of these various agencies worldwide. Perhaps one of the most important things that we have learned is that regulations have significant costs, not just benefits and that analysis of the cost and benefit of proposed legislation is an indispensable component of responsible legislation[1]. In the early 1980's this was recognised in the US when the CPSC drafted legislation requiring cost-benefit analysis to be connected to all proposed regulations[2].

Despite the recognition of the importance of cost benefit analysis prior to legislation, this is still a controversial issue, especially in light of moral issues such as establishing the value of a statistical life (VSL)[3] and whether net benefit is always necessary before invoking legislation[4]. Indeed, in 1996 it was estimated that the direct costs of federal environmental, health and safety regulation in the US was US$200 billion annually[5].

In order to make a full analysis of the costs and benefits of a particular product or activity it is important to understand the difference between hazards and risks. A hazard is a situation with a *potential* for human injury, damage to property, the environment or some combination of these. A risk is the *likelihood* of a specified undesired event occurring within a specified period or in specified circumstances arising from the realisation of a specified hazard. A risk may be expressed as either the frequency of the occurrence or the probability of occurrence, depending on the circumstances. An *individual risk* relates to the frequency at which an individual may be expected to sustain a given level of harm from the realisation of the specified hazards while a *societal risk* represents the frequency at which specified numbers of people in a given population, or the population as a whole, sustain a specified level of harm from the realisation of specified hazards[6].

Our understanding of risk and hazard is further complicated by the fact that risks can be perceived rather than real. The precautionary principle is a clear example where perceived, unquantifiable risks can be cited as the basis of regulations[7,8]. In cases where the public is in control of the risks to which they are exposed they are more likely to define them as acceptable, than when they are not in control. The choice to smoke, for example, is seen by some as an acceptable risk while exposure to a chemical additive in food or goods is not acceptable despite the fact that the risk of smoking is far greater than that of the chemical exposure.

Unfortunately there is no definition of an acceptable risk and in some ways the very word "risk" implies that it entails something that is not acceptable. It is, however, important to consider risks relative to one another. The aim should be to reduce the sum of all risks rather than reduce one risk to the detriment of another. In this context it is necessary to recognise that in the process of reducing one risk (such as through the removal of a flame retardant additive) may increase another risk (such as the risk for exposure to a fire).

In all balanced evaluations of the risks posed by a product or activity one must take a holistic approach. The most common method to assess the environmental impact of a product or activity is through the use of life-cycle assessment methodology. SP Fire Technology has, together with the Swedish Environmental Research Institute, developed a life-cycle assessment model in which the effect of the chosen level of fire safety in the functional unit is included in the overall impact assessment[9]. The Fire-LCA tool is well equipped to take into account the fact that a product with a high level of fire safety is involved in fewer and smaller fires than a product with a lower level of fire safety. This model is, however, not able to take into account the cost associated with a loss of life or the societal and individual costs and benefits associated with exposure to fires and/or chemicals.

In this project, real and perceived risks associated with exposure to flame retardants and to fires will be discussed and a monetary value placed on the costs and benefits associated with these chemicals. The model will be developed in a generic sense and then applied to a specific case study. To illustrate the case study, some background will be provided to previous work conducted using the Fire-LCA methodology and its application to TV sets, with and without DecaBDE in their external enclosures. More detailed information concerning the previous Fire-LCA TV case study can be found elsewhere[10].

This case study will provide a first attempt to establish the monetary cost and benefit of the use of flame retardants in TVs.

2 Evaluation Methodologies

2.1 Chemical exposure and health risk

This chapter will deal with the effect of chemical exposure on people, rather than flora and fauna in general, and costs to society from such exposure. Note that DecaBDE is taken as the Case study example in this chapter due to its relevance to the TV Case Study.

2.1.1 General

In order to describe the possible risks of decabromodiphenylether (DecaBDE) for humans and costs associated with this, a number of aspects must be evaluated.

Firstly, available toxicological experimental studies must be evaluated with respect to effects and specific mechanisms of action of this compound. In addition, the possible human relevance of the effects observed in toxicological studies must be shown and quantitative information from the experimental dose – effect relationships be used, for further risk assessment.

Secondly, results from exposure analysis must be put in perspective with regard to the actual experimental situations under which the toxicological studies have been performed. In this respect, different exposure situations that are relevant for the human DecaBDE exposure can be recognized, being primarily occupational, background and infant conditions.

Finally, the results from the exposure analysis must be linked with the quantitative information obtained from toxicological studies that produce information about the margin of safety actually existing for a specific situation. Here, it should be noted that exposure analysis invariably gives a range of data and the question should be raised: "which statistical parameter (e.g. median, average, maximum or 95% upper confidence interval) must be used for risk assessment?"

Once the risk for a compound has been established, which percentage of the population would suffer from adverse health effects (e.g. loss in life years or medical care) can be calculated, and what will be the associated costs. In order to do a proper risk-benefit analysis of DecaBDE this approach should ideally be followed if calculations indicate that there is an insufficient margin of safety (usually less than 100).

2.1.2 Human exposure to DecaBDE

Most of the exposure analyses done so far included only the lower brominated PBDEs that are common from a quantitative point of view in the human body and the food chain e.g. DEs 47, 99 and 153. Studies reporting about human exposure and systemic levels of DecaBDE are much more scarce and of recent origin, due to difficulties in chemical analysis. With respect to exposure, three specific situations should be recognized: occupational, background and infant exposure. The highest systemic levels for DecaBDE have been reported from Swedish workers e.g. in the cable or flame retardant rubber production with concentrations being 50 to 100 fold higher than those in the background population. Concentrations of DecaBDE in the occupationally exposed group ranged from 1 to 280 pmol/ g lipid (equiv. ~280 ng/g lipid or ~2 ng/g serum; 0.65% lipid in serum)[11].

Several other recent studies included human serum concentrations of DecaBDE in background populations, indicating extremely large individual variations for this compound for unknown reasons. Studies within these background European populations (UK and The NL) indicate similar levels of DecaBDE. Minimal measurable blood/serum levels ranged from 10 to 50 ng/g lipid[12, 13, 14], while incidental high concentrations observed as high as 200 to 300 ng/g lipid. It should be noted that individual high concentrations in background exposure situations are comparable with those observed in the occupational situations[11]. Furthermore, DecaBDE is usually measured in only 10 to 20% of the individuals examined, while detection limits are nowadays usually in the range of ~ 1 ng/g lipid.

Recently, information about the global occurrence of DecaBDE has been summarized by Londen and Van den Berg[15] (see Appendix 1). This overview shows that DecaBDE serum and milk levels between Europe and North America are not significantly different, although individual levels appear to be higher in Europe. However, this should be interpreted with caution in view of the more limited information available from North America. Based on the human levels reported above it is possible to use this as quantitative information in the risk assessment for DecaBDE. This has been done by several regulatory agencies, but also by industry and individual scientists.

2.1.3 Risk assessments done for DecaBDE

Specific risk assessments have now been done by the American Chemistry Council's Brominated Flame Retardant Industry Panel (2002)[16]; The European Union[17,18] and the WHO/FAO (2002)[12]. In their assessments two types of approaches were chosen, either based on the chronic toxicity NTP study (1986)[19] or the Viberg (2003) neurobehavioral study[20]. In addition, three different exposure scenario's could be applied: occupational, background and infant exposure.

In their risk assessment the EU selected the 1,120 mg/kg bw NOAEL for systemic effects from the NTP chronic toxicity study (1986)[19]. With respect to the neurobehavioral study, the EU (2004)[18] concluded that a NOAEL of 2.22 mg/kg day may be derived, but it was found to have many limitations and should not be used without the availability of a confirmational study. The latter study was also criticized by the EU because of the statistical analysis used by Viberg et al. (2003)[20] and the very limited number of dose groups (n=2). Using the information from the Viberg study[20], Vijverberg and Van den Berg (2004)[21] calculated with mouse specific kinetic and physiological parameters, and the brain concentration that could be expected at a 20 mg/kg LOAEL dose level. An expected brain concentration of ~ 25 µg/g lipid weight was calculated. Assuming that body distribution across organs for DecaBDE is mainly dependant on the amount of lipid and, when normalized, is approximately equal for different organs and blood, this still leaves a safety factor of two orders of magnitude between the expected concentrations in the mouse compared with the highest (occupational) serum levels in humans.

The human background exposure levels from environmental sources were originally estimated by the EU (2002)[17] at 0.05-12 µg/kg bw/d, but recent exposure analysis[12, 13, 14] indicate that these might be significantly higher for some individuals and sometimes similar to those found in occupational situations. This new exposure data has raised some concern, and the EU concluded that more research is needed in this respect (EU 2004)[18].

During occupational exposure to DecaBDE inhalation and ingestion are thought to be major routes of exposure. Due to the physico-chemical properties of DecaBDE it seems unlikely that dermal absorption plays a primary role in exposure, as experiments with skin from hairless mice showed that dermal adsorption is low and probably even less for humans[22]. The EU (2004)[18] estimated a dermal adsorption of 1-2% for humans. For inhalation and dermal exposure in the occupational setting the EU estimated 0.7 and 0.12 mg/kg day, respectively. Using the NOAEL of 1,120 mg/kg day from the chronic NTP study (1986) with an adsorption of 26% leading to and "internal" NOAEL of 291.2 mg/kg day provides a <u>safety factor of ~400 for inhalation exposure and ~2500 for dermal exposure</u>. Based on this calculation the EU concludes that the internal exposure to DecaBDE via these occupational routes is not likely to pose a health threat.

The same risk assessment and calculations can also be made using the dose level of 2.22 mg/kg day as NOAEL from the Viberg et al. (2003) study and will give <u>(safety) factors of 0.8 and 4.8 for inhalation and dermal exposure</u>, respectively[20]. These are clearly insufficient, but the question can be raised if the model used by Viberg et al. (2003) is appropriate for the adult healthy worker in an occupational setting. The type of experiment and chosen endpoints in the Viberg study might be considered more appropriate for the developing infant situation than for adults.

Risk assessments were also done by the EU (2004)[18] for children that were either breastfed or exposed to DecaBDE via house dust. As children should be considered the most sensitive subpopulation, the obtained results are highly relevant for the human population.

As a worst case approach the highest level of ~8 μg DecaBDE/kg lipid in breast milk observed by Schecter et al. (2003)[23] was used for estimating the average daily intake via breast milk. Average daily intake for infants in their first twelve months via breast milk has been estimated to be 5.2 ng/kg day[18]. Again, taking the NOAEL of 1,120 mg/kg day from the EU risk assessment and assuming that the adsorption of DecaBDE after oral exposure is the same in the rat and breastfed infant[17, 18] this leads to a <u>safety factor of approximately $2*10^8$</u>. This safety factor indicates that the exposure of young children to DecaBDE via breast milk or dust does not pose a threat. Alternatively, the 2.22 mg/kg day NOAEL of the Viberg et al. (2003)[20] study can be used as the experimental design approaches closely the breastfeeding situation. This results in a <u>safety factor of $4*10^5$</u> that is still very large and no need for concern.

Another possible major route of exposure to DecaBDE for children is household dust, which was recently addressed by Stapleton and co-workers (2005)[24]. It was estimated by these authors that with a dust intake of 0.02-0.2 g/day, the intake of DecaBDE alone could range from 180 to 1750 ng/day[24]. For a worst case scenario of 1750 ng/day and a young child (age 1-3) of ~13 kg[25] and similar adsorption of DecaBDE after oral exposure in the rat and child, the estimated daily intake would be 135 ng/kg day. Taking the NTP NOAEL of 1,120 mg/kg day[19] from the EU risk assessment[17, 18] this provides a <u>safety factor of approximately $8*10^6$.</u> Alternatively, the 2.22 mg/kg day NOAEL of the Viberg et al. (2003)[20] study can be used for this specific infant exposure situation, which results in a <u>safety factor of $1.5*10^4$</u>. No matter, which risk assessment model is applied, in both situations the margin of safety for the infant would still be very large indicating that household dust exposure, like breast milk consumption, does not pose a health threat for the infant with respect to DecaBDE exposure.

The JECFA (2005, in litt.) also did a recent risk assessment for human exposure to the total amount of PBDEs, including DecaBDE. Although this approach gave no specific information about DecaBDE, it was observed that the results of a chronic study were only available for DecaBDE, thereby providing the most detailed toxicological information. Based on dietary exposure to an average total amount of PBDEs of 0.004 μg/kg bw per day or an intake of breastfed infants of ~0.1µg/kg day it was concluded that the margin of safety was sufficient and not likely to be of significant health concern.

Finally, the American Chemistry Council's Brominated Flame Retardant Industry Panel (2002) also performed risk assessments for DecaBDE. Similar to those done by regulatory authorities and independent scientists, it was calculated that no adverse health effects were to be expected. Results from these risk assessments are shown in Table 1

Table 1: Exposure estimates for DecaBDE and Hazard quotients (HQs) based on reference dose of 4 mg/kg.day[16].

Daily intakes	Exposure Duration (yrs)	Esposure Estimate (mg/kg/d)		Hazard Quotient (RfD = 4 mg/kg/d[e])	
		Reasonable	Upper	Reasonable Estimate	Upper Estimate
Pathway-specific					
Ingestion, breast milk-manufacturer	0-2	1,9E-02[a]	3,4E-01	0,005	0,09
Ingestion, breast milk-disassembler	0-2	3,3E-06[a]	2,5E-05	8E-07	6E-06
Ingestion, consumer electronics	0-2	4,3E-06	2,5E-04	1E-06	6E-05
Ingestion, mouthing fabric (NAS)	0-2	2,6E-02	2,6E-02	0,007	0,007
General exposures	0-70	1,3E-03	3,9E-01	0,0003	0,1
Aggregate					
Infant, manufacturer	--	0,046[b]	0,76[b]	0,01	0,2
Infant, disassembler	--	0,027[c]	0,41[c]	0,007	0,1
Lifetime (0-70)	--	0,0012[d]	0,39[d]	0,0003	0,1

[a]Assumes a shorter duration for nursing (0-3 months), based on Collaborative Group on Hormonal Factors in Breast Cancer 2002.
[b]This value incorporates the intakes for ingestion of breast milk from a mother who is a manufacturer, plus ingestion from consumer electronic products, ingestion from mouthing fabric, and general exposures.
[c]This value incorporates the intake for ingestion of breast milk from a mother who is a disassembler, plus ingestion from consumer electronic products, ingestion from mouthing fabric, and general exposures.
[d]This value incorporates the intake from general exposures.
[e]The RfD is an estimate (with uncertainty spanning perhaps an order of magnitude) of a daily oral exposure to the human population (including sensitive subgroups) that is likely to be without an appreciable risk of deleterious effects during a lifetime.

2.2 Life-Cycle Assessment

2.2.1 An overview

Life-Cycle Assessment (LCA) is a versatile tool to investigate the environmental aspects of a product, a process or an activity by identifying and quantifying energy and material flows for the system. The use of a product or a process involves much more than just the production of the product or use of the process. Every single industrial activity is actually a complex network of activities that involves many different parts of the society. Therefore, the need for a system perspective rather than a single object perspective has become vital in modern research. It is no longer enough to consider just a single step in the production. The entire system has to be considered. The Life-Cycle Assessment methodology has been developed in order to handle this system approach. A Life-Cycle Assessment covers the entire life-cycle from the “cradle to grave” including crude material extraction, manufacturing, transport and distribution, product use, service and maintenance, product recycling, mechanical material recycling (not feed stock recycling) and final waste handling such as incineration or landfill. With LCA methodology it is possible to study complex systems where interactions between different parts of the system exist.

LCAs are also a much better tool to evaluate the environmental impact of a chemical substance used in a product than purely hazard based assessments. Hazard based assessments look only at the potential for environmental damage by focusing on the hazardous characteristics of a substance and worst case use scenarios without taking account of how the substance is actually used, and of possible environmental benefits or costs resulting indirectly from the function of the substance

The prime objectives are:

- to provide as complete a picture as possible of the interactions of an activity with the environment;
- to contribute to the understanding of the overall and interdependent nature of the environmental consequences of human activities; and,
- to provide decision-makers with information that defines the environmental effects of these activities and identifies opportunities for environmental improvements.

An LCA evaluates the environmental situation based on ecological effects and resource use. An LCA does not cover the economical or social effects. In an LCA, a model of the system is designed. This system is of course a representation of the real system with various approximations and assumptions.

The most widely accepted Life-Cycle Assessment methodology is based on standard LCA methodology[26, 27]. This methodology is described in the ISO standard 14040-series and other documents from different countries in Europe and the USA. Generally the method can be divided into three basic steps with the methodology for the first two steps relatively well established while the third step is more difficult and many research projects have been focused on this subject. The three steps are:

1a)	Goal definition and scoping	LCI –
1b)	Inventory analysis	Life cycle inventory
2)	Impact analysis	
3)	Valuation phase	

The *Goal Definition and Scoping* consists of defining the study purpose, its scope, project frame with system boundaries, establishing the functional unit, and establishing a strategy for data collection and quality assurance of the study. Any product or service needs to be represented as a system in the inventory analysis methodology. A system is defined as a collection of materially and energetically connected operations (e.g., manufacturing process, transport process, or fuel extraction process) that perform some defined function. The system is separated from its surroundings by a *system boundary*. The whole region outside the boundary is known as the *system environment*.

The actual data collection occurs in the *Inventory analysis* which, together with the *goal definition and scoping* make up the first step in a full LCA, i.e. the *Life Cycle Inventory*.

The *Functional Unit* is the measure of performance that the system delivers. The functional unit describes the main function(s) of the system(s) and is thus a relevant and well-defined measure of the system. The functional unit has to be clearly defined, measurable, and relevant to input and output data. Examples of functional units are "unit surface area covered by paint for a defined period of time", "the packaging used to deliver a given volume of beverage", or "the amount of detergents necessary for a standard household wash." It is important that the functional unit contains measures for the efficiency of the product, durability or life time of the product and the performance quality standard of the product. In comparative studies, it is essential that the systems be compared on the basis of equivalent function.

The most difficult part and also the most controversial part of an LCA is the *Impact Assessment*. No single standard procedure exists for the implementation of impact assessment although generally different methods are applied and the results compared.

In the *valuation* phase the different impact classes are weighed against each other. This can be done qualitatively or quantitatively. Several evaluation methods have been developed. The methods that have gained most widespread acceptance are based on either expert/verbal systems or more quantitative methods based on valuation factors calculated for different types of emissions and resources such as Ecoscarcity, Effect category method (long and short term), EPS- system, Tellus, Critical volume or Mole fraction. Due to the fact that many important emission species from fires (e.g., dibenzodioxins and furans, PAH, PCB, and DecaBDE etc) are either not dealt with in detail or not available at all, these methods are not suitable for an objective interpretation of the environmental impact of fires.

In some cases the LCA analysis is followed by an *interpretation phase* where the results are analysed. This phase provides an opportunity for the discussion of the results in terms of safety aspects. The fact that people may die in fires and that flame retardants cause reductions in the number of fire deaths cannot be included explicitly in the LCA. This should be, and is, discussed together with the results of the LCA analysis to provide a context for their interpretation and a connection to the reality of fire safety. A Cost-Benefit Analysis together with a full LCA could assist in this interpretation phase.

An LCA study has theoretical and technical limitations. Therefore the following parts of a system are usually excluded:

- *Infrastructure*: Construction of production plants, buildings, roads etc.
- *Accidental spills*: Effects from abnormal severe accidents. In the new “Fire-LCA” model, fires are included but not industrial accidents during production.
- *Environmental impacts caused by personnel*: Waste from lunch rooms, travels from residence to workplace, personal transportation media, health care etc.
- *Human resources*: Work provided by humans is not included.

An LCA analysis usually covers energy use, use of natural resources and the environmental effects. In an entire decision making process the LCA results and the environmental aspects are only a part of all the decision factors such as economic factors, technical performance and quality, and market aspects such as design. A Cost-Benefit Analysis offers clear advantages in combination with an LCA.

2.2.2 The risk assessment approach

In a *conventional* Life-Cycle Assessment the risk factors for accidental spills are excluded. For example, in the LCA data for the production of a chemical, only factors during normal operation are considered. However, there can also be, for example, emissions during a catastrophic event such as an accident in the factory. Those emissions are very difficult to estimate due to a lack of statistical data and lack of emission data during accidents. The same would apply to electric power production in nuclear power plants.

In the case of the evaluation of normal household fires the fire process can be treated as a commonly occurring activity in the society. The frequency of fire occurrences is relatively high (i.e. high enough for statistical treatment) and statistics can be found in both Europe and the USA[28]. This implies that it is possible to calculate the different environmental effects of a fire if emission factors are available. The fundamental function of flame retardants is to prevent a fire from occurring or to slow down the fire development. The introduction of flame retardants into products will thus change the occurrence of fires and the fire behaviour. By evaluating the fire statistics available with and without the use of flame retardants the environmental effects can be calculated. The benefits of the flame retardant must be weighed against the "price" society has to pay for their production and handling. To evaluate the application of flame retardants in society the Life-Cycle Assessment methodology can be modified and used. In this way a system perspective is applied. Such a model has been developed previously and applied to three case studies: TV[10], Cables[29], and Furniture[30]. Guidelines for the application of this model have also been written to facilitate its extension beyond the existing case studies[31]

The traditional application of the Fire-LCA methodology has not included a monetary evaluation of the costs and benefits of the various alternatives. This application of a holistic cost-benefit analysis will allow a realistic evaluation of the costs to society of requiring a high level of fire safety compared to those of allowing a low level of fire safety. This will be developed in more detail in the section on cost-benefit analysis.

2.2.3 The Fire-LCA system description

Schematically the Fire-LCA model can be illustrated as in Figure 1. The model is essentially equivalent to a traditional LCA approach with the inclusion of emissions from fires being the only real modification. In this model a functional unit is characterised from the cradle to the grave with an effort made to incorporate the emissions associated with all phases in the functional unit's life-cycle.

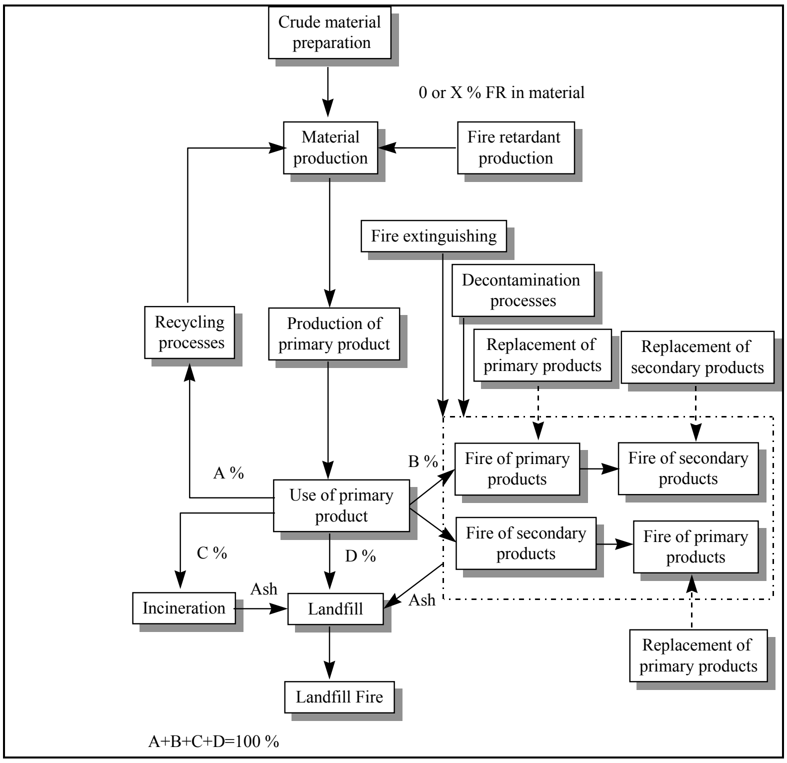


Figure 1: Schematic representation of the LCA model.

It is difficult to allocate emissions associated with accidents due to the lack of statistical data. Fires are slightly different to industrial accidents (e.g., accidental emissions during production of a given chemical) as a wealth of statistics is available from a variety of sources (such as, Fire Brigades and Insurance Companies). Differences between countries and between different sources in the same country provide information concerning the frequency of fires and their size and cause. The use of these fire statistics is discussed in more detail in the next section.

In order to facilitate the detailed definition of the Fire-LCA model shown in Figure 1 let us first define the *Goal and Scope* of the Fire-LCA and its' *System Boundaries* and discuss the possible choices of *Emissions* to include in the Fire-LCA output.

Goal and Scope: The aim of this model is to obtain a measure of the environmental impact of the choice of a given level of fire safety. Implicit in this model, is the fact that to obtain a high level of fire safety with flammable material it is necessary to include flame retardants and that the choice of flame retardant will depend on both the material and application. In order to assess the environmental impact of the presence of the flame retardant it will be necessary to compare two examples of the same functional unit: one with and one without flame retardant. The model does not necessarily aim to obtain a comprehensive LCA for the chosen functional unit. In other words only those parts of the model that differ between the flame retarded and non-flame retarded version of the product are considered in detail. All other parts are studied in sufficient detail to obtain an estimate of the size of their relative contribution. Further, present technology should be assumed throughout. In those cases where alternatives exist these should be considered as 'best' and 'worst' cases or as 'present', 'possible future' and 'state-of-the-art' technologies. These alternatives can be presented as possible scenarios and the effect of the choices made can be illuminated by comparisons between the various scenarios.

System Boundaries: According to standard practice no account is taken of the production of infrastructure or impact due to personnel. Concerning the features of the model that are specifically related to fires the system boundaries should be set such that they do not appear contrived. In general it is realistic to assume that material that is consumed in a fire would be replaced. Where possible, literature data should be used to ascertain the size of relevant contributions. In lieu of such data an estimate of the contribution is made based on experience of similar systems. In the case of small home fires, which are extinguished by the occupant without professional help, the mode of extinguishment is not included due to the difficulty in determining the extinguishing agent. In cases where the fire brigade is called to a fire, transport and deployment should be included as realistically as possible.

Emissions from fires: A wide variety of species is produced when organic material is combusted. The range of species and their distribution is affected by the degree of control in the combustion process. Due to its low combustion efficiency a fire causes the production of much more unburned hydrocarbons than does a controlled combustion. In the case of controlled combustion one would expect that carbon dioxide (CO_2) emissions would dominate. In a fire, however, a wide variety of temperature and fuel conditions and oxygen availability are present. Thus, a broader range of chemical species, such as CO, polycyclic aromatic hydrocarbons (PAH), volatile organic compounds (VOC), particles, and dibenzodioxins and furans must be considered.

The above choices provide the framework for the Fire-LCA. They should not be seen as insurmountable boundaries but as guidelines. As intimated above, in most applications of an LCA it is common to propose a variety of scenarios and to investigate the effect of the choices involved. Typically the system boundaries may be defined in different ways and the effect of this definition can be important for our understanding of the model.

2.2.4 Fire-LCA: TV Case Study

A common application of brominated flame retardants is in TV sets. A TV set application was, therefore, used as the first test of the Fire-LCA model[9, 10]. An overview of this application of the model is shown in Appendix 2. To make the figure easier to read the electric power production modules have been excluded. The model covers essentially four different parts of the life-cycle:

- TV set production (including material and component production),
- TV use,
- waste handling of the TV set, and
- TV set fires (including material replace etc).

The life-cycle of a TV set starts with the production of the different raw materials used in the TV set production. The materials are described in each module from “cradle to factory gate”. In this application, special attention was paid to the production of the flame retardants. From the production, the TV sets were distributed to the different users. In the study, the use of one million TV sets was analysed. The TV sets were then used during their entire lifetime. After their regular lifetime, the TV sets were treated in the waste handling modules. Three different waste handling possibilities were used in the model:

1. Waste (TV sets) to landfill,
2. Waste (TV sets) to incineration,
3. Waste (TV sets) to mechanical material recycling (not feed stock recycling).

In the case of mechanical material recycling the TV sets were first disassembled. The different materials were then transported to a specific material recycling process. From the disassembly process the material that were not recycling, could be transported to incineration or landfill. This process did not include feed stock recycling.

With the use of TV fire statistics, a number of different TV fires were identified. The fires potentially involved not only the TV set but also an entire room or house. From the fire statistics the number of fires per million TV sets was identified and this information was used in the model. A fire will shorten the life time of the different products involved in the fire and those products must thus be replaced. An average of 50 % life time reduction was assumed in the model. Thus, only 50 % of the material was replaced.

Some important results of this application will be provided in Chapter 4 together with an estimate of the potential fatalities and burn injuries associated with TV fires each year. This information will be provided as a background to the cost-benefit model of the adoption of a high level of fire safety in TV sets. Full information concerning the application of the Fire-LCA model to TV sets is available elsewhere and will not be provided here[9, 10].

2.3 Cost-Benefit Approach

The holistic, LCA-based, approach described above is the basis for the cost-benefit analysis (CBA) model developed in this project. This means that whereas in a traditional Fire-LCA one focuses on the emissions and energy requirements for each module this analysis will focus on the costs (positive and negative) associated with the product life-cycle. In the same way that a Fire-LCA required the definition of a functional unit we will introduce the concept of a functional unit into the CBA. This application of the CBA will focus on the costs and benefits associated with different levels of fire safety. To emphasise this we will refer to this as the Fire-CBA model.

The costs associated with the functional unit will include such things as raw material for the production, possible costs associated with the use of the product and, finally, costs associated with destruction.

In this particular analysis the base-line will be defined as the choice of a minimum level of fire safety and the cost and benefit of an incremental increase in fire safety will be calculated. In many cases the costs associated with production and use will be equivalent and therefore excluded from the comparison between products A and B. This will reduce the data collection required significantly and provide the incremental cost and benefit needed for sound decision making.

Table 2 shows the various parts of the full life-cycle and whether the costs and benefits need to be calculated to obtain a full understanding of the incremental change caused by the choice of level of fire safety.

Table 2: Collation of requirements for Fire-CBA per module.

Module	Comment
Production	Any additional costs associated with production should be included in this part of the CBA. In some cases the use of alternative material can require significant investments by industry which should be included in the cost. (Note: in the TV case study presented in the next chapter, drop-in technology is assumed. In this case only the incremental cost of the addition of the FR will be included in the Fire-CBA.)
Use	The difference assumed in the Fire-CBA do not impact on the costs associated with use and this will not be included.
Transport	As for use the costs associated with transportation of the functional unit would not be expected to change through the introduction of flame retardants.
Destruction	Additional costs associated with specific destruction plans could be included in this module, e.g. cost program in The Netherlands associate with end-of-life destruction of consumer products. Alternatively one could consider a worst case scenario with dedicated, isolated stream destruction of materials containing “hazardous” chemicals.
Fires	The cost of extinguishment, sanitation, treatment of injuries and possible deaths should be included in the costs of fires. Indeed one of the major benefits of the use of a high level of fire safety is the avoidance of fires, reduction in the size of fires that occur and reduction in injuries and loss of lives.

Clearly there are some significant problems associated with determining the costs and benefits associated with the use or avoidance of flame retardants. One of the most important issues is associated with the valuation of a life and whether this should be variable based on the age of the life lost. This specific issue is dealt with in the next section.

One aspect that is not covered in the modular treatment shown above is the cost associated with exposure to a dangerous chemical. In the case that a flame retardant could be seen to cause death or injury this should be included in the CBA. This will be dealt with in relation to DecaBDE in the next chapter.

2.3.1 Value of a Statistical Life (VSL)

The use of CBA has become more accepted in the realm of environmental and other health and safety related legislation. The primary benefit of many important environmental requirements, as determined by the dollar value of CBA, is the human lives that are saved. Thus, in determining whether a particular regulation can be justified, the central issues often revolve around the value assigned to the lives that would be saved by a specific program[32]. While the concept of placing a value on mortality risks can be seen as immoral[33] it is a necessary prerequisite of any CBA. In recognition of this a great deal has been learned about VSL over the past 30 years[34].

Estimates of the cost of legislation range from US$200 000 per statistical life saved with the US EPA's 1979 trihalomethane drinking water standard to more than US$6,3 trillion per statistical life saved for the EPA's 1990 hazardous waste listing for wood-preserving chemicals[35], US$100 000 per life saved for steering column protection regulation and US$72 billion per life saved for formaldehyde regulation[36]. Naturally, in the first case we have an example of a regulation with the potential to save many lives while in the second case we see a regulation which may be important but which has the potential to save few lives. A question one must ask then in conducting a CBA is whether it is reasonable to introduce legislation when the cost per statistical life is so high. This is, indeed the basis of CBA requirements prior to legislation.

Once legislation has been introduced it is possible to conduct a post-legislation CBA to determine the cost of a statistical life saved. When using CBA to rationalise proposed legislation one must determine an acceptable value for a statistical life and calculate whether the lives saved can justify the investment associated with the regulation.

The literature contains a wide range of proposed values of a statistical life (VSL). Table 3 contains a summary of the various VSL's that have been found and their application. The values summarised in this table are based on a Willingness to Pay (WTP) philosophy[34, 37]. Situations in which risk is, at least partially, a matter of choice provide opportunities to analyze behaviour and estimate the WTP for risk reductions (e.g., through a higher price for a product) or the Willingness to Accept (WTA) compensation for risk increments (e.g., reduction in the price of a product, or increase in payment for a risk filled occupation).

Table 3: VSL from a variety of studies (reproduced and slightly modified from reference 34).

VSL (year 2000, million US$)	Behaviour and tradeoff	Reference
$1,7	Speeds and fatalities on interstate highways with higher speed limits, 1982-1993	Ashenfelter and Greenstone (2002)[38]
Adult: $4,3 Child: 3-9: $2,9 10-14: $2,8	Based on bicycle helmet use with fatality risk reductions and costs, 1997	Jenkins, Owens and Wiggins (2001)[39]
Adult: $2,8-4,6 Child: <5: $3,7-6,0 Motorcyclist: $1,7-2,8	Based on car seat best use with fatality risk reductions and time and disutility costs, 1983	Blomquist, Miller and Levy (1996)[40]
Child, <5: $0,8	Child safety seat use with fatality risk reductions with time and money costs, 1985.	Carlin and Sandy (1991)[41]
Adult: $7,2 Child: $7,3 Elderly: $5,2	Hedonic analysis of motor vehicle prices with fatality risks, 1995	Mount et al. (2001)[42]
$3,8-$5,4	Hedonic analysis of car prices with fatality risk, 1988	Dreyfus and Viscusi (1995)[43]
$5,3	Hedonic analysis of car prices with fatality risk, 1978	Atkinson and Halvorsen (1990)[44]
$4,7[a]	Hedonic analysis of housing prices with fatality risk near Superfund Sites, 1998-1993	Gayer, Hamilton and Viscusi (2000)[45]
$5	Average value used by the CPSC when conducting CBA's as part of their rule making process.	CPSC report 2005[46]

[a]Value for a statistical cancer case.

While the values included in Table 3 correspond to VSL for a risk averted now, a life-cycle approach must take into account the fact that averting behaviour can be used now to change mortality risks in the future. This is generally done through discounting of future lives relative to present lives. In 1980, the US EPA Office of Management and Budget (OMB) strongly urged discounting the value of human lives over the period of latency of the harm[32]. At that time they recommended a discount rate of 10% but more recent CBA by the US EPA use a more moderate discount rate of 3%[46].

Discounting issues play an even more critical role in connection with harms to future generations, e.g., from climate change. Indeed, when time horizons are very long, all benefits are discounted to zero using any positive discount rate, so that a death prevented in the distant future is worth nothing at the present time. This is particularly relevant in the context of chemical exposure where the toxicological risk is of low (possibly negligible) frequency and any toxicological result (i.e. death or injury) is located in the uncertain and possibly distant future.

To avoid an unnecessarily conservative approach to the CBA presented in the TV application in this report, the results will be presented both with and without discounting.

2.3.2 End of life programs

Disposal of products that have finished their useful life-cycle can incur a variety of costs depending on the product and mode of disposal. Most of the information contained in this section has been extracted from the report "Description of Initiatives undertaken by (selected) European Countries in the field of WEEE management" produced by the Association of Cities and Regions for Recycling (ACRR)[47].

In a cost benefit analysis it is important to include any additional disposal costs due to the presence or absence of a flame retardant. In the case of a TV, it is covered by the requirements of the WEEE (Waste Electrical and Electronic Equipment) directive[48]. This directive outlines ways in which the product can be disposed of at the end of it's life cycle. In most cases the most interesting mode of disposal is "recycling". Unfortunately, recycling does not have the same meaning in different countries. For instance, in Norway, recycling includes reuse and both material recycling and energy recovery. In the Netherlands, recycling rates are defined as the proportion of materials not going to landfill or incineration.

In response to the WEEE directive a number of qualitative provisions have been set by EU countries for the management of electrical and electronic waste. These include:

- "environmentally sound treatment" (Denmark)
- "proper treatment in accordance with the regulation" (Norway)
- "no landfill without previous treatment" (Sweden, Switzerland)
- the prohibition of the incineration of products which have been taken back separately (the Netherlands)
- minimum standards for treatment like separated elimination of pollutants, recovery of metal and incineration of chemicals that cannot be recovered (Switzerland)
- the separation between re-usable and non-usable equipment (Belgium).

Most countries have developed mixed WEEE management systems based on the existing municipal management schemes in which municipalities organise collection of WEEE from households as well as the management of container parks and other collection points, while producers recycle and treat them.

In Belgium, 80% of collection points are points of sale, but they only capture 25% of the volume of waste collected, while municipal collection points (20%) capture 75% of WEEE. In the Netherlands, distribution channels (retailers and distribution centres) collect about 13% all the WEEE collected. In both countries, retailers have an access to recycling parks and RTS for small quantities of WEEE.

In most cases (with the notable exception in Switzerland), these systems are financed by local taxes for municipal collection infrastructures and/or consumers as far as further sorting, recycling and disposal costs are concerned. In Belgium, Sweden and in the Netherlands, collection costs are covered to diverse extents by producers through negotiated agreements with Local and Regional Authorities (LRAs). Producers finance part of municipal WEEE facilities (e.g. by providing specific containers), or a part of costs for transportation of WEEE from municipal collection points to Regional Transport Stations, or RTS (e.g. in the Netherlands).

If one considers recycling fees established by different producers' schemes, the part borne by final consumers appears to be higher in Switzerland (where producers set up their own management schemes), than in the Netherlands, where LRAs (compelled by the regulation) have put their collection facilities and RTS network at the disposal of manufacturers. Fees taken in some EU countries are summarised in Table 4. The source of the fees varies in that some are taken as part of the consumer price and some are extracted from taxes for waste handling.

Table 4: Summary of fees per EEE category in 4 EU countries (2004).

Product type	Belgium (RECUPEL) Fee (VAT included)	The Netherlands (NVMP)	Sweden (El-KRETSEN) Fee (VAT not included)	Switzerland SWICO/SENS Advanced Recycling Fee (VAT included)
Washing machine	10,00 €	5,-€	4,89 €	17,06 €
Coffee maker	1,00 €	3	0,4 €	0,68 €
Television	11,00 €	8,-€	0,87 – 6,52 €	10,24 €
Refrigeration appliances	20,00 €	17,-€		27,30 €

The maximum fee for television disposal is taken in Belgium while the minimum is the lower bound of the Swedish range. These fees will both be used to provide upper and lower bounds in the Fire-CBA presented in Chapter 4.

Appendix 2 contains more detailed information concerning the recovery and disposal of EEE in Belgium, Denmark, Germany, The Netherlands, Norway, Sweden, and Switzerland.

3 Evaluating a chemical risk

3.1 Mechanism of action and human relevance

With respect to the mechanism of action of DecaBDE two types of studies are relevant for the risk assessment.

The chronic National Toxicology Program (NTP) study[19] with DecaBDE (1986) is actually the only two year chronic study that has been performed to date, with a PBDE (polybrominated diphenylether). It is a classical chronic toxicity study with rats and mice that has provided quantitative information about the systemic effects of DecaBDE. The World Health Organization and Food and Agriculture Organization of the United Nations (WHO/FAO) (2002)[12] concluded that it was difficult to select a No Observed Effect Level (NOEL) from this NTP study, while the EU selected a No Observed Adverse Effect Level (NOAEL) of 1,120 mg/kg bw from this study. Nevertheless, this NTP study would be a suitable starting point when evaluating possible risks for long term background or occupational exposure of DecaBDE.

Apart from the chronic study mentioned above, several studies have been performed with different PBDEs, including DecaBDE, that addressed neurobehavioral effects after prenatal exposure in mice[20, 49, 50]. In these studies, it was observed that postnatal exposure to PBDEs 99, 153 and DecaBDE could affect the cholinergic nicotinic receptor system when administered during a critical postnatal period for brain development. In mice these effects could still be observed in adulthood. The NOAEL of DecaBDE in such a neurobehavioral study is apparently somewhere between 2 and 20 mg/kg bw[20]. The EU (2004)[18] concluded that a NOAEL of 2.22 mg/kg day may be derived from this study, but this value has too many limitations to be used without a confirmation or validation study.

With respect to the high concentrations measured in the brains of these neonatal mice it is unclear if these were in fact metabolites or parent compounds. However, in view of the fast metabolism in rodents of DecaBDE and its large molecular size the presence of metabolite(s) in the brain seems likely. If it is shown in future experiments that indeed metabolites of DecaBDE are transferred across the blood-brain barrier it must be elucidated whether the formation of the rodent metabolites are equally relevant for the human situation. With respect to hepatic cytochrome P450 composition, the major enzymes responsible for primary metabolism of PBDEs, there are significant differences between rodents and humans. Therefore, with the present state of the art, it is impossible to evaluate whether the possible role of metabolites in these neurobehavioral studies is of human relevance.

With respect to these neurobehavioral effects in rodents it should also be considered whether these, by themselves, are relevant for humans. The major penta (99) and hexa (153) PBDE have also been tested for similar neurobehavioral effects. DE 99 and 153 caused similar neurobehavioral effects in mice at approximately similar dose levels, ranging between 10 and 20 mg/kg bw[20, 50]. When reviewing the results from these three studies it is remarkable that for DecaBDE and DE 99 and 153 comparable quantitative results are found, irrespective of the degree of bromination.

Such a similarity in biological or toxicological response, irrespective of the degree of halogenation, is unusual when reviewing results from studies with other halogenated polyaromatics like dioxins, dibenzofurans and PCBs. For these compounds the toxicological responses almost invariably decrease with an increasing number of halogens[51, 52]. A very likely explanation for this lack of differentiation in quantitative response could be that the number of doses was too small with a possible large variation in individual responses causing a low statistical power. Whatever the explanation might be, the consequence is that the data available for DecaBDE are insufficient to determine a good quantitative dose – effect relationship from this neurobehavioral study that could be used for risk assessment. This conclusion is similar to that drawn by the EU in their recent risk assessment of this compound (2004).

In spite of the criticism that can be given on the studies above, the data do allow the derivation of preliminary NO(A)ELs, which have recently been summarized by Londen and Van den Berg 2005[15] (see Table 5). The information in the table has been derived from recent risk assessments by the American Chemistry Council's Brominated Flame Retardant Industry Panel (2002)[16], EU (2002, 2004)[17, 18] and WHO/FAO (2002)[12]. Most toxicologists agree that these studies can still be used to determine a first estimation of the margin of safety with respect to DecaBDE.

Table 5: NOAELs and LOAELs of DecaBDE derived from subacute, subchronic and chronic toxicity studies. NOAEL and LOAEL in mg/kg d unless stated otherwise. Adapted from references 12, 17, and 18.

Species	Duration treatment	NOAEL	LOAEL	Ref.
Mice	Single oral dose on PND 3, 10, 19. measurements after 2, 4, 6 months	PND 3 2.22 PND 10 and 19 20.1 (mg/kg bw)	PND 3 20.1 (mg/kg bw) -	Viberg et al. (2003)
	14 days	20,000	-	NTP 1986
	90 days	9,000	-	NTP 1986
	2 years	-	3,500	NTP 1986
Rats	14 days	7,500	-	NTP 1986
	28 days	80	-	Great Lakes 1977
	30 days	8	-	Norris et al. 1973
	90 days	3,350	-	NTP 1986
	2 years	1,120*	1,120**	NTP 1986

* For systemic toxicity
** For local effects

If one takes these preliminary NOAELs and LOAELs as a starting point for a risk assessment, the next question would be if the mechanism of action is relevant for the human situation. The in vivo studies, conducted with postnatal mice and several PBDEs, including DecaBDE, definitely indicate some kind of mechanism of action with respect to interaction on neurobehavior and its development. These studies[20, 49, 50] are merely descriptive in nature, and are therefore of limited quantitative value due to the lack of full dose – response relationships. Thus, they do not provide an answer concerning the human relevance of the data. However, in vitro studies with PBDEs provide more information about the possible mechanism(s) of action that might be responsible for the effects seen in vivo.

Individual BDEs 47, 99, 100 and 153 were studied in rat neuronal cultures and indicate that PBDEs are capable of changing protein kinase C and calcium homeostasis[53, 54]. These effects are very similar to those seen for *ortho* substituted (non dioxin like) PCBs. In contrast with the in vivo neurobehavioral studies with mice, potency differences between different PBDE congeners were found. The efficacy of, for example TetraDE 47, was found to be higher than the other, higher brominated PBDEs. The EC_{50} values for these PCBDs ranged from 35 µM upward, indicating that these effects should not be considered very sensitive and low level occurring. In addition, it is noticeable that effects and potency were similar for DE 47 and PCB 47, an *ortho* substituted, non dioxin like PCB[53, 54]. Comparable results have been obtained with commercial PBDE mixtures DE-71 and 79, but these have very little value for a specific risk assessment of PBDEs because of the likely presence of impurities of brominated furans and dioxins[55]. Consequently, the observed effects cannot be fully attributed to PBDEs only[53, 54, 56]. Overall, the firmest basis for a risk assessment of DecaBDE is therefore provided by the chronic NTP study (1986)[19].

In summary, it can be concluded from in vivo toxicological studies that PBDEs, including DecaBDE, can cause neurobehavioral changes in experimental animals, but the use of this information for risk assessment of the obtained in vivo dose – effect relationships has limited value from a quantitative point of view. In vitro studies indicate a mechanistic basis for the observed neurobehavioral effects that might very well be similar to non dioxin like *ortho* substituted PCBs.

The next question we must ask ourselves is whether the observed possible mechanism of action and data for neurobehavioral effects in experimental systems provide plausibility for neurobehavioral effects reported in human epidemiological studies. As the in vivo studies by Viberg *et al.*[20, 49], and Erikson *et al.*[50], focus on postnatal exposure during a sensitive period for brain development, epidemiological studies with neonates are most suitable for comparison. From the literature, there are no such studies available focusing both on PBDEs, particularly DecaBDE, and postnatal exposure.

However, studies about postnatal exposure and neurobehavioral effects are available for PCBs and dioxin like compounds from The Netherlands[57, 58, 59] and populations around the Great Lakes in North America[60]. Both studies indicate that neurobehavioral changes can be associated with higher maternal concentrations of PCBs or dioxin equivalents (TEQs). These effects appear to be long term, but reversibility has also been described[57-59]. In the case of these epidemiological studies it is difficult to discriminate the possible causal agents: dioxin like compounds including some PCBs, or *ortho* substituted PCBs, lacking a dioxin type of mechanism of action. Both types of compounds occur together in the food chain in a relatively stable ratio, making it impossible to attribute a specific effect to either group. In both studies PBDEs have not been included and consequently nothing can be said about a potential role of these compounds in the observed neurobehavioral effect in breastfed infants.

If the dioxin like compounds, including e.g. PCB 126, measured as TEQs are indeed responsible for this[57, 58, 59], it seems unlikely that PBDEs contribute, as these brominated flame retardants lack an Ah receptor mediated mechanism[55,61,62]. If the *ortho* substituted (non dioxin-like) PCBs are responsible for the neurobehavioral effects in neonates and infants, PBDEs could possibly contribute to the overall effect. However, this is merely a hypothesis because studies by Kodavanti *et al.* (2004) have indicated that there is a similarity in neurotoxic effects between *ortho* substituted PCBs and PBDEs[53]. Thus, it must be acknowledged that a biologically plausible mechanism of action exists for neurobehavioral effects of PBDEs in mammalian species that might be present in humans. However, this is not supported directly by results from epidemiological studies.

4 Evaluating a fire risk

4.1 Ignition sources

A large body of fire statistics is available world-wide concerning fires in audio-visual equipment. The available statistics are defined based on a variety of ignition sources. The available fire statistics from Europe and the US are then discussed in some detail before the specific presentation of the Fire-LCA TV fire model.

4.1.1 Internal

A recent and very thorough study, carried out by Sambrook Research International and commissioned by the UK Department of Trade and Industry (DTI)[63], identified the following causes of TV set fires, based on the historical record:

- Solder joints ageing causing arcing
- Mains switch, worn contacts
- Electromechanical stress in “heavy” components
- Overheating due to circuit component imbalances
- Capacitor failure (one design)
- Line output transformer
- Poor design of circuit layout (early TVs)
- Cathode ray tube (CRT)
- Mains lead
- Standby function, especially in old sets

While design of TVs has undoubtedly improved through the years, it remains an arduous undertaking due to the continually increasing complexity of these products. Indeed, the evidence shows that no design is totally safe. As reported in the DTI study, the history of television sets recalled by their manufacturers due to faulty design or construction, summarised in Table 6, testifies to this fact.

Table 6: Examples of TV Set Recalls, 1992-1997

Country	Manufacturer	Recall Year	Period of Manufacture	Number of Sets
Denmark	N/A	1992/3	N/A	40 000
France	Philips	1993	1983-1987	40 000
Germany	N/A	1989	N/A	200 000
Netherlands	Philips	1993	1983-1987	300 000
Sweden	Philips	1993	1983-1987	75 000
UK	Sony	1989	1985-86	N/A
UK	A	1993	1983-1986	21 models
UK	B	N/A	1986-1988	1 model
UK	C	1993	N/A	7 models
UK	D	N/A	>1992	2 models
UK	F	1993	>1992	2 models
UK	W	1993	1983-86	1 model
UK	Dixons/Matsui	1997	1993	"1 000's"

This table is indicative rather than comprehensive as no systematic record of TV set recalls is kept in any country. This example from the UK demonstrates that recalls are not uncommon.

The study concluded that faults not apparent at the time of manufacture and inevitable wear and tear present a fire hazard. Available statistics also indicate that fires in TV sets due to internal ignition sources are most common when the appliance is >10 years old.

4.1.2 External

Statistics usually exclude TV set fires if they are not clearly at the origin of the fire. The following external sources of TV set fires were identified in previous studies[63, 64]:

- Night-lights left burning without stands
- Christmas decorations
- Candles falling on the top or standing next to the set
- Lightning

The use of candles is particularly popular in Nordic countries. There is plenty of anecdotal evidence that consumers do not recognise the danger of placing a naked flame near a TV set, and when a fire occurs, the actual cause may not find its way into the statistics. One article[65] tells the story of a fire in a flat where the television had caught fire, but among the debris of the burnt television, traces of two tinned candles were found. The person who lived in the flat had not said a word about them when he explained how the television had "suddenly" burst into flames. A slight seasonal increase in TV set fires in December might be due to this tradition of setting naked lights (candles, paraffin lamps, etc.) on top of or close to TV sets.

Too often TV sets are treated like any other piece of furniture and decorated with a plant, a lamp or even a candle. TV sets can contribute significantly to the amount of combustible material available in a fire. It is estimated that a modern TV can contribute approximately 165 MJ to a fire. This is equivalent to 5 litres of gasoline.

4.1.3 Consumer misuse

Manufacturers and fire brigades inform consumers about the safe use of TV sets. They are warned against using the top of the TV set as a shelf for supporting vases, candles, or a cloth that could reduce ventilation. Consumers are warned about inadequate ventilation if the set is placed inside furniture. Nevertheless, there is evidence that most consumers do not read the manual for their TV sets, least of all the safety precautions.

Fire brigades indicate the following causes of fire due to consumer misuse[63, 64]:

- Lack of ventilation, especially when the TV sets are "boxed in" furniture
- Lack of maintenance, to remove accumulated dust (dampness can lead to electrical failure in case of dust accumulation)
- Extensive use of the standby function, especially by families with children

4.2 Fire Statistics

The criteria under which fires are counted as TV set fires can vary significantly from one country or from one statistics collecting organisation to another.

To compare statistics, the Sambrook study[63] defined a TV set fire as follows:

"A TV fire is a fire where the first point of ignition is from within the structure of the TV or ancillary equipment that forms a part of the TV, [such as] a video recorder or satellite system. [...] The resultant fire will have breached the envelope of the TV [...]. Specifically excluded are acts of vandalism, criminal damage, ignition caused by the use of accelerants and electrocution as a result of tampering."

This is in accordance with the safety standards as defined by IEC 60065 and is the definition used by National Electrical Safety Boards throughout Europe.

This definition tends to narrow statistics to fires of electrical origin, excluding most other causes. Significantly, fires that are contained within a TV set's enclosure are ignored, highlighting the important role enclosures play by providing the last barrier to any internal fire spreading outside the TV set. In addition, this definition excludes external causes such as candles.

Fire brigades and insurance companies, on the other hand, tend to report higher figures due to a broader definition of TV set fires that includes fires initiated externally. Insurance companies are generally more inclusive than other organisations in their definition of a TV fire. A recent detailed investigation of Insurance Company statistics in Sweden[64] found that approximately 50% of all TV fires as defined by insurance companies in Sweden would not qualify as TV fires according to the Sambrook definition. The discrepancy arises from the fact that fires confined only to within a TV set enclosure are included in the insurance company figures. Significantly, the Sambrook study has concluded that the occurrence of fires throughout Europe seems to be essentially the same (normalised per million TV sets) in each individual country. The Sambrook study relies on statistics from similar sources in each country. Assuming that the Sambrook conclusion is correct in indicating this similarity in fire behaviour the Swedish data can be used as a model for Europe.

At the time of the study by Sambrook the Swedish data were not available. Therefore, Sambrook has accounted for the inclusion of ‘fires’ due to external ignition sources, or due to incorrect classification of the type described above, by estimating these effects in each country studied. To this end they adjusted the reported rate of TV set fires in Denmark by subtracting 35-45% to account for fires involving candles, and for the lower rate of TV fires in smaller towns, which were extrapolated from the statistics of larger cities. An additional 25% was subtracted to account for small fires that self-extinguish. Similar adjustments were made for France (-15% and -25%), Germany (-34%), Italy (-33%), The Netherlands (-15%), Sweden (-20%), and the UK (-24%). The conclusions of the Sambrook survey suggest that about two thirds of the total number of TV set fires reported are due to internal/electrical causes and about one third to external causes. Based on their purposely conservative definition of TV set fires, Sambrook concludes that there are approximately 2208 fires in Europe per year, or 12.2 TV fires per million TV sets. They further conclude that another 6 TV fires per million TV sets are caused by external ignition.

Sweden is the first European country to make a concerted effort to reconcile the differences between fires statistics for TV fires from different sources. In the 1990’s the Insurance Federation reported approximately 6000 electrical fires per year. In 1994 (a typical year) approximately 42% of these were due to audio/visual equipment, the vast majority of which (>90%) were TV fires. This corresponds to approximately 2500 TV fires that year. At the same time the Swedish National Electrical Safety Board (SEMKO) officially estimated the total number of electrical fires to be less than 2500 (i.e., the number of TV fires according to the Insurance Federation) and the number of TV fires to be approximately 150-250 per year. In order to determine which number was most realistic an in-depth study was initiated centred around the Stockholm suburb of Vällingby. Over a 14 month period all electrical fires were investigated in detail by experts from SEMKO. The results of their findings were extrapolated to cover the whole of Sweden.

Two findings were particularly interesting. First, the Insurance federation grossly overestimated the total number of electrical fires and in particular the number of TV fires, and second, SEMKO had previously underestimated the total number of TV fires. Using SEMKO’s definition, the Vällingby study estimated that approximately 750 (or between 600-900) audio/visual fires occur per year in Sweden. These fires were all large enough to have breached the TV enclosure SEMKO concluded that the additional 1750 fires reported by the Insurance Federation were either wrongly classified, e.g., so small that they had not breached the enclosure, or were caused by an external ignition source. Assuming that approximately half of the Insurance Federation fires did not breach the housing would leave approximately 500 due to external ignition sources. These data correspond to approximately 100 TV fires/million TVs in Sweden due to internal ignition and 65 TV fires/million TVs due to external ignition, and 160 TV fires/million TVs where the fire does not beach to enclosure.

Usually, only the most severe TV set fires find their way into electrical safety board or fire brigade statistics. The authors suggest that the Vällingby project results, because of the thoroughness of the methodology, are more representative of a wider European reality. Understandably, consumers would have a financial incentive to report small TV set fires to insurance companies, while only in the event of a major fire would the consumer call the fire brigade. Therefore, it is not surprising that the Vällingby data are closer to Insurance Federation numbers than those reported in the statistics of fire protection agencies. Similarly, electrical safety boards are presumably only interested in fires of clearly electrical origin.

In conclusion, the Sambrook study provides a sound basis for comparison of fire statistics from different European countries, but it is too conservative in its estimate of the frequency of TV fires. The Vällingby data provided a better model for European TV set fire behaviour.

4.2.1 Fire-LCA TV fire model

As discussed above, the results show that a figure of approximately 100 TVs/million burn in Europe each year due to internal ignition sources and a further 65 TVs/million due to external sources. The distribution according to size of the fire is based on German results summarised in Table 7. At this point we will assume that statistics for European TVs can be related to TVs that do not contain flame retardants in the TV enclosure.

Table 7: Severity of TV set fires in Germany[64].

Severity	Frequency (%)	% used in model	# used in LCA model	Category in LCA model
Fire restricted to the TV	30-40	35	58	minor
Fire spread beyond the TV and causing damage to the property	40-60	53	88	full TV
Fire causing severe damage to the room and property	<5	5	8	full room
Fire causing major damage to the entire dwelling	<5	5	8	full house
Fire completely destroying the building	<2	2	3	full house

A further 160 TVs/million are classified as being involved in fires by insurance companies but the ‘fires’ are restricted to inside the TVs and correspond to the category of ‘minor’ primary fires in the LCA model.

It is reasonable to assume that any external ignition of TVs in the US must either pertain to a large external ignition source, or be due to the presence of a small but significant number of TV sets with HB enclosure material. This assumption is based at least in part on the results presented in the next chapter. Thus, to make the US statistics comparable to the European statistics one can assume that internal ignition will provide a high estimate of the number of fires associated with TV set housed with V0 enclosure material. This corresponds to a total of 5 TV fires/million TVs each year[66]. Again, based on experimental evidence of the fire behaviour of V0 enclosure material, one can assume that these fires are essentially minor with little damage to material other than the TV of origin.

4.3 Fire safety Context

In studying available statistics concerning TV fires in the construction of the fire part of the LCA model in this work, it quickly became apparent that TV fires cost lives. Based on their purposely conservative definition of TV set fires, Sambrook concludes that there are approximately 2208 TV fires in Europe per year, or 12.2 TV fires per million TV sets. They further conclude that another 6 TV fires per million TV sets are caused by external ignition. These results were based on information extracted from sources throughout Europe. Similarly, Sambrook estimated that a minimum of 16 people were killed, and 197 sustained injuries, each year in Europe as a direct result of TV fires[63].

As illustrated in the previous section, the Sambrook study provides a sound basis for comparison of fire statistics from different European countries, but it is too conservative in its estimate of the frequency of TV fires. The Vällingby data provided a better model for European TV set fire behaviour. These estimates are approximately a factor 10 higher than the Sambrook estimates. In terms of consumer fire safety this indicates that the number of deaths as a direct result of TV fires could also be a factor 10 higher, or closer to **160** deaths in the EC each year as a direct result of TV fires. Making a similar calculation for the number of people injured as a direct result of TV fires each year in the EC this number increases to **2000**.

These values will be used in the CBA application to TV sets to establish the possible benefit of the choice of a high level of fire safety through the potential saving of lives and avoidance of injuries.

5 TV (DecaBDE in enclosure) Case Study

5.1 Fire-LCA Application

There are two general concepts that were investigated in the scenarios presented by Simonson *et al.*[10]: the effect of brominated flame retardants and the effect of a different waste treatment in the future. The four scenarios thus form four different models. For the sake of this report only scenario comparison results are presented. Further, these results are only presented for the major species. Full results can be found in SP Report 2000:13[10].

The waste treatment is very different in different countries and it is, therefore, difficult to design a general waste treatment strategy especially for the future. As a base for the waste treatment strategies, however, an average of the OECD countries has been used. The scenarios are denoted “FR” or “NFR”, respectively, to indicate the use or non-use of flame retardants in the TV enclosure, and “today” or “future” to indicate a present day scenario for the waste treatment system or a possible future waste treatment system, designed for the year 2010. The use of flame retardants also changes the fire behaviour and thus different fire statistics and different fire models have been used for the different flame retardant scenarios. The TV waste treatment scenarios are described in Table 8 while the fire scenarios used can be found in Table 9.

Table 8 Waste handling scenarios for the TV sets in the models.

Activity	**Waste handling of today**	**Future waste handling**
TV sets to incineration	1 %	1 %
TV sets to disassembly (for recycling)	2 %	89 %
TV fire failed TV sets to incineration	1 %	1 %
TV sets to landfill	Remaining part except TV sets to fire	Remaining part except TV sets to fire

Table 9 Fire statistics used in the scenarios for NFR and FR TV sets.

Fire type	**Occurrence in NFR TV set use**	**Occurrence in FR TV set use**
TV fire failures	0.218 %	0.165 %
TV fires	0.088 %	0
TV sets in secondary fires	0.006 %	0.006 %
TV room fires	0.008 %	0
TV house fires	0.011 %	0

Let us also begin this analysis with a consideration of the energy use, as shown in Figure 2. From this figure it is clear that the relative variations in energy use is small between the different scenarios.

The small decrease in energy requirements seen in the future scenario is due to the recycling of material from the various end of life options. In all cases recycled material is assumed to reduce the requirements of virgin material for TV production.

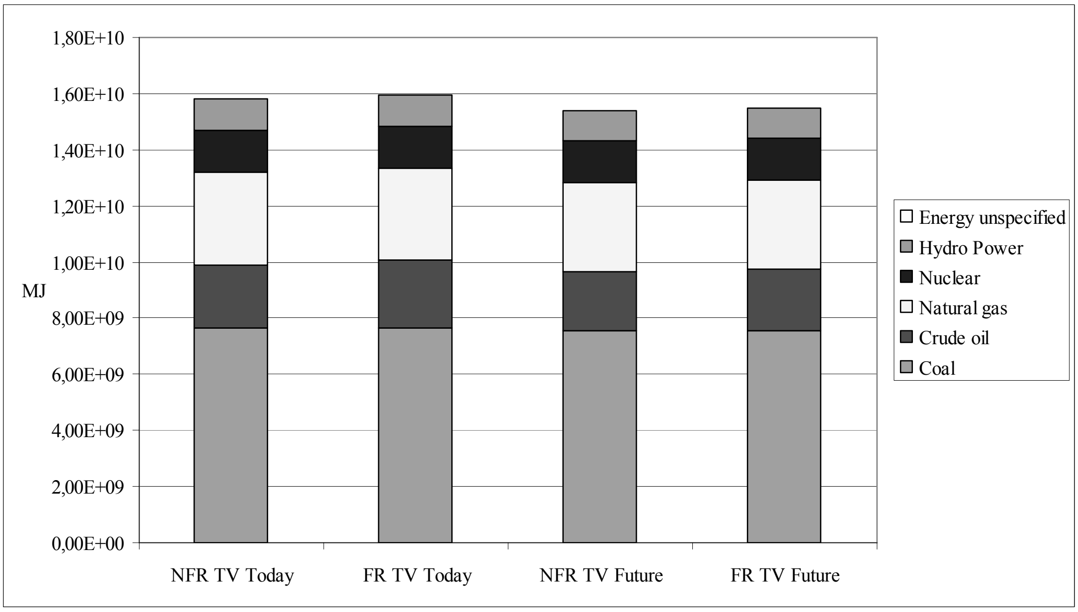


Figure 2 A comparison between the different scenarios show only small differences in the energy use.

The emissions of NO_x, SO_2, CO, CO_2 and particulates are only marginally different between the four scenarios and specific results are not presented here.

The emission of hydrocarbons shows a decreasing trend for the future scenarios, as seen in Figure 3. This is due to an increase in material recycling in the future scenario relative to the present day scenario. Further, there is a slight decrease in the emission of hydrocarbons due to the presence of flame retardants in the TV enclosure. This is due to the fact that the number of fires in the FR TV scenario is much lower than in the NFR scenario, with a correspondingly lower production of hydrocarbons from the fire part of the LCA model.

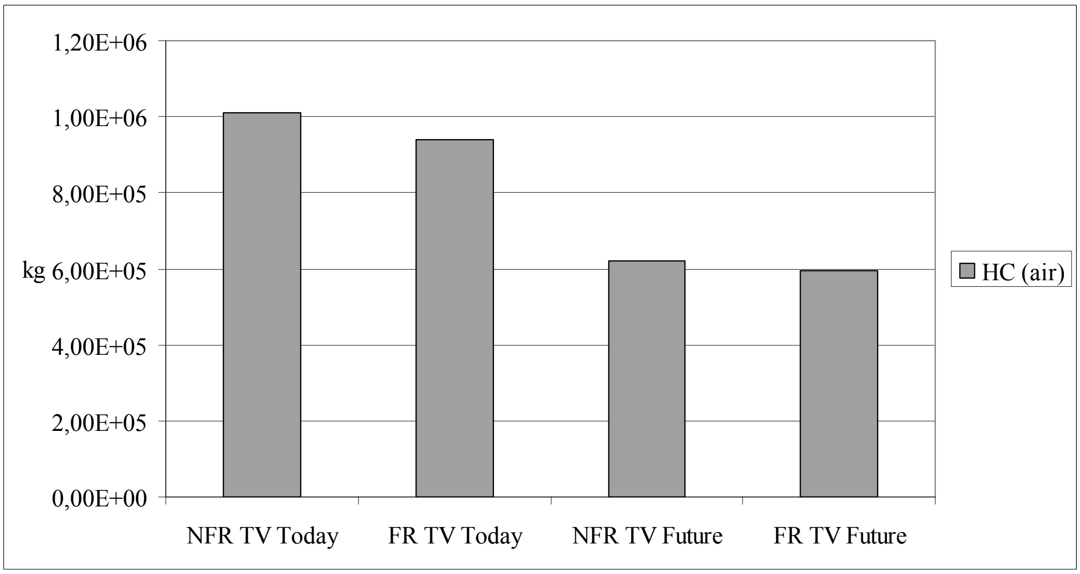


Figure 3 Comparison of hydrocarbon (HC) emissions for the four LCA scenarios.

The main PAH emissions originate largely from the different fires in the system. The use of flame retardants decreases the number of fires and, therefore, also the PAH emission, as seen in Figure 4. The difference between the present day and future scenarios is, however, small as the decrease in energy production only has a minor effect on the total PAH production.

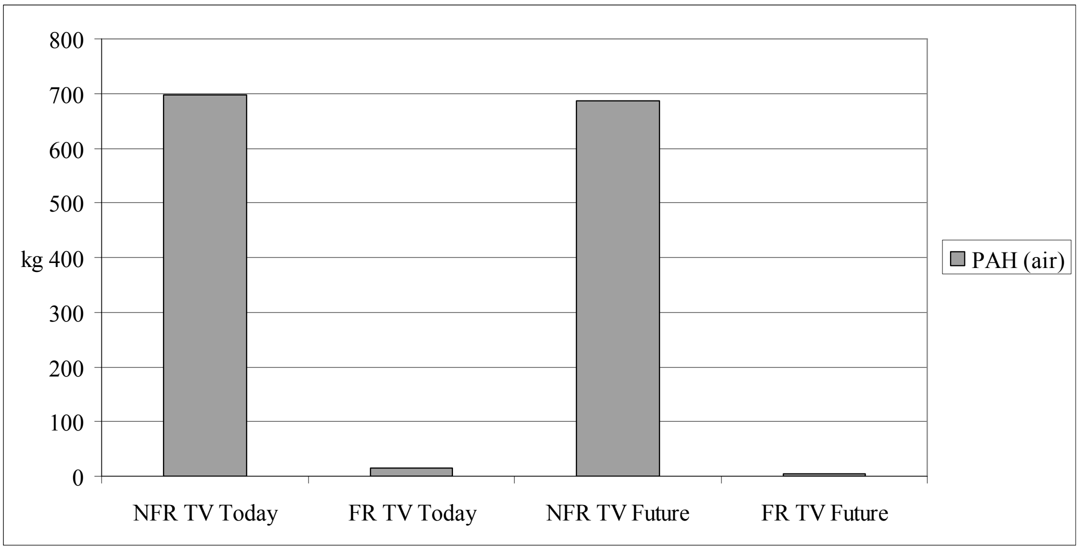


Figure 4 The main PAH emissions comes from the different fires in the system. The use of flame retardants decreases the number of fires and thus also the PAH emission.

One should also note that the production of PAH is many times higher than the production of all types of dibenzodioxins and furans. In light of the large amount of PAH produced throughout the TV life-cycles relative to dibenzodioxins and furans it is reasonable to conclude that PAH emissions represent a much greater risk to health and the environment that TCDD-equivalent and TBDD-equivalent emissions[67].

Chlorine and bromine in different materials form mainly HCl and HBr in different processes, such as combustion and fires. Only a small part of the chlorine and bromine load into the system, contribute to the formation of various chloro and bromo organic compounds. Thus, the dibenzodioxin and furan fraction, for example, is very small.

Figure 5 shows the emission of HCl, HBr and HCN. The emission of HBr is directly related to the use of flame retardants and thus higher for the FR scenarios than for the NFR scenarios. The increased HBr emission for the FR TV future scenario is the combined effect of bromine use together with a high degree of incineration. The emission of HCl is relatively constant between the different scenarios. The HCN emission is directly related to the number of fires. Thus, the emission is relatively equal between the different NFR scenarios but absent in the FR scenarios due to the small number of fires in those scenarios.

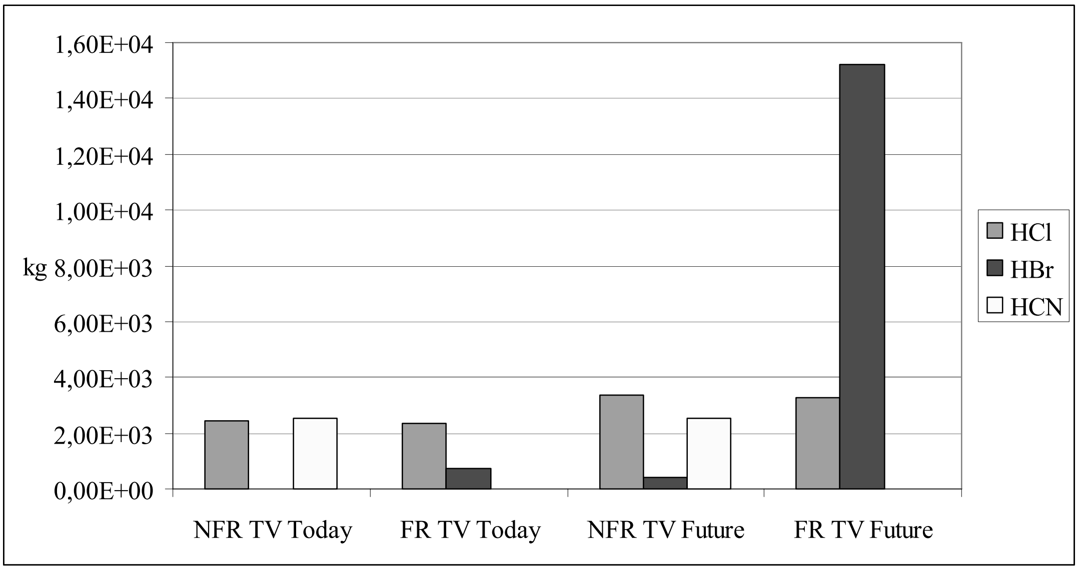


Figure 5 Comparison of HCl, HBr and HCN emissions for the four LCA scenarios.

The dibenzodioxin emissions are relatively equal for the NFR scenarios, as shown in Figure 6. The situation is, however, slightly different for the FR scenarios. The emissions in the FR TV today scenario are lower then the NFR scenarios due to a smaller number of fires. However, for the future scenario an increased emission can be expected due to an increased use of incineration.

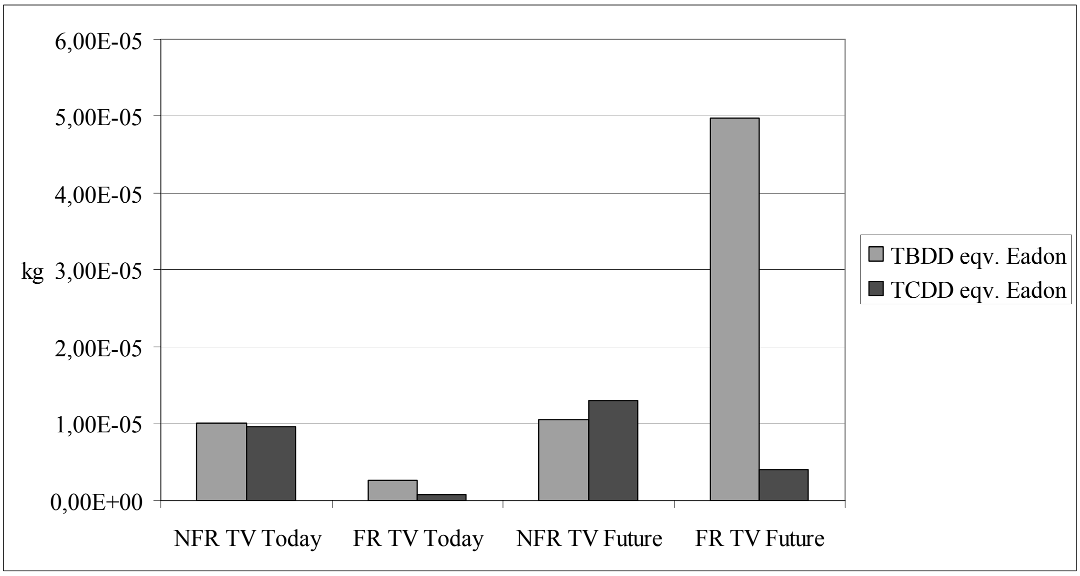


Figure 6 Comparison of dibenzodioxin equivalent emissions for the four LCA scenarios.

An allocation based on chlorine and bromine has been used for the incineration calculations. As discussed previously, very little experimental input information is available concerning the emissions of dibenzodioxins and furans from incineration. Thus, the input used is an estimate and should be interpreted as an upper limit rather than an absolute value.

In this study the impact of actual lives saved and injuries avoided cannot be included. The next section aims to remedy this situation through the development of a Fire-CBA (Fire Cost Benefit Analysis) model with a specific application to a TV set.

5.2 Fire-CBA Application

As stated in Chapter 2 there are a number of important costs that must be included to take into account the financial cost and benefit of any given proposed legislation. In the case of the application presented here it is not directly legislation that is studied but the use of a specific flame retardant (DecaBDE) to reach a high level of fire safety. This high level of fire safety is compared to a TV set with no flame retardant in the outer enclosure. Both of the TV sets are compliant to regulations in Europe but only the TV containing DecaBDE in the outer enclosure is compliant in the USA.

The costs outlined in Table 2, Chapter 2, will be dealt with in separate sub-sections below.

The TV sets used in this application will be the same as those used in the Fire-LCA TV Case Study. Detailed information concerning their composition and physical dimensions can be found elsewhere[10] and is not given here.

5.2.1 Production

The two TV sets are assumed to be identical in all respects apart from the presence or absence of DecaBDE in the outer enclosure. DecaBDE in HIPS (high impact polystyrene) is a so called drop-in technology. This implies that no additional investment is necessary for the manufacture of the TV enclosure. The manufacturer can choose to buy resin with or without DecaBDE and use the same machinery and manufacturing process to produce housing of type UL 94 V-0 (with DecaBDE) or UL 94 HB (without DecaBDE).

This implies that in the present application of the Fire-CBA model the only additional cost for the use of high fire performance material is the incremental difference in cost between the two resins. This type of information is generally confidential. We have been able to obtain a generic cost that has been used by the EU when preparing their risk assessment for DecaBDE which were based on the public work by Stevens and Mann[68]. This data suggests that the cost of adding a sufficient amount of DecaBDE to the HIPS resin would be approximately €4,40/kg, i.e. approximately US$5,30/kg.

Assuming that approximately 0,72 kg DecaBDE is present in each TV housing corresponds to a total addition cost of US$3,80/TV set. This value will be used in the cost side of the Fire-CBA calculation later.

Note that were one to use an alternative flame retardant that is not a “drop in” replacement the cost of investments in machinery and education of personnel should be included in the Fire-CBA.

5.2.2 Use

There is no additional cost during the use part of the TV life-cycle as the cost of use of the TV is the same independent of the fire performance of the TV set.

5.2.3 Fires

There are numerous costs associated with fires. These include:

- Fire fighting
- Post-fire clean-up
- Replacement of destroyed or damaged equipment
- Treatment of fire victims that do not die
- Societal losses due to fire victims that die.

This study will not attempt to include the cost of fire fighting or post-fire clean-up due to the lack of reliable data concerning these costs. Further, this study will not include the cost of replacement of destroyed or damaged equipment due to the difficulties involved in obtaining reliable data. At some later stage one could base such a calculation on insurance tables.

Thus, the benefits that will be included in this direct application of the Fire-CBA model will be the cost of lives saved through the avoidance of TV fires and treatment of injuries. A number of assumptions are necessary to allow the calculation of a total cost associated with the adoption of a low level of fire safety in a TV set. In order to make the necessary calculations three main assumptions are necessary: the value of a statistical life (VSL), the cost associated with treatment of injuries, and the discounting rate for future benefits.

In addition to these main assumptions it is necessary to establish the number of lives saved and injuries avoided each year, the total number of TV sets in Europe and the life of a TV set.

The number of lives saved through the use a high fire performance material in the TV enclosure was estimated to be 160/year in Europe each year in the Fire-LCA TV Case Study. Similarly, the number of injuries avoided was estimated, in that study, to be approximately 2000/year in Europe. This data will be used together with a VSL of US$5 million[46] and an average cost to treat a burn victim of US$200 000[69]. The discounting rate used has a significant effect on the results. Discounting up to 10% has been used in extreme cases[32] although 3% was used in the recent CPSC CBA of the proposed mattress regulations in the US[46]. To determine the sensitivity of the calculations to the discounting rate both values have been used in this study.

The average cost of a house fire was based on the average price of houses in March 2005[70].

5.2.4 Chemical exposure

Based on the data and scientific arguments presented in Chapter 2 there is measurable human exposure to DecaBDE. However, various risk assessments for background or occupation and infant exposure via breast milk or household dust indicate that no adverse health effects are to be expected due to the large margins of safety that exist. The only exception can be found for occupational settings when using the neurobehavioral effects found in neonatal mice[20]. However, it is extremely doubtful if such an experimental design using neonates is applicable for adult healthy workers.

Consequently, the cost of exposure and lack of expected associated adverse health effects to DecaBDE can be considered zero in this risk – benefit analysis.

5.2.5 End of life

A variety of recovery and recycling programs exist both in Europe and the US in response to environmental concerns connected to electrical and electronic equipment waste. The majority of schemes in the US are provided free of visible charge to the consumer as part of company stewardship programs.

In Europe the cost of disposal varies between the different European countries. A range of costs will be used in the Fire-CBA, i.e., 11 € per television (Belgium) or approximately US$13,30 per television, as this is the worst case in Europe; and 0,87 € per television (Sweden) or approximately US$1 per television, as the best case in Europe.

5.2.6 TV Life-Cycle

As in the case of the Fire-LCA TV case study it will be assumed that a TV has a working life of 10 years. To test the sensitivity of the model to this parameter both 10 year and 15 year life-cycles will be used in the CBA. A 10 year life cycle implies that 10% of the product base are replaced each year, i.e. 10% new TVs are produced and 10% old TVs are disposed of.

5.2.7 TV park in EU

An important parameter to determine for the calculation of the full cost benefit is the number of TV sets in the EU. The number of deaths and injuries has been estimated for the whole of the EU which must be related to the number of TVs replaced each year. It has not been possible to find industry estimates of the size of this market in the open literature. We do know from previous work, however, that the number of TV sets in Sweden is approximately equal to the internet penetration (i.e. the percentage of the population with access to the internet)[28].

Table 10: Summary of Internet penetration in the EU[71].

EUROPEAN UNION	Population (2005 Est.)	Internet Users, Latest Data	Penetration (% Population)
Austria	8 163 782	4 650 000	57.0 %
Belgium	10 443 012	5 100 000	48.8 %
Cyprus	950 947	298 000	31.3 %
Czech Republic	10 230 271	4 800 000	46.9 %
Denmark	5 411 596	3 762 500	69.5 %
Estonia	1 344 840	670 000	49.8 %
Finland	5 246 920	3 286 000	62.6 %
France	60 619 718	25 614 899	42.3 %
Germany	82 726 188	47 127 725	57.0 %
Greece	11 212 468	3 800 000	33.9 %
Hungary	10 083 477	3 050 000	30.2 %
Ireland	4 027 303	2 060 000	51.2 %
Italy	58 608 565	28 870 000	49.3 %
Latvia	2 306 489	810 000	35.1 %
Lithuania	3 430 836	968 000	28.2 %
Luxembourg	455 581	270 800	59.4 %
Malta	384 594	301 000	78.3 %
Netherlands	16 322 583	10 806 328	66.2 %
Poland	38 133 691	10 600 000	27.8 %
Portugal	10 463 170	6 090 000	58.2 %
Slovakia	5 379 455	2 276 000	42.3 %
Slovenia	1 956 916	950 000	48.5 %
Spain	43 435 136	16 129 731	37.1 %
Sweden	9 043 990	6 800 000	75.2 %
United Kingdom	59 889 407	37 800 000	63.1 %
European Union	**460 270 935**	**226 890 983**	**49.3 %**

It has been possible to find information concerning internet penetration throughout Europe relative to EU population. This information is summarised in Table 10. Based on the data in Table 10 it is clear that approximately 50% of the EU population have access to the internet. If we equate this to the number of TV sets in the EU this corresponds to approximately 230 million TV sets in the EU.

5.2.8 Results

The data summarised in sections 4.2.1 to 4.2.7 provide the basis for the calculation of the costs and benefits associated with the use of DecaBDE in TV set housings to provide a high level of fire safety. The information contained in these sections is summarised in Table 11.

Table 11: Summary of data used in scenario calculations.

Parameter	Value	Comment
Cost of DecaBDE	US$3,80 per TV set	Nominal cost for drop in technology[68]
Additional cost for use of FR TV set	--	Assumed there is no additional cost for using an FR TV set
VSL	US$ 5 000 000	Based on CPSC data[46]
Average cost for treatment of burn victims	US$ 200 000	Still *et al.* [69]
Discounting rate	3%, 10%	Two different discounting rates applied to test sensitivity of model[32, 46]
TV life-cycle	10 yrs	Based on data from Fire-LCA model[10]
Number of TV sets in Europe	230 million	Estimate based on Internet penetration in EU
Deaths avoided per year	160	For whole TV population. When TV has 10 yr life cycle, 10% of this number are saved for each of the next 10 yrs[10].
Injuries avoided per year	2000	For whole TV population. When TV has 10 yr life cycle, 10% of this number are avoided for each of the next 10 yrs[10].
# full house fires avoided	11/million TV sets	Based on data from Fire-LCA model[10]
Average cost per house	US$180 000	Based on the average price of a house in Sweden in march of 2005[70]
# TV fires avoided (including house fires	107/million TV sets	Based on data from Fire-LCA model[10]
Average cost per fire	US$7500	Based on statistics from the Swedish Insurance Federation. This assumes that all fires are equal, i.e. small fires and large fires are weighted equally in the statistics[72].

The input data in Table 11is used to run the CBA calculations for a number of different scenarios to test the robustness of the results. Table 12 summarises the various scenarios tested and Table 13 summarises the results of the CBA calculation.

Table 12: Summary of Scenarios for CBA calculation

Scenario	Life cycle (yrs)	Discounting	Cost of disposal	Cost of house fires included	Insurance costs included
1	10	No discount	US$1	No	No
2	10	3%	US$1	No	No
3	10	3%	US$13,3	No	No
4	10	3%	US$1	Yes	No
5	10	3%	US$1	indirectly	Yes
6	10	3%	US$13,3	Yes	No
7	10	10%	US$1	Yes	No
8	10	10%	US$13,3	Yes	No

In all scenarios, a 10 year life cycle was assumed for the TV sets. This is important as it means that 10% of the TV sets are replaced each year and benefits and costs are cumulative over the full 10 year life cycle. Three different discounting scenarios were tested (0%, 3% and 10%) to illustrate the significance of discounting rate on the impact of future lives saved or costs accrued. Two costs of disposal were used to illustrate the significance of this cost in the analysis. These two values (US$1 and US$13,3) were chosen to represent extremes present in the EU. In many cases there is no direct cost to consumers for recycling which would significantly increase the impact of the overall benefit of the avoided fires, fatalities and injuries.

Finally, the capital cost of fires (for replacement of property) is difficult to determine and has been included to a varying degree in the various scenarios. In the most conservative case the capital cost of fires has not been included. This is true of scenarios 1 – 3. Scenarios 4 and 6 – 8 include the capital cost to replace full house fires to the degree suggested in the Fire-LCA TV case study while scenario 5 includes a nominal cost per fire for all TV fires, again based on the data available in the Fire-LCA TV case study. In this case the full house fires are included indirectly as one of many types of fires.

Table 13: Results from CBA calculation.

Scenario	Gross Cost (million US$)	Gross Benefit (million US$)	Net Benefit (million US$)
1	110	1 200	1 090
2	110	1 050	940
3	393	1 050	657
4	110	1 490	1 380
5	110	1 210	1 100
6	393	1 490	1 097
7	110	1 078	968
8	393	1 078	685

6 Conclusions

A Cost-benefit model has been developed to evaluate the monetary impact of regulations aimed at the removal of flame retardants. This model has been constructed to include such costs as: incremental increases in cost to flame retard a product relative to a non-flame retarded product; additional costs for disposal of the product at the end of the product life cycle. Similarly, the model includes provisions for benefits such as: lives saved, injuries avoided, capital costs avoided through fires averted.

No cost for injuries (either to humans or the environment) due to exposure to flame retardants has been included in the TV Case Study as there was no indication that such costs exist for DecaBDE. Further, no cost has been included for anxiety due to perceived risks associated with flame retarded products.

The incremental cost of the flame retardant and recycling of the flame retarded product have been taken from international studies conducted with their base in the EU. Costs associated with lives saved and injuries avoided are based on international praxis concerning the value of a statistical life and the cost of treatment of a burn victim. Further, estimates of the number of lives saved and injuries avoided have been based on statistics for the EU. Finally, the capital costs saved through fires averted has been mainly based on European Swedish data from the Fire-LCA study and the Swedish Insurance Federation.

In all, a total of 8 scenarios were tested for the TV set application of the Fire-CBA model developed in this report. In all cases the benefits of a high level of fire performance in a TV set far outweigh the costs associated with obtaining that high level of fire safety. The net benefit is a function of the choices made in the various scenarios but ranges between 657 to 1 380 million US$ per year.

The various scenarios were chosen to illustrate the significance of the parameters included in the study as the specific value chosen for each parameter can vary depending on the assumptions made in the model.

Appendix 1: Occurrence of DecaBDE in human blood samples (concentration ng/g lipid).

Reference	**Year**	**Tissue**	**Studied group represents:**	**Male /female**	**Median age**	**Positive samples**	**Average**	**Median**	**Maximum**	**Minimum**	**LOD**	**LOQ**
Schecter (2005)[73]	1973	Serum	General Population USA	NA [x]	NA [x]	0 (0%)	NA	NA	<LOQ	ND	1	NA
Sjödin et al (2001)[74]	1988	Serum	Blood Donors USA	12 [v]	NA	5 (42%)	NA	<LOQ	33,53	<0,958	BA	0,958 [xii]
Sjödin et al (1999)[75]	1997	Serum	Hospital cleaners Sweden [viii]	0/20	48	14 (70%) [ii]	NA	<LOQ	3,7	<0,3	0,29	0,67
			Computer clerks Sweden	0/20	54	13 (65%) [ii]	NA	<LOQ	7,7	<0,3	0,29	0,67
			Electronic dismantlers Sweden	15/4	46	19 (100%) [ii]	NA	4,80	9,5	<0,3	0,29	0,67
Thuresson (2004) [76]	1999	Serum	Test facility workers Sweden	5	NA	5 (100%)	NA	2,9	5,6	1,4	NA	5×average blank or S/N ratio >10 [xi]
	1999	Serum	Shredder workers Sweden [ix]	2	NA	2 (100%)	NA	NA	2,4 – 5,2	NA	NA	5×average blank or S/N ratio >10
	1999	Serum	Smelter workers Sweden	2	NA	0 (0%)	NA	NA	NA	<LOQ	NA	5×average blank or S/N ratio >10 [xii]
Thuresson (2004)	1999	Serum	Computer technicians Sweden	15/4	35	9 (47%)	NA	1,53	6,8	<LOQ	NA	S/N ratio >10
Thuresson (2004)	2000	Serum	Abattoir workers Sweden [viii]	18	24-60	18 (100%) [iii]	NA	2,39	9,29	0,88	NA	<0,86
	2000	Serum	Rubber workers Sweden [iv]	19/1	24-60	20 (100%)	NA	28,62	268,24	1,15	NA	<0,86

Thuresson (2004)	2000	Serum	Electronic dismantlers Sweden (Lab A)	11	47	NA	NA	1,92	5,17	<0,958	NA	0,958
	2000	Serum	Office employees Sweden (Lab A)	2	NA	2 (100%)	NA [vi]	NA [vi]	1,82 – 5,17 [vi]	NA [vi]	NA	0,958
	2000	Serum	Electronic dismantlers Sweden (Lab B)	9	49	NA	NA	NA	<LOQ	ND	NA	2,87
	2000	Serum	Office employees Sweden (Lab B)	4	NA	NA	NA	NA	<LOQ	ND	NA	2,87
WWF (2003)[13]	2003	Serum	General Population UK	50/105	40,5	11 (7%)	NA	83	241	35	35 [i]	NA
WWF (2004)[14]	2003	Serum	Members of the European Parliament, 17 countries	24/23	52	16 (34%)	NA	57	2400	28	2	NA
WWF (2004)	2004	Serum	Ministers from 13 European countries	11/3	NA	3 (21%) [xiii]	3,9	NA	6,75	1,65	NA	0,005
WWF (2004)	2004	Serum	General population UK	21/12	[xv]	7 (21%)	NA	14,9	33,9	0	NA	NA
Peters et al. (2004)[61]	2004	Serum	General population The Netherlands	48/43	19-78	11 (12%)	NA	46,1	291,6	22,7	NA	22,5
Fångström (2005) [77]	1994 /1995	Serum	Mothers Faroe Islands	0/57	NA	NA	NA	1,9	9,0	0,47	NA	NA
	2001 /2002	Serum	Children Faroe Islands	42	7	NA	NA	2,5	15,3	0,57	NA	NA
Schecter (2005)	2003	Whole blood	General population USA	NA [xi]	NA [xi]	NA	NA	NA	1,4	NA	NA	NA
	2003	Whole blood	General population USA	22/17	45	18 (46%)	2,8	2,55	6,1	0,96	1-15 [xi]	NA
Takasuga et al. (2004)[78]	1998-2000	Blood	General population Japan	9/9	37-48	102 (65%) [viii]	9,2	6,9	31	1,3	1	0,1
Lopez et al. (2004)[79]	NA	Plasma	Mexican women	0/5	NA	5 (100%)	9,5	NA	14,6	4,8	NA	NA
Ryan and Patry (2001)[80, vii]	1992	Milk	Mothers' milk Canada	0/72	NA	NA	NA	NA	NA	NA	NA	NA
Schecter et al. (2003)	2001	Milk	Mothers' milk USA	0/23	26,6	7 (26%)	0,92	NA	8,24	0,48	NA	Sample level 2× blank level

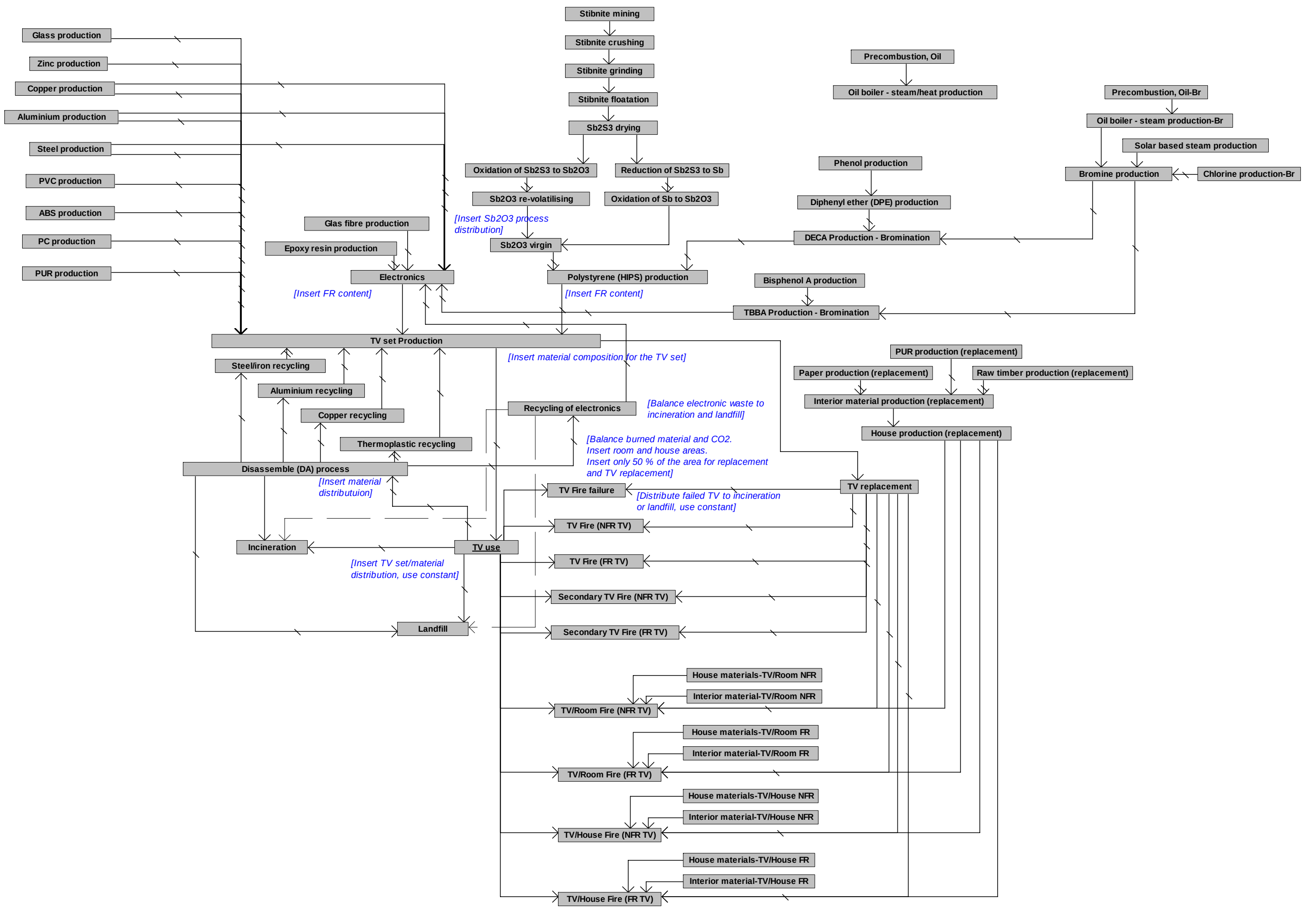
Vieth et al. (2004)[81, xiv]	2001-2003	Milk	Mothers' milk Germany	0/62	NA	25 (40%)	0,17	0,1	1	0,1	NA	0,1
	2001-2003	Milk	Mothers' milk Germany	0/31	NA	NA	0,11	NA	NA	NA	NA	0,1
Vieth et al. (2004)	2001-2004	Milk	Lactating women Germany	0/73	31,8 (18-44)	34 (47%)	0,17	0,10	1,0	<0,1	NA	0,1
Lunder and Sharp (2003)[82]	2002/2003	Milk	Mothers' milk USA	0/20	33	16 (80%)	0,24	0,15	1,23	0,08	NA	>2×99% confidence level noise
She et al. (2004)[83]	2003	Milk	Mothers' milk USA	0/16	NA	16 (100%)	0,4	0,25	1,5	0,05	NA	NA
Lopez et al. (2004)	NA	Milk	Mothers' milk Mexico	0/7	NA	7 (100%)	0,3	NA	0,6	0,1	NA	NA
	NA	Milk	Mothers' milk Sweden	0/5	NA	5 (100%)	0,4	NA	0,4	0,3	NA	NA
Polder et al. (personal comm.)	NA	Milk	Mothers' milk Norway	0/23	NA	NA	0,29	0,16	1,91	0,08	<0,01	NA
Fångström et al. (2004)[84]	1987	Milk	Mothers' milk Faroe Islands	0/10	20-29	2 (100%)	NA	NA	0,59 [xviii]	NA	NA	0,14
	1994 /1995	Milk	Mothers' milk Faroe Islands	0/10	20-29	2 (100%)	NA	NA	0,47-0,55 [xviii]	NA	NA	0,14
	1999	Milk	Mothers' milk Faroe Islands	0/10	20-29	2 (100%)	NA	NA	1,1-1,3 [xviii]	NA	NA	0,14
	1999	Milk	Mothers' milk Faroe Islands	0/9	20-29	8 (89%)	1	0,6	3,2	<LOQ	NA	0,14
Stanley et al. (1991)[85]	1987	Adipose tissue	General population USA	5/18	NA	3 (60%)	NA	NA	0,7	<LOD	NA	NA
DeCarlo (1979)[86]	NA	Hair	Barbershop near industry [xvi, xvii]	3/19	NA	1 (33%)	NA	NA	5-19	NA	NA	NA

Note: Since the lipid content in blood is about 0,65%, results for lipophilic compounds in pg/g serum may be converted into pg/g lipid by multiplying the former with a factor 150 (TNO 2004(3)) and in some articles the concentrations are expressed in pmol/g lipid this may be transferred to ng/g lipid by dividing the former by 0,958 (WWF 2004(2)).

i. Assumption that the detection limit is 35, the actual detection limit is not mentioned in WWF (2003)(1). WWF states that the samples containing BDE209 "were ranging in concentration from close to the detection limit to …". The lowest value found is 35 ng/g lipid and is therefore assumed to be close to the detection limit.
ii. In 7 cleaners, 6 clerks and 17 dismantlers the BDE 209 levels are above the LOQ.
iii. 17 results were used in the calculations, one subject was an outlier and was excluded.
iv. Rubber workers contain rubber millers and mixers, cable manufacturers (part sprayers, mantle sprayers, measurements and miscellaneous tasks). Data adapted from Thuresson et al (2004) (13). The figures found in 2000 were used.
v. Samples were taken in 1988, there is no demographic or questioner information available.
vi. The blood samples of the two office employees contained 1,82 and 5,17 ng/g lipid respectively.
vii. Little or no BDE 209 could be detected.
viii. Abottoir workers and hospital cleaners were referents with no occupational exposure and little if any computer experience.
ix. Blood from six male and three female workers was obtained but it is unclear which workers are female and which are male. The blood samples of the two schredder workers contained 2,4 and 5,4 ng/g lipid respectively, from the smelter workers also two samples were taken that were below LOQ.
x. n=100 pooled samples from archived serum (1973) and anonymous donors (2003).
xi. For each sample a LOD was calculated. The range is presented here.
xii. A S/N ratio greater than 10 was used to define the LOQ, when no interferences were present in the blank sample. If they were present the amount of analyte in the sample has to be at lease 5x the average blank level to be accepted.
xiii. Three ministers tested positive, the concentrations are: 1,65; 3,15 and 6,75 ng/g lipid.
xiv. There were 2 sampling periods, the 2^{nd} week and the 12^{th} week after delivery. Also omnivores and vegetarians/vegans were compared. No difference in levels was found: both 0,17.
xv. A multigeneration survey was performed, families with three generations: 14 children, 13 adults and 6 grandmothers. Ages ranged from 9 years to 88 years.
xvi. 5 Composite samples were selected from the NHATS study
xvii. 3 composite samples, ng/g hair
xviii. 10 samples were taken these were pooled and two samples were measured from each pool.

NA = Not Available
ND = Not Detectable

Appendix 2: Overview of the entire life-cycle inventory system.



Appendix 3: Specifics of recycling programs in EU

Belgium

Regulation
The 3 regional regulations for the management of WEEE, were implemented through one Environmental Policy Agreement, come into force in February 2001 within whole Belgium.

Scope
8 categories of appliances (covering categories 1-6 of WEEE directive):
- freezing and refrigerating equipment
- Large white goods
- Small white goods
- Brown goods
- IT- and communication equipment
- Gardening tools
- Small household appliances
- Lighting equipment (since 1/07/04)

Responsibilities
Producers bear an individual take-back duty for their own products or for similar products tendered to them. Retailers/distributors must take-back WEEE free of charge when selling a similar product.

Recycling Targets

	Recycling rates	Ferrous metals	Non ferrous metals	Plastics
Large white goods	90%	95%	95%	20% recycling (100% recovery)
Refrigerating and freezing appliances	70%			
TV and PC screens	70%			
Others	70%			

Management
Recupel Asbl is an executive management scheme gathering currently 6 sector associations, covering respectively large household appliances, consumer electronics, small household appliances, IT - Telecommunication and office equipment, electric tools and lighting equipment.

Collection infrastructure
Recupel collection scheme is organised through 50 social economy enterprises, 550 municipalities' containers parks and 1800 registered retailers. It will be further organised around about 30 Regional Transfer Stations (RTS) covering large collection areas, and where WEEE collected from municipal recycling facilities will be gathered and sorted. In 2002, container parks captured 75 % of the WEEE collected, while the share of Social enterprises was 10% and retailers and distributors gathered 15 % of the total amount collected

Financing
Financing is borne by the consumers through a visible fee which is levied on the products, worked out by sampling at recycling plants, and managed per sector to cover

the take back and treatment costs of appliances : transport from the container parks, sorting, and recycling.

Achievements

In 2002, RECUPEL collected 35 875 tons of WEEE (= 3,5 kg per inhabitant) and achieved a global 80% recycling rate.
In 2003, RECUPEL collected 45.037 tons of WEEE (= 4,5 kg per inhabitant) and achieved a global 83% recycling rate and 87% recovery rate. From these total quantities, 69% were collected in Flanders, 28% in the Walloon Region and 3% in the Brussels area.

Denmark

Regulation

Order from the Ministry of Environment and Energy, n°1067 of December 22, 1998 on Management of waste from electrical and electronic products. Following the Danish Environmental Protection Act local councils are in charge of the management of waste generated in their municipality. They can choose either to assume the task itself, or confer it to an intermunicipal waste company, or to contract operations to private companies (which is normally the case for industrial and commercial waste).

Recycling activities are also generally taken in charge by private companies but incineration (with energy recovery) is usually managed by local authorities themselves. For hazardous waste a network of intermunicipal transfer stations has been set up, which are scattered on all the Danish territory. They are managed by KommuneKemi A/S (a group held by the Danish municipalities). But KommuneKemi A/S does not handle WEEE.

Scope

The regulation essentially covers white goods, radio and television sets, IT products, office equipment and instruments of monitoring and control.

Responsibilities

Local councils were given until 1st June 1999 for providing regulations laying down detailed rules on the handling, assignment and collection of WEEE.

Recycling Targets

The regulation *should* lead to the diversion of 25 000 tonnes of WEEE from incineration and landfilling to recycling and so allow to recover for instance 40% of the landfilled copper (*Source* : Waste 21).

Management

Local authorities ensure that waste electrical and electronic equipment is collected and assigned to separate treatment and approved companies. About 30 SME have so developed an expertise and specialised in the processing of WEEE.

Upon request, producers may be granted permits by local council to take back free of charge their own or similar products.

Since 1999, distributors and retailers may offer a take back service in the scope of municipal waste management schemes.

Financing

Costs for implementing the WEEE legislation until now have been met by local governments. Total costs of treatment of WEEE are estimated between 100 and 200 million DKK per year (13,5 – 27 millions €). The regulation should also induce a rise in the annual fee paid by households of about 5,4 €.

Germany

Regulation

From August 2005 consumers can hand in their waste free of charge at communal collection points: thereafter, manufacturers must assume responsibility for the waste.

Manufacturers must provide a guarantee that they will finance the management of WEEE for equipment brought on to the market after August 2005.

Management

For management of waste equipment in the future, suitable elements of current usual practice should be taken into consideration. This includes in particular separate collection of waste equipment from private households (shared product responsibility) which is already practiced in many local authorities, but also good practice within a purely professional sphere (business to business).

Collection infrastructure

Proven local authority collection structures are taken into account, local authorities retain financial responsibility for the collection of all waste equipment from private households. The future ordinance stipulates that the public waste management companies must place the different categories of waste electronic and electrical equipment in a specified number of collection containers ready for the producers to collect.

In addition it is possible for **distributors** to voluntarily take back waste equipment
A distributor can pass on waste equipment he has voluntarily taken back to the public companies free of charge.

In accordance with the principle of shared product responsibility, the **manufacturer** assumes responsibility from the local authority collection points onwards.

In order to ensure that producers throughout the country meet their obligation to collect WEEE without distortions of competition and under the same conditions, the ordinance commits them to organise collection consistently with competition in a central contact office for the local authorities which the producers organise on a private law basis and finance (coordination office). This office takes all information on containers ready for collection and requests the manufacturers or their commissioned waste management companies to collect each of the reported WEEE containers, according to a rota specified by this office. It thus ensures that WEEE does not remain in any of the local authorities, and prevents “selective collection” (collection only at attractive points with a high level of specific waste equipment types).

Financing

As a countermove to the local authorities' assuming responsibility for collection from households, from the local authority collection point onwards the producers finance waste management (recovery/recycling/environmentally sound disposal) for all waste equipment from private households.

Achievements

In Germany, the quantity of WEEE, including domestic appliances and consumer electronics, has grown from 1,188,000 tonnes in 1992 to 2,099,000 in 2000.

The Netherlands

Regulation

Decree of 21 April 1998, entered into force partly on 1st June 1998 and completely on 1st January 1999.

Scope

14 categories of Electrical and Electronic equipment, including CFC products, which were regulated in two steps (large goods from 1st January 1999, and the remaining categories one year later).

Responsibilities

Producers/importers have to take back and recycle:
- WEEE ***of their own brand*** from Local Authorities' collection points
- WEEE ***of their own brand*** tendered to them by a repair company
- WEEE tendered to them by a retailer when supplying a new "similar" product. (From January 1, 2005, this "old for new" duty will become a "brand-related" one).

Manufacturers and importers can be exempted from their individual duties by the signature of a Covenant with the Ministry of Environment and by joining a collective scheme.

Since July 1999, **Local Authorities** must provide for household WEEE separate collection (either kerbside collection, or collection on sites), and for the creation and maintenance of a site within the municipality or the municipalities' association they are part of, where suppliers can leave a product taken back from a private household. As a corollary of the producers' obligations, municipalities are responsible for orphan products, and have to sort WEEE by brand to leave them at manufacturers' disposal.

Retailers have to take back WEEE coming from consumers on the "old for one" basis. It is prohibited to retain for commercial purposes freezers or refrigerators discarded after use.

Recycling Targets

	Recycling rates
TV sets	69%
Large white goods appliances	73%
Refrigerating and freezing equipment	75%
“Small” appliances	53%

Management

For white goods, 5 main producers’ sector organisations have joined within NVMP (Netherlands Association for Disposal of “Metalelectro” Products) while V-ICT (or ICT-Milieu) has been set up for the management of grey goods (IT equipment, Paper printing equipment and telecommunication goods).

Both take, through official carriers, goods discarded by consumers from RTS, retailers and repair companies, to their recycling partners.

Associated within the NVRD (Nederlands Vereniging vor Afval en Reinigingsmanagement) since 1996, local authorities ensure the collection and the transport of WEEE to one of the 69 Regional Transfer Stations where WEEE are sorted out and put to the disposition of manufacturers and importers of EEE. Since they provide manufacturers/importers with such a logistical structure, these have agreed that Local Authorities are neither obliged to sort WEEE by brand (unless they are paid for this service), nor to take care of orphan products.

Like repair companies, retailers have access to the municipal facilities. Regional Transfer Stations accept also waste tendered to them directly by retailers, but may charge them for the service.

In 2001, 87% of the products collected by NVMP originated from Regional Transfer Stations. The role of the Distribution Centres has stabilised at 3-4% of the total collection, while the Retail Sector collected directly 10% of the total amount. This channel seemed to display particular growth.

Collection infrastructure

Financing

Local authorities (LRA) bear the costs for the collection and transport of WEEE until the “municipality limit”; other transport and sorting costs are financed by the manufacturers’ organisations. LRA finance WEEE separate collection by levying local taxes. Following the kind of service agreed with the Regional Transfer Station, this amounts to 0,16 € on average per inhabitant.

When buying an electrical or electronic equipment, consumers pay a removal contribution in addition to the purchase price. With the removal fees, **NVMP** pays :

- the Regional Transfer Station : manufacturers/importers pay a fee per item that is transferred to them; this fee varies from 1,80 € to 3,40 €.
- the logistic and recycling partners.

Retailers receive also a proportional compensation for their take-back service (one-off 10% on the payment of removal fees).

ICT Milieu : ICT manufacturers and importers had opted for an individual producer responsibility and been paying for the real costs of treated grey goods of their own brand and of their share of orphans (individual responsibility, worked out by the recycling partners). But because of too many sorting constraints, and significant amounts of orphan products, a new financing scheme was introduced from 2003 based on collective producer responsibility : producers will pay for the treatment of the effective items collected and processed in proportion of their current market share. .

It has been estimated by the Ministry of Environmental Protection that altogether, the recovery of WEEE in the Netherlands costs about 1,- € per inhabitant.

Achievements

According to NVMP the collection results in 2001 corresponded to an amount of 4,13 kg WEEE/ capita.

Norway

Regulation

The Regulation regarding scrapped electrical and electronic products promulgated on 16 March 1998 entered into force on 1st July 1999.

Scope

No **categories** have been determined and all products containing electrical or electronic components are in principle embraced by the regulation with the exceptions of products permanently installed in means of transport or large devices (ex : lifts, escalators…) where only the components should be regarded as EE products.

The **re-use** of the EE product in its original form for its original purpose means that the product has not to be regarded as scrapped and is not covered by the regulation requirements.

Responsibilities

Manufacturers/importers are obliged to ensure that the EEE they introduce on the Norwegian market are collected when they end up as waste, and are recycled or otherwise properly handled. They are obliged to arrange for the collection of WEEE free of charge in geographical areas corresponding to those in which the products are sold, were sold or supplied trough suitable logistic systems that do not cause “unreasonably high transport costs for any municipality”. The frequency of collection points must take into account the needs of the municipality, and their capacity correspond to the share of manufacturers’ sales in the area.

Municipalities (LRAs) are obliged to receive all WEEE through accessible (regarding number, site, opening hours…) facilities. They may demand a charge for business waste, but consumer waste have to be managed with the annual municipal tax.

All **distributors/retailers** in Norway are required to accept consumer WEEE free of charge. Distributors are also only obliged to accept WEEE of products belonging to the same products range they are selling at the time these discarded appliances are handed in. The “old for new” condition only applies to waste from companies.

Recycling Targets

In 1998 a sector agreement was signed with the Ministry of Environment setting a target of 80% WEEE collection for the 1 July 2004.

Management

National suppliers have established two management enterprises for consumers' WEEE:

7 Hvitevareretur AS (large and small household appliances)
8 Elecktronikkretur AS (IT&T, Consumers Electronics, toys, medical…).
9

They have joined within El-retur in order to implement a collective logistic and recycling scheme.

In the El-retur system, WEEE is collected from about 4,000 collection points:

- 350 municipal collection facilities
- 3000 retailers
- from about 650 other sources like workshops, offices and various waste management companies.

6 logistic subcontractors are responsible for all logistic tasks, including the provision of cages and containers free of charge for collection facilities included. WEEE are then delivered to nine recycling plants dedicated to specific areas of the country.

Collection infrastructure

Financing

LRA finance the municipal collection facilities with local taxes.

Hvitevareretur AS levies a recycling fee per unit through the Norwegian Custom and Excise System (the fee is paid with each company's monthly taxes and duties), which forwards then the recycling fees to the system. The funds allow to pay the logistic and recycling costs as well as the kick backs to retailers and distributors.

For Consumers Electronics, **Elektronikkretur AS** members (447 businesses affiliated in 2001) pay a recycling fee per unit put on the market, through their branch associations. For brown and white goods, the recycling fee is prepaid, but for IT goods, total real management costs (for collection and treatment) are subdivided onto members' market shares (net volumes in kg) within the different product groups. Funds are managed by Elektronikkretur AS to pay the logistic and recycling partners.

Making the fee visible or not at the purchase is left to the distributor's discretion.

Achievements

In 2001, 7,2 kg of WEEE per capita were collected and processed by **El-retur**, which following the definition of recycling within the Norwegian regulation, achieved a recycling target of 82%.
Euro Vironment, an independent system, was set up by 14 IT companies (including Compac and Dell which are together 50% of the IT Norwegian market). By collecting about 3,250,000 kg IT products in 2002 they achieved a collection rate of 0,7 kg per capita.

Sweden

Regulation
The Producer Responsibility for Electrical and Electronic Products Ordinance (2000:208) came into effect on 1 July 2001.

Scope
10 categories of products are allocated to the Producer Responsibility. Refrigerators and freezers are excluded, since there is a municipal responsibility for these products.

Responsibilities
Manufacturers, importers and retailers are jointly responsible. When selling a new product, they are obliged to take back at the place of supply or at another suitable designated place, a *"similar"* product handed to them and *serving essentially the same purpose as the product sold*. This obligation is related to the same number of products as the products sold. Producers may designate suitable collection points only after consultation with the municipality.

Recycling Targets

Management
To avoid collection in shops, El-Kretsen AB (service provider owned by 20 trade associations in the electrical and electronic sectors) has made agreements with all 289 Swedish local authorities to use their collection schemes.

Collection infrastructure
The collection of electric and electronic waste at recycling centres is the most common method in Sweden. The addresses and opening hours of about 1000 collection points (about 650 for households, and 350 for businesses) are presented on www.elkretsen.se.

These collection points are often supplemented with on-site collection from housing estates. Since 1st July 2003, customers have been entitled to return their old television, radio or video recorder to the shop when they buy a new one. In response to this, El-Kretsen started a new free service under which they collect these old products at the shops and transport them to a recycling plant. About 3000 shops have signed up for this collection service.

WEEE are also collected by El-Kretsen from :

- Service workshops for home electronics and IT products
- Large customers of light sources
- All the Swedish hospital (system under development).

Products are sorted according to 5 main types prior to their transport :

- large white goods are handled as such
- small and medium
- sized equipment are placed in cages that hold approximately 400 kg
- fluorescent tubes (under 60 cm) are collected by means of plastic boxes of 1490 units
- fluorescent tubes (over 60 cm) are collected by means of metal boxes of 1150 units

- idem for low-energy and discharge lamps, and bulbs (plastic boxes of 1400 lamps or 2500 bulbs).

Financing

Collection stations are run at the own initiative and expense of local authorities (exception to the producer responsibility principle).

Producers (through El-Kretsen) finance the further collection and the recovery of WEEE, but historical electronic waste from households is the responsibility of the municipalities. As the Swedish law demands products show the total price, visible fees are forbidden.

Recycling fees are very complex and depend on the return rates, weight of appliances, methods and costs of treatment, material composition. El-Kretsen uses three different financing models:
Standard : recycling fee per unit put onto the market. A preliminary cost is fixed and the accounts are settled for each product type at the end of the year.
ICT : the real costs of collection and treatment of ICT-WEEE are charged each month to the manufacturers according to their market share.
There exist also **fixed annual fees** for some products.

The funds are managed by the system to pay the different partners of the system, and the recycling costs.

On average, the costs of WEEE collected and treated are about 3,90 SEK/ kg (c.a. 0,42 €), with 72% for treatment, 19% for transport/loading boxes, and 7% for administration / information costs.

Achievements

In 2001 during the six months when producer responsibility applied, about 30.000 tons of this waste were collected by El-Kretsen from households and industry, equivalent to 7kg WEEE per inhabitant.

In 2003, El-Kretsen collected approximately 80.000 tons of WEEE. In terms of the population as a whole, it corresponds to 9 kg/inhabitant with a cost of about 30 SEK (3,26 €) / inhabitant. The 23.500 tons of refrigerators and freezers (2,5 kg/inhabitant) handled by the municipalities shall be added to this.

Sources : Collecting and recycling of electrical and electronic products in Sweden, 2003-2004, El-Kretsen AB.

Switzerland

Regulation

Ordinance on the return, the take-back and the disposal of electric and electronic appliances (OREA), in place since 1 July 1998.

Scope

The OREA addresses appliances which depend on electricity and specifically mentions consumer electronics, office, information and telecommunication equipment, and household appliances.

Responsibilities

Manufacturers or importers have to take back appliances of their own brand or of the brand they sell.

Municipalities have no mandatory take-back obligation, and are thus not obliged to provide for separate collection or for collection points. If they are willing to, local authorities can do it on a voluntary basis, knowing that electrical or electronic appliance cannot be dealt with anymore together with bulky waste collections, and that the OREA decree states that disposal of appliances must be financed by market actors.

Retailers must take back appliances similar to those they sell from final consumers.

Management

2 mains voluntary schemes have been set up :

- SWICO has been dealing with "office equipment" and consumers electronics from 2002
- SENS deals with refrigerating and freezers.

Both have been working together from 1st January 2003 within a global solution for WEEE management.

Collection infrastructure

In Switzerland, the retailers network is considered to offer enough taking back opportunities in itself, and returning equipment to the dealer or the manufacturer is strongly recommended by SWICO, as they are specialists to assess the possibility to recycle the equipment or parts of it. With this approach, 5-15% of discarded equipment can be reused. Retailers take back discarded appliances from private and business users free of charge.

Pick up services are organised on request by the manufacturers associations from private households, points of delivery or (re-)distribution centres.

Financing

The manufacturers have set up a Convention for Recycling and Disposal, that obliges participants to impose an Advanced Recycling Fee (ARF) on the sale of new equipment. Manufacturers transfer the fees on a recycling account held by SWICO.

There are 2 different models to calculate the ARF (which includes also the Advanced Disposal Tax for batteries) :

1. IT and office products : fee conditional on the equipment value
2. consumer electronics : each piece of equipment has a specified fee.

Consumers goods which price is not higher than c.a. 35,-€ are not subjected to the ARF.

Achievements

The current figure for collected WEEE in Switzerland is 8kg/ capita. More than 75% of end-of-life equipment is recycled, approximately 20% are incinerated, and 3% end up in landfills.

References

[1] R.W. Hahn (Ed.), "Risks, Costs and Lives Saved. Getting Better Results from Regulation", Oxford Univ. Press (1996).
[2] "Commission Initiated Formal Rule for Assessing Costs and Benefits of its Regulations", Press Release from US CPSC, http://www.cpsc.gov/CPSCPUB/PREREL/prhtml81/81010.html (1981).
[3] G.C. Blomquist, "Self-Protection and Averting Behavior, Values of Statistical Lives and Benefit Cost Analysis of Environmental Policy", *Review of Economics of the Household*, **2**, pp 89-110 (2004).
[4] A.M. Finkel, "Myths, Chicanery, and Blundering on the Risk Assessment Front", *AIHA Journa*l, **57**, pp 793-798 (1996).
[5] K.J. Arrow, M.L. Cropper, G.C. Eads, R.W. Hahn, L.B. Lave, R.G. Noll, P.R. Portney, M. Russel, R. Schmalensee, V.K. Smith, and R.N. Stavins, "Is There a Role for Benefit-Cost Analysis in Environmental, Health, and Safety Regulation?", *Science*, **272**, pp 221-222 (1996).
[6] C.F. Clement, "The Characteristics of Risks of Major Disasters", *Proc. R. Soc. Lond. A*, **424**, pp 439-459 (1989).
[7] N. Ashford, K. Barrett, A. Bernstein, R. Costanza, P. Costner, C. Cranor, P. deFur, K. Geiser, A. Jordan, A. King, F. Kirschenmann, S. Lester, M M'Gonigle, P. Montague, J. Peterson Myers, M. O'Brien, D. Ozonoff, C. Raffensperger, P. Regal, P. Resor, F.Robinson, T. Schettler, T. Smith, K-R. Sperling, S. Steingraber, D. Takvorian, J. Tickner, K. von Moltke, B. Wahlstrom, J. Warledo, "Wingspread Statement on the Precautionary Principle", Wingspread Conference Centre, Racine, Wisconsin (1998).
[8] J. Tickner, C. Raffensperger, and N. Myers, "The Precautionary Principle in Action: A Handbook", *Science and Environmental Health Network*, http://www.biotech-info.net/handbook.pdf (2000).
[9] M. Simonson, A. Boldizar, C. Tullin, H. Stripple, and J-O Sundqvist, "The Incorporation of Fire Considerations in the Life-Cycle Assessment of Polymeric Composite Materials: A Preparatory Study", SP Report 1998:25, ISBN 91-7848-731-5 (1998).
[10] M. Simonson, P. Blomqvist, A. Boldizar, K. Möller, L. Rosell, C. Tullin, H. Stripple and J.O. Sundqvist, "Fire-LCA model: TV case study", SP Report 2000:13, ISBN 91-7848-811-7 (2000).
[11] K. Thuresson, "Occupational Exposure to Brominated Flame Retardants with Emphasis on Polybrominated Diphenyl Ethers." E*nvironmental Chemistry*, pp. 69. Stockholm University, Stockholm (2004).
[12] WHO/FAO, J. "Polybrominated Diphenyl Ethers." (2002).
[13] World Wildlife Fund, W. "Contamination: The Result of WWF'S Biomonitoring Survey", (http://www.wwf.org.uk/filelibrary/pdf/biomonitoringresults.pdf) (2003).
[14] World Wildlife Fund, W. "Contamination: the next generation. Results of the family chemical contamination survey." (2004).
[15] K. van Londen, M. van den Berg, "DecaBromodiphenyl ether: occurrence, toxicokinetics, and human risk assessment.", Report for BSEF by IRAS, Utrecht University.
[16] American Chemistry Council's Brominated Flame Retardant Industry Panel, B. Decabromodiphenylether (A.K.A. Decabromodiphenyl Oxide, DBDPO) CAS # 1163-19-5. In Voluntary Children's Chemical Evaluation Program (VCCEP), Arlington, (2002).
[17] EU, "European Union Risk Assessment Report: Bis(pentabromophenyl)ether", pp. 1-294 (2002).
[18] EU, "Update of the risk assessmnet addendum of Bis(Pentabromodiphenyl)Ether (Decabromdiphenylether) Human Health Draft." (2002).
[19] National Toxicology Program, N. "NTP Toxicology and Carcinogenesis Studies of Decabromodiphenyl Oxide (CAS No. 1163-19-5) In F344/N Rats and B6C3F1 Mice (Feed Studies)", *Natl Toxicol Program Tech Rep Ser 309*, pp 1-242 (1986).
[20] H. Viberg, A. Fredriksson, E. Jakobsson, U. Orn, P. and Eriksson, "Neurobehavioral derangements in adult mice receiving decabrominated diphenyl ether (PBDE 209) during a defined period of neonatal brain development" *Toxicol Sci,* **76**, 112-120 (2003).
[21] H. Vijverberg, and M. van den Berg,Letter to the editor. *Toxicological Sciences* **79**, 205-206 (2004).
[22] M.F. Hughes, B.C. Edwards, C.T. Mitchell, B. and Bhooshan, "In vitro dermal absorption of flame retardant chemicals", *Food and Chemical Toxicology*, **39**, 1263-1270 (2001).

[23] A. Schecter, M. Pavuk, Päpke, J.J. Ryan, L. Birnbaum, and R. Rosen, "Polybrominated Diphenyl Ethers (PBDEs) in U.S. Mothers' Milk", *Environ Health Perspect*, **111**, 1723-1729 (2003).
[24] H.M. Stapleton, N.G. Dodder, J.H. Offenberg, M.M. Schantz, and S.A. Wise, S. A., "Polybrominated diphenyl ethers in house dust and clothes dryer lint." *Environ Sci Technology*, **39**, 925-931 (2005).
[25] United States Environmental Protection Agency, U. E. "Child-Specific Exposures Factors Handbook"; Interim Report. Available from: http://cfpub.epa.gov/ncea/cfm/recordisplay.cfm?deid=55145, pp. table 11-16. National center for Environmental Assessment, Washington (2002).
[26] F. Consoli, D. Allen, I Boustead, J. Fava, W. Franklin, A.A. Jensen, N. de Oude, R. Parish, D. Postlethwaite, B. Quay, J. Séguin, and B. Vigon, "Guidelines for Life-Cycle Assessment: A Code of Practice", SETAC (1993).
[27] L-G. Lindfors, K. Christiansen, L. Hoffman, Y. Virtanen, V. Juntilla, O-J. Hanssen, A. Rönning, T. Ekvall, and G. Finnveden, "Nordic Guidelins on Life-Cycle Assessment ", Nord 1995:20, Nordic Council of Ministers, Copenhagen (1995).
[28] M. de Poortere, C. Schonback and M. Simonson, "The Fire Safety of TV Set Enclosure Materials, A Survey of European Statistics", *Fire and Materials*, **24**, 53-60 (2000).
[29] M. Simonson, P. Andersson, L. Rosell, V. Emanuelsson, H. Stripple, "Fire-LCA Model: Cables Case Study", SP Report 2001:22 (2001).
[30] P. Andersson, M. Simonson, L. Rosell, P. Blomqvist, H. Stripple, "Fire-LCA Model: Furniture Study", SP Report 2003:22 (2003).
[31] P. Andersson, M. Simonson, C. Tullin, H. Stripple, J.O. Sundqvist and T. Paloposki, "Fire LCA Guidelines", SP Report 2004:43 (2004).
[32] R.L. Revesz, "Environmental Regulation, Cost-Benefit Analysis, and the Discounting of Human Lives", *Columbia Law Review*, **99**(4), 941-1017 (1999).
[33] S.I. Brandt-Rauf and P.W. Brandt-Rauf, "Occupational Health Ethis: OSHA and the Courts", *J. of Health Politics, Policy and Law*, **5**, pp 523-534 (1980).
[34] G. C. Blomquist, "Self-Protection and Averting Behavior, Values of Statistical Lives, and Benefit Cost Analysis of Environmental Policy", *Rev. of Ec. Of the Household*, **2**, pp 89-110 (2004).
[35] K.J. Arrow, M.L. Cropper, G.C. Eads, R.W. Hahn, L.B. Lave, R.G. Noll, P.R. Portney, M. Russell, R. Schmalensee, V. K. Smith, and R.N. Stavins, "Is there a role for Benefit-Cost Analysis in Environmental, Health, and Safety Regulation?", *Science*, **272**, pp 221-222 (1996).
[36] R.W. Hahn, "The Economic Analysis of Regulation: A Response to the Critics", *The University of Chicago Law Review*, **71(3)**, pp 1021-1054 (2004).
[37] D. Seligman, "How much is money is your life worth?" *Fortune*, **113(5)**, pp 25-28 (1986).
[38] O. Ashenfelter and M. Greenstone, "Using Mandated Speed Limits to Measure the Value of a Statistical Life", presented at the session in honor of the memory of Sherwin Rosen at the AERE/ASSA meetings, available as NBER Working Paper 9094 (2002).
[39] R. R. Jenkins, N.Owens, and L.B. Wiggins, "Valuing reduced risk to children: The case of bicycle safety helmets.", *Contempary Economic Policy*, **19(4)**, pp 397-408 (2001).
[40] G.C. Blomquist, T.R. Miller, and D.T. Levy, "Values of risk reduction implied y motorist use of protection equipment: New evidence from different populations", *J. of Transport Economics and Policy*, **30**, pp 55-66 (1996).
[41] P. S. Carlin, and R. Sandy, "Estimating the value of a young child's life", *Southern Economic J.*, **58**, pp 186-202 (1991).
[42] T. Mount, W. Weng, W. Schulze, and L. Chestnut, "Automobile safety and the Value of Statistical Life in the family: Valuing reduced risk for children, adults and the elderly", Presented at the Assoc. of Environ. and Resource Economists Workshop, "Assessing and Managing Environmental and Public Health Risks", Bar Harbor, ME, June 13-15 (2001).
[43] M.K. Dreyfus and W.K. Viscusi, "Rates of time preference and consumer valuation of automobile safety and fuel efficiency", *J. of Law and Economics*, **38**, pp 79-105 (1995).
[44] S.E. Atkinson, and R. Halvorsen, "The valuation of risks to life: Evidence from the market for automobiles", *Review of Economics and Statistics*, **72**, pp 133-136 (1990).
[45] T. Gayer, J.T. Hamilton and W.K. Viscusi, "Private values of risk tradeoffs at superfund sites: Housing market evidence on learning about risk", *Review of Economics and Statistics*, **82**, pp 439-451 (2000).

[46] C.L. Smith, "Preliminary regulatory analysis of a draft proposed rule to address cigarette and small open flame ignitions of upholstered furniture", Directorate for Economic Analysis, US CPSC, November 2005.

[47] Association of Cities and Regions for Recycling (ACRR) website: http://www.acrr.org .

[48] http://europa.eu/eur-lex/pri/en/oj/dat/2003/l_037/l_03720030213en00240038.pdf

[49] H. Viberg, A. Fredriksson, and P. Eriksson, "Neonatal exposure to the brominated flame retardant 2,2',4,4',5-pentabromodiphenyl ether causes altered susceptibility in the cholinergic transmitter system in the adult mouse", *Toxicol Sci,* **67**, pp 104-107 (2002).

[50] P. Eriksson, H. Viberg, E. Jakobsson, U. Orn, and A. Fredriksson, "A brominated flame retardant, 2,2',4,4',5-pentabromodiphenyl ether: uptake, retention, and induction of neurobehavioral alterations in mice during a critical phase of neonatal brain development", *Toxicol Sci*, **67**, 98-103 (2002).

[51] U.G. Ahlborg, G.C. Becking, L.S. Birnbaum, A. Brouwer, H.J.G.M. Derks, M. Feeley, G. Golog, A. Hanberg, J.C. Larsen, A.K. Liem, A. K., et al., e. "Toxic equivalency factors for dioxin-like PCBs: Report on WHO-ECEH and IPCS consultation" *Chemosphere,* **28**, pp 1049-1067 (1994).

[52] M. Van den Berg, L. Birnbaum, A.T. Bosveld, B. Brunstrom, P. Cook, M. Feeley, J.P. Giesy, A. Hanberg, R. Hasegawa, S.W. Kennedy, T. Kubiak, J.C. Larsen, F.X. van Leeuwen, A.K. Liem, C. Nolt, R.E. Peterson, L. Poellinger, S. Safe, D. Schrenk, D. Tillitt, M. Tysklind, M. Younes, F. Waern, and T. Zacharewski, "Toxic equivalency factors (TEFs) for PCBs, PCDDs, PCDFs for humans and wildlife", *Environ Health Perspect*, **106**, pp 775-792 (1998).

[53] P.R. Kodavanti, T.R. Ward, G. Ludewig, L.W. Robertson, and L.S. Birnbaum, "Polybrominated diphenyl ether (PBDE) effects in rat neuronal cultures: 14C-PBDE accumulation, biological effects, and structure-activity relationships", *Toxicol Sci*, **88**, pp 181-192 (2005).

[54] P.R. Kodavanti, and T.R. Ward, "Differential effects of commercial polybrominated diphenyl ether and polychlorinated biphenyl mixtures on intracellular signaling in rat brain in vitro", *Toxicol Sci*, **85**, pp 952-962 (2005).

[55] J.M. Sanders, L.T. Burka, C.S. Smith, W. Black, R. James, and M.L. Cunningham, "Differential expression of CYP1A, 2B, and 3A genes in the F344 rat following exposure to a polybrominated diphenyl ether mixture or individual components", *Toxicol Sci*, **88**, pp 127-133 (2005).

[56] C. Dufault, G. Poles, and L.L. Driscoll, "Brief postnatal PBDE exposure alters learning and the cholinergic modulation of attention in rats" *Toxicol Sci*, **88**, pp 172-180 (2005).

[57] M. Hausman, C. Koopman-Esseboom, V. Fidler, M. Hadders-Algra, C.G. van der Paauw, L.G. Tuinstra, N. Weisglas-Kuperus, P.J. Sauer, B.C. Touwen, and E.R. Boersma, "Perinatal exposure to polychlorinated biphenyls and dioxins and its effect on neonatal neurological development" *Early Hum Dev*, **41**, pp 111-127 (1995).

[58] C. Koopman-Esseboom, M. Hausman, B.C. Touwen, E.R. Boersma, A. Brouwer, P.J. Sauer, and N. Weisglas-Kuperus, "Newborn infants diagnosed as neurologically abnormal with relation to PCB and dioxin exposure and their thyroid-hormone status", *Dev Med Child Neurol*, **39**, p 785 (1997).

[59] S. Patandin, C.I. Lanting, P.G. Mulder, E.R. Boersma, P.J. Sauer, and N. Weisglas-Kuperus, "Effects of environmental exposure to polychlorinated biphenyls and dioxins on cognitive abilities in Dutch children at 42 months of age" *J Pediatr*, **134**, pp 33-41 (1999).

[60] J.L. Jacobson, S.W. Jacobson, and H.E. Humphrey, "Effects of in utero exposure to polychlorinated biphenyls and related contaminants on cognitive functioning in young children" *J Pediatr*, **116**, pp 38-45 (1990).

[61] A.K. Peters, K. van Londen, A. Bergman, J. Bohonowych, M.S. Denison, M. van den Berg, and J.T. Sanderson, "Effects of polybrominated diphenyl ethers on basal and TCDD-induced ethoxyresorufin activity and cytochrome P450-1A1 expression in MCF-7, HepG2, and H4IIE cells", *Toxicol Sci*, **82**, pp 488-496 (2004).

[62] R.J.B. Peters, "Man-Made Chemicals in Human Blood", *TNO/Greenpeace Netherlands*, Apeldoorn, The Netherlands (2004).

[63] Sambrook Research International, TV Fires (Europe), Department of Trade and Industry (UK), March (1996), http://www.dti.gov.uk/homesafetynetwork/pdf/tvfires.pdf .

[64] I. Enqvist (Ed.), "Electrical Fires – Statistics and Reality. Final report from the 'Vällingby project'", Electrical Safety Commission, available in Swedish only (1997).

[65] K. Stålbrand, ”Common household appliances cause thousands of fires”, Aktuell Säkerhet, 1, pp 24-28 (1997). Available in Swedish only.
[66] J.R. Hall, ”The US Home Product Report, 1990-1994 (Appliances and Equipment)”, NFPA (1997).
[67] M. Simonson, C. Tullin and H. Stripple, “Fire-LCA study of TV sets with V0 and HB enclosure material”, Chemosphere, 46(5), pp 737-744 (2002).
[68] G.C. Stevens and A.H. Mann, “Risks and Benefits in the Use of Flame Retardants in Consumer Products”, A report for the Department of Trade and Industry, DTI Ref: URN 98/1026 (1999).
[69] J. Still, H. Orlet, E. Law, and C. Gertler, “Lawn Mower Related Burns”, *J. Burn Care & Rehabilitation*, September/October, 403-405 (2000).
[70] http://www.scb.se/templates/pressinfo____126504.asp
[71] http://www.internetworldstats.com/stats4.htm#eu
[72] http://www.forsakringsforbundet.com/common/browse.asp?id=1717
[73] A. Schecter, O. Papke, K.C. Tung, J. Joseph, T.R. Harris, and J. Dahlgren, “Polybrominated diphenyl ether flame retardants in the U.S. population: current levels, temporal trends, and comparison with dioxins, dibenzofurans, and polychlorinated biphenyls”, *J Occup Environ Med*, **47**, pp 199-211 (2005).
[74] A. Sjödin, D.G. Patterson, and Å. Bergman, ”Brominated Flame Retardants in Serum from U.S. Blood Donors”, *Environmental Science and Technology*, **35**, pp 3830-3833 (2001).
[75] A. Sjödin, L. Hagmar, E. Klasson-Wehler, K. Kronholm-Diab, E. Jakobsson, and Å. Bergman, ”Flame retardant exposure: polybrominated diphenyl ethers in blood from Swedish workers”, *Environ Health Perspect*, **107**, pp 643-648 (1999).
[76] K. Thuresson, “Occupational Exposure to Brominated Flame Retardants with Emphasis on Polybrominated Diphenyl Ethers” *Environmental Chemistry*, pp. 69. Stockholm University, Stockholm (2004).
[77] B. Fängström, “Human exposure to organohalogen compounds in the Faroe Islands”, Stockholm University, Department of Environmental Chemistry. Doctoral Thesis (2005).
[78] T. Takasuga, K. Senthilkumar, H. Takemori, E. Ohi, H. Tsuji, and J. Nagayama, “Impact of fermented brown rice with Aspergillus oryzae (FEBRA) intake and concentrations of polybrominated diphenylethers (PBDEs) in blood of humans from Japan”, *Chemosphere*, **57**, pp 795-811 (2004).
[79] D. López, M. Athanasiadou, I. Athanassiadis, L.Y. Estrada, F. Díaz-Barriga and Å. Bergman, “A Preliminary Study on PBDEs and HBCDD in Blood and Milk from Mexican Women”, The Third International Workshop on Brominated Flame Retardants (2004).
[80] J.J. Ryan, and B. Patry, “Body Burdens and Food Exposure in Canada for Polybrominated Diphenyl Ethers (BDEs)”, *Organohalgen Compounds*, **51**, pp 226-229 (2001).
[81] B. Vieth, T. Herrmann, H. Mielke, B. Ostermann, O. Päpke, T. Rüdiger, ”PBDE Levels in human Milk: The Situation in Germany and Potential Influencing Factors - A Controlled Study”, *Organohalogen Compounds*, **66**, pp 2613-2618 (2004).
[82] S. Lunder, and R. Sharp, “Mothers' Milk. Record Levels of Toxic Fire Retardants found in American Mothers' Breast Milk”, available from: http://www.ewg.org/reports/mothersmilk/es.php Environmental Working Group (2003).
[83] J. She, A. Holden, M. Sharp, M. Tanner, C. Williams-Derry, K. Hooper, “Unusual Pattern of Polybrominated Duphenyl Ethers (PBDEs) in US Breast Milk”, *Organohalogen Compounds*, **66**, pp 3895-3900 (2004).
[84] B. Fängström, A. Strid, I. Athanassiadis, P. Grandjean, P. Weihe, and Å. Bergman, “A Retrospective Time Trend Study of PBDEs and PCBs in Human Milk from the Faroe Islands”, *Organohalgen Compounds*, **66**, pp 2795-2799 (2004).
[85] J.S. Stanley, P.H. Cramer, K.R. Thornburg, J.C. Remmers, J.J. Breen, and J. Scwemberger, “Mass Spectral Confirmation of Chlorinated and Brominated Diphenylethers in Human Adipose Tissues”, *Chemosphere*, **23**, pp 1185-1195 (1991).
[86] V.J. DeCarlo, ”Studies on brominated chemicals in the environment”, *Ann N Y Acad Sci*, **320**, pp 678-681 (1979).

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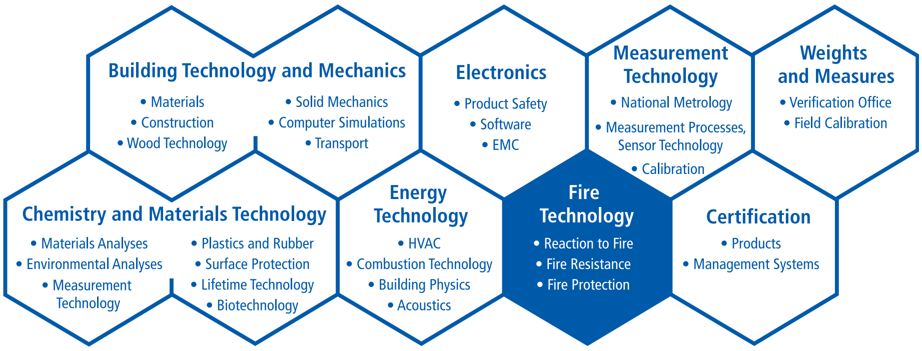

SP Fire Technology
SP REPORT 2006:28
ISBN 91-85533-13-0
ISSN 0284-5172

SP Swedish National Testing and Research Institute
Box 857
SE-501 15 BORÅS, SWEDEN
Telephone: + 46 33 16 50 00, Telefax: +46 33 13 55 02
E-mail: info@sp.se, Internet: www.sp.se

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