

Supplementary material to:

Life cycle assessment of flame retardants in an electronics application.

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Life Cycle Assessment of Environment-Compatible Flame Retardants (Prototypical Case Study)

Collaborative Project in "Environment (including Climate Change)"

WP8 D8.5 LCA report

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Abbreviations

ABS	Acrylonitrile butadiene styrene terpolymer
Alpi	Aluminium diethylphosphonate
APP	Ammonium polyphosphate
ATH	Aluminium trihydroxide
ATO	Antimony trioxide
BDP	Bisphenol-A bis(diphenylphosphate)
BFR	Brominated flame retardant
BPS	Brominated polystyrene
BSEF	Bromine Science and Environmental Forum
CAS	Chemical abstracts service
CF	Characterisation factor
CSR	Corporate social responsibility
DALY	Disability-adjusted life year
DCB	Dichlorobenzene
DecaBDE	Decabromo diphenylether
DOPO	9,10-dihydro-9-oxa-10-phosphaphenanthrene-10-oxide
E&E	Electrical and electronic equipment
EoL	End of Life
EPR	Epoxy resin
EPS	(expanded) polystyrene foam
ESF	Enfiro stakeholder forum
(EU)–SIA	(EU) Social Impact Assessment
EVA	Ethylene vinyl acetate
FR	Flame retardant
HBCDD or HBCD	Hexabromocyclododecane
HFFR	Halogen free flame retardant
HIPS	High-impact polystyrene
LCA	(environmental) Life Cycle Assessment
LCI	Life Cycle Inventory
LCIA	Life Cycle Impact Assessment
MPP	Melamine polyphosphate
MSWI	Municipal solid waste incineration
NMVOC	Non-methane volatile organic compounds
OctaBDE	Octabromodiphenylether
OEM	Original equipment manufacturer
PBB	Polybrominated biphenyls
PBDD/PBDF	Polybrominated dibenzodioxins / polybrominated dibenzofurans
PBDE	Polybrominated diphenyl ethers
PBT	Polybutylene terephthalate
PC	Polycarbonate
PCDD/PCDF	Polychlorinated dibenzodioxins / polychlorinated dibenzofurans
PET	Polyethylene terephthalate
PFR	Phosphor-based flame retardant
PIN	Phosphorus, Inorganic and Nitrogen
PPE	Polyphenylene ether
PS	Polystyrene
PWB	Printed wiring board
RDP	Resorcinol bis(diphenylphosphate)

REACH	Regulation on Registration, Evaluation, Authorization and Restriction of Chemicals
RoHS	Restriction of Hazardous Substances
SETAC	Society of Environmental Toxicology and Chemistry
SAN	Styrene acrylonitrile copolymer
SLCA	Social Life Cycle Assessment
TBBPA	Tetrabromobisphenol A
TEF	Toxic Equivalency Factor
TEQ	Toxic Equivalent
VECAP	Voluntary Emissions Control Action Programme
WEEE	Waste Electrical and Electronic Equipment
ZHS	Zinc hydroxystannate
ZS	Zinc stannate

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1. Introduction

1.1 The ENFIRO project

Flame retardants are incorporated into materials (plastics, textiles) to save lives. However, some of them (certain brominated flame retardants, BFRs) pose a risk to the environment, and should therefore be substituted by less harmful ones (halogen-free flame retardants, HFFRs). The EU funded project ENFIRO aims to find out which alternatives are most suitable, from both a technical and environmental point of view.

Environmental scientists, chemists, toxicologists, material scientists and fire safety researchers have joined forces to evaluate all relevant aspects of alternative flame retardants, in order to determine which alternatives are most viable.

One of the studies within this framework is an environmental life cycle assessment (LCA).

1.2 The ENFIRO LCA study

In order to obtain insight into the overall consequences of the substitution of BFRs with HFFRs for the environment, a life cycle assessment (LCA) is conducted. With this method, an evaluation is made of all environmental impacts in all stages of the life cycle of the products considered, from the production of raw materials via production, use and maintenance to the waste treatment phase.

This LCA is conducted according to the procedures described in ISO14044.

An LCA study is divided into 4 phases (see Figure 1):

1. The goal and scope phase, in which the purpose of the study is stated, the level of detail and study boundaries are defined, and methodological choices are made.
2. The life cycle inventory phase (LCI phase) results in an inventory of input/output data with regard to the system being studied. It involves the collection of the data necessary to meet the goals of the defined study.
3. The life cycle impact assessment phase (LCIA) has the purpose to transform the large body of data on inputs and outputs from the LCI into a limited number of environmental effect scores.
4. In the Life cycle interpretation phase, the results of the LCIA are summarized and discussed as a basis for conclusions and recommendations, in accordance with the goal and scope definition. In addition, the robustness and validity of the data and results are checked by performing sensitivity analyses.

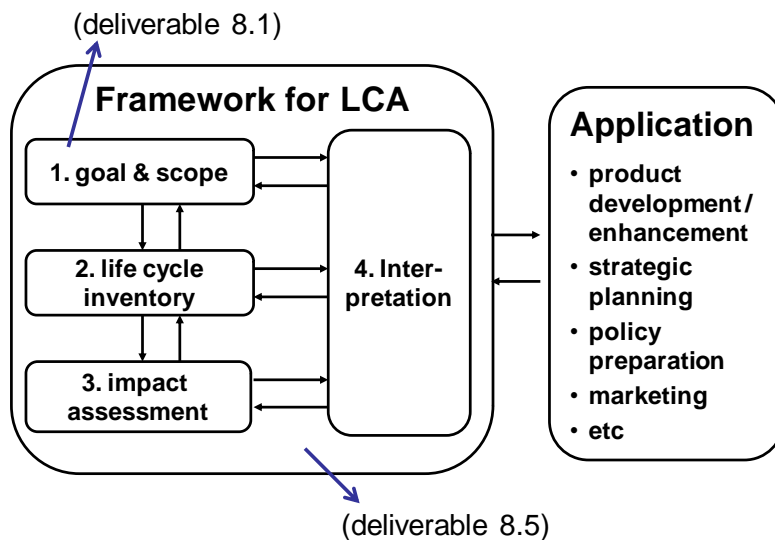


Figure 1 General framework of an LCA study, with the specific deliverables for the ENFIRO study indicated.

As figure 1 shows, the arrows between the phases go in both directions, indicating the iterative nature of the LCA performance. It is possible that due to new insights obtained in one study phase, some aspects in another phase need to be adjusted. For example, the inventory data gathered may indicate that an environmental impact category that was expected to be irrelevant during the goal & scope definition is actually of high importance, which would then lead to inclusion of that effect category in the assessment criteria as defined in the goal & scope phase.

1.3 Scientific literature on LCA studies of flame retardants

Most LCA literature in which polymers are studied, does not consider polymer additives such as flame retardants (or stabilizers and plasticizers such as phthalates).

Some studies exist which focus on flame retardants and fire, notably some efforts in which the occurrence of fires is incorporated into the environmental impact calculations (a specific LCA-application named Fire-LCA) (Andersson 2003, Andersson 2004, DePoortere 2000, Hamzja 2008, Simonson 2002).

One LCA study has been found (although not a full LCA-report or peer-reviewed study) in which TBBPA is compared with a nanoclay flame retardant. This cradle-to-gate study found a lower environmental impact for the nanoclay product than for TBBPA (DeBenedetti et al. 2008).

Some additional studies have been found on aspects of the life cycle of Printed Wiring Boards (PWBs), which is an important application of flame retardants to be addressed in the current study (Adams 2006, Alaei 2003, Andrae 2005, Bergendahl 2005, DjE 2009, Geibig 2005, Iji 1997, Rochat 2007, Scharnborst 2005). None of these studies was a full LCA study according to the ISO14044 standard.

2. Goal & Scope

2.1 Intended application

This study intends to compile the knowledge on the environmental effects of specific brominated and halogen-free flame retardants over their whole life cycle, and to demonstrate which of the alternatives has the lowest overall environmental impact. In addition, the study shows in which stages of the life cycle of all alternatives the highest environmental gain may be achieved by future improvements.

2.2 Reasons for carrying out the study

Flame retardants are indispensable in a large range of consumer products. The function of several types of flame retardant is related to the prevention and/or adsorption of radicals formed during a combustion process. Consequently, these substances are often designed to be stable up to relatively high temperatures, and are therefore not easily degradable. This persistency in the environment, in combination with the toxicity of some BFRs are a cause of concern.

In recent years, a number of alternative flame retardants (HFFRs) have been developed as possible substitution products for BFRs. In the ENFIRO project, a large set of functional and environmental properties of these HFFRs are determined.

As the BFRs have been used and studied for several decades, the knowledge on the environmental and human effects of several brominated flame retardants is already extensive. The current LCA study uses the existing and new data on FRs to compare the selected BFRs and HFFRs, and to demonstrate where in the life cycle the most significant environmental impacts occur, and which alternative has the most positive overall environmental profile.

2.3 Intended audience

The result of this study is a publicly available report which is useful for policy makers, producers of FRs, downstream users of FRs such as formulators of polymers and producers of consumer products, as the detailed evaluation of the bottlenecks in the environmental performance of flame retarded products will help to focus policy and improve products.

As the ENFIRO project as a whole is a 'prototypical case study', the outcome is of further interest to policy makers and scientists, as it shows how scientific studies combined with intensive stakeholder communication leads to concrete policy advice. The study may serve as an example for future substitution studies.

All aggregated data on the LCI and the impact assessments is publicly available, with the exception of some data on FR production, where Non-Disclosure Agreements with industry partners have been signed.

2.4 Product system

The product system to be considered is a flame retardant system incorporated in a commercial electronics application. A comparison is made between a brominated flame retardant system and a non-brominated one. Any adjustments that must be made to the non-brominated application in order to meet the same property requirements as the brominated application (e.g. fire safety requirements or physical properties of the polymer) are incorporated into the study.

2.5 Functional unit

The LCA studies focus on FRs in the following applications:

- printed circuit boards
- electronic components
- injection moulded products

The study of the 3 applications above can be combined into the analysis of an electronics product. As a case study, a laptop computer was chosen.

The functional unit of this study is defined as: ‘the complete life cycle of a laptop containing flame retarded polymers’.

A comparison is made between a laptop containing BFRs and a laptop containing HFFRs. For a fair comparison of FRs, a number of criteria have to be met. These criteria are summarized in Table 1. The results from other Work Packages in the ENFIRO project (notably WP 2, 5, and 6) provide the answer which FR-polymer combinations meet these requirements.

In work package 2, a first selection has been made of suitable polymer-FR combinations, and the main areas where these combinations can be applied. These results formed the basis for a further selection of FRs and their applications, as well as for the detailed functional and environmental behaviour studies of the ENFIRO project.

Table 1: Requirements which must be fulfilled by the functional unit (adapted from Deliverable 2.1).

<i>Property</i>	<i>Requirement</i>
Fire safety	<p>Required fire safety level must be achieved, according to the legal requirements and corresponding fire tests for the specific product.</p> <p>Requirement for the flame retardant system:</p> <ul style="list-style-type: none"> - The flame retardant should commence thermal activity in the temperature range of the thermal decomposition of the polymer - The flame retardant should not generate any toxic gases beyond those produced by the degrading polymer itself and should not increase the smoke density of the burning polymer
Mechanical properties	<p>Requirement for the flame retardant system:</p> <ul style="list-style-type: none"> - Not significantly alter the mechanical properties of the host polymer - Be easy to incorporate into the host polymer - Be compatible with the host polymer - Should be stable under processing and service of life conditions
Physical properties	<p>Requirement for the flame retardant system:</p> <ul style="list-style-type: none"> - Be resistant towards ageing and hydrolysis - Not cause corrosion - Should not bleed or bloom - Should be stable under processing and service of life conditions <p>And depending on the application:</p> <ul style="list-style-type: none"> - Be colourless or at least have non-discolouring properties - Have good light stability
Electrical properties (in electronic products)	<p>Requirement for the flame retardant system:</p> <ul style="list-style-type: none"> - Should not adversely affect electrical properties in printed board laminates and plastic encapsulated devices - Must fulfil safety requirements in terms of current tracking and arcing (for connectors, plugs etc.)

Health and environmental properties	Requirement for the flame retardant system: - Not have harmful health effects - Not have harmful environmental properties
Commercial viability	Requirement for the flame retardant system: -commercially available and cost efficient

2.6 System boundaries

In this full life cycle assessment, all relevant processes in the life cycle are included (cradle-to-grave). This includes raw material extraction, fossil fuel extraction, electricity generation, transport, material manufacturing, consumer product manufacturing, product use, maintenance and waste treatment.

For the waste treatment phase, the options of recycling, waste incineration and landfill are considered, as well as the waste disposal of electronic products in non-EU countries.

The current study is a comparative LCA, and therefore aspects which are identical for the compared scenarios may be left out of the calculations. This could be the case with the application of FRs in electronics, where the energy consumption during the use phase will probably be equal for all scenarios. The energy consumption in the use phase of electronic products will give a relatively large contribution to the total environmental profile over the life cycle of the product (estimated to be about half of the overall environmental impact), and will probably not be influenced by the choice of FR.

However, in order to gain insight into the environmental impact contributions of the different phases in the life cycle of the electronics product, the use phase including energy consumption is included.

In the calculations, a distinction is made between biogenic CO₂ (originating from biomass) and fossil CO₂ (originating from fossil fuels). The environmental impact of biogenic CO₂ is set to 0, as the short cycle of biogenic CO₂ is assumed not to influence the long-term CO₂ concentration in the air. Biogenic CO₂ is not expected to play a significant role in the current study.

When process data from LCA databases are used (e.g. for transport, electricity generation or plastics blow moulding), regional specific process data for Europe are chosen, as far as possible.

When an accidental fire occurs, the different flame retardants in the products may result in different emissions, and therefore differences between the scenarios. For this reason, the occurrence of fire and the corresponding emissions are included in the study. Earlier studies including fire into LCA calculations serve as guidance for these specific calculations (the 'Fire-LCA' model (*Simonson, 2000*)).

The system boundaries of this study are set such that a cut-off is applied at End-of-Life recycling, which means that the potential costs and benefits of recycling are allocated to the production of the recycled metal or plastic, and not to the End-of-Life phase of the laptop. The fact that emissions from incineration of plastics are considered, while recycling will have a score of 0, will still lead to a better environmental performance of the (to be) recycled material.

2.7 Allocation procedures

In an LCA study, allocation procedures must be chosen for three situations:

- multi-input processes, such as waste treatment;
- multi-output processes, such as chlorine production (in which, besides chlorine, also hydrogen, NaOH and NaOCl are produced);
- allocation of avoided emissions or production, in recycling processes.

In this study, for multi-input and multi-output processes an economic allocation is used. This means that the contribution to the environmental impact of the production chain of each ‘co-product’ is considered to be proportional to its relative financial value.

2.8 LCIA methodology and software

The software used for the LCA study is SimaPro 7.3, developed by PRé consultants, Amersfoort, The Netherlands. This dedicated LCA-software is used to compile all inventory data, calculate the environmental impacts, and perform sensitivity analyses.

The impact assessment method ReCiPe 1.03 is applied in this LCA. The full range of environmental impact categories from this method is considered. The ReCiPe method was published in 2009, and was developed by PRé Consultants, CML Leiden, Radboud University Nijmegen and RIVM, commissioned by the Dutch ministry of Housing, Spatial Planning and the Environment (VROM). It is a combined midpoint and endpoint method, building on the internationally often used methods Eco-indicator 99 and CML2. The ReCiPe method is discussed in more detail in section 5.

For the calculation of toxicity characterization factors for FRs which are not included in the ReCiPe method, the Excel based calculation tool USES-LCA2 was used. This Uniform System for the Evaluation of Substances adapted for LCA purposes (USES-LCA) is a multimedia fate, exposure and effects model, and is based on the (E)USES model for risk assessment and multimedia fate model SimpleBox 3. It was developed at Radboud University, The Netherlands (*Van Zelm, 2009*).

2.9 Data requirements

The project aims to collect a large amount of foreground data from producers and other stakeholders, notably from the ENFIRO Stakeholder Forum. Other foreground data is produced by the ENFIRO partners, in particular data on the environmental fate of the FRs. These data are used in the calculations of the ecotoxicological and human toxicological endpoints in the LCIA phase of the study.

The ENFIRO studies on the functional and fire safety properties of the FR-polymer combinations serve to further specify which combinations comply with the requirements for the functional unit.

Data from scientific literature or publicly available patents are used in cases where actual data from stakeholders cannot be obtained.

For background data, such as emissions due to transport or electricity generation, the LCA-database Ecoinvent 2.0 is used.

When data on production processes is confidential, a decision is made in consultation with the provider of the data how to present these data in the report. In some cases, confidentiality agreements have been signed.

2.10 Critical review

ISO14044 requires that LCA studies making comparisons between products, which will be disclosed to the public, are peer reviewed by an external independent LCA expert. For the current study, Swerea IVF (Mölndal, Sweden) acts as independent expert.

The objective of the review is to challenge and test the assumptions, methods, results and conclusions of the study. Consensus with respect to the study conclusions is sought, but cannot be guaranteed.

Stakeholders and the external LCA expert have been involved early on in the project, to prevent new major criticisms arising in the end phase of the project. In order to make optimal use of the review process, a “goal & scope“ report describing the design of the study has been reviewed, then a second report describing the inventory phase, and then the final report including all results and conclusions have been reviewed.

The review report has been added to this report in Annex 5.

2.11 The ReCiPe method

The result of the inventory phase is a long list of emissions, materials consumed, and other data. It is not possible to decide from this list which of the scenarios studied is more environmentally benign. Therefore, the life cycle impact assessment (LCIA) transforms these inventory data into a set of scores on a limited amount of environmental impact categories. In this study, LCIA results will be calculated with the relatively new method ReCiPe.

Impact categories indicate the severity of an environmental effect. ReCiPe considers impact categories at three levels:

1. Eighteen midpoint level categories
2. Three endpoint level categories
3. One single score indicator

ReCiPe uses environmental mechanisms as the basis for modeling. An environmental mechanism can be seen as a chain of causes and effects that together result in a certain level of damage to e.g. human health or ecosystems. For example, for climate change it is known that certain substances cause an increase in radiative forcing, which reduces the radiation of heat from earth to space. The result is that more energy remains in the atmosphere, and average temperatures increase. This will result in changes in natural habitats of organisms, and possibly to extinction of a number of species. This cause-effect chain is illustrated in Figure 2.

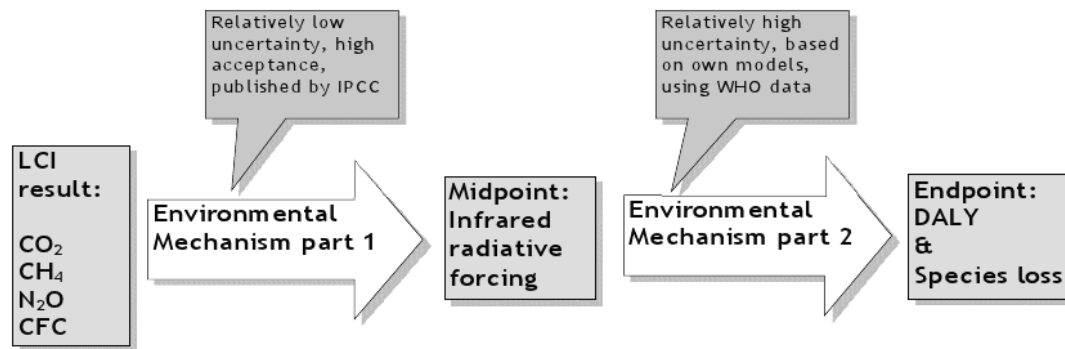


Figure 2 Example of a harmonized midpoint-endpoint model for climate change, coupled to ecosystem damage [source: www.lcia-recipe.net]

This example demonstrates that when the environmental mechanism (cause-effect chain) becomes longer, the uncertainties increase. Radiative forcing (“IPCC equivalency factors” in figure 1) is a physical property, which can be experimentally determined relatively easily. Global average temperature increase as a consequence of radiative forcing is more difficult to determine, as this is caused by a large number of processes, including positive and negative feedback mechanisms. Our understanding of the resulting changes in natural habitats is also incomplete, and therefore the expected species loss is even harder to predict.

The advantage of considering only the midpoint level impact categories (e.g. CO₂-equivalents) is the relatively low uncertainty, but the disadvantage is that a value for radiative forcing does not tell us anything about the actual damage caused. Therefore a midpoint level score is more difficult to interpret and compare to other environmental effects.

In ReCiPe, factors are calculated for 18 relatively robust midpoint level categories, as well as three more uncertain endpoint categories (Fig. 2 and Table 2). The reason to include endpoint categories is that the list of midpoint categories is still hard to interpret, partly because there are so many, and partly because they are rather abstract. How does one compare radiative forcing with soil saturation numbers (a measure for acidification)? The endpoint level categories are meant to facilitate interpretation of the results, as they are only three and are easier to understand.

The endpoint level categories are:

- Damage to human health (expressed as (disability adjusted) life years lost)
- Damage to ecosystems (species lost * time)
- Depletion of resources (expressed as “increased cost of extraction”)

The user can decide at which level the LCIA results should be shown, thereby choosing between higher uncertainties in the values themselves or higher uncertainty of a correct interpretation of results by the reader.

In the current study, both midpoint and endpoint level results are shown.

Figure 3 shows the general structure of the method.

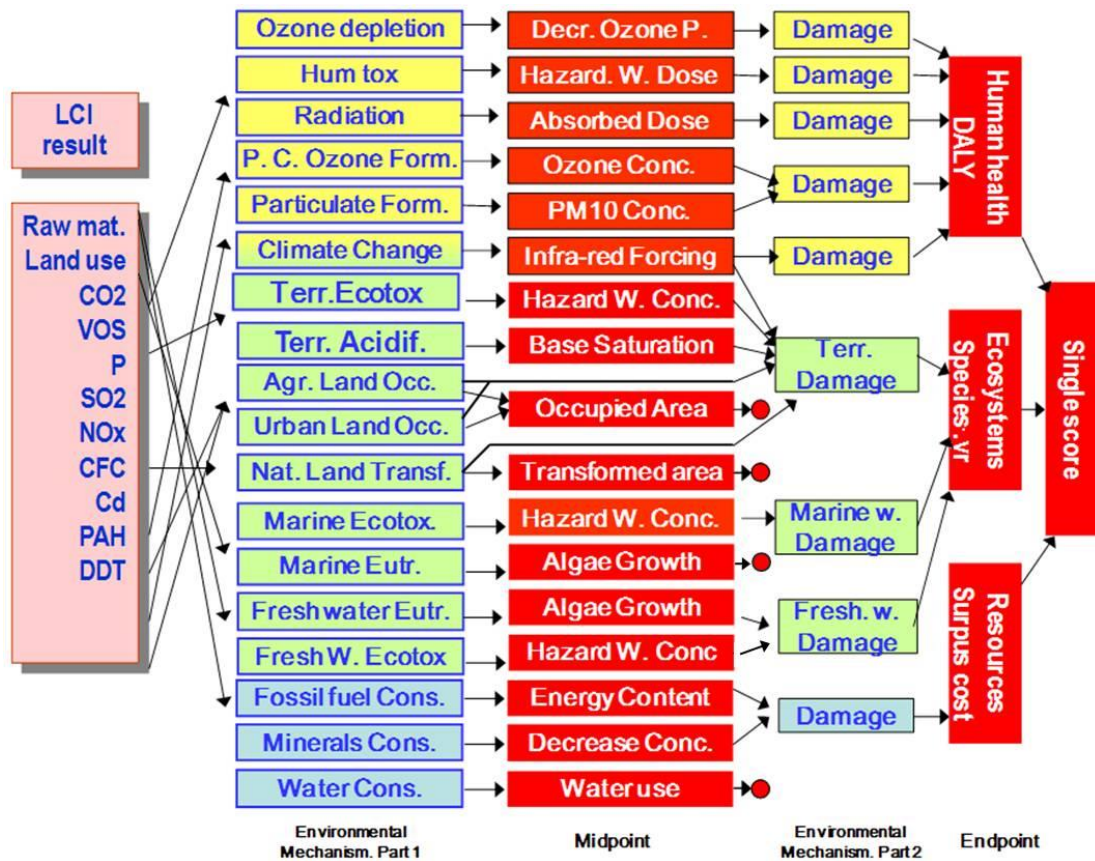


Figure 3 General structure of the ReCiPe method [source: www.lcia-recipe.net]

Note that water consumption and marine eutrophication are not contributing at the endpoint level. Climate change (CO₂-equivalents) is subdivided into two midpoint level categories: one contributing to the endpoint Human Health and one contributing to the endpoint Ecosystem Damage.

Table 2 lists an overview of the impact categories, and their (midpoint and endpoint) units. A brief description of each impact category can be found in Annex 1.

To arrive at one final Single Score of the Total impact, the endpoint categories must be weighed. The ReCiPe H/A weighing set will be used, with normalization for Europe. This weighing set gives a weight of 40% to human health and ecosystems, and a weight of 20% to resource depletion.

Table 2 Midpoint level impact categories in ReCiPe.

	<i>Impact category</i>	<i>Midpoint unit</i>	<i>Endpoint unit</i>
1	Climate change - Human Health	kg CO ₂ -eq.	DALY ^(a)
	Climate change - Ecosystems	kg CO ₂ -eq.	species.yr
2	Ozone depletion	kg CFC-11-eq.	DALY
3	Terrestrial acidification	kg SO ₂ -eq.	species.yr
4	Freshwater eutrophication	kg P-eq.	species.yr
5	Marine eutrophication ^(b)	kg N-eq.	
6	Human toxicity	kg 1,4-DCB-eq.	DALY
7	Photochemical oxidant formation	kg NMVOC	DALY
8	Particulate matter formation	kg PM10-eq.	DALY
9	Terrestrial ecotoxicity	kg 1,4-DCB-eq.	species.yr
10	Freshwater ecotoxicity	kg 1,4-DCB-eq.	species.yr
11	Marine ecotoxicity	kg 1,4-DCB-eq.	species.yr
12	Ionising radiation	kg ²³⁵ U-eq.	DALY
13	Agricultural land occupation	m ² a	species.yr
14	Urban land occupation	m ² a	species.yr
15	Natural land transformation	m ²	species.yr
16	Water depletion ^(b)	m ³	
17	Minerals depletion	kg Fe-eq.	\$
18	Fossil fuel depletion	kg oil-eq.	\$

(a) Disability-adjusted life year.

(b) These categories are not considered at the endpoint level.

3. Life Cycle Inventory

In this chapter, the detailed inventory of inputs and outputs for each phase in the life cycle of the selected flame retardants is described.

3.1 Approach

The life cycle of FRs was constructed on the basis of many contacts in industry, NGO and academia, notably through the ENFIRO Stakeholder Forum.

Details on specific phases in the life cycle were retrieved from (freely accessible) LCA studies on electronics products, specific scientific literature and experimental results from the ENFIRO project.

For details on the production of FRs, contacts were sought with the main European FR producers, and questionnaires were distributed, inquiring on the use of raw materials and energy, emissions and waste production. Some of the producers responded, and provided detailed data on their production processes. This was normally done under a Non Disclosure Agreement.

When no specific industry data was received, an inventory was produced based on publicly available literature, such as public LCA studies and Ullman's Encyclopedia of Industrial Chemistry. This inventory was then sent to the FR producer, giving them an opportunity to correct or complete the data.

For background processes, the Ecoinvent LCA-database was used. This is internationally the most widely used LCA-database, which contains data on a large number of industrial processes, materials, transport systems and waste treatment methods. These descriptions are mostly based on Western European data. These background processes mainly concern production of energy, mining and processing of raw materials, and transport.

Concerning the production of the laptop, details on processes or parts in which FRs do not play a significant role were copied from the laptop production data in Ecoinvent. This includes among others the batteries and LCD module. As these processes are not affected by the choice of FR, they will score equally in both scenarios, and therefore do not need to be studied in detail.

A check was made whether plastics were present in the data sheets of the production of these parts, and when present, the plastics amounts were set to 0 in those processes, to ensure they would not be double counted.

3.2 Description of the life cycle of a flame retardant in an electronics application

The product system to be considered is a combination of several flame retarded polymers incorporated in an electronics product. A laptop computer has been selected as case study.

Some literature on LCA studies of electronics is available (*Dodbiba et al., 2008, Duan et al., 2009, Citroth et al., 2011*). Results vary considerably, depending on system boundaries, data sources and assumptions.

A common view in the LCA community is that 'any product with a plug' will have its main environmental impact in the use phase. Although this is probably true for ovens and washing machines, it does not hold for all electronics. In some studies, environmental impact scores for the manufacturing phase and use phase are comparable, and sometimes the manufacturing phase scores higher.

The score for the waste phase depends very much on the scenario: dumping on a landfill can have a score similar to the manufacturing and use phase, while the score for proper WEEE treatment scores lower or even negative (environmental benefit due to recycling). No quantitative LCA scores have been found for the improper treatment of WEEE in developing countries.

The environmental impact of FRs is normally not discussed in LCA studies on electronics. Sometimes, the plastics used in electronics are considered to be simple polymers, ignoring any additives that may be used (Ciroth *et al.*, 2011, Dodbiba *et al.*, 2008). In those studies, the environmental impact of the metals used in electronics by far outweighs the impact of the plastics used.

Figure 4 shows the life cycle for an FR that is incorporated as additive into a polymer, which is then used in an electronics product.

The life cycle of a FR starts with mining/extraction of raw materials such as bromine, phosphor and oil from nature. These raw materials are processed to form the base chemicals from which the FR is produced.

The FR is then incorporated into a material such as a plastic or resin. Possibly, several materials are subsequently combined to produce components such as printed wiring boards (PWBs). Components and materials are then used to produce the finished product, in this case study the laptop.

In the use phase of the product, energy use, maintenance and cleaning occur.

At the end of the use phase, the discarded product is collected, or possibly a fraction of the products is disposed of illegally. The discarded product may then be exported, disassembled or separated in some way.

Finally, part of the FR-containing product may be recycled, landfilled or burned in a waste incineration plant (where electricity and heat may be generated). Recycled material will enter the same production phase again, or may be used as material in a different type of product.

Each of the phases described in this life cycle may include transport, energy use and use of other chemicals or materials, which all contribute to the environmental impact of the life cycle of the FR.

Emission of FRs to the environment may occur in several phases of the life cycle.

Firstly, during the FR production phase emissions may occur. In 2004, a program called VECAP (voluntary emissions control action programme) was initiated by the 3 largest BFR producers and 130 BFR users' sites, in order to monitor and reduce these emissions. VECAP reports that residues of BFRs in packaging waste are the main emission route during the production phase (VECAP 2009).

A second phase in the life cycle where FR emissions can be expected is the use phase. Indications for this are reports on relatively high concentrations of FRs in house dust (Harrad *et al.* 2010, Stapleton *et al.* 2008, Stapleton *et al.* 2009). However, OECD emission scenarios indicate that the emission percentage is rather low: an estimated 0.05% of the FRs are expected to volatilize during the lifetime of the product (OECD, 2004).

In the waste phase, FR emissions can occur during all of the possible waste treatment options (Morf *et al.* 2005, Leung *et al.* 2007, Guo *et al.* 2009). Leaching of FRs from materials after illegal dumping or storage in landfills can occur (Osako *et al.* 2004, Choia *et al.* 2009). When discarded products are incinerated (in European waste incineration plants or over open fires after export outside the EU), FRs may be released, or toxic substances like dioxins and furans may be formed out of the FRs (Gullett *et al.* 2007).

Emissions of other substances such as CO₂, NO_x and SO_x can occur in each phase of the life cycle, and are quantified in this study by using existing databases, reports on emissions by companies and scientific literature.

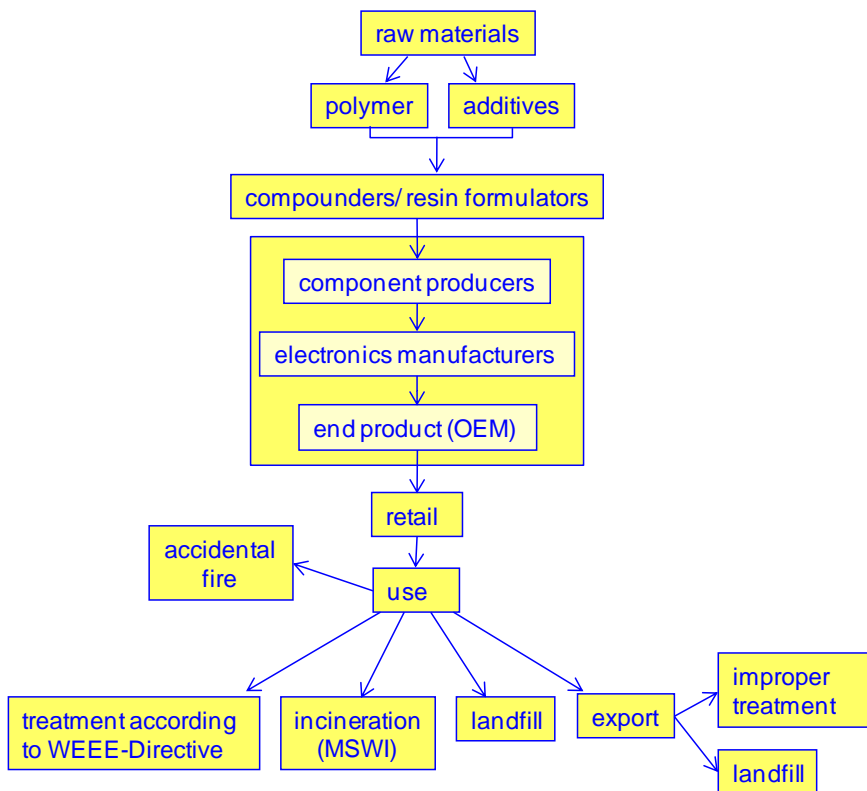


Figure 4 The life cycle of flame retardants in an electronics application.

3.3 Flame retarded polymers in a laptop

Based on the work done in other ENFIRO work packages (2 to 5), a selection was made of polymer-FR combinations to be studied in detail. Based on this selection, the LCA case study was chosen. A laptop computer was considered as an ideal case study, as in this product several of the polymer-FR combinations are incorporated, i.e. PA6,6 in connectors, EVA in cables, PPE/HIPS or PC/ABS in the computer casing, and epoxy resins in the printed wiring boards.

Amounts of plastic in a laptop (total laptop weight 3.15 kg) are based on several (LCA) studies from literature (see for further details paragraph 3.6 and Annex 2). The FR loadings are based on fire safety tests performed within the ENFIRO project, and apply to materials passing a V0 test.

The amounts of the polymer-FR systems considered in laptop are shown in table 3.

Since the values in table 3 are averages based on several studies, they refer to a theoretical case and not any existing laptop brand or model. According to some industry sources, some laptops are in production which do not contain flame retardants at all.

In the column with HFFR options in Table 3, in most cases two alternative options are given. In each case, the first option is selected for the main HFFR scenario. The second alternative has been modelled in a sensitivity analysis (see paragraph 5.1.1).

Further discussions on the composition and production of a laptop can be found in paragraph 3.5.

Table 3 The polymer-FR systems studied. Amount of polymer present in the laptop and % loading of the FR in the polymer are indicated.

Polymer	Amount in laptop (kg)	BFR scenario		HFFR scenario		Part type
		compound	% weight	compound	% weight	
PA6,6 + GF	0.114	brominated PS + ATO	18 + 7	Alpi + MPP, (Alpi + MPP + ZS)	16.7 + 8.3 , (13.3 + 6.7 + 3)	connectors, switches
EVA	0.159	decaBDE + ATO	13 + 4	ZHS-coated ATH	70	cables
PPE/HIPS	0.516	decaBDE + ATO	14 + 6	RDP, (BDP)	15 , (15)	casing
PC/ABS	0.651	decaBDE + ATO,	10 + 5	BDP, (RDP)	20, (20)	casing
epoxy resin	0.293	reactive TBBPA	20	DOPO, (Alpi + ATH)	25, (15 + 45)	PWB

3.4 Production of flame retardants

For the detailed inventory on the production of FRs, the main European FR producers were contacted, and questionnaires were distributed, inquiring on the use of raw materials and energy, emissions and waste production. Some of the producers responded, and provided detailed data on their production processes (normally under a Non Disclosure Agreement).

When no specific industry data was received, an inventory was made using publicly available literature, such as public LCA studies, patent descriptions and Ullman's Encyclopedia of Industrial Chemistry. This inventory was then sent to the FR producer, giving them an opportunity to correct or complete the data. When using general literature, amounts of chemicals used were often based on stoichiometric ratios in the chemical reactions involved. This corresponds to the minimum amount of chemicals needed: in practice the use of chemicals will always be higher due to non-complete conversion of chemicals, spillage, and a fraction of the product being of inferior quality. As it is impossible to estimate these losses without actual industry data, the stoichiometric amounts were used as objective data.

For the production of base chemicals, mining of raw materials, energy production and transport, data from the Ecoinvent database were used.

Emissions of FRs during production of FRs were in most cases estimated using the EU Technical Guidance Document on risk assessment (2003). In some cases, more specific data was available, as is discussed in the paragraphs below.

3.4.1 Antimony trioxide (ATO)

Data on the production route of ATO was taken from the EU risk assessment on ATO (2008) and from an LCA study on TVs (Simonson, 2000).

The first step in ATO production is the mining of the mineral stibnite (Sb_2S_3). Stibnite is present at several locations in the world, but the largest production occurs in China. In the current study, the stibnite is assumed to originate from China. Processing steps up until the production of crude ATO occur near the mine, therefore the Chinese energy production mix is used for those processing steps.

No data sheet for stibnite production was available in Ecoinvent, therefore the data sheet on the mining of tin was chosen as a starting point. This data set was adjusted for the specific energy consumption, as reported by Simonson (2000). In addition, the extraction of tin from the ground was replaced by the extraction of antimony, and transport to Europe was deleted in this process step.

As specific emissions to air and water are not very detailed in the tin mining process, an average was taken here of four mining processes from Ecoinvent: tin, zinc, copper and lead.

The first processing step is crushing the ore from particles of around 125 mm to around 12.5 mm. The Ecoinvent process for crushing rock is used, adjusted for specific energy consumption reported by Simonson (2000). Then, a further grinding step follows, to arrive at particles of around 0.1 mm. Sb_2S_3 is then extracted from the stibnite ore using flotation techniques. A drying/dewatering step then follows, often using a press filter. Energy consumption for each of these steps is taken from Simonson (2000).

Processing of purified stibnite to ATO can be performed via two production routes. Both routes are of equal importance, and are each followed for 50% in this study.

The first route involves oxidation of Sb_2S_3 to Sb_2O_3 at around 850-1000°C, which results in emissions of SO_2 . This is followed by a purification process called revolatilization, in which the Sb_2O_3 is vaporized and condensed. In the second route, Sb_2S_3 is reduced to Sb using iron and coal, to and then oxidized to Sb_2O_3 . The inventory data for both production routes are taken from Simonson (2000).

Finally, ATO is transported from China (Hong Kong) to Europe (Rotterdam, NL) by freight ship.

The main inventory data for the production of purified stibnite and ATO are listed in table 4 and 5, respectively.

Table 4 Main inventory data for mining and purification of stibnite (1 ton purified stibnite). Further data is used from the corresponding Ecoinvent data sheets as described above.

<i>amount</i>	<i>product</i>	<i>Process step</i>
1000 kg	Antimony, in ground (mineral depletion)	Mining of stibnite
2.41 MJ	Energy, from oil	Mining
3 kWh	Electricity, production mix China (process sheet 'Crushing, rock/RER')	Crushing
15 kWh	Electricity, production mix CN	Grinding
4.5 kWh	Electricity, production mix CN	Flotation
0.5 kWh	Electricity, production mix CN	Drying/dewatering

Table 5 Main inventory data for production of ATO out of purified stibnite (1 ton ATO). Further data is used from the corresponding Ecoinvent data sheets as described above.

<i>amount</i>	<i>product</i>	<i>Process step</i>
1165 kg	Purified stibnite input (stoichiometric)	input
5.83 MJ	Electricity, production mix CN	oxidation of Sb ₂ S ₃ to Sb ₂ O ₃ , route 1 (50%)
330 kg	Emission of SO ₂ . 659 kg/ ton Sb ₂ O ₃	oxidation of Sb ₂ S ₃ to Sb ₂ O ₃ , route 1 (50%)
100 MJ	Electricity, production mix CN	revolatilization of Sb ₂ O ₃ , route 1 (50%)
450 MJ	Heat, at hard coal industrial furnace	revolatilization of Sb ₂ O ₃ , route 1 (50%)
2.91 MJ	Electricity, production mix CN. 5 MJ/1000 kg Sb ₂ S ₃ .	Reduction of Sb ₂ S ₃ to Sb, route 2 (50%)
291 MJ	Heat, at hard coal industrial furnace. 0,5 MJ/kg Sb ₂ S ₃	Reduction of Sb ₂ S ₃ to Sb, route 2 (50%)
287 kg	Pig iron	reduction of Sb ₂ S ₃ to Sb, route 2 (50%)
0.5 MJ	Electricity, production mix CN: 1 kWh/1000 kg Sb ₂ O ₃	Oxidation of Sb to Sb ₂ O ₃ , route 2 (50%)
18500 tkm	Transport, transoceanic freight ship/OCE	Hong Kong - Rotterdam: 18500 km

3.4.2 Bromine

Bromine is produced out of bromide-rich brines, following the Kubierschky process. This is a steaming-out process, in which the brine is heated to about 92°C, and chlorine is passed through it, oxidizing the bromide ions to molecular bromine (Ullman's, 2002).

Production of bromine occurs mainly in Israel and in the USA. The brines at these locations have different bromide concentrations: around 13000 ppm in Israel (Dead Sea water), and 5000 ppm in the USA. Consequently, more brine is needed to produce bromine in the USA than in Israel: around 200,000 and 77,000 L, respectively. This also leads to a higher energy consumption for this process in the USA than in Israel.

In the current study, bromine is considered with 50% production in Israel and 50% in the USA.

As bromine is a hazardous compound, industry's general policy is to limit transports of pure bromine. It is therefore assumed that the bromine is processed into BFRs in close proximity of the bromine production plant.

Table 6 Main inventory data for production of bromine out of brine (1 ton bromine).

<i>amount</i>	<i>product</i>	<i>Process step</i>
444.3 kg	Chlorine, liquid, production mix	Oxidation of bromide ions to molecular bromine
23.2 GJ	Heat, natural gas, industrial furnace >100 kW	Heating of brine, Israel (50%)
60.2 GJ	Heat, natural gas, industrial furnace >100 kW	Heating of brine, USA (50%)

3.4.3 Brominated polystyrene (BPS)

The production of brominated polystyrene (BPS) occurs by brominating polystyrene at elevated temperatures, with at least part of the process occurring at 300°C (Albemarle, 2000).

The degree of bromination of the PS chain can vary per product or manufacturer. In the current study, an existing product was chosen with 2.7 bromine atoms per PS unit (ICL, 2011).

For the polymer, the Ecoinvent process sheet for polystyrene (general purpose, GPPS) production was used.

No further production details have been found. Therefore, as an estimation, energy consumption data were used from an Ecoinvent process sheet on a similar process involving halogenation of aromatic compounds: the production of mono- and dichlorobenzene.

Table 7 Main inventory data for production of brominated polystyrene out of polystyrene and bromine (1 ton BPS).

<i>amount</i>	<i>product</i>	<i>Process step</i>
680.2 kg	bromine	Bromination of polystyrene (stoichiometric amount)
328.2 kg	Polystyrene, general purpose	Bromination of polystyrene (stoichiometric amount)
32 kWh	Electricity, medium voltage	Bromination of polystyrene (copied from production process for mono- and dichlorobenzene (Ecoinvent))
808 MJ	Heat, unspecific, in chemical plant	Bromination of polystyrene (copied from production process for mono- and dichlorobenzene (Ecoinvent))

3.4.4 Decabromo diphenylether (decaBDE)

DecaBDE is produced by bromination of diphenyl ether, at temperatures around 59°C, in the presence of a base, and using a catalyst (ECB, 2007).

The starting material diphenyl ether is produced out of phenol and bromobenzene (Ullman's, 2002). For this process, the Ecoinvent production process sheet "Diphenylether-compounds" was used.

As further production details were lacking for the bromination of diphenyl ether, values for estimated energy consumption were copied from an Ecoinvent process sheet on a similar process involving halogenation of aromatic compounds: the production of mono- and dichlorobenzene.

Table 8 Main inventory data for production of decaBDE out of diphenyl ether and bromine (1 ton decaBDE).

<i>amount</i>	<i>Product</i>	<i>Process step</i>
833 kg	Bromine	Bromination of diphenyl ether (stoichiometric amount)
178 kg	Diphenylether compounds (Ecoinvent process)	Bromination of diphenyl ether (stoichiometric amount)
500 kg	sodium hydroxide (50% in water, production mix)	Bromination of diphenyl ether
32 kWh	Electricity, medium voltage	Bromination of diphenyl ether (copied from production process for chlorobenzenes (Ecoinvent))
808 MJ	Heat, unspecific, in chemical plant	Bromination of diphenyl ether (copied from production process for chlorobenzenes (Ecoinvent))

3.4.5 Tetrabromobisphenol-A (TBBPA)

TBBPA is produced by bromination of bisphenol-A (EC, 2007).

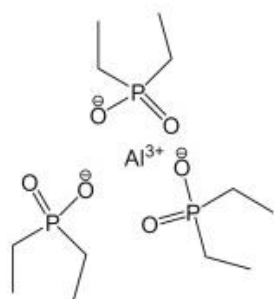
Bisphenol-A (BPA) is formed in a condensation reaction of phenol and acetone. In the Ecoinvent database a production process sheet is present for BPA, which is used in this study.

Similar to decaBDE and decaBDethane, values for estimated energy consumption in the bromination of BPA were copied from an Ecoinvent process sheet on a similar process involving halogenation of aromatic compounds: the production of mono- and dichlorobenzene.

Table 9 Main inventory data for production of TBBPA out of bisphenol-A and bromine (1 ton TBBPA).

amount	Product	Process step
588 kg	Bromine	Bromination of diphenyl ethane (stoichiometric amount)
420 kg	Diphenylether compounds (Ecoinvent process)	Bromination of diphenyl ethane (stoichiometric amount)
500 kg	sodium hydroxide (50% in water, production mix)	Bromination of diphenyl ethane
32 kWh	Electricity, medium voltage	Bromination of diphenyl ethane (copied from production process for chlorobenzenes (Ecoinvent))
808 MJ	Heat, unspecified, in chemical plant	Bromination of diphenyl ethane (copied from production process for chlorobenzenes (Ecoinvent))

3.4.6 Aluminium diethylphosphinate (Alpi)



Aluminium diethylphosphinate is a metal phosphinate salt prepared from diethylphosphinic acid and aluminium hydroxide or aluminium sulfate (Clariant, 2002).

The raw material diethylphosphinic acid is produced from sodium hypophosphite (NaH_2PO_2) and ethylene in the presence of a free-radical initiator (Clariant, 2002).

Clariant is the main producer of Alpi, and has shared detailed data on raw materials, energy consumption and production processes. These data are confidential and not shown in this report. The data were used in the current study.

3.4.7 Aluminium trihydroxide (ATH)

Mining of bauxite (a mixture of $\text{Al}(\text{OH})_3$, $\text{AlO}(\text{OH})$, and some other minerals) is the first step in the production of ATH. Bauxite is then refined using the Bayer process, which involves a digestion step by washing with a hot solution of sodium hydroxide at around 175 °C. The $\text{Al}(\text{OH})_3$ dissolves in the hydroxide solution, while other components do not dissolve. After filtration and cooling, the aluminium hydroxide is precipitated. The solid impurities in the filtration residu are called red mud, which are usually disposed in large ponds, and neutralize in contact with air over several years.

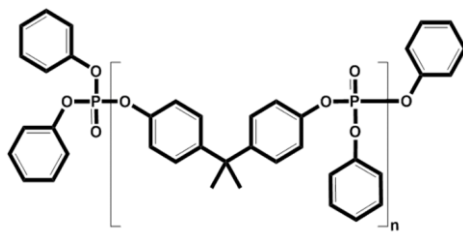
The energy intensive process steps necessary to produce metallic aluminium (calcination and electrolysis) are not required for the production of the FR ATH.

In this study, the production of ATH is modeled using the Ecoinvent data sheet “Aluminium hydroxide, at plant”.

Table 10 Main inventory data for production of ATH out of bauxite (1 ton ATH). Further data is used from the corresponding Ecoinvent data sheet as described above.

amount	product	Process step
1400 kg	bauxite	
39 kg	sodium hydroxide, 50% in H ₂ O	Refining of bauxite
30 kg	quicklime	Refining of bauxite
140 kWh	Electricity, medium voltage	Refining of bauxite
4038 MJ	Heat, from hard coal, natural gas, and light fuel oil	Refining of bauxite

3.4.8 Bisphenol-A bis(diphenylphosphate) (BDP)



BDP is a polymeric bisarylphosphate, consisting of an oligomer mixture with different chain lengths. For this study, the product has been simplified to a single compound of $n=1$ (see figure). In a product catalogue of one of the main producers, the given phosphorus content of this product (8.9%) also refers to the $n=1$ situation, suggesting this is an acceptable simplification (ICL, 2011).

Bisarylphosphates are produced from phosphoryl chloride (phosphorus oxychloride) and various phenols, in this case bisphenol-A and phenol (EFR4, 2012). For production of the three starting materials, the corresponding product sheets available in the Ecoinvent database were used.

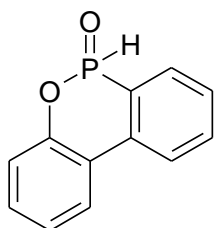
No further details on production processes could be found. For the reaction of bisphenol-A, phenol and phosphoryl chloride to RDP, as an estimated energy consumption values were copied from two Ecoinvent process sheets on processes also involving a phenolic reaction: the production of cyclohexanol¹.

¹ In another Ecoinvent sheet concerning a reaction with phenols, production of and bisphenol-A, equal energy consumption amounts are reported.

Table 11 Main inventory data for production of 1 ton BDP.

amount	product	Process step
330 kg	bisphenol-A	Condensation reaction bisphenol-A, phenol and phosphoryl chloride (stoichiometric amount)
544 kg	phenol	Condensation reaction bisphenol-A, phenol and phosphoryl chloride (stoichiometric amount)
443 kg	phosphoryl chloride	Condensation reaction bisphenol-A, phenol and phosphoryl chloride (stoichiometric amount)
333 kWh	Electricity, medium voltage	Condensation reaction bisphenol-A, phenol and phosphoryl chloride (copied from production processes involving other phenolic reactions (Ecoinvent))
2000 MJ	Heat, natural gas, at industrial furnace >100kW	Condensation reaction resorcinol, phenol and phosphoryl chloride (copied from production processes involving other phenolic reactions (Ecoinvent))

3.4.9 9,10-dihydro-9-oxa-10-phosphaphenanthrene-10-oxide (DOPO)



DOPO is a cyclic hydrogen phosphinate FR made from *o*-phenylphenol and phosphorus trichloride (EPA, 2008). Further details are lacking for this reaction, and therefore values for estimated energy consumption were copied from Ecoinvent process sheets on other production processes involving phenolic reactions.

The starting material ortho-phenylphenol is produced in a two-step process out of cyclohexanone. The first stage is a condensation of cyclohexanone using sulphuric acid at around 100°C to form *o*-cyclohexenyl cyclohexanone. After neutralization with sodium carbonate, the reaction mixture is dehydrogenated at around 280°C to form *o*-phenylphenol (King, 1977). Only the energy needed for heating of the reactants is included in the inventory.

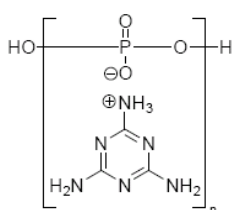
Table 12 Main inventory data for production of 1 ton *o*-phenylphenol.

amount	product	Process step
1153 kg	cyclohexanone	Condensation of cyclohexanone (stoichiometric amount)
231 kg	sulphuric acid	Condensation of cyclohexanone
250 kg	sodium carbonate	Neutralization of reaction mixture
149 MJ	Heat, unspecified, in chemical plant.	Energy just for heating to 100°C (condensation step).
348 MJ	Heat, unspecified, in chemical plant.	Energy just for heating to 280°C (dehydrogenation step).

Table 13 Main inventory data for production of DOPO out of o-phenylphenol (1 ton DOPO).

amount	product	Process step
787 kg	o-phenylphenol	DOPO production (stoichiometric amount)
635 kg	Phosphorous chloride (Ecoinvent process)	DOPO production (stoichiometric amount)
333 kWh	Electricity, medium voltage	DOPO production (copied from production processes involving other phenolic reactions (Ecoinvent))
2000 MJ	Heat, natural gas, at industrial furnace >100kW	DOPO production (copied from production processes involving other phenolic reactions (Ecoinvent))

3.4.10 Melamine polyphosphate (MPP)



To prepare melamine polyphosphate, melamine and phosphoric acid are mixed and heated at a temperature of 260-280°C (Lin et al., 2010; Fu et al., 2011). No further details on energy consumption were found, and therefore values were copied from a different Ecoinvent production processes involving phosphate reactions: the production of ammonium nitrate phosphate.

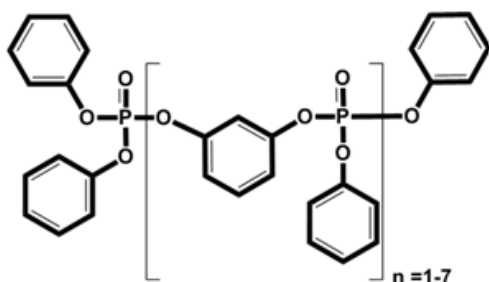
The production sheets for melamine and phosphoric acid were both taken from the Ecoinvent database.

Table 14 Main inventory data for production of MPP out of melamine and phosphoric acid (1 ton MPP).

amount	product	Process step
563 kg	melamine	MPP production (stoichiometric amount)
514 kg	phosphoric acid, industrial grade, 85% in H ₂ O	MPP production (stoichiometric amount, corrected for dilution in water)
253 kWh	Electricity, medium voltage (copied from production processes involving phosphate reactions, Ecoinvent database)	MPP production (copied from a different production process involving a phosphate reaction (Ecoinvent))
2119 MJ	Heat, natural gas, at industrial furnace >100kW (copied from production processes involving phosphate reactions, Ecoinvent database)	MPP production (copied from a different production process involving a phosphate reaction (Ecoinvent))

3.4.11 Resorcinol bis(diphenylphosphate) (RDP)

RDP is a polymeric bisarylphosphate, consisting of an oligomer mixture with chain lengths between 1 and 7. For this study, the product has been simplified to a single compound of n=1 (see figure). In a product catalogue of one of the main producers, the given phosphorus content of this product (10.7%) also refers to the n=1 situation, indicating that this simplification is acceptable (ICL, 2011).



Bisarylphosphates are produced from phosphoryl chloride (phosphorus oxychloride) and various phenols, in this case resorcinol and phenol (EFRA, 2012).

No further details on production processes could be found. For the reaction of resorcinol, phenol and phosphoryl chloride to RDP, as an estimated energy consumption values were copied from two Ecoinvent process sheets on a

processes also involving phenolic reactions: the production of bisphenol-A and the production of cyclohexanol. In both sheets, equal energy consumption amounts are reported.

The starting material resorcinol is produced mainly in the USA and in Japan. The production route starts with the sulfonation of benzene. The reaction mixture is neutralized, and then the benzene disulfonic acid is treated with sodium hydroxide at high temperature (a process called alkali fusion). The reaction mixture is then acidified to yield resorcinol (*Durairaj, 2005*).

Details on energy consumption in these processes could not be found. The estimated energy consumption was therefore copied from an Ecoinvent process sheet on a processes also involving benzene: the production of dichlorobenzene.

Table 15 Main inventory data for production of 1 ton resorcinol.

amount	product	Process step
709 kg	Benzene	Resorcinol production (stoichiometric amount)
1782 kg	Sulphuric acid, liquid	Resorcinol production (stoichiometric amount)
1450 kg	Sodium hydroxide, 50% in H ₂ O, production mix	Resorcinol production (stoichiometric amount, corrected for the dilution in water)
32 kWh	Electricity, medium voltage	Resorcinol production (copied from production processes involving other benzene reaction (Ecoinvent))
808 MJ	Heat, unspecific, in chemical plant	Resorcinol production (copied from production processes involving a different benzene reaction (Ecoinvent))

Table 16 Main inventory data for production of RDP out of resorcinol (1 ton RDP).

amount	product	Process step
192 kg	resorcinol	Condensation reaction resorcinol, phenol and phosphoryl chloride (stoichiometric amount)
655 kg	phenol	Condensation reaction resorcinol, phenol and phosphoryl chloride (stoichiometric amount)
534 kg	phosphoryl chloride	Condensation reaction resorcinol, phenol and phosphoryl chloride (stoichiometric amount)
333 kWh	Electricity, medium voltage	Condensation reaction resorcinol, phenol and phosphoryl chloride (copied from production processes involving other phenolic reactions (Ecoinvent))
2000 MJ	Heat, natural gas, at industrial furnace >100kW	Condensation reaction resorcinol, phenol and phosphoryl chloride (copied from production processes involving other phenolic reactions (Ecoinvent))

3.4.12 Zinc hydroxystannate (ZHS)

Zinc hydroxystannate (ZnSn(OH)₆) is produced in a decomposition reaction by combining in an aqueous solution a water soluble hydroxystannate (often sodium hydroxystannate) and a water soluble zinc salt. (often zinc chloride) (*Cusack et al., 1990*). After separation of the product by filtration or centrifugation, a drying step is often included at around 110°C. The energy use for heating is included in the inventory.

The starting material $ZnCl_2$ is prepared from zinc oxide and hydrogen chloride (*Brynestad 1978*). No specific energy consumption data were found, therefore the energy consumption was copied from an Ecoinvent process sheet for production of a different metal chloride: lithium chloride. Zinc production occurs in mainly China, followed by Australia and Peru (*USGS, 2011*). Zinc is transformed into zinc oxide by the Indirect (French) method (*International Zinc Association, 2011*). To model zinc oxide production, the Ecoinvent process sheet “Zinc oxide, at plant” was modified by exchanging iron scrap by the stoichiometric amount of zinc metal as raw material². For the zinc metal, a mixture of 85% primary zinc and 15% iron scrap was used, which reflects the current recycled material content of zinc. Sodium hydroxystannate is produced from tin and sodium hydroxide (*McIlbenny, 1929*). No further details are known for this process. The largest tin production occurs in China, followed by Indonesia and Peru (*USGS, 2011*). The Ecoinvent process sheet was used for production of this raw material.

In the case study, ZHS is present as a coating on ATH. This ZHS-coated ATH is assumed to consist of 25% ZHS and 75% ATH by weight (*Hornsby, 2003*).

Table 17 Main inventory data for production of 1 ton zinc chloride.

amount	Product	Process step
597 kg	Zinc oxide (modified Ecoinvent process)	Production of $ZnCl_2$ (stoichiometric amount)
1780 kg	Hydrochloric acid, 30% in H_2O	Production of $ZnCl_2$ (stoichiometric amount, corrected for dilution)
2050 MJ	Electricity, medium voltage	Production of $ZnCl_2$ (copied from a production process for a different metallic chloride salt (Ecoinvent))

Table 18 Main inventory data for production of 1 ton sodium hydroxystannate.

amount	product	Process step
445 kg	Tin, at regional storage	Production of $Na_2Sn(OH)_6$ (stoichiometric amount)
1803 kg	Sodium hydroxide, 50% in H_2O , production mix	Production of $Na_2Sn(OH)_6$ (stoichiometric amount, corrected for dilution)

Table 19 Main inventory data for production of ZHS out of sodium hydroxystannate and zinc chloride (1 ton ZHS).

amount	product	Process step
476 kg	$ZnCl_2$	Production of ZHS (stoichiometric amount)
933 kg	$Na_2Sn(OH)_6$	Production of ZHS (stoichiometric amount)
79 MJ	Heat, natural gas, at industrial furnace >100kW	Energy just for heating to 110°C (drying step)

² The iron scrap in Ecoinvent was possibly based on the assumption that the Direct (American) production route is followed. This production route has been important for many years, but has declined in recent years. The Indirect (French) method uses metallic zinc, and results in a higher purity product.

3.4.13 Zinc stannate (ZS)

Zinc stannate ($ZnSnO_3$) is produced by controlled thermal dehydration of ZHS, at a temperature in the range of 250-350°C (*William Blythe, 2012*).

Table 20 Main inventory data for production of ZS out of ZHS (1 ton ZS).

amount	Product	Process step
1233 kg	Zinc hydroxystannate	dehydration of ZHS (stoichiometric amount)
246 MJ	Heat, natural gas, at industrial furnace >100kW	Energy just for heating to 300°C

3.5 Production of flame retarded polymers

For the description of the production of polymers, existing process sheets from the Ecoinvent database have been used. These descriptions are based on the most recent data from PlasticsEurope (*PlasticsEurope, 2012*). These data are representative for Western European production processes. Table 21 shows the main starting materials for the plastics considered, as well as the Ecoinvent process sheets used.

For PPE, no Ecoinvent process sheet is available. Therefore, a process sheet for a related polymer from the phenyl ether polymer class was taken as starting point, and adjusted for the differing monomer, as shown in Table 22. Hydrogen sulphide and dichlorobenzene were replaced by dimethylphenol. The monomer dimethylphenol is produced from phenol and methanol (*Weber, 2010*).

Table 21 Main starting materials for the investigated polymers, and corresponding Ecoinvent process sheets.

Polymer	Main starting materials	Ecoinvent process sheet used
PA6,6 + GF	Hexamethylenediamine, adipic acid	Nylon 66, glass-filled, at plant/RER U
EVA	Ethylene, vinyl acetate	Ethylene vinyl acetate copolymer, at plant/RER U
PPE	Dimethylphenol	Polyphenylene sulfide, at plant/GLO U (adjusted)
HIPS	Styrene	Polystyrene, high impact, HIPS, at plant/RER U
PC	Bisphenol-A, phosgene	Polycarbonate, at plant/RER U
ABS	Acrylonitrile, styrene, butadiene	Acrylonitrile-butadiene-styrene copolymer, ABS, at plant/RER U
epoxy resin	Bisphenol-A, epichlorohydrin	Epoxy resin, liquid, at plant/RER U

Table 22 Main inventory data for production of 1 ton PPE out of dimethylphenol. Further data is used from the Ecoinvent data sheet for PPS as described above.

amount	Product	Process step
1000 kg	Dimethylphenol	Polymerization of dimethylphenol
926 kg	NaOH, 50% in H ₂ O, production mix	Polymerization of dimethylphenol
6 kg	N-methyl-2-pyrrolidone	Polymerization of dimethylphenol (catalyst)
333 kWh	Electricity, medium voltage	Polymerization of dimethylphenol
2000 MJ	Heat, natural gas, at industrial furnace >100kW	Polymerization of dimethylphenol

The polymers PPE and HIPS are considered to be used in a 55/45 mixture. For the polymers PC and ABS, a 70/30 mixture is considered. These ratios are taken from the ENFIRO fire tests, and therefore ensure that the material is practicable and has a V0 fire safety rating.

The flame retardants are assumed to be added during the extrusion process.

Based on experience in the ENFIRO material testing, the energy consumption of extrusion comprises mainly the heating of the materials. As a consequence, a slightly different processability caused by a different FR in the plastic will hardly influence the energy consumption of the extrusion process. Therefore, the general Ecoinvent process for plastics extrusion was not modified for different FRs or loading percentages, and has been used for all FR-polymer combinations.

As already shown in Table 3, the loading rates of FRs are again listed in Table 23.

Table 23 Loading rates (weight) for the investigated FRs in different polymers.

amount	BFR	Loading % (weight)	HFFR	Loading % (weight)
PA6,6 + GF	brominated PS + Sb ₂ O ₃	18 + 7	Alpi + MPP 2nd alternative: Alpi + MPP + ZS	16.7 + 8.3 13.3 + 6.7 + 3
EVA	decaBDE ethane + Sb ₂ O ₃	13 + 4	ZHS-coated ATH	70
PPE/HIPS	decaBDE + Sb ₂ O ₃	14 + 6	RDP 2nd alternative: BDP	15 15
PC/ABS	decaBDE + Sb ₂ O ₃ ,	10 + 5	BDP 2nd alternative: RDP	20 20
epoxy resin	reactive TBBPA ^a	20	DOPO ^a 2nd alternative: Alpi + ATH	25 15 + 45

^a: not tested in ENFIRO fire tests.

3.6 Production of a laptop

Laptops are produced in a wide range of designs, materials and chemicals used. The extent to which FRs are used in laptops can vary as well. It is therefore difficult to define the composition of a 'typical laptop'. Several studies are available showing the presence of FRs in laptops (*Brigden 2007, Destailats 2008*).

No examples have been found of laptops without FRs. However, a number of laptops exist on the market which are BFR-free (and consequently must contain alternative FRs) (*ChemSec, 2010*).

In certain laptops, FRs may not be present in all polymer materials. Most certainty exists on the presence of FRs in printed wiring boards (PWBs) and connectors (*Brigden 2007, EFRA 2010*).

In the current case study, it is assumed that all main polymer materials are flame retarded, including PWBs, connectors and switches, cables and casing.

Starting point for the exact laptop composition was the Ecoinvent process “Laptop computer, at plant/GLO U”. Data in that process sheet are based on a HP laptop from 2003 (Hischier, 2007). The weight of the laptop is 3.15 kg.

Some additional studies were found reporting detailed laptop compositions. These data are listed in Annex 2. The averages of the weight percentages of different polymers for these studies were determined, and when this average was higher than the weight percentage from the existing Ecoinvent process, the higher value was chosen for our LCA scenarios. This was done to ensure that no relevant polymers would be left out if they were coincidentally not present in the laptop considered by Ecoinvent.

The resulting amounts of polymer present in the ‘theoretical’ laptop considered in this study are likely to be somewhat higher than in individual ‘existing’ laptops, and can be considered a worst case scenario. The values are shown in Table 24.

In the Ecoinvent laptop process sheet (as well as in underlying process sheets on laptop components), the amounts of polymers were set to 0. The plastics from table 3 were added to the LCA scenario as separate processes. In this way, the environmental impact of the FR-polymers can be specifically shown, and double-counting of the polymers is avoided.

Energy use and auxiliary materials needed for the production of the laptop and its components (e.g. battery, LCD-display, DVD-drive) are all covered in the Ecoinvent process sheet. It is assumed that these values are independent of the types of FRs used.

Table 24 Amounts of polymer used in a laptop, based on Ecoinvent and other literature.

material	Amount of flame retarded polymer in laptop	
	kg	%
PA66	0.114	3.6
EVA	0.159	5.0
PPE/HIPS	0.516	16
PC/ABS	0.651	21
epoxy resin	0.293	9.3
total laptop weight	3.15	

3.7 The use phase

3.7.1 Energy consumption of the laptop

Calculation of total electricity use during the use phase of the laptop has been based on a study by Ciroth et al. (2011). We consider a laptop that is used only in an office environment. The laptop is in active mode for 7 hours during week days. When the laptop is switched off, it is assumed to stay plugged in, which means there is still some energy consumption. The lifetime of the laptop is assumed to be 4 years.

Electricity consumption of different laptop modes has been taken from the Ecoinvent database.

The total electricity consumption over the lifetime of the laptop amounts to 215 kWh. A breakdown of this number is shown in Table 25.

Table 25 Loading rates (weight) for the investigated FRs in different polymers.

Computer mode	Energy use (kWh/ hour)	Hour/ day	Days/ week	Weeks/ year	Total electricity consumption in 4 years (kWh)
Active mode	0.025	7	5	48	168
Standby mode	0.004	2	5	48	7.7
Off during work days	0.0015	15	5	48	21.6
Off during weekend	0.0015	24	2	48	13.8
Off during holidays	0.0015	24	7	4	4.0
Total over lifetime of laptop					215 kWh

3.7.2 Volatilization of flame retardants from plastics

During the use phase of electronic products, FRs can volatilize out of the polymers and thus be released to the environment. A number of studies report concentrations of FRs in house dust (*Harrad et al. 2010, Stapleton et al. 2008, Stapleton et al. 2009*). The sources of FRs in house dust will be a combination of e.g. electronic products, furniture upholstery, carpets and insulation material. Therefore, FR emissions from a laptop cannot be directly calculated from data on concentrations in house dust.

According to OECD emission scenarios on plastics additives, the estimated emission factor due to volatilization is 0.05% over the lifetime of the product (*OECD, 2004*). These emission factors are also used in some EU Risk Assessments on FRs (*European Commission, 2008b*). This results in total FR emissions in the µg range for the laptop case study.

Preliminary results from experiments in the ENFIRO project, using plastic plates at 20-100°C, indicate that the emission factor due to volatilization may be several orders of magnitude lower, which would result in a total emission in the µg range over the lifetime of a laptop. As the experiments are not completed yet, these results have not been used.

Earlier studies commissioned by UBA on volatilization of FRs with emission test chambers also showed generally low emissions, in the ng/m²/hour range, with the exception of TCPP³ which was emitted up to the high µg/m²/hour range (*Kemmlin 2003a and 2003b*). Those results correspond to a total FR emission in the µg range over the lifetime of a laptop. For very low-volatility additives, the tests suffer from practical problems (sink effects, sorption of additives to wall surfaces of emission test chambers).

For organic additive FRs, we consider a worst case emission factor of 0.05% due to volatilization, according to the OECD scenario. Additionally, the OECD scenario estimates an emission factor to water of 0.05% (for indoor service), which is also included in the current study. For inorganic FRs (for indoor service), emission factors are estimated to be lower: 0.01% to water and 0 to air (*OECD, 2004*).

The EU risk assessment on ATO also uses the emission factor of 0.01% to water (*European Commission, 2008a*).

For decaBDE, more specific data are available from the EU risk assessment (*European Commission, 2002*). In that report, a use phase emission of 0.038% per year is determined. This amounts to an emission factor of 0.0019 over the laptop lifetime, and this value is used in the current study.

For reactive FRs (DOPO and TBBPA in EPR), emission factors were assumed to be 0 in the use phase, as their covalent link to the polymer makes emission unlikely.

³ Tris (2-chloro-isopropyl)phosphate

A different approach was used in a Swedish study on emission of additives from plastics (*Westerdahl, 2010*). A screening model was developed, in which volatilization was related to the diffusion coefficient in plastics (Fick's law). Input data for the model were sales figures, average thickness/shape of products and molecular weight of the additive. This resulted in relatively high additive emissions of around 2% per year. As the model was not fully developed yet, and the discrepancy in predicted emissions with other literature is high, it was decided not to use these values in the current study.

Some literature suggests that physical weathering of small particles from flame retarded polymer materials may be a relevant source of FRs to the (indoor) environment (*Harrad, 2010*). Skin contact with electronic products has been suggested as a potential human exposure route to FRs. Due to insufficient quantitative data, these processes have not been included in the current study.

LCA considers environmental effects on a global scale, and therefore FR emissions to air from volatilization in the use phase are considered as contributions to the total stress on ecosystems and human health by toxic compounds. An LCA study does not consider local effects, such as exposure to FRs at the workplace (which may be locally be relatively high). To evaluate specific local effects, a risk assessment should be performed.

The emission factors of FRs in the use phase used in this study are summarized in Table 38.

3.7.3 Occurrence of fire

Within the field of flame retardants, the occurrence of fire is obviously an important subject. When considering the life cycle of a laptop however, it is clear that only a minor fraction of them will actually catch fire. Although incidents or unintended processes are often ignored in LCA studies, it was decided that for the current study the occurrence of fire would be included.

In a recent study, a fire occurrence in Europe of 2-2.5 million domestic fires per year was reported (*Nibra, 2009*). This would correspond to 3.9 fires per 1000 European inhabitants. Fire statistics are collected in different ways in different European countries, and not for every European country data are available. A range in fire occurrence per country is reported from 0.8 to 8.9 per 1000 inhabitants ⁴ (for The Netherlands and Estonia, respectively; *Nibra, 2009*).

Research on the causes of domestic fires show that between 3 and 31% were caused by electrical appliances (*Nibra, 2009*). From these data, it is not possible to establish whether the presence of a laptop would increase the chance of fire. Due to the relatively low power usage of a laptop (compared to a TV or washing machine), it can be expected that laptops are only a minor cause of fires.

We assume the presence of one laptop per household, and take the average European fire occurrence per household as a value in the current study for the fraction of laptops that will be burnt. This fraction is 0.0094 (9.4 per 1000 households). The fire statistics used can be found in Annex 3.

In earlier LCA studies on burning electronic products from Sweden, a lower incidence of 1.4 serious fires per 1000 dwellings was used (*Simonson, 2001*).

For the UK, detailed data on fire occurrence is available (*Department for Communities and Local Government, 2011*). The study reports 412,000 fires in 2004 (close to the number used in *Nibra 2009*). However, most of these (311,000) were outdoor fires, and the number of actual dwelling fires in Britain totalled 57,000. Which type of fires are included for the other countries in the *Nibra* study is unknown. It is possible that the domestic fire occurrence used in the current study is a considerable overestimation.

⁴ Average fire occurrence 3.9/ 1000 inhabitants, median 3.2/ 1000 inhabitants.

In all LCA scenarios, the investigated polymers have a V0-rating for fire safety. We can therefore assume that fire occurrence will be equal in all scenarios.

3.7.4 Emissions from fire

During a fire, complex and unpredictable chemical processes take place. The complex chemical composition of a laptop makes it even harder to model emissions during fire in a reliable way. In the last decade, a series of studies have been performed on emissions from burning electronic products by SP, the Swedish National Testing and Research Institute (*Simonson, 2000; Andersson, 2003, 2004, 2005*). In those studies, gases produced during the combustion of electronic products were sampled and chemically analyzed. The studies focused on the comparison of flame retarded (using BFRs) and non-flame retarded products and did not compare different types of FRs. Results of the studies were used in the current study as an indication of possible emissions from electronics during a fire. For the case of laptops, no comparable studies could be found. Therefore, the emission data for TV sets were assumed to be roughly representative for the laptop as well. Because of these assumptions and uncertainties, the data in this paragraph should be considered as rough estimates. In table 26, averaged emission data from the tests with TV sets are listed (*Simonson, 2000*). These data were used in both the BFR and HFFR scenarios. In the same study, brominated dioxins were analyzed as well. Emissions of 0.67 and 6.3 µg/kg combustible material were found for the non-FR and BFR-TV, respectively (*Simonson, 2000*). These emissions were used in the current study as well, for the HFFR and BFR scenario, respectively.

Table 26 Averaged emission data during the combustion of a TV set, taken from (Simonson, 2000).

<i>Chemical emission</i>	<i>g per kg combustible material</i>
PAHs	5.7
benzene	9.6
phenol	1.4
styrene	26
toluene	7.2
naphthalene	3.1
Non-methane volatile organic compounds (NMVOC)	11
PCBs	$25 \cdot 10^{-6}$
Chlorinated dioxins (2,3,7,8-TCDD equivalents)	$4 \cdot 10^{-6}$
Hydrogen chloride	13

A fraction of the FRs in the polymers may be emitted during combustion. The organic FRs will burn almost completely, while a larger fraction of the inorganic FRs may be emitted intact. In the SP study on TV sets, a fraction emitted to air for decaBDE is $3.4 \cdot 10^{-9}$, while a fraction for TBBPA of $2.7 \cdot 10^{-5}$ is given. As there are no values available for other organic FRs, we assume for all organic FRs an in-between value of $3 \cdot 10^{-7}$ for the fraction of FR emitted to air during a fire (including decaBDE and TBBPA). The SP study on TV sets reports a high emission factor of antimony to air, with a fraction of 0.835 of ATO present in the material being emitted to air (*Simonson, 2000*). This is in line with the mode of action of ATO, which

acts primarily in the vapor phase by forming highly volatile antimony halides or oxyhalides (Cusack, 2008). In contrast, the inorganic FRs ZHS and ZS are mainly active in the condensed phase, where they contribute to char formation (Cusack, 2008). Therefore, the emission factors to air of these inorganic FRs are assumed to be 10 times lower than for ATO.

As the different FRs have different flame retarding mechanisms, the emissions of other combustion products may also differ between the scenarios. The SP studies do not give information on this matter. ENFIRO tests performed on the burning behaviour of different FR-polymer combinations give some insights into these phenomena. For 3 polymers, the emission of CO, CO₂ and smoke were measured when different FRs were present. Ratios of emissions for the polymers with HFFR and BFR are shown in Table 27.

Table 27 Results from ENFIRO fire tests for 3 polymers, comparison of emissions of polymer with BFR and polymer with HFFR.

Polymer	Ratio of emission with HFFR/ with BFR		
	CO	CO ₂	Smoke
PC/ABS	0.76	1.15	0.84
PPE/HIPS	0.73	2.18	0.90
PA6,6	0.37	0.89	0.37

For CO and CO₂ emissions from burning electronics, the same data source was used as for Table 27 (Simonson, 2000). Smoke or particulate matter were not reported in the SP study. In a more recent Swedish study, emissions of particulate matter during combustion of plastics and PWB laminates of 0.05 g/g were reported (Blomqvist, 2005). As in the SP study BFRs were used, those data are used in the current study for the BFR scenario as well (see Table 28). For the emission of CO, CO₂ and smoke in the HFFR scenario, the values were multiplied by the ratios from Table 27, and with the fractions these polymers take in the total polymer weight in the laptop (the total polymer weight is considered as the total combustible material in the laptop). For the fraction of polymers not tested on emissions within ENFIRO, the emissions were assumed to remain unchanged (EVA and epoxy resin). The resulting emissions for the HFFR scenario are shown in the last column of Table 28: CO₂ has a higher emission in the HFFR scenario, while CO and particulate matter are lower.

Table 28 Emissions from burning electronics, for the scenario with BFRs and with HFFRs. Adjustments are based on the ratios in Table 27, and the amount of each polymer in the laptop.

Chemical emission	g per kg combustible material	
	– BFR scenario	– HFFR scenario
CO ₂	2400	3300
CO	86	68
Particulate matter, <10µm	50	44

Emissions in Tables 26 and 28 apply to 1 kg of combustible material. For the laptop, the total amount of plastics (1.73 kg) is considered as the total amount of combustible material.

The fate of the other materials (mainly aluminium, steel, copper, glass) during fire was not further investigated (and is less relevant for a comparison of flame retardants). It is assumed that these non-combustible materials remain in the ashes after the fire, and these are treated in a hazardous waste

incinerator. The corresponding Ecoinvent process sheet was used to model this hazardous waste treatment.

3.8 End of Life scenarios

The End of Life scenarios considered for the laptop are:

- collection and treatment according to the WEEE Directive
- disposal with municipal waste, followed by incineration in municipal solid waste incinerator (MSWI)
- disposal with municipal waste, followed by dump on landfill
- export to non-EU countries, followed by substandard waste treatment

The additional End of Life scenario of fire (assumed to be 0.94% of the total) has been described above in paragraphs 3.7.3 and 3.7.4.

The percentages of WEEE ending up in the different treatment systems is a subject of heated international debate. Numbers reported by different stakeholders differ. According to Greenpeace, as much as 60-75% of electronics waste is unaccounted for (*Cobbing, 2008*). According to the European Commission in 2008, the indications were that there is separate collection over 85% of WEEE arising, even though only 33% is officially reported as 'separately collected', but less than half of this - approximately 40% of all WEEE arising - is treated and reported according to the requirements of the WEEE-Directive; the remainder goes to substandard treatment in the EU or is illegally exported out of the EU (*European Commission, 2008c*). Eurostat has some on-line data available, but these are incomplete, extremely variable between countries and seem to be not uniformly collected ⁵ (*Eurostat, 2012*).

An important organization reporting European data on the collection of WEEE is the WEEE forum. They divide WEEE into 10 categories, and for this study category 3a "IT&T equipment (excl. monitors)" is relevant. According to their data, 417000 tons of IT&T (excl. monitors) were put on the market in 2009, while 131000 tons were collected as WEEE, which results in a collection rate of 31% (*WEEE, 2010*). The data also show large differences between countries / WEEE Forum members, with a range of collected amounts from 26 kg to less than 1 kg per inhabitant per year. The report also states that the amount of WEEE collected has been steadily increasing over the years.

Probably the most detailed investigation so far on WEEE flows was recently performed by Huisman et al. for The Netherlands (*Huisman, 2012*). This study investigated the official WEEE collection systems, but in addition quantified the 'unofficial' complementary WEEE collection streams. The size of these complementary collection streams turns out to be similar to the officially documented WEEE collection. Export streams of WEEE plus (still functioning) used equipment (EEE) were also quantified. In the study, WEEE was divided into 8 categories, of which 'IT equipment' is relevant for the current study. The WEEE amounts quantified for this category are shown in Table 29.

It is unsure how representative these data are for the whole EU. However, as this is the most in-depth study available, these data are used for the current study, with the some adjustments described below. In The Netherlands, hardly any municipal waste goes to landfill. On average in the EU, 19% of municipal waste is incinerated, and 38% goes to landfill, while the rest is recycled or composted (*European Commission,*

⁵ For IT and telecommunications equipment in 2008, these data would suggest a percentage of waste treated outside the EU of 0.2%, 3.6% and 16% for The Netherlands, Denmark and Bulgaria, respectively. Due to the incoherence of the data, these were not used in the current study.

2010). As the WEEE ending up in municipal waste cannot be recycled anymore (and neither composted), we assume for the WEEE in European municipal waste a split of 67% landfill and 33% incineration. The Huisman study does not specifically indicate which WEEE quantities are exported out of the EU and treated under substandard conditions. It is known that part of the used EEE that is exported is in fact not functioning or repairable (so should be classified as WEEE) or when functioning, will be treated improperly after disposal (*VROM Inspectie, 2007*). To obtain an estimation of improperly treated WEEE, we assume that half of the exported EEE flow will end up in improper treatment (and the other half ending up in municipal waste, with a split of 67% landfill and 33% incineration). We further assume that half of the WEEE indicated in the study as ‘not documented’ follows the improper treatment route (and the other half ending up in municipal waste, with a split of 67% landfill and 33% incineration). This results in a fraction of WEEE treated improperly of $(0.5 \cdot 0.18 + 0.5 \cdot 0.20) = 0.19$.

The following paragraphs describe how the different waste treatment methods were modeled.

Table 29 Amounts of WEEE ending up in different waste treatment methods for the subcategory IT (based on Huisman et al., 2012).

	<i>kg/ inhabitant / year</i>	<i>fraction of WEEE+ used EEE</i>	<i>further specification, based on own assumptions</i>		<i>fraction</i>
EEE put on market (POM)	3.03				
WEEE+used EEE generated	3.00				
export of used EEE	0.60	0.20	municipal waste	incineration landfill	0.033* 0.066*
			substandard treatment		0.10*
WEEE generated	2.40	0.80			
Collection and treatment according to WEEE Directive:					
via official WEEE route	0.61				
via complementary routes, households	0.29				
via complementary routes, businesses	0.41				
total proper WEEE treatment	1.31	0.44			
WEEE ending up in municipal waste	0.54	0.18	incineration landfill		0.06* 0.12*
not (yet) documented	0.55	0.18	municipal waste	incineration landfill	0.03* 0.06*
			substandard treatment		0.09*
Total fractions of waste streams			treated according to WEEE Directive		0.44*
			incineration		0.12*
			landfill		0.25*
			substandard treatment		0.19*

**: these values are not mentioned in Huisman, 2012, but based on own assumptions, see main text.*

3.8.1 Treatment according to WEEE Directive

Foreground data on WEEE treatment according to the WEEE-Directive were obtained from Stena Metall, a Swedish recycling company. One of their main activities is the dismantling, recycling and incineration of WEEE in full compliance with WEEE-Directive requirements. The company is also active in developing new treatment standards.

These data are seen as state-of-the-art WEEE treatment. The average European WEEE treatment may be somewhat less advanced (i.e. higher emissions), but no detailed data could be found for the average European situation.

The processes in this waste treatment scenario are described in the following paragraphs.

Pretreatment of electronics waste

The electronic waste is collected and stored in metal cages that can be placed under roof or not. Rainfall can cause leaching of additives from the plastic which is considered as emissions to industrial soil. Air emissions of additives from the plastics are considered as emissions to low populated air. All emissions are set to a default value of 1/10000 of the assumed concentration of the additive in the plastics where no specific data is available. For this process, only emissions of FRs are included in this study.

Average distance in Sweden for electronic waste was 112.5 km from the collection site to the pretreatment site. The model assumes that each vehicle is filled to 100% on the way to the pretreatment site (112.5 km) and is empty on the way back to the collection site (112.5 km). This modeled as a 14t truck 50% filled for 112.5 km. This distance is a worst case assumption, proposed by the Swedish WEEE compliance scheme El-Kretsen (*Seeger, 2009*). This transport was modeled with the Ecoinvent process sheet “Transport, lorry 7.5-16t, EURO4/tkm”.

Some pretreatment plants have manual dismantling for some products before shredding and some have only shredding. Two options are therefore modeled, one with a manual step of dismantling, and one where the whole process is automated. According to the Ecoinvent process “Disposal, laptop computer, to WEEE treatment”, the manual route accounts for 23% of laptops, while 77% follows the automatic route. These values apply to the Swiss situation in 2005, but are assumed in this study to apply to the European situation as well, for lack of better data.

Manual dismantling and shredding

When the WEEE arrives at the pretreatment site it is transferred to a conveyor belt and a first visual and manual sorting is performed to avoid problematic WEEE in the process. Air emissions of additives from the plastics should actually be considered as emissions to indoor air, but as CFs are lacking for this emission route, a general emission to air is used. All emissions are set to a default value of 1/1000 of the concentration of the FR in the plastics, as no specific data is available.

An electricity consumption (high voltage) for visual sorting and dismantling of 43 kWh/ton waste is reported, which includes running the conveyor belt, lighting, heating and ventilation (*Sjölin, 2012*).

Although these data originate from a Swedish plant, we use the European electricity production mix in the model, to improve the representativeness for Europe.

The main aim for manual dismantling is to remove mercury containing light sources and other problematic materials or unknown items for which the continued processes are not designed. The light sources etc. go to further treatment which is not included here. It is assumed that 1 wt % of the material is sorted out and goes to landfill, modeled with the Ecoinvent process “Disposal, hazardous waste, 0% water, to underground deposit”.

The shredding of dismantled products is performed in a closed system and the outlet air is treated before release into the outside air. The air emissions from the shredding are therefore seen as negligible.

The data represents a shredder that operates a certain presorted electronic waste fraction from a specific market. Other waste shredders, and electronic waste fractions from other markets, may have different properties. The electricity consumption of the shredder is 75 kWh/ton waste (*Sjölin, 2012*).

Automated shredding

Automated shredding of non-dismantled products is performed in a closed system and the outlet air is treated before release into the outside air. The air emissions from shredding are therefore seen as negligible.

For the shredder, data from a large scale pilot plant for shredding of possible mercury containing electronic goods are used. This plant does shredding and sieving in an automated process with an electricity consumption of 380 kWh/ton waste (Sjölin, 2012).

Sorting of fragmented materials and further treatment

Sorting of fragmented scrap by flotation will render sorted material fractions depending on material properties. The bromine containing plastic fraction is sorted out from the non-bromine fraction by flotation. Water can be used as flotation media to sort out plastics, and the electricity consumption for the whole process, including ventilation and feeding is around 10 kWh/ton waste (Sjölin, 2012).

It is assumed that no emissions occur in this process. The non-brominated plastic fraction will go to material recycling which is generally performed in Asia. The brominated plastic fraction is incinerated as hazardous waste.

The incineration of the brominated plastic fraction was modeled with the Ecoinvent process “Disposal, hazardous waste, 25% water, to hazardous waste incineration”. In this process, the brominated ash will go to hazardous waste landfill. A different treatment option for the brominated ash is distillation to hydrogen bromide (HBr), but this was not modeled.

Recycling

We assume that material recycling occurs of both precious metals and the non-brominated plastic fraction. However, the system boundaries of this study are set such that we apply a cut-off at End-of-Life recycling, which means that the potential costs and benefits of recycling are allocated to the production of the recycled metal or plastic, and not to the End-of-Life phase of the laptop. The fact that emissions from incineration of (bromine containing) plastics are considered (see above), while recycling (of non-brominated plastics) will have a score of 0, will still lead to a better environmental performance of the (to be) recycled material.

The inventory for the End-of-Life treatment of a laptop in compliance with the WEEE Directive is summarized in Table 30. The emission factor of the FRs (0.001) is also listed in Table 38.

Table 30 Main inventory data for the End-of-Life treatment of a laptop in compliance with the WEEE Directive.

<i>amount</i>	<i>product</i>	<i>process step</i>
0.0312 kWh	Electricity, high voltage	Manual dismantling (23%)
0.0543 kWh	Electricity, high voltage	Shredding after manual dismantling (23%)
0.922 kWh	Electricity, high voltage	Automated shredding non-dismantled WEEE (77%)
0.0315 kWh	Electricity, high voltage	Sorting of fragmented scrap by flotation (100%)

3.8.2 Municipal solid waste incineration

As shown before in Table 30, 12% of the laptops are assumed to end up at municipal solid waste incineration (MSWI). No dismantling or separation steps take place in this process.

The laptop is collected with municipal waste in a municipal waste lorry, and transported over 40 km. This is modeled using the Ecoinvent process sheet “Transport, municipal waste collection, lorry 21t”.

For modeling the non-plastics (1.417 kg), adjustments were made to the Ecoinvent process sheet ‘Disposal, residues, mechanical treatment, laptop computer, in MSWI’. In the original process sheet, emissions of antimony and bromine were listed, but as the non-plastic materials are not expected to contain these elements, these emissions were deleted from the process sheet.

For the plastics in the laptop (1.733 kg), the starting point was the Ecoinvent process sheet ‘Disposal, plastic, consumer electronics, to municipal incineration’. The EU risk assessment on ATO reports the following emission factors from MSWI incineration: 0.001 to air and 0.003 to water. These emission factors are used in the current study for all inorganic FRs. For organic FRs, it is unlikely that emission from MSWI will take place, as they will be completely incinerated. However, as a worst case approach, the emission factor of organic FRs to air is assumed here to be 10 times lower than the estimated emission factor during accidental fire, which amounts to $3.0E-8$.

No recycling of metals from the incineration ashes is accounted for in this process. This cut-off at recycling means that the potential benefit and costs of recycling are allocated to the production of the recycled metals, and not to End-of-Life metal.

Recovery of metals from ashes is common practice in some European countries, but not in all of them. For the comparison of scenarios, the potential recycling of steel and (metallic) aluminium does not make any difference, and can therefore be left out of the model.

3.8.3 Landfill

For the European situation, 25% of the laptops are assumed to end up on a landfill (see Table 30). No specific waste scenarios for electronics on a landfill are available in Ecoinvent, and therefore a combination of two more general processes were used for the laptop: for plastics (1.73 kg), the process sheet “Disposal, plastics, mixture, to sanitary landfill” was used, and for the other materials (1.42 kg), the process “Disposal, inert material, to sanitary landfill” was selected. It is possible that due to these choices some relevant emissions (e.g. heavy metals) are ignored. However, these possible omissions will be the same for all scenarios, and therefore not influence their comparison.

A number of publications exist which report the presence of flame retardants in landfill leachates (review of Weber *et al.*, 2011). In addition, research within the ENFIRO project has shown that most of the investigated flame retardants have a tendency to leach from polymer materials. The fraction of flame retardants in the material which eventually reach the environment due to leaching from landfills is however hard to estimate. Proper landfill management should ensure that most of the leached chemicals do not enter the environment.

The Technical Guidance Document and the OECD emission scenarios do not provide standard approaches to quantifying leaching from landfill, apart from stating that “The maximum potential loss could be calculated from the amount of additive remaining in the plastic at disposal, but it is very unlikely that this amount would be released.”

EU risk assessments on FRs do not address emissions from a landfill in detail. The EU risk assessment on a different type of plastic additive, DEHP (a phthalate plasticizer), provides some data on leaching of this substance additive from PVC. An average DEHP leaching percentage of 0.81%/year was found for cables buried in soil. This could be considered as a badly managed landfill with insufficient treatment of leachates.

From the same risk assessment on DEHP, an estimation of emission factors from landfills can be made based on the reported total consumption, total emissions and the fraction of waste going to landfill. The emission factor thus calculated is 4.1E-5.

Within the ENFIRO project, laboratory experiments were performed in which leaching of FRs was studied using flame retarded plastic plates during 20 days. Results showed a wide range in leaching rates for different FRs as well as for different polymers: leaching percentages between $<2.5E-7$ and 2.0% were found (for decaBDE from PC/ABS and MPP from PA6,6, respectively). These 20-day leaching values were extrapolated to 25 years, which is the assumed residence time on the landfill. The resulting emission factors to surface water are listed in Table 31.

Leaching of reactive FRs (TBBPA and DOPO in the current case study), was not experimentally measured. It was therefore assumed that 1% of FR would be present as unreacted (additive) FR, which would then be prone to leaching at the rate of the most similar FR that had been experimentally measured. The values in Table 31 were used in the landfill waste scenarios.

Table 31 Leaching percentages of FRs from polymers in a laptop during 25 years on a landfill. Data are extrapolated from 20-day leaching experiments.

<i>polymer</i>	<i>BFR</i>	<i>Percentage of total leached (%) in 25 years</i>	<i>HFFR</i>	<i>Percentage of total leached (%) in 25 years</i>
PA6,6+GF	BPS	0	Alpi	8.7
			MPP	100
EVA	ATO	96	ZHS	96
	decaBDE	3.8E-4	ATH	96
PPE/HIPS	ATO	96	RDP	0.17
	decaBDE	6.5E-4		
PC/ABS	ATO	96	BDP	5.7E-3
	decaBDE	1.1E-4		
epoxy resin	reactive TBBPA	3.8E-6	DOPO	8.8E-4

3.8.4 Export of e-waste to non-EU countries

Although no exact numbers are available, it is clear that a significant fraction of WEEE in Europe is transported out of Europe, to a number of possible countries in Asia and Africa. After arrival, rudimentary recycling techniques are applied to recover part of the metals. These practices often take place with little regard for worker safety or environmental consequences (Kuper, 2008; Nnorom, 2008; Nordbrand, 2009; Sepúlveda, 2010). The fraction of WEEE ending up in this End-of-Life scenario was derived indirectly from the study by Huisman *et al.* (2012), as discussed in paragraph 3.8. In this study, we assume that 19% of the laptops end up in this waste scenario.

Often in LCA studies, only 'official' waste treatment scenarios are included. Aspects such as littering or illegal export are often left out. However, as the export of WEEE is one the most hotly debated issues in the life cycle of electronics, and as it is clear this waste stream is of considerable size, we do include it in this study.

LCA databases do not provide data on energy use or emissions during substandard waste treatment. Other types of studies do give information on the general processes taking place during these treatment steps. Data on some specific emissions are available as well.

Therefore, we take as starting point the municipal solid waste incineration as described in paragraph 3.8.2, and add the specific emission data that was found in literature about illegal WEEE treatment practices. Some of these emissions are independent of the type of FR (and will be equal in both scenarios), and some emissions are specifically linked to FRs.

The uncertainties in the resulting inventory for this waste scenario are relatively high.

The first phase of this waste scenario is considered to be transport of WEEE to China over 19.000 km per freight ship (China was chosen as example; many other countries receive European WEEE as well). This is modelled with the corresponding Ecoinvent process sheet ‘Transport, transoceanic freight ship’.

Emissions from the substandard recycling of a laptop can be expected in the following processes (*Sepúlveda, 2010; Wang 2012*):

- 1) Open burning of cables to retrieve copper wires (lead and cadmium, chlorinated dioxins from e.g. PVC)
- 2) Re-melt plastics for re-application; open burning of circuit boards to retrieve metals (brominated dioxins from BFRs)
- 3) Desoldering of printed wiring boards by heating them on a small stove (lead/tin vapors generated)
- 4) Acid leaching of printed wiring boards to retrieve precious metals (acid emissions to water, acid fumes, cyanide emissions)
- 5) Manual dismantling of flat panel display with mercury-containing lamps (mercury)

During the recovery of precious metals, around 0.5 g sodium cyanide (NaCN) is needed per g metal (*Keller, 2006*). One of these metals to be recovered is gold, which has a high standard reduction potential and is therefore the last metal to be dissolved. This implies that first all other metals will have to be dissolved by NaCN. From the WEEE composition listed in the Ecoinvent process sheets ‘Disposal, plastic, consumer electronics, to MSWT’ and ‘Disposal, residues, mechanical treatment, in MSWT’ the total amount of metals was calculated, and this was multiplied by 0.5 to arrive at the consumed amount of NaCN: 62 g NaCN per laptop. It was assumed that 10% of the NaCN is emitted to air, and 90% to water. The emissions are expressed as cyanide emissions in the model.

In similar recovery processes, strong acids such as nitric acid are used as well (*Keller, 2006*). The assumption was made that the consumption of acid was equal to the consumption of NaCN. Again, a 10% emission to air and 90% emission to water was assumed. In the ReCiPe model, the only strong acid that has toxicity characterization factors is hydrogen fluoride (HF), and therefore the emission of HF was used as a proxy for the emission of nitric acid.

As emissions from combustion of the organic components of the laptop, the same data was used as for the combustion during accidental fire (see paragraph 3.7.4). The data from Table 27 were copied for the current process, with the exception of dioxins, for which more specific data were available (see below). The fraction of organic FRs that may be emitted during substandard WEEE treatment was assumed to be equal to those in the accidental fire scenario: $3E-7$ for organic FRs. For inorganic FRs, more specific data were available (see below).

The only study in which actual emission factors during simulated WEEE recycling experiments were calculated was performed by Gullett et al. (2007). In that study, air and residual ash was analyzed during experiments of open, uncontrolled combustion of PWBs and insulated wires, under conditions

comparable to rudimentary WEEE recycling operations. The average reported emissions of particulate matter and some metals are used in this study, as listed in Table 33.

The emissions of antimony, zinc and tin (reported as elemental composition of the fly ash) reported by Gullett can be used as an estimation of the emission of inorganic FRs. Assuming a WEEE composition in Gullett's experiments similar to the laptop in the current study, the emission factors to air are calculated as 0.0052 for ATO and 0.037 for ZHS and ATH. The resulting FR emissions for the laptop are shown in Table 35 and 36.

For relevant metals not listed in the Gullett study, it was assumed that 1% is emitted to air. Emission to water is assumed to be 20%. For the concentrations of heavy metals in WEEE, the compositions listed in the Ecoinvent process sheets "Disposal, plastic, consumer electronics, to MSWI" and "Disposal, residues, mechanical treatment, in MSWI" were used. The derived emissions of metals are shown in Table 33 and 34.

The formation of dioxins is one of the most discussed hazard issues concerning substandard WEEE treatment. Brominated dioxins can be formed out of BFRs, but (chlorinated) dioxins can also be formed out of other WEEE components, such as PVC. The study by Gullett *et al.* (2007) reports (chlorinated) PCDD/PCDF⁶ emissions of 92 ng TEQ per kg PWB, and 11900 ng TEQ per kg insulated wire. The log-average of these two values is used for the current study, which amounts to an emission of 1000 ng TEQ per kg WEEE. This emission is considered to originate from sources other than FRs, and is therefore incorporated in both the BFR and the HFFR scenario.

The Gullett study further reports an emission of brominated PBDD/PBDF⁷ from PWBs 100x higher than the chlorinated PCDD/PCDF. This emission of 9200 ng TEQ per kg PWB would originate from BFRs.

In other literature, lab studies are reported which give an indication how much dioxin can be formed out of BFRs. A review showed a wide range of 0,04% to 33% dioxin yield (Weber, 2003). Based on the averaged data in that review, for each BFR in the current study a specific brominated dioxin formation yield during incineration was determined from literature: 1.4% for decaBDE, and lower for the other BFRs (see Table 32).

For the 5 polymers studied, the amounts present in the laptop are multiplied by the specific BFR loading rates.

To express dioxin emissions in 2,3,7,8-TCDD-equivalents (the most toxic dioxin, which is the unit used in the Simapro software), a toxic equivalency factor (TEF) could be derived from the Weber review (2003), in which emissions are expressed in both grams and TEQ. The derived average TEF is 0.039.

Total emission of TCDD-eq. would then amount to 91 mg for a laptop in this waste scenario.

⁶ PCDD/PCDF: polychlorinated dibenzodioxins / polychlorinated dibenzofurans

⁷ PBDD/PBDF: polybrominated dibenzodioxins / polybrominated dibenzofurans

Table 32 Amount of dioxins formed during improper waste treatment of 1 laptop, based experimental results, reported by Weber (2003).

<i>polymer</i>	<i>BFR</i>	<i>loading %</i>	<i>kg polymer / laptop</i>	<i>kg decaBDE</i>	<i>kg TBBPA</i>	<i>kg other BFR</i>
EPR	TBBPA	20	0.293		0.0586	
EVA	decaBDE	13	0.159	0.0207		
PPE/HIPS	decaBDE	14	0.516	0.0722		
PA66	Br-PS	18	0.114			0.0205
PC/ABS	decaBDE	10	0.651	0.0651		
total (kg):				0.158	0.059	0.021
fraction dioxin formation out of specific BFR:				0.014	0.00038	0.0033
dioxin formation out of specific BFR (kg):				0.0022	0.000022	0.000068
total dioxins emitted (kg):						0.0023
TEF-value for dioxins formed:						0.039
total dioxins emitted from laptop (mg TEQ):						90.5
total dioxins emitted per kg WEEE (mg TEQ):						28.7

The two values for dioxin emissions from improper WEEE treatment derived from the different studies differ considerably: 9.2 µg per kg WEEE from the Gullett study and 28.7 mg per kg WEEE for the Weber study. For the current study we take the log-average of these two values, which amounts to 0.51 mg TEQ per kg WEEE. The resulting emission of PBDD/PBDF is listed in Table 35. The strongly differing results from literature indicate the large uncertainty of this value.

Table 33 Inventory of emissions to air during substandard waste treatment of 1 laptop.

<i>substance</i>	<i>unit</i>	<i>Based on literature source</i>
particulate matter	52.1 g	Gullett, 2007
PCDD/PCDF	3.3 µg TEQ	Gullett, 2007
lead	3.38E-3 kg	Gullett, 2007
arsenic	2.51E-2 kg	Gullett, 2007
copper	1.28E-3 kg	Gullett, 2007
cadmium	3.92E-6 kg	Ecoinvent
mercury	3.52E-8 kg	Ecoinvent
chromium	1.62E-6 kg	Ecoinvent
strong acid (nitric acid)	6.18 g	Keller, 2006
cyanide	3.28 g	Keller, 2006

Table 34 *Inventory of emissions to water during substandard waste treatment of 1 laptop.*

<i>substance</i>		<i>unit</i>	<i>Based on literature source</i>
lead	1.98E-4	kg	Ecoinvent
arsenic	5.39E-6	kg	Ecoinvent
copper	6.85E-3	kg	Ecoinvent
cadmium	7.84E-5	kg	Ecoinvent
mercury	7.03E-7	kg	Ecoinvent
chromium	3.23E-5	kg	Ecoinvent
strong acid (nitric acid)	55.7	g	Keller, 2006
cyanide	29.5	g	Keller, 2006

Table 35 *Inventory of emissions to air during substandard waste treatment of 1 laptop: FR related emissions in the BFR scenario.*

<i>substance</i>		<i>unit</i>	<i>Based on literature source</i>
ATO	406	mg	Gullett, 2007
decaBDE	4.74E-8	kg	Simonson, 2000
TBBPA	1.76E-8	kg	Simonson, 2000
BPS	6.16E-9	kg	Simonson, 2000
PBDD/PBDF	1.62	mg	Gullett 2007 / Weber 2003

Table 36 *Inventory of emissions to air during substandard waste treatment of 1 laptop: FR related emissions in the HFFR scenario.*

<i>substance</i>		<i>unit</i>	<i>Based on literature source</i>
ZHS	1.02	g	Gullett, 2007
ATH	3.08	g	Gullett, 2007
Alpi	5.71E-9	kg	Simonson, 2000
MPP	2.84E-9	kg	Simonson, 2000
RDP	2.32E-8	kg	Simonson, 2000
BDP	3.91E-8	g	Simonson, 2000
DOPO	2.20E-8	g	Simonson, 2000

3.9 Environmental fate and effects of flame retardants

3.9.1 Calculation of toxicity characterization factors using the USES-LCA model

Most of the FRs in the current study do not have characterization factors (CFs) for ecotoxicity and human toxicity in the ReCiPe model. For other substances, ReCiPe uses the model USES-LCA2 to determine these CFs, and the same model was used to determine toxicity CFs in this study. USES-LCA2 is a multimedia fate, exposure and effects model, based on the EU risk assessment tool EUSES/Simplebox 3 (Van Zelm, 2009).

As input the model requires physicochemical properties of a substance, as well as environmental properties such as partition coefficients (K_{ow} , K_{oc} , K_{aw}), bioconcentration factors, degradation rates, and toxicological properties. When values are missing, certain parameters are derived using simple QSARs in the model.

The model then calculates the distribution of the substance over different environmental compartments after it has been emitted to air, water or soil, and subsequently calculates the possible exposure of organisms to the substance. This is then multiplied by the effect factor which is based on the toxicological properties.

The outcome of the model is a Midpoint and Endpoint CF for the freshwater, marine and terrestrial ecotoxicity and human toxicity for a substance. The Midpoint CF is expressed as 1,4-dichlorobenzene equivalents per kg emitted (a relative unit, analogous to the unit for climate change, CO₂-equivalents / kg). Different CFs are calculated for emission to water, air and sediment (because the environmental fate can be different for these different emissions). The Midpoint CF is converted into an Endpoint CF, which expresses the potential damage the substance can do to Ecosystems (unit of CF: potentially disappeared fraction of species * year / kg) and Human Health (unit of CF: DALY / kg).

The toxicity CFs thus calculated for environmental emissions of FRs were added to the existing ReCiPe method.

3.9.2 Physicochemical and environmental properties of FRs

The main source for data on physicochemical properties of FRs was the literature review on this subject written within the ENFIRO project (*Waaaijers et al., 2012*). Additionally, some properties were derived from results of fugacity-based multimedia modeling within ENFIRO (*Kong et al., 2011*).

The values used in this study are shown in table 37. A number of values were not found in literature. These values are then assumed to be 0 in the model.

Table 37a Main physicochemical properties of FRs used in the USES-LCA2 model.

Compound	Molar Weight (g/mol)	Melting Point (°C)	Water Solubility (g/m ³ ;25°C)	Vapour Pressure (Pa;25°C)	log K _{OW}	log K _{OC} (L/kg)	Log K _p (soil) (L/kg)	Log K _{aw} (m/s)	Degradation rate constant in air (s ⁻¹)
RDP	574.4 ^a	90	1.1E-4	1.6E-6	7.4	4.6		-10.9	1.6E-5
BDP	692.6 ^a	90	1.9E-4	1.3E-4	6	4.5		-11.7	1.8E-5
DOPO	216.2	118	1.4E3	2.2E-3	1.9	1.7		-5.7	4.5E-6
AlPi	390.3	330	1.8E3	1E-6	-0.44				
ATH	78.0	230	0.015	1E-6	-	5.0			
APP	97 ^a	275	10000	1E-6	-2.2				
MPP	224.1 ^a	400	100	1E-6	-2.3			-10.9	
ZS	232.1	483	13	1E-6	-				
ZHS	286.1	190	1	1E-6	-0.5				
ATO	291.5	656	26	1E-6	-		2.4		
decaBDE	959.1	300	0.025	4.6E-6	6.3				
TBBPA	543.9	181	0.24	6.2E-6	5.9	5.7			2.2E-6
BPS	317.2 ^a	221.4	1E-20	2E-05	n.d.				

^a: this value refers to the monomer.

Table 37b Main physicochemical properties of FRs used in the USES-LCA2 model.

Compound	hydrolysis rate constant (s ⁻¹)	biodegradation rate constant in water (s ⁻¹)	aerobic biodegradation rate constant in sediment (s ⁻¹)	anaerobic biodegradation rate constant in sediment (s ⁻¹)	biodegradation rate constant in soil (s ⁻¹)	Bioconcentration factor (L/kg)	Aquatic ecotoxicity NOEC (mg/L)	Human toxicity Effect concentration (kg/kg/day)
RDP	2.1E-7		2.4E-8		1.1E-7	969	3.0 ⁽¹⁾	-
BDP	1.3E-7		1.5E-8		6.7E-8	1445	5.0 ⁽²⁾	-
DOPO	2.1E-7		2.4E-8		1.1E-7	5.4	2.0 ^(a, 2)	6.5E-3 (ED50) ⁽⁵⁾
APIi							10 ⁽²⁾	1.0E-3 (NOEL) ⁽⁵⁾
ATH							25 ^(a, 2)	5.0E-3 (ED50) ⁽²⁾
APP							88 ⁽²⁾	2.0E-3 (ED50) ⁽⁵⁾
MPP							72 ⁽²⁾	-
ZS							3.3 ^(b)	-
ZHS							3.3 ^(a, 2)	-
ATO							1.1 ⁽³⁾	2.0E-2 (NOEL) ⁽³⁾
decaBDE		4.4E-8	2.2E-8	5.0E-9	2.2E-8		-	1.0E-6 (NOEL) ⁽⁶⁾
TBBPA						485	0.16 ⁽⁴⁾	1.0E-4 (LOEL) ⁽⁷⁾
BPS							-	-

^a: this value is an extrapolation based on the lowest EC50 found in literature.

^b: value for ZS assumed to be equal to ZHS, for lack of better data.

References: 1: Brooke, 2009; 2: Waaijers, 2012; 3: European Commission, 2008a; 4: European Commission, 2007; 5: EPA, 2008; 6: USES-LCA database, 2009; 7: Leisewitz, 2001

3.9.3 Summary of the environmental emissions of FRs

In the previous paragraphs, the emission factors of flame retardants in different phases of the life cycle have been mentioned already. This paragraph gives an overview of these data.

Starting point for the derivation of emission factors are the EU Technical Guidance Document on the risk assessment of chemicals (*European Commission, 2003*), and the OECD document on emission scenarios for plastics additives (*OECD, 2004*). These emission factors are generally based on a worst case approach. Where additional information was available, based on data reported by industry, emission factors were calculated based on those data. This was the case for flame retardants which have been subjected to an EU risk assessment, and flame retardants which are part of the VECAP initiative. The emission factors used in this study are listed in table 38.

For the FR production phase, emission factors on BFRs were used from the industry initiative VECAP (2011). Data for ATO were taken from the EU risk assessment on this compound (2008). For all other FRs, the EU TGD on risk assessment (2003) was followed, which assumes a worst-case emission factor of 0.003 to wastewater.

Possible emissions during handling and transport of the FRs have been estimated using the OECD emission scenarios for plastics additives (2004).

For industrial processes, including the formulation of the flame retarded polymer, blending, moulding etc, the OECD emission scenarios were followed as well, except for decaBDE, TBBPA and ATO, for which the more specific emission data from their risk assessment reports were used.

The same sources were used for emission factors during the use phase. For reactive FRs, emission factors were assumed to be 0 in the use phase.

Emissions of FRs during accidental fire have been based on the LCA studies of Simonson *et al.* (2000).

The leaching rates of FRs from electronics on a landfill are based on laboratory experiments performed within the ENFIRO project. These experiments resulted in 20-day leaching values, which were extrapolated to 25 years, which is the assumed residence time on the landfill.

The emission factors from MSWI incineration for inorganic FRs are all assumed to be equal to the values reported in the EU risk assessment on ATO. For organic FRs, the emission factor to air is assumed to be 10 times lower than the estimated emission factor during accidental fire.

Table 38 Emission factors for flame retardants in different phases of the life cycle, as used in this study.

Life cycle phase, compound	Compartment	Emission factor	details	reference
Production of FR decaBDE	Air	1.1E-5		VECAP, 2011
	Water	1.8E-5		
	land	3.1E-5		
TBBPA	Air	1.0E-6		VECAP, 2011
	Water	2.0E-7		
	land	0		
ATO	Air	7.6E-5		EU risk assessment, 2008
	Surface water	2.8E-7		
	wastewater	4.7E-8		
Other FRs	wastewater	0.003		EU TGD on risk assessment, 2003

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<i>Industrial processes</i>			Including formulation of flame retarded polymer, extrusion, moulding etc	
decaBDE	Air	1.0E-4		EU risk assessment, 2002
	Water	5.0E-4		
TBBPA	Air	1.00E-5		EU risk assessment, 2007
	Water	1.01E-5		
ATO	Air	1.63E-6		EU risk assessment, 2008
	wastewater	3.5E-4		
	Solid waste	1.6E-2		
Inorganic FRs	Water	2.0E-4	Assumed to be closed processes	OECD, 2004
	Air	0		
Low volatility organic FRs	Water	1.2E-4	Assumed to be closed processes	OECD, 2004
	Air	2.0E-5		
medium volatility organic FRs	Water	2.0E-4	Assumed to be closed processes	OECD, 2004
	Air	1.0E-4		
<i>Use phase</i>				
decaBDE	Air	0.0019	0.038%/year; service life 5 years	EU risk assessment, 2002
ATO	wastewater	1.0E-4		EU risk assessment, 2008
Inorganic FRs	Water	1.0E-4	Indoor service	OECD, 2004
	Air	0		
Low volatility organic FRs	Water	5.0E-4	Indoor service	OECD, 2004
	Air	5.0E-4		
Reactive FRs			No emission	
<i>Accidental fire</i>				
Organic FRs	Air	3.0E-7		Simonson, 2000
ATO	Air	0.84		Simonson, 2000
Other inorganic FRs	Air	0.084		Cusack, 2008
<i>Waste phase, treatment according to WEEE Directive</i>				
All FRs	Air	0.001	Assumption	
<i>Waste phase, landfill</i>				
All FRs	Surface water	Between 3.8E-8 and 1	Based on ENFIRO leaching experiments, see paragraph 3.8.3	ENFIRO Work Package 4
<i>Waste phase, incineration</i>				
ATO	Air	0.001		EU risk assessment, 2008
	Surface water	0.003		
	Air	0.001	Assumed to be equal to ATO	
Inorganic FRs	Surface water	0.003		
	Air	3.0E-8	Assumed to be 10 times lower than the estimated emission factor during accidental fire	
<i>Waste phase, substandard treatment</i>				
Organic FRs	Air	3.0E-7	Equal to emission factor in accidental fire	Simonson, 2000
ATO	Air	0.005		Gullett, 2007
ZHS	Air	0.037		Gullett, 2007
ATH	Air	0.037		Gullett, 2007

4. Life Cycle Impact Assessment

In this chapter, the inventory of inputs and outputs described in chapter 3 is linked to environmental effects using the impact assessment method ReCiPe (described in detail in paragraph 2.11). This method has been augmented with toxicity characterization factors for the studied FRs, as these were not available yet in the ReCiPe method (see paragraph 3.9.1).

In the first paragraphs, results for separate phases in the life cycle are described, and then the results for the complete life cycles of the BFR and HFFR scenario are compared.

The main aim of the ENFIRO project is to find FRs which are more environmentally benign than the environmentally harmful BFRs. Main questions for the current LCA comparison are therefore: does the overall ecotoxicological and human toxicological profile over the whole life cycle improve by this substitution, and are there other environmental impacts that are negatively affected by this substitution (is there a shift of burdens)?

From this scope, it follows that the impact categories freshwater, marine and terrestrial ecotoxicity and human toxicity are very relevant. Therefore, these scores are shown in separate figures.

Main impact categories that are expected to be affected by the substitution are Climate Change, Particulate matter formation and Resource Depletion, as the HFFRs may have different energy consumption during production. In most cases, these three energy related impacts will be proportional to each other, and therefore it suffices to show only one of these categories (an exception may be particulate matter formation during mining, which would be independent of the others). Climate change was chosen as the representative for energy consumption, as this category has usually the highest overall impact.

In the ReCiPe Single Score results figure, all other impact categories are visible as well, and if other categories have significant contributions to the overall score, this will be visible in this graph.

The comparison of scenarios is done using both the separate toxicity and climate change results and the overall Single Score results. This comparison method is in line with the scope of the study.

The ISO 14044 standard states that weighting and aggregating of LCA results across different impact categories are an option to represent results in a simplified way, but is based on (subjective) value choices and should not be used when comparisons between products are made in a report to be disclosed to the public.

As a test, a comparison was made using the three ReCiPe Endpoint scores, without weighting and aggregating these scores (and therefore fully in line with the ISO 14044 standard). It was found that this graph mainly reflects climate change in all 3 damage categories: impact on Human Health, Ecosystems and Resource depletion. As this representation does not add relevant information to the separate climate change results already shown, it was decided not to use the representation with three Endpoint scores for the discussion of the further results.

4.1 Production of FRs

Figure 5 and 6 show a comparison of the production of 1 kg of each of the FRs investigated ('cradle-to-gate').

The separate toxicity impact categories in Figure 5 show that ATO scores highest for human toxicity. This is caused by heavy metal emissions during the mining phase. During mining of zinc and tin ore, heavy metal emissions also lead to a relatively high human toxicity score for ZHS and ZS.

Relatively high human toxicity, terrestrial and freshwater ecotoxicity scores are also observed for RDP, BDP and DOPO: this is caused by the emission of white phosphorus to air, during production of of

phosphoryl chloride (POCl_3) and phosphorous chloride (PCl_3) out of ‘phosphorus white, liquid’, which are intermediary products in the production route. The Ecoinvent datasheet which lists this white phosphorus emission (the datasheet for POCl_3 and PCl_3 production), mentions that the value is based on estimations, and not measurements. The emission should therefore be treated with caution. In paragraph 5.1.4, a sensitivity analysis will be done to check how relevant this emission is for the overall results. Although the FR Alpi also contains phosphorus, its production does not involve PCl_3 , and therefore the white phosphorus emission did not come up in that production route. No emissions of white phosphorus have been reported by Alpi producers.

For marine ecotoxicity, ZHS and ZS have the highest scores (Figure 5), which is caused by emissions of ZS and ZHS. These emissions also cause the relatively high score of ZS and ZHS for freshwater ecotoxicity.

In Figure 6, total LCA scores for the production of 1 kg FR are shown (the corresponding values can be found in Annex 4.1). Large differences are found between FRs, with a factor of 32 between the highest and lowest scores (ZS and ATH, respectively).

For most FRs, the largest contributions to the score are energy use related, with relatively high scores for climate change and fossil depletion.

The inorganic FRs ATO, ZS and ZHS have additionally high scores for particulate matter formation and metal depletion, which are both related to the mining phase.

It should however be kept in mind that the comparison of FR production per kg is of limited practical use: FRs cannot be simply interchanged, as most FRs are compatible with only a limited amount of polymers. In addition, the necessary loading rates to achieve the required fire safety performance (V_0) varies with FRs.

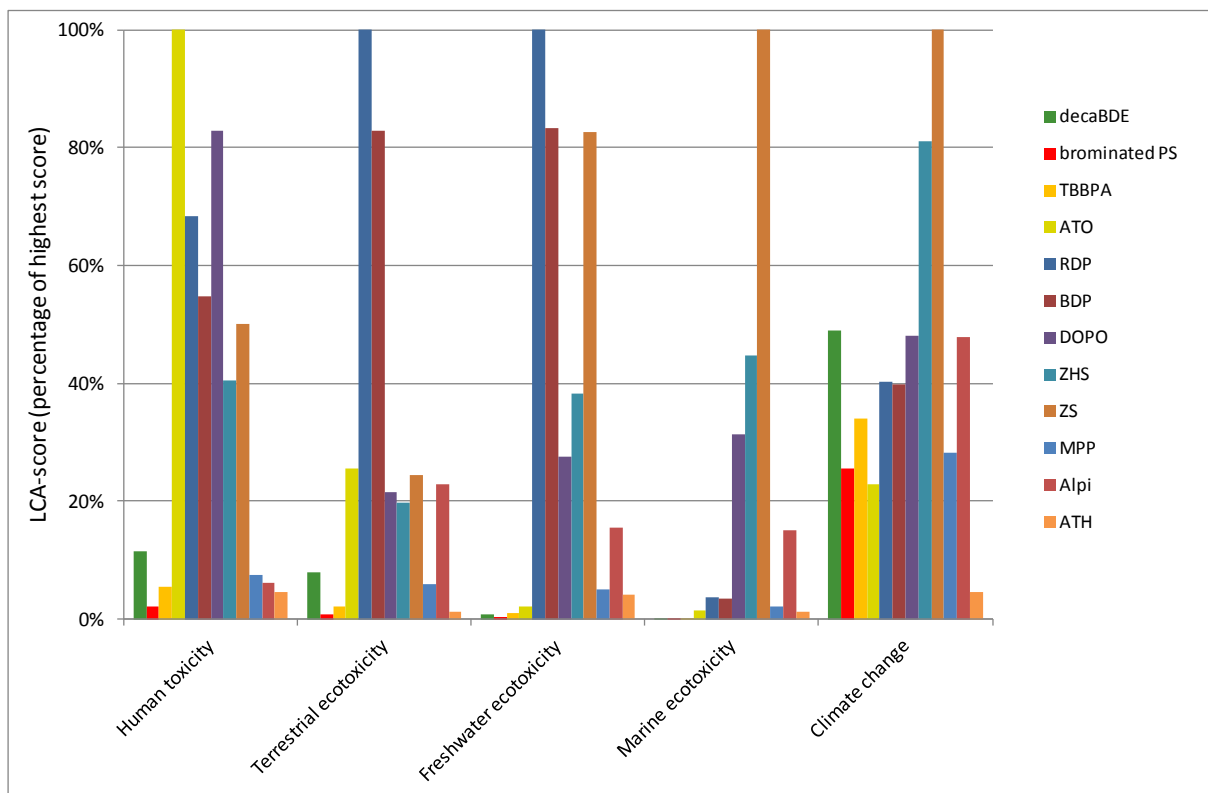


Figure 5: LCA scores for the production of 1 kg flame retardant, cradle-to-gate, for the four toxicity impact categories and climate change of the ReCiPe method. Each category is expressed as fraction of the FR with the maximum score.

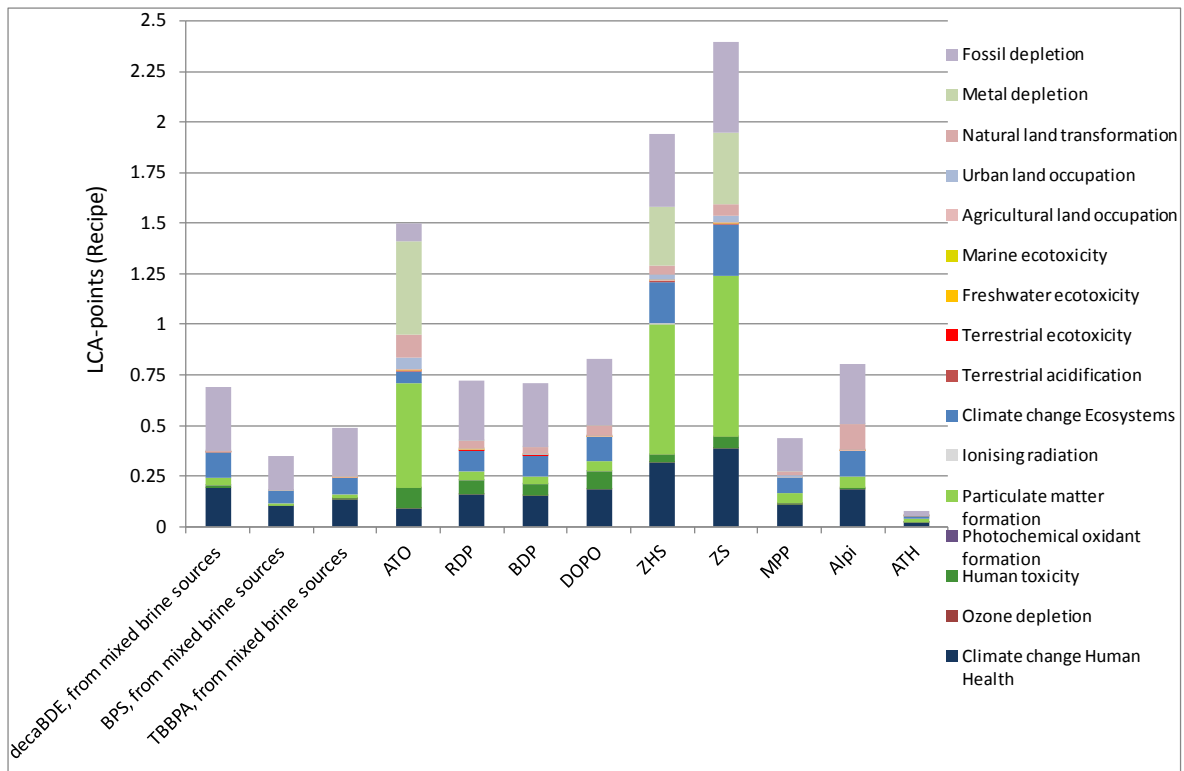


Figure 6: Environmental impact for the production of 1 kg flame retardant, cradle-to-gate. Scores are expressed in LCA-points according to the ReCiPe method. Colours indicate the contributions of the different environmental impact categories.

4.2 Flame retarded polymers

Of more practical significance than a direct comparison of FRs, is the comparison of compounded FR-polymer combinations, which fulfil the V0 fire safety requirements.

A comparison of flame retarded polymers is shown in figures 7 and 8 shown (the corresponding values are listed in Annex 4.2). FR loading rates are as described in paragraph 3.3. The life cycle phases considered are from raw materials via production of polymer and FR to blending of the polymer with FR (cradle-to-gate). For comparison, for each of the five polymers the scores of a non-flame retarded version are shown as well (polymer without additive).

Scores for human toxicity largely reflect the findings from paragraph 4.1: relatively high scores are observed for polymers containing RDP, BDP and DOPO, which is caused by the emission of white phosphorus to air during production of the intermediates $POCl_3$ and PCl_3 . Polymers containing ATO score relatively high on human toxicity as well, due to heavy metal emissions during the mining phase. The emissions of white phosphorus also dominate the scores for terrestrial and freshwater ecotoxicity, leading again to relatively high scores for polymers containing RDP, BDP and DOPO.

Emissions of ZHS lead to relatively high scores on freshwater and marine ecotoxicity for EVA+ZHS-ATH.

The relatively high score on marine ecotoxicity for EPR-DOPO is caused by the emission of DOPO. The impact category Climate Change shows only small variations within polymers (BFR, HFFR or no FR), indicating that the FR has only a minor influence on this impact category. For PA6,6 and PC/ABS,

the polymer without FR has a slightly higher score for Climate Change than the flame retarded version, which indicates that the production of the corresponding FRs has a lower ‘carbon footprint’ than the polymer itself.

The comparison of total LCA scores for the production of 1 kg flame retarded polymer is shown in Figure 8. Variation between the BFR and HFFR versions of a polymer are relatively small: a maximum difference of 16% is observed.

For PA6,6, no significant difference in total score between the BFR and HFFR version is found. For EVA and EPR, the HFFR version has a higher score (16 and 10%, respectively), and for PPE/HIPS and PC/ABS, the BFR version scores higher (9 and 5%, respectively).

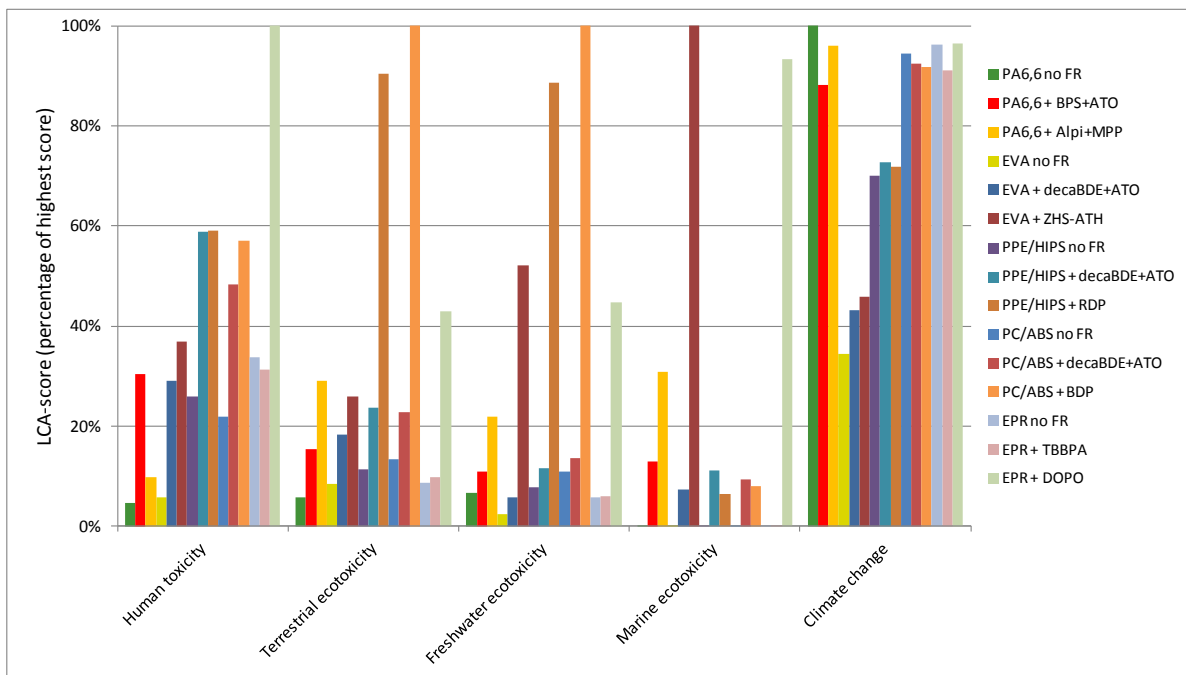


Figure 7: Environmental impact for the production of 1 kg flame retarded polymer, cradle-to-gate. Different loading rates required for different FRs are included in these calculations. The four toxicity impact categories and climate change of the ReCiPe method are shown. Each category is expressed as fraction of the FR with the maximum score (100%).

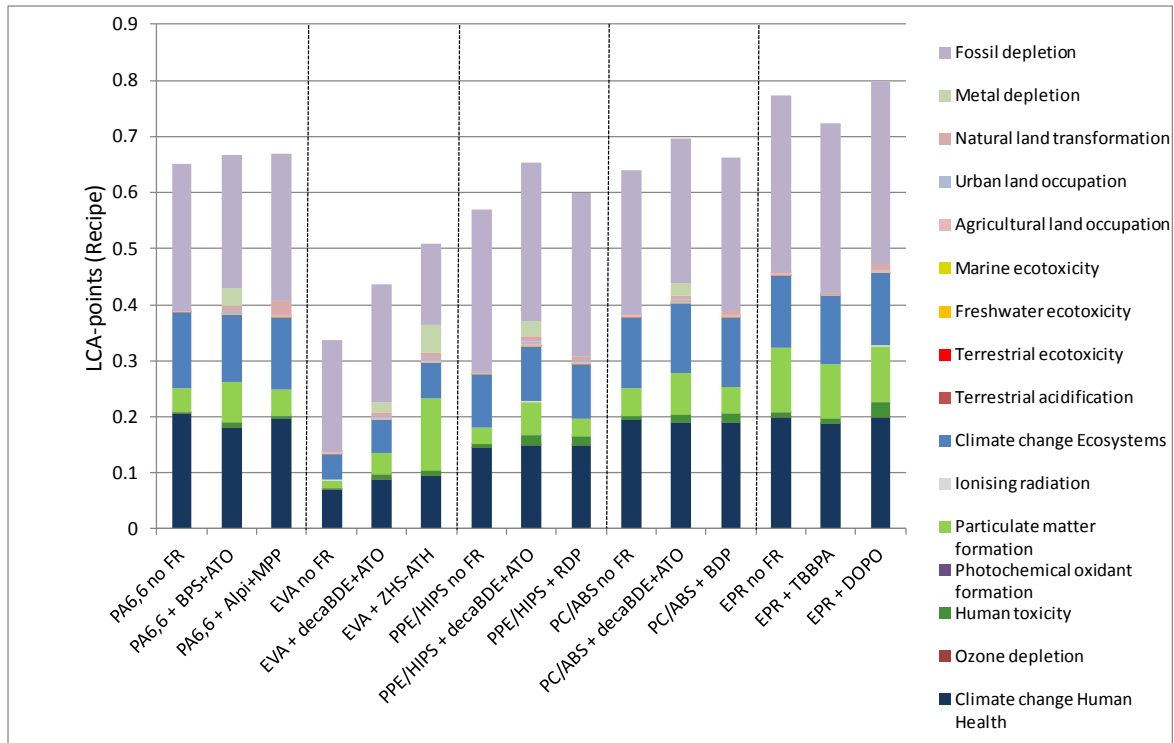


Figure 8: Environmental impact for the production of 1 kg flame retarded polymer, cradle-to-gate. Different loading rates required for different FRs are included in these calculations. Scores are expressed in LCA-points according to the ReCiPe method. Colours indicate the contributions of the different environmental impact categories.

4.3 Use phase

A relevant environmental impact of a laptop during the use phase is its electricity use. This impact is incorporated into the model, but as it is not influenced by the choice of FRs, it will be a constant score in all scenarios, and will not be specifically discussed here. The environmental impact of electricity is shown in the whole life cycle results as discussed in paragraphs 4.5 and 4.6.

4.3.1 Emissions during use

As described in paragraph 3.7.2, due to volatilization of FRs from the polymers, emissions to air and water can occur during the use phase.

A comparison of this life cycle phase for the BFR and HFFR scenarios is shown in figure 9 and 10 and Table 39.

Human toxicity and ecotoxicity are the only impact categories relevant in this process, as the only emissions concern the FRs, and these contribute only to the toxicity impact categories.

As can be seen in figure 9, human toxicity and freshwater ecotoxicity score highest in the BFR scenario, while the terrestrial and marine ecotoxicity scores are highest for the HFFR scenario.

Figure 10 shows that the total score is highest for the BFR scenario, which is mainly due to the human toxicity score of the emission of decaBDE (4.2 mPt). The terrestrial ecotoxicity score in the HFFR scenario (0.023 mPt) is mainly due to emissions of Alpi and MPP, while the scores for freshwater and marine ecotoxicity are caused mainly by RDP, BDP, Alpi and ZHS (HFFR scenario) and ATO (BFR scenario).

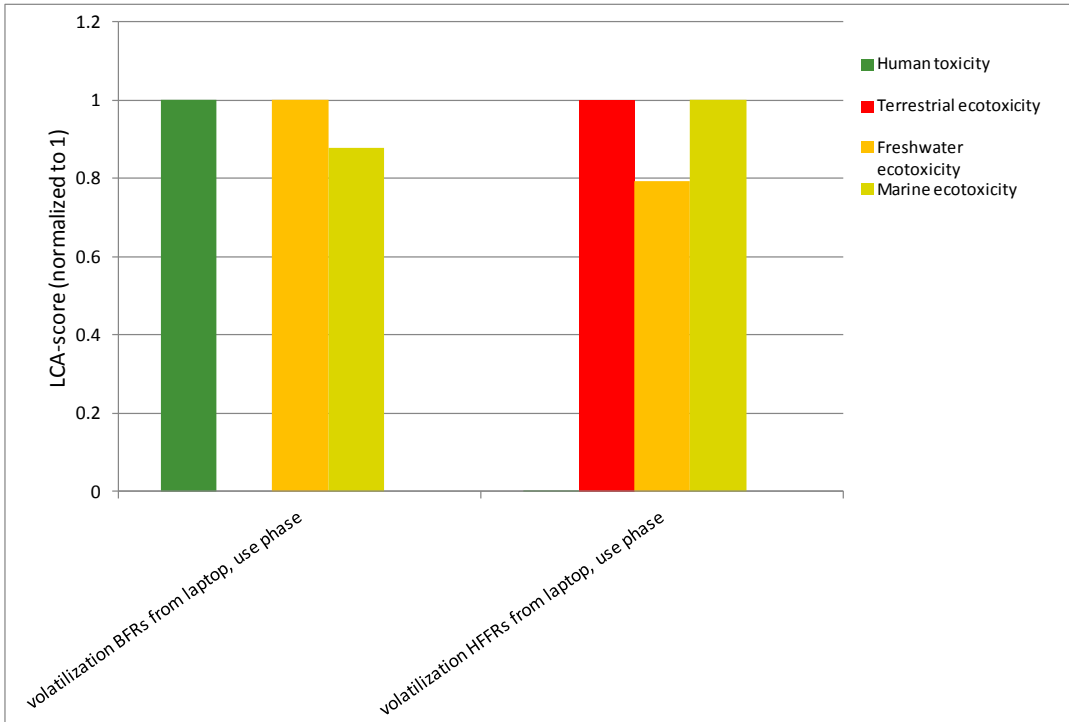


Figure 9: Environmental impact for chemical emissions during the use phase of a laptop with BFRs and a laptop with HFFRs, for the four toxicity impact categories and climate change of the ReCiPe method. Each category is expressed as fraction of the FR with the maximum score .

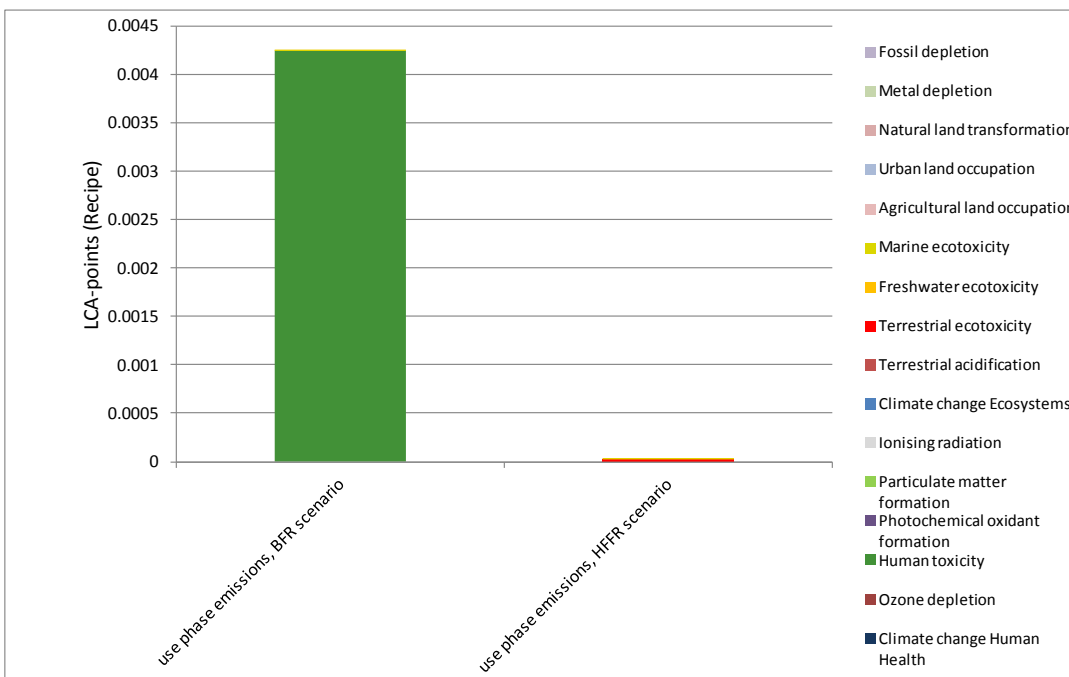


Figure 10: Environmental impact for chemical emissions during the use phase of a laptop with BFRs and a laptop with HFFRs. Scores are expressed in LCA-points according to the ReCiPe method. Colours indicate the contributions of the different environmental impact categories.

Table 39 *Environmental impact for chemical emissions during the use phase of a laptop with BFRs and a laptop with HFFRs.*

	BFR scenario (LCA points, ReCiPe method)	HFFR scenario (LCA points, ReCiPe method)
Total	4.2E-03	2.5E-05
Climate change Human Health	0	0
Ozone depletion	0	0
Human toxicity	4.2E-03	1.2E-08
Photochemical oxidant formation	0	0
Particulate matter formation	0	0
Ionising radiation	0	0
Climate change Ecosystems	0	0
Terrestrial acidification	0	0
Terrestrial ecotoxicity	3.4E-26	2.3E-05
Freshwater ecotoxicity	1.1E-06	8.5E-07
Marine ecotoxicity	9.3E-07	1.1E-06
Agricultural land occupation	0	0
Urban land occupation	0	0
Natural land transformation	0	0
Metal depletion	0	0
Fossil depletion	0	0

4.3.2 Accidental fire

Based on European statistics on fire occurrence, a small fraction of the laptops is estimated to end up in accidental fire (see paragraph 3.7.3). Some of the emissions during combustion of a laptop are expected to vary with the presence of different FRs, while other emissions are independent of the FRs.

As explained in paragraph 3.7.4, the emissions of CO and smoke are expected to be higher in the BFR scenario than in the HFFR scenario, while the CO₂ emission is expected to be lower in the BFR scenario.

Figure 11 shows total LCA scores for the combustion of a complete laptop, for the BFR and HFFR scenario (the corresponding values can be found in Annex 4.3).

A relatively high score for terrestrial ecotoxicity in the BFR scenario is observed. This is caused mainly by the emission of ATO, which due to its mode of action (active in the vapor phase), has a higher emission factor to air than other FRs (see paragraph 3.7.4).

The higher rate of smoke formation in the BFR scenario leads to a higher score for particulate matter formation.

CO₂ emissions are higher in the HFFR scenario than in the BFR scenario. This leads to higher scores for climate change.

The total score for accidental fire is 26% higher for the BFR scenario than for the HFFR scenario.

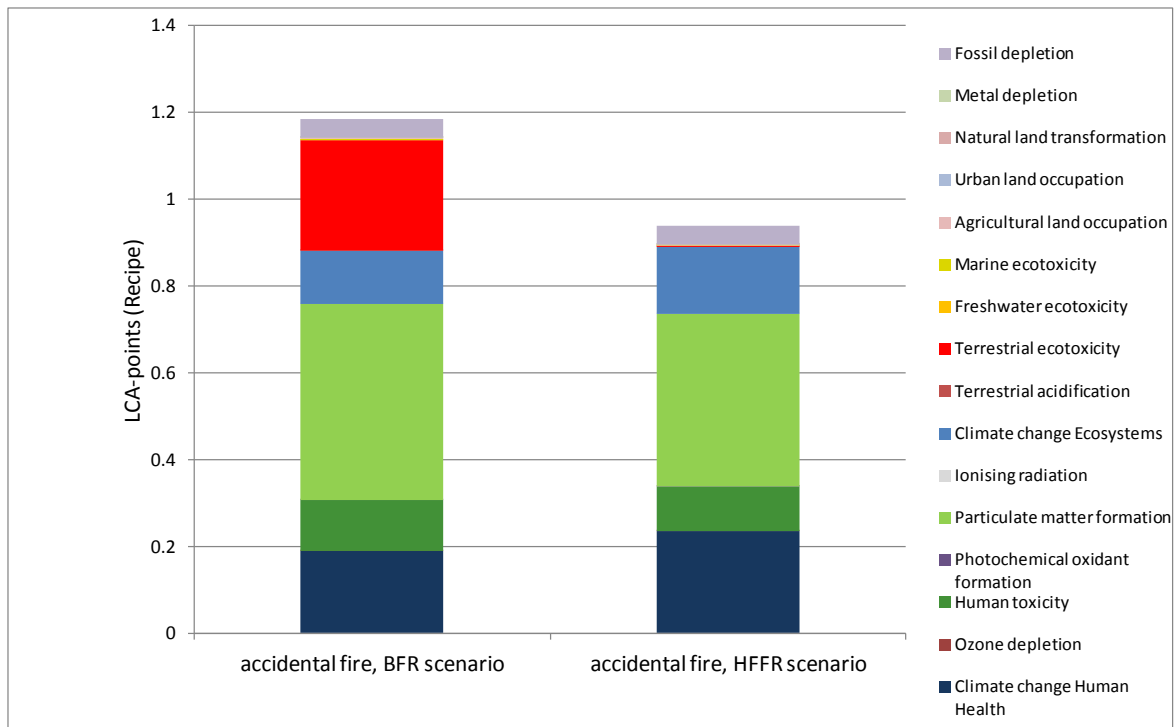


Figure 11: Environmental impact for accidental combustion of a laptop with BFRs and a laptop with HFFRs. The fraction of laptops actually ending up in an accidental fire is not incorporated in this figure. Scores are expressed in LCA-points according to the ReCiPe method. Colours indicate the contributions of the different environmental impact categories.

4.4 End of Life scenarios

Paragraph 3.8 described the four different End of Life options included in the model.

In figure 12 and 13, LCA scores of the waste treatment options for a complete laptop are shown (the corresponding values are shown in Annex 4.4). The fraction of laptops ending up in the different waste treatment scenarios is not incorporated in this comparison.

Figure 12 shows that for all toxicity impact categories, the BFR scenario scores higher than the HFFR scenario.

For human toxicity and terrestrial ecotoxicity, the improper treatment options score highest for both the BFR and HFFR version.

Human toxicity for improper treatment in the BFR version is mainly caused by the formation of dioxins during incomplete incineration of BFRs. Human toxicity for improper treatment in the HFFR version comes mainly from emissions of lead, arsenic, hydrogen fluoride, (chlorinated) dioxins and PAHs to air, and arsenic to water (all of which also occur in the BFR version).

A number of emissions contribute to the scores for terrestrial ecotoxicity: emissions of bromine, copper, cyanide, PAHs, and the FRs ATO, ZHS and ATH.

Scores for freshwater and marine ecotoxicity are mainly caused by ATO, bromine and several heavy metals (BFR scenario), and the FRs ZHS, ATH and Alpi (HFFR scenario).

In the impact category Climate Change, the landfill option has the lowest score for both the BFR and HFFR scenario, as no incineration takes place in this option, and therefore CO₂ emissions are low. The MSWI and improper treatment option have the highest Climate Change scores, as in both cases the laptop is incinerated, leading to CO₂ emissions. Climate Change scores for the WEEE-compliant treatment are

higher for the BFR version than for the HFFR version, because in the BFR version, a higher percentage of the polymers (all bromine containing polymers) is incinerated instead of recycled.

For the total scores of the waste treatment options shown in Figure 13, it is clear that the improper treatment option has the highest total score in both the BFR and HFFR scenario, mainly due to high contributions of the human toxicity impact category. The BFR scenario scores higher than the HFFR scenario for this waste treatment option, due to the formation of (brominated) dioxins out of BFRs. In both scenarios, the second highest score is for incineration in a normal MSWI, followed by the WEEE-compliant treatment procedure. Landfill has the lowest total impact. The impact category Climate Change has the largest contribution to these total scores.

This comparison of waste treatment options shows that even when WEEE does not contain BFRs, the export and improper treatment remains a serious threat to human health.

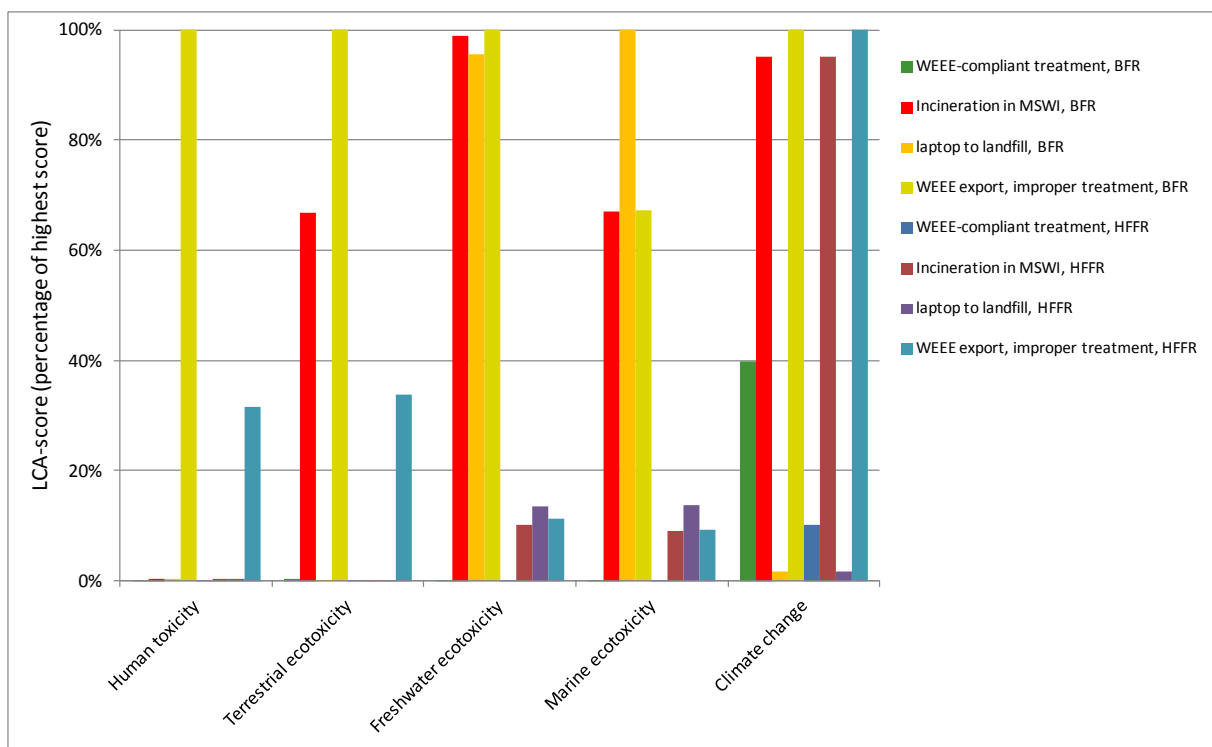


Figure 12: Environmental impact for the End-of-Life treatment of 1 laptop. A laptop with BFRs and a laptop with HFFRs are compared for the 4 End-of-Life options. Scores are shown for the four toxicity impact categories and climate change of the ReCiPe method. Each category is expressed as fraction of the FR with the maximum score. The fraction of laptops actually ending up in the different EoL scenarios is not incorporated in this figure.

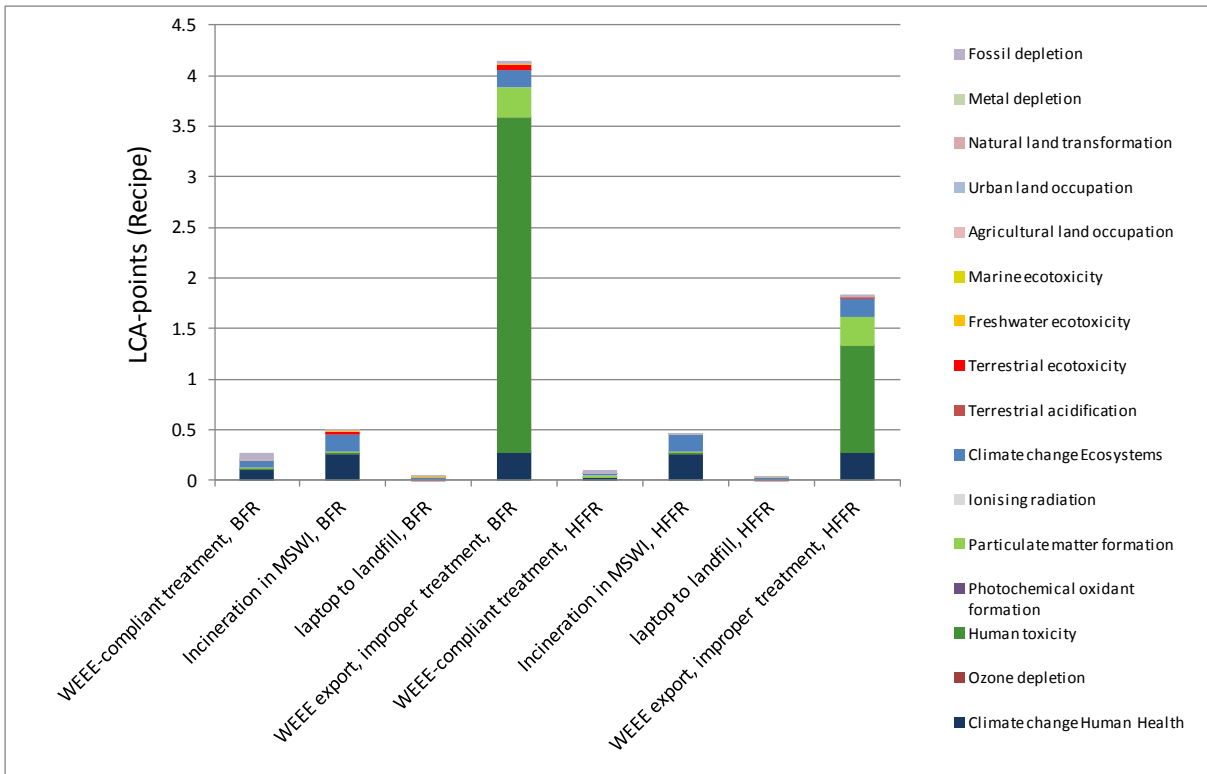


Figure 13: Environmental impact for the End-of-Life treatment of 1 laptop. A laptop with BFRs and a laptop with HFFRs are compared for the 4 End-of-Life options. The fraction of laptops actually ending up in the different EoL scenarios is not incorporated in this figure. Scores are expressed in LCA-points according to the ReCiPe method. Colours indicate the contributions of the different environmental impact categories.

4.5 Full life Cycle of a laptop with BFR

LCA results for the full life cycle scenario of BFRs in a laptop are shown in Figure 14 and 15 (the corresponding values are listed in Annex 4.5). The environmental impacts for each phase in the life cycle are shown.

The proper amounts of flame retarded polymer and the proper fractions of the laptop ending up in the different end of life options or in accidental fire have all been incorporated into the calculations, as discussed in chapter 3.

Significant scores are observed for the production of different laptop components, mainly the PWBs, LCD screen and batteries (as described in paragraph 3.6, the corresponding Ecoinvent process sheets were adjusted, as all polymer materials in those components were set to 0, and modelled as separate processes). Another high contribution to the overall environmental impact comes from electricity use of the laptop during the use phase. The scores in these processes are all independent of the choice of FRs.

Figure 15 zooms in on the processes that are influenced by the FRs. Of these processes, the highest impact occurs during improper waste treatment due to the emission of dioxins formed out of BFRs. The second highest score is for the production of flame retarded PC/ABS followed by flame retarded PPE/HIPS and EPR. The relative impact of the production of the five flame retarded polymers reflects the amounts of polymer present in the laptop (see paragraph 3.3).

The contribution to the overall environmental impact by volatilization of FRs in the use phase and emissions from accidental fire are low.

Of the four End-of-Life options, the highest impact occurs for export followed by improper treatment. The End-of-Life option with the second highest impact is WEEE-Directive compliant treatment. This is mainly due to the high percentage of WEEE ending up in this waste scenario (as shown in paragraph 4.4, the impact *per kg* is lower for the WEEE-Directive compliant treatment than for incineration in an MSWI). Landfill is the End-of-Life option with the lowest impact.

For all phases of the life cycle except improper waste treatment, the impact scores are dominated by the impact categories climate change, fossil depletion and particulate matter formation, which are all three mainly related to fossil energy consumption. In the improper waste treatment phase, human toxicity has the largest contribution to the score.

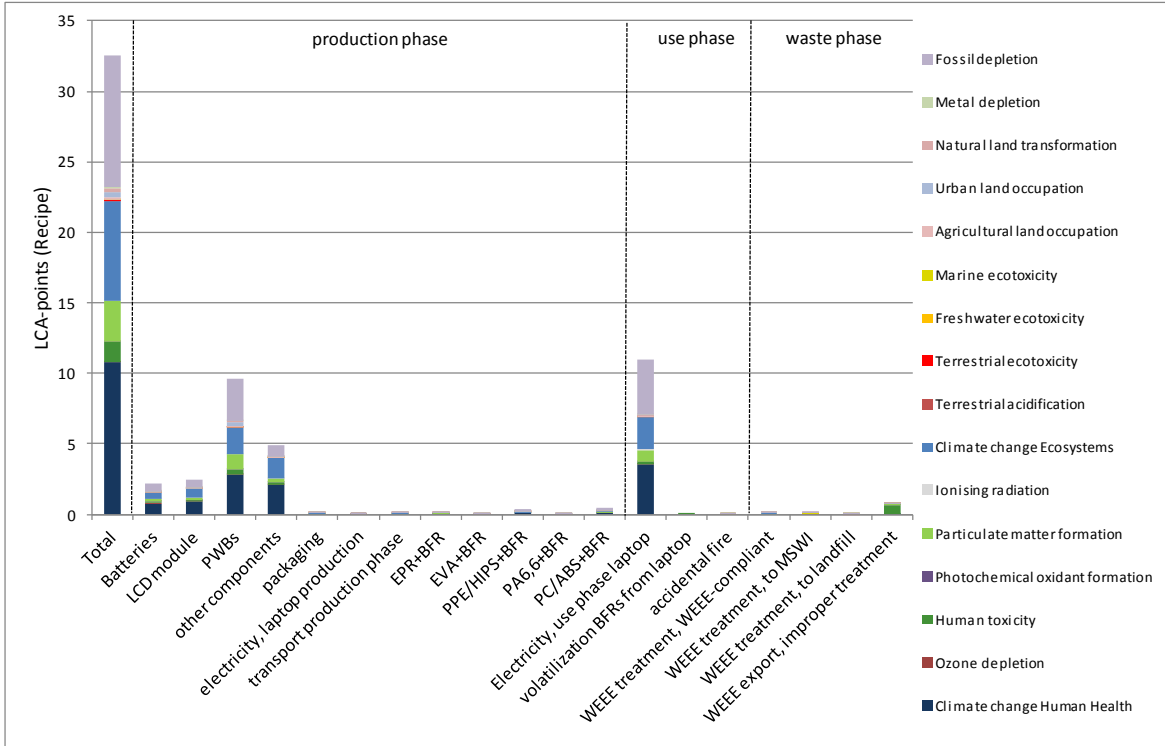


Figure 14: Environmental impact for each phase of the life cycle of the laptop scenario containing BFRs. Scores are expressed in LCA-points according to the ReCiPe method. Colours indicate the contributions of the different environmental impact categories.

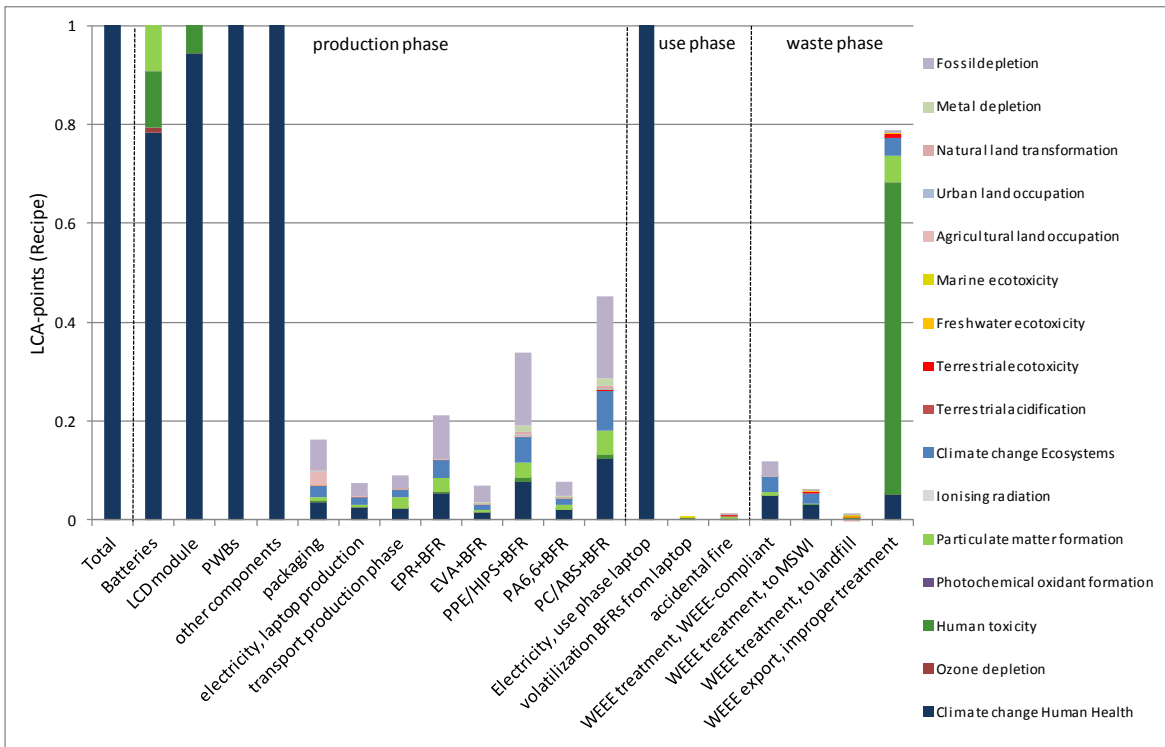


Figure 15: Environmental impact for each phase of the life cycle of the laptop scenario containing BFRs. Y-scale has been adjusted to show the processes/phases with lower scores. Scores are expressed in LCA-points according to the ReCiPe method. Colours indicate the contributions of the different environmental impact categories.

4.6 Full life cycle of a laptop with HFFR

In figure 16 and 17, the environmental impacts of the laptop life cycle scenario with HFFRs for each phase in the life cycle are shown (the corresponding values can be found in Annex 4.6). The proper amounts of flame retarded polymer and the proper fractions of the laptop ending up in the different end of life options or in accidental fire have all been incorporated into the model.

The processes which are independent of the choice of FRs have scores equal to those of the scenario with BFRs (paragraph 4.5): significant scores are observed for the production of different laptop components, such as PWBs, the LCD screen and batteries (in which the amounts of polymer materials were set to 0), and electricity use of the product during the use phase.

In Figure 17, the adjusted y-axis makes the processes that are influenced by the FRs more visible. Of these processes, the production of flame retarded PC/ABS has the highest impact. The improper waste treatment phase has the second highest score, followed by the production of flame retarded PPE/HIPS and EPR.

The FR emissions due to volatilization in the use phase and emissions from accidental fire have only a minor contribution to the overall environmental impact of the complete laptop life cycle.

Of the four End-of-Life options, the highest impact occurs for export followed by improper treatment, as was the case for the BFR scenario. For the HFFR scenario, the End-of-Life option with the second highest impact is incineration in an MSWI, which is different from the BFR scenario. Although a higher percentage of WEEE ends up in WEEE-Directive compliant treatment than in MSWI incineration, the impact *per kg* is considerably lower for the WEEE-Directive compliant treatment than for MSWI incineration, and as a consequence the total impact of WEEE-Directive compliant treatment is lower as well. Landfill is the End-of-Life option with the lowest impact (as was the case for the BFR scenario).

Dominating impacts in all phases of the life cycle are the fossil energy related categories climate change, fossil depletion and particulate matter formation. The fourth highest score is for human toxicity, which is the main impact category in improper waste treatment phase, due to toxic emissions of lead, arsenic, hydrogen fluoride, PAHs and other chemicals.

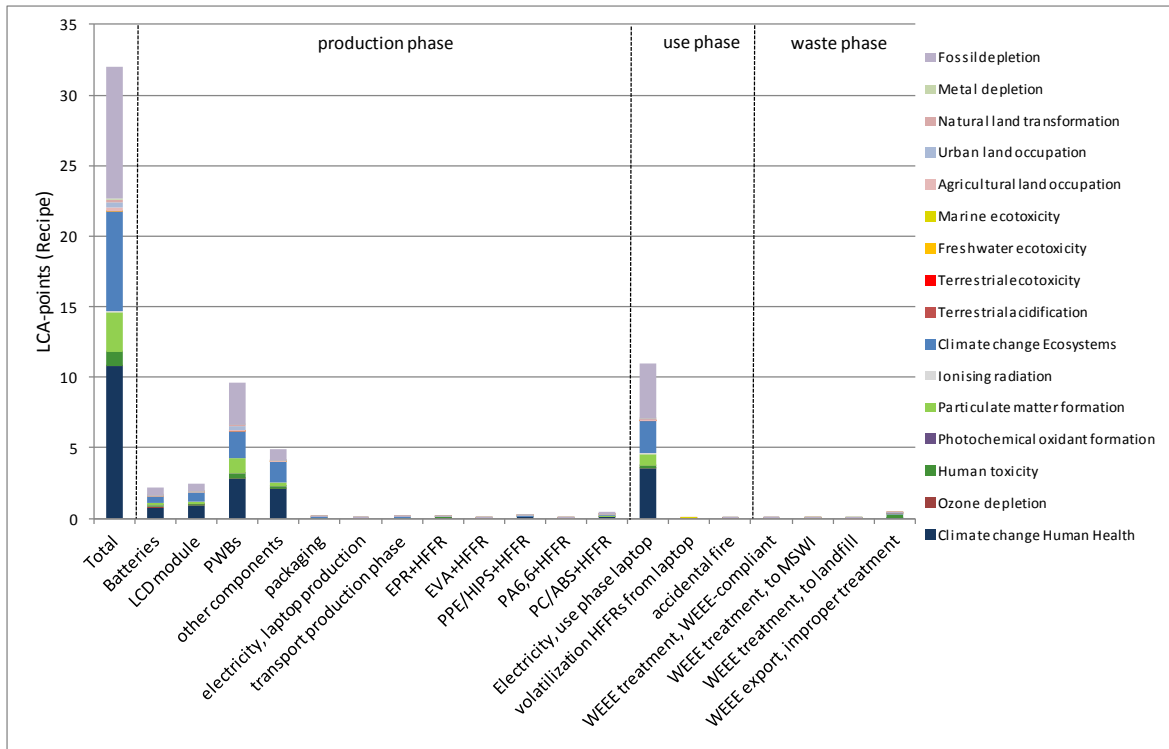


Figure 16: Environmental impact for each phase of the life cycle of the laptop scenario containing HFRs. Scores are expressed in LCA-points according to the ReCiPe method. Colours indicate the contributions of the different environmental impact categories.

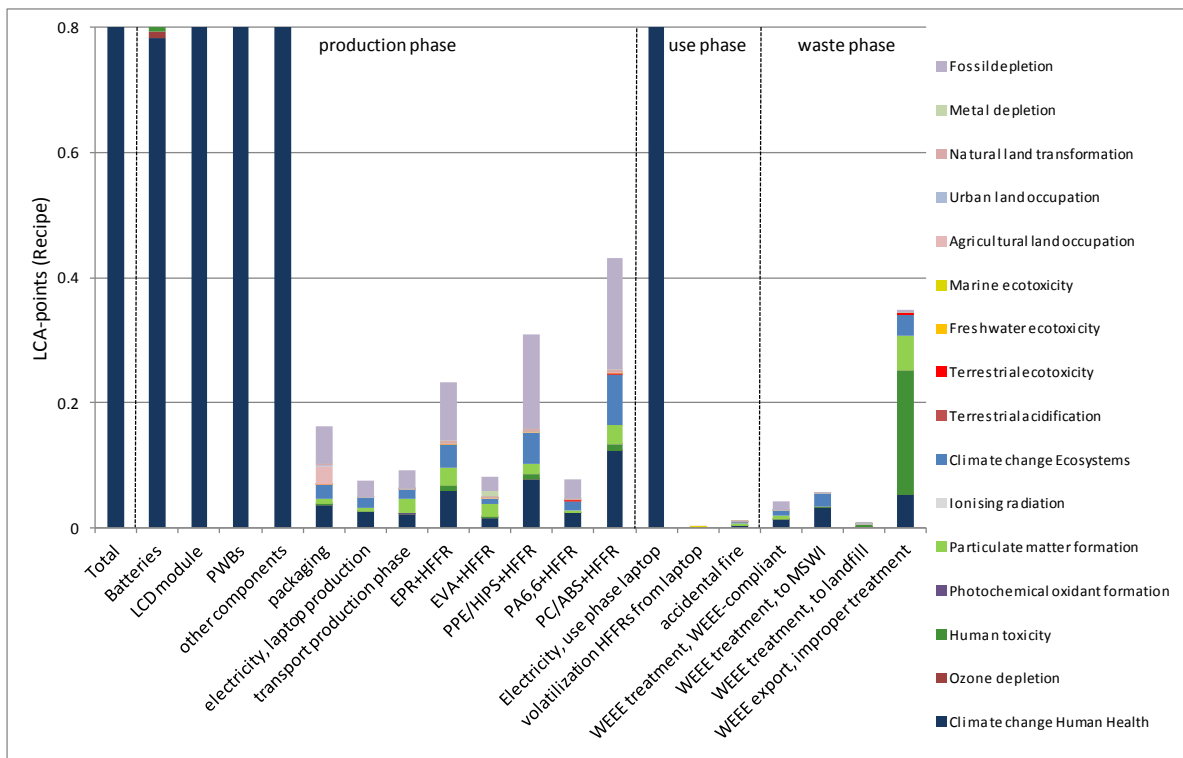


Figure 17: Environmental impact for each phase of the life cycle of the laptop scenario containing HFRs. Y-scale has been adjusted to show the processes/phases with lower scores. Scores are expressed in LCA-points according to the ReCiPe method. Colours indicate the contributions of the different environmental impact categories.

4.7 Comparison of BFR and HFFR scenarios

As was shown in paragraphs 4.5 and 4.6, there are many similarities between the results for the BFR and HFFR scenarios. Figure 18 shows the scores of both scenarios for each impact category, expressed as fraction of the highest score (100%). For many impact categories, the scores of both scenarios are almost equal. The main differences between the scenarios are observed for the impact categories human toxicity, terrestrial, marine and freshwater ecotoxicity and metal depletion. For these five impact categories, the impact is lower in the HFFR scenario.

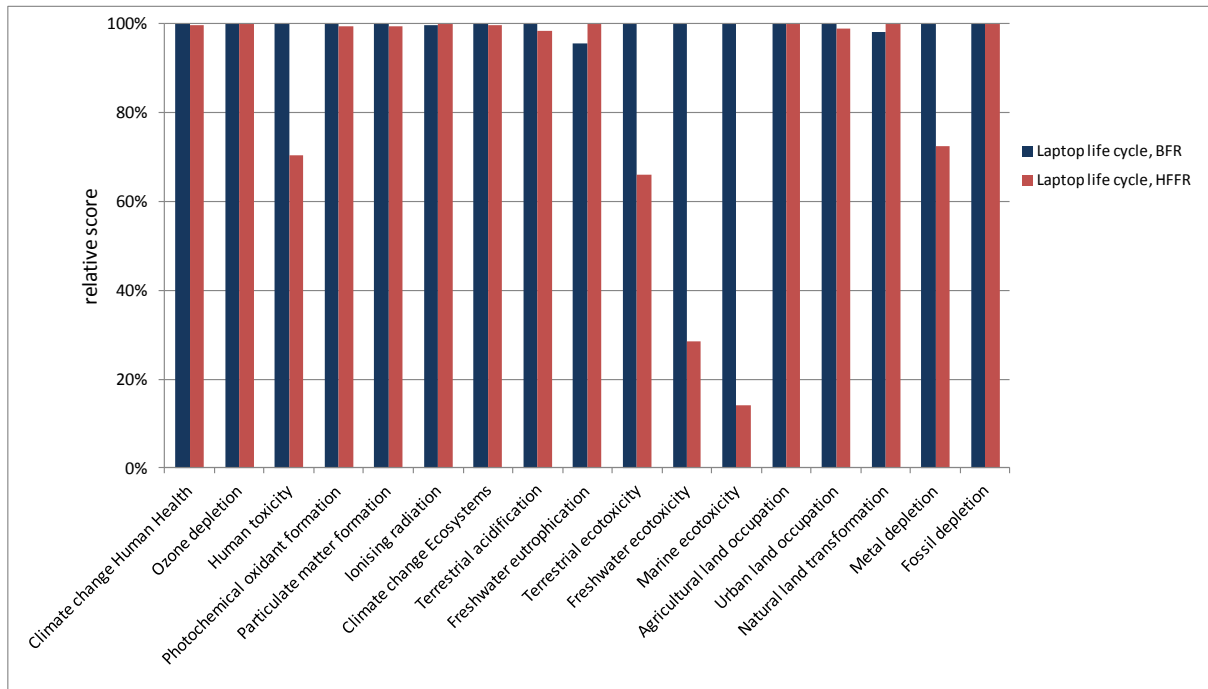


Figure 18: Comparison of scenarios: normalized characterization results for the complete life cycle of the scenario with BFRs and with HFFRs.

The influence of the substitution of BFRs by HFFRs was further evaluated by subtracting the complete laptop scenario with HFFRs from the complete scenario of the laptop with BFRs. The result indicates the difference in environmental performance in each phase of the life cycle, and is shown in figures 19 and 20. A positive score indicates a higher environmental impact for BFRs, and a negative score signifies a higher impact for HFFRs.

The total score is positive, indicating the overall environmental impact of the life cycle of the BFR containing laptop is larger than the score of the HFFR containing laptop. The largest differences are found for human toxicity in the improper treatment phase. A relatively large difference in the WEEE-Directive compliant waste treatment phase is found as well.

In the production phase of the flame retarded polymers, smaller differences are found, with net positive scores for PPE/HIPS and PC/ABS (higher score for the BFR scenario), and net negative scores for EPR, and EVA (higher score for the HFFR scenario). For PA6,6, the net score is almost 0 (equal impact of the HFFR and BFR version).

Relatively small differences are found between the BFR and HFFR scenario for FR volatilization during the use phase, emissions during accidental fire, during incineration in a MSWI, and for the WEEE fraction

ending up on a landfill. The differences for these life cycle phases are positive, indicating a slightly higher impact for the BFR scenario.

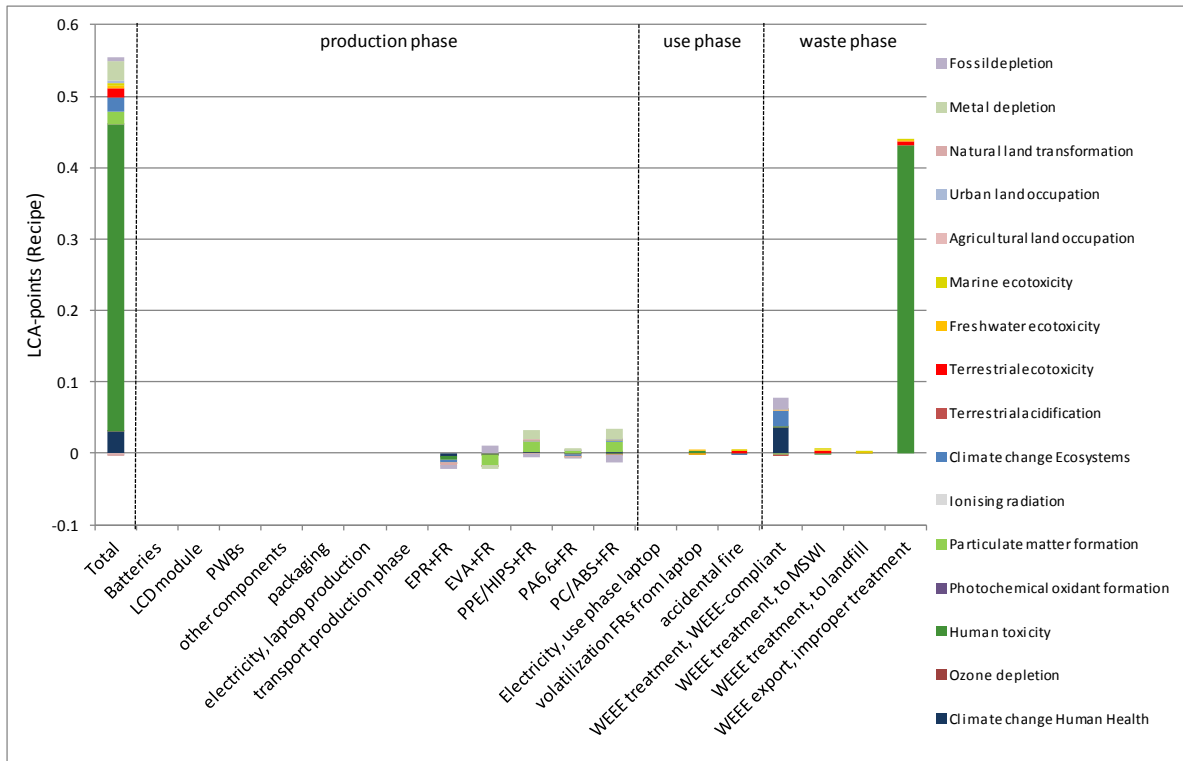


Figure 19: Comparison of scenarios: the results of the scenario with HFFRs subtracted from the scenario with BFRs. Each phase in the life cycle of the laptop is indicated separately. A positive score indicates a higher environmental impact for BFRs, and a negative score signifies a higher impact for HFFRs.

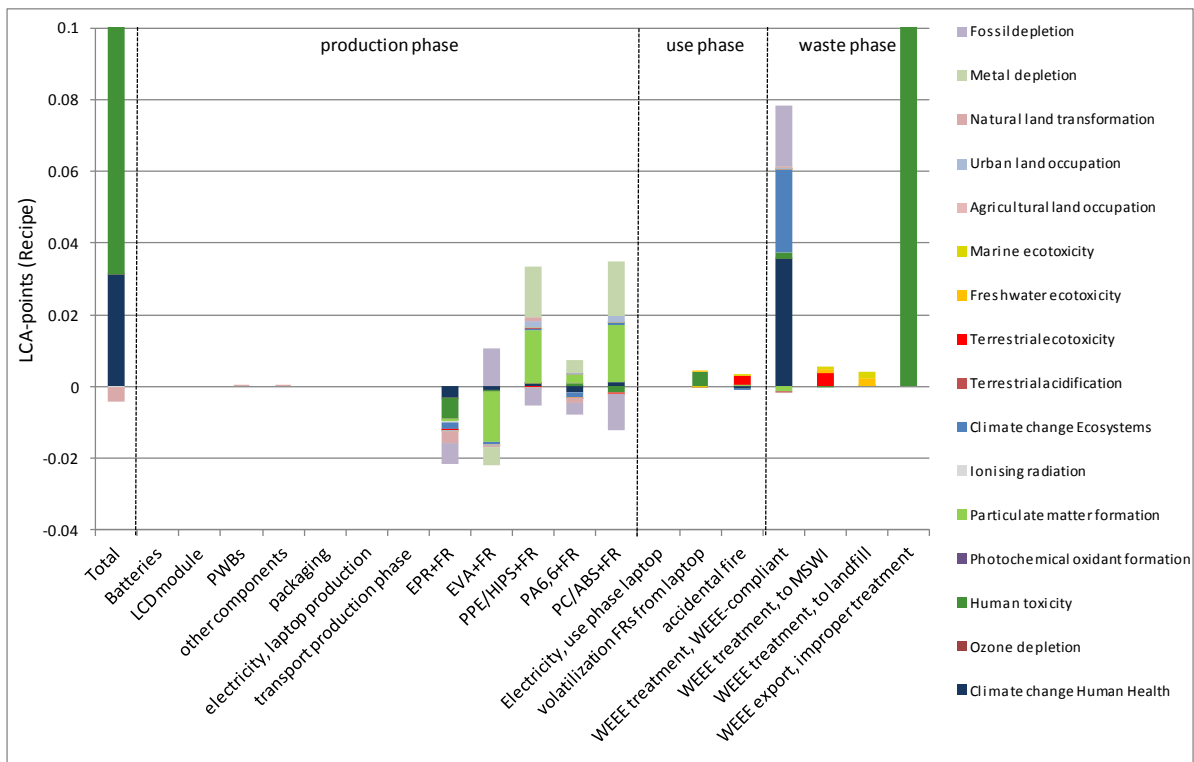


Figure 20: Comparison of scenarios: the results of the scenario with HFFRs subtracted from the scenario with BFRs. Y-scale has been adjusted to show the processes/phases with lower scores. Each phase in the life cycle of the laptop is indicated separately. A positive score indicates a higher environmental impact for BFRs, and a negative score signifies a higher impact for HFFRs.

5. Interpretation and discussion

5.1 Sensitivity analysis

In chapter 3, the assumptions and choices made during the construction of the scenarios were explained. In the current chapter, some of the key choices are evaluated, and their influence on the LCA result is investigated.

The following parameters are varied, in order to find out how robust the results from the LCA scenarios are.

- different sets of HFFRs were selected for 4 of the 5 polymers
- electricity mix was varied from the average European production mix to the Northern European mix, as well as to the Chinese production mix
- the source of brine for the production of bromine (50% enriched and 50% unenriched brine in the BFR scenario) was varied from 100% enriched to 100% unenriched
- the emissions of white phosphorus during the production of some of the phosphorus containing FRs (based on Ecoinvent data) was varied
- the formation of brominated dioxins/furans during improper WEEE treatment has been varied between the lowest to the highest estimations from literature.
- in the future, it is expected that a higher percentage of WEEE will be properly treated according to the WEEE-Directive. The influence of this expected change on the environmental profile has been modelled.

5.1.1 Different set of HFFRs

The selection of HFFRs and their loading rates in the HFFR scenario was based on the fire safety tests and material performance tests of the ENFIRO project. Several other FR-polymer combinations were found to also fulfil the requirements. In a sensitivity analysis, the HFFR scenario was constructed with these alternative FR-polymer combinations, to check whether this leads to different results in environmental performance. These alternatives were already mentioned in Table 3 (paragraph 3.3), and are listed again in the Table below.

Table 40 Sensitivity analysis: the FR-polymer combinations and loading rates of the main HFFR scenario and the alternative HFFR scenario.

Polymer	Amount in laptop (kg)	main HFFR scenario		alternative HFFR scenario	
		compound	% weight	compound	% weight
PA6,6 + GF	0.114	Alpi + MPP	16.7 + 8.3	Alpi + MPP + ZS	13.3 + 6.7 + 3
EVA	0.159	ZHS-coated ATH	70	ZHS-coated ATH	70
PPE/HIPS	0.516	RDP	15	BDP	15
PC/ABS	0.651	BDP	20	RDP	20
epoxy resin	0.293	DOPO	25	Alpi + ATH	15 + 45

Figures 21 and 22 show the LCA results for the scenario with alternative HFFRs. The only significant difference with the original HFFR scenario is found for the production of flame retarded EPR. The high loading rate of ATH in the alternative HFFR scenario reduces the environmental impact for this life cycle

phase. The reduced environmental impact for FR-EPR production is also visible in Figure 22, where the difference between the BFR and HFFR scenario for this process is now positive (environmental impact for this process higher in the BFR scenario than in the HFFR scenario), whereas this value was negative in the original scenario (figure 20, paragraph 4.7).

This sensitivity analysis indicates that by changing the HFFR-polymer combinations, differences in the environmental impact for the production phase can occur, while for all other phases (use phase, End-of-Life), significant changes are not like likely. The overall result that the substitution of BFRs by HFFRs leads to environmental benefit seems to be robust.

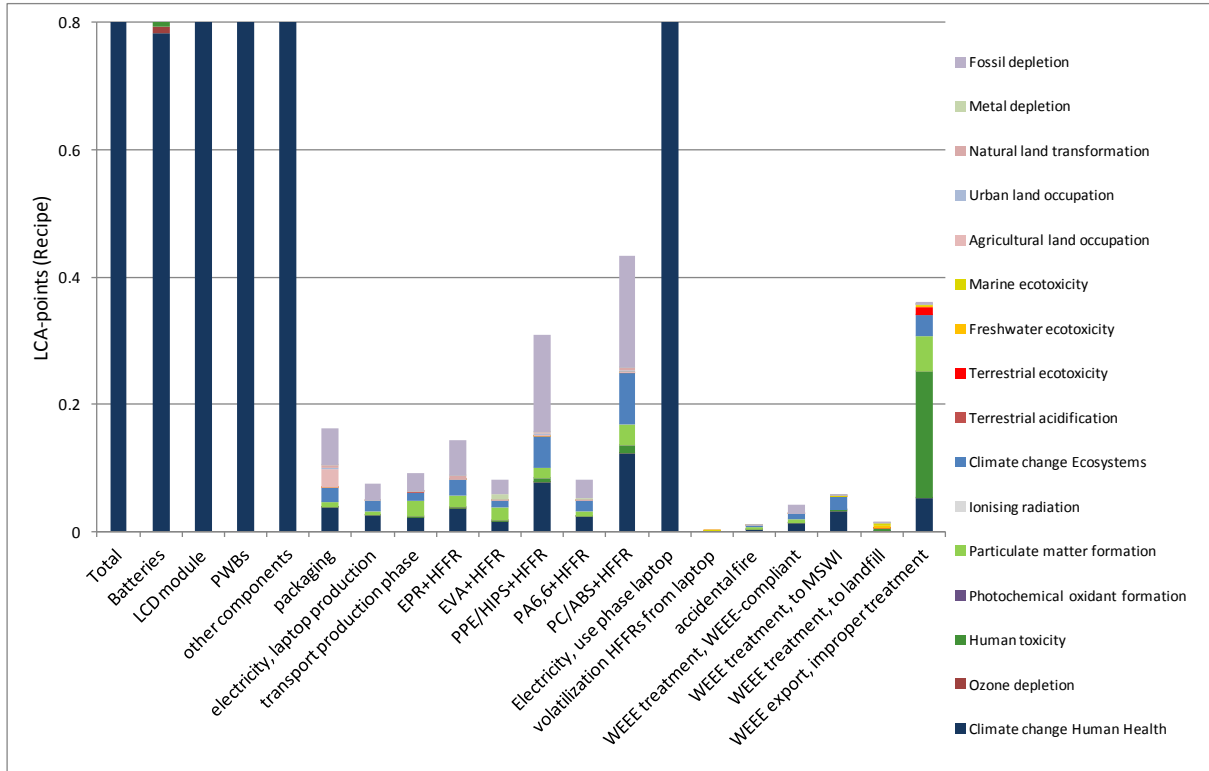


Figure 21 Sensitivity analysis: the LCA results of the alternative HFFR scenario. Y-scale has been adjusted to show the processes/phases with lower scores. Each phase in the life cycle of the laptop is indicated separately.

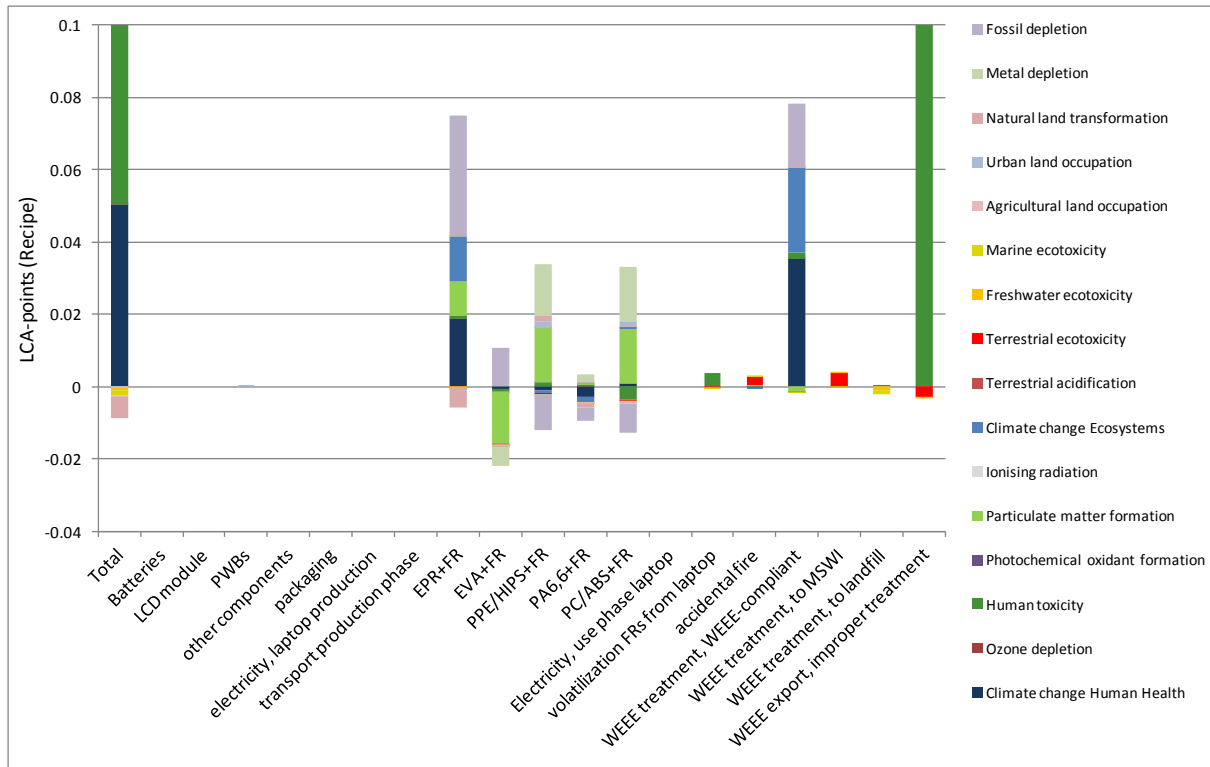


Figure 22 Comparison of scenarios, sensitivity analysis: the results of the scenario with HFFRs (2nd alternatives) subtracted from the scenario with BFRs. Y-scale has been adjusted to show the processes/phases with lower scores. Each phase in the life cycle of the laptop is indicated separately. A positive score indicates a higher environmental impact for BFRs, and a negative score signifies a higher impact for HFFRs.

5.1.2 Electricity mix for FR production

The collection of data on the production processes, energy consumption and emissions of the FRs was performed in a uniform way, as described in paragraph 3.4. Uncertainties in these data are therefore similar for the different FRs. One uncertainty is the yields of FR production: no details were known for any of the FRs, and therefore yields of 100% were assumed. Some companies have stated that unreacted chemicals are reused within the factory.

LCA results for the production phase show that energy consumption has an important contribution to the environmental impact for all FRs. For most FRs, production in Western Europe is assumed, and therefore the average European electricity production mix was used in the model. As a sensitivity analysis, different electricity production mixes were selected, to evaluate the influence on the LCA result when the FR production would take place in other regions. A scenario was constructed in which the Northern European electricity mix is used during FR production, and a scenario in which the Chinese electricity mix is used.

The comparison between production in China and production in Europe is shown in Figure 23. Due to the higher environmental impact per kWh electricity for the Chinese electricity mix, the impact per kg FR produced is between 8 and 53% higher for the Chinese version, compared to the European version (on average 21% higher).

In the current sensitivity analysis, only the electricity production mix was varied, as these were both present in the Ecoinvent database. It is however likely that other industrial processes differ as well between the European and Chinese situation (e.g. heat generation, chemical emissions). As no comparative data were available, these possible differences were not incorporated into the calculations.

In the comparison between the BFR and HFFR scenarios (figure 24), the higher impacts of production of both BFRs and HFFRs partly cancel each other out. Only marginal changes are observed in the production phase of the five FR-polymer combinations, with an increase in impacts slightly larger in the HFFR scenario than in the BFR scenario for flame retarded EPR, EVA, PA6,6 and PC/ABS. For PPE/HIPS, the impact increase in the BFR scenario is slightly larger than the increase in the HFFR scenario.

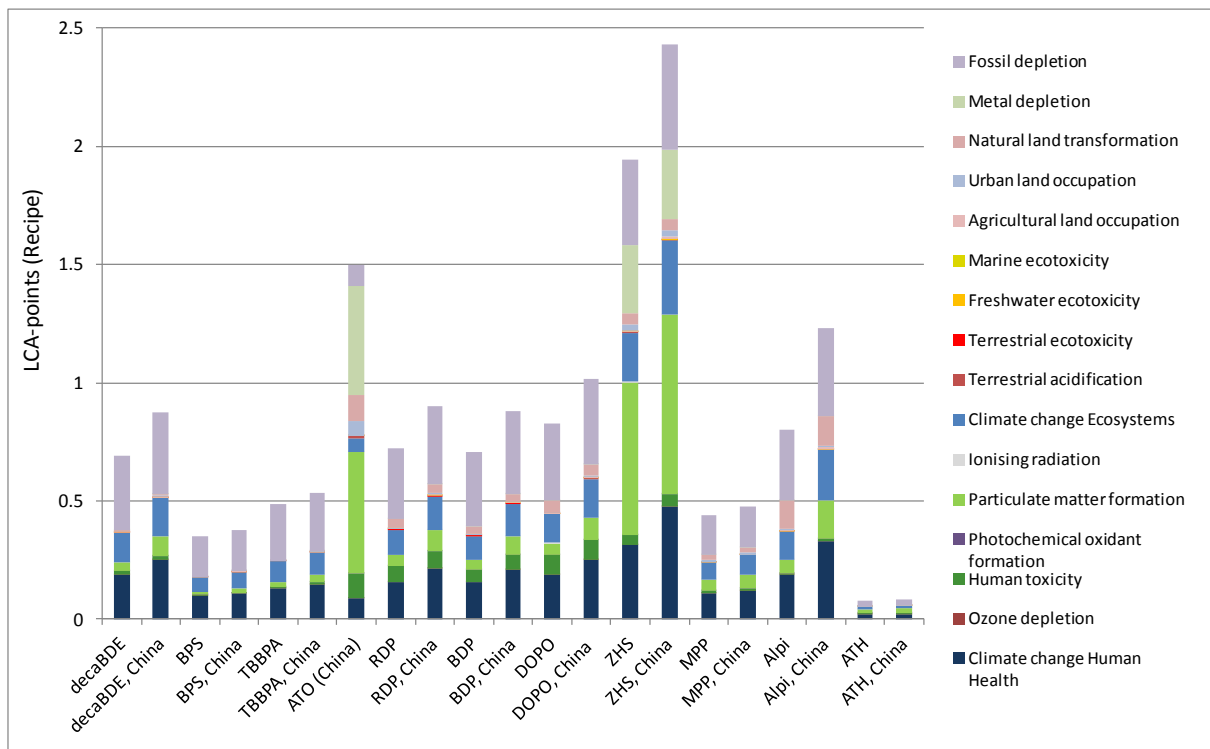


Figure 23 Sensitivity analysis: comparison of FR production using the European and Chinese electricity mix.

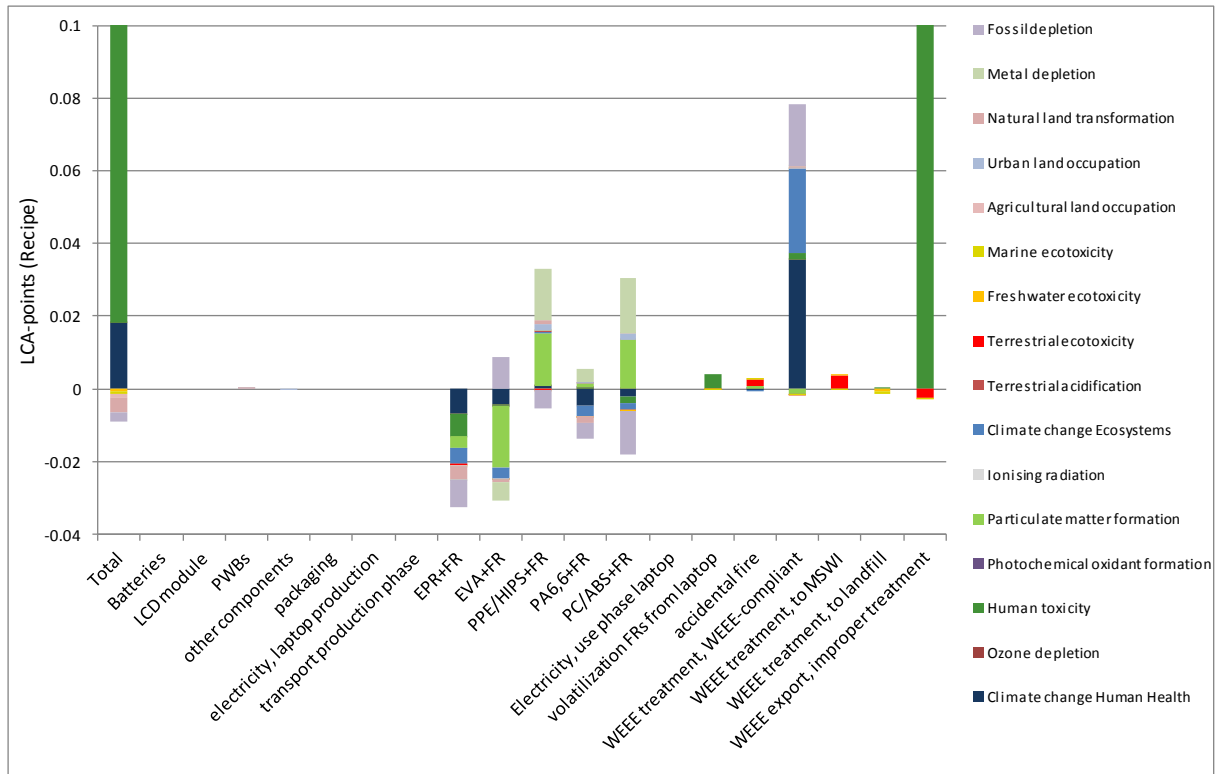


Figure 24 Sensitivity analysis: comparison of FR production using the European and Chinese electricity mix. Each phase in the life cycle of the laptop is indicated separately. The results of the scenario with HFFRs are subtracted from the scenario with BFRs. A positive score indicates a higher environmental impact for BFRs, and a negative score signifies a higher impact for HFFRs. The Y-scale has been adjusted to show the processes/phases with lower scores.

5.1.3 Source of brine for bromine production

For the production of bromine, large amounts of brine are used. The amounts of brine needed, and the energy required for heating the brine, depend on the bromide concentration in the brine. Brine from Dead Sea water contains around 13000 ppm bromide, while this concentration is around 5000 ppm for brine used in the USA (see also paragraph 3.4.2). In the main scenario, the use of a 50/50 mixture of enriched and un-enriched brine is assumed.

A sensitivity analysis is performed to show the influence of the use of different brines on the final LCA result, by varying the brine for bromine production from 100% unenriched to 100% enriched.

Figure 25 shows that scores for the toxicity impact categories show a relatively small variation with variation of the brine source: maximum 5% difference for decaBDE and TBBPA, and slightly larger for BPS (maximum 15%). For the climate change score, variations are larger when the brine source is changed: between 16 and 25%.

Total LCA scores for BFR production are shown in figure 26. When unenriched brine is used instead of the brine mix of the main BFR scenario, total scores increase by 15 and 23% for decaBDE/TBBPA and BPS, respectively. When enriched brine is used, scores decrease by the same percentages.

It is clear from this sensitivity analysis that the brine source has a significant influence on the environmental impact of BFR production. When the values are compared to the scores for the production of the other investigated FRs (figure 6, paragraph 4.1), their order in terms of environmental impact (from highest to lowest impact) is not affected, when the variability due to brine sources is taken into account.

Therefore, the main conclusions of this study are not affected by the variability discussed in this sensitivity analysis.

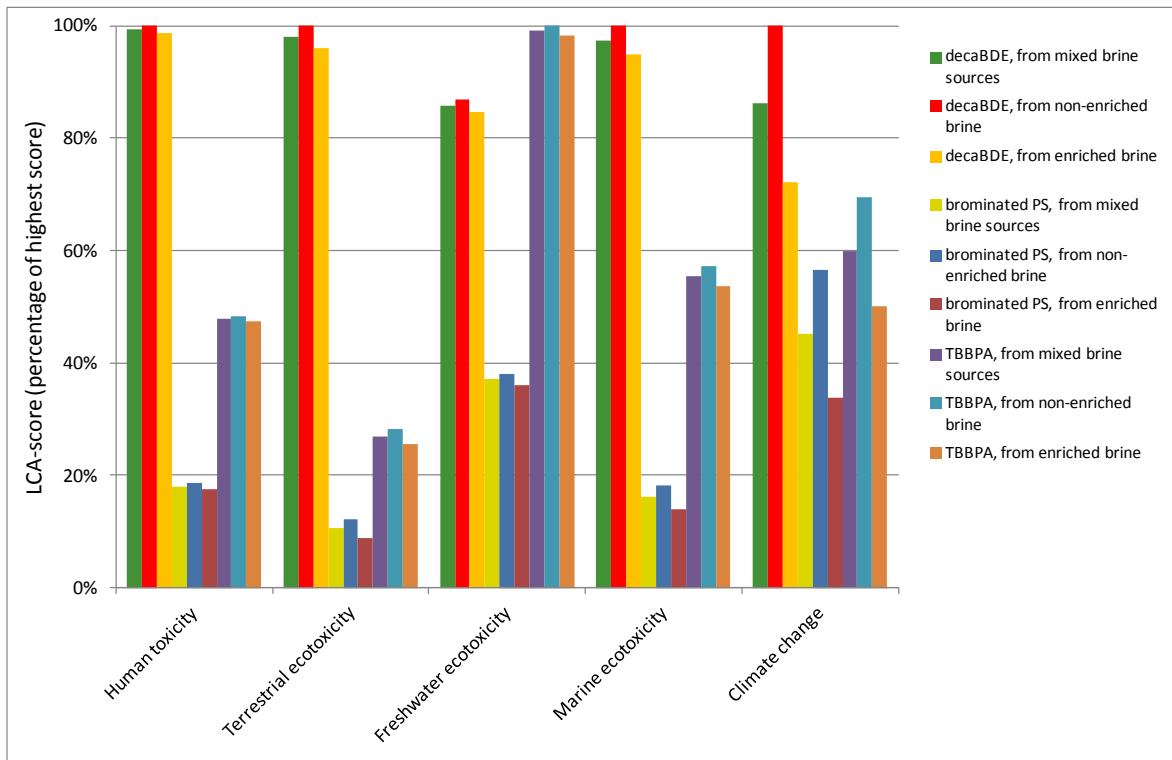


Figure 25: Sensitivity analysis: environmental impact for the production of BFRs, using different brine sources. Results for the four toxicity impact categories and climate change of the ReCiPe method are shown. The score in each category is expressed as fraction of the FR with the maximum score (100%).

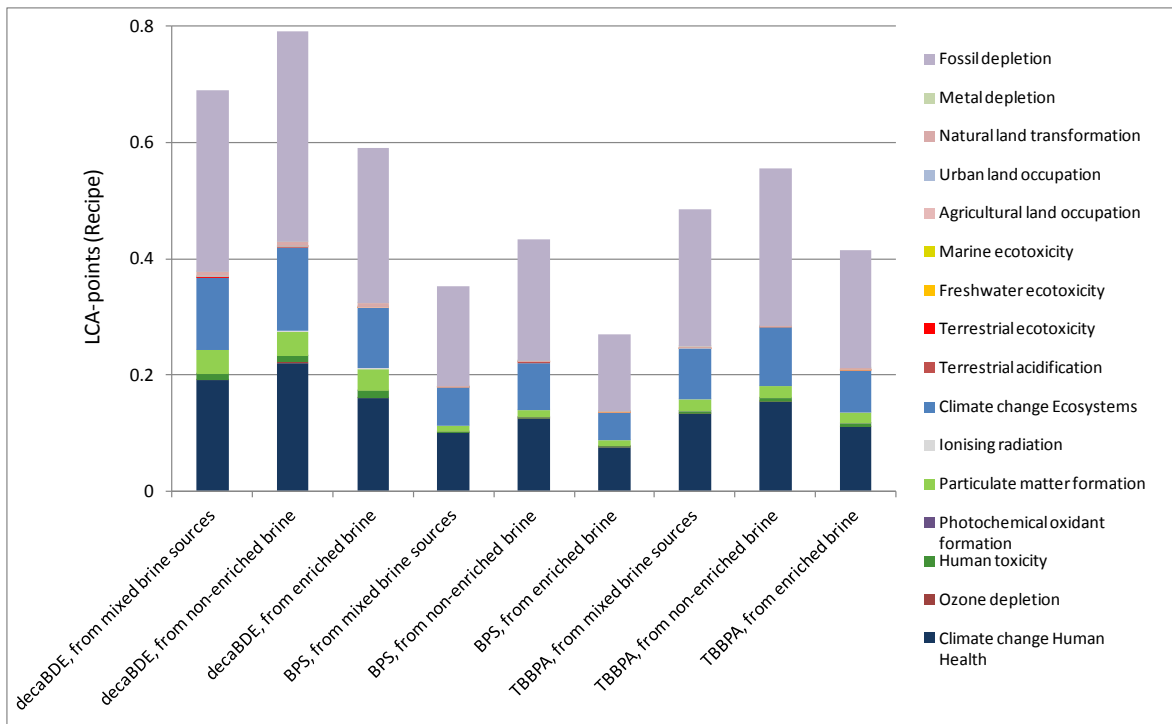


Figure 26: Sensitivity analysis: the total environmental impact for the production of BFRs, using different brine sources.

5.1.4 Phosphorus emissions during FR production

In paragraph 4.1, it was mentioned that for the production of three phosphorus-containing FRs - RDP, BDP and DOPO - relatively high scores were found for human toxicity and terrestrial ecotoxicity. This could be traced to the emission of small amounts of white phosphorus during the production of POCl_3 (phosphoryl chloride) and PCl_3 (phosphorous chloride), which are intermediary products in the production route⁸. This emission originates from the datasheet for POCl_3 and PCl_3 production from the Ecoinvent database, and was not specifically reported by producers.

The influence of these white phosphorus emissions on the overall result was investigated in a sensitivity analysis by constructing equal production scenarios in which the emission of white phosphorus ('phosphorus') was replaced by 'phosphorus, total' (which has a considerably lower characterization factor for toxicity).

The results for the four toxicity impact categories are shown in figure 27. Results for the production of all investigated FRs is shown, with two scenarios each for RDP, BDP and DOPO. It is clear that for all three FRs the scores for human toxicity and terrestrial and freshwater ecotoxicity is strongly reduced when the original emission of white phosphorus is replaced by 'phosphorus, total'. As expected, the substitution of the phosphorus emissions has no influence on the impact category 'climate change'.

The influence of the phosphorus emissions on total environmental impact of FR production is shown in figure 28. Again, results for the production of all investigated FRs are shown, with two scenarios each for RDP, BDP and DOPO. Substitution of emissions of white phosphorus by 'phosphorus, total' leads to a reduction in the total score of 7 to 9% (for BDP and DOPO, respectively).

This sensitivity analysis shows that chemical emissions in any phase of the production process can have a significant influence on results, when looking at specific impact categories. The case of white phosphorus is rather extreme, as this substance has high characterisation factors for human toxicity and terrestrial ecotoxicity. For most (less toxic) substances, the contribution to the toxicity score would be considerably lower.

As the Ecoinvent database indicates that the emission of phosphorus was not based on measurements but on estimations, it is well possible that the lower impact found in this sensitivity analysis is more appropriate.

Figure 28 shows that the order in terms of total environmental impact for FR production (from highest to lowest impact) is not affected, when the alternative production scenario with lower impact from phosphorus emissions is chosen. Therefore, the main conclusions of this study are not affected by the variability discussed in this sensitivity analysis.

⁸ The FR Alpi also contains phosphorus, but its production route does not involve PCl_3 , and no emissions of white phosphorus have been reported by producers.

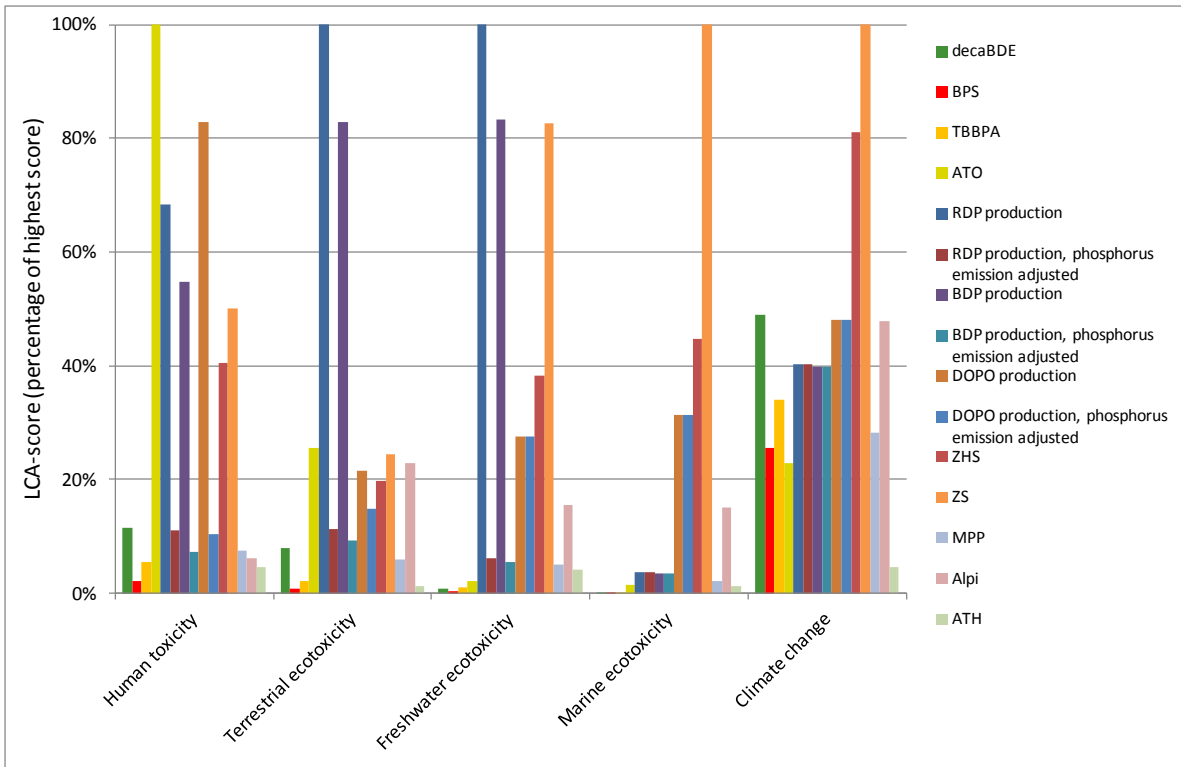


Figure 27: Sensitivity analysis: influence of emissions of white phosphorus during FR production on the scores of the four toxicity impact categories and climate change of the ReCiPe method. The score in each category is expressed as fraction of the FR with the maximum score (100%).

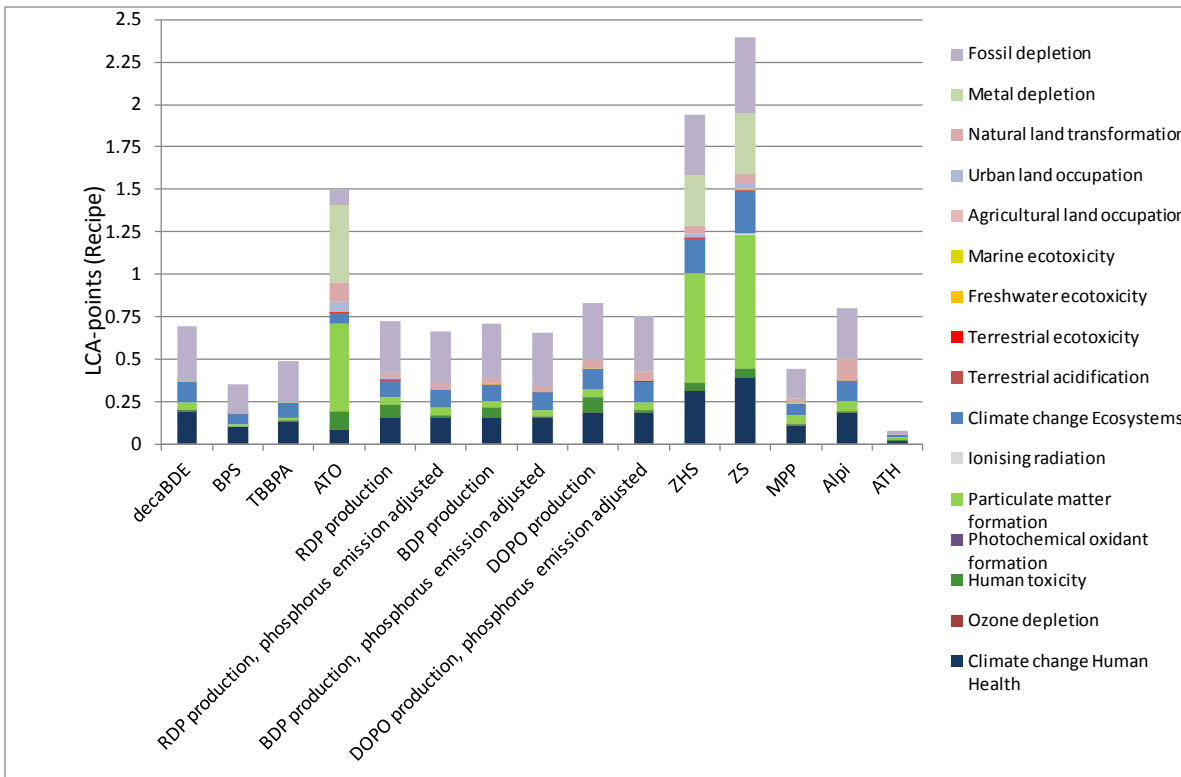


Figure 28: Sensitivity analysis: influence of emissions of white phosphorus during FR production on the total environmental impact for the production of FRs.

5.1.5 Dioxin formation during improper WEEE treatment

Of the four investigated End-of-Life options, the export of WEEE followed by improper treatment was shown to have the highest environmental impact (see paragraph 4.4).

Emissions during improper WEEE treatment are hard to estimate, but from literature describing field studies, simulation tests and laboratory experiments, this waste treatment scenario could be described in sufficient detail. The formation of dioxins was found as a main issue determining the impact of this scenario. However, estimations in literature of the extent of dioxin formation vary by orders of magnitude. In this sensitivity analysis, waste scenarios using the lowest and highest reported yield of (brominated) dioxin formation are compared. As described in paragraph 3.8.4, these values are 9.2 µg per kg WEEE (Gullett, 2007) and 28.7 mg per kg WEEE (Weber, 2003).

It is clear from the results in Figure 29 that the higher estimate of dioxin formation leads to a much higher score for human toxicity. For the other toxicity categories, hardly any change in score occurs, and as expected, the change in dioxin formation does not affect the score for climate change at all.

Figure 30 shows the total scores for the waste treatment options. The extent of dioxin formation has a strong influence on the total score for improper WEEE treatment. In all cases, this End-of-Life option remains the one with the highest score, although for the lower estimation for dioxin formation, the BFR scenario scores only a fraction higher than for the improper WEEE treatment in the HFFR scenario.

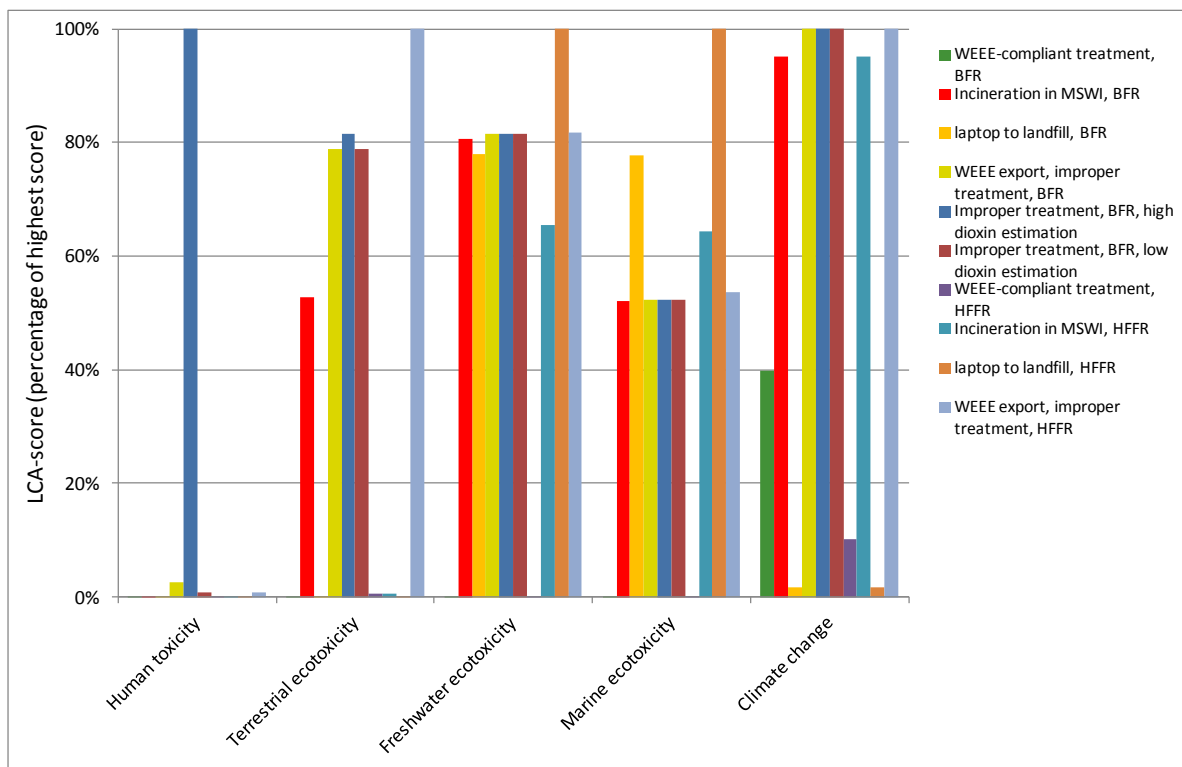


Figure 29 Sensitivity analysis: influence of dioxin formation during improper WEEE treatment of one laptop on the scores of the four toxicity impact categories and climate change of the ReCiPe method. The score in each category is expressed as fraction of the FR with the maximum score (100%).

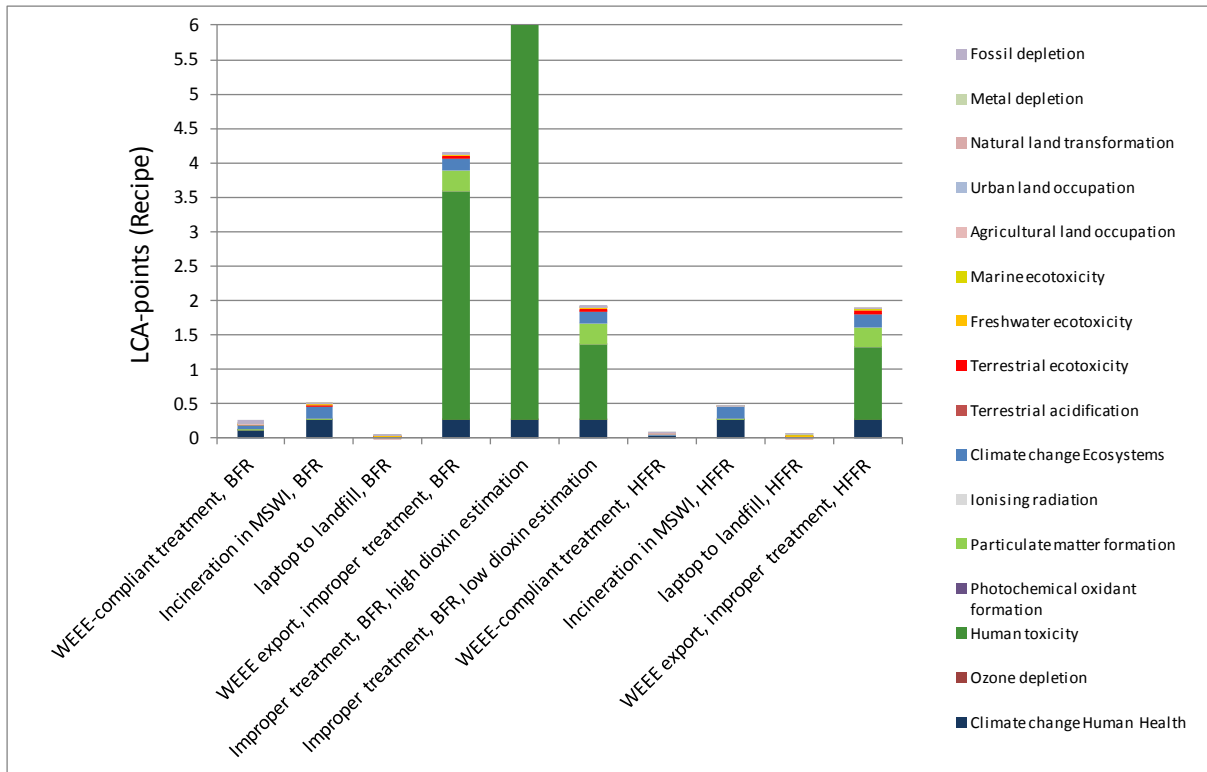


Figure 30 Sensitivity analysis: influence of dioxin formation during improper WEEE treatment on the total environmental impact of the waste treatment of one laptop. The fraction of laptops actually ending up in the different EoL scenarios is not incorporated in this figure.

5.1.6 Future increase in WEEE-compliant treatment

According to the most reliable data available, 44% of IT equipment is currently treated according to the WEEE-Directive (see paragraph 3.8). Of the four End-of-Life options, this is the largest fraction. However, the aim of the European Union is to improve WEEE collection and treatment such that (almost) all of the WEEE will be treated properly. This sensitivity analysis shows how the overall environmental score changes when the percentage of properly treated WEEE doubles to 88%. The rest of the WEEE ends up in the other three End-of-Life options, in the same ratios as in the main scenario. This results in a distribution of waste as shown in Table 39.

Table 41 Sensitivity analysis: adjusted percentages of IT waste ending up in different waste scenarios.

End-of-Life option	Percentage of IT waste ending up in this waste scenario
WEEE-compliant treatment	88%
Incineration (MSWI)	2.6%
landfill	5.3%
Export, improper treatment	4.1%

Figure 31 shows how the scores change with the adjusted End-of-Life percentages. The impact of improper WEEE treatment is reduced by 78%, while the impact of WEEE-compliant treatment has been doubled, as a double amount of BFR-containing plastics will be incinerated as chemical waste. The overall score for the complete life cycle of the laptop changes only slightly: from 32.5 to 32.0 Pt.

Figure 32 shows the same changes for the HFFR scenario: the adjusted End-of-Life percentages result in a 78% impact reduction for improper WEEE treatment, and a doubled impact of WEEE-compliant treatment. Over the complete life cycle of the laptop, the total score decreases only marginally, from 32.0 to 31.7 Pt.

In Figure 33, the differences between the BFR and HFFR scenario are shown, using the End-of-Life percentages from Table 29. With increasing WEEE-compliant waste treatment, the difference between the BFR and HFFR scenario becomes smaller. The overall score of the BFR scenario remains higher than the overall score of the HFFR scenario.

This sensitivity analysis also shows that as WEEE-compliant treatment becomes more successful, the issue of recyclability of flame retarded plastics becomes more prominent. According to the data used in this study (based on a Swedish state-of-the-art WEEE treatment facility), BFR-containing plastics are incinerated as chemical waste, while HFFR-containing plastics undergo some form of material recycling. For both types of plastic, future improvements in recycling possibilities are expected.

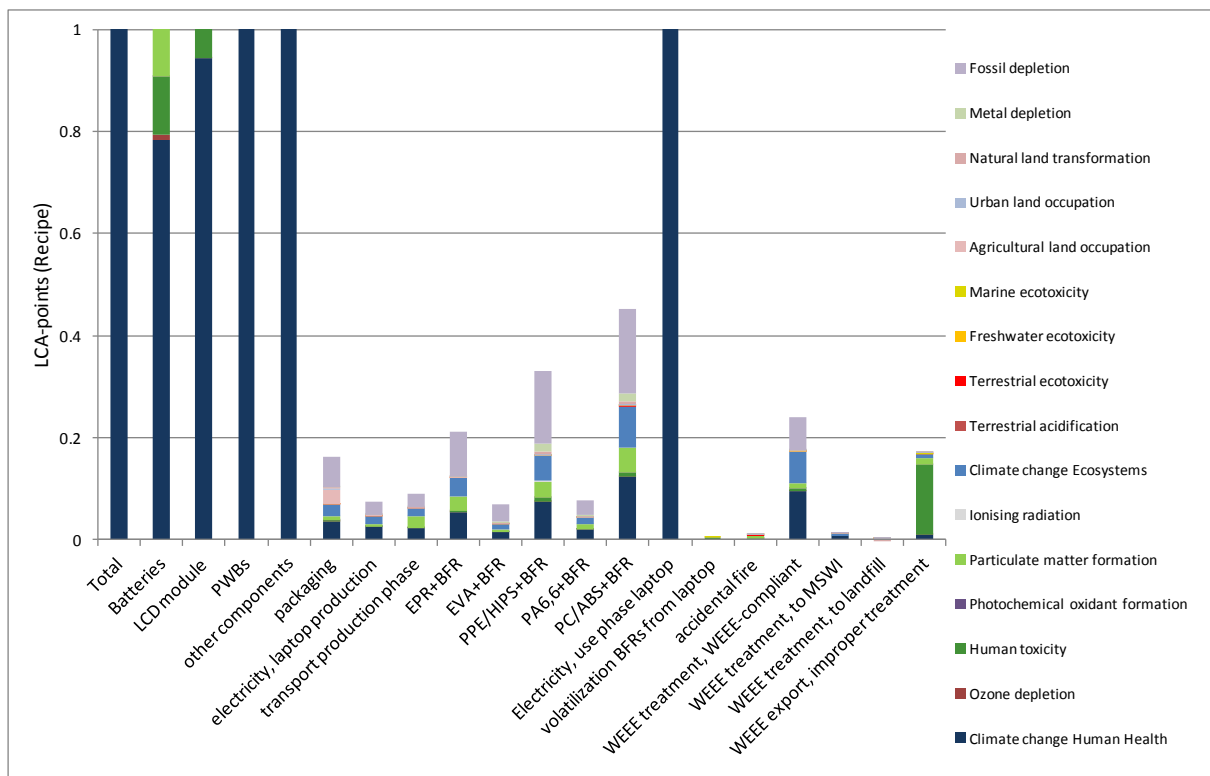


Figure 31: Sensitivity analysis: environmental impact for each phase of the life cycle of the laptop scenario containing BFRs, with WEEE-compliant waste treatment set at double the current percentage. Y-scale has been adjusted to show the processes/phases with lower scores. Scores are expressed in LCA-points according to the ReCiPe method. Colours indicate the contributions of the different environmental impact categories.

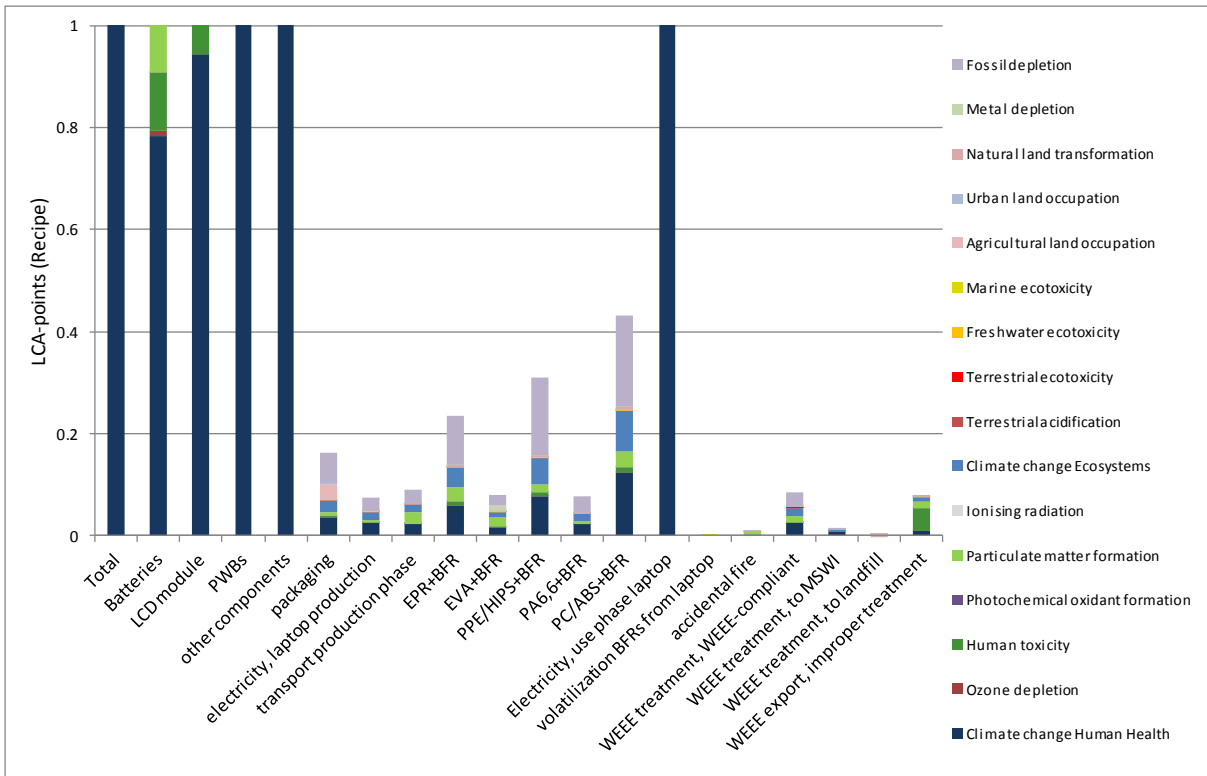


Figure 32: Sensitivity analysis: environmental impact for each phase of the life cycle of the laptop scenario containing HFFRs, with WEEE-compliant waste treatment set at double the current percentage. Y-scale has been adjusted to show the processes/phases with lower scores. Scores are expressed in LCA-points according to the ReCiPe method. Colours indicate the contributions of the different environmental impact categories.

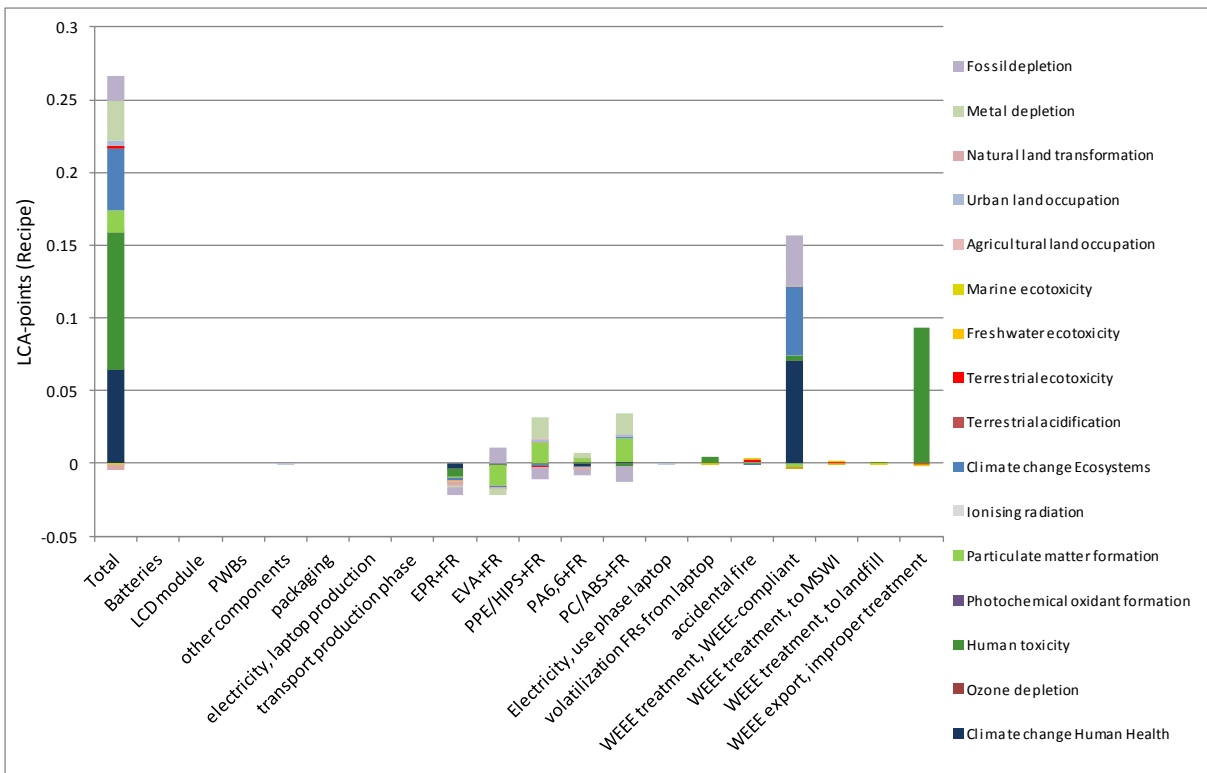


Figure 33: Sensitivity analysis, with WEEE-compliant waste treatment set at double the current percentage: the LCA results of the scenario with HFFRs subtracted from the scenario with BFRs. Each phase in the life cycle of the laptop is indicated separately. A positive score indicates a higher environmental impact for BFRs, and a negative score signifies a higher impact for HFFRs.

5.2 Discussion

5.2.1 Data quality

In this LCA study, a diverse set of processes concerning the life cycle of FRs are addressed. A consequence of this diversity is that the data used in the study has a wide range of sources and quality. The ‘chaotic’ nature of some of the processes modelled is such that precise data on chemical emissions is impossible to obtain. This is the case for accidental fire and improper WEEE treatment. The best possible literature or experimental data was used in these cases, and therefore the present study reflects the current state of knowledge on these issues.

For some processes, experimental data generated within the ENFIRO project could be used to model emissions. This was the case for emissions due to leaching from a landfill, and emissions generated during accidental fire. For a realistic composition of flame retarded polymers, the ENFIRO fire safety test results were used.

Not all data generated by the ENFIRO team could be used in the LCA. For example, it was not possible to properly include results of neurotoxicity experiments, as the models used for the determination of toxicity characterisation factors could not properly incorporate these data. As Table 37 shows (paragraph 3.9.2), toxicity is summarized in LCA by two parameters: a NOEC for aquatic ecotoxicity and an effect concentration for human toxicity (ED50, LOEL or NOEL), and it is hardly possible to comprehensively reflect all knowledge on different toxicological endpoints in these parameters.

All data on production of FRs have been based on scientific literature, publicly available patents or the LCA-database Ecoinvent. For only a few FRs, data directly delivered by companies could be used. It is therefore not entirely certain to which extent the production data used reflects the actual industrial production processes.

As far as possible, the data used concern the average European situation. In some cases, reliable European data were not available. For example, no reliable European data could be found on the percentages of WEEE ending up in the different End-of-Life options. Therefore, a in-depth report describing the situation for the Netherlands was used, and an assumption was made that this was representative for Europe (an adjustment was made for the average European ratio of household waste ending up in incineration and on landfill).

For all modelled processes, the data used was of similar quality for both the BFR and HFFR scenario. Therefore, the comparability of the scenarios remains ensured.

5.2.2 Processes normally not included in LCA

The following three issues are normally not treated in LCA studies:

- indoor human exposure via dust (local impact)
- human exposure to FRs in Asia when burning electronics waste in open fires to recycle the metals (occupational health issue)
- fire occurrence (incident)

These issues have been incorporated in the current study. The LCA results show the impact of these processes within the complete life cycle of the product. Although this ‘big picture’ type of result gives relevant insights, further evaluations of these topics are necessary. More focused risk assessment studies would yield more detailed knowledge of the risks involved in these processes.

The high impact of improper WEEE treatment as modelled in this study shows the potential relevance of ‘informal’ End-of-Life options. Future LCA studies should increasingly include these issues, because ignoring them may result in a quite incomplete description of the full environmental impact of a product’s life cycle.

5.2.3 Toxicity in LCA

The focus of the ENFIRO project was to find viable alternatives to BFRs which are less persistent and toxic. This focus was expressed in this LCA study by determining specific characterization factors of all FRs for human toxicity and freshwater, marine and terrestrial ecotoxicity for emissions to air and water. LCA results were discussed for the specific toxicity impact categories as well as the total LCA scores. To be able to compare the complete life cycle of FRs, a product was chosen as case study. The life cycle of a complete laptop was considered, for which scenarios with BFRs and with HFFRs were compared. By considering a complete laptop, a number of processes and life cycle phases are introduced which are not directly related to the toxicity of FRs. Specifically energy use during production of all laptop components (including materials such as metals and glass) and energy consumption during the laptop use phase score high on impact categories Climate change and fossil depletion, and have a dominating contribution to the overall LCA score. This may distract the attention from toxicity related effects.

By focusing the discussion of the results on the differences between scenarios, toxicity scores (although sometimes small compared to the total LCA scores) were given the attention fitting to the focus of the ENFIRO project.

By also showing the toxicity scores in proportion to other environmental impacts, a ‘big picture’ overview was obtained, which can be considered as an evaluation parallel to the hazard based approach of the search for viable FRs to substitute hazardous BFRs.

6. Conclusions

In this LCA study, a comparison is made between the overall environmental impact of a laptop containing BFRs and a laptop containing HFFRs. The main question to answer was: “Will the substitution of BFRs by HFFRs lead to higher environmental impacts in any phase of the life cycle, and will the overall environmental impact of a product containing HFFRs be different from a product containing BFRs?”. This question can only be addressed by choosing a representative case study, in which all relevant phases of the life cycle are properly described, so that a general answer can be given. For FRs incorporated in polymers, an electronics application was considered to be most representative. A laptop was chosen as case study, in which most of the FR-polymer combinations studied in the ENFIRO project are represented.

6.1 Conclusions

The LCA findings lead to the following conclusions.

- In most phases of the life cycle of FRs, fossil energy use related impact categories dominate the LCA score: Climate change, Fossil depletion and Particulate matter formation.
- The life cycle phases in which human toxicity and ecotoxicity play the largest role are:
 - export of WEEE followed by improper waste treatment. In this waste scenario, the formation of dioxins during improper incineration of BFR containing plastics has the largest contribution to human toxicity.
 - emissions of FRs during volatilization (or wearing/abrasion) in the use phase have LCA scores only in the toxicity impact categories. Emission factors of FRs are considered to be low.
 - during accidental fire, emission of ATO to air has a relatively high score for terrestrial ecotoxicity. However, when considering the complete life cycle, only a small fraction of the laptops will end up in an accidental fire, and therefore accidental fire has only a small contribution to the total score.
 - for the waste scenarios MSWI incineration and landfill, contributions to the toxicity impact categories come from both FR related (ATO, bromine, ZHS, ATH and Alpi) and non-FR related emissions (heavy metals).
- The environmental impact in the production phase of FRs (cradle-to-gate, per kg) varies considerably. The highest impacts are found for ZS, ZHS and ATO. Lower total impacts are found for decaBDE, RDP, BDP, DOPO and Alpi, with differences in total scores within 20%. Then follow TBBPA, MPP and BPS, and the lowest impact is found for the production of ATH. For the three FRs with the highest environmental impact (ZHS, ZS, ATO), the raw material mining phase has a high contribution to the total score.
- For the production of flame retarded polymers (cradle-to-gate, per kg), differences in environmental impact between BFR and HFFR are not very large, with maximum differences of 16%. This is partly due to the fact that the production of polymer has a high contribution to the score (which varies only moderately between scenarios, due to different FR loading rates), and partly because the FRs with high impact per kg are used at lower loading rates, while FRs with low impact per kg are used at higher loading rates.

For PA6,6, no significant difference in total score between the BFR and HFFR version is found. For EVA and EPR, the HFFR version has a higher score (16 and 10%, respectively), and for PPE/HIPS and PC/ABS, the BFR version scores higher (9 and 5%, respectively).
- Emission of FRs in the use phase through volatilization, wearing or abrasion cause the highest impact on human toxicity and freshwater ecotoxicity for the BFR scenario, and the highest impact on

terrestrial and marine ecotoxicity for the HFFR scenario. In the overall score, the BFR scenario has the highest score, through the contribution of human toxicity.

The emissions of FRs in this phase have only a small contribution to the overall impact over the complete life cycle, but this phase is still likely to be the most important exposure route for humans

- In the case of accidental fire, the BFR scenario has a higher overall impact than the HFFR scenario due to a higher rate of smoke formation and a higher terrestrial ecotoxicity score. In the HFFR scenario, the score for Climate change is higher than in the BFR scenario due to higher CO₂ emissions (more complete combustion)
- Of the four End-of-Life scenarios for WEEE, the option ‘export followed by improper treatment’ has the highest environmental impact for both the BFR and HFFR scenarios. In the BFR scenario, this high impact is mainly caused by the formation of (brominated) dioxins during improper WEEE incineration. Rates of formation of dioxins out of BFRs is reported in literature show a wide range. A sensitivity analysis showed that the extent of dioxin formation has a strong influence on the total score for improper WEEE treatment. In all cases, this End-of-Life option remains the one with the highest score, although for the lower estimation for dioxin formation, the BFR scenario scores only a fraction higher than for the improper WEEE treatment in the HFFR scenario.
- The high LCA score for improper WEEE treatment in the HFFR as well as the BFR scenario shows that even when BFRs are substituted by HFFRs, these practices are still quite harmful, as there is a range of toxic emissions during improper treatment, including lead, arsenic, hydrogen fluoride, (chlorinated) dioxins and PAHs.
- the End-of-Life option with the second highest impact per kg (after the improper WEEE treatment option) is incineration in an MSWI. WEEE treatment according to the WEEE-Directive has a lower impact per kg, while the lowest impact is found for the landfill option.
- The main differences between the two full life cycle scenarios are found in the scores for the impact categories human toxicity, ecotoxicity and metal depletion. For these five impact categories, the impact is lower in the HFFR scenario. For the other impact categories, the scores of both full life cycle scenarios are almost equal.
- The life cycle phases of the laptop showing the largest differences between the BFR and HFFR scenarios are improper WEEE treatment and WEEE-Directive compliant treatment. In both life cycle phases, the BFR scenario has a higher impact. Smaller differences are found in the production phase of the flame retarded polymers: for PPE/HIPS and PC/ABS the BFR scenario has a higher net score, while for EPR and EVA the HFFR scenario scores higher. For PA6,6, no significant difference is found. For the impacts of volatilization in the use phase, emissions during accidental fire, WEEE treatment in a MSWI and WEEE on a landfill, relatively small differences between scenarios are found, with the BFR scenario having a slightly higher impact in these life cycle phases.
- When considering the life cycle of a complete laptop, the contribution of FRs to the total environmental impact is minor.
- The substitution of BFRs by HFFRs in this laptop case study gives clear environmental benefits in the waste phase, and smaller environmental benefits in other life cycle phases. In two life cycle phases (production of flame retarded EPR and EVA) the environmental impact with HFFRs is higher than with BFRs.

6.2 Recommendations

This study shows that for improvements of the life cycle environmental performance of FRs, the waste treatment phase is critical. When developing new FR systems, their behavior in different End-of-Life options should be taken into account.

Export and improper treatment of WEEE has the highest impact of all waste treatment options for both the BFRs and HFFR scenarios, and efforts should continue (or be intensified) to reduce the amount of European WEEE ending up in this scenario.

Recycling of WEEE is the key to closing the material cycles of electronic products. Currently, the focus is mainly on recycling the precious metals in electronics. For further improvement of the environmental performance of FRs, these activities should be expanded to include recycling of plastics, in such a way that their flame retarding (and mechanical) properties are maintained.

An option that should always be kept in mind (but is not treated in this study) is the possible redesign of electronics so that there is less need for FRs. A separation between energy sources and combustible materials by e.g. metal barriers could reduce the need for FRs. These design challenges are influenced by a number of processes: reduced energy consumption of electronics may reduce the risk of fire, stricter fire safety regulations increase the need for FRs, while the high price of these specific additives (compared to the price of the plastic itself) ensures that a producer will use FRs only when they are really necessary.

This study shows that processes which are often ignored in LCA studies can have an important contribution to the environmental performance of a product (in this case the improper WEEE treatment phase). It is therefore recommended to broaden the scope and system boundaries of future LCA studies to include unofficial or illegal scenario options (specifically in the End-of-Life phase) to provide a more complete description of the full environmental impact of a product's life cycle, and thereby contribute to relevant discussions in society and policy.

The current LCA study is part of the ENFIRO project, which follows mainly a hazard based approach: the search for safe alternatives to hazardous BFRs. The LCA approach uses (measured or estimated) emissions to calculate environmental impacts, and therefore goes beyond the hazard-based approach. By including a number of other environmental impact categories, a different type of evaluation is obtained, which is complementary to the hazard-based evaluation. Although the emphasis of the LCA results is on different aspects than hazardous FRs, they do also confirm that substitution of BFRs by HFFRs lead to a reduction of (eco)toxicological impacts. In research projects focusing on the substitution of hazardous chemicals, LCA analyses produce valuable complementary information which allows a more complete evaluation of the viability and sustainability of alternatives.

Acknowledgements

During the environmental LCA study of the ENFIRO project, a large number of stakeholders and experts were contacted. Many of them delivered useful insights into the life cycle of flame retardants in electronics. Several stakeholders provided quantitative data on specific processes, or pointed to useful literature sources. The ENFIRO stakeholder forum, which was established in order to inform stakeholders on project progress, and gain valuable feedback from the field, held specific meetings throughout the project, while the members were also invited to the public ENFIRO workshops. These meetings yielded a lot of useful information for this LCA study. The ENFIRO stakeholder forum consisted of 17 members, including FR producers, formulators, OEMs, NGOs, and a waste recycler. All stakeholders and experts who shared their knowledge and experience with us, are deeply acknowledged.

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Annex 1: Short description of ReCiPe impact categories.

A short description of the impact categories considered in the ReCiPe model is given in this Annex. More background information can be found in the ReCiPe literature (*Goedkoop, 2009*).

Climate change (human health and ecosystems)

The increased greenhouse effect is causing a number of environmental mechanisms that affect both human health and ecosystems. As these endpoints are expressed in different units (DALY and Species.yr), this impact category has been subdivided at the midpoint level. Carbon dioxide and methane are the best-known greenhouse gases.

Ozone layer depletion

The concentration of ozone between 15 and 30 km height in the atmosphere is called the ozone layer. This layer absorbs a large fraction of the harmful ultraviolet radiation from the sun. The thickness of the ozone layer has decreased, especially since the 1980s. In spring, the ozone above the Antarctic decreases with almost 50%. Also above Europe, the ozone layer has decreased. The ozone layer is affected by certain chemicals such as chlorofluorocarbons (CFCs). In the ozone layer, these compounds disintegrate, and chlorine atoms are formed which break down ozone to chloromonoxyde and oxygen ($\text{Cl} + \text{O}_3 \rightarrow \text{ClO} + \text{O}_2$). Subsequently, the chloromonoxyde molecule falls apart under influence of UV-radiation, and the chlorine atom again reacts with an ozone molecule.

Terrestrial acidification

Acidification of soil or water is a consequence of emissions from factories, agriculture, electricity plants and transport. The main substances responsible are sulphur dioxide (SO_2), nitrogen oxides (NO_x), ammonia (NH_3) and volatile organic compounds (VOCs). The substances reach the soil via air and water, and enter plants via leaves and roots, making them more susceptible to disease. Acid deposition also affects rivers and lakes, and the organisms that reside there.

Freshwater eutrophication

Eutrophication is the increase of nutrient concentrations in water and soil. This leads to increased growth of specific organisms, resulting usually in a decrease in biodiversity. Eutrophication occurs in freshwater due to run-off of manure and fertilizer, resulting in increased concentrations of nitrogen and phosphates. This leads to strong algae blooms. Eutrophication can also lead to hypoxia: decreased oxygen concentrations in the water.

Human toxicity

This impact category comprises the emissions to air, water or soil that (eventually) lead to toxic effects for humans.

Taken into account are the environmental fate and persistence, the accumulation in the human food chain (exposure) and toxicity (based on a disease-specific and a chemical-specific component) of a chemical.

Fate and exposure are calculated by means of 'evaluative' multimedia fate and exposure models.

The chemical 1,4-dichlorobenzene (emitted to urban air) is used as a reference substance in the midpoint calculations.

Photochemical oxidant formation

Smog, a combination of the words smoke and fog, is air pollution by smoke and exhaust gases, causing mists in certain periods, that can cause human health effects. Substances influencing the formation of smog are mainly ozone and fine dust and to a lesser extent nitrogen dioxide and sulphur dioxide.

Particulate matter formation

Fine particulate matter consists of airborne particle with a diameter of less than 10 micrometer, of various chemical compositions. They can cause health problems when inhaled. Chronic exposure to fine particulate matter can cause the increase of symptoms of people with respiratory and cardiovascular diseases. Norms for fine particulate matter are exceeded on many locations in Europe, especially along busy roads.

Ecotoxicity, (terrestrial, freshwater, marine)

Ecotoxicity comprises emissions to air, water or soil that result in toxic effects to organisms in soil, freshwater or marine water.

This impact category takes into account the environmental fate and persistence, the exposure of organisms to the chemical and toxicity (based on a toxic mode of action-specific and a chemical-specific component). Fate and exposure are calculated with 'evaluative' multimedia fate and exposure models. As a reference substance, the chemical 1,4-dichlorobenzene (emitted to freshwater for freshwater ecotoxicity, to seawater for marine ecotoxicity and to industrial soil for terrestrial ecotoxicity) is used in the midpoint calculations.

Ionising radiation

Ionising radiation (also called radioactive radiation) is the result of the decay of radioactive atoms such as Uranium-235, Krypton-85 and Iodine-129. There are two types of ionising radiation: particle radiation (alfa radiation, beta radiation, neutrons, protons) and high energy electromagnetic radiation (X-rays, gamma rays). Ionising radiation can cause DNA damage and can be carcinogenic.

Land occupation (agricultural and urban)

The impact category land occupation reflects the damage to ecosystems caused by the effects of land occupation over a certain time period. The midpoint indicator simply books the square meters of agricultural or urban land occupation without further discriminating between uses.

Natural Land Transformation

Transforming land from one type of use to another, will influence the species diversity of the area. The damage to ecosystems can be calculated by taking into account the time needed for the transformed area to restore to a land-use type with a similar diversity.

Land transformation is linked to land occupation: often occupation follows a transformation. However, often occupation occurs in an area that has already been converted (transformed). In such cases we do not allocate any of the transformation impact to the production system that occupies an area.

Depletion (minerals and fossil fuel)

The impact of the use of minerals and fossil fuels is proportional to the scarcity of the resource. The unit is 'marginal cost increase of extraction' (dollars per kg).

Annex 2: Data on detailed laptop compositions from different literature sources.

Only the polymers were considered in this comparison. Other materials are listed only to indicate the ranges reported in literature.

material	Deng, 2011	Kahhat, 2011	Tekawa, 1998	IVF, 2007	Shrivastava, 2005	HP, 2012	Fredholm, 2008	Ecoinvent 2 (Hischier, 2007)	average	Used in this study (weight %)	Used in this study (kg/laptop)
total plastics	36%	28%	40%	30%	44%	39%	38%	34%	36%		
PC/ABS	21%		31%	15%	29%	27%		2%	21%	21%	0,651
HIPS	<9%		<5,3%	0,11%				16%	8%	16%	0,516
PA66	<9%		<5,3%	10%	0,01%			0,8%	3,6%	3,6%	0,114
EVA	<9%		<5,3%		0,16%			5,0%	3%	5,0%	0,159
epoxy resin	6%		4%	0,11%	4%	3%		9,3%	4%	9,3%	0,293
PWB material (incl epoxy)		12%						15%	13%		
aluminium	14%	14%	4%	1%	16%	8%	4%	11%	9%		
steel	23%	16%	3%	17%	12%	10%	20%	18%	15%		
copper	7%	2%	11%	3%	4%	8%	12%	14%	7%		
glass	8%		10%	13%	16%	7%	10%		11%		

Annex 3: Data on fire occurrence.

European data on domestic fire occurrence. One laptop per household is assumed, and therefore fire occurrence per household is used for the current study. Although exact data on the number of inhabitants per European country is available, the number of households could not be found for many countries. In addition, fire statistics are not available for each European country. Therefore only a limited list of countries is included in this overview.

Data correspond to the year 2004 (*Nibra 2009, Dol et al., 2010*).

	Fires per 1000 inhabitants	Population	Total fires in 2004	Number of households	Fires per household
The Netherlands	0.8	16.280.000	13025	7.091.000	0.0018
Slovakia	1.9	5.380.000	10227	2.072.000	0.0049
Hungary	2.1	10.110.000	21225	3.837.000	0.0055
Germany	2.2	82.520.000	181536	39.178.000	0.0046
Finland	2.2	5.230.000	11502	2.430.000	0.0047
Sweden	2.7	8.990.000	24283	4.441.000	0.0055
Austria	4.3	8.170.000	35139	3.475.000	0.010
Latvia	4.3	2.310.000	9945	905.000	0.011
Lithuania	4.7	3.440.000	16147	1.354.000	0.012
France	5.5	62.530.000	343929	25.876.000	0.013
UK ^a	7.4	59.870.000	443022	24.200.000	0.018
Estonia	8.9	1.350.000	12009	567.000	0.021
				average	0.0094
				median	0.0078

^a: a different report mentions that the occurrence of actual dwelling fires is much lower in the UK, with total of 57 000 for 2004 (*Department for Communities and Local Government, 2011*).

Annex 4: LCA scores.

The following Tables show the LCA results corresponding to figures in the main text. Scores are expressed in LCA-points according to the ReCiPe Endpoint method (1.03, Europe, H/A, with toxicity characterization factors added for the investigated FRs).

Annex 4.1: Environmental impact for the production of 1 kg flame retardant, cradle-to-gate. These results correspond to Figure 6.

	decaBDE, from mixed brine sources	BPS, from mixed brine sources	TBBPA, from mixed brine sources	ATO	RDP	BDP	DOPO	ZHS	ZS	MPP	Alpi	ATH
Total	0,69	0,35	0,49	1,50	0,72	0,71	0,83	1,94	2,40	0,44	0,80	0,08
Climate change Human Health	1,90E-01	9,99E-02	1,32E-01	8,95E-02	1,57E-01	1,55E-01	1,87E-01	3,15E-01	3,89E-01	1,10E-01	1,86E-01	1,83E-02
Ozone depletion	8,81E-05	4,94E-05	4,72E-05	8,41E-06	5,53E-05	4,64E-05	7,71E-05	8,48E-05	1,05E-04	2,72E-05	1,93E-05	4,11E-06
Human toxicity	1,19E-02	2,16E-03	5,73E-03	1,02E-01	7,00E-02	5,62E-02	8,50E-02	4,15E-02	5,12E-02	7,66E-03	6,25E-03	4,68E-03
Photochemical oxidant formation	1,02E-05	4,67E-06	8,67E-06	9,74E-05	1,50E-05	1,51E-05	1,46E-05	6,61E-05	8,15E-05	7,51E-06	1,66E-05	2,22E-06
Particulate matter formation	4,00E-02	1,08E-02	1,96E-02	5,16E-01	4,64E-02	3,80E-02	4,92E-02	6,46E-01	7,96E-01	4,99E-02	5,68E-02	1,72E-02
Ionising radiation	5,81E-04	8,85E-05	3,00E-04	5,36E-05	7,26E-04	5,59E-04	6,52E-04	1,91E-03	2,35E-03	2,10E-04	1,37E-03	3,87E-05
Climate change Ecosystems	1,25E-01	6,54E-02	8,65E-02	5,85E-02	1,03E-01	1,01E-01	1,23E-01	2,06E-01	2,55E-01	7,20E-02	1,22E-01	1,20E-02
Terrestrial acidification	3,28E-04	8,68E-05	1,49E-04	5,96E-03	3,21E-04	2,42E-04	3,27E-04	2,63E-03	3,24E-03	4,37E-04	2,28E-04	4,45E-05
Terrestrial ecotoxicity	2,84E-04	3,05E-05	7,77E-05	9,22E-04	3,61E-03	2,99E-03	7,78E-04	7,14E-04	8,80E-04	2,15E-04	8,25E-04	4,20E-05
Freshwater ecotoxicity	3,45E-06	1,49E-06	4,00E-06	8,81E-06	3,99E-04	3,33E-04	1,10E-04	1,53E-04	3,30E-04	2,05E-05	6,16E-05	1,68E-05
Marine ecotoxicity	2,45E-08	4,05E-09	1,39E-08	4,16E-06	9,98E-06	9,42E-06	8,58E-05	1,23E-04	2,74E-04	5,80E-06	4,14E-05	3,67E-06
Agricultural land occupation	1,32E-03	2,67E-04	1,14E-03	2,11E-03	2,80E-03	2,33E-03	2,90E-03	4,67E-03	5,76E-03	1,84E-03	3,72E-03	9,29E-05
Urban land occupation	8,13E-04	1,49E-04	5,61E-04	6,15E-02	2,59E-03	2,19E-03	3,02E-03	2,44E-02	3,01E-02	8,88E-03	2,69E-03	2,54E-04
Natural land transformation	6,74E-03	1,16E-03	1,65E-03	1,13E-01	3,69E-02	3,07E-02	4,74E-02	4,66E-02	5,75E-02	2,00E-02	1,24E-01	5,63E-04
Metal depletion	9,76E-05	2,15E-05	9,55E-05	4,57E-01	1,78E-04	1,58E-04	1,66E-04	2,89E-01	3,56E-01	1,84E-04	5,35E-05	2,27E-05
Fossil depletion	3,14E-01	1,71E-01	2,37E-01	8,82E-02	3,00E-01	3,18E-01	3,30E-01	3,64E-01	4,50E-01	1,68E-01	2,98E-01	2,27E-02

Annex 4.2: Environmental impact for the production of 1 kg flame retarded polymer, cradle-to-gate. Different loading rates required for different FRs are included in these calculations. These results correspond to Figure 8.

	PA6,6 +			EVA +			PPE/ HIPS +			PC/ ABS +					
	PA6,6	PA6,6 +	PA6,6 +	EVA no	decaBD	EVA +	PPE/ HIPS	decaBD	PPE/ HIPS +	PC/ ABS no	decaBD	PC/ ABS +	EPR no	EPR +	EPR +
	no FR	BPS + ATO	Alpi + MPP	FR	ATO	ZHS- ATH	no FR	ATO	RDP	FR	ATO	BDP	FR	TBBPA	DOPO
Total	0,65	0,67	0,67	0,34	0,44	0,51	0,57	0,65	0,60	0,64	0,70	0,66	0,77	0,72	0,80
Climate change Human Health	2,06E-01	1,81E-01	1,97E-01	7,06E-02	8,89E-02	9,40E-02	1,44E-01	1,50E-01	1,48E-01	1,94E-01	1,90E-01	1,89E-01	1,98E-01	1,87E-01	1,98E-01
Ozone depletion	1,60E-06	1,11E-05	7,07E-06	6,05E-06	1,71E-05	1,99E-05	1,09E-05	2,19E-05	1,78E-05	1,62E-06	1,08E-05	1,09E-05	1,63E-06	1,11E-05	2,09E-05
Human toxicity	1,32E-03	8,67E-03	2,79E-03	1,66E-03	8,31E-03	1,05E-02	7,40E-03	1,68E-02	1,69E-02	6,26E-03	1,38E-02	1,63E-02	9,67E-03	8,97E-03	2,86E-02
Photochemical oxidant formation	1,38E-05	1,83E-05	1,40E-05	6,23E-06	1,06E-05	1,53E-05	1,25E-05	1,75E-05	1,31E-05	1,38E-05	1,78E-05	1,43E-05	3,39E-05	2,91E-05	2,94E-05
Particulate matter formation	4,48E-02	7,25E-02	4,80E-02	1,44E-02	3,84E-02	1,29E-01	2,85E-02	6,01E-02	3,17E-02	5,05E-02	7,32E-02	4,87E-02	1,15E-01	9,67E-02	9,95E-02
Ionising radiation	7,29E-05	9,25E-05	3,19E-04	1,76E-04	2,36E-04	4,57E-04	3,54E-04	3,82E-04	4,20E-04	7,33E-05	1,34E-04	1,85E-04	7,33E-05	1,33E-04	2,36E-04
Climate change Ecosystems	1,35E-01	1,19E-01	1,29E-01	4,62E-02	5,82E-02	6,15E-02	9,44E-02	9,80E-02	9,67E-02	1,27E-01	1,24E-01	1,23E-01	1,30E-01	1,22E-01	1,30E-01
Terrestrial acidification	3,26E-04	6,84E-04	3,25E-04	1,03E-04	3,71E-04	5,32E-04	2,16E-04	5,82E-04	2,35E-04	2,78E-04	5,71E-04	2,76E-04	5,38E-04	4,66E-04	4,92E-04
Terrestrial ecotoxicity	3,81E-05	1,03E-04	1,96E-04	5,60E-05	1,23E-04	1,74E-04	7,68E-05	1,60E-04	6,09E-04	8,99E-05	1,53E-04	6,73E-04	5,85E-05	6,54E-05	2,89E-04
Freshwater ecotoxicity	4,77E-06	7,97E-06	1,60E-05	1,69E-06	4,22E-06	3,82E-05	5,66E-06	8,53E-06	6,49E-05	7,88E-06	9,97E-06	7,32E-05	4,22E-06	4,39E-06	3,27E-05
Marine ecotoxicity	8,37E-09	3,23E-06	7,73E-06	6,43E-09	1,85E-06	2,50E-05	1,75E-08	2,78E-06	1,60E-06	7,47E-09	2,31E-06	2,02E-06	1,06E-08	1,97E-08	2,34E-05
Agricultural land occupation	2,52E-03	2,70E-03	3,28E-03	3,34E-03	3,44E-03	3,59E-03	3,60E-03	3,68E-03	3,85E-03	2,71E-03	2,91E-03	3,13E-03	2,53E-03	2,75E-03	3,24E-03
Urban land occupation	1,90E-04	4,51E-03	1,37E-03	4,65E-04	2,98E-03	4,66E-03	1,22E-03	4,81E-03	1,45E-03	1,96E-04	3,35E-03	6,28E-04	2,11E-04	3,14E-04	9,55E-04
Natural land transformation	3,44E-04	8,48E-03	2,28E-02	1,11E-03	6,39E-03	9,03E-03	1,13E-03	8,71E-03	6,56E-03	3,54E-04	6,69E-03	6,49E-03	3,30E-04	6,66E-04	1,22E-02
Metal depletion	1,44E-05	3,20E-02	3,80E-05	6,85E-05	1,84E-02	5,06E-02	1,07E-04	2,75E-02	1,20E-04	1,84E-05	2,29E-02	4,88E-05	1,57E-05	3,41E-05	5,63E-05
Fossil depletion	2,62E-01	2,37E-01	2,63E-01	1,99E-01	2,12E-01	1,45E-01	2,89E-01	2,83E-01	2,93E-01	2,58E-01	2,57E-01	2,73E-01	3,17E-01	3,04E-01	3,24E-01

Annex 4.3: Environmental impact for accidental combustion of a laptop with BFRs and a laptop with HFFRs. The fraction of laptops actually ending up in an accidental fire is not incorporated in these results. Scores are expressed in LCA-points according to the ReCiPe method. These results correspond to Figure 11.

	accidental fire, BFR scenario	accidental fire, HFFR scenario
Total	1.18	0.94
Climate change Human Health	1.88E-01	2.34E-01
Ozone depletion	7.88E-06	7.88E-06
Human toxicity	1.17E-01	1.03E-01
Photochemical oxidant formation	4.58E-05	4.47E-05
Particulate matter formation	4.53E-01	3.99E-01
Ionising radiation	9.33E-05	9.33E-05
Climate change Ecosystems	1.23E-01	1.53E-01
Terrestrial acidification	5.06E-05	5.06E-05
Terrestrial ecotoxicity	2.55E-01	4.69E-03
Freshwater ecotoxicity	1.01E-03	2.25E-05
Marine ecotoxicity	3.44E-03	5.21E-05
Agricultural land occupation	3.35E-04	3.35E-04
Urban land occupation	3.52E-04	3.52E-04
Natural land transformation	1.12E-03	1.12E-03
Metal depletion	2.58E-05	2.58E-05
Fossil depletion	4.22E-02	4.22E-02

Annex 4.4: Environmental impact for the End-of-Life treatment of 1 laptop. A laptop with BFRs and a laptop with HFFRs are compared for the 4 End-of-Life options. The fraction of laptops actually ending up in the different EoL scenarios is not incorporated in these results. Scores are expressed in LCA-points according to the ReCiPe method. These results correspond to Figure 13.

	WEEE-compliant treatment, BFR	Incineration in MSWI, BFR	laptop to landfill, BFR	WEEE export, improper treatment, BFR	WEEE-compliant treatment, HFFR	Incineration in MSWI, HFFR	laptop to landfill, HFFR	WEEE export, improper treatment, HFFR
Total	0.27	0.51	0.04	4.15	0.10	0.46	0.03	1.84
Climate change Human Health	1.08E-01	2.59E-01	4.78E-03	2.72E-01	2.76E-02	2.59E-01	4.78E-03	2.72E-01
Ozone depletion	1.17E-05	1.93E-06	5.08E-07	4.44E-06	4.18E-06	1.93E-06	5.08E-07	4.44E-06
Human toxicity	4.66E-03	1.46E-02	1.17E-02	3.32E+00	6.76E-04	1.46E-02	1.17E-02	1.05E+00
Photochemical oxidant formation	4.28E-06	3.41E-06	3.45E-07	4.79E-05	5.56E-06	3.41E-06	3.45E-07	4.79E-05
Particulate matter formation	1.29E-02	5.28E-03	5.79E-04	2.93E-01	1.60E-02	5.28E-03	5.79E-04	2.93E-01
Ionising radiation	2.57E-04	9.69E-06	2.45E-06	3.51E-05	1.58E-04	9.69E-06	2.45E-06	3.51E-05
Climate change Ecosystems	7.09E-02	1.70E-01	3.13E-03	1.78E-01	1.81E-02	1.70E-01	3.13E-03	1.78E-01
Terrestrial acidification	9.70E-05	3.25E-05	3.13E-06	1.96E-04	1.30E-04	3.25E-05	3.13E-06	1.96E-04
Terrestrial ecotoxicity	1.32E-04	3.11E-02	2.08E-05	4.65E-02	7.80E-05	6.27E-05	2.08E-05	1.58E-02
Freshwater ecotoxicity	1.15E-05	1.07E-02	1.03E-02	1.08E-02	7.32E-07	1.11E-03	1.47E-03	1.23E-03
Marine ecotoxicity	1.03E-06	5.96E-03	8.89E-03	5.99E-03	6.66E-07	7.98E-04	1.23E-03	8.21E-04
Agricultural land occupation	6.87E-04	3.98E-05	3.11E-05	8.18E-05	3.02E-04	3.98E-05	3.11E-05	8.18E-05
Urban land occupation	5.66E-04	1.30E-04	5.81E-04	2.26E-04	2.11E-04	1.30E-04	5.81E-04	2.26E-04
Natural land transformation	1.70E-03	2.12E-04	-4.68E-04	9.95E-04	8.76E-04	2.12E-04	-4.68E-04	9.95E-04
Metal depletion	3.73E-05	4.59E-06	1.29E-06	8.52E-06	8.27E-06	4.59E-06	1.29E-06	8.52E-06
Fossil depletion	7.19E-02	9.35E-03	2.34E-03	2.63E-02	3.30E-02	9.35E-03	2.34E-03	2.63E-02

Annex 4.5a: Environmental impact for each phase of the life cycle of one laptop containing BFRs. Scores are expressed in LCA-points according to the ReCiPe method. These results correspond to Figure 14.

	Total	Batteries	LCD module	PWBs	other components	packaging	electricity, laptop production	transport production phase	EPR+BF R	EVA+BF R	PPE/HIP S+BFR	PA6,6+B FR	PC/ABS +BFR
Total	32.5	2.16	2.47	9.58	4.89	0.16	0.08	0.09	0.21	0.07	0.34	0.08	0.45
Climate change Human Health	10.79	7.83E-01	9.44E-01	2.87E+00	2.14E+00	3.66E-02	2.45E-02	2.19E-02	5.48E-02	1.41E-02	7.72E-02	2.07E-02	1.24E-01
Ozone depletion	0.01	1.00E-02	6.04E-05	5.70E-04	9.33E-05	5.10E-06	2.14E-06	4.73E-06	3.24E-06	2.72E-06	1.13E-05	1.26E-06	7.06E-06
Human toxicity	1.45	1.14E-01	6.55E-02	3.26E-01	1.30E-01	1.44E-03	7.40E-04	4.47E-04	2.63E-03	1.32E-03	8.67E-03	9.88E-04	9.00E-03
Photochemical oxidant formation	0.00	4.53E-05	5.90E-05	2.92E-04	6.21E-05	3.98E-06	1.43E-06	9.10E-06	8.52E-06	1.68E-06	9.03E-06	2.08E-06	1.16E-05
Particulate matter formation	2.87	1.63E-01	2.00E-01	1.09E+00	3.11E-01	7.40E-03	5.89E-03	2.46E-02	2.83E-02	6.10E-03	3.10E-02	8.26E-03	4.77E-02
Ionising radiation	0.06	8.57E-04	1.88E-03	1.59E-02	3.78E-03	4.78E-05	2.27E-04	3.66E-05	3.90E-05	3.75E-05	1.97E-04	1.05E-05	8.72E-05
Climate change Ecosystems	7.06	5.13E-01	6.18E-01	1.88E+00	1.40E+00	2.39E-02	1.61E-02	1.44E-02	3.59E-02	9.25E-03	5.05E-02	1.35E-02	8.10E-02
Terrestrial acidification	0.02	1.34E-03	1.53E-03	9.05E-03	1.66E-03	5.38E-05	4.89E-05	2.01E-04	1.36E-04	5.91E-05	3.00E-04	7.80E-05	3.72E-04
Terrestrial ecotoxicity	0.03	2.93E-03	1.06E-03	7.96E-03	1.99E-03	6.56E-05	1.95E-05	2.15E-05	1.92E-05	1.96E-05	8.25E-05	1.17E-05	9.99E-05
Freshwater ecotoxicity	0.01	5.18E-04	1.02E-04	2.67E-04	2.41E-04	2.19E-06	5.13E-07	3.82E-07	1.28E-06	6.71E-07	4.40E-06	9.08E-07	6.49E-06
Marine ecotoxicity	0.00	7.64E-07	2.28E-07	1.52E-06	8.17E-07	4.30E-09	2.06E-09	4.86E-09	5.77E-09	2.94E-07	1.43E-06	3.68E-07	1.50E-06
Agricultural land occupation	0.22	7.85E-03	2.13E-02	4.16E-02	7.12E-02	2.88E-02	2.93E-04	6.87E-05	8.04E-04	5.48E-04	1.90E-03	3.08E-04	1.89E-03
Urban land occupation	0.35	4.81E-03	2.42E-02	2.62E-01	2.85E-02	9.06E-04	1.29E-04	2.10E-04	9.21E-05	4.74E-04	2.48E-03	5.15E-04	2.18E-03
Natural land transformation	0.20	1.13E-02	2.09E-02	8.78E-02	2.00E-02	3.68E-03	2.98E-04	1.23E-03	1.95E-04	1.02E-03	4.50E-03	9.67E-04	4.35E-03
Metal depletion	0.10	1.77E-03	4.05E-03	4.47E-02	1.13E-02	1.87E-05	5.22E-06	8.70E-06	9.98E-06	2.92E-03	1.42E-02	3.65E-03	1.49E-02
Fossil depletion	9.34	5.43E-01	5.65E-01	2.95E+00	7.64E-01	5.98E-02	2.70E-02	2.83E-02	8.90E-02	3.36E-02	1.46E-01	2.70E-02	1.68E-01

Annex 4.5b: Environmental impact for each phase of the life cycle of one laptop containing BFRs. Scores are expressed in LCA-points according to the ReCiPe method. These results correspond to Figure 14.

	Electricity, use phase laptop	volatilization BFRs from laptop	accidental fire	WEEE treatment, WEEE- compliant	WEEE treatment, to MSWI	WEEE treatment, to landfill	WEEE export, improper treatment
Total	11.0	0.0042	1.11E-02	0.12	0.061	0.010	0.79
Climate change Human Health	3.54E+00	0.00E+00	1.77E-03	4.76E-02	3.11E-02	1.20E-03	5.17E-02
Ozone depletion	3.09E-04	0.00E+00	7.41E-08	5.16E-06	2.32E-07	1.27E-07	8.43E-07
Human toxicity	1.47E-01	4.24E-03	1.10E-03	2.05E-03	1.75E-03	2.93E-03	6.30E-01
Photochemical oxidant formation	2.10E-04	0.00E+00	4.30E-07	1.88E-06	4.09E-07	8.64E-08	9.11E-06
Particulate matter formation	8.81E-01	0.00E+00	4.26E-03	5.69E-03	6.34E-04	1.45E-04	5.56E-02
Ionising radiation	3.26E-02	0.00E+00	8.77E-07	1.13E-04	1.16E-06	6.13E-07	6.67E-06
Climate change Ecosystems	2.32E+00	0.00E+00	1.16E-03	3.12E-02	2.04E-02	7.82E-04	3.39E-02
Terrestrial acidification	7.24E-03	0.00E+00	4.76E-07	4.27E-05	3.90E-06	7.83E-07	3.72E-05
Terrestrial ecotoxicity	3.79E-03	3.37E-26	2.39E-03	5.81E-05	3.73E-03	5.20E-06	8.84E-03
Freshwater ecotoxicity	7.92E-05	1.08E-06	9.47E-06	5.05E-06	1.28E-03	2.59E-03	2.06E-03
Marine ecotoxicity	3.73E-07	9.28E-07	3.23E-05	4.53E-07	7.16E-04	2.22E-03	1.14E-03
Agricultural land occupation	4.72E-02	0.00E+00	3.15E-06	3.02E-04	4.78E-06	7.78E-06	1.55E-05
Urban land occupation	2.28E-02	0.00E+00	3.31E-06	2.49E-04	1.56E-05	1.45E-04	4.30E-05
Natural land transformation	4.45E-02	0.00E+00	1.05E-05	7.49E-04	2.54E-05	-1.17E-04	1.89E-04
Metal depletion	2.70E-03	0.00E+00	2.43E-07	1.64E-05	5.50E-07	3.24E-07	1.62E-06
Fossil depletion	3.90E+00	0.00E+00	3.96E-04	3.17E-02	1.12E-03	5.85E-04	5.00E-03

Annex 4.6a: Environmental impact for each phase of the life cycle of one laptop containing HFFRs. Scores are expressed in LCA-points according to the ReCiPe method. These results correspond to Figure 16.

	Total	Batteries	LCD module	PWBs	other components	Packaging	electricity, laptop production	transport production phase	EPR+HF FR	EVA+HF FR	PPE/HIP S+HFFR	PA6,6+H FFR	PC/ABS +HFFR
Total	32.0	2.16	2.47	9.58	4.89	0.16	0.075	0.091	0.23	0.081	0.31	0.076	0.43
Climate change Human Health	1.08E+01	7.83E-01	9.44E-01	2.87E+00	2.14E+00	3.66E-02	2.45E-02	2.19E-02	5.81E-02	1.50E-02	7.63E-02	2.25E-02	1.23E-01
Ozone depletion	1.11E-02	1.00E-02	6.04E-05	5.70E-04	9.33E-05	5.10E-06	2.14E-06	4.73E-06	6.12E-06	3.17E-06	9.20E-06	8.06E-07	7.09E-06
Human toxicity	1.02E+00	1.14E-01	6.55E-02	3.26E-01	1.30E-01	1.44E-03	7.40E-04	4.47E-04	8.38E-03	1.68E-03	8.70E-03	3.18E-04	1.06E-02
Photochemical oxidant formation	7.24E-04	4.53E-05	5.90E-05	2.92E-04	6.21E-05	3.98E-06	1.43E-06	9.10E-06	8.60E-06	2.43E-06	6.74E-06	1.60E-06	9.29E-06
Particulate matter formation	2.85E+00	1.63E-01	2.00E-01	1.09E+00	3.11E-01	7.40E-03	5.89E-03	2.46E-02	2.92E-02	2.05E-02	1.64E-02	5.48E-03	3.17E-02
Ionising radiation	5.58E-02	8.57E-04	1.88E-03	1.59E-02	3.78E-03	4.78E-05	2.27E-04	3.66E-05	6.92E-05	7.27E-05	2.17E-04	3.64E-05	1.20E-04
Climate change Ecosystems	7.04E+00	5.13E-01	6.18E-01	1.88E+00	1.40E+00	2.39E-02	1.61E-02	1.44E-02	3.80E-02	9.79E-03	4.99E-02	1.47E-02	8.03E-02
Terrestrial acidification	2.18E-02	1.34E-03	1.53E-03	9.05E-03	1.66E-03	5.38E-05	4.89E-05	2.01E-04	1.44E-04	8.46E-05	1.21E-04	3.71E-05	1.80E-04
Terrestrial ecotoxicity	2.18E-02	2.93E-03	1.06E-03	7.96E-03	1.99E-03	6.56E-05	1.95E-05	2.15E-05	8.47E-05	2.77E-05	3.14E-04	2.23E-05	4.38E-04
Freshwater ecotoxicity	2.04E-03	5.18E-04	1.02E-04	2.67E-04	2.41E-04	2.19E-06	5.13E-07	3.82E-07	9.58E-06	6.07E-06	3.35E-05	1.83E-06	4.76E-05
Marine ecotoxicity	5.79E-04	7.64E-07	2.28E-07	1.52E-06	8.17E-07	4.30E-09	2.06E-09	4.86E-09	6.84E-06	3.98E-06	8.25E-07	8.81E-07	1.31E-06
Agricultural land occupation	2.24E-01	7.85E-03	2.13E-02	4.16E-02	7.12E-02	2.88E-02	2.93E-04	6.87E-05	9.49E-04	5.72E-04	1.99E-03	3.74E-04	2.04E-03
Urban land occupation	3.47E-01	4.81E-03	2.42E-02	2.62E-01	2.85E-02	9.06E-04	1.29E-04	2.10E-04	2.80E-04	7.41E-04	7.51E-04	1.56E-04	4.09E-04
Natural land transformation	2.05E-01	1.13E-02	2.09E-02	8.78E-02	2.00E-02	3.68E-03	2.98E-04	1.23E-03	3.57E-03	1.44E-03	3.38E-03	2.60E-03	4.22E-03
Metal depletion	7.27E-02	1.77E-03	4.05E-03	4.47E-02	1.13E-02	1.87E-05	5.22E-06	8.70E-06	1.65E-05	8.05E-03	6.18E-05	4.33E-06	3.17E-05
Fossil depletion	9.34E+00	5.43E-01	5.65E-01	2.95E+00	7.64E-01	5.98E-02	2.70E-02	2.83E-02	9.48E-02	2.30E-02	1.51E-01	3.00E-02	1.78E-01

Annex 4.6b: Environmental impact for each phase of the life cycle of one laptop containing HFFRs. Scores are expressed in LCA-points according to the ReCiPe method. These results correspond to Figure 16.

	Electricity, use phase laptop	volatilization HFFRs from laptop	accidental fire	WEEE treatment, WEEE- compliant	WEEE treatment, to MSWI	WEEE treatment, to landfill	WEEE export, improper treatment
Total	11.0	2.47E-05	8.82E-03	0.043	0.055	0.006	0.35
Climate change Human Health	3.54E+00	0.00E+00	2.20E-03	1.21E-02	3.11E-02	1.20E-03	5.17E-02
Ozone depletion	3.09E-04	0.00E+00	7.41E-08	1.84E-06	2.32E-07	1.27E-07	8.43E-07
Human toxicity	1.47E-01	1.15E-08	9.70E-04	2.98E-04	1.75E-03	2.93E-03	2.00E-01
Photochemical oxidant formation	2.10E-04	0.00E+00	4.20E-07	2.45E-06	4.09E-07	8.64E-08	9.11E-06
Particulate matter formation	8.81E-01	0.00E+00	3.75E-03	7.04E-03	6.34E-04	1.45E-04	5.56E-02
Ionising radiation	3.26E-02	0.00E+00	8.77E-07	6.94E-05	1.16E-06	6.13E-07	6.67E-06
Climate change Ecosystems	2.32E+00	0.00E+00	1.44E-03	7.95E-03	2.04E-02	7.82E-04	3.39E-02
Terrestrial acidification	7.24E-03	0.00E+00	4.76E-07	5.74E-05	3.90E-06	7.83E-07	3.72E-05
Terrestrial ecotoxicity	3.79E-03	2.28E-05	4.41E-05	3.43E-05	7.53E-06	5.20E-06	3.00E-03
Freshwater ecotoxicity	7.92E-05	8.54E-07	2.11E-07	3.22E-07	1.33E-04	3.67E-04	2.34E-04
Marine ecotoxicity	3.73E-07	1.05E-06	4.89E-07	2.93E-07	9.57E-05	3.08E-04	1.56E-04
Agricultural land occupation	4.72E-02	0.00E+00	3.15E-06	1.33E-04	4.78E-06	7.78E-06	1.55E-05
Urban land occupation	2.28E-02	0.00E+00	3.31E-06	9.27E-05	1.56E-05	1.45E-04	4.30E-05
Natural land transformation	4.45E-02	0.00E+00	1.05E-05	3.85E-04	2.54E-05	-1.17E-04	1.89E-04
Metal depletion	2.70E-03	0.00E+00	2.43E-07	3.64E-06	5.50E-07	3.24E-07	1.62E-06
Fossil depletion	3.90E+00	0.00E+00	3.96E-04	1.45E-02	1.12E-03	5.85E-04	5.00E-03

Annex 5: LCA review report.

LCA critical review report ENFIRO

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Preface

This report presents the result of the critical review of the Life cycle assessment (LCA) performed within the European funded project ENFIRO. In the LCA, calculations were made for a laptop with different flame retardant systems. This critical review report consists of three parts; review of Scope and Goal definition, review of data collection (LCI) and a Final Assessment review of life cycle impact assessment (LCIA) and interpretation of the LCA.

Summary

This report presents the results of a critical review of the LCA carried out in the European Commission funded project ENFIRO. The review has been performed by LCA experts at Swerea IVF. The review has been made against the standard ISO 14044 which ensures quality and transparency of the LCA.

This critical review report consists of three parts; review of Scope and Goal definition performed in spring 2010, review of data collection (LCI) performed in June 2012, and a Final Assessment review of life cycle impact assessment (LCIA) and interpretation of the LCA performed in September to December 2012.

The LCA was found to be performed according to ISO 14044.

Goal and scope review

In this study the goal was to examine the environmental impact for new alternative flame retardants (FRs) for electronic equipment in comparison to the existing halogenated (brominated) flame retardants (BFRs). As one part toxicity parameters have been studied and have been stated to be of high importance.

The goal and scope was reviewed in May 2010 at the scoping stage of the study. Comments and suggestions referred to: choice of functional unit (1 kg instead of 1000 kg was discussed on material level and then a laptop in use for five years), whether or not to include the use stage as such and fires in the use stage (use phase was included as well as fire), separation of product systems, the replacement product when calculating avoided burdens and description of category indicators. All comments and suggestions were treated satisfactorily in a revised goal and scope definition document.

The main aim of the ENFIRO project is to find FRs which are more environmentally benign than the environmentally harmful BFRs. Main questions for the current LCA comparison are therefore: does the overall ecotoxicological and human toxicological profile over the whole life cycle improve by this substitution, and are there other environmental impacts that are negatively affected by this substitution (is there a shift of burdens)?

From this scope, it follows that the impact categories freshwater, marine and terrestrial ecotoxicity and human toxicity are very relevant in addition to climate change. Therefore, by advice from the reviewers, these scores are shown in separate figures in the result section of the main report.

Relevant environmental impact categories and characterisation factors

According to discussions within the project and with the LCA reviewer some environmental impact categories and characterisation factors have been pointed out as more relevant and further studied and described. In the following list the environmental impact categories chosen for the study are presented.

Environmental impact categories

Climate change - yes

Ozone - no

Energy - no

Eutrophication - no

Acidification - no

Terrestrial ecotoxicity - yes

Freshwater ecotoxicity - yes

Marine ecotoxicity - yes

Human toxicology

Human toxicology - yes

Data collection review

For the data collection review, performed in June 2012, some major findings or discussions points will be described in this report. Apart from these findings some text/spelling corrections have been proposed and some other minor details have been raised. All comments and suggestions were treated satisfactorily in a revised data collection document.

Input

The ENFIRO project has focused upon environmental issues of alternative non-halogenated flame retardants in comparison to halogenated flame retardants, with special attention to toxicity. The inputs in this LCA study are mainly from databases but as much as possible specific data, especially on toxicity, have been received from the rest of the project team of ENFIRO. Furthermore some of the data has actually been produced within the project by tests and experiments conducted, giving a strength to the LCA.

The input on specific chemical characteristic data and toxicity data has been included in the USES-LCA method. This method of including data is today somewhat beyond state of the art and is mainly performed by frontrunners in the area. Where data gaps are present it is affecting the overall analysis due to difficulties of handling this in the USES-LCA method. This can be compared to the IPPC-Tegewa chemical risk assessment model which can/should score high for data gaps (dealing with waste water streams). Nevertheless, it is a large improvement to be able to perform the assessment with the input that has been able to be achieved and used in the model.

A lot of effort has been put from both the reviewer and the LCA practitioner to ensure the quality of the data concerning the production routes for the different FRs.

Final Assessment review

During the assessment review, performed in September 2012, some parts were discussed thoroughly. In this assessment part of the LCA all discussion points brought up in the review were treated satisfactorily in the final project report.

Discussions of Waste scenario

In the assessment the assumption have been made that halogenated free flame retarded (HFFR) plastics are more likely to be recycled compared to halogenated flame retarded plastics. This assumption has been discussed in detail and the conclusion has been that this is true as long as the additives/flame retardants used in the plastics are non-toxic. The more recent addition to the End of waste directive will affect possibilities for recycling of plastic in relation to hazardous content.

Discussions about presentation of results

According to the goal and scope the following presentation of results have been discussed and agreed upon:

Main impact categories, apart from toxicity, that are expected to be affected by the substitution are Climate Change, Particulate matter formation and Resource Depletion, as the HFFRs may have different energy consumption during production. In most cases, these three energy related impacts will be proportional to each other, and therefore it suffices to show only one of these categories Climate change was chosen as the representative for energy consumption.

The comparison of scenarios is done using both the separate toxicity and climate change results and the overall Single Score results with the ReCiPe method. This comparison method is in line with the scope of the study.

According to the standard (ISO 14044) weighting (as in the case with the Single Score) is an optional element but shall not be used in LCA studies intended to be used in comparative assertions intended to be disclosed to the public. In this context one question is if these products are commercial? The alternative flame retardants can be considered as raw material that can be purchased from many different suppliers. Therefore it was decided that the results could, in addition to being presented in non weighted impact categories (climate change and toxicity), also be presented as weighted results.

Sensitivity analysis

Production of FR

When discussing the modelling of the production phase there were several questions raised such as if the production route chosen is the most relevant and which processes within this route are most relevant. Another way of modelling the production phase may lead to both increase as well as decrease in the environmental impact. A sensitivity analysis was therefore discussed for production. The question was if the production becomes more or less important in the overall assessment if it would be modelled differently. For the impact of production of solely the FRs, the difference may be significant but for polymers+FR, the difference would be less. In figures 14-17 in the LCA report, the impacts of production of polymers+FRs are relatively small, and a double increase of the FR alone would not change the picture much.

In addition the fact that 100 % yield was assumed for all chemical processes was discussed. The same assumption was made for all FR and this systematic assumption would not give different result compared to if a 50 % yield was to be assumed. The yield for different production routes and therefore FR may however differ. This is not included in the LCA report, but was discussed between the reviewers and the practitioner.

Service Life of product

A sensitivity scenario has been discussed for the service life of the final product using different FR-polymer systems. This sensitivity analysis was agreed not to be performed more than an addition of explanation of what happens if the service life is shortened by one year out of the five. From a technical perspective the service life has been assessed and the result was that five years seem adequate.

Final report

The final review was performed during November and December 2012. In the final report all parts brought up in the review were treated satisfactorily.

Final results

Even though the flame retardants scores low in the overall perspective it is important to keep in mind that the weight percentage is very low for these compounds in the whole product (laptop), and still it is possible to see some impact. However, in order to see hot spots for different phases and parts of the LCA, different zooming of the study is presented.

Some studies benchmark the results from the LCA by e.g. calculating the total avoided (saved) climate impact (CO₂) in Europe from the use of an improved product, e.g. a laptop. In other situations the goal of 1000 kg CO₂ per inhabitant is used to benchmark a potential improvement due to for instance a different material or process. The reviewer and the LCA practitioner came to consensus not to include such a benchmark in the final report since climate change is not the focus of the study, or of ENFIRO.

Conclusion

The review has been made against ISO 14044 which ensures quality and transparency of the LCA.

The overall conclusions presented in the LCA report have been regarded to be correct and relevant for the study.

The LCA was found to be performed according to ISO 14044.