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Life cycle assessment of the treatment and recycling of refrigeration equipment containing CFCs and hydrocarbons

Final report

IMPORTANT NOTE

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Commissioned by
RAL Gütegemeinschaft Rückproduktion von FCKW-haltigen Kühlgeräten e.V.
[RAL Quality Assurance Association for the Demanufacture of Refrigeration Equipment]

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Life cycle assessment of the treatment and recycling of refrigeration equipment containing CFCs and hydrocarbons

Final report

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AltautoV	German Ordinance Governing the Transfer, Collection and Environmentally Sound Disposal of End-of-Life Vehicles (*Verordnung über die Überlassung, Rücknahme und umweltverträgliche Entsorgung von Altfahrzeugen*)
CED	cumulative energy demand
CFC	chlorofluorocarbon
eq	equivalent
EU	European Union
GEMIS	Global Emission Model for Integrated Systems
HC	hydrocarbon
HCFC	hydrochlorofluorocarbon
HFC	hydrofluorocarbon
H-T	high-temperature
IPCC	Intergovernmental Panel on Climate Change
ISO	International Standards Organization
KrW-/AbfG	German Waste Avoidance, Recycling and Disposal Act (*Kreislaufwirtschafts- und Abfallgesetz*)
NMVOC	non-methane volatile organic compound
ODP	ozone depletion potential
PM_{10}	particulate matter
POCP	photochemical ozone creation potential
PS	polystyrene
PU	polyurethane
PVC	polyvinylchloride
R11	trichlorofluoromethane
R12	dichlorodifluoromethane
RAL	German Institute for Quality Assurance and Certification (*Deutsches Institut für Gütesicherung und Kennzeichnung e.V.*)
SHF	shredder heavy fraction

SLF	shredder light fraction
UV-B	ultraviolet B radiation
VOC	volatile organic compound
WHO	World Health Organization

1 Summary

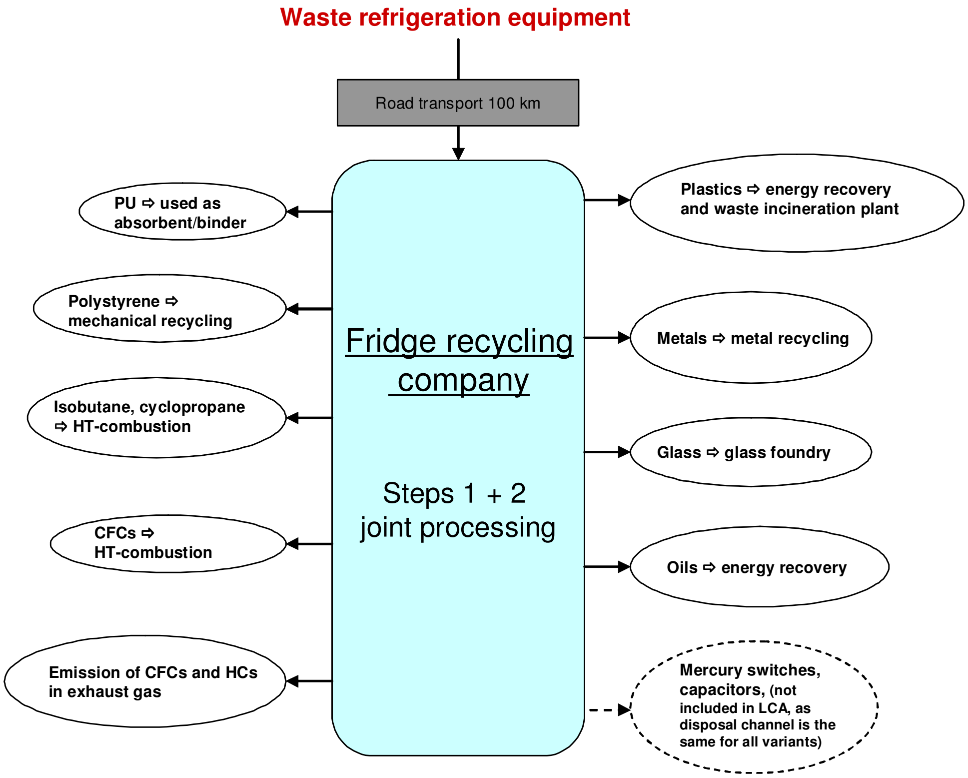
1.1 Background and subject of study

This life cycle assessment (LCA) was commissioned by the RAL Quality Assurance Association for the Demanufacture of Refrigeration Equipment. Its objective is to make an ecological comparison of different disposal channels for waste domestic refrigeration appliances containing CFCs and hydrocarbons. The disposal channels considered include those that could arise from potential changes to the European WEEE Directive. The study was carried out in accordance with the ISO 14040 and 14044 standards. It also includes a critical review by Mr Giegrich of the Institute for Energy and Environmental Research (*Institut für Energie- und Umweltforschung*) in Heidelberg. Assistance was also provided by Dr Keri of the Austrian Ministry of Agriculture, Forestry, Environment and Water Management (*Bundesministerium für Land- und Forstwirtschaft, Umwelt und Wasserwirtschaft*), Mr Schmit of the Luxembourg State Environmental Agency (*Umweltamt*) and Mr Hornberger and Ms Janusz-Renault of the Fraunhofer Institute for Manufacturing Engineering and Automation (*Fraunhofer-Institut für Produktionstechnik und Automatisierung*).

1.2 Description of processing variants

1.2.1 Variant 1: Joint processing

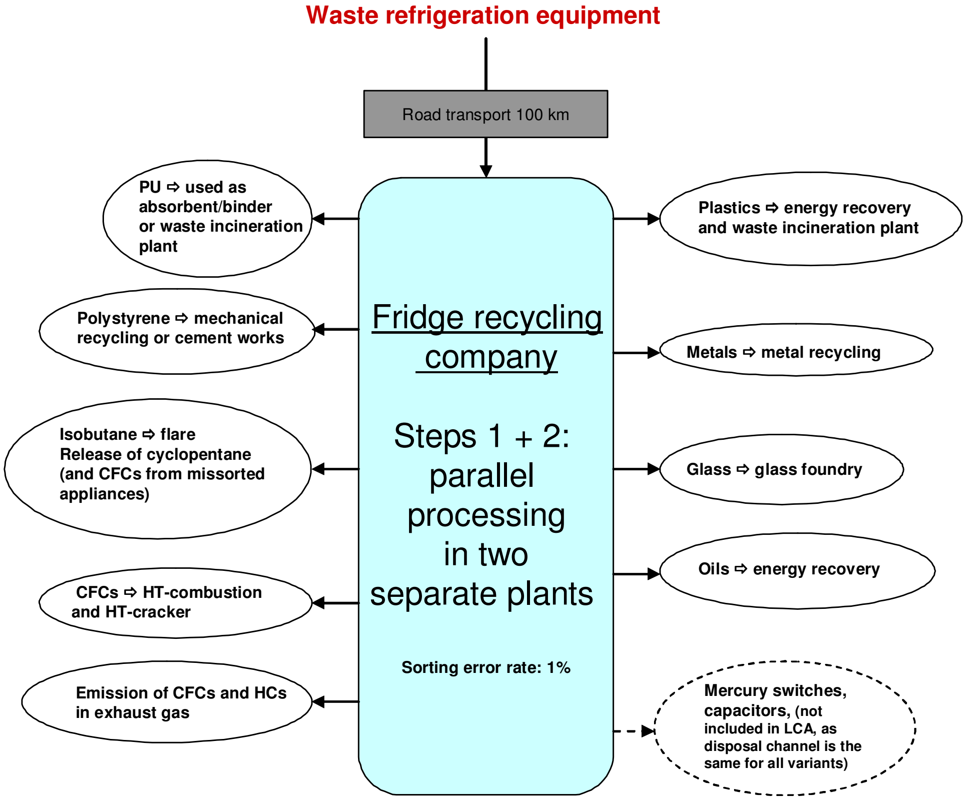
The material flows associated with joint processing are shown in the following figure:



In variant 1, the waste refrigeration appliances are all treated at the premises of the fridge recycling company. This involves the simultaneous treatment of waste appliances containing hydrocarbons (HCs) and those containing chlorofluorocarbons (CFCs) in a single plant. The resulting polyurethane powder is reused as an absorbent (chemical and oil binder). The highly purified polystyrene fraction is mechanically recycled. Metals are sent for metal recycling. Less pure plastic fractions go to waste incinerator plants or for combsution in cement works.

1.2.2 Variant 2: Parallel processing

The material flows in parallel processing are shown in the following figure:

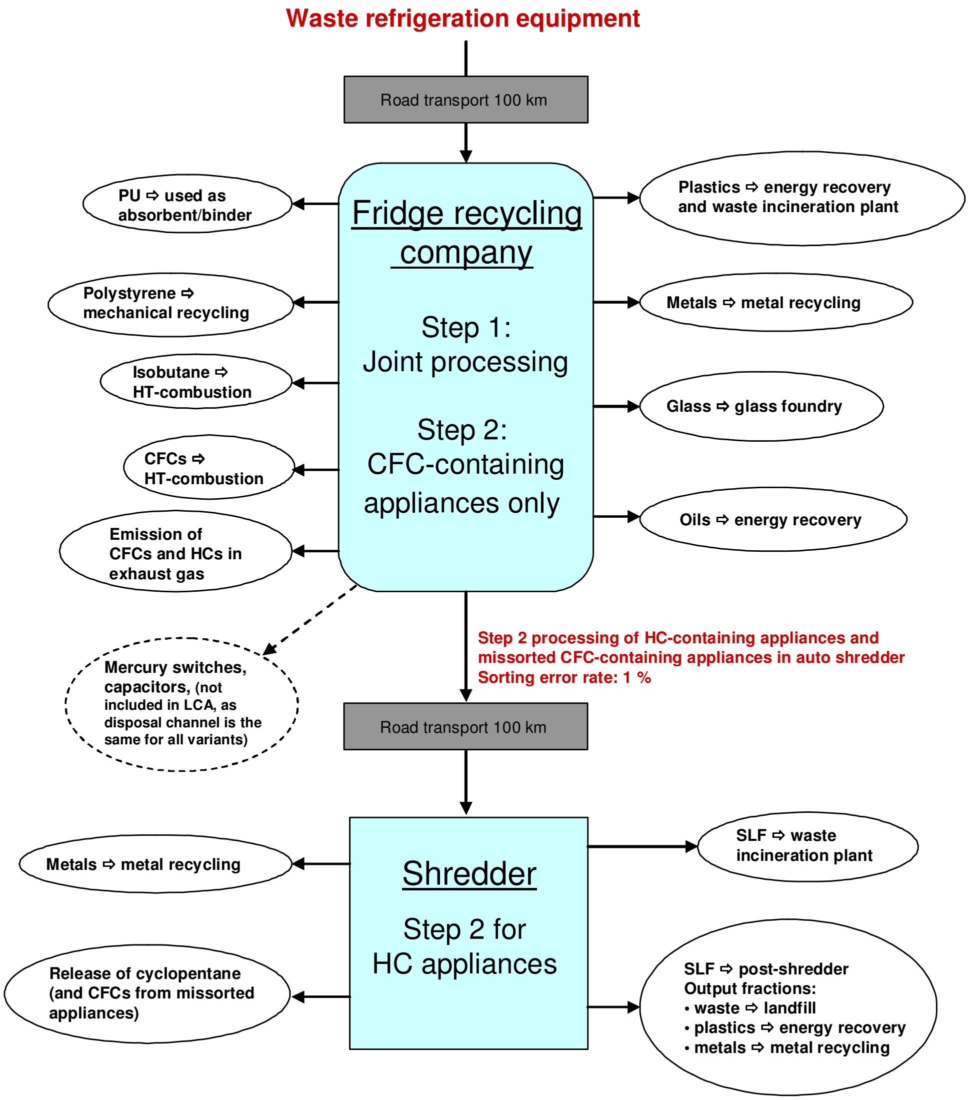


In this variant, the waste refrigeration appliances are all treated at the premises of the fridge recycling company. The HC-containing and CFC-containing appliances are processed in separate plants (no batch operation). The CFC line operates essentially as in variant 1 (joint processing).

The computational model of the material and energy flows assumes that the cyclopentane outgassed and collected during the crushing or shredding process (30 % of the total quantity in the foam) is subsequently released. The polyurethane flakes and chunks are incinerated. The model also assumes that 1 % of CFC-containing appliances are missorted and thus processed together with the hydrocarbon units.

1.2.3 Variant 3: Step 2 processing of HC-containing appliances in an auto shredder

The material flows in variant 3 are depicted below:

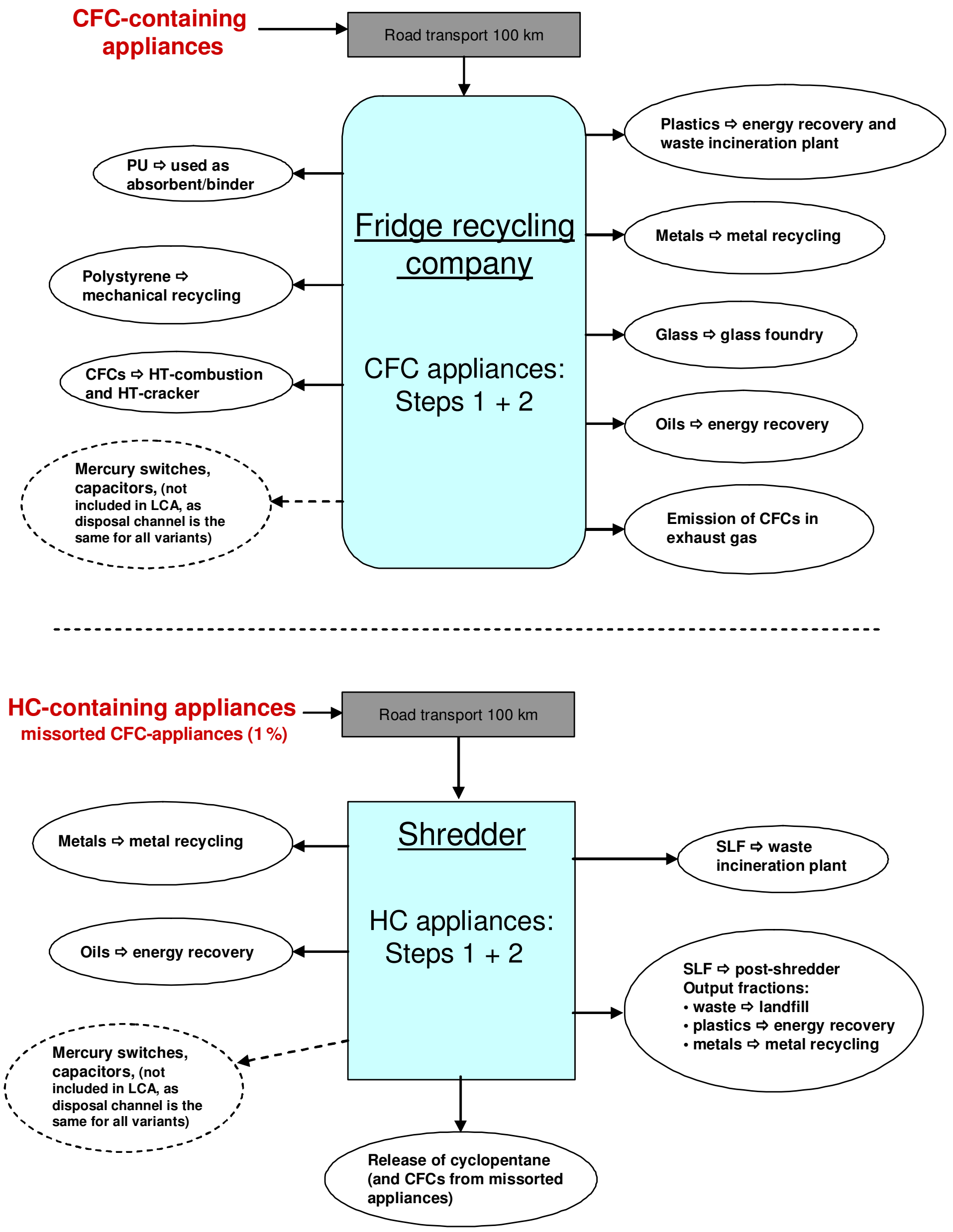


Variant 3 assumes that step 1 processing is carried out at the premises of the fridge recycling company with HC-containing and CFC-containing appliances being processed jointly to a high environmental standard. Apart from those incorrectly sorted appliances, the CFC-containing units are then treated at the premises of the fridge recycling company as in variant 1.

The HC-containing appliances and the missorted CFC-containing appliances (assumed sorting error: 1 %) are transported to an auto shredder facility for disposal. Fine shredding of the appliances results in the release of 70 % of the cyclopentane, or 70 % of the R11 in the case of the missorted CFC-containing appliances. Subsequent treatment occurs in post-shredder equipment or an incinerator.

1.2.4 Variant 4: Step 1 and step 2 processing of HC-containing appliances in auto shredder

The material flows in variant 4 are shown in the following figure:



Variant 4 assumes that HC-containing and CFC-containing appliances are sorted at the collection point, for example a public waste-collection depot. Apart from the missorted appliances, the CFC-containing appliances are treated at the premises of the fridge recycling company to a high environmental standard.

The HC-containing devices and the missorted CFC-containing appliances are treated in an auto shredder as in variant 3.

1.3 Sorting errors

The number of missorted CFC-appliances that end up in the hydrocarbon line by mistake has a significant influence on the results of the life cycle assessment. The error rate depends on many factors. In order picking, error rates are typically between 0.1 and 3 %. However in fridge recycling, there are three additional factors that have an important effect on sorting errors: the lack of labelling of many refrigerator appliances (estimated at 20–30 %), the lack of feedback to the sorter when an appliance has been incorrectly sorted, and potential problems with the recycling plant (e.g. explosion protection) if too many HC-containing are erroneously sorted into the CFC-appliance line. This suggests that the error rates in sorting refrigeration appliances are considerably higher than in order picking. The authors therefore estimate that a sorting error rate of 1 % represents a realistic lower limit that can only be achieved if all possible measures are taken to avoid incorrect sorting. In addition to the baseline calculation that assumes a sorting error rate of 1 %, sensitivity calculations with sorting error rates of 5 % have been performed for variants 2 to 4. For variant 4 a further sensitivity calculation has been made assuming a 10 % sorting error rate because, in this case, sorting takes place at local waste-collection centres where trained staff may not be available and members of the public may have to sort the appliances themselves.

1.4 Results

The baseline calculation of this life cycle assessment assumes that 20 % of the waste appliances contain hydrocarbons and 80 % are CFC-containing appliances. This corresponds to the proportions to be expected in the near future.

The following table gives an overview of the results of the impact assessment for all seven impact categories considered. Positive values indicate adverse environmental impact while negative values indicate favourable environmental impact. In the latter case, the credits from the recycling processes outweigh the adverse environmental effects. The best variant from an environmental point of view is joint processing (variant 1) and the values are shown on a dark grey background. The worst variant in each impact category is shown against a light grey background.

Table 1-1 Results of impact assessment (absolute values; proportion of HC-containing appliances: 20 %)

Variant		1	2	3	4
		Joint processing	Parallel processing	HC appliances: Step 2 in shredder	HC appliances: Steps 1+2 in shredder
Global warming potential	1000 t CO_2 eq per year	-193	-169	-155	-128
Ozone depletion potential	kg R11 eq per year	1,207	4,116	6,573	8,609
Photochemical ozone creation potential	kg ethylene eq per year	-15,032	3,828	28,221	38,035
Acidification	t SO_2 eq / year	-967	-959	-947	-948
Eutrophication	t PO_4 eq / year	-62	-62	-60.3	-60.9
Particulate matter	t PM_{10} eq / year	-1035	-1,027	-1,013	-1015
Cumulative energy demand (CED)	PJ	-2.64	-2.63	-2.60	-2.60

The table shows that the results for the impact categories 'ozone depletion potential', 'global warming potential' and 'photochemical ozone creation potential' differ considerably from one another. These impact categories are considered in more depth below.

The results for the categories 'acidification', 'eutrophication', 'particulate matter' and 'cumulative energy demand' lie very close together, with deviations of between 2 and 4 %. In view of the general level of uncertainty in the data, such small deviations cannot be taken as clear indications of an environmental advantage or disadvantage associated with a particular variant.

1.4.1 Global warming potential

The main factors responsible for the global warming potential are energy-related CO_2 emissions and CFC emissions. Because of the credits from recycling processes, the overall result is a favourable environmental impact that represents between 0.013 and 0.019 % of Germany's total greenhouse gas emissions. Expressed in tonnes, the annual benefit is between 128,000 and 193,000 tonnes of CO_2 equivalent.

The best variant from an ecological point of view is the joint processing of CFC-containing and HC-containing appliances. The worst is variant 4 in which HC-containing appliances are disposed of entirely in an auto shredder.

The following figure shows the extent to which the different processes contribute to the result. The sum for all processes is shown by the dark right-hand bar in each case.

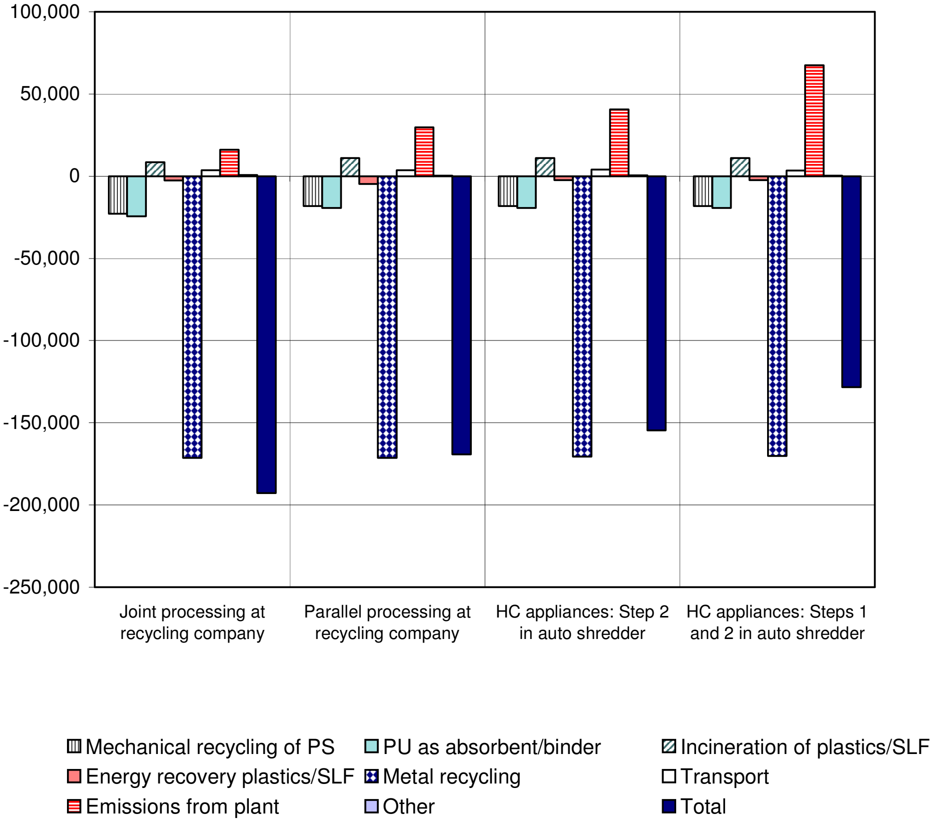


Figure 1.1 Results for the global warming potential and contribution of each individual process (proportion of HC-containing appliances: 20 %) in t CO_2 equivalent per year

The greatest differences with respect to greenhouse activity stem from the emissions from the fridge recycling plants, shredders and post-shredders, and the CFC emissions and hydrocarbon emissions from the relevant output streams (emissions from the post-processing of polyurethane and the CFC-containing refrigerator oil). The crucial factors determining the difference in the global warming potential are the emissions of the CFCs R11 and R12.

1.4.2 Ozone depletion potential

The ozone depletion potential is determined exclusively by the R11 and R12 emissions. The diagram shows the ozone depletion potential for the individual processing variants and the different sensitivity analyses.

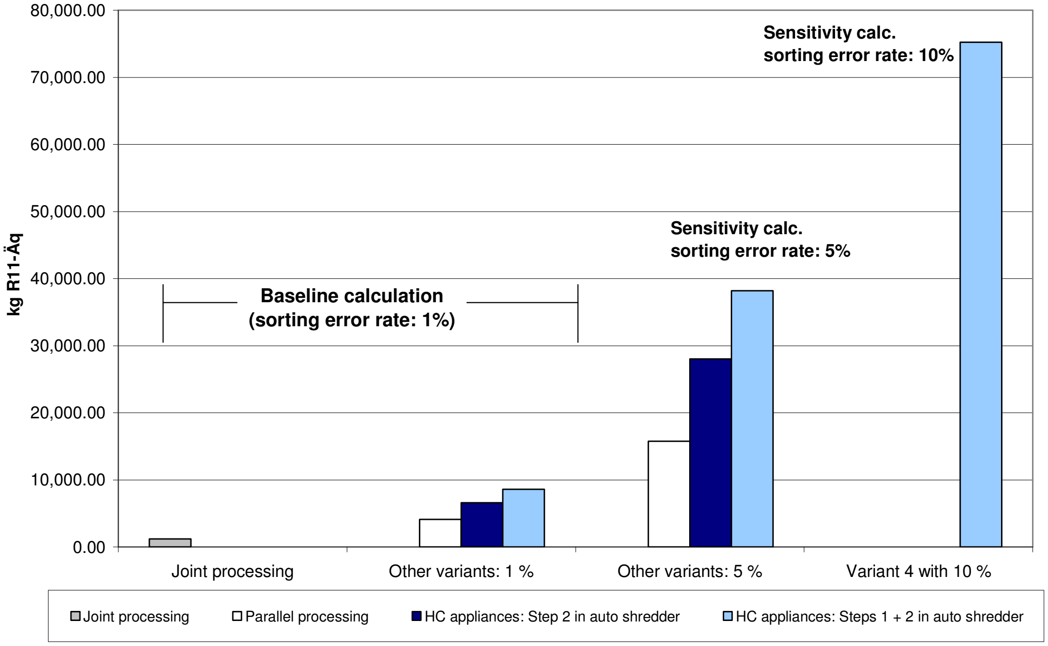


Figure 1.2 Ozone depletion potential for the baseline calculation and for the sensitivity calculations that assume different sorting error rates

In the baseline calculation, which assumes a sorting error rate of 1%, joint processing yields the lowest ozone depletion potential (ODP) of about 1200 kg R11 equivalent/year. The ODP for parallel processing and for variant 3 (HC-containing appliances: step 2 processing in auto shredder) is considerably higher, at around 4000–6500 kg R11 equivalent/year. Variant 4 (HC-containing appliances treated entirely in an auto shredder) is even higher, with about 8600 kg R11 equivalent/year. The differences are almost entirely due to emissions from missorted CFC-containing appliances.

The different processing variants account for between 0.4 and 2.8% of the emission potential associated with recently introduced ozone depleting substances. Relative to Germany's overall ODP burden (i.e. old emissions and potential new emissions), the values lie between 0.04 and 0.08% for variant 1 (joint processing) and 0.3 and 0.6% for variant 4 (HC-containing appliances treated entirely in an auto shredder). Greater precision is not possible because only rough quantitative estimates can be made regarding old emissions.

The diagram above also shows that the ODP increases dramatically at higher sorting error rates. In variant 4 the ODP rises to 38,000 and 75,000 kg R11 equivalent when sorting error rates of 5 and 10% are assumed. These values are 32 and 62 times that for joint processing and represent up to 25% of Germany's ODP (relative to the emission potential of recently introduced ozone depleting substances) or up to 3% of the national ODP (relative to total emissions, i.e. previous emissions and potential new emissions).

1.4.3 Photochemical ozone creation potential

The results for the impact category ‘photochemical ozone creation potential’ (POCP) are shown in table 1-1 and demonstrate that variant 1 yields a total environmental benefit of about -15,000 kg ethylene equivalent per year. By contrast, the other variants result in environmental burdens. The critical factors here are the isobutane and cyclopentane emissions. As a result, the greatest environmental burden is associated with variant 4.

Although these values represent only a very small fraction of Germany’s total POCP burden (0.006 % for variant 4), it is important to realize, when interpreting these results, that summer smog formation is a local and temporary process. Hence even small quantities of photochemical ozone precursors can make a significant contribution to local ground-level ozone formation for a limited period of time. This means that even low levels of hydrocarbon emissions should be avoided if at all possible.

1.4.4 Sensitivity analyses

Sensitivity analyses were carried out for the following parameters:

- Fraction of HC-containing appliances (baseline calculation: 20 %; sensitivity analysis: 50 %)
- Sorting error rate (baseline calculation: 1 %; sensitivity analyses: 5 and 10 %)
- Disposal of the CFCs recovered (baseline calculation: 100 % high-temperature combustion; sensitivity analysis: 50 % high-temperature combustion / 50 % high-temperature cracking)
- CFC destruction rates during downstream processing of polyurethane foam
- Treatment of polystyrene in parallel processing (baseline calculation: incineration in cement works; sensitivity analysis: mechanical recycling)
- Missorting of HC-containing appliances.

The sensitivity analyses show that the sorting error rate has a decisive influence on the LCA results. The other parameters also had an effect on the result but do not alter the overall conclusion. The results of the sensitivity analyses for the different sorting error rates in terms of ozone depletion potential are shown in section 1.4.2.

1.5 Conclusion

The irrefutable conclusion drawn from the life cycle assessment is that variant 1 (i.e. joint processing in a single recycling plant) is the most environmentally friendly treatment process.

With respect to the global warming potential, variant 4 achieved only about 66 % of the savings in CO_2 equivalent that are obtainable with variant 1. Variant 1 represents a saving of about 0.02 % of total greenhouse gas emissions in Germany. The use of variant 1 therefore results in additional savings of about 24,000 to 65,000 tonnes CO_2 equivalent per year compared to the use of variants 2 to 4. As climate protection is seen as a particularly important and urgent issue, all measures that achieve reductions in greenhouse gas emissions of this magnitude are significant.

In the case of ozone depletion potential, the difference between the variants is even more marked. Variant 1 differs from variants 2 to 4 by a factor of 3 to 7 at a sorting error rate of 1 %. The ODP levels for the four processing variants are still high and represent in the worst case almost 3 % of Germany's emission potential associated with recently introduced ozone depleting substances. At a sorting error rate of 5 %, the ODP for variant 4 would be about 32 times higher than that for joint processing. An additional sensitivity calculation was performed for variant 4 with an even higher sorting error rate of 10 % as in this variant refrigeration devices are sorted at local waste-collection depots. If the staff at the numerous waste-collection centres are not suitably trained, or if the sorting is left to members of the public, a sorting error rate of this magnitude is realistic. In this case the ozone depletion potential would be 62 times higher than in joint processing and would represent about 25 % of Germany's current emission potential associated with recently introduced ozone depleting substances.

In terms of photochemical ozone creation, variant 1 results in a net environmental benefit while all other variants lead to an additional environmental burden. While the emission of these photochemical ozone precursors is low (< 006 %) relative to total emission levels for Germany, the fact that even small amounts of these photochemically active substances can contribute to the formation of ground-level ozone, it is imperative that all avoidable hydrocarbon emissions are eliminated.

The differences between the variants with respect to the impact categories acidification, eutrophication, PM_{10} and energy consumption (expressed as cumulative energy demand) are so small that within the precision achievable in a life cycle assessment the results can be treated as effectively equal.

2 Aims and scope of study

2.1 Background

At present CFC-containing appliances make up the great majority (80–90 %) of waste refrigeration appliances from private households. The German CFC/Halon Prohibition Ordinance *[FCKW-Verordnung 1991]*, the European regulation on ozone depleting substances *[EU 2037/2000]* and the EU directive on waste electrical and electronic equipment (WEEE) specify that all CFCs must be removed from waste appliances and destroyed in a non-harmful way *[WEEE 2003]*. In Germany the WEEE directive was implemented as part of the Electrical and Electronic Equipment Act *[ElektroG 2005]*. Section 11 paragraph 2 of this law, in combination with appendices III and IV, and section 3 paragraph 12 of the Waste Avoidance, Recycling and Disposal Act (KrW-/AbfG) specifies that all fluids and all CFCs, HCFCs, HFCs and hydrocarbons (HCs) must be removed from end-of-life appliances and then destroyed or recovered in accordance with section 10 paragraph 4 of KrW-/AbfG.

Hydrocarbon-containing appliances, most of which contain cyclopentane and isobutane, currently make up 10–20 % of waste refrigeration appliances from private households. This proportion is set to increase to about 50 % over the next five to ten years (see section 2.3). At present the HCs have to be removed from waste appliances. The question of whether this requirement should be omitted in the next round of changes to the WEEE legislation is currently the subject of controversy. For practical purposes this would mean that such appliances would be processed directly in (auto)shredder facilities or in refrigerator recycling plants without the complete prior removal of the hydrocarbons, which would as a result be released into the environment. It is also to be expected that numerous missorted CFC-containing appliances, especially those with inadequate labelling, would find their way directly into shredders without having undergone proper prior treatment by a refrigerator disposal specialist. In such cases it can be assumed that the CFCs would not be safely removed in advance and that most of the CFCs in these appliances would be emitted into the environment.

This life cycle assessment (LCA) was commissioned by the RAL Quality Assurance Association for the Demanufacture of Refrigeration Equipment. The study considers the different potential disposal channels for household refrigeration appliances containing CFCs and HCs that could arise from possible future changes to the WEEE legislation and that are currently the subject of vigorous debate. Specialists in this field refer to appliances containing HCs both as VOC refrigerators and as HC-containing refrigerators. In the present report appliances that contain the refrigerant isobutane or the blowing agent cyclopentane are referred to as ‘HC-containing appliances’ or ‘HC appliances’.

The study has been carried out in accordance with the ISO 14040 and 14044 standards *[ISO 14040; ISO 14044]* that regulate procedures for performing life cycle assessments (LCAs). The study includes a critical review by Jürgen Giegrich of the Institute for Energy and Environmental Research in Heidelberg (ifeu). Assistance was also provided by Dr Keri of the Austrian Ministry of Agriculture, Forestry, Environment and Water Management (*Bundesministerium für Land- und Forstwirtschaft, Umwelt und Wasserwirtschaft)*, Mr Schmit of the Luxembourg State Environmental Agency (*Umweltamt*) and Mr Hornberger and Ms Janusz-Renault of the Fraunhofer Institute for Manufacturing Engineering and Automation (*Fraunhofer-Institut für Produktionstechnik und Automatisierung*).

2.2 Aims of study and target audience

The focus of this study is an ecological comparison of the different processing and disposal options for waste domestic refrigeration appliances containing CFCs and HCs that could arise from possible future changes to the WEEE legislation and how these changes might affect the resulting emissions of HCs and CFCs. The study has been written for decision makers in politics, in European environmental agencies and in the business sector. The environmental impact analysis is intended to aid the decision making process. The following processing variants were investigated:

Variant 1: Joint processing at the premises of the fridge recycling company for all HC- and CFC-containing appliances[1].

Variant 2: Parallel processing at the premises of the fridge recycling company for all HC- and CFC-containing appliances[2].

Variant 3: Joint step 1[3] processing of all HC- and CFC-containing appliances. In step 2[4] all CFC appliances, except those that have been missorted, remain at the premises of the fridge recycler. The HC-containing devices and the missorted CFC appliances are processed in auto shredder facilities.

Variant 4: Appliances containing HCs are separated from those containing CFCs at public waste collection depots. The CFC appliances are processed at the premises of the fridge recycler. The HC appliances and the missorted CFC appliances are processed in an auto shredder.

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[1] In joint processing all appliances are treated in a single plant without prior sorting.

[2] In parallel processing the appliances are sorted before further treatment and are then processed in separate plants (no batch operation).

[3] Step 1 involves removal of the refrigerant and the refrigerator oil from the cooling.

[4] In step 2 the appliance carcass is shredded and then separated into the different material fractions for recycling, reuse or disposal.

Variants 2, 3 and 4 are subject to the proviso that it is permissible to treat HC-containing appliances without the prior removal and disposal of cyclopentane. Variant 4 is subject to the additional proviso that the isobutane refrigerant does not have to be extracted and removed before further treatment takes place[5].

The following environmental impact categories are evaluated and compared for each variant: ozone depletion potential, global warming potential, acidification, eutrophication, particulate matter, photochemical ozone creation potential (summer smog) and energetic resources.

2.3 Function and functional unit

The function to be fulfilled by the system under investigation is the disposal of refrigeration appliances containing CFCs and HCs. The quantity of domestic refrigeration appliances requiring disposal each year has been estimated at 120,000 tpa *[ifeu 2005]*. This is in approximate agreement with the figures given by the German Statistical Office (*Statistische Bundesamt*) for the quantity in 2001 *[StaBu 2003]*. Assuming an average weight of 40 kg per standard fridge, as defined in section 3.1, a total of three million domestic refrigeration appliances require disposal each year. Of these three million appliances about 10 % are not considered in this LCA because they represent appliance types that are not relevant to this investigation (appliances with defective cooling circuits, absorber-type refrigerators containing ammonia, appliances with polystyrene or fibreglass insulation, appliances containing the refrigerant R134a). For the purposes of this LCA the functional unit is therefore taken to be the treatment and disposal of 2.7 million CFC- and HC-containing domestic refrigeration appliances made up of a mixture of domestic fridges, fridge-freezers, and chest and upright freezers[6].

2.4 Timeline

The results of the LCA assume that 20 % of the appliances contain HCs and 80 % CFCs. This corresponds to the proportions expected in the near future[7]. The situation in about 5–10 years' time is considered by carrying out a sensitivity analysis. By that time the proportion of waste refrigeration appliances containing HCs is expected to

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[5] For variants 3 and 4 it is assumed that fridges only make up a small proportion of the total shredder input so that shredder plants comply with the emission limits set by the national air quality control regulations (*TA Luft*).

[6] The CFC content assumed for appliances in this study is based on the mix of appliances defined in *[RAL 2003]*: 60 % domestic fridges, 25 % domestic fridge-freezer combination units and 15 % domestic chest and upright freezers.

[7] In *[RAL 2005]* the current proportion of HC appliances is estimated at 10 %. The analysis in *[Gabriel 2005]* yields a figure of the same order of magnitude. Estimates made in 2000 for Switzerland assume an earlier rise in the proportion of HC appliances *[SENS 2000]*.

have risen to about 50%[8] as a result of the ban on the use of CFCs in refrigeration equipment that came into effect in 1994.

2.5 Description of the processing variants studied

The processing of end-of-life refrigeration equipment is carried out by fridge recyclers as a two-step process. The recycling process used depends mainly on the type of refrigerant, the type of insulation material and the blowing agent used in the insulation material.

Step 1 involves the vacuum extraction of the CFCs or HCs and the refrigerator oil from the refrigerant circuit and the removal of glass, capacitors, mercury switches, compressors, cover boards and loose plastic components. The majority of waste appliances still contain the CFC R12 which was used until the mid 1990s. Waste appliances made since then contain alternative refrigerants, particularly isobutane (R600a) and, to a lesser extent, R134a. The refrigerant R134a has no ozone depletion potential but does possess a very high global warming potential of 1300 CO_2 equivalents. However, as R134a plays only a minor part in domestic refrigeration appliances and is no longer used in new appliances by German manufacturers *[UBA 2004b]*, it is not considered in this LCA. Almost all new appliances contain isobutane as the refrigerant. It has no ozone depleting potential and its global warming potential is only 3 CO_2 equivalents. As isobutane is an explosive gas, recyclers have to take appropriate precautions. Special ammonia/water absorption refrigerators are also used in the hotel and camping sectors. However according to *[UBA 2004b]* the number of ammonia absorption appliances is very low, a finding confirmed by data from plant operators[9]. Absorption appliances are not considered further in this study.

Step 2 involves the shredding and separation of the polyurethane (PU) foam from the carcass materials, the breaking down and degassing of the PU foam and the further separation of the different output fractions (metals, various plastics, CFCs, etc.). Until the mid 1990s, R11 was the main blowing agent used for the PU foam. It was then largely replaced by cyclopentane. For a while refrigeration appliances employing the blowing agents R-141b and R-134a were also used in Germany. However, as they constitute only a very small proportion *[UBA 2004b]* they are not considered further.

Appliances insulated with mineral wool/fibreglass or polystyrene constitute another group that accounts for about 10% of the total[10]. These appliances are not considered because, although their refrigerant circuits often contain CFCs, they do not cause

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[8] According to estimates made by RAL *[RAL 2005]* the proportion of HC-containing appliances will grow to 50% within about 5 – 10 years. Exactly when the 50% level is reached is not relevant for the present study.

[9] See section 3.2 for details of company data used.

[10] Estimates from data in *[Gabriel 2005]*

emissions of either CFCs or HCs in step 2 and are therefore only moderately relevant for this study.

The present study covers steps 1 and 2 for all variants and the subsequent downstream processing or recycling of the output fractions.

An important condition for all variants is that high environmental standards are maintained when processing CFC-containing appliances, at least where these appliances can be recognized as such and are not missorted into other treatment channels.

In the present study the following four variants were evaluated and their environmental costs compared:

Variant 1: Joint processing

Variant 2: Parallel processing

Variant 3: Step 2 processing of HC-containing appliances in an auto shredder

Variant 4: Step 1 and step 2 processing of HC-containing appliances in an auto shredder

All model calculations assume that 2.7 million CFC and HC-containing domestic refrigeration appliances are processed each year in Germany. It is assumed that, initially, 20 % of these appliances contain HCs. A sensitivity analysis also considers a 50 % proportion of HC-containing appliances.

The removal of the refrigerator oils, mercury switches and capacitors is identical in all variants. Even in variant 4, it is assumed that these components are removed by hand before the appliance enters the shredder.

2.5.1 Variant 1: Joint processing

Material flows for joint processing are shown in the following figure:

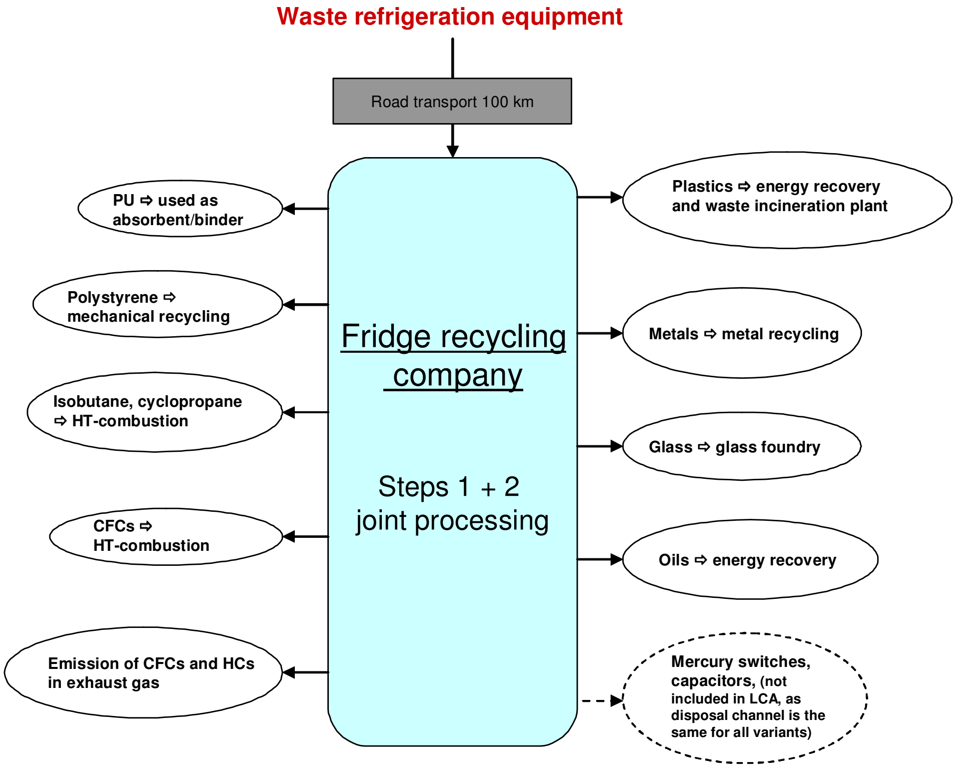

Figure 2.1 Material flow diagram for variant 1 – Joint processing

In variant 1, the waste refrigeration appliances are all treated at the premises of the fridge recycling company. This involves the simultaneous treatment of waste appliances containing hydrocarbons (HCs) and those containing chlorofluorocarbons (CFCs) in a single plant. The process engineering needs to meet two basic requirements. Firstly, appropriate precautions must be taken for dealing with the explosive materials isobutane and cyclopentane. Secondly, as far as possible all the CFC in the cooling circuits and the PU foam must be collected and disposed of. This requires a complex processing technique that ensures only minimal amounts of PU foam remain attached to the other output fractions, and involves either grinding the PU foam to a fine powder or destroying its structure in some other way to maximize degassing of CFC and cyclopentane from the foam. The CFCs and HCs recovered are then destroyed together by high-temperature incineration. In addition to the PU powder, a high purity polystyrene fraction is produced that can be recycled as polystyrene granules.

2.5.2 Variant 2: Parallel processing

The material flows in parallel processing are shown in the following figure:

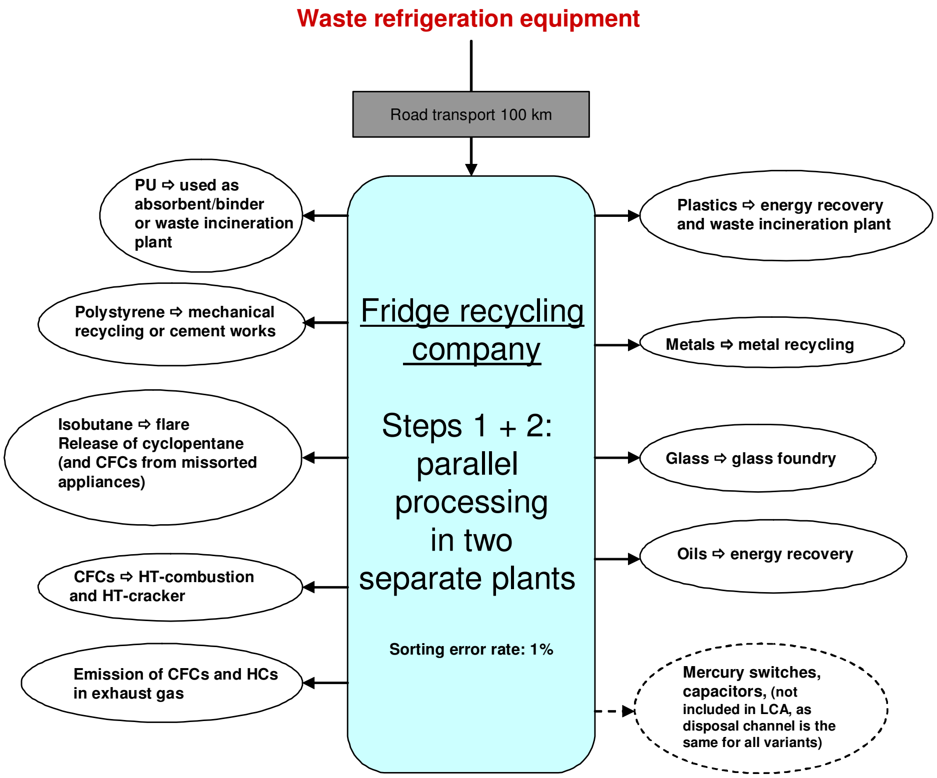

Figure 2.2 Material flow diagram for variant 2 – Parallel processing

In this variant, the waste refrigeration appliances are all treated at the premises of the fridge recycling company. The HC appliances and those containing CFCs are processed in separate plants (no batch operation). The CFC line operates essentially as in variant 1 (joint processing). As the CFCs and the explosive HCs are not collected together, they can be disposed of either by high-temperature combustion or by high-temperature cracking. However this is only possible if relatively few HC-containing appliances end up in the CFC line as a result of inadequate labelling. If a high proportion of explosive HCs is present, the mixture cannot be treated by high-temperature cracking. In this LCA it is assumed in baseline calculations that high-temperature combustion is used for disposal. In the sensitivity analysis a disposal mix is considered involving 50% high-temperature combustion and 50% high-temperature cracking.

The treatment of HC-containing appliances is markedly different from that in joint processing. In step 1 the isobutane from the HC-containing appliances is extracted separately. It is assumed for the purposes of this LCA that it can then be destroyed on site by controlled incineration[11]. It is also assumed that the cyclopentane collected during shredding is subsequently released. The PU foam from HC appliances is not further degassed or ground down but remains in pieces. Because the treatment is technically simpler than in joint processing, it is assumed here that the PU and polystyrene fractions cannot be mechanically recycled, but that the PU foam is incinerated and the polystyrene energetically recycled in a cement works. An alternative scenario, in which parallel processing produces a pure polystyrene fraction that can be mechanically recycled, is considered in a sensitivity analysis.

The material and energy balance inventory also takes account of possible missorting of CFC-containing devices. In the baseline calculation, it is initially assumed that one in every hundred CFC-containing appliances is missorted prior to treatment in step 1 and is processed along with the HC-containing appliances (see discussion on missorting rates in section 3.4).

2.5.3 Variant 3: Step 2 processing of HC-containing appliances in an auto shredder

The material flows relevant to variant 3 are shown in the figure below:

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[11] Isobutane can be recycled but this is not normal practice at present. With 2.7 million appliances, 20 or 50 % of which contain HCs, recycling the isobutane would yield credits of 100–300 t CO_2 eq as a result of reducing CO_2 emissions by not releasing or burning the hydrocarbon. These credits offset costs that would need to be taken into account in a more detailed investigation.

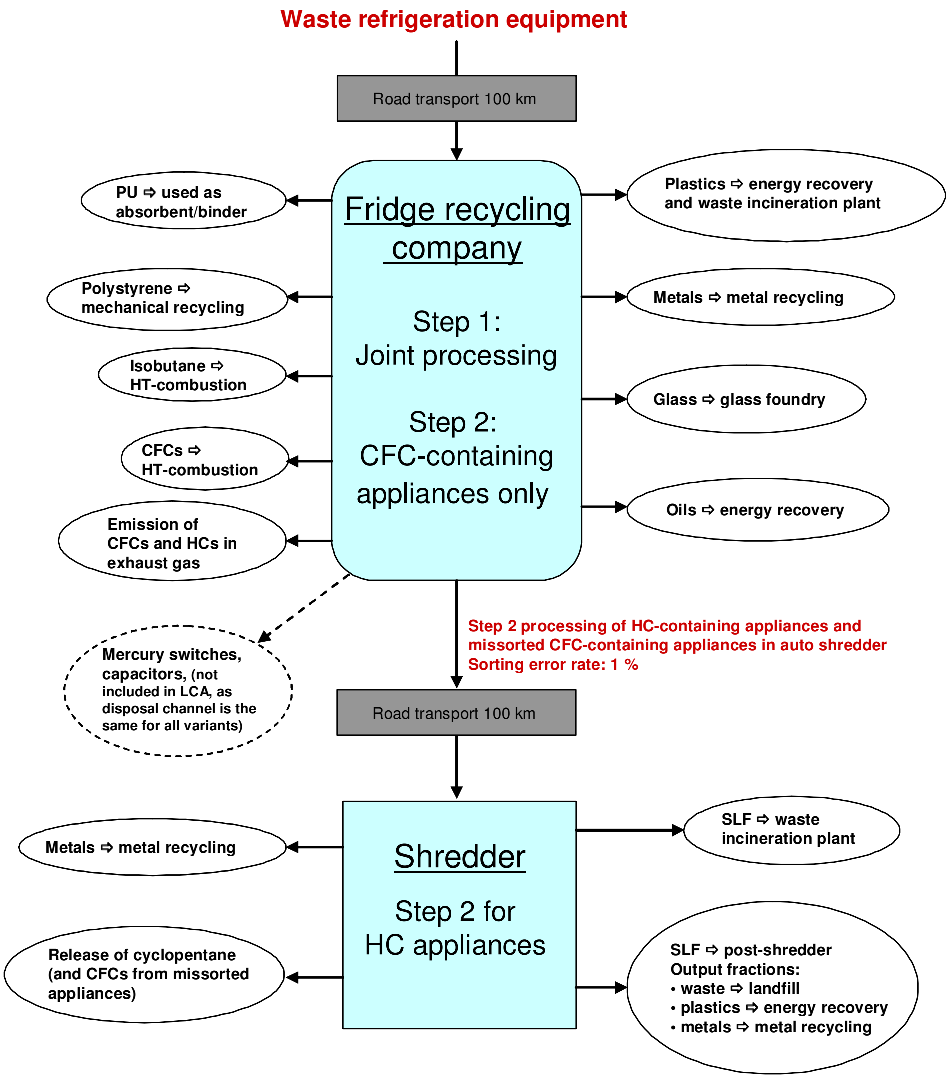

Figure 2.3 Material flow diagram for variant 3 – Step 2 processing of HC-containing devices in auto shredder

Variant 3 assumes that step 1 processing is carried out at the premises of the fridge recycling company with joint processing of HC-containing and CFC-containing appliances. The appliances are sorted into those containing HCs and those containing CFCs before step 2 processing begins. Apart from incorrectly sorted appliances, the CFC-containing units are then treated at the premises of the fridge recycling company as in variant 1.

The HC-containing appliances and the missorted CFC-containing appliances (assumed sorting error: 1 %)[12] are transported to an auto shredder facility for disposal. As well as the metal fractions, which are recycled, a so-called shredder light fraction (SLF) is produced. This fraction is treated in different ways (60 % is incinerated and 40 % is processed in post-shredder equipment). In the sensitivity analysis that considers the situation in 5 to 10 years time, where the proportion of HC-containing appliances has risen to 50 %, it is assumed that new post-shredder capacity will be created and that only 20 % of the SLF will be incinerated while 80 % is processed in post-shredder plants. Post-shredder processing involves further separation into different fractions for metal and energetic recycling. Energetic recycling is carried out in blast furnaces, reflecting current disposal practice.

It is assumed that the CFCs are disposed of by high-temperature combustion. CFCs from joint step 1 processing cannot be treated by high-temperature cracking because of the proportion of HCs present. It is also assumed that for reasons of logistics the CFCs from step 2 follow the same disposal route.

2.5.4 Variant 4: Step 1 and step 2 processing of HC-containing appliances in auto shredder

The material flows associated with variant 4 are shown in the following figure:

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[12] See discussion of sorting error rates in section 3.4.

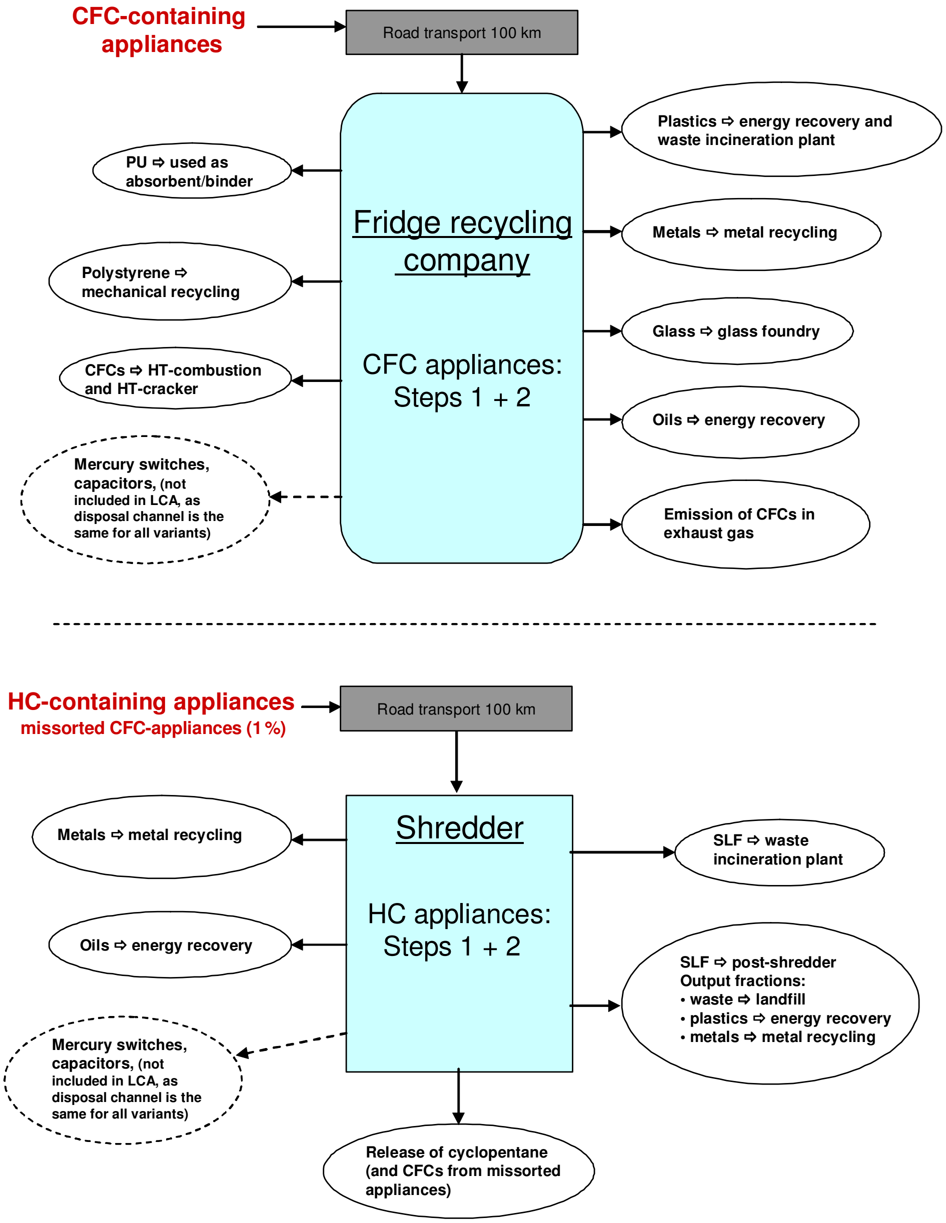

Figure 2.4 Material flow diagram for variant 4 – Step 1 and step 2 processing of HC-containing appliances in auto shredder

Variant 4 assumes that HC-containing and CFC-containing appliances are sorted at the collection point, for example a public waste-collection depot. Apart from missorted appliances, the CFC-containing appliances are then treated at the premises of the fridge recycling company as in the previous variants.

The HC-containing appliances and the missorted CFC-containing appliances (assumed sorting error: 1 %; see discussion in section 3.4) are processed entirely at an auto shredder facility. It is a fundamental assumption of this variant that permission will be given for appliances to be treated in the auto shredder without prior removal of the HC-containing refrigerant[13]. The rest of the disposal process is as described for variant 3.

With respect to CFC disposal, the same conditions apply as for parallel processing. The baseline calculation therefore assumes that 100 % of the CFC is subjected to high-temperature combustion.

The following table gives an overview of the recycling routes for the output fractions of the different variants.

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[13] In this variant it is assumed that fridges only make up a small proportion of shredder input so that the plant complies with the limits set by national air quality control regulations (*TA Luft*) despite the release of isobutane and cyclopentane.

Table 2-1 Overview of material treatment channels in variants 1-4

Variant	1 Joint processing	2 Parallel processing		3 HC appliances: Step 2 in shredder		4 HC appliances: Steps 1+2 in shredder	
		HC appliances	CFC appliances	HC appliances	CFC appliances	HC appliances	CFC appliances
Cyclopentane	H-T combustion	Released		Released		Released	
Isobutane	H-T combustion	Flare		H-T combustion		Released	
Refrigerant R12	H-T combustion		H-T combustion**		H-T combustion		H-T combustion
Blowing agent R11	H-T combustion		H-T combustion**		H-T combustion		H-T combustion
PU	Absorbent/binder	Waste incinerator	Absorbent/binder	SLF Waste incinerator and post-shredder*	Absorbent/binder	SLF Waste incinerator and post-shredder*	Absorbent/binder
PS	Mechanical recycling	Cement works	Mechanical recycling		Mechanical recycling		Mechanical recycling
Other plastics	Incineration and cement works	Incineration and cement works	Incineration and cement works		Incineration and cement works		Incineration and cement works
Metals	Metal recycling						
Oil	Thermal treatment / Energy recovery						
Glass	Glass recycling					SLF	Glass recycling

* SLF in baseline calculation: 60% waste incinerator, 40% post-shredder
* SLF in sensitivity analyses: 20% waste incinerator, 80% post-shredder
** Sensitivity analysis assumes 50% processed by high-temperature combustion and 50% by high-temperature cracking

2.6 System limits

The processing of domestic refrigeration appliances containing CFCs and HCs was investigated within the following limits:

- The analysis of material and energy balances in the treatment of waste refrigeration appliances begins at the central collection point (e.g. public waste-collection site).
- The production of auxiliary and operating materials and the provision of the energy needed to treat and process refrigeration appliances are calculated from the raw material stage.
- The environmental burden resulting from recycling is considered up to the production of a marketable or potentially marketable product. This can take the form of either material products or energy. These products are typically secondary raw materials or industrial precursors and are not consumer end products.
- For these secondary raw materials or recycled products a complementary process is defined based on primary raw materials. LCI calculations for these products are made from the raw material stage.
- For each unit process, we adopt the commonly used convention of setting a 1 % cut-off for the input materials and their associated upstream impact. This means that the costs of production or supply are only taken into account for those auxiliary and operating materials that make up more than 1 % of the mass of the relevant output of a process. The sum of such materials not taken into account should not exceed 5 % of the mass of the relevant output of a process.
- The comparative LCA does not consider the disposal of capacitors and mercury switches as these components are treated identically in all variants.
- The following refrigerator types are not considered (see explanation in section 2.5): appliances with defective cooling circuit, ammonia/water absorption appliances, appliances containing R134a and R141b and those insulated with mineral wool/fibreglass or polystyrene.
- The effects of possible explosions during auto shredder processing of refrigeration appliances are not considered.
- The limits for the individual processes are described in the sections on modelling the individual variants.

2.7 Data quality

The ISO 14040 and 14044 standards (*[ISO 14040], [ISO 14044]*) make the following requirements with respect to data quality:

Timespan: Data relating to the quantities of different types of waste refrigeration appliances and their treatment are from the period between 2002 and 2006. The data for the subsequent downstream processing stages are from the period between 2000 and 2006.

Geographic range: The LCA refers to the situation in Germany. In other countries, different recycling infrastructures may result in different processing channels and different transport distances. Imports are taken into account when modelling upstream processes.

Technological range: Assumptions regarding the quantities and types of waste domestic refrigeration appliances containing CFCs and HCs are based on current figures for Germany. However, the variants described here do not reflect current recycling practice in which several different technology standards exist in parallel. Instead each variant illustrates the material flows that would result if all waste refrigeration equipment was to be processed in that manner.

Data categories: In this study only material flow and energy input are quantified and used in calculations. The energy resource consumption and atmospheric emissions are both taken into account.

Data quality: Overall, this study is based on data of a quality appropriate to its subject matter and aims. The modelling procedures and data used are described in detail in chapter 3. Additional sensitivity analyses are performed for important parameters such as the proportion of HC-containing appliances and the sorting error rate.

2.8 Impact assessment methods

2.8.1 Impact categories

The different environmental impacts of the pollutants released during the fridge treatment process must be taken into account when assessing the overall environmental impact of the various processing variants. The task of life cycle impact assessment is to investigate the data obtained in the life cycle inventory analysis with regard to particular environmental effects – so-called impact categories – and thus deliver additional information to aid evaluation. The various bodies working to develop life cycle analysis methods have reached a level of agreement on which impact categories should be considered in LCAs (see Table 2-2) *[CML 2001] [UBA 1995]*.

Table 2-2: Impact categories

Resource consumption	**Photochemical ozone creation potential**
Global warming potential	Land use
Stratospheric ozone depletion	Smell nuisance, noise
Toxic burden for humans[14]	Workplace burdens
Ecotoxicological burden[15]	Waste heat and refuse
Acidification of ecosystems	Radiation load
Eutrophication of ecosystems	Threats to natural beauty and diversity

The pollutant-related environmental burdens shown above in **bold type** were selected, together with particulate (PM_{10}) emissions, as the impact categories best suited for the current life cycle assessment of the fridge recycling process.

The **consumption of energy resources** is represented by the cumulative energy demand (CED). CED is a measure of the total amount of energy resources used to make a product or provide a service. It also includes the energy contained in the product itself. The CED identifies all non-renewable and renewable energy resources as primary energy values, with the higher heating value (HHV) of the various fuels used in the calculations. No characterization factors are used. This means that the consumption of energy resources is not an impact category based on different impact factors, but a life cycle inventory parameter. Metal resources are not represented because the processing variants studied differ only marginally with respect to metal recycling.

The calculation of **global warming potential** in terms of CO_2 equivalents is a generally accepted approach. In addition, the Intergovernmental Panel on Climate Change (IPCC) is a specialist international committee that generates and updates the computational methods and the corresponding values for each climatically relevant substance. The calculation of CO_2 equivalents takes into account the length of time that the gases spend in the troposphere. It is therefore necessary to establish the period of time to be used for the climate model for the purposes of the LCA. Germany's Federal Environment Agency (*Umweltbundesamt*) recommends modelling on a 100-year basis as this is most likely to reveal long-term results of the greenhouse effect. The substances taken into account in the calculation of the global warming potential associated with the fridge recycling process – CO_2, N_2O, CH_4,

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[14] According to *[UBA 1999]* no methodological concepts exist for characterizing toxicity to humans or ecotoxicity. For many materials that are ecologically toxic or toxic to humans the available data are so poor that it is difficult to draw sound conclusions. However, the toxicity of isobutane and cyclopentane is taken into account in impact category photochemical ozone creation potential.

[15] See footnote 14.

R11 and R12 – are listed with their CO_2 equivalence values as specified in *[IPCC 2005]*. The values for isobutane and cyclopentane were taken from *[StaBW 2004]* on the assumption that the global warming potential of pentane is equal to that of cyclopentane.

Eutrophication means the over-enrichment of a water body or the soil with nutrients. In the context of this study, only terrestrial eutrophication is considered. The eutrophication potential of nutrient emissions is calculated from the aggregation of phosphate equivalents according to *[CML 2004]*.

Acidification can also occur in both terrestrial and aquatic systems. Emissions of acid-forming waste gases are responsible. Acid formation potentials *[CML 2001]* are calculated with the characterization factors given in *[CML 2004]*.

Particulates with an aerodynamic diameter < 10 µm are suspected of having an especially high toxicity potential for humans. Many studies have found a relationship between particle levels and mortality or morbidity. The results showed that increases in emission concentrations of PM_{10} were associated with greatly increased mortality from respiratory and cardiovascular diseases *[WHO 2002]*. The (large-area) burden from particles of diameter $\leq$ 10 µm (PM_{10}) results from both direct emissions of particulate materials and from secondary particles forming from precursor substances. These include NO_x, SO_2, ammonia and NMVOCs. They are classified according to *aerosol formation factors* which are used in EU reporting *[EEA 2002]* and are also recommended by the WHO as an indicator of air quality *[WHO 2002]*. The results of calculations performed on this basis are expressed in PM_{10} equivalents. The average value of the PM_{10} potential for NMVOC emissions in Switzerland derived by Heldstab et al. *[Heldstab et al. 2002]* is 0.012.

The impact category **photochemical ozone creation potential** (POCP) concerns the formation of summer smog or ground-level ozone. It describes the formation of ground-level ozone (O_3) in the lower layers of air. Ozone causes damage to forests and other vegetation. At higher concentrations it is toxic to humans (causing irritation of respiratory organs, asthma, coughs and eye irritation) *[Schmid et al. 2006]*. Ozone formation is a complex process in which an ozone creation potential can be allocated to each hydrocarbon. Exact potentials apply only for a defined environment with a particular light intensity, a particular NO_x concentration and defined meteorological conditions. In this LCA the impact factors shown in Table 2-3 are taken as average values that relate to 1 kg ethene equivalent. It is assumed that as there is already an existing background level of the pollutants NO_x, SO_2 and CO, these substances are not included in the LCI analysis. For isobutane, cyclopentane and methane, the current characterization factors specified in *[CML 2004]* and

[Jenkin et al. 2000] were adopted. For non-specific NMVOCs, the characterization factor from *[CML 2001]* was used[16].

The **ozone depletion potential (ODP)** describes ozone reduction in the stratosphere located about 15 to 40 km above the surface of the earth. This ozone layer is damaged by persistent halohydrocarbons, particularly CFCs. Increased destruction of ozone brings an increase in UV-B radiation at the earth's surface because the stratospheric ozone layer filters out a large part of the UV-B radiation. Increased UV-B radiation at the earth's surface can lead to an increase in the occurrence of skin and eye diseases and to immune-system impairment in humans. Even a small increase in the average level of UV-B irradiation is likely to cause damage to ecosystems and to have an adverse influence on the food chain *[UBA 2006d]*. Aggregation is expressed in terms of R11 equivalents.

The following table shows the impact factors used.

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[16] For non-specific NMVOCs *[CML 2004]* makes a worst-case estimate and assumes a characterization factor of 1. This is equivalent to double the average of all other individual substances. However, this approach has a distorting effect on the question of interest here: the individual contributions of isobutane and cyclopentane. We have therefore assumed the general characterization factor of 0.416 specified for non-specific NMVOCs in *[CML 2001]*.

Table 2-3: Impact factors for the calculation of impact potentials

Impact category	Agent	Impact factor	Source
Global warming potential kg CO2 equivalent (time horizon: 100 years)	CO_2 fossil	1	IPCC 2005
	CH4 fossil	23	IPCC 2005
	N_2O	296	IPCC 2005
	R11	4,680	IPCC 2005
	R12	10,720	IPCC 2005
	Isobutane (R600a)	3	StaBW 2004
	Cyclopentane	11	UNEP 2002
Acidification kg SO_2 eq/kg	SO_2	1.2	CML 2004
	NOx	0.5	CML 2004
	NH_3	1.6	CML 2004
Eutrophication, terrestrial kg PO_4 eq/kg	NOx	0.13	CML 2004
	NH_3	0.35	CML 2004
PM_{10} (particulates) kg PM_{10} eq/kg	Primary particles	1	EEA 2002, WHO 2002
	SO_2	0.54	EEA 2002, WHO 2002
	NOx	0.88	EEA 2002, WHO 2002
	NH_3	0.64	EEA 2002, WHO 2002
	NMVOC	0.012	Heldstab et al. 2002
Photochemical ozone precursors kg ethylene eq/kg	NMVOC	0.416	CML 2001
	Isobutane	0.307	CML 2004
	Cyclopentane	0.515	Jenkin et al. 2000
	CH_4	0.006	CML 2004
Ozone depletion potential (ODP) kg R11 eq/kg	R11	1	CML 2004
	R12	0.82	CML 2004

2.8.2 Normalization

The values obtained were set against the overall burden for Germany to show the relative significance of the various environmental influences and to allow potentially opposing results to be weighed against each other. Normalization makes it possible to identify those impact categories that make a particularly significant contribution to the current environmental situation.

2.8.2.1 Total emissions in Germany

Table 2-4 shows the overall burden in Germany and the aggregation of the individual impact categories.

Table 2-4: Total emissions and consumption in Germany and their aggregated environmental effects as a basis for normalization

Impact category	in 1000 t/a	Source
Global warming potential (CO_2 eq)	1,017,000	*[UBA 2005a]* for 2003
N_2O	205	*[UBA 2006a]* for 2003
CO_2	865,000	*[UBA 2006a]* for 2003
CH4 fossil	3.582	*[UBA 2006a]* for 2003
Acidification (SO_2 eq)	2.415	calculated*
SO_2	616	*[UBA 2006a]* for 2003
NOx	1.428	*[UBA 2006a]* for 2003
NH_3	601	*[UBA 2006a]* for 2003
Eutrophication, terrestrial	394	calculated*
NOx	1,428	*[UBA 2006a]* for 2003
NH_3	601	*[UBA 2006a]* for 2003
PM_{10} (particulates)	2,262	calculated*
Primary particles	271	Pregger 2006 for 2000
SO_2	616	*[UBA 2006a]* for 2003
NOx	1,428	*[UBA 2006a]* for 2003
NMVOC	1,460	*[UBA 2006a]* for 2003
NH_3	601	*[UBA 2006a]* for 2003
Photochemical ozone precursors	629	calculated*
NMVOC	1,460	*[UBA 2006a]* for 2003
CH_4	3,582	*[UBA 2006a]* for 2003
Ozone depletion potential (New use of potentially ozone depleting substances)	0.305	*[UBA 2006a]* for 2003
CED	14,334 PJ	*[UBA 2004a]* for 2003

* Calculated with the impact factors shown in Table 2-3.

2.8.2.2 Normalization of the ozone depletion potential values

Germany's overall emissions of substances that can damage the stratospheric ozone layer are of particular interest for this study. As statistical data are not available on present levels of emissions from waste appliances, building materials and from landfill, only very rough estimates of the current emissions situation are possible.

According to a study carried out in 1995 by Ökorecherche *[Ökorecherche 1995]*, the total ODP for that year was around 5,500 t R11 eq. We can assume that today's level is markedly lower as a result of Germany's CFC/Halon Prohibition Ordinance *[FCKW-Verordnung 1991]* and comes from three main sources: rigid foam insulating boards containing CFCs that are still present in buildings or have been landfilled, CFC-containing refrigeration and air-conditioning equipment (domestic, industrial, vehicular), and new applications.

Outgassing of CFCs from rigid foam boards that are still present in buildings or have been landfilled was estimated at 1,300 t R11eq/a based on an assumed half-life of 100 years. Because of this long half-life we expect current CFC emissions from rigid foam boards to be of the same order of magnitude as in 1995. A few hundred tonnes of CFC emissions may also result from incorrect disposal of waste refrigeration and air-conditioning appliances. Potential emissions from new uses of CFCs have been estimated in *[UBA 1996]* to be about 305 t R11eq/a. From these figures, we can make a rough estimate that the overall ODP in Germany is of the order of between 1,500 and 2,500 t R11eq/a.

Because this estimate is very approximate, the normalization used in this LCA relates to current new uses of those ozone-depleting substances that can be expected to become relevant to emissions during their life cycle. This approach is consistent with the normalization procedure used in other impact categories. Except for global warming potential, the other impact categories do not consider emissions from end-of-life materials. While methane emissions from landfill sites are taken into account when calculating annual CO_2 equivalent greenhouse gas emissions, emissions from CFC-containing materials that are either in landfill or still in use are not included in the calculation of Germany's greenhouse gas inventory *[UBA 2005b]*.

According to *[UBA 2006b]*, just over 39,000 tonnes of ozone depleting substances were used in 2003. They were used mainly as source materials for making other chemical products (76.7 %) and were completely destroyed or converted in the course of the manufacturing process. This proportion is therefore not relevant in terms of stratospheric ozone loss. However, there are still 9,023 t of material that may be relevant for emissions. They are used mainly (97.7 %) as a refrigerant in fridges, deep freezes and air-conditioning appliances and as a blowing agent for aerosols, plastics and foam materials. Most of these substances are used in closed systems so that damage to the ozone layer and an effect on the climate is only relevant in the event of their release. These substances are therefore described as potentially relevant to emissions. In 2003, the ODP of these potentially relevant substances was 305 t R11 eq.

2.9 Allocation process

In LCAs, allocation refers to the procedures that are necessary if the systems being observed produce several recyclable products or if material or energy from external systems flows into the unit processes being observed.

In the present LCA no allocation of material or energy flows is necessary for the processes taking place in shredders or at fridge recyclers' premises. It is nevertheless possible that allocations had already been made for some of the data sets obtained from other sources relating to upstream activities (e.g. electricity generation, metal production). Details are not provided here but can be obtained from the relevant sources.

2.10 Limitations and unintended applications

The present study uses models to assess the recycling of CFC- and HC-containing refrigeration appliances from private households. This means that assumptions and simplifications have been made that must be considered when interpreting the results. In particular the following points should be borne in mind:

- The study deals exclusively with the treatment of domestic appliances. It does not include the large area of commercial refrigeration and cooling. It also excludes appliances with damaged cooling circuits, with mineral wool/fibreglass and polystyrene insulation, appliances that use ammonia or R134a as the refrigerant, and appliances containing R141b as the foam blowing agent.
- The study describes and compares model processing variants to illustrate the environmental impact under the conditions described in each case.
- The modelling of the joint processing variant is based on company data that was made available by three recycling firms. These were recycling companies maintaining high environmental standards and obtaining high-quality recycling fractions.
- Modelling of auto shredder processing does not reflect current practice or the present legal situation. To meet the model's conditions the requirements in Appendix II to the WEEE directive would need to be changed to allow HC-containing appliances to be processed using auto shredder equipment.
- The effects of possible explosions when waste refrigeration appliances are treated in an auto shredder have not been considered.

2.11 Critical review

To comply with the LCA standards ISO 14040 and 14044, any comparative LCA study intended for publication must be evaluated and commented upon in a critical review. A critical review of the present study was carried out as specified in chapter 6 of ISO 14044.

Jürgen Giegrich (ifeu - *Institut für Energie- und Umweltforschung Heidelberg GmbH*) was appointed as Critical Reviewer. He was provided with the report, the company data used and the calculation files (software used: Umberto).

Assistance was also provided by Dr Keri of the Austrian Ministry of Agriculture, Forestry, Environment and Water Management (*Bundesministerium für Land- und Forstwirtschaft, Umwelt und Wasserwirtschaft*), Mr Schmit of the Luxembourg State Environmental Agency (*Umweltamt*) and Mr Hornberger and Ms Janusz-Renault of the Fraunhofer Institute for Manufacturing Engineering and Automation (*Fraunhofer-Institut für Produktionstechnik und Automatisierung*).

3 Modelling and input data

The computer program Umberto was used for life cycle inventory modelling and impact assessment. The modelling process and the input data are described in the following sections.

3.1 Material composition of refrigeration appliances

Data on the materials used in the manufacture of domestic refrigeration appliances were taken from a number of sources, including *[Gabriel 2005]*, *[IPA 2005]*, *[ifeu 1999]*, *[Zeiler 2000]*, *[Öko-Institut 2005a]*, *[Öko-Institut 2005b]* and *[ESU 2005]*. Table 3-1 below provides an overview of the data from each source and gives detailed information about the individual fractions.

Table 3-1 Material composition of an average refrigerator according to different data sources

Material	**Recycler 1a** **kg/unit**	**Recycler 1b** **kg/unit**	**Recycler 2a** **kg/unit**	**Recycler 2b** **kg/unit**	**Recycler 3a** **kg/unit**	**Recycler 3b** **kg/unit**	**Gabriel 2005** **kg/unit**	**Zeiler 2000** **kg/unit**	**IPA 2005 (small HC appliances)** **kg/unit**	**IFEU 1999** **kg/unit**
Scrap steel	25	19	16.41	17.3	16.1	17	19	18.4	12.6	16.8
Compressors		7.7		10.5	9.0	9	8.84		7.7	8
Non-ferrous fraction from carcass		1.7					1.5			
Mixed fraction non-ferrous metals / plastics					10.0					
Cables		0.18		0.44			0.09		0.16	
Aluminium	2.5		2.21	2.3		3.1		1.96	0.46	2.4
Copper				0.07				1.28	0.22	0.24
Plastics	5.5	7.5	6.23	6.7		4	6.2	13.6	6.7	6
PU (powder)	4.0	4.4	3.97	4.0	4.2	4.5	3.4		3.69	3.2
R11	0.45 (R11 + R12)		0.286		0.297	0.32				0.24
R12			0.118		0.126	0.13				0.08
R600a (isobutane)									0.048	
Cyclopentane			0.196						0.20	
Glass	0.25	0.34	0.25	1.2	0.17	0.24	0.27	1.28	3.82	
Water	0.75		0.0326		0.25		1.27			
Oil	0.25	0.19	0.166			0.25	0.194		0.13	0.24
Remainder		0.04				1.46	0.17	3.44	2.90	3.56
Total (excluding refrigerant)	**38.3**	**41.1**	**29.3**	**42.5**	**39.7**	**39.6**	**40.9**	**40.0**	**38.4**	**40.4**
			excluding compressors and cables	excluding oil and water	excluding glass and cables					

Most of these values, especially those provided by the recycling companies and those in *[Gabriel 2005]*, are average values for all end-of-life domestic fridges, fridge-freezers, chest freezers and upright freezers sent for processing. The data from these different sources are not always comparable because individual sources group some of the fractions differently and consider different size categories. It is nevertheless apparent that the masses provided by the different data sources for the most important fractions (steel, PU, plastics) are of the same order of magnitude. Using these data, an average value for the material input, i.e. the amount of material delivered to recycling operators, was obtained for the LCA model. Its composition is shown in table 3-2. Studies by Recycler 2 showed that CFC-containing and HC-containing appliances both contain about the same amount of PU. In terms of LCA modelling, these two kinds of appliances therefore differ only in the refrigerant and blowing agent used but not in their construction. The specific amount of cyclopentane contained in PU foam was given in *[IPA 2005]* as 54 g per kg PU and this was taken as the cyclopentane content of the standard appliance used in the calculations. Its isobutane content was derived from the amount specified in *[IPA 2005]* for a small fridge with no freezer compartment. This value was multiplied by a factor of 1.25 to obtain a representative isobutane quantity for an 'average' refrigeration appliance based on the typical mix of appliance types found in Germany.

Table 3-2 Average composition of 1,000 'standard' refrigeration appliances as delivered to the recycler

Material	**Standard refrigeration appliance used in LCA** **tonnes per thousand units**
Steel scrap excluding compressors	17.0
Compressors	9.0
Non-ferrous fraction from carcass	2.0
Cables	0.15
Plastics excluding PU	6.2
PU	4.0
R11 (CFC appliance)	0.340
R12 (CFC appliance	0.115
Isobutane (HC appliance)	0.060
Cyclopentane (HC appliance	0.22
Glass	0.25
Water	0.25
Oil	0.20
Remainder	0.17
Total (excluding refrigerant)	**39.2**

It is assumed, as in *[Gabriel 2005]*, that the 6.2 kg of plastics per appliance can be sorted into the following fractions.

Table 3-3 Plastics fractions produced during the recycling process

Plastics fraction	**Quality / treatment options**	**Mass**	
		kg/appliance	**%**
Polystyrene	High purity, mechanical recycling possible	2.8	45.2
Plastics from initial disassembly	Various plastics, also contains other material e.g. plywood; usually incinerated	0.2	3.2
Plastics for energy recovery	Mixture of plastics suitable for energy recovery in cement works	0.7	11.3
Residual materials from carcass	Various plastics and impurities; usually incinerated	2.5	40.3
Total		6.2	100.0

3.2 Processes at the recycling company's premises

The modelling of environmentally sound fridge recycling procedures used operating data supplied by three recycling companies. The data published in *[Gabriel 2005]* describing material flows in a trial carried out at the company AVE GmbH & Co. KG were also evaluated. The joint processing model was derived from the analysis of these data. For the parallel processing variant, LCA calculations for the CFC line were based on the parameters used in the joint processing model. However different assumptions were made for the HC line of the parallel processing variant (see sections 2.5.2, 3.7.1 and 3.7.2). As some of the company information is confidential, the relevant data are not published in this study. All the data were nevertheless made available to the Critical Reviewer.

Processing by recycling operators generally occurs in two stages (typically referred to as 'step 1' and 'step 2') as described in section 2.5. The following processes were included in the material and energy balance analyses used for this LCA study.

Step 1

Step 1 involves the vacuum extraction/removal of the CFCs or HCs and the refrigerator oil from the cooling circuit, as well as the removal of glass, capacitors, mercury switches, compressors, cover boards and loose plastic components.

The fuel and energy costs used in modelling both the joint and parallel processing variants were based on data supplied by operators: 0.03 l of diesel and 0.1 kWh of electricity per appliance.

Direct emissions of isobutane and R12 into the air only occur to a small extent. For the level of exhaust gas emissions and the fate of the residual amounts of isobutane and R12 in the refrigerator oil, see section 3.5 (CFC material balance).

In variant 2 (parallel processing), in which the model assumes flaring of the extracted isobutane, the resulting CO_2 emissions are included in the LCI calculations.

The output fractions include isobutane, R12, cable, glass, refrigerator oil, the compressor and mixed plastics with impurities. LCI analyses are performed for the following treatment channels:

Table 3-4 Output fractions from step 1 at fridge recycling company and their treatment

Output fractions	Treatment
Isobutane	High-temperature combustion (variants 1 and 3: joint processing); On-site incineration (variant 2: parallel processing);
R12	High-temperature combustion or cracking
Cable	PVC insulation in waste incinerator; copper in copper smelter
Glass	Glass foundry
Refrigerator oil	Energy recovery
Compressor	Metal recycling
Mixed plastics with impurities	Waste incinerator
Capacitors and mercury switches	Not included in assessment, as disposal channel is the same for all variants

Step 2

In step 2 of the fridge recycling process, the appliance carcass is shredded in an encapsulated shredder and then separated into various fractions: PU foam, various plastics (polystyrene and other plastics fractions of varying purity), scrap iron and a fraction containing non-ferrous metals. If the PU foam contains CFCs, it is ground or treated in some other way in order to destroy the polymer's pore structure and maximize the release of the CFCs contained in the foam. The exhaust gas from the recycling facility is collected and purified.

The amounts of fuel, electric power and operating materials used when modelling the joint processing variant were 0.88 kg of nitrogen[17], 0.03 l of diesel and 5 kWh of electricity per appliance. For the HC line, the specific electricity consumption is taken to be 25 % lower in parallel processing than in joint processing because a less complex separation technique can be assumed.

It is assumed that all the cyclopentane freed during shredding in variant 2 (parallel processing) is then released to the atmosphere (30 % of the total cyclopentane, see section 3.6.2). In variant 1 (joint processing), by contrast, the cyclopentane is processed together with the CFCs by high-temperature combustion.

Emissions of cyclopentane and CFCs result from the residual adhesion of PU foam to the metal and plastics fractions and from the CFC and HC residues in the shredded or milled PU foam. The LCI calculations relating to CFCs and HCs are described in detail in sections 3.5 and 3.6.

In metal separation it is assumed that 99 % of the iron and 90 % of the non-ferrous metals are recycled.

The following output fractions and recycling routes are taken into account in this process:

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[17] Nitrogen consumption is low by comparison with *[IPA 2005]* because the data used in the present study came from a company that has optimized its nitrogen use with the help of suitable technology.

Table 3-5 Output fractions from step 2 of the fridge recycling process and their treatment

Output fractions	Treatment
Cyclopentane	High-temperature combustion (variant 1) Release (variant 2) Small amount released from residues in PU foam
R11	High-temperature combustion or cracking Small amount released from residues in PU foam In parallel processing (variant 2) - release in step 2 from missorted CFC-containing appliances
Iron and non-ferrous metals	Metal recycling
PU	PU powder from joint processing and CFC-containing appliances: recycled as absorbent/ binder PU flakes and chunks from parallel processing (HC line): incinerator
Polystyrene	Mechanical recycling as polystyrene granules In parallel processing: energy recovery in cement works
Mixed plastics	Plastics fraction pure enough for energy recovery: cement works Plastics fraction with a high level of impurities: incinerator

3.3 Shredder processes

When processed in an auto shredder, refrigeration appliances are shredded together with a very varied range of ferrous waste (vehicles, electrical appliances, miscellaneous scrap metal). The output fractions produced are shredder scrap, shredder heavy fraction (containing non-ferrous metals) and shredder light fraction. The following shredder material flows are critical for this LCA:

1. Metals

The LCA assumes that 96% of the iron and 80% of the non-ferrous metals are separated and recycled. Fridge recycling operators typically achieve transfer rates of 99% for iron and 90% for non-ferrous metals, because high standards are set for the purity of the material fractions in order to minimize CFC emissions. When the shredder light fraction is processed using post-shredder equipment with assumed transfer rates of 80% for iron and 60% for non-ferrous metals (see description of post-shredder treatment in section 3.7.4), the whole system – shredder plus post-shredder – achieves yields of about 99 % for iron and 92 % for non-ferrous metals. This puts it slightly above the metal yields achieved by fridge recycling companies.

The detailed material flows assumed for metals in shredder treatment are presented in detail in the appendix (see section 7.1.8).

2. Plastics

Almost all of the plastics end up in the shredder light fraction. Their treatment is described in section 7.1.8 of the appendix.

Fuel consumption and emissions

- Non-published company data were used for electricity consumption and particle emissions. These data were made available to the Critical Reviewer[18]. Diesel consumption was assumed to be 50 % of the specific consumption of the model fridge recycler (0.37 l/t)[19]. These levels of consumption are of minor significance for the results of the LCA because the total resource consumption by the fridge recycling plant, shredder and postshredder equipment together makes up less than 1 % of the overall result.
- The effects of possible explosions during auto shredder processing of refrigeration appliances are not considered.

3.4 Missorting

In variants 2 (parallel processing) and 3 (step 2 processing of HC-containing appliances in an auto shredder), the waste appliances are sorted at the recycler's premises into those containing CFCs and those containing HCs. The number of missorted CFC-containing appliances that arrive in the hydrocarbon line by mistake is very important for the results of the life cycle assessment.

Missorting rates depend on many factors. Important factors include:

- level of qualification and motivation of operating personnel,
- personnel have to cope with repetitive, monotonous work and time pressures,
- design of workplace e.g. lighting,
- state of appliances (how many labels are missing or illegible?),

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[18] Company data were compared with values from the Fraunhofer Institute for Manufacturing Engineering and Automation (*Fraunhofer-Institut für Produktionstechnik und Automatisierung*) and are of the same order of magnitude.

[19] Consumption by recycler in step 2: 0.03 l diesel per appliance. Converted to consumption per tonne of input at a mass of 40 kg per appliance.

- feedback from operating staff when missorting occurs (generally minimal as HC line and shredder operators are unlikely to provide feedback).

According to *[Lolling 2001]*, error rates in order picking are around 0.1 to 3 %[20]. So-called ‘mispicks’, in which the wrong article is selected, is the error most relevant to the sorting of refrigeration appliances. Mispicks make up 40 % of all order picking errors. Other kinds of errors include quantity errors, omission errors and status errors.

In the case of fridge recycling, however, there is another important source of error: many appliances are inadequately labelled or have no labelling at all which makes correct classification impossible. From the reports of recycling company staff and sampling surveys in the retail trade sector *[R-Plus 2006]*, *[Bresch 2006]*, *[Remondis 2006]* and *[RAL 2006]* it appears that about 20 to 30 % of waste refrigeration appliances can be expected to be wrongly labelled or have no label at all.

If error rates for CFC-containing appliances are to be kept low, staff must be encouraged and motivated to sort appliances into the CFC line in doubtful cases. If a CFC-containing appliance is nevertheless missorted into the HC line in variants 2 and 3 the recycling company is not usually informed. This is either because the shredder operator has not noticed the error or because missorting is not detectable in parallel processing. If appliances are sorted at local waste-collection centres, as in variant 4, feedback is even less likely. As a result of this lack of feedback, sources of error are not identified and are therefore not remedied.

The opposite applies to sorting HC-containing appliances. If a large number of HC units find their way into the CFC line by mistake, and the necessary explosion protection measures are not in place, the plant's sensors will detect the release of HCs and trigger a warning signal (explosion hazard). However, the fact that the sorting staff receive feedback on missorted HC appliances means that they will be more likely to allocate a doubtful appliance to the HC line. The sorting error rate for CFC-containing appliances increases accordingly.

In summary it can be seen that by comparison with order picking there are three important additional factors that lead to sorting errors: the lack of labelling on many waste refrigerators, the lack of feedback to sorters when a device has been incorrectly sorted, and potential problems at the plant (explosion protection) when too many HC appliances are missorted into the CFC line. It can therefore be assumed that error rates when sorting refrigeration appliances are considerably higher that in order picking. For this reason the authors feel that a sorting error rate of 1 % represents a realistic lower limit. The baseline calculation was nevertheless initially performed using this 1 % lower limit in order to identify environmentally relevant differences between the variants when the necessary effort is made to prevent the error rate rising much above 1 %.

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[20] Lolling investigated 75 order picking systems in a very wide range of industry sectors.

Variant 4 was also initially based on a sorting error rate of 1 %. In this case, sorting occurs at local waste-collection centres. With the staffing practices now typical at public waste collection depots, considerably higher sorting error rates should be assumed. The personnel are not trained to make proper distinctions between fridge types. Furthermore, as a result of efforts to save costs at local waste-collection centres, those delivering the appliances often do the sorting themselves. One waste-collection centre operator described the situation as follows *[Mühlherr 2007]*: 'From the depot operator's point of view it is absolutely unthinkable to let members of the public sort waste fridges into those that do and those that do not contain CFCs without providing additional trained staff. If we were to set up another container for CFC-free appliances, in addition to the fridge container currently provided, we would need extra staff. Even then, one hundred percent separation couldn't be guaranteed.' In the case of variant 4, achieving the assumed sorting error rate of 1 % would require higher staffing levels at waste-collection centres and extensive additional staff training. Without the extra staff and training it is certainly feasible that error rates of 5 to 10 % could occur during variant 4 processing.

As well as the baseline calculation with a sorting error rate of 1 %, sensitivity calculations were performed for variants 2 to 4 with a sorting error rate of 5 %. As a result of the additional problems described above for variant 4, a further sensitivity calculation was performed for this processing with an assumed error rate of 10 %.

3.5 Material and energy balance analysis: CFCs

3.5.1 Refrigerant R12 in step 1

For the LCA calculations we assume that the model CFC appliance contains 115 g of R12 refrigerant in its cooling circuit[21]. This quantity is almost entirely removed by fridge recyclers in accordance with the requirements of the German Federal Environment Agency (*Umweltbundesamt*). The RAL limit of 0.1 %[22] is taken as the upper limit for the proportion of R12 remaining in the refrigerator oil. Operating data from fridge recycling companies show that an even lower R12 content can be achieved.

The LCI calculations for the joint processing variant assumed that the R12 extracted is then processed by high-temperature combustion (see description in the appendix section 7.1.4) with a destruction rate of 99.99 %. In the parallel processing variant, the R12 is not extracted together with the (explosive) isobutane and can therefore be treated by either high-temperature cracking or high-temperature combustion. A sensitivity analysis was performed to determine whether 50 % high-temperature

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[21] Value as specified in the RAL standard *[RAL 2003]* or the UBA guideline *[UBA 1998a]*. This value is in good agreement with data supplied by fridge recycling companies.

[22] 0.1 % refers to the mass of R12 relative to the mass of refrigerant oil.

combustion and 50 % high-temperature cracking has an effect on the LCI results (for description see appendix, section 7.1.5).

Exhaust gas emissions are included in the calculations for step 2.

In the case of missorted CFC-containing appliances in variant 4 that are sent for shredding prior to step 1, all the R12 and isobutane[23] is released into the atmosphere.

3.5.2 Blowing agent R11 in step 2

The model assumes that R11 is the only blowing agent used in CFC-containing appliances. The amount present is taken to be 85 g per kg PU foam[24]. Estimates of the fate of the R11 were based on company data and the expectation values in the RAL GZ-728 quality assurance and test specifications *[RAL 2003]*. The LCI analysis includes the following R11 release channels into the environment:

The majority stems from residual emissions from the degassed PU foam. The R11 content of degassed PU foam was assumed to be 0.2 %[25], which agrees with the maximum value permitted by RAL *[RAL 2003]* and real values supplied by fridge recycling companies.

Residual PU adhering to the metal and plastics fractions are a further source of emissions. A maximum value of 0.5 %[26] (as specified in *[RAL 2003]*) was assumed for the material and energy balance calculations in this study. According to data from fridge recycling companies, actual values are at or below these levels. The residual PU foam adhering to metal and plastic scrap still contains 30 % of the original quantity of R11 (*[UBA 1998a]*, *[Elektrolux 1989]*, *[iuta 2005]*[27]). The same applies for the missorted CFC-containing appliances which are shredded in variants 3 and 4. For these appliances the assumption is made that 70 % of the R11 originally contained in the foam is released. The remaining 30 % is then either emitted or destroyed depending on how the metal and plastics scrap is then treated. If sent for landfill, it is assumed that all the R11 is released into the environment over a period of years. If incinerated, an R11 destruction rate of 90 % is assumed (see explanation in section 3.5.3). In the case of parallel processing, information from recycling companies indicates that gentle shredding into larger chunks only releases 30 % of

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[23] This assumes that waste refrigeration equipment forms such a small proportion of total shredder input that emissions remain within air quality control limits (*TA-Luft*).

[24] Expected value according to the UBA guideline *[UBA 1998a]*. Values from fridge recycling companies are of the same order of magnitude.

[25] 0.2 % refers to the mass of R11 relative to the mass of PU.

[26] 0.5 % refers to the mass of residual PU still adhering to metal and plastics relative to the mass of the metal and plastics fractions respectively.

[27] See UBA guideline *[UBA 1998a]*. This assumes that 20–30 % of the R11 remains in the shredded foam. Investigations by a recycling plant operator *[iuta 2005]* showed that 28–34 % of the R11 was still present within foam after shredding with rotary shears. Studies cited by *[Elektrolux 1989]* showed that, after shredding of PU foam into 10–15 mm pieces, 37 % of the R11 was still contained in the foam.

the R11 originally contained in the foam, with the other 70 % remaining in the foam chunks.

The proportion of CFCs released in the course of subsequent processing of the foam is discussed in section 3.5.3.

Using data from recycling companies, the model assumes exhaust gas emissions of 4 mg R11/m^3 with 9 m^3 of exhaust gas being produced per standard appliance.

It is also assumed that the R11 collected is subjected to high-temperature combustion (see description in appendix, section 7.1.4) with a destruction rate of 99.99 %. A sensitivity calculation was also performed for the parallel processing variant in which 50 % of the R11 was treated by high-temperature combustion and 50 % by high-temperature cracking.

3.5.3 Destruction of residual CFCs in refrigerator oil, PU foam and other fractions

A small proportion of the CFC refrigerant R12 remains in the refrigerator oil fraction which goes for energetic recovery. According to *[Vehlow et al. 2003]*, R12 is not completely destroyed by incineration because of its high thermal stability. The following degradation rates were used for different incineration quality levels:

- A destruction rate of 98 % was assumed for high-quality incineration in a refinery or cement works in which an adequate residence time at a temperature of at least 850 °C could be guaranteed (sensitivity analyses for 95 % and 99.5 % were also performed).
- A destruction rate of 90 % was assumed for R12 from missorted appliances in parallel processing (sensitivity analyses for 80 % and 99 %). This R12 is destroyed in the flare intended for the destruction of isobutane.

Two properties affect the incineration of the R11 contained in PU foam. Firstly, R11 is less stable than R12 and is therefore more easily destroyed. Secondly, PU is very light and, depending on furnace construction, can be carried away so quickly by the exhaust gas flow that it never reaches the hot areas of the furnace or does not remain in them for long enough. Unlike the R12 in the refrigerator oil; R11 is also emitted from foam during transport, storage and any further shredding. For these reasons the following destruction rates are assumed for the downstream treatment of PU (waste incineration, energy recovery or metal recycling plus the associated transport logistics):

- degassed milled PU: 98 % (sensitivity analyses for 95 % and 99.5 %).
- PU chunks: 90 % (sensitivity analyses for 80 % and 99 % respectively).

Destruction rates for PU chunks are lower here because they have not been subjected to degassing. Emissions during transport and storage are therefore likely to be higher.

In contrast to the treatment of residual CFCs described above, the CFCs recovered during the main fridge recycling process are almost completely destroyed by high-temperature combustion or, as modelled in the sensitivity analyses, by high-temperature cracking (destruction rate 99.99 %).

3.6 Material and energy balance analysis: Hydrocarbons

3.6.1 Refrigerant R600a

a) R600a emissions in fridge recycling plants

According to table 3-2, the standard HC appliance used in the calculations is assumed to contain 60 g of isobutane (R600a) as the refrigerant. As with CFC-containing appliances, it is assumed that the refrigerant is almost completely extracted. In joint processing, the isobutane is processed together with R12 by high-temperature combustion. In parallel processing, the isobutane is destroyed on site by controlled incineration.

It is assumed, in analogy to the CFC-containing appliances, that the refrigerator oil contains a maximum of 0.1 %[28] isobutane. However, this residual isobutane is destroyed almost completely (99.99 %) when the refrigerator oil is used for energy recovery.

In accordance with data from fridge recycling plant operators, isobutane gas emissions were assumed to be 4 mg/m^3 with 4.5 m^3 of exhaust gas produced per appliance.

b) R600a emissions from auto shredders

As defined in this study, variant 4 assumes that it is permissible to shred HC-containing appliances without the prior controlled removal of the isobutane. For this to become a real alternative the present laws would need to be changed appropriately. It is assumed that in variant 4 all the R600a in the appliance is released to the environment.

3.6.2 Cyclopentane as blowing agent

a) Cyclopentane emissions in fridge recycling plants:

According to table 3-2, the standard appliance used in the calculations is assumed to contain 220 g of the blowing agent cyclopentane at the time of delivery to the

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[28] 0.1 % refers to the ratio of the mass of isobutane to the mass of refrigerant oil.

recycling plant. This is equivalent to 55 g per kg of PU foam. In joint processing most of the cyclopentane emitted is in the form of residual emissions from degassed PU foam. Investigations at one recycling plant found the HC content of degassed PU foam to be 0.22 %[29]. The PU fraction is processed thermally (incineration, energy recovery or use as an absorbent/binder with subsequent incineration), in the course of which the cyclopentane is almost completely (99.99 %) destroyed.

In parallel processing the PU foam is only shredded into chunks. Unlike the procedure for CFC-containing appliances, no milling is involved. According to information provided by fridge recycling companies and investigations by Elektrolux *[Elektrolux 1989]*, only 30 % of the original cyclopentane escapes from the PU foam matrix[30] in the shredding process. The cyclopentane that is outgassed during shredding is all released. In contrast to the procedure in joint processing, no further degassing takes place after shredding. This means that 70 % of the original cyclopentane is still present in the shredded foam. As in the joint processing variant, the cyclopentane is assumed to be almost entirely destroyed in the subsequent thermal processing of the PU foam.

The PU residues adhering to the metal and plastics fractions are also considered as an additional emissions channel in joint processing. Like the CFC-containing appliances, we assume a figure of 0.5 %[31] for the residual PU adhering to plastic and metal scrap.

According to data from fridge recycling companies, the amounts of cyclopentane emitted in the exhaust gas stream can be calculated from the specific emission level of 4 mg/m^3 and exhaust gas production of 9 m^3/appliance.

b) Cyclopentane emissions from auto shredders

The cyclopentane that outgasses during shredding (70 %, analogous to the R11 in the residual PU foam adhering to metal and plastic scrap in joint processing, see section 3.5.2) is assumed to be released entirely to the atmosphere. The cyclopentane remaining in the PU fraction is almost completely (99.9 %) destroyed in subsequent thermal processing

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[29] Study carried out by a plant manufacturer (200 appliances, mean value from 9 individual samples). The result is of the same order of magnitude as the residual CFC content of foam (0.2 %) permitted by the RAL standard. These values refer to the mass of blowing agent in relation to the mass of PU.

[30] Studies by Elektrolux *[Elektrolux 1989]* showed that, when PU foam was shredded into 20–30 mm pieces, the proportion of blowing agent remaining in the foam was 69 %. When the foam was shredded into 15–20 mm pieces this proportion was only 37 % .

[31] 0.5 % refers to the mass of adhering PU in relation to the mass of the metal and plastics fraction.

3.7 Recycling of plastics

3.7.1 Recycling of the PU fraction

The main recycling route for the PU fraction from refrigeration appliances (around 4 kg per standard appliance) involves re-use as an oil absorber and as a binder for paint sludge and similar hazardous waste, with subsequent energy recovery. In these applications PU acts as a substitute for sawdust or plastic- or mineral-based oil binding material *[Beirat LTwS 2004]*. Another recycling option for PU that could be considered in future is its use in adhesive compressed board made from recycled materials as an alternative to wood and chipboard[32] *[IVPU 1996]*. In LCA calculations this form of recycling would be similar to use as a substitute for sawdust.

The processing of PU waste to generate the raw material polyol is not practised at present.

In the present study we assume that all the PU obtained from fridge recycling is used as a substitute for sawdust. Because of its low weight and good absorption properties, 1 kg of PU powder replaces 3 kg of sawdust *[Beirat LTwS 2004]*. The assumptions made about the destruction of CFC residues in the PU are presented in section 3.5.3.

This is a conservative approach because, when recycled PU is used as an oil binder it often replaces binders made from primary plastics (especially PU and polypropylene) *[Beirat LTwS 2004]* whose substitution would bring higher credit values.

Where PU is used as a substitute for sawmill waste, credit is given for avoiding production of wood shavings and sawdust in the wood and chipboard industries (see section 7.2.4 of the appendix for a description of these processes). Wood shavings and sawdust are currently used mainly for the manufacture of chipboard and heating pellets. Because of the high level of demand for wood shavings and sawdust there is no unused potential. Increased use of PU powder as an absorbent/binder could correspondingly reduce the amount of sawdust used for this purpose. This would make more wood shavings/sawdust available for chipboard production, which uses both forestry wood and recycled wood products as raw materials, and could thus reduce the use of forestry wood (see analysis of sources and uses of sawmill waste *[Knappe et al. 2006]*.

In parallel processing, there is no complex treatment of HC-containing appliances to produce a pure PU powder fraction. It is therefore assumed that the PU chunks produced in the HC line are incinerated. This is a conservative approach, as the further milling of the PU foam needed to achieve higher quality recycling would also lead to a greater release of cyclopentane from the foam and, in the case of missorted

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[32] According to companies manufacturing compressed board from recycled materials, it would be possible to use the PU fraction provided the material was sufficiently pure. At present, however, the process uses only CFC-free and very clean scrap from PU foam board production.

CFC-containing appliances, to increased release of R11. This would have an especially marked effect on both the ozone depletion potential and photochemical ozone creation potential.

The PU foam that is shredded contributes to the shredder light fraction. The disposal of this fraction is described in section 3.7.4.

3.7.2 Recycling of the polystyrene fraction

In joint processing and in the CFC line of parallel processing, fridge recycling companies use complicated separation techniques to produce a very pure polystyrene fraction. This fraction amounts to just over 4 kg per standard appliance and can be recycled mechanically (polystyrene granules). It is assumed that every 100 kg of polystyrene on the input side can be offset by 95 kg of polystyrene granules on the output side. The LCI result is calculated with data from the Boustead database *[Boustead 2006]*.

The separation of the material fractions from HC appliances is less complicated in parallel processing than in joint processing because there is no need to minimize the amount of (CFC-containing) PU residue still adhering to metals and other plastics. For the purpose of the LCI analysis, it is therefore assumed that the polystyrene fraction is not sufficiently pure for mechanical recycling, but is used for energy recovery in a cement works. Energy recovery by incineration in a cement works is described in more detail in section 7.1.6 of the appendix. In addition, a sensitivity analysis is performed to assess the mechanical recycling of polystyrene in parallel processing.

3.7.3 Recycling of mixed plastics fractions

If the composition of the standard appliance presented in section 3.1 is assumed, fridge recycling will produce two other mixed plastics fractions of variable quality in addition to the PU and polystyrene fractions.

One mixed plastics fraction with a yield of about 0.7 kg per appliance, is too heterogeneous to be suitable for mechanical recycling but is pure enough to be used for energy recovery. It is assumed that this energy recovery takes place in a cement works. The relevant LCI data are described in the appendix, section 7.1.6.

A further mixed plastics fraction is made up of plastics and wood materials from initial disassembly of the appliance together with miscellaneous sorting residues from step 2 processing. This fraction, amounting to about 2.7 kg per appliance, is incinerated. The LCI data on incineration are described in the appendix, section 7.1.3.

3.7.4 Disposal of the shredder light fraction

According to the specifications in the German regulations on domestic waste (*TA Siedlungsabfall*), shredder light fraction (SLF) will no longer be landfilled from April 2007; in contrast to 2006, when about 90 % went to landfill *[EUWID 2006]*.

It is therefore assumed in this LCA study that 40 % of the SLF receives post-shredder treatment and 60 % is incinerated. The high-calorie post-shredder fraction substitutes heavy oils in blast furnaces.

Incineration and blast furnace applications are described in the appendix in sections 7.1.3 and 7.1.7.

An example of post-shredder technology is the VW-SiCon process that has only recently been brought into large-scale use at a few sites. In this process, ferrous and non-ferrous metal fractions are separated out from the input. Their total metal content is around 10 % depending on the input source. A post-shredder plant is also currently being operated by Scholz AG. There is expected to be a marked increase in post-shredder capacity in the next few years. The sensitivity calculation to model the possible fridge recycling situation in 5-10 years' time (50 % HC-containing devices) therefore assumes that 80 % of the SLF will be treated in post-shredder plants and only 20 % will be incinerated.

The following transfer rates are assumed for the metals and plastics from postshredder treatment sent for metal recycling or for energy recovery.

- Transfer rate for plastics from the SLF into fractions for energy recovery (shredder granulate and shredder fluff): 98 %. Use in blast furnaces (replaces heavy oils).
- Transfer rate for ferrous metals and non-ferrous metals from the SLF into fractions for metal recycling: 80 % for ferrous and 60 % for nonferrous metals.

As no data are available on operating emissions (PM_{10}) and fuel and electric power consumption, the specific consumption values for shredders were used.

3.8 Recycling of metal fractions

The LCI analysis assumes further processing of scrap iron and aluminium into steel and secondary aluminium respectively. Steel and aluminium are obtained as the output products from these processes.

The environmental costs associated with producing the same amount of steel and aluminium from primary sources (complementary processes) are saved and therefore credited to the system.

The processes for the treatment of scrap to produce secondary metals are described in the appendix, sections 7.1.1 and 7.1.2. The complementary processes are described in sections 7.2.1 and 7.2.2.

In a study of primary and secondary copper production processes *[RWTH-IME 1995]* it was shown that the primary energy demand for smelting and refining was of the same order of magnitude for primary and secondary copper (deviation of 6%). A precise distinction between the smelting of primary and secondary materials is not possible in view of the conditions in Germany, because the large German copper smelting plants are charged with a mixture of primary and secondary material. For these reasons, only the prior stages of mining and ore processing are credited in the recycling of copper-containing materials in the present study. It is assumed that the environmental effects of the subsequent stages – smelting and refinement – are similar for primary and secondary copper. The processes involved in copper ore processing are described in section 7.2.3.

3.9 Transport

It is assumed that fridges are transported to recycling or auto shredder facilities in 40 t articulated trucks each of which takes 140 appliances.

The following transport distances were used:

- collection centre – fridge recycling facility: 100 km
- collection centre – auto shredder plant: 50 km
- fridge recycling facility – auto shredder plant 50 km
- auto-shredder – postshredder treatment plant 200 km
- transport of scrap metal fractions for recycling: 500 km (80 % road; 20 % rail)

About 20 fridge recyclers and 40 large-scale shredder plants in Germany are certified in accordance with the national regulation on the treatment of end-of-life vehicles (*Altauto V*) (see *[BDSV 2007]*). The average transport distance to a fridge recycling facility is estimated to be about 100 km. As there are more auto shredder plants, a shorter distance of 50 km is assumed for this stage of transport.

Metal is transported considerably further because there are only a few smelting plants for iron and non-ferrous metals. An average distance of 500 km is assumed here.

Road transport is modelled according to *[ifeu/UBA 2006]*. Specific vehicle emissions assuming that 50% of the transport capacity is used (trucks return empty) are detailed in section 7.3 of the appendix. The use of transport capacity was set at the low level of 50% in order to obtain the upper limit for emissions from transport. As calculations show that transport does not have a decisive influence on the LCI results, no additional sensitivity calculations were performed.

3.10 Other processes

3.10.1 Energy and fuel

Calculations of electricity consumption are based on a typical German power station including associated upstream processes. For other energy sources and fuels and their related upstream activities, data sets relevant to Germany were used. The processes used are tabulated in section 7.4 of the appendix.

3.10.2 Landfill

Landfill is of only minor importance for material flows in the treatment of end-of-life refrigeration appliances. It is only included in the model for a minor fraction of the post-shredder residue (corresponding to 2% of the SLF input). All the CFC and cyclopentane residues present in the PU foam in post-shredder residue are released in the course of time *[Scheutz Kjeldsen 2002]*.

Apart from CFC and cyclopentane emissions, only diesel consumption and diesel engine emissions are considered in the material flow and energy balance calculations. Further gas emissions from landfill are usually so low that they are below the cutoff level. The water path is not included in the model because it does not affect any of the impact categories.

Schwing *[Schwing 1999]* gives data for diesel consumption in landfill tipping which lie between 0.5 and 2 l per tonne of waste. In the present study an average diesel consumption of 1.5 l/t is assumed.

4 Impact assessment results

4.1 Overview

The table below gives an overview of the results of impact assessments for all seven impact categories considered. Positive values indicate adverse environmental impact while negative values indicate favourable environmental impact. Most of the values are negative, indicating that the credits from the recycling processes outweigh the adverse environmental effects. The best variant from an environmental point of view is joint processing (variant 1) and the associated values are shown on a dark grey background. The worst variant in each impact category is shown against a light grey background.

The following table gives an overview of the results of the impact assessments for all seven impact categories considered. Positive values indicate adverse environmental impact while negative values indicate favourable environmental impact. In the latter case, the credits from the recycling processes outweigh the adverse environmental effects. The best variant from an environmental point of view is joint processing (variant 1) and the values are shown on a dark grey background. The worst variant in each impact category is shown against a light grey background.

Table 4-1 Results of impact assessment (absolute values; proportion of HC-containing appliances: 20 %)

Variant		1 Joint processing	2 Parallel processing	3 HC appliances: Step 2 in shredder	4 HC appliances: Steps 1+2 in shredder
Global warming potential	1000 t CO_2 eq per year	-193	-169	-155	-128
Ozone depletion potential	kg R11 eq per year	1,207	4,116	6,573	8,609
Photochemical ozone creation potential	kg ethylene eq per year	-15,032	3,828	28,221	38,035
Acidification	t SO_2 eq / year	-967	-959	-947	-948
Eutrophication	t PO_4 eq / year	-62	-62	-60.3	-60.9
Particulate matter	t PM_{10} eq / year	-1035	-1,027	-1,013	-1015
Cumulative energy demand (CED)	PJ	-2.64	-2.63	-2.60	-2.60

Table 4-2 shows normalized results. The absolute positive and negative environmental impacts are considered in relation to the overall impact for the whole of Germany.

Table 4-2 Impact assessment results as a percentage of the total impact in Germany (proportion of HC-containing appliances: 20 %)

Variant	1 Joint processing	2 Parallel processing	3 HC appliances: Step 2 in shredder	4 HC appliances: Steps 1+2 in shredder
Global warming potential	**-0.019**	**-0,017**	**-0.015**	**-0.013**
Ozone depletion potential*	**0.40**	**1.35**	**2.15**	**2.82**
Photochemical ozone creation potential	**-0.002**	**0.001**	**0.004**	**0.006**
Acidification	**-0.040**	**-0.040**	**-0.0392**	**-0.0392**
Eutrophication	**-0.0157**	**-0.0155**	**-0.0152**	**-0.0154**
Particulate matter	**-0.0457**	**-0.0454**	**-0.0448**	**-0.0449**
Cumulative energy demand (CED)	**-0.0184**	**-0.0183**	**-0.0181**	**-0.0182**

* Normalized relative to recently introduced ozone depleting substances

The tables show that the results for the impact categories ‘ozone destruction potential’, ‘global warming potential’ and ‘photochemical ozone creation potential’ differ markedly from one another and are therefore considered in more depth in the following sections.

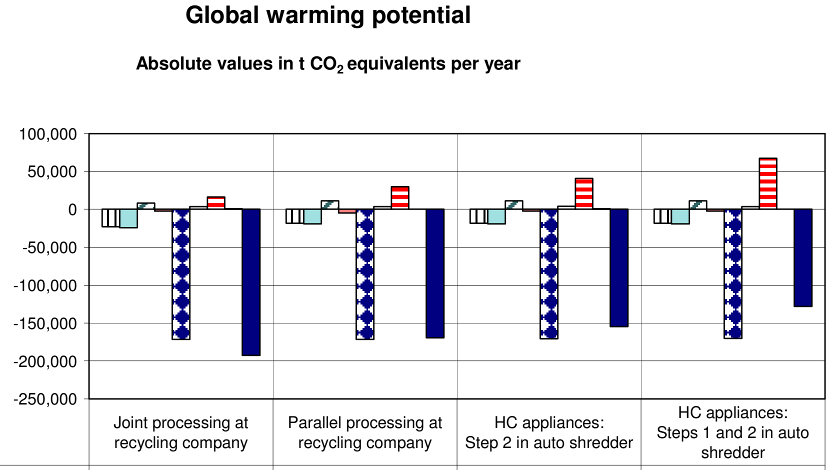
Only minor differences (between 2 and 4 %) are observed between the variants for the impact categories ‘acidification’, ‘eutrophication’, ‘particulate matter’ and ‘cumulative energy demand’. In view of the general level of uncertainty of the data, such small deviations cannot be taken as clear indications of an environmental advantage or disadvantage associated with a particular variant. These impact categories are therefore not considered in greater detail.

4.2 Global warming potential

The main factors responsible for the global warming potential of a particular processing variant are energy-related CO_2 emissions and CFC emissions. Because of the credits from recycling processes, the overall result is a favourable environmental impact representing between 0.013 and 0.019 % of Germany’s total greenhouse gas emissions. The annual benefit is between 128,000 and 193,000 tonnes CO_2 equivalent.

The best variant from an ecological point of view is the joint processing of CFC-containing and HC-containing appliances. The worst is variant 4 in which appliances containing HCs are disposed of entirely in an auto shredder.

The diagram below shows the extent to which the different processes contribute to the result.



	Joint processing at recycling company	Parallel processing at recycling company	HC appliances: Step 2 in auto shredder	HC appliances: Steps 1 and 2 in auto shredder
▥ Mechanical recycling of PS	-22,993	-18,211	-18,211	-18,211
□ PU as absorbent/binder	-24,451	-19,365	-19,365	-19,365
▨ Incineration of plastics/SLF	8,379	10,986	10,965	10,913
□ Energy recovery plastics/SLF	-2,607	-4,824	-2,450	-2,473
◆ Metal recycling	-171,493	-171,493	-170,562	-170,250
□ Transport	3,462	3,462	3,971	3,355
▤ Emissions from plant	16,144	29,774	40,619	67,393
□ Other	683	254	499	282
■ Total	-192,876	-169,418	-154,534	-128,356

Figure 4.1 Results for the global warming potential and contributions of specific processes (proportion of HC-containing appliances: 20 %)

Results for each of the four variants are shown in the separate sections of the diagram. For each variant the (solid dark blue) right-hand bar shows the sum total or overall result and the eight bars to its left represent the contributions from individual processes.

It is apparent that metal recycling makes the greatest contribution to the favourable environmental impact[33]. Because the extent of metal recycling is similar in all four variants, distinctions between the variants are due primarily to the following other processes:

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[33] The favourable environmental impact shown is a net impact, i.e. the environmental costs resulting from the recycling of secondary metals minus the credits for substitution of primary metals.

Plant emissions

The greatest differences with respect to greenhouse activity stem from the emissions from the fridge recycling plants, shredders and post-shredders, and the CFC emissions and hydrocarbon emissions from the relevant output streams (emissions from the post-processing of polyurethane and the CFC-containing refrigerator oil). The crucial factors determining the differences in global warming potential are the emissions of the CFCs R11 and R12. Other emissions relevant to the global warming potential, particularly CO_2 emissions, do not differ significantly. The figure below shows the levels of CFC emissions. The calculations are shown in detail in the appendix (see table 7-7).

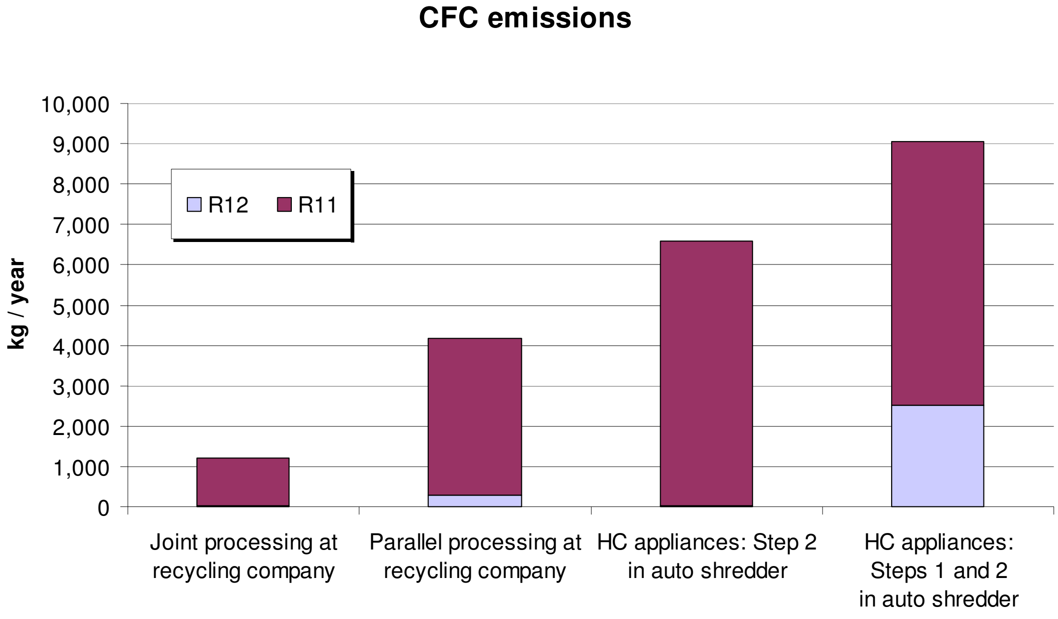

Figure 4.2 CFC emissions for each variant (proportion of HC-containing appliances: 20 %)

Joint processing (variant 1) yields emissions of about 1,200 kg CFCs per year. These consist mainly of R11 emissions from polyurethane residues adhering to metal and plastic scrap and residual R11 still present in the degassed foam, small proportions of which are emitted in the course of further treatment (storage, transport, use as an absorbent and subsequent incineration). Calculations assume that the model recycling plant works to high environmental standards (see section 3.2).

In **parallel processing** (variant 2) CFC emissions are the same as those in joint processing. However, substantial emissions resulting from the missorting of appliances must be added. It was assumed that sorting takes place before step 1 and that 1 % of CFC-containing appliances are allocated to the HC line by mistake. The R12 present in these appliances is flared on site together with isobutane. As combustion in the flare does not take place at a high-temperature and is not designed for treating R12, it must be assumed that the R12 following this route is not completely destroyed (see life cycle inventory information in section 3.5). The R11 from the missorted appliances is released on shredding and is emitted together with the cyclopentane.

In the **step 2 treatment of HC appliances in an auto shredder** (variant 3) the CFC emissions are higher than in parallel processing. As well as the emissions from normal treatment of CFC-containing appliances there are also high levels of R11 emissions from shredding missorted CFC-containing appliances. These are higher than in parallel processing because the HC line of a parallel processing set-up is designed to produce larger chunks of PU than in an auto shredder in order to reduce the amount of blowing agent released from the PU foam (see explanation in section 3.5.2).

In the **complete processing of HC-containing appliances by shredding** (variant 4), the emissions of R11 from shredding the appliance carcass are the same as those for variant 3 (step 2 processing of HC-containing appliances occurs in an auto shredder). However, additional R12 emissions occur as a result of incorrect treatment of missorted CFC-containing appliances in step 1. The diagram above consequently shows high levels of R12 emissions that result from the release of R12 from their cooling circuits on shredding.

Joint processing is the best variant as far as plant operating emissions are concerned and was found to be more favourable by 13,000–51,000 t CO_2 equivalent per year than the other variants.

Recycling of the PU fraction

Like the CFC lines of the other variants, joint processing produces PU powder that is used as an absorbent/binder. It is assumed in this LCA that PU powder can be substituted for sawdust at a ratio of 1 : 3. In contrast, parallel processing produces PU chunks that are incinerated. In the variants involving shredding, the PU becomes part of the shredder light fraction and is treated in various ways. Baseline calculations assume that 60 % is incinerated as waste and 40 % receives post-shredder treatment prior to energy recovery in a blast furnace. A sensitivity analysis considers the situation in 5-10 years when 50 % of waste refrigeration appliances will then contain HCs and post-shredder treatment will have increased to 80 %. In terms of global warming potential, recycling of PU as an absorber or binder is the best option. This option is better than parallel processing, where the PU is incinerated, by about 8,000 t CO_2 equivalent when the sums of the two subprocesses ('PU used as absorbent/binder' and 'plastics → waste incineration plant') are compared. The advantages are even greater when variant 1 is compared to the 'shredder variants' 3 and 4. In these variants, most of the PU ends up in the SLF that is then used as a substitute for heavy oils in blast furnaces. The amounts of CO_2 emitted during the burning of oil and PU are similar.

Recycling of the polystyrene fraction

Joint processing and the CFC lines of the other variants produce a very pure polystyrene fraction that is mechanically recycled. The left-hand bars in Fig. 4-1 show the relevant credits in each case. The model assumes that the polystyrene from the HC line in the parallel processing variant is used for energy recovery in a cement works[34]. Because use in a cement works brings less credit than mechanical recycling, joint processing is better than parallel processing by about 3,000 t CO_2 equivalent per year as a result of the more advantageous recycling of polystyrene (comparison of sub-processes 'mechanical recycling of polystyrene' and 'energy recovery from plastics'). If the polystyrene from parallel processing was treated in a waste incinerator there would be a net deficit for polystyrene disposal that would further worsen the overall result for the parallel processing variant.

The advantages are even greater when variant 1 is compared to the 'shredder variants' 3 and 4. In these variants, most of the PU ends up in the SLF that is then used as a substitute for heavy oils in blast furnaces. The amounts of CO_2 emitted during the burning of oil and PU are similar.

Other processes

Waste incineration produces an overall burden in terms of global warming potential because the resulting CO_2 emissions are higher than those that are saved by substituting for electricity and district heating produced by conventional means[35]. The amount of plastics incinerated is smallest in joint processing. In parallel processing and in the shredder variants more plastics are incinerated, in the form of PU chunks and SLF.

Transport only contributes 2–3 % of the result in this impact category.

High-temperature combustion, oil recycling and glass recycling make relatively small positive or negative contributions (less than 1 %) to the global warming potential.

4.3 Ozone depletion potential

The ozone depletion potential depends entirely on the R11 and R12 emissions. The figure below shows the ozone depletion potential for each variant. The left-hand and central bars show the ozone depletion potential for each variant resulting from

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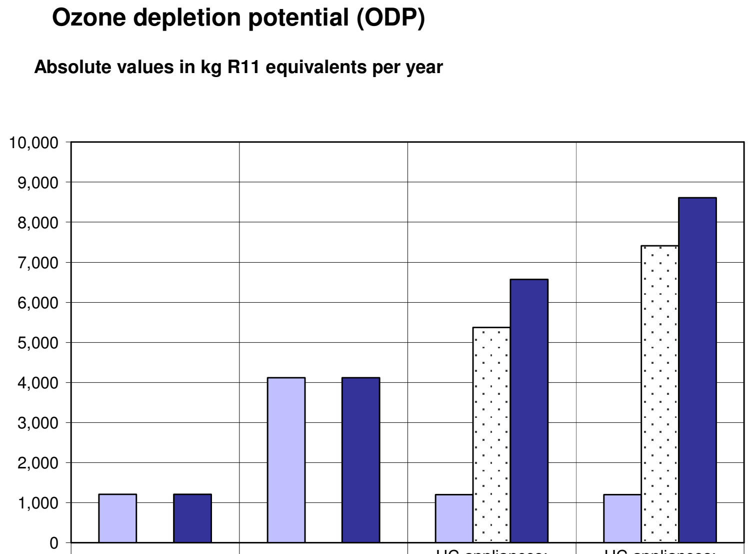
[34] The production of a high-purity polystyrene fraction in the HC line of the parallel processing variant and its mechanical recycling is considered in the sensitivity analysis discussed in section 4.5.6.

[35] The environmental burden shown is a net burden, i.e. weighted emissions from the combustion of fuels and from their related upstream processes minus credits for the electricity and heat that are replaced.

treatment in a fridge recycling plant and in a shredder respectively, including emissions associated with downstream processing. The right-hand bar shows the sum total ODP for each variant.



	Joint processing at recycling company	Parallel processing at recycling company	HC appliances: Step 2 in auto shredder	HC appliances: Steps 1 and 2 in auto shredder
□ Fridge recycling plant + downstream emissions	1,207	4,116	1,196	1,195
⬚ Auto shredder + downstream emissions	0	0	5,377	7,414
■ Total	1,207	4,116	6,573	8,609

Figure 4.3 Ozone depletion potentials and contributions from specific processes (proportion of HC-containing appliances: 20 %)

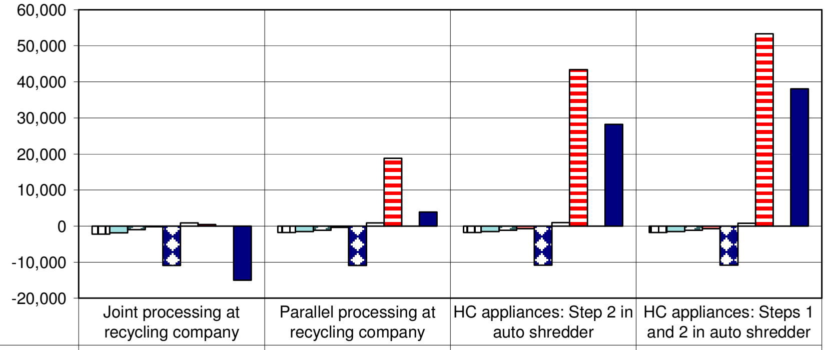
The origins of the CFC emissions have already been described in section 4.2 on the global warming potential. It can be concluded that joint processing has the lowest ozone depletion potential (ODP) with about 1,200 kg R11 equivalents per year. The ODP for parallel processing and for variant 3 (HC-containing appliances: step 2 processing in auto shredder) is considerably higher, at around ,4000–6,500 kg R11 equivalent/year. At 8,600 kg R11 equivalent/year, the ODP for variant 4 (HC-containing appliances treated entirely in an auto shredder) is higher still. The differences are almost entirely due to emissions from missorted CFC-containing appliances.

The different processing variants account for between 0.4 % (variant 1: joint processing) and 2.8 % (variant 4: HC appliances processed entirely in an auto shredder) of the emission potential associated with recently introduced ozone depleting substances. Relative to Germany's overall ODP burden (i.e. old emissions and potential new emissions), the values lie between 0.04 and 0.08 % for variant 1 and 0.3 and 0.6 % for variant 4. Greater precision is not possible because only rough quantitative estimates can be made regarding old emissions (see explanations in section 2.8.2.2).

4.4 Photochemical ozone creation potential (POCP)

The results for the impact category POCP are shown in the diagram below.

Photochemical ozone creation potential
Absolute values in kg ethylene equivalents per year



	Joint processing at recycling company	Parallel processing at recycling company	HC appliances: Step 2 in auto shredder	HC appliances: Steps 1 and 2 in auto shredder
◫ Mechanical recycling of PS	-2,254	-1,785	-1,785	-1,785
☐ PU as absorbent/binder	-1,893	-1,499	-1,499	-1,499
◪ Incineration of plastics/SLF	-1,002	-1,186	-1,185	-1,179
☐ Energy recovery plastics/SLF	-173	-320	-721	-735
☐ Metal recycling	-10,958	-10,958	-10,901	-10,878
☐ Transport	868	868	995	841
▤ Emissions from plant	437	18,743	43,362	53,305
☐ Other	-57	-35	-45	-34
■ Total	-15,032	3,828	28,221	38,035

Figure 4.4 Results for photochemical ozone creation potential and contributions from specific processes (proportion of HC-containing appliances: 20 %)

This impact category is dominated by the environmental benefits from metal recycling and plant emissions. However, the credits from metal recycling are very similar for all the four variants so that the observed differences between the variants are essentially due to differences in plant emission levels.

Emissions of isobutane and cyclopentane are of prime importance here. The calculations are shown in detail in the appendix (see table 7-7).

In joint processing the plant emissions are so low that, when they are set against the credits resulting from metal recycling, an overall environmental benefit of just over

15,000 t ethylene equivalent per year results (proportion of total for Germany: 0.002 %).

In parallel processing and variant 3 (step 2 treatment of HC appliances in auto shredder) the cyclopentane released during shredding is emitted, making weighted plant emissions higher than the credits from metal recycling. The overall result for these variants is therefore an environmental burden of about 4,000–28,000 t ethylene equivalent per year (proportion of overall burden for Germany: 0.001–0.004 %).

Variant 4 (HC appliances treated entirely in auto shredder) results in an even greater environmental burden amounting to some 38,000 t ethylene equivalent per year. This represents about 0.006 % of the overall burden for Germany. In this variant the cyclopentane emissions are supplemented by the isobutane released from the appliance cooling circuits.

Although these values represent only a very small fraction of Germany's total POCP burden, it is important to realize, when interpreting these results, that summer smog formation is a local and temporary process. Hence even small quantities of photochemical ozone precursors can make a significant contribution to local ground-level ozone formation for a limited period of time. This means that even low levels of hydrocarbon emissions should be avoided if at all possible.

4.5 Sensitivity analyses

4.5.1 Proportion of HC-containing appliances: 50 %

A sensitivity analysis was performed in which the proportion of HC-containing appliances was taken to be 50 % rather than 20 %. This reflects the situation expected in 5 to 10 years' time. The sensitivity calculation also assumes changes in the processing of the shredder light fraction. It is likely that post-shredder treatment will be widely used in 5 to 10 years. The model therefore assumes that 80 % of the shredder light fraction receives post-shredder treatment and is subsequently recycled while 20 % is incinerated.

The results for all impact categories are set out in the table below. To facilitate comparison, the results of the baseline calculation (20 % HC-containing appliances) are given in brackets.

Table 4-3 Results of sensitivity analysis assuming 50 % HC-containing appliances (values in brackets relate to 20 % HC-containing appliances)

Variant		**1 Joint processing**	**2 Parallel processing**	**3 HC appliances: Step 2 in shredder**	**4 HC appliances: Steps 1+2 in shredder**
Global warming potential	**1000 t CO_2 eq per year**	**-195**	**-162**	**-156**	**-141**
		-(193)	**-(169)**	**-(155)**	**-(128)**
Ozone depletion potential	**kg R11 eq per year**	**755**	**2,573**	**4,118**	**5,391**
		(1,207)	**(4,116)**	**(6,573)**	**(8,609)**
Photochemical ozone creation potential	**kg C_2H_4 eq per year**	**-15,040**	**32,179**	**92,666**	**117,137**
		-(15,032)	**(3,828)**	**(28,221)**	**(38,035)**
Acidification	**t SO_2 eq per year**	**-968**	**-943**	**-936**	**-944**
		-(967)	**-(959)**	**-(947)**	**-(948)**
Eutrophication	**t PO_4eq per year**	**-62**	**-60**	**-59**	**-60**
		-(62)	**-(62)**	**-(60)**	**-(61)**
Particulate matter	**t PM_{10}eq per year**	**-1,035**	**-1,012**	**-1,000**	**-1,008**
		-(1,035)	**-(1,027)**	**-(1,013)**	**-(1,015)**
Cumulative energy demand (CED)	**PJ**	**-2.64**	**-2.58**	**-2.67**	**-2.69**
		-(2.64)	**-(2.63)**	**-(2.60)**	**-(2.60)**

The table shows that the underlying ranking of the variants remains unchanged under the conditions chosen for the sensitivity analysis. It also shows that only the impact categories ‘ozone depletion potential’ and ‘photochemical ozone creation potential’ differ significantly in the baseline and sensitivity analysis conditions. Both of these impact categories are influenced primarily by the material flows for CFCs and for isobutane and cyclopentane respectively. This means that when the proportion of CFC-containing appliances is reduced from 80 % to 50 %, the ozone depletion potential falls by a factor of 0.6 while the impact category ‘photochemical ozone creation potential' increases across variants 2 to 4 by a factor of 3 to 8. The latter effect is mainly due to an increase in plant emissions.

Diagrams are included in the appendix (section 8.1) giving details of sensitivity analyses for the results in the impact categories 'global warming potential', 'photochemical ozone creation potential' and 'ozone depletion potential'. Calculations for emissions of CFCs, isobutane and cyclopentane are presented in section 7.6.

4.5.2 Destruction of CFCs in the recycling process

Section 3.5.3 considers the extent of destruction of R11 and R12 during the following treatment processes: energy recovery from refrigerator oil, burn off in an isobutane flare, incineration in a waste incinerator, incineration in a cement works, incineration in a blast furnace, and metal recycling. Because of the high level of uncertainty associated with the available data, two sensitivity analyses were performed, one using a lower and one a higher rate of destruction. The following destruction rates were assumed in the sensitivity analysis for the thermal processes including upstream logistics (i.e. prior transport and storage).

- CFC destruction rates during energy recovery from refrigerator oil: 95% and 99.5% (baseline calc.: 98%)
- CFC destruction in an isobutane flare: 80% and 99% (baseline calc.: 90%)
- Destruction of CFCs from degassed milled PU during energy recovery following use as an absorbent/binder: 95% and 99.5% (baseline calc.: 98%)
- Destruction of CFCs from PU chunks in incinerators, cement works, blast furnaces and metal recycling: 80% and 99% (baseline calc.: 90%)

The figure below shows the results obtained for the four variants.

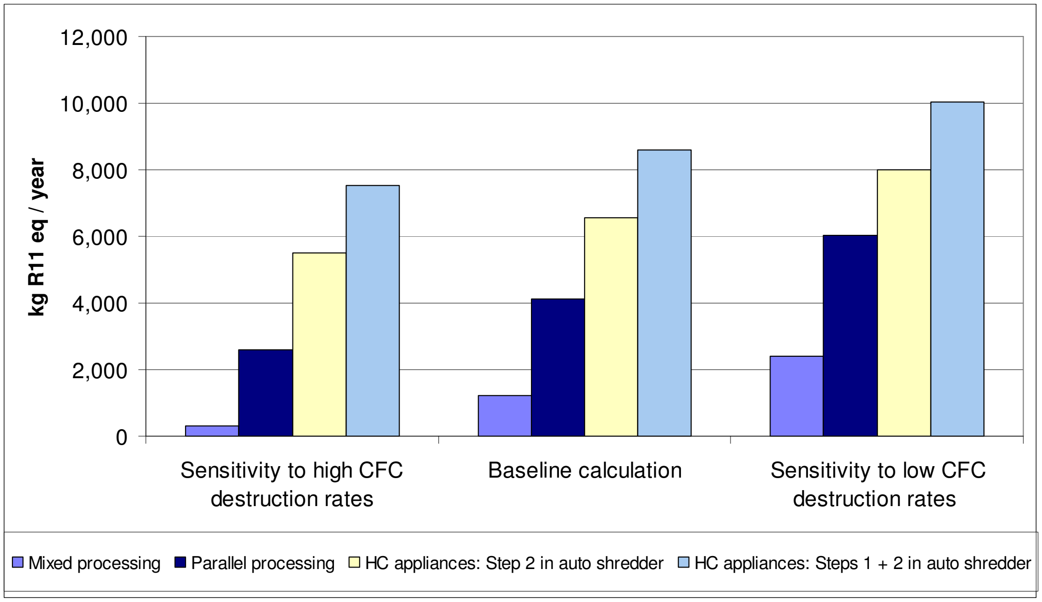


Figure 4.5 Ozone depletion potentials for sensitivity analyses assuming CFC destruction rates in the downstream treatment processes that are lower or higher than the rates assumed in the baseline calculations.

The diagram shows that the underlying differences between the variants do not change significantly when different assumptions are made regarding CFC destruction rates. On average, higher destruction rates led to a reduction in the ODP of 1,110 kg R11 equivalent per year compared to baseline levels, while lower destruction rates led to an increase in ODP of 1,400 kg R11 equivalent per year.

4.5.3 High-temperature cracking

A sensitivity calculation was also performed for the treatment of the CFCs collected in parallel processing. It was assumed that 50 % of the CFCs were treated by high-temperature combustion and 50 % by high-temperature cracking, rather than being treated entirely by high-temperature combustion. The following table shows the results for the impact categories 'global warming potential' and 'photochemical ozone creation potential'.

Table 4-4 Results for treatment of CFCs in parallel processing assuming 100 % high-temperature combustion and a combination of 50 % high-temperature combustion and 50 % high-temperature cracking.

	100% H-T combustion	**50% H-T combustion / 50% H-T cracking**
GWP	-169,418	-170,325
POCP	3,828	3,788

As the table shows, the differences in the overall result are small. The option including high-temperature cracking is slightly better because HCl and HF are recovered. This result is based on average assumptions for high-temperature combustion and high-temperature cracking (see sections 7.1.4 and 7.1.5). The results are so similar that they cannot be used to make a clear environmental distinction between two real fridge recycling plants. To answer that sort of question, exact local conditions (e.g. power and fuel consumption, electricity and heat extraction, HCl and HF output yields) would need to be established and used in the corresponding calculations.

4.5.4 Missorting of CFC-containing appliances

The results already presented have shown that the ozone depletion potential plays an extremely significant role in the LCA. Firstly, ozone depletion potential is the impact category in which emissions from fridge recycling activities make the greatest relative contribution to current emission levels in Germany. The significance of ODP-relevant CFC emissions from fridge recycling is made particularly clear in calculations that show that these emissions can constitute up to 3 % of Germany's emission potential associated with recently introduced ozone depleting substances. Secondly, the differences in ODP between the variants are very high. The ODP for variant 4 is about seven times as high as that for variant 1.

The most crucial parameter for this result is the missorting rate for CFC-containing appliances. This is described in detail in section 3.4. A sorting error rate of 1 % was initially assumed for the LCA. This rate is considered to represent a lower limit that can only be achieved when every effort is made to minimize the missorting of CFC-containing appliances. Higher sorting error rates appear more realistic. A rate of 5 % was also considered in variants 2 to 4 with up to 10 % considered for variant 4 in which sorting takes place at local waste-collection depots (see explanations in section 3.4). The following diagram shows the results of these sensitivity analyses.

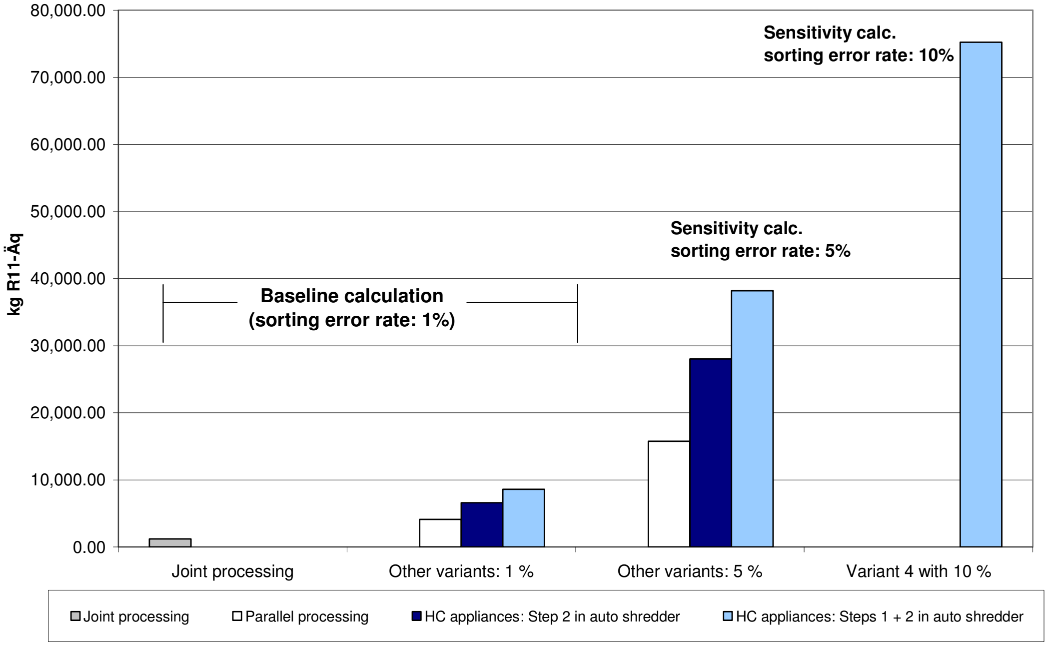


Figure 4.6 Ozone depletion potentials from sensitivity calculations assuming different sorting error rates

The diagram also shows that the ODP increases dramatically at higher sorting error rates. In variant 4 the ODP rises to 38,000 and 75,000 kg R11 equivalent per year when the sorting error rate is increased from 5 to 10 %. These values are 32 and 62 times that for joint processing. They represent up to 25 % of Germany's ODP (relative to the emission potential of recently introduced ozone depleting substances) or up to 3 % of the national ODP (relative to total emissions, i.e. previous emissions and potential new emissions).

4.5.5 Missorting of HC-containing appliances

It is also to be expected that some of the HC-containing appliances will be missorted. However, these errors are only of minor importance in the LCA and have no effect whatsoever on the ozone depletion potential, the most significant impact category. Missorting of HC appliances would in fact lead to a slight improvement for the impact categories 'photochemical ozone creation potential' and 'global warming potential' in variants 2–4 as these appliances would be processed in the CFC line in which a somewhat higher quality of mechanical recycling is achieved. In addition, because these missorted appliances are processed in the CFC line, the HCs they contain would not be released during shredding (variants 3 and 4) or parallel processing (variant 2) but would be destroyed together with the CFCs. However the rank order of the variants and the absolute order of magnitude of the result would not change.

4.5.6 Mechanical recycling of polystyrene in parallel processing

It was assumed for the baseline calculations in this LCA that the HC line of the parallel processing variant doses not involve any complex material separation and that the polystyrene produced is therefore not sufficiently pure for mechanical recycling. A sensitivity analysis was carried out in which a complex separation process was also included in the HC line of parallel processing so that the polystyrene produced is suitable for mechanical recycling. This would have the following effects on the result of the LCA:

- No change in the ozone depletion potential – the most significant impact category when comparing the four processing variants.
- The impact categories ‘global warming potential’ and ‘photochemical ozone creation potential’ would improve slightly for variant 2 (parallel processing). However the order of magnitude of the results would remain unchanged.

5 Conclusion

The irrefutable conclusion drawn from this life cycle assessment study is that variant 1 (i.e. the treatment of all types of waste refrigeration appliance in a single recycling plant) is the most environmentally friendly treatment process for end-of-life refrigeration equipment. That a single process came out on top for all the evaluation criteria used is an extremely rare occurrence in a life cycle assessment. Comparing the variant treatments is in this case a simple matter as there is no need to carry out the usually challenging task of weighing up a number of different environmental criteria against each other.

Furthermore, when the variant processes are compared in terms of the most important evaluation criteria, highly relevant differences become apparent. For example, variant 4 achieves only about 66 % of the savings in CO_2 equivalents that can be achieved with variant 1. Using variant 1 can save about 0.02 % of Germany's total emissions of greenhouse gases. Compared to variants 2 to 4, variant 1 saves an additional 24,000 to 65,000 tonnes of CO_2 equivalents annually. As climate protection is regarded as a particularly important and urgent environmental issue, all measures that contribute to achieving reductions in greenhouse gas emissions of this magnitude are significant. This is particularly the case when, as here, all other environmental compartments profit as well.

The disparity between the variant processing methods is even more apparent when the highly significant environmental criterion 'ozone depletion potential' is considered. The difference between variant 1 and variants 2 to 4 is a factor of 3 to 7. Furthermore, the latter three variants 2 to 4 make up to 3 % of the current emission potential from recently introduced ozone depleting substances in Germany, a significantly higher contribution than that from variant 1. The most important underlying cause is the faulty sorting of CFC-containing appliances in variants 2 to 4. It is difficult to specify an accurate figure for the assumed error sorting rate. The sorting error of 1 % used in the present life cycle assessment was deliberately chosen at the low end of the range of sorting error rates derived from an analysis of available data. In fact, this low level of sorting error can only be achieved if all conceivable measures are taken to minimize sorting errors.

The sensitivity of the results to assuming a higher sorting error was also analysed. A higher sorting error is to be expected unless very elaborate control and feedback mechanisms are in place and every effort is made to eliminate known sources of error. Many waste refrigeration appliances are insufficiently clearly labelled. Recycling plant operators estimate that between 20 and 30 % of all end-of-life fridges and freezers are inadequately labelled. If an error sorting rate of 5 % is assumed, the ozone depletion potential for variant 4 is about 32 times greater than that for joint processing. An additional sensitivity calculation was performed for variant 4 with an even higher sorting error rate of 10 % as in this variant the refrigeration devices are sorted at local waste-collection centres. If the staff at the numerous waste-collection centres are not suitably trained, or if the sorting is left to members of the public, a sorting error rate of this magnitude is realistic. In this case the ozone depletion potential would be 62 times higher than in the joint processing variant and would represent about 25 % of Germany's current emission potential associated with recently introduced ozone depleting substances.

If photochemical ozone creation is considered, variants 2 to 4 are in complete contrast to the results for variant 1. While the recycling of waste refrigeration appliances using variant 1 results in a net environmental benefit, all other variants lead to a net environmental burden through the production of summer smog precursors. The fractional contribution to total emissions of photochemical ozone precursors in Germany is relatively low (< 006 %). However, it is important to recall that summer smog formation is a local and temporary process. Hence even small quantities of photochemical ozone precursors can make a significant contribution to local, ground-level ozone formation for a limited period of time. It is therefore imperative that all avoidable hydrocarbon emissions are eliminated.

The differences between the variants with respect to acidification, eutrophication, PM_{10} emissions and consumption of energetic resources (expressed in terms of the cumulative energy demand) are so small that within the precision achievable in a life cycle assessment the results can be treated as effectively equal.

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7 Appendix A: Life cycle inventory information

7.1 Recycling and treatment processes

7.1.1 Production of steel from scrap iron

7.1.2 Production of secondary aluminium

In this LCA it is assumed that the separated aluminium scrap is sent for secondary smelting. Calculations were performed using the GEMIS process 'Metall/Alumin-DE-sekundär' which models the smelting process with an output of 910 kg aluminium per 1000 kg of aluminium scrap. It takes account of energy input, consumption of fuel and auxiliary materials and the major upstream processes.

7.1.3 Waste incineration

A study performed in 2002 evaluated data from a large number of German waste incineration plants regarding emissions, use of fuels and auxiliary materials and energy balance *[Öko-Institut 2002]*. Mean values were obtained from these data. As the quality of data available for N_2O and NH_3, was poor, estimates were made based on values reported in the literature. The mean or estimated values for an average German waste incineration plant are listed below.

Table 7-1 Emissions assumed for waste incineration

Parameter	Mean value[mg/m^3]
N_2O	2.0
Particulates	1.11
PM_{10}*	1.05
HCl	1.97
HF	0.11
SO_2	6.19
NOx	94.09
NH_3	0.5

* It is assumed that PM_{10} makes up 95 % of all particulate matter.

The following mean values were assumed for materials used in flue gas scrubbing (source: *[Öko-Institut 2002]*).

Table 7-2 Consumption of auxiliary materials used in the waste incineration process

Parameter	Mean value [kg/Mg waste]
Activated lignite	5.01
Ammonia solution	4.6
NaOH	stoichiometric quantity depending on chlorine content of waste

Auxiliary energy sources, such as heating oil or natural gas, are not included in calculations for the present case because the plastics fractions involved have a much higher energy content than average waste incinerator input.

The treatment of residues from waste incineration (fly ash and bottom ash) is not considered in this LCA. Firstly, only a very small amount of ash is produced because the plastics contain no inert matter and only the inorganic impurities produce ash. Secondly, residues from waste incineration play only a minor role in the impact categories considered (energy consumption for transport, tipping and backfill) and therefore lie below the cutoff limit.

The table below presents the average utilization levels for the energy recovered from waste incineration as published in *[Öko-Institut 2002]*. These values represent net efficiencies for the production of electricity, district heating and process heat and were calculated by subtracting from the energy content of the waste material input the energy losses and the energy consumed by the waste incineration plants themselves.

Table 7-3 Net utilization levels for the energy recovered during waste incineration

Parameter	Mean value
Electricity	10 %
District heating	19 %
Process steam	4 %

It is assumed for the LCA that the energy extracted from waste incineration can be substituted for energy supply processes – the complementary processes are shown in the following table.

Table 7-4 Complementary processes assumed for energy extraction from waste incineration

Energy source	Complementary process
Electricity	Coal-fired condensation power station (GEMIS process 'Strom-bonus-el-Stk-DE-2000')
District heating	Gas-fired heating plant (GEMIS process 'Gas-HW-mittel-DE-2000')
Process steam	Industrial gas boiler (GEMIS process 'Gas-Kessel-DE-2000')

When used to produce electricity, waste incineration plants mainly generate baseload power and they thus compete with the typical baseload providers: hydroelectric, nuclear and coal-fired power stations. Substituting electricity from nuclear or hydroelectric plants with electricity from waste incineration is nevertheless not a viable option because of the low variable costs of these generation methods. Lignite-fired power stations either operate with very low variable costs or are bound by long-term contracts so that substitution of electricity from waste incineration is also unlikely in their case. In the short and medium term, therefore, electricity produced from waste incineration can only substitute for electric power generated in coal-fired power stations that burn imported coal.

For heat production, the reference process was gas heating plants. It is assumed that the additional provision of district heating power from waste incineration substitutes primarily for gas heating plants or leads to an increase in the density of existing district heating networks with a consequent increase in sales. Any increase in network density is likely to occur in conurbations, where natural gas is currently the primary heating fuel. In the production of process steam it is assumed that heat from waste incineration substitutes for process heat from a gas-fired boiler as gas now also dominates in this sector, with the use of oil falling sharply.

7.1.4 High-temperature combustion

It is assumed that high-temperature combustion takes place primarily in incinerators burning hazardous waste and that some high-energy waste is mixed in so that additional supplementary firing is not necessary. It is also assumed that the specific emissions and the consumption of auxiliary materials are identical to those in domestic waste incineration (see section 7.1.3). The same level of electricity production is assumed as in domestic waste incineration. However, less of the energy recovered in high-temperature combustion is utilized for district heating than is the case in domestic waste incineration. This lower level of 10% is assumed because (high-temperature) hazardous-waste incinerators are often in locations where there is no or only low demand for heat.

7.1.5 High-temperature cracking

Company data are used to model energy consumption and the output of HCl and HF in high-temperature cracking. The appropriate credits are given for the HCl and HF output fractions. Energy and material balance calculations for HCl are made using the Boustead database *[Boustead 2005]*. The calculations for HF use the Ecoinvent dataset 283 *[Ecoinvent 2006]*.

7.1.6 Energy recovery in cement kilns

The plastic fractions combusted in cement kilns are assumed to serve as a substitute for coal.

In terms of direct emissions of substances considered in this LCA, with the exception of CO_2, (i.e. NO_x, SO_2, N_2O, CH_4, HCl, HF, NH_3, NMVOC, PM_{10}) it is assumed that there are no relevant differences to when coal is burnt.

According to figures from the German cement industry, emissions of acid air pollutants and nitrogen oxides are independent of the type of fuel used *[Öko-Institut 2001]* and depend primarily on the way the process is operated. For example, at process temperatures in rotary cement kilns of about 2000 °C, NO_x emissions are very high and depend on various factors such as temperature, excess air and flame geometry. Sulphur is almost entirely incorporated into the clinker. The fluorides brought into cement kilns with the alternative fuels (i.e. combustible waste) are converted into inorganic fluoride compounds and neutralized by the alkaline rawmix. About 88–98 % of the fluorides become incorporated into the clinker *[UBA Österreich 1995]*. Chlorides are also neutralized but are hardly incorporated into the clinker. They react to form volatile alkali metal and heavy metal salts that condense in the cold part of the furnace. Because of the alkaline environment in cement kilns, emissions of HCl and HF are negligible and are therefore neglected in the LCA. It is also assumed that the extent of combustion and particle production when burning plastics and coal are of similar orders of magnitude and that therefore the two fuels do not differ with respect to emissions of CH_4, NH_3, NMVOC and PM_{10}.

CO_2 emissions are calculated on the basis of an elemental analysis.

Coal substitution is calculated using the GEMIS dataset 'Kohle-mix-Import-DE-2000'.

7.1.7 Energy recovery in blast furnaces

Where SLF is used as a substitute for heavy oil in blast furnaces it is assumed that the amount of oil replaced is that with the same energy content as the plastics. It is also assumed that the emissions from the blast furnace do not change as a result of the change in input fuel.

Differences between burning SLF and burning heavy oil in blast furnaces therefore only arise from the upstream processes associated with the extraction and supply of heavy oil. Calculations for the substitution of heavy oil were performed according to the GEMIS dataset 'Raffinerie\Öl-schwer-DE-2000'.

CO_2 emissions were calculated on the basis of an elemental analysis.

7.1.8 Auto shredders

In general, shredder plants are divided into shredders, zerdirators, kondirators and mills. These types differ in the details of the shredding technology used *[AK 16 1995]* but all are included under the heading 'shredder plants'. The central elements of a shredder plant are the hammer mills in which the input material is torn into pieces by rotating hammers that strike against an anvil-like breaker plate. The material stays in the shredder until the pieces are sufficiently small to pass through a screen and be removed. A large proportion of the light fraction is sucked out of the shredder during shredding with the aid of dust extraction equipment. Shredded metal and the remaining shredder residues are transported to a separating drum or air classifier. An electromagnetic drum separator then separates the scrap steel from the non-ferrous metals and coarse impurities such as plastics and rubber *[AK 2 1990]*. Shredder plants produce three output streams:

- shredder scrap
- shredder heavy fraction (contains non-ferrous metals),
- shredder light fraction (including shredder dust).

The relative amounts of these three output fractions vary depending on the composition of the input material and the operating mode of the plant. For instance, the purity of the shredder scrap can be increased by double shredding to an iron content of 99%. However, this causes increased amounts of iron to be discharged in the SLF (decreasing yield of increasing purity, *[AK 16 1995]*). Shredder scrap is high-quality scrap characterised by high apparent density, high purity and approximately uniform piece size.

The shredder heavy fraction SHF includes about 50% non-ferrous metals. These are separated and sent for metal recycling. The other 50% of the shredder heavy fraction consists of plastics and other materials. Once the non-ferrous metals have been removed, the residual SHF has to be processed along with the SLF *[Reinhardt Richers 2004]*.

The constituents of the shredder light fraction are mainly non-metallic. Shredding of commercial mixed scrap has been found to produce only about 15% SLF *[Paßvoß 2000]* while recycling of end-of-life vehicles produces about 25% and treatment of electrical and electronic scrap yields as much as 40% SLF. However these are ideal values as shredder plants are usually operated with a planned mixed input.

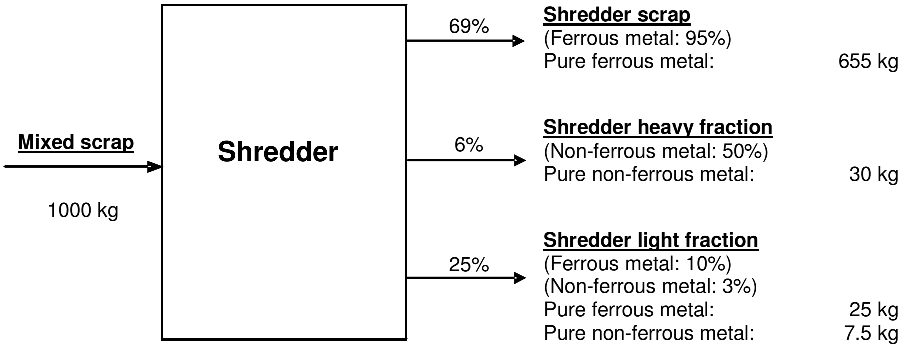
If the potentially large variations in the quantities of different input materials are ignored, the approximate relative amounts of the three output streams can be estimated as follows (see *[AK 2 1990]*):

- shredder scrap: 69 %
- shredder heavy fraction (contains non-ferrous metals): 6 %
- shredder light fraction (including shredder dust): 25 %

The shredder heavy fraction contains about 50 % non-ferrous metals which are recycled following separation. The other 50 % is disposed of together with the shredder light fraction. In 2006 most of the shredder light fraction (including shredder dust) was still landfilled (90 % in 2006 *[Euwid 2006]*) or used for energy recovery (6 % in 2006 *[Euwid 2006]*). The composition of the SLF varies considerably depending on the proportions of EOL vehicles, electrical appliances and other mixed scrap in the input. According to data from various sources (*[Dehoust et al. 1999]*, *[Paßvoß, T. 2000]*), the proportions of different metals vary as follows:

0.5–4 %	aluminium
1–3 %	copper
3–13 %	iron

In the present LCA study, it is assumed that the SLF includes 10 % iron and 3 % non-ferrous metals. On the basis of these data and assumptions the following material flow diagram can be derived:



In this model an input of 1000 kg yields an output fraction containing 718 kg of metals. This corresponds almost exactly to the average refrigeration appliance, whose metal content is about 70 %. This diagram is therefore applicable to the treatment of refrigeration appliances.

According to the assumptions made above, 3.7% of the iron input and 20% of the non-ferrous metal input reach the shredder light fraction. Transfer rates from the shredder input to the recycling fractions shredder scrap and shredder heavy fraction are therefore taken to be 96% for iron and 80% for non-ferrous metals.

7.2 Credits for substituted primary materials

7.2.1 Primary steel production

Primary steel production begins with iron ore which is mined, concentrated and conditioned in other countries and imported into Germany by sea. The fine ore is sintered with coke breeze and heating gas. Ores, coke and additional fuels (coal, oil) are converted in blast furnaces to molten iron, slag and heating gas. Steel is produced from the molten iron in a basic oxygen converter. The steel is then cast into slabs.

The emission factors for primary steel production are calculated using the GEMIS process ‘Metall\Stahl-D-Oxygen’. All data relate to Germany.

7.2.2 Primary aluminium production

Primary aluminium production is modelled using the GEMIS process ‘Metall/Alumium-mix-DE’. This takes account of the various aluminium manufacturing locations producing the aluminium that was consumed in Germany in 1994 (about one third manufactured in Germany and two thirds imported from various parts of the world). All upstream processes are taken into account, specifically: bauxite mining, refining the aluminium-containing bauxite ore into alumina, melt electrolysis to obtain primary aluminium from the alumina, and transport.

7.2.3 Production of concentrated copper ore

Germany’s only primary copper smelter uses about 10% secondary raw materials *[NA 2005]*. It is therefore not possible to conduct LCI calculations for copper production in Germany from purely primary sources. A study that compared primary and secondary copper production *[RWTH-IME 1995]* found that primary energy use for smelting and refining was of the same order of magnitude for primary and secondary copper (deviation of 6%). For this reason only the upstream mining and ore concentration processes are credited for the recycling of copper-containing materials in the present study. It is assumed that the environmental effects of the subsequent stages – smelting and refinement – are similar for primary and secondary copper.

The data was taken from the GEMIS dataset 'Xtra-Abbau\Kupfer-Erz(Konz.)-generisch', which is based on the study mentioned above *[RWTH-IME 1995]*. Both *[RWTH-IME 1995]* and GEMIS assume a model ore with a copper content of 30%.

7.2.4 Production of woodchips

Where woodchips are substituted, credit is given for the production processes in the chipboard industry that are thereby rendered unnecessary. The GEMIS process 'Industrie-Fichte-atro' was used to calculate the costs involved in felling and transporting timber. The NMVOC value for wood production was taken from the Ecoinvent dataset 2477 *[Ecoinvent 2006]*.

A value of 39 kWh per m³ of raw chipboard was assumed for the energy consumed by debarking and chipping operations (see *[Ressel 1986]*). If 67.5% of the raw timber is used, the result is about 51 kWh per dry tonne of wood.

7.3 Transport

Assuming that the 40 t articulated trucks are 50% full (empty return journey) and that each truck can accommodate 140 refrigeration appliances, the following specific emissions were calculated on the basis of ifeu data *[ifeu/UBA 2006]*:

Pollutant	Specific emissions [g/10^3 km]
CH_4	0.055
CO	0.54
CO2	316
N_2O	0.0041
NMVOC	0.19
NOx	2.77
Particulates	0.054
SO_2	0.18

The cumulative energy demand was estimated to be 4,875 kJ per thousand kilometres on the basis of GEMIS data for diesel fuel.

7.4 Fuels and auxiliary materials

The following datasets were used for the energy and material balance calculations:

Table 7-5 Datasets used for modelling the consumption of fuels and auxiliary materials

Fuels and auxiliary materials	GEMIS dataset
Electricity	Netz-el-DE-Verteilung-NS-2000
Diesel	Diesel-D-2000
Liquid nitrogen	Xtra-generisch\N2 (flüssig)
Activated carbon	Activated carbon
NaOH	NaOH-mix-DE
Ammonia	Ammonia

7.5 Elemental analysis

The following material data were used:

Table 7-6 Material data

	Elemental analysis						Water content	Lower heating value	Source
	C	H	O	Cl	S	N		MJ/kg	
Polystyrene	87.80%	7.90%	4.00%	0.10%	0.10%	0.20%	2.00%	37.6	Öko 2001
PVC	42.80%	5.30%	1.70%	50.00%	0.10%	0.10%	2.00%	19.7	Öko 2001
Mixed plastics	82.70%	13.30%	3.80%	0.10%	0.00%	0.10%	2.00%	40.9	Öko 2001
Polystyrene	60.40%						2.00%	25.5	ASTRA 1997
Anthracite	73.00%							27.5	Gemis 2006
Heavy oil	87.50%							40.7	Gemis 2006
Machine oil	85.00%							42	ASTRA 1997 and own estimates
Activated carbon	90.00%							40	ASTRA 1997 and own estimates

7.6 CFC material and energy balance

The following tables show the CFC emissions calculated for the different variants, and compares the baseline values (where HC appliances are assumed to make up 20 % of all refrigeration appliances) with the results of the sensitivity analysis in which the proportion of HC-containing appliances is assumed to be 50 %.

Table 7-7 Baseline calculation of CFC emissions assuming 20 % HC-containing appliances

	Variant 1: Joint processing		**Variant 2: Parallel processing**		**Variant 3: Step 2 (HC) in auto shredder**		**Variant 4: Steps 1+2 (HC) in auto shredder**	
	CFC	**HC**	**CFC**	**HC**	**CFC**	**HC**	**CFC**	**HC**
Missorted CFC appliances	None	None	1% from Step 1	None	1% in Step 2		1%	
Step 1 (R12, isobutane)	Vac. Extraction of R12, Remainder: max. 0.1% w/w in refrigerator oil, thermal treatment --> destruction rate: 98%	Vac. Extraction of isobutane Remainder: max. 0.1% w/w in oil, destruction rate: 99.9%	Vac. Extraction of R12, Remainder: max. 0.1% w/w in refrigerator oil, thermal treatment --> destruction rate: 98%	Vac. Extraction of isobutane Remainder: max. 0.1% w/w in oil, destruction rate: 99.9%	Vac. Extraction of R12, Remainder: max. 0.1% w/w in refrigerator oil, thermal treatment --> destruction rate: 98%	Vac. Extraction of isobutane Remainder: max. 0.1% w/w in oil, destruction rate: 99.9%	Vac. Extraction of R12, Remainder: max. 0.1% w/w in refrigerator oil, thermal treatment --> destruction rate: 98%	Isobutane released to atmosphere
	R12 collected: 100% H-T combustion; destruction rate: 99.99%	Isobutane collected: 100% H-T combustion	R12 collected: 100% H-T comb.; destruction rate: 99.99%	Isobutane collected: controlled combustion	R12 collected: 100% H-T comb.; destruction rate: 99.99%	Isobutane collected: 100% H-T combustion; destruction rate: 100%	R12 from missorted appliances: released to atmosphere	
	Exhaust air: contained in Step 2	Exhaust air: 4 mg/m3 at 4.5 m3/appliance	Exhaust air: contained in Step 2		Exhaust air: contained in Step 2		Exhaust air: contained in Step 2	
			Controlled combustion of R12 from missorted appliances: destruction rate: 90%				R12 collected: 100% H-T comb.; destruction rate: 99.99%	
Step 2 (R11, cyclopentane)	Degassing of PU foam. Loss: max. 0.2% w/w R11. Recycled as absorbent/binder. Destruction rate in subsequent incineration: 98%	Degassing of PU foam Residual content: 0.22% w/w of PU foam; recycled as absorbent/binder. Destruction rate in subsequent incineration: 99.9%	Degassing of PU foam. Loss: max. 0.2% w/w R11 Recycled as absorbent/binder. Destruction rate in subsequent incineration: 98%	Release of 30% of cyclopentane during shredding. No further milling and degassing.	Degassing of PU foam. Loss: max. 0.2% w/w R11 Recycled as absorbent/binder. Destruction rate in subsequent incineration: 98%	Release of 70% of cyclopentane during shredding.	Degassing of PU foam. Loss: max. 0.2% w/w R11 Recycled as absorbent/binder. Destruction rate in subsequent incineration: 98%	Release of 70% of cyclopentane during shredding.
	0.5% PU residue adhering to metals and plastics, containing 30% of original R11; destruction rate: 90%	0.5% PU residue adhering to metals and plastics, containing 30% of original cyclopentane; destruction rate: 90%	0.5% PU residue adhering to metals and plastics, containing 30% of original R11; destruction rate: 90%	Further 70% incinerated with foam; Destruction rate in waste incinerator: 99.9%	0.5% PU residue adhering to metals and plastics, containing 30% of original R11; destruction rate: 90%		0.5% PU residue adhering to metals and plastics, containing 30% of original R11; destruction rate: 90%	
	Exhaust air: 4 mg/m3 at 9 m3/appliance	Exhaust air: 4 mg/m3 at 4.5 m3/appliance			Exhaust air: 4 mg/m3 at 9 m3/appliance		Exhaust air: 4 mg/m3 at 9 m3/appliance	
	R11 collected: 100% incinerated; destruction rate: 99.99%		R11 collected: 100% incinerated; (destruction rate: 99.99%); Missorted appliances: 30% released, 70% incinerated with PU chunks in waste incinerator; degradation rate: 90%		R11 collected: 100% incinerated; (destruction rate: 99.99%); R11 from missorted appliances: 70% released in shredder; 30% into SLF. 60% of SLF is incinerated (destruction rate: 90%) and 40% receives post-shredder treatment (98% –> blast furnace with 90% destruction, and 2% –> landfill, i.e. released)	30% end up in the SLF. 60% of SLF incinerated (destruction rate: 99.9%) and 40% receives post-shredder treatment (98% –> blast furnace with 99.9% destruction, and 2% –> landfill, i.e. released)	R11 collected: 100% incinerated; (destruction rate: 99.99%); R11 from missorted appliances: 70% released in shredder; 30% into SLF. 60% of SLF is incinerated (destruction rate: 90%) and 40% receives post-shredder treatment (98% –> blast furnace with 90% destruction, and 2% –> landfill, i.e. released)	30% end up in the SLF. 60% of SLF incinerated (destruction rate: 99.9%) and 40% receives post-shredder treatment (98% –> blast furnace with 99.9% destruction, and 2% –> landfill, i.e. released)

Results in kg:	**Variant 1: Joint processing**		**Variant 2: Parallel processing**		**Variant 3: Step 2 (HC) in auto shredder**		**Variant 4: Steps 1+2 (HC) in auto shredder**	
	CFC	**HC**	**CFC**	**HC**	**CFC**	**HC**	**CFC**	**HC**
Step 1:	9	0.11	9	0.11	9	0.11	9	32,400
R12 & isobutane	25	10	25	10	25	10	2,484	
			248				25	
Total for Step 1	**33**	**10**	**282**	**10**	**33**	**10**	**2,517**	**32,400**
Step 2:	335	4.6	331	35,640	331	83,160	331	83,160
R11 & cyclopent.	694	1.1	687	83	687		687	
	78	10	77		77		77	
	73		2,791		5,450	320	5,450	320
Total for Step 2	**1,180**	**15**	**3,886**	**35,723**	**6,546**	**83,480**	**6,546**	**83,480**

Table 7-8 Calculation of CFC emissions for the sensitivity analysis that assumes 50 % HC-containing appliances

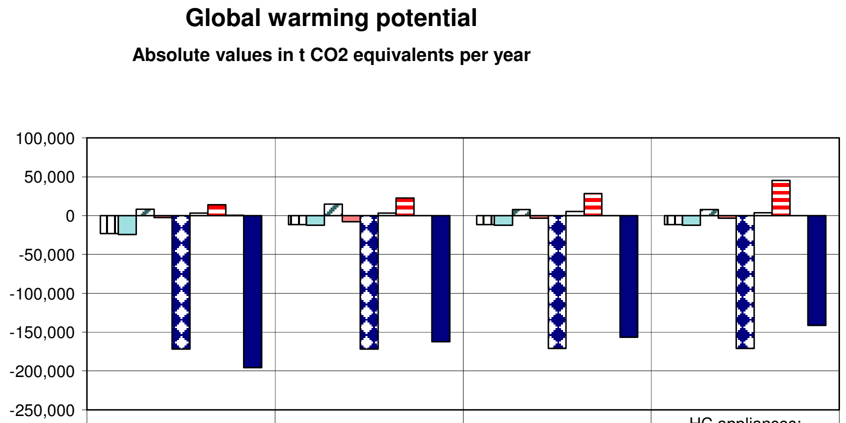
	Variant 1: Joint processing		Variant 2: Parallel processing		Variant 3: Step 2 (HC) in auto shredder		Variant 4: Steps 1+2 (HC) in auto shredder	
	CFC	**HC**	**CFC**	**HC**	**CFC**	**HC**	**CFC**	**HC**
Missorted CFC appliances	None	None	1% from Step 1	None	1% in Step 2		1%	
Step 1 (R12, isobutane)	Vac. Extraction of R12, Remainder: max. 0.1% w/w in refrigerator oil, thermal treatment --> destruction rate: 98%	Vac. Extraction of isobutane Remainder: max. 0.1% w/w in oil, destruction rate: 99.9%	Vac. Extraction of R12, Remainder: max. 0.1% w/w in refrigerator oil, thermal treatment --> destruction rate: 98%	Vac. Extraction of isobutane Remainder: max. 0.1% w/w in oil, destruction rate: 99.9%	Vac. Extraction of R12, Remainder: max. 0.1% w/w in refrigerator oil, thermal treatment --> destruction rate: 98%	Vac. Extraction of isobutane Remainder: max. 0.1% w/w in oil, destruction rate: 99.9%	Vac. Extraction of R12, Remainder: max. 0.1% w/w in refrigerator oil, thermal treatment --> destruction rate: 98%	Isobutane released to atmosphere
	R12 collected: 100% H-T combustion; destruction rate: 99.99%	Isobutane collected: 100% H-T combustion	R12 collected: 100% H-T comb.; destruction rate: 99.99%	Isobutane collected: controlled combustion	R12 collected: 100% H-T comb.; destruction rate: 99.99%	Isobutane collected: 100% H-T combustion; destruction rate: 100%	R12 from missorted appliances: released to atmosphere	
	Exhaust air: contained in Step 2	Exhaust air: 4 mg/m3 at 4.5 m3/appliance	Exhaust air: contained in Step 2		Exhaust air: contained in Step 2		Exhaust air: contained in Step 2	
			Controlled combustion of R12 from missorted appliances: destruction rate: 90%				R12 collected: 100% H-T comb.; destruction rate: 99.99%	
Step 2 (R11, cyclopentane)	Degassing of PU foam. Loss: max. 0.2% w/w R11. Recycled as absorbent/binder. Destruction rate in subsequent incineration: 98%	Degassing of PU foam Residual content: 0.22% w/w of PU foam; recycled as absorbent/binder. Destruction rate in subsequent incineration: 99.9%	Degassing of PU foam. Loss: max. 0.2% w/w R11 Recycled as absorbent/binder. Destruction rate in subsequent incineration: 98%	Release of 30% of cyclopentane during shredding. No further milling and degassing.	Degassing of PU foam. Loss: max. 0.2% w/w R11 Recycled as absorbent/binder. Destruction rate in subsequent incineration: 98%	Release of 70% of cyclopentane during shredding.	Degassing of PU foam. Loss: max. 0.2% w/w R11 Recycled as absorbent/binder. Destruction rate in subsequent incineration: 98%	Release of 70% of cyclopentane during shredding.
	0.5% PU residue adhering to metals and plastics, containing 30% of original R11; destruction rate: 90%	0.5% PU residue adhering to metals and plastics, containing 30% of original cyclopentane; destruction rate: 90%	0.5% PU residue adhering to metals and plastics, containing 30% of original R11; destruction rate: 90%	Further 70% incinerated with foam; Destruction rate in waste incinerator: 99.9%	0.5% PU residue adhering to metals and plastics, containing 30% of original R11; destruction rate: 90%		0.5% PU residue adhering to metals and plastics, containing 30% of original R11; destruction rate: 90%	
	R11 collected: 100% incinerated; destruction rate: 99.99%	Exhaust air: 4 mg/m3 at 4.5 m3/appliance	R11 collected: 100% incinerated; (destruction rate: 99.99%); Missorted appliances: 30% released, 70% incinerated with PU chunks in waste incinerator; degradation rate: 90%		R11 collected: 100% incinerated; (destruction rate: 99.99%); R11 from missorted appliances: 70% released in shredder; 30% into SLF. 20% of SLF is incinerated (destruction rate: 90%) and 80% receives post-shredder treatment (98% –> blast furnace with 90% destruction, and 2% –> landfill, i.e. released)	30% end up in the SLF. 20% of SLF incinerated (destruction rate: 99.9%) and 80% receives post-shredder treatment (98% –> blast furnace with 99.9% destruction, and 2% –> landfill, i.e. released)	R11 collected: 100% incinerated; (destruction rate: 99.99%); R11 from missorted appliances: 70% released in shredder; 30% into SLF. 20% of SLF is incinerated (destruction rate: 90%) and 80% receives post-shredder treatment (98% –> blast furnace with 90% destruction, and 2% –> landfill, i.e. released)	30% end up in the SLF. 20% of SLF incinerated (destruction rate: 99.9%) and 80% receives post-shredder treatment (98% –> blast furnace with 99.9% destruction, and 2% –> landfill, i.e. released)

Results in kg:	Variant 1: Joint processing		Variant 2: Parallel processing		Variant 3: Step 2 (HC) in auto shredder		Variant 4: Steps 1+2 (HC) in auto shredder	
	CFC	**HC**	**CFC**	**HC**	**CFC**	**HC**	**CFC**	**HC**
Step 1:	5	0.27	5	0.27	5	0.27	5	81,000
R12 & isobutane	16	24	16	24	16	24	1,553	
			155				16	
Total for Step 1	**21**	**25**	**176**	**25**	**21**	**25**	**1,573**	**81,000**
Step 2:	209	11.5	207	89,100	207	207,900	207	207,900
R11 & cyclopent.	434	2.8	429	208	429		429	
	49	24	48		48		48	
	46		1,744		3,416	1,513	3,416	1,513
Total for Step 2	**737**	**39**	**2,429**	**89,308**	**4,101**	**209,413**	**4,101**	**209,413**

8 Appendix B: Impact assessment

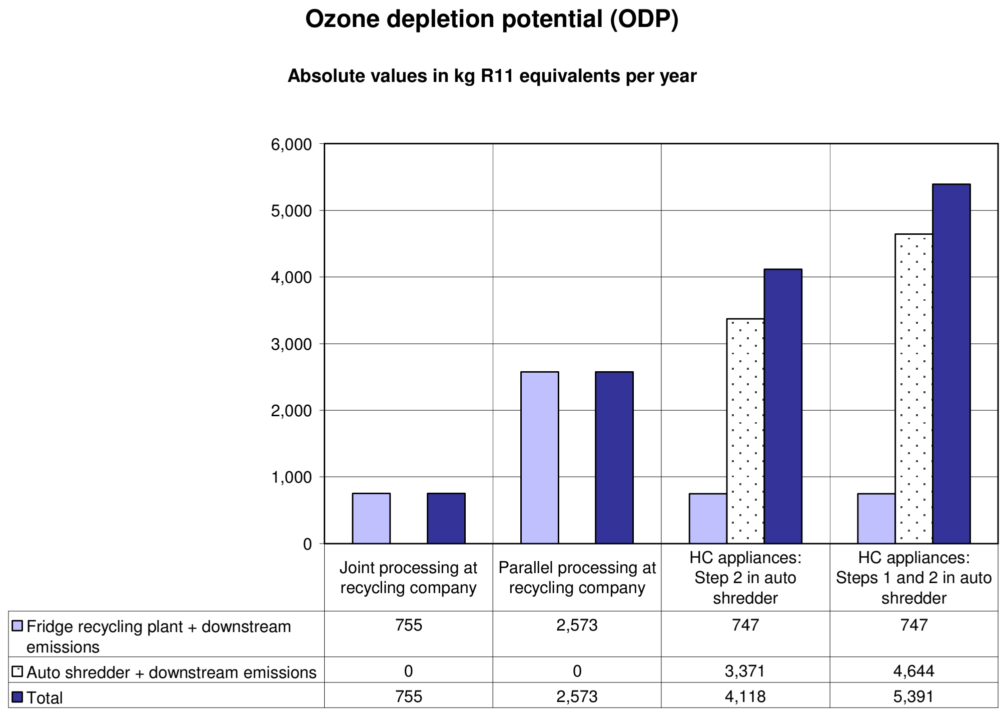
8.1 Results assuming 50% HC-containing appliances

The following diagrams show the results for the individual sub-processes assuming that 50% of the appliances contain hydrocarbons and that the shredder light fraction is disposed of by incineration (60%), in cement kilns (20%) and by post-shredder processing (20%). The results are shown for the impact categories 'global warming potential', 'ozone depletion potential' and 'photochemical ozone creation potential'.



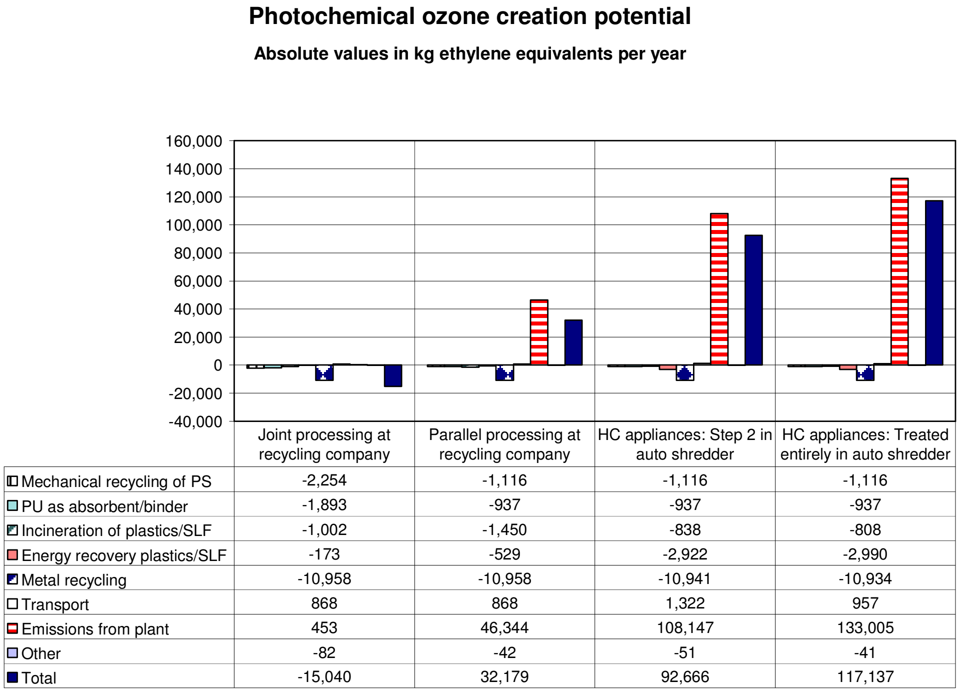
	Joint processing at recycling company	Parallel processing at recycling company	HC appliances: Step 2 in auto shredder	HC appliances: Steps 1 and 2 in auto shredder
Mechanical recycling of PS	-22,993	-11,382	-11,382	-11,382
PU as absorbent/binder	-24,451	-12,103	-12,103	-12,103
Incineration of plastics/SLF	8,379	14,710	7,860	7,609
Energy recovery plastics/SLF	-2,607	-7,991	-3,163	-3,272
Metal recycling	-171,493	-171,493	-171,215	-171,108
Transport	3,462	3,462	5,275	3,816
Emissions from plant	13,939	22,560	28,431	45,452
Other	382	-181	-62	-117
Total	-195,383	-162,418	-156,360	-141,105

Figure 8.1 Results for the impact category 'global warming potential' assuming 50% HC-containing appliances (sensitivity analysis)



	Joint processing at recycling company	Parallel processing at recycling company	HC appliances: Step 2 in auto shredder	HC appliances: Steps 1 and 2 in auto shredder
▨ Fridge recycling plant + downstream emissions	755	2,573	747	747
☐ Auto shredder + downstream emissions	0	0	3,371	4,644
■ Total	755	2,573	4,118	5,391

Figure 8.2 Results for the impact category ‘ozone depletion potential’ assuming 50 % HC-containing appliances (sensitivity analysis)



	Joint processing at recycling company	Parallel processing at recycling company	HC appliances: Step 2 in auto shredder	HC appliances: Treated entirely in auto shredder
◫ Mechanical recycling of PS	-2,254	-1,116	-1,116	-1,116
□ PU as absorbent/binder	-1,893	-937	-937	-937
◩ Incineration of plastics/SLF	-1,002	-1,450	-838	-808
□ Energy recovery plastics/SLF	-173	-529	-2,922	-2,990
◩ Metal recycling	-10,958	-10,958	-10,941	-10,934
□ Transport	868	868	1,322	957
▤ Emissions from plant	453	46,344	108,147	133,005
□ Other	-82	-42	-51	-41
■ Total	-15,040	32,179	92,666	117,137

Figure 8.3 Results for the impact category ‘photochemical ozone creation potential’ assuming 50 % HC-containing appliances (sensitivity analysis)