Free dissemination

(Contract 021262)

SusTools
Tools for Sustainability:
Development and Application of an Integrated Framework

Evaluation of treatment options for municipal solid waste
and Stakeholder Workshops & Multicriteria Analysis

Rudi Torfs[1], Ann Van der Linden[1], Ari Rabl[2], Assaad Zoughaib[2], Tim Taylor[3], Steve Arnold[3], Danae Diakoulaki[4], George Mavrotas[4] and Mike Holland[5].

[1]: Vito, Belgium
[2]: Ecole des Mines de Paris, France
[3]: University of Bath, UK
[4]: National Technical University of Athens, Greece
[5]: EMRC, UK

Study commissioned by EC DG Research

2005/IMS/R/002

VITO

January 2005

TABLE OF CONTENTS

LIST OF TABLES

LIST OF FIGURES

SUMMARY

Part 1 of this report summarizes the results of the waste treatment case studies of the SusTools project of the EC. They serve as basis for the multicriteria analysis that has been carried out during the workshop with the stakeholders. The case studies concern the waste that remains after selective collection and recycling. The case study in France compares the external costs of land fill and incineration. The case study in Flanders (Belgium) zooms in on several technologies for incineration, presenting both private and external costs. The results highlight the importance of avoided pollution due to energy recovery (and to a lesser extent materials recovery). Emissions due to the transport of waste play a negligible role compared to the direct emisisons from the incinerator or landfill. The comparison of damage costs between landfill and incineration depends very stongly on the efficiency of energy recovery and the fuels that are dispaced.

In part 2 a multicriteria analysis is made. The methodological background to develop the MCA framework, organizational details for the workshop, and the results obtained for each case study during the workshop are summarised in this chapter. The participation of the stakeholders in the workshops has offered the opportunity to communicate the consequences of alternative policy options for waste treatment. Their involvement was critical in highlighting different aspects of the problems and to identify aspects needing further research. Moreover, the stakeholders were confronted with the dilemmas raised and tried to solve them through a structured preference elicitation procedure that helped balanc the pros and cons and reach the most satisfactory compromise. Preferences have been elaborated with an interactive platform based on Multi-Criteria Analysis (MCA) model during the workshop and the results have been presented to stakeholders to initiate further discussion. The transparency of the procedure and the open discussion on critical points led to the common approval of this compromise solution (which for all case studies turned out the same as the one chosen intuitively by most participants).
In any case, an interactive-iterative procedure of stakeholders' involvement is necessary for the assessment of policy options, and the workshops have demonstrated the value of the framework proposed in the SusTools project.

PART 1 :
EVALUATION OF TREATMENT OPTIONS FOR MUNICIPAL SOLID WASTE

1 INTRODUCTION

This report presents the results of the waste treatment case studies of the SusTools project of the EC. SusTools is the acronym of the project ‘Tools for sustainability – Development and application of an integrated framework’, and was funded by the European Commission under the EESD Programme (5th framework programme, 1998-2002, contract N° EVG3-CT-2002-80010).

The project addresses one of the main difficulties with formulating policies for sustainable development, namely the need to assess the numerous and complex tradeoffs that are inevitably present and whose overall consequences are often not obvious. There are costs and benefits, winners and losers; careful analysis of each specific policy option is needed to evaluate the consequences and decide whether the benefits outweigh the costs. Over the years various tools have been developed for the evaluation of environmental policy options, but often without much interaction or exchange. Today, policy makers need an integrated approach using the most appropriate tools in a consistent manner. The present project develops therefore an integrated framework, based on the tools and lessons learned in different scientific disciplines and policies areas.

SusTools demonstrates a combination of different tools and methods for widely different environmental issues, with varying degree of information available for the policymaker. The focus is not on inventing new instruments, but rather to present a framework of existing instruments which can be used in a systematic way. Combining elements of life cycle assessment (LCA), impact pathway approach (IPA), cost-benefit analysis (CBA) and multi-criteria decision analysis (MCDA), the SusTools methodology brings together state-of-the-art science with recent advances in decision analysis. The case studies presented are for demonstration purposes only and add to an already existing discussion in the EU on the management of municipal solid waste (see also chapter 2). It does not seek complete solutions for future policies, but the SusTools approach enables a clear analysis of policy options, alternative solutions and the selection in consensus with different stakeholders in an imperfect world with uncertain information and data. In an extended impact assessment of EU policies, for example, this approach deserves a place. But also on a smaller scale, tackling local issues, it can prove useful.

General methodological issues for the case studies are described in chapter 3. An application of tools like LCA, IPA and others are demonstrated in chapter 4 where a review of literature and studies has been done, to derive indicative numbers on costs and external costs of waste treatment in Europe. The first case study on waste describes a national site dependant issue of selecting either landfill or incineration in France (chapter 5). At different sites the optimal choice might be different. Whereas the first case study uses a rather generic incinerator as contrast to landfill, the second waste case study zooms in on the different incineration technologies in Flanders (Belgium). Purpose is to demonstrate a more detailed level of choice (chapter 6). Once the choice has fallen on incineration instead of landfill, there are many options available on the market. Apart from economical and legal aspects a policy maker also wants to have insight in environmental and social aspects of the different technologies, and weigh the options according to his/her preferences (e.g. more materials recovered as a goal). Both case studies use available data, and have therefore limited their options to those technologies for which the relevant data is available. If in the future the methodology is further used to test other policies or new technologies, data gathering is an important part of the preparation.

Both case studies here also prove that with different levels of detail the methodology still can arrive at a decent and realistic evaluation. Both case studies have been developed in close cooperation with the waste industry and with support from the waste authorities. This approach has been very successful. The case studies concern the waste that remains after selective collection and recycling.

2 WASTE POLICY IN THE EU

The production of waste means a loss of valuable resources and energy. Transportation and the disposal of waste can have varying impacts on the environment, particularly on human health and amenity. In order to minimize these burdens, a ranking of preference concerning the waste management options is agreed upon. The criteria that were used in developing these rankings are based on environmental impact considerations– it is not necessarily a case of economic efficiency. The hierarchy is shown in Figure 1.

Minimisation	Most preferred
Re-use	
Composting	
Recycling	
Incineration with energy recovery	
Landfill	Least preferred

Figure 1: Waste hierarchy.

However this is not an absolute guide for all circumstances. The reasoning behind this hierarchy is disputed – landfilling is cheaper in many parts of the EU and the scarcity of land, which drives moves away from landfill, is not necessarily such a key issue across all member states. The European Union's approach to waste management takes into account the following principles:

- Prevention principle, avoiding or minimizing waste;
- Producer responsibility and polluter pays principle;
- Precautionary principle, avoiding anticipating on risks;
- Proximity principle, avoiding or minimizing transportation of waste.

The European waste management strategy is structured according to the waste hierarchy:

- *Waste prevention, waste minimization or re-use of waste*: the easiest way to reduce the overall impact of waste on society is to avoid producing waste. This is a key factor in any waste management strategy. Both reduction of the total amount of waste or the reduction of hazardous components in waste is tackled. Waste prevention is closely linked with improving manufacturing methods, product policies and influencing consumers to demand greener products and less packaging. It has close ties with the EU directive on Integrated Pollution Prevention and Control (IPPC), and the Integrated Product Policy strategy.
- *Recycling and reuse*: If waste cannot be prevented, as many of the materials as possible should be recovered, preferably by recycling. The European Commission has defined several specific 'waste streams' for priority attention, the aim being to reduce their overall environmental impact. This includes packaging waste, end-of-life vehicles, batteries, electrical and electronic waste. EU directives now require Member States to introduce legislation on waste collection, reuse, recycling and disposal of these waste streams. The commission is developing targets for prevention and recycling as a further step to move away from incineration and landfill. It is acknowledged that quantifiable targets are not yet possible, and that data and broadly accepted methods are needed in order to define the optimal mix of different approaches to waste. It is noted that recycling might even be a bigger burden than disposal in some cases.

- *Improving final disposal and monitoring*: Where possible, waste that cannot be recycled or reused should be safely incinerated, with landfill only used as a last resort. Both these methods need close monitoring because of their potential for causing severe environmental damage. The EU has recently approved a directive setting strict guidelines for landfill management. It bans certain types of waste, such as used tyres, and sets targets for reducing quantities of biodegradable rubbish. Another recent directive lays down tough limits on emission levels from incinerators.

The case studies presented in this study only deal with the final disposal of waste.

3 METHODOLOGIES

The concern about waste is driven by the knowledge of the potential damages of waste (either through landfill or incineration), and by the fact that waste production per capita is still increasing. Today, the major challenges to make a generic waste strategy operational and to implement it, are:

- How to compare and choose between different waste treatment options, especially since a number of them are new?
- How to implement such a strategy, taking into account local aspects

For both points, specific tools are available. To choose between different technology options, one can use a number of instruments, especially related to economic information (costs of the technologies), to environmental issues (LCA, Life cycle impact assessment (LCIA), health impacts assessment with a focus on collective exposures or doses, IPA, external costs,…), related to technology risk assessment, or to energy recuperation. By means of cost-benefit analysis and multi-criteria analysis (MCA), the different indicators can be compared, and optimal choices can be made. The results may be expressed in indicators per kg waste treated. These tools have in common that they look at costs and benefits for *society as a whole*, and allow for trade-off within different themes (e.g. environment, minimising total exposure and health impacts) and between themes (economic and environmental damage costs). The policy discussion can be at a national or regional level.

LCA provides an inventory of the pollutants emitted by the different activities (stages) involved in the treatment of waste. The external costs due to these pollutants are calculated by means of an IPA, taking into account the pathways by which the pollutant disperses in the environment (e.g. the passage of dioxins into the food chain). LCA and IPA are both used in a review of costs and external costs in section 4 and in the case studies to define burdens and impacts.

The damage costs per kg of pollutant are shown in Figure 2, based on the IPA methodology of ExternE. Damage costs for PM_{10}, NO_x and SO2 depend on emission site and values are shown for incinerator emissions in typical cities in France and in Flanders. For the other pollutants in Figure 2 there is little or no variation with emission site. The cost of greenhouse gases is estimated as 0.019 €/kg of CO_{2equiv} (for CH_4 a global warming potential of 20 is assumed).

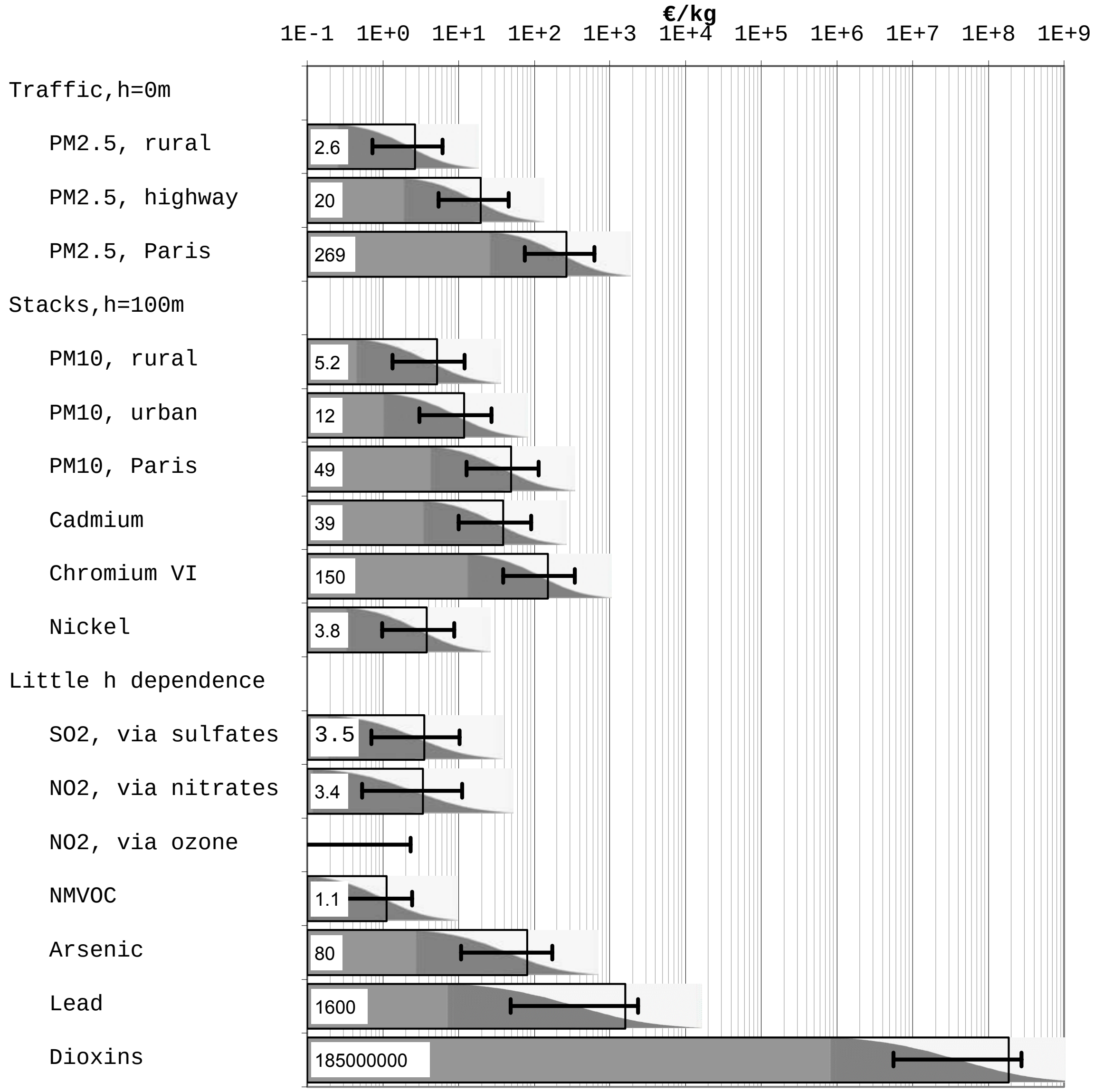


Figure 2: Damage cost per kg of pollutant emitted by incinerators in typical sites of France, assumed for the present study. The numbers for Flanders are essentially the same, except for PM_{10} (19 €/kg) and SO_2 (8 €/kg).

Amenity impacts (odour, noise, visual intrusion) are not considered in the case studies; they should however be taken into account by local decision makers when faced with a specific choice. For a review of amenity impacts of waste management options see also the section 4.8 and the annex to this report.

The external costs are obtained by aggregating the impact of individual facilities over all regions and populations of Europe (bearing in mind that air pollutants disperse over great distances). Thus they do not show the contribution of groups or individuals near an installation who may be exposed to much higher doses than the rest of the population. Since protection against excessive risks is also an important criterion for decisions, it has to be taken into account separately. However, in practice under current regulations even the largest individual risks imposed by waste treatment facilities are so small that this consideration is unlikely to impose a constraint on a decision.

4 A REVIEW OF THE BURDENS AND IMPACTS OF WASTE TREATMENT TO PERFORM AN ECONOMIC EVALUATION

This section reviews the available literature and data on burdens and impacts of different waste treatment technologies. The review serves as a basis for deriving a full cost of waste treatment including the external cost of all burdens and impacts (the environmental externalities). This section presents a brief summary of the main findings of the paper, that is attached as an annex to this report (Taylor, Arnold and Markandya, 2004).

4.1 Methodology

An in depth review of secondary data on the costs of various waste management options was conducted. This built on earlier work for the Commission by Coopers and Lybrand (1993) and Eunomia (2004). It is obvious that care should be taken when applying such estimates for formal cost-benefit analysis of waste treatment options – as technologies and standards change over time so do costs. Thus the cost estimates contained in this section of the report should be taken as being illustrative rather than definitive. In addition, many of the external cost estimates are site-specific and thus care should be taken when applying these to other areas or to Europe as a whole.

4.2 Definitions and data

Municipal Solid Waste (MSW) is a by-product of the consumption of goods and services by households. The costs involved range from the financial costs faced by the providers of waste management services to external costs including environmental and health costs.

Finding data on financial costs is difficult as, on the whole, such costs are not published due to commercial sensitivity. For external costs, the impact pathways are not always clear, and where they are the costs may be difficult to distinguish due to a lack of studies (as is the case with amenity impacts of incineration for example).

This study will focus on the costs of landfill and incineration of MSW. A summary of the costs, benefits and outcomes of the technologies are shown inTable 1. The rest of the document will show figures for the financial costs and show the magnitude of the non-financial effects. It is assumed that all possible recycling and composting takes place, and also that all technologies are at their technical optima - for example, all landfills have state-of-the-art leachate/gas collections systems.

Table 1: A summary of the main costs of the options of incineration and landfill.

	Financial	**Non-financial**
Incineration		
Construction	General construction costs	Political good will/bad will
Operation	Monitoring Abatement Labour Pre-treatment Fuel	Disposal of bottom ash Fly ash Gas emissions Traffic Health and safety risks
Post-operation	Decontamination	Visual eyesore of chimney
Benefits	Heat and electricity can be generated and distributed from the incinerator and could replace more polluting sources. The volume of waste is reduced. Ferrous metal can be removed from the bottom ash and the remaining bottom ash can be used in construction.	
Landfill		
Construction	Land Construction costs	Political goodwill/badwill Land use changes
Operation	Labour Fuel Machinery Monitoring	Leachate Gas Health and safety Rats/seagulls/vermin Traffic congestion Litter Health and safety risks Odour
Post-operation	Monitoring	Gas Waste land
Benefits	Landfill gas can be collected and used for electricity/heat generation, or refined and sold as fuel. After a sufficient period the land can be reused.	

4.3 Financial costs

An important indicator of the financial cost to waste managers are the so-called gate fees charged by site operators for waste treated at incinerators and landfills.Table 2 shows how these vary across Europe. The gate fees excluding tax are a good indication of the financial costs facing the site operators, even though they do not provide information on how these costs are made up. As the table shows, gate fees for both landfill sites and incinerators vary quite widely. A broad trend is that fees vary across countries to similar extents for landfill and incineration. However, in Luxembourg and the Netherlands, both countries with high population densities, landfill costs are seen to be higher than incineration. This is probably due to the increased opportunity cost of land use for landfill in these countries.

Table 2: Gate fees and taxes across Europe.

Country	Gate fee excl. tax (€/t)	Source	Tax (€/t)	Source	Total gate fee (€/t)
Incineration					
Austria	105	(3)	14-71	(3)	**148**
Belgium	70	(3)	3.7-22.3	(1)	**83**
Denmark	42-75	(3)	38 with energy recovery else 44	(1)	**97 or 103 (no energy recovery)**
Finland	52	(3)			**52**
France	88	(3)			**88**
Germany	88	(3)			**88**
Luxembourg	132				**132**
Netherlands	84	(2)	0	(1)	**84**
Spain	32	(3)			**32**
Sweden	35	(3)	0	(1)	**31**
	17-45	(2)			
UK	50	(3)			**50**
Landfill					
Austria	92	(3)	21.8-29.1 (43.9 in 2001)	(1)	**117**
Belgium	68-83	(1)	3.7-22.3	(1)	**63**
	50	(3)			
Denmark	34	(3)	50	(1)	**84**
Finland	12	(3)	15	(1)	**27**
France	32-55	(2)	9	(1)	**52.5**
Germany	26-153	(2)	proposed 12.7	(1)	**90**
	60-90	(1)			
Greece	10	(3)	0	(1)	**11**
	06--15	(1)			
Ireland	44-51	(2)	0	(1)	**47.5**
Italy	10.3-25.8	(1)	10	(1)	**28**
Luxembourg	162	(3)	0	(1)	**162**
Netherlands	75	(1)	12.4-64.3	(1)	**113**
Portugal	6--15	(1)	0	(1)	**11**
Spain	15-30	(1)	0	(1)	**23**
	9	(2)			
	15	(3)			
Sweden	23-90	(2)	29	(1)	**86**
UK	18-33	(1)	17.6	(1)	**43**

(1) Database of environmental taxes, http://europa.eu.int/comm/environment/enveco/env_database
(2) National expert
(3) European Environment Agency 1999, 'Environment in the EU at the turn of the century', chapter 3.7.
Note: The stated total may not be the total of Gate fees stated + tax due to different sources being named.
Source: OECD/EU (2003)

After paying the gate fees and taxes, the authorities who deal with the waste management also have to pay for collection and pre-treatment (sorting and compressing, transport).
Table 3 presents findings from a variety of municipalities in the UK detailing their total costs of dealing with one tonne of MSW. These range from under £30 to over £45, with a mean

cost of £34 (€55). The proportions of MSW treated in different ways are shown in the *Notes* column. These data do not reveal much about the relative costs of landfill versus incineration since few municipalities in the UK incinerate large proportions of their MSW. However, they do give an indication of the price paid for the treatment and disposal of a tonne of MSW in the UK.

Table 3: Total costs of waste treatment. Sources: County Council Websites

Cost/ tonne	Source	Year	2000 prices £	2000 prices €	Notes
£34.41	Bexley	1997/98	**£35.69**	**€ 58.81**	Not including recycled, composted
£36.31	Bexley	1998/99	**£37.08**	**€ 61.10**	
£27.45	Warwickshire	2001/02	**£27.12**	**€ 44.68**	9.1%R, 4.3%C, 3.2% I, 83.4% L
£28.15	Warwickshire	2002/03	**£27.45**	**€ 45.23**	10.0% R, 4.9% C, 4.5% I, 80.6% L
£29.50	Warwickshire	2003/04	**£28.37**	**€ 46.75**	target; 11.0% R, 9% C, 3.5% I, 76.5%L
£33.07	Derbyshire	2001/02	**£32.67**	**€ 53.83**	8.44% R, 3.44 %C, 0.03% I, 87.79% L
£38.90	Hampshire	1998/99	**£39.73**	**€ 65.46**	
£40.83	Hampshire	2001/02	**£40.33**	**€ 66.46**	
£47.73	Hampshire	2002/03	**£46.54**	**€ 76.68**	cost of disposal
£30.36	Gateshead	2001/02	**£29.99**	**€ 48.12**	539kg/person; 6.8%R, 93.2%L
£33.60	Gateshead	2002/03	**£32.76**	**€ 52.86**	534.90kg.person; 3.25%R, 95.19%L (a)
£31.95	Gateshead	2003/04	**£30.73**	**€ 44.03**	541.62kg/person; 4.53%R, 95.21%L (a)
		Mean	***£34.04***	***€55.33***	
		Median	***£32.72***	***€53.35***	

a) no data for remainder - probably incinerated without energy recovery

Key: R = Recycled, C = Composted, I = Incinerated, L= landfill

Sources: County Council Websites

Table 4 presents the range and mean of the costs found in the review. These are presented in Section 2 of the annex by technology type and by type of cost. The data presented is from a range of EU states, but primarily for the UK. It shows that for incineration the capital costs vary widely whereas landfill and pre-treatment plants are more uniform in their capital costs. The total costs are based on data from both average total net costs in Coopers and Lybrand (1996) and Eunomia (2001). Interesting to note are the weighted means, which were derived using population data for 2000 (1998 for Belgium) to give a less biased average. Perhaps the most obvious thing to note about the table is the lack of data - showing the scarcity of financial data for many waste treatment options. However, the table does show that landfill is generally cheaper than incineration, particularly in terms of running costs. It also shows the great variability in costs, even within one particular technology or process. This is because waste management is usually at a local level whereby authorities or municipalities decide what type of solution they require and at what scale. The result is that the variations in collection, treatment and disposal - along with reclaimed materials and energy generated - lead to many different costs.

The summary table gives averages for the total costs. The range of values shown is taken from the sources presented in the annex, and gives an indication of how widely costs vary. The two important variables that affect total cost seem are location and the size of the plant, particularly for incinerators. The average figures shown are taken from two studies, notablyCoopers and Lybrand (1996) and Eunomia (2001). The Coopers and Lybrand report estimated costs net of electricity production by country and found that landfilling had lower costs than incineration in almost every case. Because some of the smaller nations have much higher per tonne costs, the averages have been population weighted, and this is probably why the costs are higher than the gate fees shown earlier. In comparison, the Eunomia study presented costs for incineration net of tax and energy revenue, and landfill including tax and the population weighted average costs are shown below. There are fewer taxes on incineration in Europe than on landfilling as a consequence of the use of incentives to support the waste hierarchy. The main thing to note here is the large difference between average landfill costs. One reason for this is likely to be the time difference between the two studies; the total cost of landfilling has been rising in Europe over recent years both due to financial pressures (scarcity of sites) and legislative pressures (landfill tax, more stringent standards on leachate and gas collection).

Table 4: MSW treatment costs in EU

Costs in €, for 2001	**Moving Grate**	**Fluidised Bed**	**Landfill**	**Pre-treatment**
Capital				
Range of Costs	80-508m	4.2m-80m	21.9-36m	18-21.8m
Mean	236.6m	42.1m	30.56m	19.9m
Capital per tonne per year				
Range of Costs	52.26-63.61	26.02-28	8-17.81	3.91-13.95
Mean	57.94	27.01	13.66	8.93
O&M				
Range of Costs	57.66/t		3-39/t	
Mean	57.66/t		10.92	
Pre-tax Costs before revenues from recovered materials or energy reduce the net cost.				
Range of Costs	37-83			
Mean	53.81			
Heat Production Prices €/kWh				
Range of Prices	0.018-0.02			
Mean	0.019			
Electricity Production Prices €/kWh				
Range of Prices	0.014-0.05			
Mean	0.036			
Electricity Profits				
Range of Profits	16.27/t			
Mean	16.27/t			
Electricity Production				
Range of Output/capacity	0.051-0.108kW/t			
Mean	0.078kw/t			
Total Costs				
Range of Costs	37.07-141.66		28.78-135.5	
Mean (unweighted)	73.03		73.092	
Mean (C&L, pop. weighted)	72.56		28.96	
Mean (Eunomia, pop. weighted)			83.01 (1)	52.50 (2)

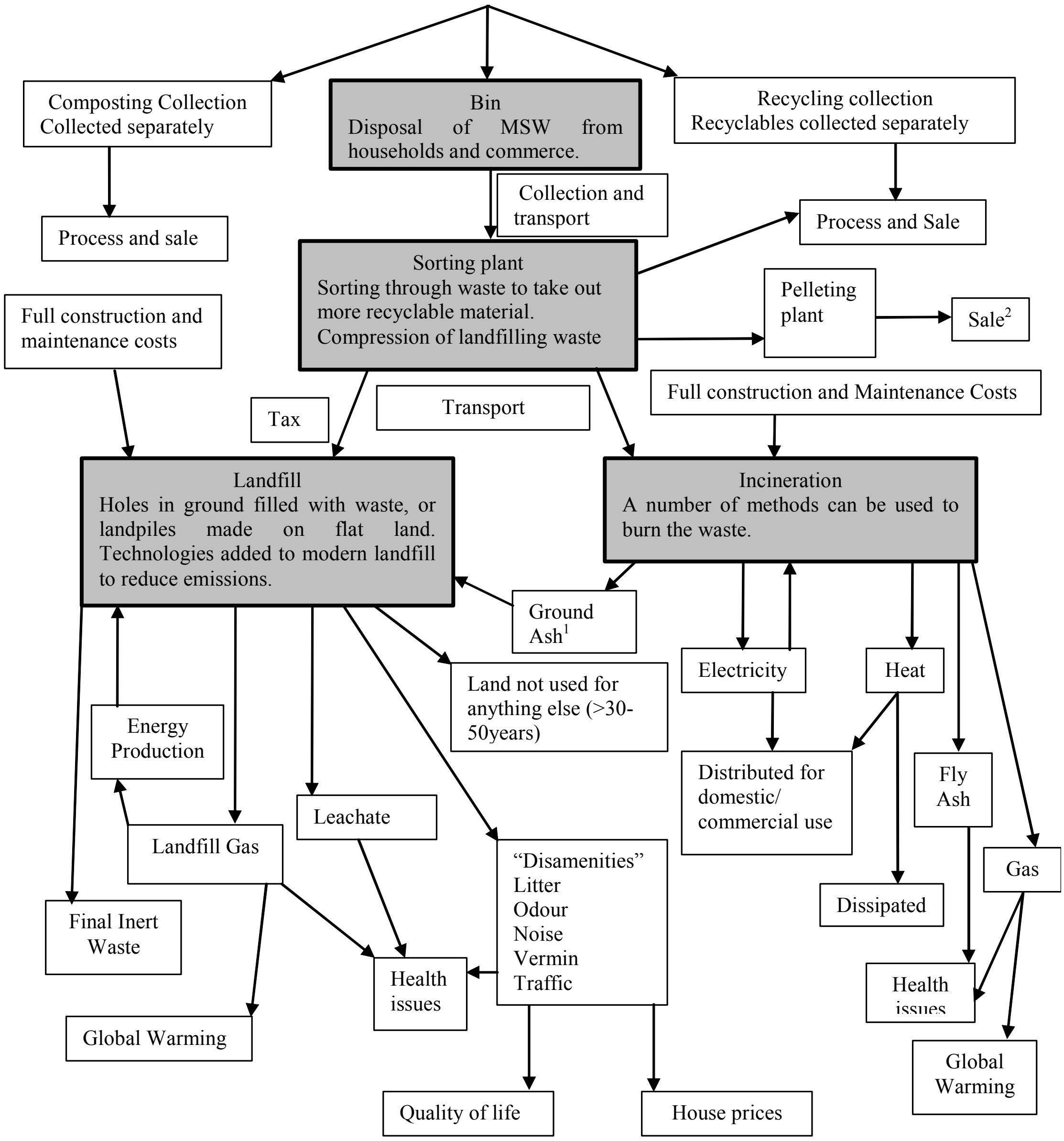
(1) Costs before tax. (2) Costs after tax

4.4 External costs of MSW treatment options

The processes and technologies used generate different external impacts. These external effects are often hard to quantify, as some are political or social phenomena whereas others are scientifically/epidemiologically hard to measure. However, the pathways resulting in effects that may be identified in the literature are shown in Figure 3. This diagram shows the stages from the 'creation' of waste, but a full study of the effects of implementing the waste hierarchy would have to look at the complete life-cycle of the products that are disposed in MSW to identify the effects of waste reduction and recycling. It is important to note that not all the effects are negative – for example energy can be used for local heating or providing energy to the national grid, and the disposal of waste in itself may be seen as a benefit (with the resultant reductions in health and other impacts that result from poor waste management).

We now turn to examine the major impacts, through air and soil/water emissions.

Figure 3: The costs of waste



[1]*This ash will probably be landfilled in a site for hazardous waste, and so be taken out of the MSW pathway.*
[2] *Pelleted waste is often used as fuel (RDF = refuse derived fuel) and burnt for energy but this process is not looked at here since it was not in the Sustools brief.*

4.5 Emissions to air

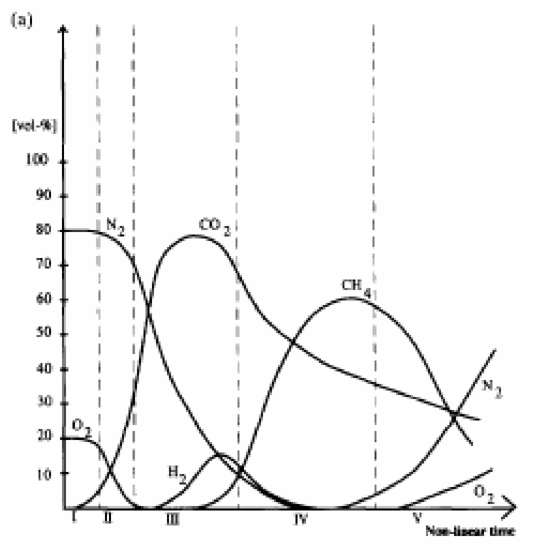
4.5.1 Landfill

As waste in a landfill site biodegrades, landfill gas (LFG) is emitted. This gas is made up of, mainly, methane and CO_2, and so has consequences for global warming. LFG can result even in cases where there is compostible collection[1]. Abatement options include pre-treatment by separation and thermal treatment, or gas collection. Since LFG can be produced years after the landfill is closed, collection systems are more of a short-term solution, but many existing sites do not have adequate facilities. In the USA, landfill sites are the largest source of atmospheric methane, (Atcha and Van Son, 2002). In the UK, it is estimated that up to 40% of atmospheric methane is from this source (Waste Online, 2004).

In addition to methane, up to 130 different chemicals have been identified in LFG, many of which can be hazardous (Hickman, 1988, Gendebien et al, 1992). LFG components can also cause damage to machinery and pipework, including the potential of chlorinated organic compounds to cause engine failure at high concentrations (IEA, 2000:70). LFG odours can be a source of disamenity to local residents and workers, and the build up of gas can cause explosions (SWPHO, 2002). Since the composition of the gas changes over time and depends on the waste in the landfill, there is too much variation to put a simple level of each chemical in a representative quantity of LFG – this would require laboratory experiments for a given mix of waste.

Figure 4 is a schematic representation of the LFG composition over time. A landfill can emit LFG for thousands of years (Finnveden et al 1995:192), so the x-axis on the diagram has magnified the earlier stages for clarity. The 5 stages shown are marked to give a indication of the main gases emitted during that stage, and the main changes in gas composition - ; the initial stage, the aerobic stage, the acid anaerobic stage, the methane producing stage and the maturation stage.

[1] The collection of garden, kitchen and other biodegradable waste which is taken to a composting plant where it is allowed to break down into a substance which can be used as a fertilizer/soil enricher. The technology used in composting can be to simply wait and allow natural processes to occur, or to speed up the process in a number of ways. The gases given off are sometimes collected.



Source: Finnveden et al 1995

Figure 4: Landfill gas composition over time

4.5.2 Incineration

Incineration involves the emission of particulates and chemicals suspended in smoke. These emissions may lead to a range of health and other external impacts, including amenity effects. Abatement can happen throughout the process, in terms of screening the inputs, controlling the incineration process, and in removing particles as they leave the incinerator.

Table 5 shows the main emissions from incineration and landfill, compared to UK long- and short-term air quality standards. This shows incineration complies with air quality standards. However, the regulations concerning the emissions from MSW incinerators are getting stricter all the time, and many organisations point to dangers not shown in these figures, such as the variation in emission rates and the dispersion and deposition effects.

4.5.3 Summary

Concentrations of chemicals from incinerators and landfill are presented inTable 5. These are associated with a range of health effects – notably respiratory impacts and cancers. The external costs related with these are discussed later.

Table 5: Calculation of 'worst case' peak airborne ground level concentrations at 800m from the edge of the waste disposal site.

	Incinerator ($\mu g/m^3$)	*Landfill without gas collection ($\mu g/m^3$)*	*Landfill with 40% gas collection ($\mu g/m^3$)*	*LAQS values ($\mu g/m^3$)long term standards +*	*SAQS values ($\mu g/m^3$)short term standards +*
Particulates	0.7	-	-	40 ##	50##
CO	20	32.4	19.6	550	11600##
VOCs	4	5276	-	14500#	-
SO2	20	-	125	**	125##
HCl	5	-	-	**	500
HF	0.4	-	-	**	30
NO2	80	-	-	40##	200##
Cd	0.02	-	-	1.5	1.5
Hg	0.02	-	-	1	15
Pb	0.42	-	-	0.5##	-
Cu	0.2	-	-	10	200
Cr	0.2	-	-	4.5	-
Ni	0.2	-	-	0.2	6
Mn	0.2	-	-	1	1500
As	0.2	-	-	0.2	60
Sn	0.2	-	-	10	400
Dioxin (ng TEQ)	4.00E-09	4.20E-08	8.00E-09	#	#
Benzene	-	204	122	16.25	160
Vinyl Chloride	-	259	154	60	600
III Trichorethane	-	38.2	23	1110	11110
Tetrachlorethane	-	-	76	3350	33500
Toluene	-	3337	1429	1910	19100
Xylene	-	2650	1135	4410	44100
Dichlorobenzene	-	153	65.3	1500	1500
Ethylbenzene	-	127	545	4410	44100
Styrene	-	625	268	860	8600

LAQS - Long-run Air Quality Standards

SAQS - Short-run Air Quality Standards - This is higher than the LAQS to allow for short bursts of the emission caused by, for example, a change in the waste input composition. However, in the longer-term, such emissions are not tolerated.

* There is no standard for VOCs. The value indicated is based on aliphatic hydrocarbons as represented by n octane

** Short term standard only set

Values are UK Air Quality Objectives

No standard has been set

- No data

Source: Bridges, Bridges and Potter (2001)

4.6 Emissions to soil and water

4.6.1 Landfill

There are a large number of substances that could be emitted from landfills, carried by the leachate that escapes through the soil (see Table 7). Costs of leachate were looked at in the COWI report (2000) for the European Commission which examined three previous studies of landfill leachate costs. Their results are shown in Table 6, but the writers of the report stress that the three are not very comparable. Each study used a different methodology to generate the cost of the leachate. The CSERGE study (1993) used a marginal damage cost approach, which looked at the mortality and morbidity costs of As, Cd, Cr, Cu, Ni, Pb and Hg emissions. Miranda and Hale (1997) based their valuation on the clean up costs, whilst ECON (1995) used a control cost methodology alongside linked environmental values and indices.

Table 6 shows the costs of leachate as estimated by three studies analysed by COWI (2000). These studies used different techniques to analyse the costs and so the results cannot be compared to any great detail. What they do show is that these costs are relatively low, particularly if a landfill site has good leachate collection and treatment systems.

Table 6: Estimated costs of leachate

Substance	*Cost per kg emission to water (€)*[1]	*Cost per kg emission to soil (€)*[1]	*Cost per tonne MSW landfilled (€)*
Leachate			0-1.09[2] 0.77 (0-1.54)[3]
Lead (Pb)	178	5	
Cadmium (Cd)	622	1,514	
Mercury (Hg)	1,022	37	
Dioxins	62,824,889	n.a.	
Antimony (Sb)	121,366	121,366	
Arsenic (As)	308	12	
Barium (Ba)	31	37	
Beryllium (Be)	44,928	44,928	
Copper (Cu)	5	1	
Chromium (Cr)	17,479	320	
Nickel (Ni)	12	4	
Selenium (Se)	16,125	16,125	
Zinc (Zn)	1	1	

[1] Source: COWI: 46, study 1 (CSERGE 1993) based on cleanup costs, £1= 1.71233euro
[2] Source: COWI: 46, study 2, (Miranda and Hale 1997) based on marginal damage cost. Leachate consists of As, Cd, Cr, Cu, Ni, Pb, Hg. £1=1.11495euro
[3] Source: COWI 46, study 3 ECON 1995[2].

4.6.2 Incineration

The major emissions from incineration to soil and water arise from the disposal of ash. This ash often contains the same hazardous chemicals found in landfilled waste, but at more concentrated forms. There are two forms of ash resulting from incineration - fly ash and bottom ash. Fly ash consists of particles caught in the flue, such as smoke particles. These can

[2] It should be noted that the writers of the COWI report regard this study as having a dubious methodology.

be very toxic, and the level of formation depends on the incineration technology and processes, and the abatement technology used. Bottom ash is the waste that, during or after incineration falls to the bottom of the incinerator. This is usually landfilled as a 'final' waste management option, and since it is inert, i.e. it does not decompose, does not add to LFG. However, it often contains toxic metals and is prone to leaching. Therefore it is regarded as hazardous waste and so must be landfilled in special sites. Some sources suggest that bottom ash can be combined with concrete and used as construction aggregate - the concrete eliminates or reduces the threat from leaching.

A UK Environment Agency report into ash generation found that between 2.6 and 3.1 million tonnes of bottom ash and 312,000 tonnes of fly ash was generated in the UK over a five year period from 10.4m tonnes of waste incinerated (Environment Agency, 2002). 79% of the bottom ash was landfilled whilst 21% was treated and went to the following destinations:

- bulk fill, road base, rubble, used on landfill sites - 36%;
- metals extracted - 3%;
- stock pile - 15%;
- blocks - 6%;
- asphalt - 2%;
- and 38% of the reprocessed ash went to landfills (Environment Agency 2002:17)

In the Netherlands, 100% of bottom ash is reused, and 70% in Denmark. By 2000, 42% of bottom ash was being reprocessed in England (Environment Agency, 2002: 18)-. In some cases, fly ash caught in the abatement technology is reprocessed with bottom ash but the quantities involved vary widely and no clear data was available. The conclusions of the Environment Agency are that no clear health risk arose from the storage and treatment of ash (2002:31). Bottom and fly ash have both been used as construction materials by adding to concrete or vitrification at high temperatures of up to 3000^{o}C. Whilst some authors claim this captures the various chemicals and therefore stops leaching, other authors dispute this, particularly when the effects of time and erosion are added. (compare Valenti, 1999 and Allsopp et al, 2001). Allsopp et al (2001) records an incidence when fly and bottom ash was used as fertilizer on allotments. Subsequent analysis found that the allotments were "contaminated with extremely high levels of heavy metals and dioxins." (Allsopp et al, 2001: 12).

Many of the abatement technologies used in incinerators are used to reduce the amount of harmful particles emitted to the atmosphere, not just reduce the global warming potential of the emissions. Therefore these captured particles, i.e. fly ash, contain a high amount of hazardous and toxic chemicals.

Cost estimates done in the past are discussed later in the section *External Cost Estimates*.

4.7 Occupational health

When considering the costs of MSW management the effects on the health of workers must be examined. The numbers of workers at risk and the exposures they face will vary according to the management processes used, implementation of good practices and the composition of the waste. However, there is a lack of data confirming the likely frequency of accidents or infections related to these risks and associated costs estimates. The range of risks faced are outlined below. More detailed information can be found in the ILO's Encyclopaedia of Occupational Health and Safety (ILO, 1998).

4.7.1 Landfill

The following list outlines possible occupational hazards arising on a landfill site. It will not, however, quantify them as they will vary widely across sites due to the working practices and policies at each site. Occupational hazards may arise from the following elements of landfilling MSW:

- Transport - accidents
- Loading and unloading - injuries whilst sheeting, walking on loads, reversing/visibility
- Machinery - such as conveyors, balers
- Slipping and tripping
- Hazards of bacteria and chemicals, dust - can cause asthma, skin irritation.
- Leptospirosis (from rat's urine)
- Manual handling - injuries

4.7.2 Incineration

The risks from the incineration of MSW vary according to the incinerator's operation method and whether any manual sorting of waste occurs at the incinerator site. The more manual the process, the greater the risk of infection from the waste but there are lower risks of injury through machinery. Generally, risks include:

- The heat of an incinerator gives rise to risks of burns, fires, carbon monoxide poisoning.
- Hazardous and poisonous substances may be released during incineration - especially risks to the circulation and respiratory systems
- Physical exertion may lead to back pains, etc.
- The hot, humid environment may cause discomfort.
- Risk of accidents from ladders, slippery surfaces, debris, ashes; risk of cuts, punctures, burns, poisoning.
- Exposure to high levels of noise, heat, and large temperature changes.
- Exposure to MSW and ash with possible health effects resulting such as silicosis, lung function and respiratory system effects, increased blood lead levels (Schilling, 1988 and Malkin et al, 1992 in ILO, 1998)
- Exposure to obnoxious, sometimes offensive odour.
- General tiredness as a result of work in a hot, dirty, noisy environment.
- Stress from low salary, low status, monotony, shift work etc.

4.8 Disamenity

Disamenities can be an important part of the valuation of the effects of waste management. Particularly when a new site is proposed, residents' objections and reactions can add to the cost, for example in public relations or in re-designing parts of the plans, and can even prevent construction altogether.

Whilst there has been little study on the impact of incineration plants and none on waste transfer/MRFs, the relatively large amount of research concerning landfill sites shows that there is a definite cost placed on living near a landfill site. This is often shown through hedonic pricing by a decrease in the house prices near to a landfill site. A CV study carried out in the UK showed that people who were living near a well-established landfill site would be willing to pay to reduce the noticeable effects of odour and litter but not noise (Garrod and Willis, 1998).

Putting a precise figure on the disamenities requires consideration of the local conditions, such as population density, longevity of site, other sources of disamenity and so on. However, the results here show that whilst day-to-day disamenities are quite low (€0.15-0.28 per day per household affected) aggregating up for a whole site, or nation, can lead to a significant cost. Taking the figures from Cambridge Econometrics et al (2003:55), a landfill site in the UK results in disamenity values between €2.38-€3.41 per tonne of waste landfilled[3]. This implies a fixed disamenity of between €532,630 and €750,160 per site.

4.8.1 Incineration

Few studies exist for the amenity impacts of incineration. Kiel and McClain (1995) present an hedonic analysis for a site in North Andover, Massachusetts. They find when construction began in 1981, houses further from the site were more expensive by $2283/mile (€2916/mile in 2000 prices) - this does cover disamenities from the construction itself as well as those associated with the future operation. From the time it went into operation the premium was $8100/mile (€10345/mile) though after the plant had been in operation for four years this fell to $6606/mile (€8437/mile) in the ongoing operation phase. This could mean that full adjustment has not been made or the site is viewed as a continuous disamenity.

4.9 Health impacts

4.9.1 Landfill

As has been shown earlier, the treatment and disposal of waste carries the risk of health impacts on both the workers and the local population. This is due to hazardous substances present in the waste or formed through the treatment processes of combustion or biodegradation. As has been shown earlier, the treatment and disposal of waste carries the risk of health impacts on both the workers and the local population. This is due to hazardous substances present in the waste or formed through the treatment processes of combustion or biodegradation. Table 7 outlines the hazards present in landfilling and incineration plants, but offers little indication of the possible scale of the risks. Many of the emissions for incinerators are captured by state-of-the-art abatement technology and are landfilled in a special hazardous waste site, whereas MSW landfills will emit a larger range of substances through gas, dust or

[3] This is for all landfills regardless of the waste accepted. However, the study found that landfills accepting inert and specialised-biodegradable waste showed "no significant stock disamenity with distance" whilst co-disposal sites which mix MSW and hazardous waste (55% of the sites in the survey) generate similar disamenities to special/hazardous waste sites. Therefore this range can be taken as representative of untreated MSW.

leachate as the table shows. Calculating the quantities of the substances emitted through the different pathways is difficult as they will vary according to the lifecycle of the site, the substances in the MSW, the technology and surroundings of the site and so on. Also, the impact that any substances have is unclear.

A number of studies have looked at the health effects of living near landfill sites. The following have been subjected to debates regarding methodologies applied, but the main findings were:

- Elliott et al, *BMJ* 2001
    - Study found excess risks of congenital abnormalities and low birth rate near landfill sites in Great Britian. However, these were lower than other studies have found.
- Dolk et al, Lancet, 1998
    - This study found a raised risk in congenital abnormalities in babies whose mothers lived near to landfill sites. However, the causal effects of this need to be investigated.
- Vrijheid et al, 2002
    - This study found a higher risk for chromosomal abnormalities in people who lived closer to a hazardous landfill (0-3km) than those who live further away.

The cancer risks of landfill sites are obviously a key factor in determining costs, but are difficult to accurately gauge. Vinyl Chloride and Benzene are known human carcinogens and are emitted from landfill sites. The US EPA's generic risk assessment for landfill sites with gas control found an upper risk estimate of 100 to 10000-cancer cases per million people exposed for a lifetime (1988, in Bridges et al: 331) But this is a questionable estimate. Eschenroeder et al. (1990) assess that the cancer risk for landfills without gas collection is almost two orders of magnitude higher than for incineration but no account is taken of dioxin emissions from landfills. Also, many of the studies (and public perception) are based on sites using out of date technologies which emit more substances than those sites examined in the rest of the report.

The Enviros et al report (2004) carried out a thorough review of health and environmental impacts from waste management practices in the UK, and found that while there is some evidence of congenital birth defects being more likely near to landfills (e.g. Elliot et al 2001) there is no proof that the landfill sites are the cause.

4.9.2 Incineration

There is less evidence on the health effects of incineration than for landfill. Enviros et al (2004: 259) conclude that there is no "consistent evidence for significantly elevated levels of ill-health in populations potentially affected by emissions from MSW incineration," especially since regulation has tightened the emissions standards of incinerators. Whilst there are a number of potentially toxic emissions to air the quantity of these substances emitted is small when compared to other sources such as road transport. The South-West Public Health Observatory also conclude that there is "insufficient" evidence to link health outcomes with either working in or living near a MSW incinerator, though they note that this might be either due to no link existing or the link being "not detectable" using existing methods and data (SWPHO:vi and 31).

Table 7: Health effects of landfill and incineration. Source: from SWPHO 2001

Factor	Landfill	Incineration
Physical Hazards		
Organic chemicals	The main ones are: Polycyclic aromatic hydrocarbons Benzene, benzo(a)anthracene, benzo(a)pyrene, chrysene, heptachlor, polychlorinated biphenyls, tetrachloroethylene, dichlorodiphenyltrichoroethane, trichloroethylene. VOCs PCDD/F dioxins and furans PCBs Alkanes, chlorinated saturated and unsaturated hydrocarbons	Dioxins and furans, PCBs, chlorinated benzenes, halogenated phenols, polychlorinated dibenzothiophenes, PAHs, VOCs
Heavy metals	Chromium Arsenic, cadmium, chromium, mercury and lead	Mercury, lead, cadmium, arsenic, chromium.
Dust	Lead dust Dust from hazardous waste site	Particulate matter
Microbial pathogens	Clostridium botulinum type C	Unlikely to be associated with incineration
Vermin	Risk of spreading disease but low probability of occurrence. Residential complaints about rodents, flies and birds near landfill operations	Unlikely to be associated with incineration
Radionuclides	Radium	Not mentioned
Inorganic compounds	Hydrogen sulphide	Acidic gases -SO2, HCl and HNO3. Acidic aerosols with H2S04.
Road Traffic	Transportation risks associated with removing soil during remediation of hazardous waste site. Residential complaints about landfill operations	Possibility
Fire and explosion	Explosion in Loscoe, Derbyshire. 31 cases in USA 1967-1987	Fire at Dundee incinerator
Psychosocial factors		
Quality of life	Nuisances - bad smells all year round, preventing residents from opening windows or going for walks, windblown rubbish in gardens, hundreds of flies in houses, flocks of seagulls and crows defecating on washing left outside. Residential complaints about odour, litter, noise, hours, mud, dust, traffic near landfill operations. Noise from traffic and machine operation - may lead to disturbed sleep if work starts early morning	Residential perception of neighbourhood quality. Disruption of cohesion, neighbourhood change, increase or decrease in population, lack of community control.

Psychological factors	Concerns, stress, worry by residents living near hazardous waste sites.	Public concerns, distrust of government and scientific institutions, anxiety, stress, feeling of powerlessness and alienation, increase in self esteem and social connection due to involvement in community activism against site.

4.10 Employment

Employment impacts may be important when making a decision on the implementation of a waste management option. As Table 8 shows, some activities are much more labour-intensive than others, with incineration being more labour intensive than landfill per tonne. Of course, if more people are working in waste management, then there are more people exposed to the health and safety risks associated with the sector.

Table 8: Waste management activities and job quantities

Activity	*Jobs per 100,000 tonnes*	*Tonnes per Job*	*Source*
Collection-Packaging	466	214	Cottica and Kaulard (1995)
Reprocessing	162	617	Murray (1998)
Recycling	241	415	Based on Murray (1998)
Landfill	8 - 12	7 885 - 15 246	Cottica and Kaulard (1995)
Incineration	19 - 37	2 692 - 5 397	Cottica and Kaulard (1995)
Aluminium	-	28 - 933	From Murray (1998)
Ferrous Metals	-	162 - 2 102	From Murray (1998)
Incineration	40	2500	Eunomia (2001)

From EC 2001:34 and Eunomia 2001

4.11 External cost estimates

Comparative external cost estimates are presented in a number of studies (ExternE, 1999; COWI, 2000). Table 9 below presents the results from the ExternE study. As can be seen from the table, incineration was found to be generally more costly in terms of environmental impacts than landfill.

The COWI study evaluated external costs by landfill and incineration, using case studies based on compliance or non-compliance with emissions standards. They find that, based on their assumptions, the “best” estimates for net environmental costs for incineration range from –43 Euro/tonne to 77 Euro/tonne. The negative costs show that for some options there is an environmental benefit caused by displacing emissions from more damaging sources of energy, hence these are highly dependent on the source of energy being displaced (eg old coal power plants). For landfill they find a range of 11-20 Euro/tonne. Hence, the analysis is quite sensitive to assumptions on technologies and assumptions on the type of energy being displaced (i.e. if coal then the greater the benefit for incineration).

Table 9 presents the summary of the ExternE (1999) report into the external effects of electricity generation through waste management processes. Despite it being a study into a specific part of MSW management it shows estimates for the costs of other stages, for example transporting waste to the treatment site and the occupational hazards for workers there. It clearly shows that incineration has greater public health costs than landfill, though landfill sites have greater costs for the other stages in the 'fuel cycle'.

Table 9: Damages and avoided costs per tonne of the waste incineration cycle (ExternE, 1999)

			MSW Incinerator 2000€/t	Landfill 2000€/t
POWER GENERATION				
Public Health (mortality and morbidity),			24.12	0.26
of which:				
	TSP		*1.03*	*0.00*
	SO_2		*3.45*	*0.05*
	NO_x		*17.97*	*0.19*
	NO_x via ozone		*1.65*	*0.02*
	Cd-pCDD/F		*0.02*	*nr*
	Benzene		*nr*	*0.00*
Crops			*0.50*	*0.01*
Ecosystems			nq	nq
Materials			0.22	0.01
	Monuments		*nq*	*nq*
Noise			nq	nq
Amenity losses			ng	11.09
Global Warming				
(illustrative restricted range)				
	CO2			
		low	20.16	3.87
		mid	51.52	9.77
	CH4			
		low	nr	7.23
		mid	nr	9.37
OTHER FUEL CYCLE STAGES				
Public Health			*1.49*	*5.70*
	TSP		*1.47*	*4.63*
	SO_x		*0.00*	*0.01*
Road Accidents			0.14	0.69
Occupational Health			0.86	0.08
Ecological Effects			ng	ng
Road damages			nq	nq
Global Warming				
		low	0.01	0.04
		mid	0.03	0.09

Yoll= mortality impacts based on 'years of life lost' approach, VSL - impacts evaluated based on 'value of statistical life' approach

ng: negligible; nq: not quantified; iq: only impact quantified; nr: not relevant

4.12 Overall net economic costs – estimates for Europe

Some of the previous studies (including Coopers and Lybrand, 1993) present data on the net economic costs of waste management options. A summary of the results presented in the Coopers and Lybrand (1993) study is shown in Table 10. It should be noted that this table indicates the range of estimates, which broadly represents the ranking of technologies across countries. This should not be used as a direct representation of which technology is "best" due to the site-specific nature of the cost estimates. In addition, these costs are based on old technologies.

Table 10: Coopers and Lybrand results: Ranges (Euro 2000/tonne)

1993 mixed refuse collection, bring system for recyclable and organic materials	Max	Min
Landfill- no gas recovery	4.6	23.0
Landfill- gas flared	4.6	21.8
Landfill- energy generation (displacing old coal)	3.4	21.8
Landfill- energy generation (displacing average EU electricity)	3.4	21.8
Landfill- no transfer	3.4	20.7
Incineration- electricity generation (displacing old coal)	-28.7	-13.8
Incineration- electricity generation (displacing average EU electricity)	12.6	26.4
Recycling	-324.1	-19.5
Composting	18.4	146.0
1993 co-collection of mixed refuse and recyclable and organic materials (blue box)		
Landfill	3.4	19.5
Incineration- electricity generation (displacing old coal)	-31.0	-12.6
Incineration- electricity generation (displacing average EU electricity)	-23.0	29.9
Recycling	-319.5	-90.8
Composting	3.4	146.0
1993 separate collection of mixed refuse and recyclable and organic materials (wheelie bins)		
Landfill	3.4	16.1
Incineration- electricity generation (displacing old coal)	-31.0	-12.6
Incineration- electricity generation (displacing average EU electricity)	12.6	26.4
Recycling	-264.4	-47.1
Composting	6.9	114.9

Source: Coopers and Lybrand, 1993

4.13 Summary

This section has presented a review of the literature on the costs, both private and external, of landfill and incineration. No attempt has been made to combine this data in the form of a large scale cost-benefit analysis or similar of the waste management system in Europe, because the analysis shows wide discrepancies and uncertainties. First, location is a major determinant of a range of costs – as a consequence the amenity and health impacts in particular are strongly site-specific. In general we can conclude the following:

- Landfill is typically characterised by a lower financial cost than incineration. However, energy recovery may reduce this difference;
- The financial costs vary widely by site and technology – as such extreme care should be taken when transferring cost estimates from one site to another;
- The cost of leachate for landfill is an area of some dispute, though where abatement technologies fail these may be significant;
- There is some evidence of health effects arising from living near landfill. The degree of uncertainty in these varies. These include birth defects (uncertain) and cancers.
- Incineration is more labour intensive than landfill.
- External cost estimates to date show a range of impacts for landfill and incineration. A number of key assumptions underlie these calculations, notably the type of energy being displaced. The estimates presented here cannot be used for a Europe-wide CBA of landfill vs. incineration as the impacts are highly site-specific.

5 COMPARING LANDFILL AND INCINERATION IN FRANCE

At the start of an LCA the boundaries of the analysis must be chosen. For a comparison of landfill and incineration the most appropriate choice is to start at the point where the waste has been collected and sorted. From here the waste must be transported to the landfill or incinerator; we have included the emissions due to possible differences in transport distance by showing a hypothetical difference of 100 km, for the purpose of illustration. In addition to the emission of pollutants from the landfill or incinerator, the emissions avoided by recovery of energy and materials are also taken into account, based on the LCA data of ADEME [2000].

Impacts from residues of incineration and from soil and water pollution by leachates are not included, because these items appear to be negligible for modern landfills. If the landfill is operated according to regulations there are no such impacts during the foreseeable future because the operator has the obligation to maintain and safeguard the facility for 30 years after closure. Even after that period any conceivable leak can affect only the local zone in the vicinity of the landfill and the contribution to the external cost would be negligible (as shown by more detailed studies of specific sites). The assumptions of the analysis are summarized in table 1.

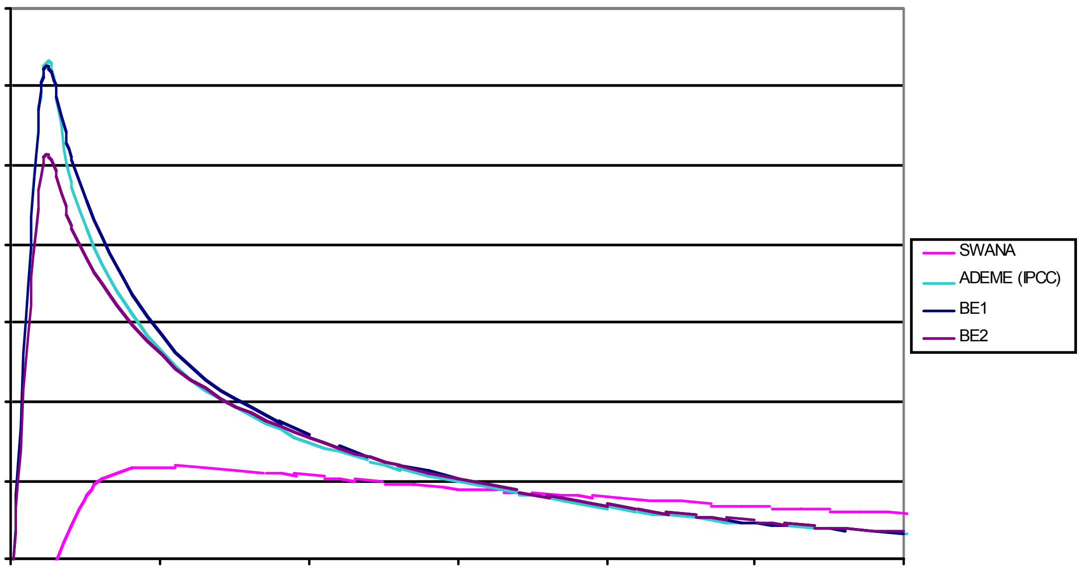
Table 11: Assumptions of the analysis of incineration and landfill of municipal waste.

Stages taken into account in LCA	Transport of waste; Emissions from landfill or incinerator; Avoided emissions due to energy recovery; Avoided emissions due to materials recovery.
Emissions from incinerator	Equal to limit values of Directive EC [2000], in reality the average emissions are lower.
Impact pathway analysis	Assumptions and results of ExternE [2004]
Impacts taken into account	Human health; Losses of agricultural crops; Damage to materials and buildings; Global warming.
Impacts not taken into account	Soil and water pollution due to leachates, probably negligible; Impacts from residues of incineration; Amenity impacts (visual intrusion, noise, odor).

The principal emissions from landfill are CH_4 and CO_2. Figure 2 shows the total greenhouse gas emissions of a municipal solid waste landfill versus time. CH_4 is expressed as equivalent CO_2, using a GWP (global warming potential) of 20. Note that a modern landfill is divided into a large number of individual compartments; they are filled one after another and sealed when they are full. The time in figure 2 is measured from the date that a compartment is sealed.

There may also be emissions to soil and to water. Emissions to soil can occur from slag and from leaking liners, under the landfill and under the storage site of the fly ash from incinerators. Emissions to water arise from certain types of flue gas treatment and from the extraction of leachates under a landfill. Emissions to soil are difficult to estimate because they depend on integrity of the liners in the future. In any case their impacts remain limited to the immediate vicinity of the landfill, and the experience with studies of this type of impact [ExternE 1995] suggest that they are negligible. In the present version impacts of emissions of leachates to soil or water have not yet been included.

a) Calculated by four different models [ADEME [2003].



b) approximation in four steps, according to ADEME [2003], for 1 twaste and 70% recovery of CH4.

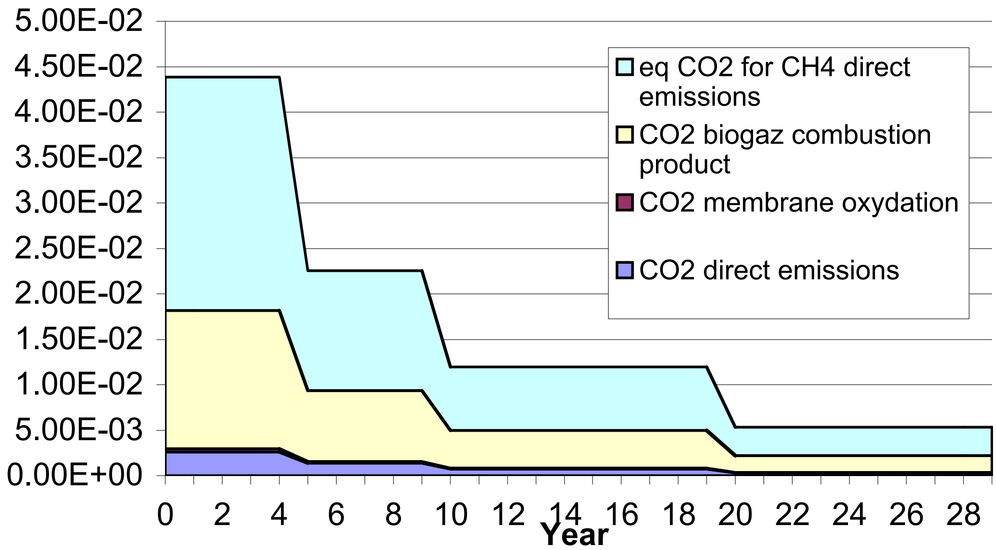


Figure 5: Total greenhouse gas emissions from a municipal solid waste landfill versus time.

The external costs and the comparison between landfill and incineration turn out to be extremely sensitive to assumptions about energy recovery. For that reason we consider a fairly large number of options (indicated in the figures by labels such as E=c&o):

for incineration

- recovery of heat and electricity for typical installations in France, according to ADEME [2000],
- recovery of electricity only,
- recovery of heat and electricity for Saint Ouen (Paris), the installation with the best energy recovery;

for landfill:

- no energy recovery,
- recovery of electricity only, by motor,
- recovery of electricity only, by turbine,
- recovery of heat only.

For each of these we consider several suboptions (indicated by the labels in the captions):

- the recovered electricity displaces coal and oil fired power plants, 50% each (E=c&o),
- the recovered electricity displaces nuclear power plants (E=n),
- the recovered heat displaces gas and oil fired heating systems, 50% each (H=g&o),
- the recovered heat displaces only oil fired heating systems (H=o).

Note that for the purpose of this analysis the benefit of recovered electricity is essentially zero if it displaces nuclear because the damage costs of nuclear are very small compared to those of oil or coal; thus this option is equivalent to no electricity production at all.

A summary of the total damage cost for all the options is shown in figure 3. More detailed results for some of the options can be found in figure 4, showing the contribution of each stage and of the major pollutants (micropollutants such as dioxins and toxic metals are shown as “Other”). For all the options the benefits of materials recovery make a negligible contribution to the total damage cost. The damage costs of waste transport, assumed to be by a 16 tonne truck over 100 km roundtrip, is also negligible. The only significant contributions come from direct emissions (of the landfill or incinerator) and energy recovery.

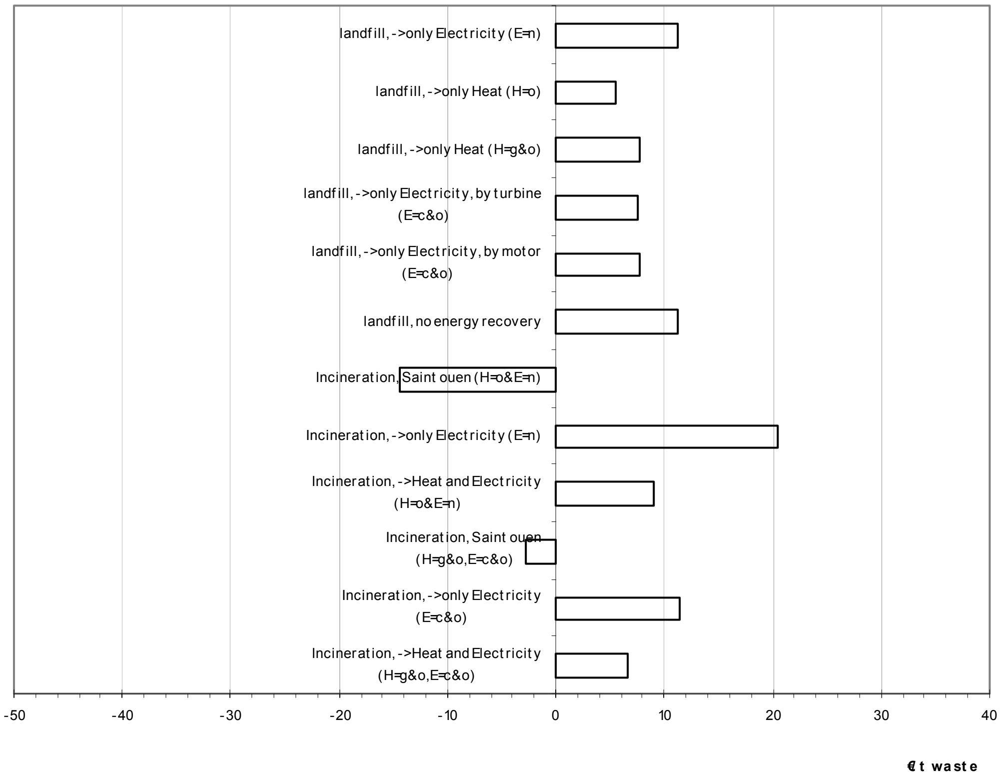
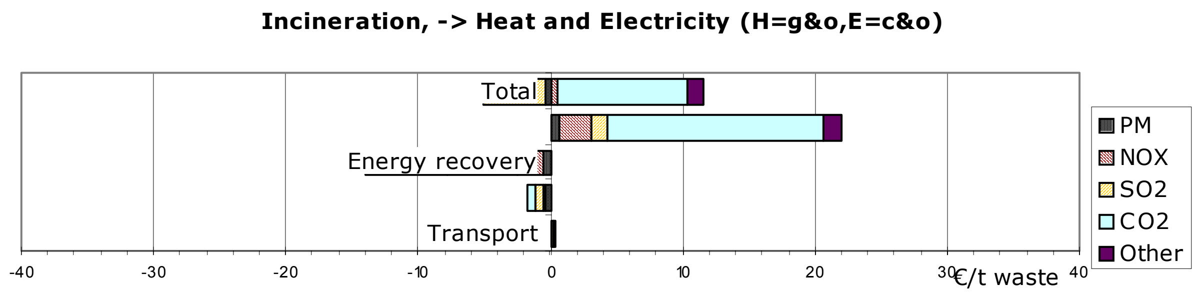
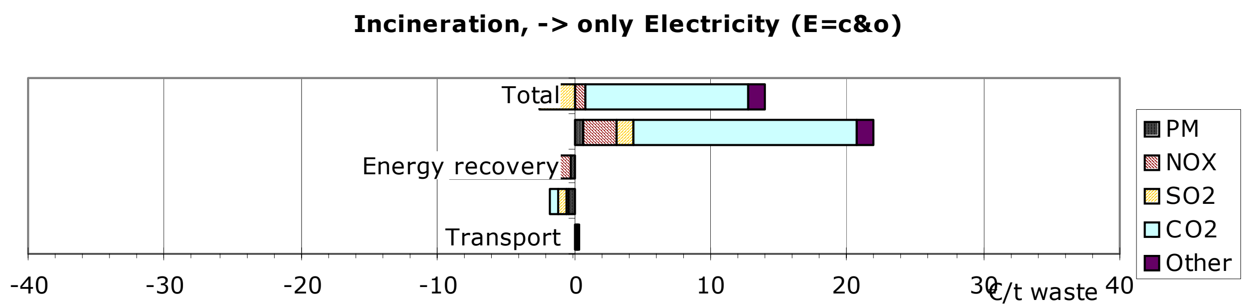
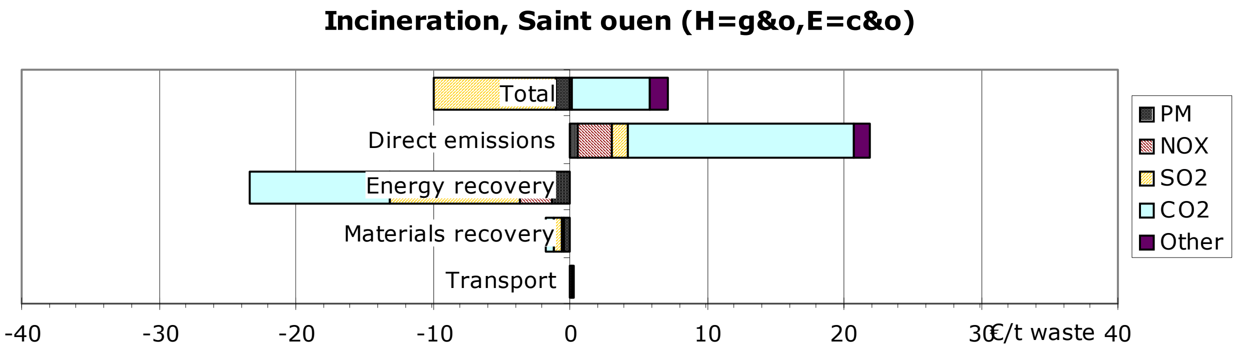
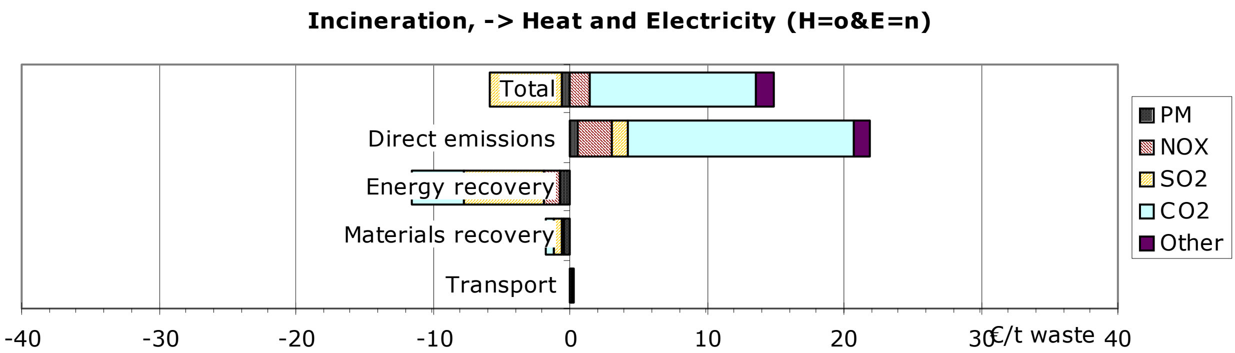


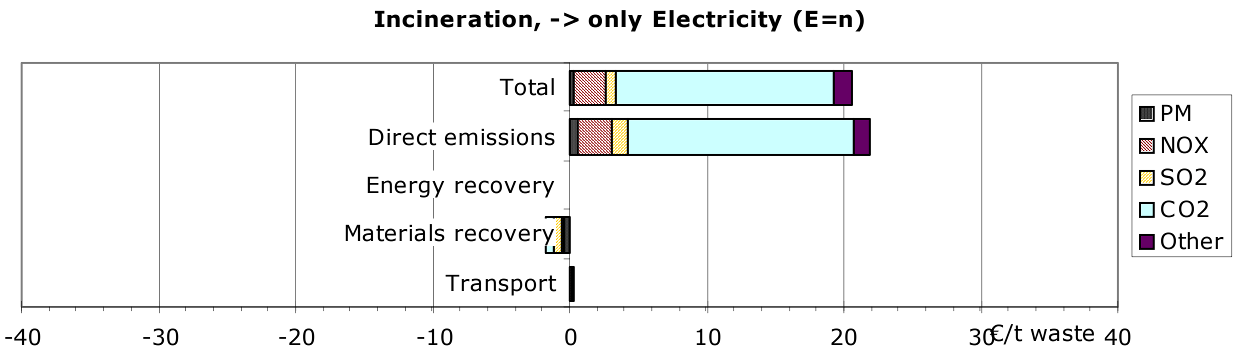
Figure 6: Results of total damage cost for all options.

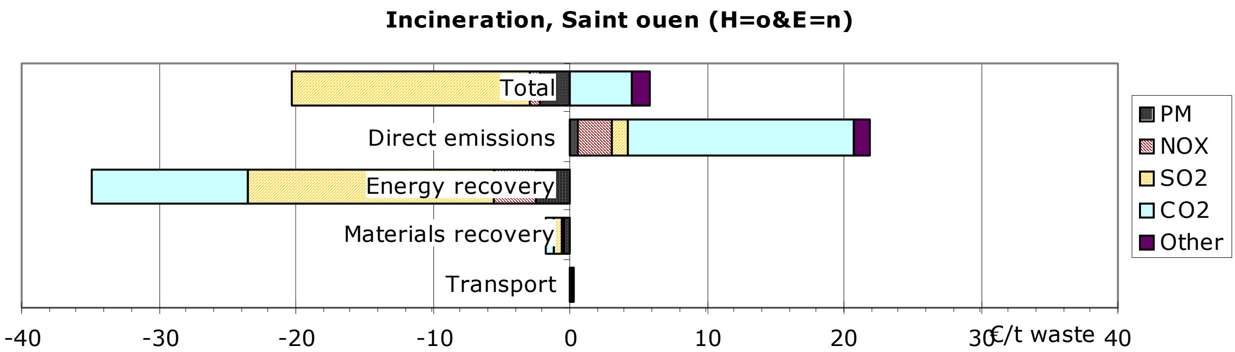




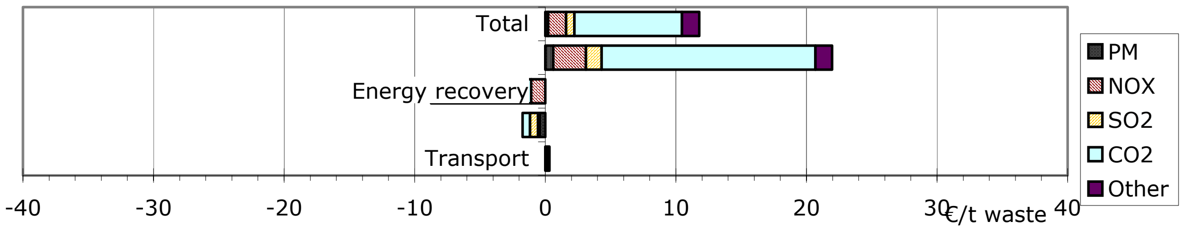

Incineration, Saint ouen (H=g&o,E=c&o)
Total
Direct emissions
Energy recovery
Materials recovery
Transport
PM
NOX
SO2
CO2
Other
-40 -30 -20 -10 0 10 20 30 40
€/t waste


Incineration, -> Heat and Electricity (H=o&E=n)
Total
Direct emissions
Energy recovery
Materials recovery
Transport
PM
NOX
SO2
CO2
Other
-40 -30 -20 -10 0 10 20 30 40
€/t waste

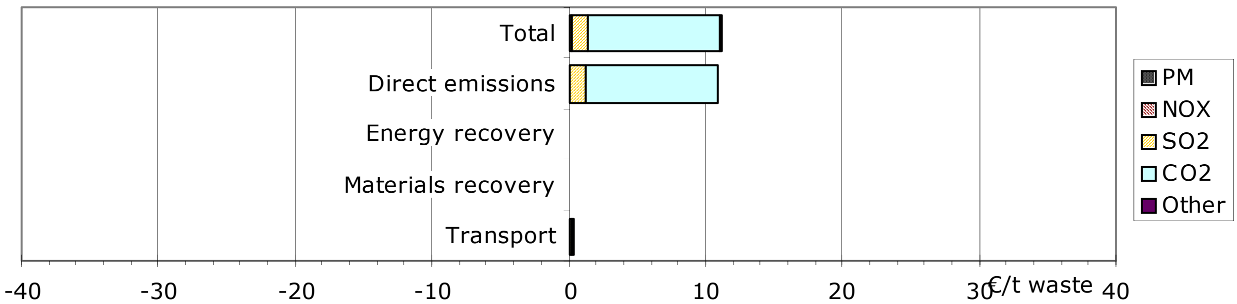

Incineration, -> only Electricity (E=n)
Total
Direct emissions
Energy recovery
Materials recovery
Transport
PM
NOX
SO2
CO2
Other
-40 -30 -20 -10 0 10 20 30 40
€/t waste


Incineration, Saint ouen (H=o&E=n)
Total
Direct emissions
Energy recovery
Materials recovery
Transport
PM
NOX
SO2
CO2
Other
-40 -30 -20 -10 0 10 20 30 40
€/t waste

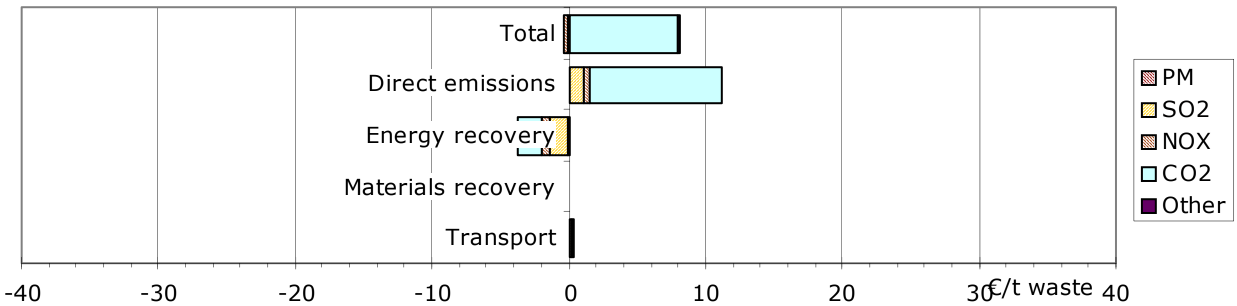
Incineration, Saint ouen (H=g&E=n)



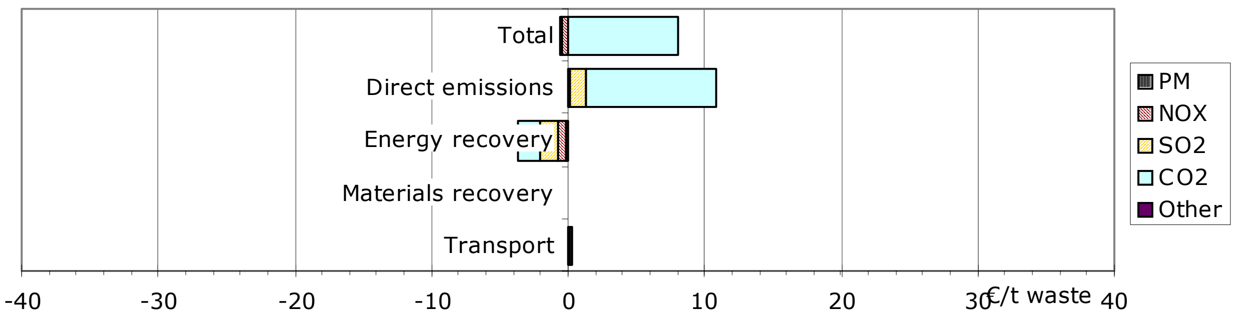
landfill, no energy recovery



landfill, -> only Electricity, by motor (E=c&o)



landfill, -> only Electricity, by turbine (E=c&o)



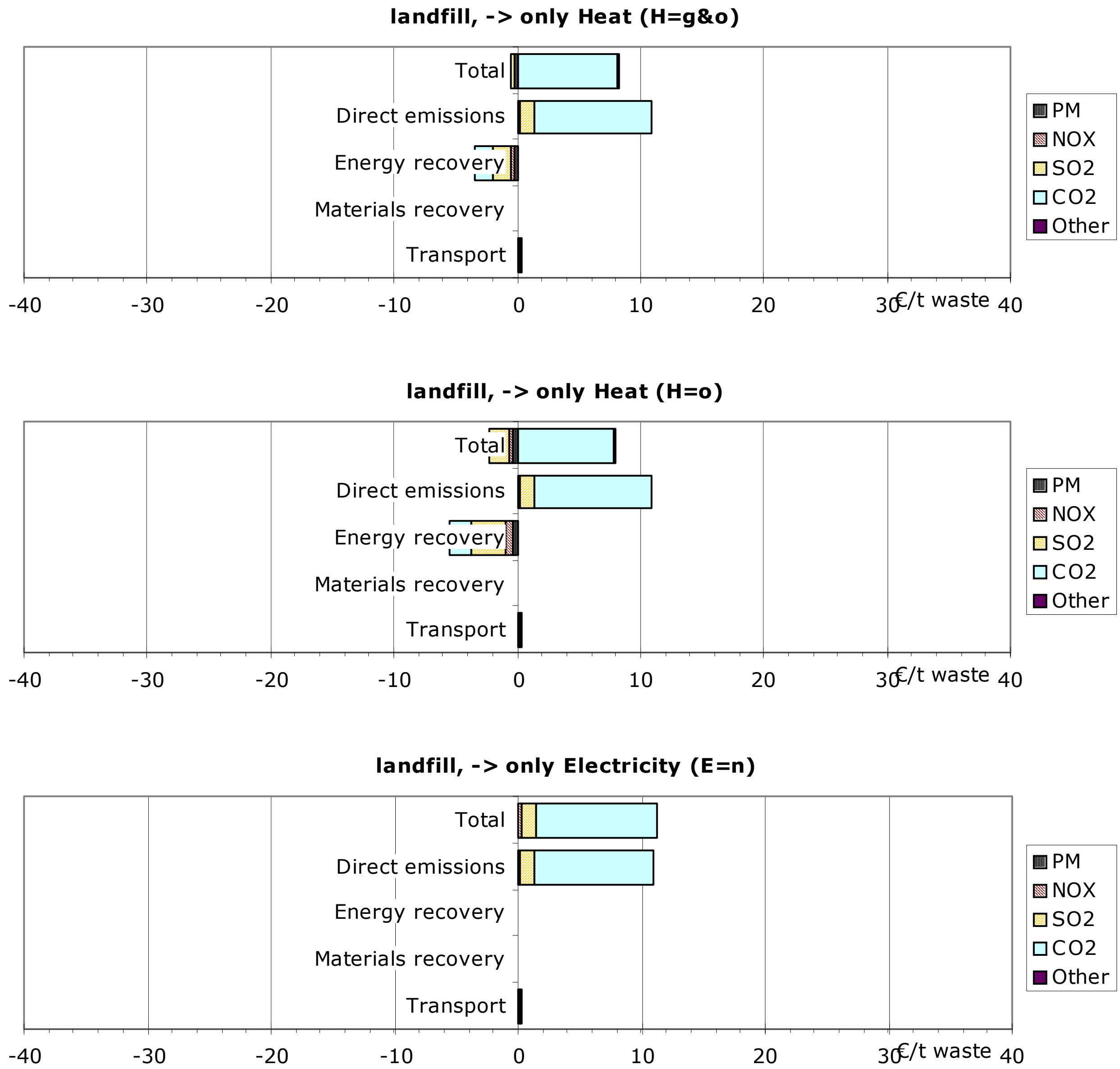


Figure 7: Detailed results, by stage and pollutant. "Other" = dioxins, other organic carcinogens and toxic metals.

For landfill the cost is dominated by greenhouse gas emissions because only about 70% the CH_4 can be captured. Energy recovery from a landfill is not very significant. By contrast, energy recovery is crucial for the damage cost of incineration. Under very favorable conditions (the case for the Saint Ouen incinerator of Paris) the total external cost can even be negative, i.e. a net benefit. Recovery of electricity does not bring significant benefits in France, by contrast to most other countries, because it contributes base load power and all the base load power is produced by nuclear; the options where it displaces coal or oil are not realistic in France because these fuels are used only during the heating season.

The reader should keep in mind that the uncertainties of damage costs are very large, typically a factor of about three (i.e. the true damage could be three times large or smaller than the values shown) [Rabl & Spadaro 1999]. For some pollutants, in particular greenhouse gases and dioxins, the uncertainty is even larger. These uncertainties have different effects on different treatment options. Some comparisons, in particular those between landfill and incineration, are especially sensitive to this kind of uncertainty because greenhouse gases play a larger role for landfills.

6 COMPARING INCINERATION TECHNOLOGIES IN FLANDERS

6.1 Methodology

In 2000 Vito carried out a study on the authority of OVAM (the Flemish waste authority) in which various treatment scenarios for the rest fraction of municipal solid waste where discussed and compared on 5 different criteria (environmental impact, energy, materials, costs and operation) (Vrancken et al., 2000; Theunis et al., 2003). The treatment scenarios that where evaluated involve the processing of waste into electricity, reusable materials, non-reusable materials and emissions. This treatment is performed in integrated process or by means of a combined pre-treatment and thermal valorisation of the refuse derived fuel.

The goal of the study was to make a comparative evaluation of various treatment scenarios that could be implemented in the Flemish waste market on short term and in accordance with the Flemish legal framework. The method as described below is a simplification of the methodology that was developed for the mentioned study. We thank the Flemish waste authority for kindly approving the use of the method in this EU project. We also wish to stress that without the help of technology providers this kind of evaluation is impossible, because of the specific data that are required.

We now use this information to build a life cycle inventory (LCI) of energy and auxiliaries and their life cycle data (emissions, raw materials) that goes into the waste treatment system, and of emissions and products that come out of the system. We do not perform a full LCA, but we will use the data in combination with IPA and the ExternE methodology to derive external costs of the treatment scenarios. We can combine this information with a cost evaluation to make a full cost assessment, and take into account that energy recovery leads to avoiding energy production and emissions elsewhere. We also examine how sensitive the full cost ranking is to the assumptions on avoided electricity mix. We also use emission data to give an assessment of the adherence to limit values for the different incineration technologies. Finally we use the LCI and IPA results in combination with other decision criteria like costs and material recovery to generate values for a wide range of criteria that can be used in a multi-criteria decision analysis (MCDA). Results of the stakeholder workshop and the MCDA exercise are given in the report on MCDA for SusTools. The case study thus demonstrates extensively how different tools work together to give a broad overview of environmental, economical and other impacts of different waste treatment options to solve the policy question of choosing the best technology for the final treatment of municipal solid waste. The broad overview does not give clear and consistent answers, but enables a more transparent and objective multi-criteria decision analysis.

6.2 System boundaries and assumptions

The technologies described are set in a certain context. Pre-treatment technologies that produce a calorific fraction (RDF, fibre, …) are linked to a fluidised bed combustion, whereas other possibilities exist for these fractions, that might or might not be better alternatives. This is however out of scope in this case study. In this case study we propose a focus on:

- grate furnace technologies;
- (biological) pre-treatment with combustion of the produced calorific fractions.

The goal of the evaluation is to compare the burdens and impacts caused during the *disposal of 1 tonne of municipal solid waste*. In this model the residual fraction of municipal solid waste is studied, with a specific composition (see also section 6.3). By this residual fraction of municipal solid waste we understand the fraction that remains after selective collection of specific waste streams. In Flanders following fractions are collected separately : organic waste, metal and plastic beverage packaging, beverage cartons, glass and paper. The European average for the residual fraction is about 320 kg per capita. In Flanders this fraction has been reduced to about 160 kg per capita and per year.The situation in Flanders is perhaps atypical for an assessment in other EU countries, but is representative for a future situation in the EU, and therefore consistent with principles and policies of the EU (see also section 2). The situation in Flanders compared to the EU is shown in the following figure.

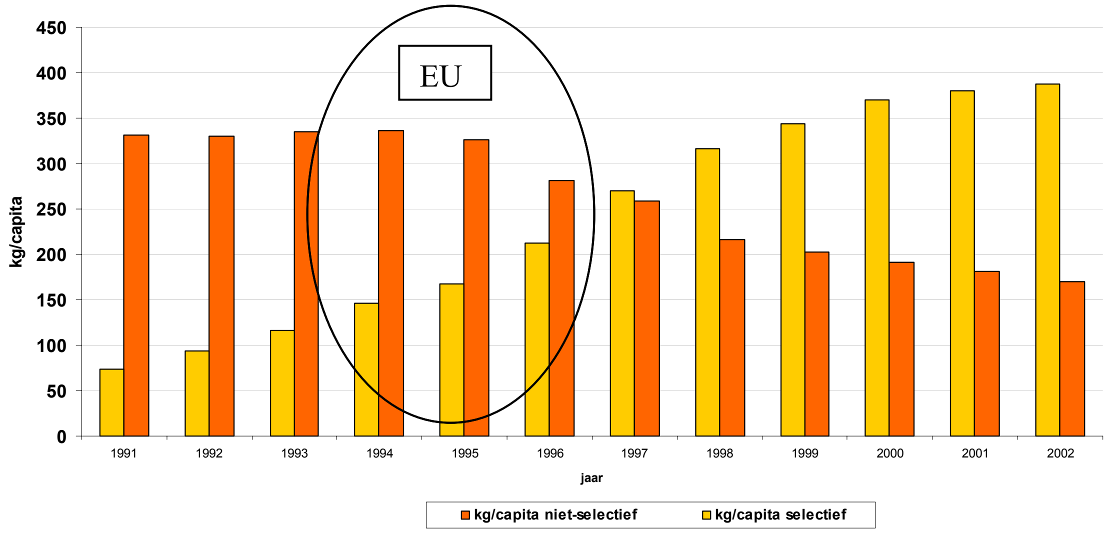


Figure 8 : Evolution of waste collection per capita in Flanders, compared to the EU

In the evaluation the use and processing of auxiliary materials and fuels are taken into account, by making a life cycle inventory of emissions. Together with the emissions (for more details see section 6.5) caused by the processing of the waste itself they form the basis for the comparison. The supply of waste and transportation needed to be able to process the waste (kerbside collection etc.) are not taken into account because they are strongly location related. If transportation of calorific fractions from the pre-treatment plant to the thermal valorisation plant is needed, the emissions caused by this transport are included. In a study VITO made on behalf of LISOM (Theunis, 2003), the impact of transportation of RDF was investigated. It was shown in this study that the transport of RDF over 100 km causes maximum 5% of the total impact on the environment. This leads to the conclusion that transportation of the RDF is not a significant parameter for the evaluation. Energy produced by waste avoids energy production and emissions elsewhere. This has been taken into account as well (section 6.6).

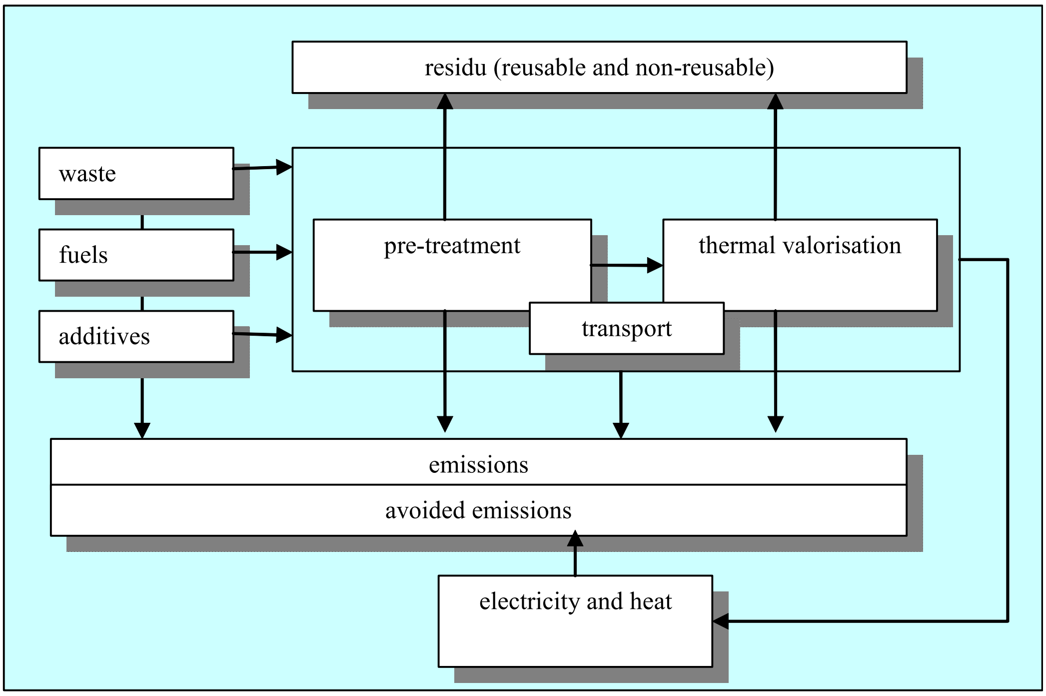


Figure 9: System boundaries

.

6.3 Waste composition

To be able to compare the different techniques or scenarios it is necessary to assume that the incoming waste has a similar and known composition. The composition of the waste has a significant influence on the result of a technique or scenario, especially on the direct emissions and produced energy, but also on the recovery of materials and costs. Not only the relative contribution of the different waste fractions is important but also the elementary composition of these fractions. Following table (Table 12) gives an overview of waste compositions in Europe. This table shows that the waste composition across Europe can differ a lot as shown by the column ‘range EU15’. It also shows that the composition of household waste is changing, especially the amount of organic waste and paper and cardboard.

Table 12: Overview of different municipal waste compositions in Europe

fraction	study Flanders 1997	range EU15 1999	average EU15 1999	average EU15 1995
organic (food and garden waste)	45%	19 – 48%	37%	29%
paper and cardboard	16%	16 – 43%	20%	26%
glass	5%	5 – 13%	7%	7%
metals	5%	3 – 8%	5%	4%
plastics	9%	5 – 14%	10%	9%
textiles	2%	} 9 – 33%	} 21%	5%
other	17%			20%

The waste composition that is used in this caste study (study Flanders) is somewhat different from the EU15 average of 1999 as well as of 1995. It is however within the ranges of the EU15 composition.

Table 12 gives an overview of the waste composition used in this study. This composition was determined by a sorting analysis of residual waste in the spring of 1997. The waste that was analysed originated from Haviland[4] in Flanders. This composition does not respond to the current average composition of residual household waste anymore. The differences can be explained by the fact that the analysis was done in 1997 on the one hand. As shown in the previous table the waste composition is changing rapidly the last few years. On the other hand VITO demonstrated in a study for OVAM that the season as well as the area where the waste is collected has its influence on the composition. This has its influence on the calorific value of the waste. Calorific values of the waste varied between 7,9 en 11,4 MJ/kg depending on the season and the collection aria. This is endorsed by the findings of Göerner. According to his study the lower heating value of municipal waste has increased from about 8 kJ/kg to between 9 and 11 kJ/kg, depending on the county, the catchment area and the pre-treatment of waste. The waste as described below has a calorific value of 8,5 MJ/kg. The calorific value of residual household waste in Flanders is currently around 10,5 MJ/kg. We used this composition nevertheless, because of it's detailed mass balance. This is a great advantage for environmental comparisons.

Effects of the waste composition on the results of the case study can be:

- The use of drier waste with a higher calorific value will have a positive influence on the energy recovery of the evaluated system. A lower moisture content (e.g. because of a lower content of organic matter) can result in a higher production of RDF. The consequence of these higher amounts of RDF is a higher energy recovery. The same goes for MSW with a high content of fractions that end up in the RDF (e.g. plastics and paper);
- Higher amounts of ferrous-, non-ferrous metals and incombustibles lead to lower amounts of RDF, and higher amounts of recovered material or disposables. This leads to lower amounts of recovered energy and therefore less avoided emissions. It can also lead to higher costs (for disposal) and benefits (from material recovery);
- The elementary composition (more specific the carbon and heavy metal content) of the waste has an influence on the direct emissions of a technology or scenario. This is because they are used to calculate the emissions resulting from these elements. The content of heavy metals in the RDF can differ depending on the used pre-treatment technology, depending on which fractions are incinerated or disposed and which are recovered for recycling;
- Higher amounts of organic matter can also lead to a higher production of biogas and/or fibres in installations which produce these fractions. Because of this it is possible that these installations produce more electricity, which qualifies as renewable energy. This can have consequences on the economics of the technology/scenario. A higher production of biogas also leads to a higher overall energy efficiency, because the use of biogas has a high energy efficiency.

These effects will probably not change the ranking of the scenarios. This means that the ranking of the evaluated systems will probably not vary when using a different waste

[4] Haviland Intercommunale is a collaboration agency between municipalities for regional development set up in 1965 by all 35 municipalities of the district of Halle-Vilvoorde (the area surrounding Brussels) and the province of Flemish-Brabant. In the mean time the public welfare centres of 18 municipalities and 3 other intermunicipal agencies of the area joined and the intercommunale offers also services to 8 police zones in the district.

composition for the evaluation. When however different types of waste are used for the different scenarios in the evaluation, it is very likely that the ranking will change or that differences between scenario's are enlarged or reduced.

fraction	weight	water	ash (dry)	HHV(1)	LHV(2)	chloride	sulphate	C	H
	%	%	%	J/g	J/g	mg/l	mg/l	%	%
organic fraction	**45,2**	**75,3**	**24,1**	**16.119**	**15.031**			**37,5**	**5,7**
organic kitchen waste	39,63	75,96	17,62	17.614	2.071	31,16	39,1	41	6,15
garden waste	5,58	70,25	61,43	7.538	325	8,37	33,2	17,5	3,31
paper and cardboard	**16,4**	**15,9**	**10,6**	**17.313**	**13.048**			**39,4**	**6,2**
reusable paper non packaging	5,15	7,24	16,11	15.235	12.811	2,88	11,3	38,8	5,66
reusable paper packaging	2,17	6,27	6,07	24.363	21.356	1,26	5,1	39,7	6,49
reusable cardboard packaging	2,83	6,13	13,1	15.872	13.552	3,41	6,6	35,3	5,85
non reusable paper and cardboard packaging	3,12	27,63	6,65	17.611	10.901	9,38	6,1	43,3	7,44
non reusable paper and cardboard other	3,11	33,93	3,55	16.704	9.339	8,01	6,7	41,4	6,07
glass	**4,8**	**1,1**	**100**	**0**	**-26**			**0**	**0**
glass packaging	4,11	1,25	100		-30				
glass other	0,73		100		0				
metals	**5,1**	**3,8**	**100**	**0**	**-91**			**0**	**0**
metals ferro packaging	2,24	5,27	100		-128				
metals non ferro packaging	0,72	9,94	100		-241				
metals others	2,1	0,02	100		0				
plastics	**9,1**	**10,7**	**3,6**	**39.958**	**33.900**			**71,8**	**7,9**
plastic bottles - PET	0,56	1,06	0,31	22.965	20.997	1,87	5	59,3	7,87
plastic bottles - PVC	0,02	0,63	0,6	23.220	21.357	2,22	5	51,7	7,85
plastic bottles - HDPE	1,06	19,15	8,92	36.571	27.819	5,47	18	68,1	7,28
plastics - foils	3,63	16,44	3,44	42.422	33.700	4,17	6	82,7	7,4
plastics others - packaging	2,53	4,86	2,43	40.763	36.840	6,48	5	61,3	8,79
plastics others - non packaging	1,33	3,48	4,05	42.483	39.318	2,39	5	73,6	7,61
textiles	**2,17**	**22,99**	**2,06**	**19.681**	**12.953**	**4,66**	**26,6**	**46,4**	**9,8**
hazardous household waste	**0,32**	**25,33**	**17,54**	**34.440**	**24.011**	**5,52**	**18,1**	**64,2**	**6,7**
mixed fraction	**7,6**	**35,9**	**22,3**	**16.807**	**8.795**			**38,5**	**8**
mixed fraction - hygienic fraction	4,83	50,12	31,57	13.001	4.442	7,92	11,2	33,9	7,61
mixed fraction - beverage cartons	0,75	17,86	7,66	22.388	16.636	21,1	10,7	45,8	7,37
mixed fraction - other packaging	2,05	8,87	15,23	19.874	16.182	56	7,6	41,9	8,62
others	**6,08**	**5,93**	**22,17**	**23.905**	**21.086**	**3,65**	**34,2**	**42**	**6,13**
inert residue	**3,16**	**34,72**	**62,35**	**6.235**	**2.787**	**7,03**	**39,6**	**18,2**	**3,1**
rubbish bags		**17,09**	**8,2**	**33.190**	**25.778**	**4,53**	**5**	**77,8**	**7,33**
total weighted average	**99,98**	**42,6**	**31,1**	**17740**	**8464**			**36,7**	**5,4**

(1) higher heating value
(2) lower heating value

6.4 Range of application

This method can be used tot evaluate new pre-treatment installations which produce one or more energetic fractions or to evaluate integrated thermal valorisation installations. If necessary, the method can be extended to be able to evaluate new thermal valorisation techniques which are developed to process separated fractions from residual waste. In this case study the method was used to evaluate three grate furnaces and three pre-treatment techniques. The pre-treatment techniques are evaluated with and without valorisation of the thermal fractions.

6.4.1 Integrated system

Three variants of one integrated system are evaluated. The basic technology is a grate furnace with reciprocating grates, energy recovery, and bottom ash treatment. Trucks deliver the waste material in bulk. They transport the waste to a waste reception hall and tip their loads in the bunker. A crane operator carefully spreads the waste supplied over the bunker as efficient mixing provides ideal incineration. The waste from the bunker is filled in the feeding chutes at a controlled rate. The waste drops from the chutes onto push and feed tables, which spread it over the moving incineration grates. Primary air is blown by fans into the areas below the grate, where its distribution can be closely controlled. This primary air is taken from the waste bunker. To secure complete incineration secondary air is blown into the incineration chamber.

- **Flue gas cleaning**

The installed flue gas cleaning system is to a large extend the same for the three systems. All three have a semi-wet DeSOx-process, injection of activated carbon and a fabric filter. The only difference is the installed deNOx installation. In the first variant no deNOx is installed. In the second variant a selective non-catalytic reduction process (SNCR) reduces the nitrogen oxides. The reducing agent that is used is urea. In the third variant NOx are reduced by a selective catalytic reduction (SCR). In this process the reduction agent is ammonia. It is mixed with air and added to the flue-gasses. The flue-gasses are then passed over a catalyst.

Table 13: Grate furnace, emissions to air.

	GF	GF SNCR	GF SCR
emissions	kg/tonne waste	kg/tonne waste	kg/tonne waste
CO2	7.64E+02	7.64E+02	7.64E+02
NOx	2.12E+00	1.06E+00	4.24E-01
SO2	5.30E-02	5.30E-02	2.65E-02
PM	1.22E-02	1.22E-02	1.22E-02
others			
As	5.25E-07	5.25E-07	5.25E-07
Cd	2.76E-07	2.76E-07	2.76E-07
Cr	2.23E-05	1.89E-05	2.23E-05
Pb	5.52E-05	5.52E-05	5.52E-05
Ni	7.08E-06	7.08E-06	7.08E-06
Hg	8.40E-07	8.40E-07	8.40E-07
Dioxins	2.12E-10	2.12E-10	1.80E-10

- **Energy recovery**

During incineration, as much energy as possible is recovered. The heat generated by the incineration process is fed to a steam boiler, which is integrated in the furnace. In this steam boiler steam is produced of 40 bar, at 400°C. Afterward this steam is expanded in a turbine to produce electricity. The system uses oil to start-up. In addition, the grate furnace with selective catalytic reduction of NO_x uses natural gas to start up the SCR-installation. Both the SCR and SNCR use electricity, therefore the net energy production is lower.

Table 14: Grate furnace, energy inputs and output.

	GF	GF SNCR	GF SCR
energy use	MJ/tonne waste	MJ/tonne waste	MJ/tonne waste
oil	169	169	169
gas			289
	kWh/tonne waste	kWh/tonne waste	kWh/tonne waste
electricity		80	85
net energy output	kWh/tonne waste	kWh/tonne waste	kWh/tonne waste
electricity	560	480	475

- **Materials**

During the incineration different types of residues are formed: bottom ashes, fly ashes and reaction products. The incineration ashes are treated in various cut, sieve and wash units. A robust bar sieve first separates the large pieces of metal and stones. A rotary sieve then separates other large pieces, which are de-ironed and sent back to the grate incinerator. The ashes are then separated into three fractions in the wash and sieve unit. Ferrous separators retrieve the iron from the two largest fractions. A non-ferrous separator retrieves mainly aluminium. The inert fraction is converted into granulates, which are used as secondary materials in construction. The smallest fraction is dehydrated and deposited in a landfill class 1 site. The boiler ashes and reaction residues are disposed on a category 1 landfill site. Before disposal the reaction residues are solidified. The ferrous and non ferrous fraction can go to recycling.

Table 15: Grate furnace, material outputs.

	GF	GF SNCR	GF SCR
recuperation	kg/tonne waste	kg/tonne waste	kg/tonne waste
Metals	**34**	**34**	**34**
ferrous	32	32	32
non ferrous	3	3	3
Minerals	**81**	**81**	**81**
bottom ashes	81	81	81
Organics	**n/a**	**n/a**	**n/a**
residue to landfill			
minerals	**150**	**150**	**150**
bottom ashes	112	112	112
fly ashes	21	21	21
flue gas cleaning residue	17	17	17

6.4.2 Pre-treatment 1: SORDISEP

SORDISEP consists of a SORting, DIgestion and SEParation of municipal and industrial waste for the recovery of recyclables and the production of energy by means of the DRANCO digestion process. Ferro, non-ferro and RDF are separated from the waste. The remainder is the digested. During this digestion biogas is formed. After the digestion wet separation is performed.

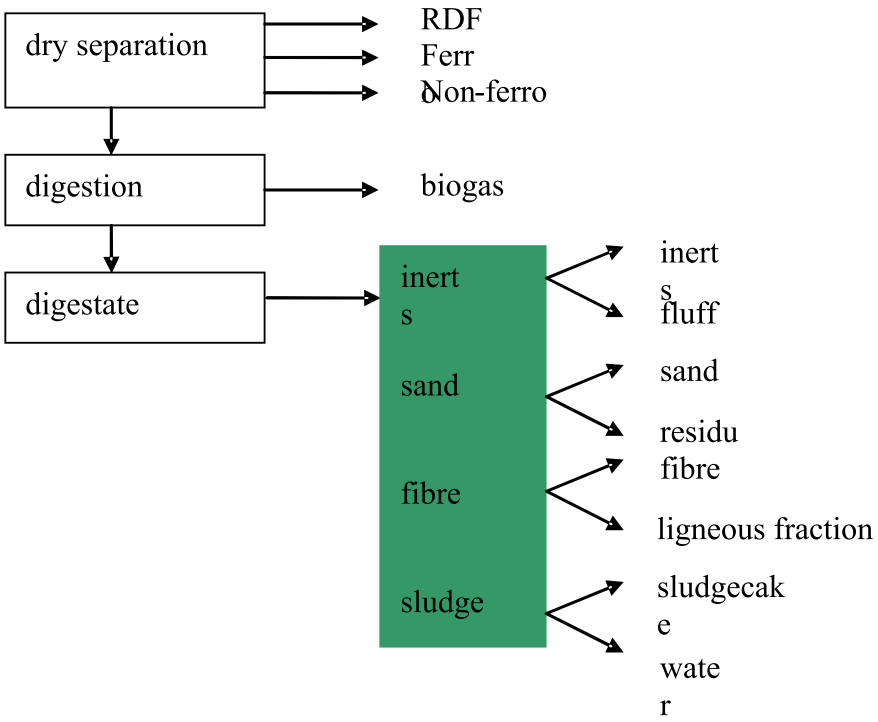


Figure 10 : Flow-chart SORDISEP

- **Flue gas cleaning**

During the digestion biogas is formed. This biogas is burned in a biogas-engine to produce electricity. The flue gases from this engine are cleaned using a SCR-catalyst, activated carbon filtration and thermal oxidation. The SCR reduces the nitrogen oxides (< 100 mg/Nm³). During the thermal oxidation CO and hydrocarbons are reduced (respectively < 100 and 50 mg/Nm³).

Table 16: SORDISEP, emissions.

emissions	kg/tonne waste
CO2	1.81E+02
NOx	7.23E-02
SO2	2.80E-02
PM	0.00E+00
others	
Dioxins	4.00E-11

- **Energy**

The biogas engine produces electricity with an efficiency of 35%. The system uses 84 kWh_e per tonne of waste, which is produced in its own installation. When the biogas production exceeds the available capacity (biogas engine and storage capacity) the excess of biogas is burned off.

Table 17: SORDISEP, energy inputs and output.

energy use	kWh/tonne waste
electricity	84
energy output	kWh/tonne waste
electricity	164

- **Materials**

After digestion, several fractions are produced. The inert and sand fractions can be reused as secondary materials in construction. The fibre fraction can be reused for soil improvement. The wood fraction is thermally valorised together with the produced RDF and fluff. The ferro and non ferro that is separated goes to recycling. The residues need to be disposed on a landfill category 2.

Table 18: SORDISEP, materials.

recovery	kg/tonne waste
Metals	**45**
ferrous	39
non ferrous	6
Minerals	**136**
inert	52
sand	84.2
Organics	**70**
for soil improvement	70
	MJ/tonne waste
Thermal fractions	**7059**
biogas	1816
RDF	5243
residue to landfill	kg/tonne waste
Minerals	**15**
residue	14.9
Organics	**83**
sludge cake	83

6.4.3 Pre-treatment 2: HERHOF

The Herhof-process (or Herhof Stabilat® Process) is a drying process that allows an effective capture of materials for recycling and a valuable fuel product called Trockenstabilat®.

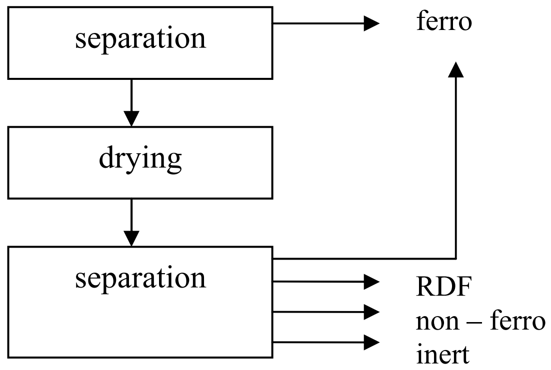


Figure 11 : Flow-chart HERHOF

- **Flue gas cleaning**

Because of repeated recirculation of air in this process the concentration of carbon compounds increases. A thermal oxidizer oxidises the organic compounds in a combustion chamber. Clean air contents of < 5 mg TOC/m3 waste air can be guaranteed.

Table 19: HERHOF, emissions.

emissions	kg/tonne waste
CO2	1.13E+02
PM	2.70E-03
others	
Hg	1.00E-05
Dioxins	6.00E-12

- **Energy**

During the Herhof-process there is no production of electricity. The installation however uses electricity, gas and diesel.

Table 20: HERHOF, energy inputs and output.

energy use	MJ/tonne waste
gas	141
diesel	7.2
	kWh/tonne waste
electricity	100
energy output	kWh/tonne waste
electricity	0

- **Materials**

All the fractions that are separated in this process can theoretically be reused. The ferrous and non ferrous metals can go to recycling. The inert fraction can be reused in construction. The separated RDF can be thermally valorised.

Table 21: HERHOF, materials.

recuperation	kg/tonne waste
Metals	**49**
ferrous	39
non ferrous	10
Minerals	**61**
inerts	61
Organics	**n/a**
	MJ/tonne waste
Thermal fractions	**8887**
RDF	8887
residue to landfill	kg/tonne waste
minerals	**0**

6.4.4 Pre-treatment 3: FIBRECYCLE

Here the waste is treated in an autoclave with saturated steam. Afterward the waste is separated. In the scenario as described below is assumed that all thermal fractions are thermally valorised. According to the supplier it is however possible to recycle the fibre fraction.

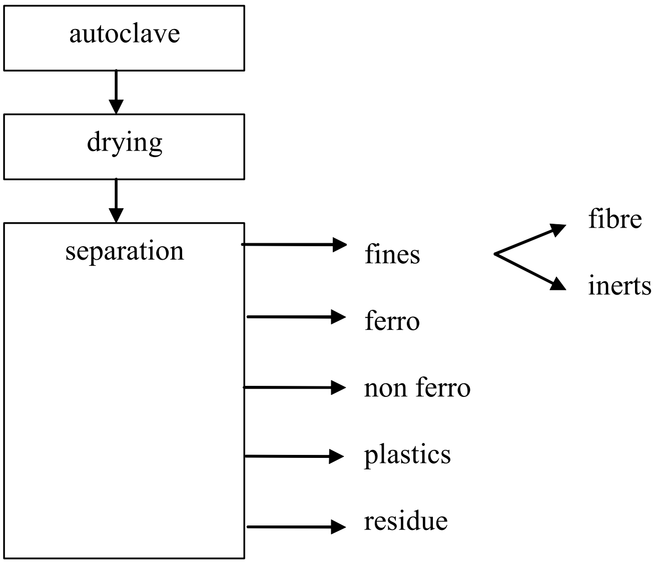


Figure 12 : Flow-chart FibreCycle

- **Flue gas cleaning**

During this process there are, according to the supplier, no emissions to air. Emissions from the use of gas and oil are not reported.

▪ **Energy**

During this process there is no production of energy. The installation however uses electricity, oil and. a significant amount of natural gas.

Table 22: FibreCycle, energy inputs.

energy use	MJ/tonne waste
oil	35
gas	1522
	kWh/tonne waste
electricity	106
energy output	kWh/tonne waste
electricity	0

▪ **Materials**

The fine fraction is separated from the material that is discharged from the autoclave. This fine fraction is then separated in a fibre fraction and an inert fraction. The fibre fraction can be reused of thermal valorised. The inert fraction can be used in construction. Ferro and non ferro are removed from the oversized fraction. These two fractions can be recycled. Afterwards a plastic fraction is separated, that can be thermal valorised in a CFB. The remainder is a residue. This residue can be valorised in a grate furnace. In the basic scenario all thermal fractions are valorised. In the sensitivity analysis the fibre fraction is being reused.

Table 23: FibreCycle, materials.

recuperation	kg/tonne waste
Metals	**46**
ferrous	39
non ferrous	7
Minerals	**48**
inerts	48
Organics	**n/a**
	MJ/tonne waste
Thermal fractions	**5981**
fibre	5981
plastics	2505
residue	672
residue to landfill	kg/tonne waste
minerals	**0**

6.4.5 Thermal valorisation : CFB

To process RDF from municipal solid waste an external circulating bed is the optimal type of fluidised bed. This is due to its high flexibility and aptitude to process fuels with high calorific value. A fluidised bed incinerator is a lined combustion chamber in the form of a vertical cylinder. In the lower section, a bed of inert material (e.g. sand) on a grate or distribution plate is fluidised. The waste is continuously fed into the fluidised sand bed. The temperature in the free space above the bed is generally between 850 and 950 °C. In the bed itself the temperature is lower, around 650 °C. Because of the well-mixed nature of the reactor, fluidised bed incineration systems generally have a uniform distribution of temperatures and oxygen, which results in stable operation. During incineration the fluidised bed contains the unburned waste and the ash produced. Because of the high speed of the gas, the greater part of the solid material particles are removed from the bed chamber with the flue-gas. The ashes in the flue-gasses are then separated from the sand in a downstream cyclone. The sand is then returned to the bed.

- **Flue gas cleaning**

The installed flue gas cleaning systems includes a cyclone or ESP, a wet scrubber, a SNCR, activated carbon filtration an a fabric filter.

Table 24: Pre-treatment + CFB, emissions.

	pre 1 + CFB	pre 2 +CFB	pre 3 + CFB
emissions	kg/tonne waste		
CO2	4.56E+02	7.65E+02	7.56E+02
NOx	4.53E-01	6.01E-01	6.97E-01
SO2	5.73E-02	4.62E-02	5.07E-02
PM	6.74E-03	1.33E-02	1.04E-02
others			
As	1.55E-07	2.26E-07	2.86E-07
Cd	2.18E-07	5.41E-07	4.37E-07
Cr	1.20E-05	1.54E-05	1.75E-05
Pb	3.96E-05	4.22E-05	6.07E-05
Ni	2.30E-06	5.54E-06	6.86E-06
Hg	1.09E-06	1.14E-05	1.46E-06
Dioxins	1.57E-10	1.91E-10	2.03E-10

- **Energy**

During incineration, as much energy as possible is recuperated. The heat generated by the incineration process is fed to a steam boiler, which is integrated in the furnace. In this steam boiler steam is produced of 40 bar, at 400°C. Afterward this steam is expanded in a turbine to produce electricity. The fluidised bed boiler uses approximately 20 MJ natural gas per tonne RDF. In addition, the system uses 142 kWh_e per tonne of RDF. This electricity is produced in the installation itself.

Table 25: Pre-treatment + CFB, energy inputs and output.

	pre 1 + CFB	pre 2 +CFB	pre 3 + CFB
energy use	MJ/tonne waste		
oil			35
gas	6	150	1522
diesel		7.2	
	kWh/tonne waste		
electricity	128	169	106
energy output	kWh/tonne waste		
electricity	531	522	601

- **Materials**

During incineration and flue gas cleaning several residues are formed: bed and boiler ash, cyclone ash and flue gas cleaning residue. The bed and boiler ash can be reused as a secondary material in construction. The cyclone ash and flue gas cleaning residue need to be land filled.

Table 26: Pre-treatment + CFB, materials.

	pre 1 + CFB	pre 2 +CFB	pre 3 + CFB
recovery	kg/tonne waste		
Metals	**45**	**49**	**46**
ferrous	39	39	39
non ferrous	6	10	7
Minerals	**160**	**101**	**92**
inert	52	61	48
sand	84.2		
bottom ashes			6
bed ashes	15	26	25
boiler ashes	8.5	14	14
Organics	**70**	**n/a**	**n/a**
for soil improvement	70		
residue to landfill			
Minerals	**42**	**69**	**53**
residue	14.9		
bottom ashes			8
fly ashes			2
cyclone ashes	26	45	43
flue gas cleaning residue	15	25	25
Organics	**83**		
sludge cake	83		

6.5 Emissions

For the evaluation of pre-treatment techniques the supplier of the technique is asked to give emission data. The data they have to provide refer to the pre-treatment installation, and not to the installations in which the formed thermal fractions are being processed.

In what follows is explained how the emissions caused by the thermal valorisation are determined. A distinction is made between emissions that mainly depend on the composition of the fuel (CO_2 and heavy metals) and emissions that depend on the process control. Emission data for additives and fuels are taken form LCA databases (Simapro database).

6.5.1 Heavy metals

- **Determination of the heavy metal content of the energetic fractions**

Emissions of heavy metals are related to the content of heavy metals in the combusted fractions. We made a metal balance for the different energetic fractions that are separated during the pre-treatment. This balance is made using data provided by the supplier concerning the composition of the produced fractions. When the supplier did not have a composition of the different fractions we made assumptions concerning the composition. The composition of the different fractions used here are given in Table 27.

Table 27: Composition of the calorific fractions.

kg/ton waste	WASTE	OWS			HERHOF	FIBRCYCLE		
fraction		RDF	residue	wood fract	RDF	fibre	plastics	residue
organic fraction	128,18	4,18	1,12	0,22	103,22	128,18	0,00	0,00
garden waste	4,56	1,02	0,00	0,04	3,67	4,56	0,00	0,00
wood	32,00	28,59	1,24	1,21	25,77	32,00	0,00	0,00
paper and cardboard	134,48	16,50	0,00	0,67	108,29	134,48	0,00	0,00
glass	45,60	3,21	3,84	0,29	8,05	10,40	0,00	0,00
metals - ferro	38,00	6,02	0,00	0,25	0,99	0,85	0,00	0,00
metals - non ferro	10,45	1,76	0,10	0,08	1,20	4,28	0,00	0,00
plastics	78,26	66,99	6,45	2,99	78,26	1,13	77,13	0,00
textiles	15,40	14,72	0,44	0,62	15,40	0,00	0,00	15,40
leather - rubber	9,00	8,92	0,33	0,38	9,00	0,00	0,00	9,00
carpets	8,50	8,33	0,00	0,34	8,50	0,00	0,00	8,50
hazardous houselhold waste	2,25	0,42	1,40	0,07	2,25	0,00	0,00	2,25
mixed fraction - hygenic fraction	23,04	0,93	0,00	0,04	23,04	21,62	0,00	1,42
mixed fraction - beverage cartons	6,56	1,88	0,53	0,10	5,28	0,00	0,00	6,56
others - combustible	7,90	3,78	3,30	0,29	7,90	0,00	0,00	7,90
others - non combustible	8,50	2,01	4,25	0,25	8,50	0,00	0,00	8,50
inerts	20,80	10,75	0,14	0,44	3,67	10,40	0,00	0,00
DM	573,48	180,04	23,13	8,26	413,00	347,90	77,13	59,53
wet	1000	257,20	37,30	14	486	386	90	103
DM%	57%	70%	62%	59%	85%	90%	86%	58%

Table 28: Heavy metal content of the different components of MSW.

mg/kg DM	**As**	**Co**	**Cu**	**Hg**	**Ni**	**V**	**Zn**	**Pb**	**Cd**	**Cr**
organic fraction	6,3	36	73	0,3	220	29	362	176	0,5	204
garden waste	0,6	7	18	0,1	16	1	157	277	1,2	58
wood	0,6	7	43	0,1	16	1	157	277	1,2	58
paper and cardboard	0,2	2	73	0,1	9	0	138	13	1,1	7
glass	28,7	8	24	0,0	19	7	91	381	1	253
metals - ferro	54,0	64	6116	0,0	744	276	589	52	1	1730
metals - non ferro	4,8	11	146000	0,0	582	35	13615	11635	1,9	331
plastics	0,3	6	256	0,1	54	1	404	302	38,8	85
textiles	0,2	3	67	0,1	15	1	315	42	1,6	93
leather - rubber	1,3	3	41	0,0	16	2	3886	190	38	1995
carpets	1,5	9	32	0,0	56	4	980	120	9,5	76
hazardous houselhold waste	26,0	127	2295	267,0	605	0	76000	0	99	0
mixed fraction - hygenic fraction	0,2	2	41	0,1	9	0	138	13	1,2	7
mixed fraction - beverage cartons	0,2	2	41	0,1	9	0	138	13	1,1	7
others - combustible	0,0	0	0	0,0	0	0	0	4000	0	0
others - non combustible	0,0	0	0	0,0	0	0	0	2000	0	0
inerts	9,1	45	100	0,0	31	43	565	1967	34	82

- **Calculation of the emissions**

The emissions of heavy metals formed during the combustion of energetic fractions are calculated. This calculation is based on the content of heavy metals in the combusted fraction and element transfer coefficients to the exhaust gas. This transfer coefficient depends on the flue gas cleaning installation that is used. The transfer coefficients that are used in these calculations are given in Table 29.

Table 29: Transfer coefficients to exhaust gas for heavy metals.

	fraction emitted to air
Cd	0,01
Hg	0,22
Pb	0,0329
Cr	0,028
Cr(III)	0,028
Cr (VI)	0,028
Cu	0,0883
As	0,02
Co	0,0059
Ni	0,017
V	0,1
Zn	0,01

6.5.2 CO2

- **Determination of the C content of the energetic fractions**

The carbon content of the different energetic fractions that are separated was also calculated. The same method was used as described in section . The C-content of the different fractions is given in Table 12.

- **Calculation of the emissions**

C-emissions are calculated using the same method as described above. The calculation is based on the C content of the different fractions and the element transfer coefficients tot the exhaust gas. The transfer coefficient used for C is 98%. This means that 98% of the carbon content of the combusted fraction is emitted as CO_2.

6.5.3 Remaining emissions

Some emissions (NO_x, SO_2, dust and dioxins) are not entirely related to the composition of the combusted fractions. They also depend on process conditions and flue gas cleaning. For these emissions the data that are used are given in the following table.

Table 30: Used emission data to air for NOx, SO2, dioxin and dust.

emission data		GF	GF SNCR	GF SCR	CFB
NOx	mg/Nm³	400	200	80	130
SO2	mg/Nm³	10	10	5	10
PM	mg/Nm³	2,3	2,3	2,3	2,3
dioxins	ng TEQ/Nm³	0,04	0,04	0,034	0,04

The data used for NO_x are based on the flue gas cleaning system that is installed and, for the CFB, emission limit values that are set for NO_x for a new CFB that is being build in Flanders. The values for SO_x, dust an dioxins that are used are based on average measured emission values.

6.6 Avoided emissions from electricity production

When producing a certain amount of electricity from waste, then -under constant electricity demand- production of electricity elsewhere is avoided. A lot of discussion is then spend on how to include this benefit from waste production. In real circumstances the instant marginal technology for electricity production is the one that is avoided by waste to energy production. This depends on time of day, time of year and marginal price considerations. For a theoretical model this kind of information is not available and mostly not very helpful. Therefore we consider the avoidance of energy production in general terms. We assume an average kWh is to be avoided. Because we also use yearly averaged waste and emission data, this is a good basis for comparison. The average kWh however changes from year to year and from country to country. The use of limit values (from the Large Combustion Plant Directive) is better suitable for policy decisions. They are the same in every country and they provide a longer term perspective for comparison. However limit values tend to overestimate the effect of avoided emissions by an order of magnitude! Real emissions, like particle emissions, are very often much lower then the limits. results are given in section 6.14.

6.7 Costs

6.7.1 Methodology

The methodology that is used is based on the method environmental costs of the VROM and the directives of the European Environmental Agency. These Guidelines aim to establish a common framework and vocabulary for documenting and using data on the costs of possible environmental protection measures to define a minimum set of information which will enable data users to understand the contexts in which data comparisons are valid or not valid. For the calculations an estimation of the investment costs and the operation cost is made. The costs are given as annual cost, and final as cost per tonne. Annual cost are those cost which can be assigned to a specific year. To do so, the investment expenses are converted to annual investment costs. Besides these investment costs the annual costs enclose operational costs. These operational costs are corrected to include savings and/or profits. Investment costs are the purchasing costs and all additional cost. The purchasing cost is the price that has to be paid by the buyer of the installation. Additional costs are costs that have to be made to make installation operational, such as preparation cost, start up cost,... These additional costs can vary from 30 to 250 percent of the purchasing cost. Operational cost are all other costs that have to be made to make and keep the installation operational. In most cases the exploitation of an installation leads to profits and/or savings which are referred to as negative costs. The total annual cost is calculated using following formula:

$$\text{Total annual cost} = I_0 \left[\frac{r(1+r)^x}{(1+r)^n - 1} \right] + OC$$

I_o = total investment expenses
OC = annual operational cost
r = discount rate
n = economical life span of the installation

6.7.2 Data

Data related to cost depend strongly on the given context. They need to be taken with the greatest caution. To be able to use similar, recent and reliable date, we asked suppliers to fill in a questionnaire regarding costs. These data where used as a base for the calculations. They where supplemented and refined with information gathered during visits to companies, literature and own estimations. In Table 31 the different calculated costs for the treatment of one tonne of waste in the different scenarios are given. The costs that are used refer to the year 2000. Each scenario includes an overhead cost of 15%. The prices for ferrous and not ferrous metals that are used in the calculations are 45 and 496 €/tonne respectively for the thermal scenarios (grate furnace and Fibrecycle). The prices used for the mechanical pre-treatment scenarios are 35 €/tonne for the ferrous fraction and 390 €/tonne for the non ferrous fraction. These prices were given by different suppliers. These data where the most realistic data that were available, and refer to the year 2000. More recent data where not obtained. The costs that are used for landfilling are 150 €/tonne for a landfill for hazardous waste and 50 €/tonne for a landfill for non hazardous waste (see also section 6.8). These are also costs that refer to the year 2000. As for the ferrous and non ferrous fraction, these data are the most realistic data available. All the cost calculations have to be taken with the greatest cautions. Crucial data on investment costs and operational costs have been provided by the technology suppliers who all reported in their own way, and have not been compared to the costs of actual projects. Costs do not include taxes, levies or subsidies. Neither do they include the cost of capital or a risk premium. The calculated cost is not a full commercial market price. Revenues caused by the sale of recovered products by the grate furnace scenarios are not given in following table. The benefits from the sale of ferrous and non ferrous metals are included in the operational cost.

Table 31: Comparison of calculated costs for the different scenarios.

	GF	GF-SNCR	GF-SCR	pre 1	pre 2	pre 3	pre 1+ CFB	pre 2 + CFB	pre 3 + CFB
COST									
Capital cost	34,03	34,63	37,15	25,52	22,81	9,85	39,78	45,3	35,7
pre-treatment				25,52	22,81	9,85	25,52	22,81	9,85
final treatment							14,25	22,49	22,03
Operational cost	38,45	38,74	39,96	33,34	26,02	15	46,28	45,36	41
pre-treatment				33,34	26,02	15	33,34	26,02	18,17
final treatment							12,96	19,34	18,72
Overhead cost	8,79	9,22	9,8	7,69	6,69	2,98	10,61	11,18	8,52
Total cost	81,27	82,59	86,91	66,55	55,53	27,83	96,67	101,85	85,23
REVENUE									
Sales recovered products				-3,56	-4,22	-4,97	-3,56	-4,22	-4,97
Electr.production	-13,88	-11,9	-11,77	-4,07			-13,16	-12,94	-14,91
Total revenue	-13,88	-11,9	-11,77	-7,63	-4,22	-4,97	-16,73	-17,16	-19,87
Net cost	**67,39**	**70,69**	**75,13**	**58,92**	**51,3**	**22,86**	**79,94**	**84,68**	**65,35**

6.8 Recovered fractions : reusable

The aspect of material recovery is evaluated on the basis of the produced quantities of reusable material on the one hand and, the environmental and technical quality of the possible reusable fractions on the other hand. The different scenarios produce different types of products. The processing of fractions with a calorific value of the pre-treatment techniques is discussed in section 6.4.5 The fractions that are discussed in this section are other fractions that can be reused. The amounts given are in kg/tonne residual waste. Al scenarios produce ferrous, non ferrous and inert fractions. The amounts differ between the different scenarios. Following table gives an overview of the amounts of materials that are produced by the different scenarios per tonne of residual waste input.

Table 32: Amounts of reusable materials produced by the different scenarios per tonne of residual waste.

	GF	GF SNCR	GF SCR	pre 1	pre 2	pre 3	pre 1 + CFB	pre 2 +CFB	pre 3 + CFB
recovery	**kg/ton waste**								
Metals	**34**	**34**	**34**	**45**	**49**	**46**	**45**	**49**	**46**
ferrous	32	32	32	39	39	39	39	39	39
non ferrous	3	3	3	6	10	7	6	10	7
Minerals	**81**	**81**	**81**	**136**	**61**	**48**	**160**	**101**	**92**
inert				52	61	48	52	61	48
sand				84			84		
bottom ashes	81	81	81						6
bed ashes							15	26	25
boiler ashes							9	14	14
Organics	**n/a**	**n/a**	**n/a**	**70**	**n/a**	**n/a**	**70**	**n/a**	**n/a**
for soil improvement				70			70		
	GJ/ton waste								
Thermal fractions				**7**	**9**	**9**			
biogas				2					
RDF				5	9				
fibre						6			
plastics						3			
residue						1			

6.8.1 Metals

- **Ferrous metals**

The amounts of ferrous metals recovered in the pre-treatment-scenarios are higher than in the grate furnace. This is due to the high recovery rate that is used for these scenarios compared to the integrated systems (90 versus 75%). This has not been checked with empirical data. The quality of this fraction is assessed. In thermal systems (GF and Pre3 (FibreCycle)) this fraction has been cleaned and stripped of any impurities through the thermal process. Ferrous fractions from an incineration process is oxidized. The fractions recovered in non-thermal processes have not been stripped from impurities. The recovery rate as mentioned above is uncertain. However, the effect of this recovery rate on the total amount of recovered materials or on the costs is small compared to the effect of other uncertainties in this kind of analysis (e.g. cost data, waste composition, …)[5] . The price used in the calculations for this fraction is given in section 6.7. The amounts recovered are around 31,5 kg/tonne waste for the grate furnace, and 39 kg/tonne for the pre-treatment technologies.

- **Non ferrous metals**

The amounts of recovered non-ferrous metals is comparable in the pre-treatment scenarios. These amounts are higher than for the grate furnace. As with the ferrous metals, this is due to the fact that other recovery rates are used. Regarding the quality of this fraction, we refer to the quality of the ferrous metals. The same goes for the considerations made about the recovery rate. The price used in the calculations for this fraction is given in section 6.7. The recovered amounts are nearly 3 kg/tonne for the grate furnace, and between 6 to 10 kg/tonne waste for the pre-treatment techniques.

6.8.2 Minerals

By minerals we understand the different reusable inert fractions produced during the different processes (inerts from pre-treatment and different types of ashes produces during the thermal treatment of the energetic fractions). Minerals can be reused in Flanders if the environmental quality meets the VLAREA-legislation (Flemish legislation on recovery of materials from waste). According to this legislation the leachability of heavy metals and the total concentration of organic components needs to be determined and compared with the limiting values. Besides this environmental quality, the technical applicability of this fraction has to be demonstrated before this fraction can be reused. Inert fractions, which comply with the Flemish regulation on secondary raw materials (VLAREA), are marketed as ‘secondary building materials’. From this we can conclude that two types of mineral fractions are separated. One the one hand there are reusable minerals, on the other hand non-reusable minerals. The reusable minerals are further discussed in this section, and will be called inerts. The non reusable minerals (further referred to as residue) will be discussed in section 6.10.1.

- **Minerals from pre-treatment**

All evaluated pre-treatment technologies separate at least one mineral fraction (pre-treatment 1 produces two mineral fractions, referred to as a sand fraction and an inert fraction, pre-treatment 2 and 3 both produce one mineral fraction.

---

[5] A recovery rate of 75% in stead of 90% for ferrous and non-ferrous metals yields 6.5 kg of ferrous metals less and 1 kg of non-ferrous metals less. The resulting difference in calculated costs is less than 1€/tonne waste.

The sand-fraction, separated in a pilot installation of the SORDISEP technology (Pre1), exceeds the limiting leaching value for copper. We assume that by scaling up and optimizing the technology the leaching value for copper will comply with the regulation. The high leaching value for copper can be explained by the presence of organic components in this fraction. This is due to the fact that copper leaches under the form of organo-copper complex. Therefore it may be needed that the inerts from this pre-treatment technology needs to be washed ad aged (storing in open air). During the aging carbonisation and condensation reduce the leachability. The inerts from the HERHOF-process comply with the VLAREA limit values. The quality of the inerts obtained in he FibreCycle process needs yet to be determined. If the inerts are to be reused, additional metals separation on the inert fraction will be needed. Even then it is possible that a contamination of the inerts with fibre leads to the same leaching problems as mentioned for the SORDISEP technology.

About the technical applicability of their inert fractions from the studied technologies is currently no information available. Regardless these uncertainties, we assume that the quality (environmental as well as technical) of the inert fractions obtained during the different pre-treatment methods, without further treatment, is potentially the same. And therefore the applicability of these fractions is comparable. We assume that all mineral fraction obtained during pre-treatment of residual waste, can meet the VLAREA limiting values and can be reused. However this has to be confirmed for some fractions. In the cost calculations we assumed that these fractions can be marketed at zero costs. Meaning there are no costs, but also no benefits from these fractions.

The recovered amounts lie between 48 kg/tonne for pre-treatment 3 and 136 kg/tonne for pre-treatment 1. Pre-treatment 2 lies in between with 60 kg/tonne. These differences are caused by the recovery rate of the different systems. The high amounts recovered by pre-treatment 1 can be explained by the fact that this technique is tuned to recover as much as possible and that these fractions contain probably some non-inert impurities. But also by the fact that the non-mineral fractions from pre-treatment 2 and 3 probably contain some inerts.

- **Minerals from thermal treatment**

During the thermal treatment of waste, different types op ashes are produced. In this study we named these ashes: bottom ashes, bed ashes, boiler ashes, fly ashes and cyclone ashes. Bottom ashes and fly ashes are formed in the grate furnace. Bed, boiler and cyclone ashes result from the fluidised bed reactor.

Only the applicability of treated bottom ashes from a grate furnace has been thoroughly studied. These studies indicate that these ashes comply with the VLAREA regulations and are therefore suited for reuse. According to experts, the quality of the bed and boiler ashes from the fluidised bed reactor would also comply with the VLAREA regulation. The reuse of the produced bottom, bed and boiler ashes is assured.

The fly ashes and cyclone ashes contain large amounts of heavy metals and are therefore not suited for reuse. These two types of ashes are further discussed with the flue gas cleaning residue in section □.

The quantity of bottom ashes obtained during the thermal treatment of the thermal fraction(s) is related to the amount and the ash content of this (these) fraction(s). Processing of waste in the grate furnace results in 81 kg/tonne reusable bottom ashes. The processing of thermal fractions in the pre-treatment scenario leads to 23 kg/tonne waste in the pre-treatment 1 scenario to 44 kg/tonne in the pre-treatment 3 scenario of reusable mineral fractions (bed, boiler and bottom ashes). Pre-treatment 2 scenario lies in between with 40 kg/tonne waste processed. These differences are caused by the different amounts of thermal fractions that are valorised, the difference in ash content of these fractions, and for pre-treatment 3 scenario, the fact that one thermal fraction is valorised in a grate furnace in stead of the fluidised bed (see section 6.9).

6.8.3 Organics

Pre-treatment scenarios 1 and 3 produce one or more organic fraction(s). Pre-treatment 1 produces a fibre fraction, a residue, a ligneous fraction and a sludge cake. Pre-treatment 3 produces a fibre fraction.

The ligneous fraction of pre-treatment 1 is thermally valorised because of its calorific value. (see section 6.9). The residue obtained during the cleaning of the sand fraction and the sludge cake are both discussed in section □.

The composition of fibre fraction of pre-treatment 1 complies with the Flemish VLACO-legislation for compost from organic kitchen and garden waste. The quality of this fraction was also tested. The fibres have an equal quality as e.g. Finnish peat.

Pre-treatment 3 produces also a fibre fraction. According to the supplier different processing options exist theoretically for this fibre fraction: anaerobic digestion, incineration, gasification, fibreboard production, possibly after drying and/or pelletising. Some of these options are being investigated. These investigated options are the use in a material that can be used as permanent shuttering, the use in a type of tile and the use in fibreboard. Also the use of the fibre in soil improvers and in co-incineration or stand-alone incineration plants are investigated. Based on the available data it is not possible to judge the environmental quality of the fibre potential reuse applications. None of the possibilities mentioned has been demonstrated yet on a large scale. The mentioned options should be regarded at as potential recycling options. In this study however we assume that the fibre fraction is thermally valorised in a fluidised bed reactor. When however the reuse potentials are thoroughly investigated and applicable, it is very well possible that this fraction is not thermally valorised, but is reused. This fraction is further discussed in section 6.9

Only the fibre fraction of pre-treatment 1 is regarded an organic fraction that can be reused. The total amount produced is nearly 70 kg/tonne waste processed.

6.9 Thermal fractions

The different pre-treatment technologies produce different thermal fractions. Pre-treatment 1 produces biogas, an RDF, a fluff fraction and a ligneous fraction. The solid thermal fractions are put together to make a RDF-mix. Pre-treatment 3 produces a residue, a plastic fraction and fibres. Pre-treatment 2 produces RDF. The RDF mix and RDF produced by pre-treatments 1 and 2 are valorised in a circulating fluidised bed. The biogas of pre-treatment 1 is combusted in a biogas engine The fibre and plastic fraction from pre-treatment 3 are valorised in the circulating fluidised bed. The residue is treated in a grate furnace with non catalytic NOx-reduction an bottom ash treatment.

The amount recovered thermal fractions are expressed in GJ/tonne of waste processed. The amounts vary from 7 (pre-treatment 1) to 9 (pre-treatment 2 and 3) GJ/tonne waste. This difference between pre-treatment 1 and the other techniques can be explained by the fact that some organic fractions are not thermally valorised in pre-treatment 1.

6.10 Recovered fractions : not reusable

6.10.1 Minerals

- **Minerals from pre-treatment**

Only pre-treatment 1 produces a mineral fraction that is not reusable, further called residue. This residue is formed during the cleaning of the sand fraction. This fraction does not comply with VLAREA limiting values and can therefore not be reused as a secondary material. This fractions is disposed of on a landfill for non-hazardous waste.

- **Minerals from thermal treatment**

During the thermal valorisation different types of ashes are formed, as mentioned before. The fly ashes (grate furnace) and cyclone ashes (fluidised bed) contain large amounts of heavy metals and do not comply with the limit values of VLAREA. These residues are disposed of at a landfill for non-hazardous waste.
An other residue that is formed during the thermal processes are flue gas cleaning residues. This fraction are land filled at a landfill for hazardous waste after immobilisation.

The types of landfills that are mentioned for the different types of minerals above are not stringent. In reality they depend on the leachability of the fractions and how they relate to the regulations that are set for the different types of landfill sites in Flanders (VLAREM II). In this study we made assumptions for the different fractions based on available data and own estimates.

6.10.2 Organics

The sludge cake is landfilled. Pilot tests on this fraction however indicated that chemical cleaning of this sludge could lead to a fraction which complies tot the Flemish limit values for compost (VLACO-regulation). However this needs to be confirmed in a full scale installation. We assumed in this study that this fractions is landfilled on a landfill for non-hazardous waste.

Table 33: Amounts of non-reusable materials produced by the different scenarios per tonne of residual waste.

	GF	GF SNCR	GF SCR	pre 1	pre 2	pre 3	pre 1 + CFB	pre 2 +CFB	pre 3 + CFB
residue to landfill					**kg/ton waste**				
Minerals	**150,11**	**150,11**	**150,11**	**14,9**			**41,93**	**69,42**	**52,88**
residue				14,9			14,9		
bottom ashes	111,54	111,54	111,54						8,17
fly ashes	21,12	21,12	21,12						1,55
cyclone ashes							26,49	44,65	43,16
flue gas cleaning residue	17,45	17,45	17,45				15,44	24,77	25,44
Organics				**82,5**			**82,5**		
sludge cake				82,5			82,5		

6.11 Defining decision criteria

Based on the methodology and described above, we can perform a life cycle inventory. We can give insight on costs (private and external), and on the energy efficiency of the treatment options. This information can now be combined in a multi-criteria analysis. We discuss here the different criteria and the values for these criteria. This information has been used in a workshop with stakeholders on October 14th 2004 in Brussels. Results of this MCDA workshop are given in the report on MCDA.

The added value of MCDA is that a methodology is provided for stakeholders in order to evaluate a variety of burdens and impacts of waste treatment. Evaluation criteria for these burdens and impacts cover the environmental, economical and technical aspects of waste treatment. These criteria have different units and therefore they cannot be compared directly. The following criteria are withheld in this study. Numerical or qualitative values for the criteria and for the different alternatives are given in section 6.13.

- Impacts from greenhouse gases can either be defined by their global warming potential, or can be expressed in terms of damage costs. Here a damage cost of 19 € per tonne of CO_2 is used.
- Impacts on health deal with the impacts from air pollution following the emission of NOx, SO_2, particles, metals and dioxins. They are expressed in external costs. The external costs are calculated following an impact pathway approach from ExternE (2003-2004). The costs per kg of pollutant are given in Figure 13. They vary according to the location. Estimates for a typical location in Flanders and for a typical city in France for example are different because of the difference in regional dispersion of pollutants and because of differences in exposed population. They can also be expressed in physical terms (YOLL, years of life lost).
- Emissions of pollutants that have a potential contribution to acidification of ecosystems are grouped through their acidification potential and expressed as ‘Acidifying equivalents’ (Aeq).
- After treatment a final residue that is only suited for landfill is produced: the amount of residue to landfill is an important criteria in choosing waste treatment scenarios, especially when policies are designed to move away from landfilling.
- From the point of sustainable use of resources, prevention and reuse of materials is important. The recovery of ferrous and non-ferrous metals, of organic material (e.g. for soil improvement) and of inert material (e.g. for construction purposes) form another group of criteria. They might be combined in one criterion, but information on the quality of reuse is lost (construction applications being inferior to the recycling of metals for example).
- Impacts from auxiliaries and fuels used to process the waste are grouped and expressed as external costs.
- Avoided emissions through the production of energy from waste are also given in terms of externalities. They are negative, and will lower the total impact from waste treatment.
- Two qualitative indicators were excluded, because they didn’t give enough contrast between the treatment options. One deals with amenity issues (noise, odour,…). In an extensive study comparing very different and new technologies, amenity issues can differ strongly. We deal here with more- or less proven technologies, making this

criterion less distinctive. Operational risks give a qualitative value to the technical, hygienic and occupational risk involved with the waste treatment scenario.
- The economic burden of choosing one waste treatment scenario over another is measured by the net costs per tonne. they include investments, operational costs and profits.

The approach is flexible however to include other criteria that are deemed to be important to stakeholders.

6.12 Economic evaluation of waste treatment

6.12.1 Damage costs

Some criteria can be compared directly, i.e. those expressed in an external cost per tonne of waste. Using these criteria gives a first impression of the differences between the alternative choices, in terms of potential damages. In Figure 14 below the results are put together for the six waste treatment scenarios:

- GF : grate furnace with particle filter, deSOx and activated carbon injection
- GF SNCR: grate furnace with particle filter, deSOx, activated carbon injection and Selective Non-Catalytic Reduction (NOx reduction).
- GF SCR: grate furnace with particle filter, deSOx, activated carbon injection and with Selective Catalytic Reduction (NOx reduction).
- Pre-treatment 1 + CFB: biological pre-treatment combined with circulating fluidised bed combustion for RDF
- Pre-treatment 2 + CFB: biological pre-treatment combined with circulating fluidised bed combustion for RDF
- Pre-treatment 3 + CFB: thermal pre-treatment combined with circulating fluidised bed combusting for fibre and plastics and grate furnace with SNCR for residue.

The base case uses external costs in Flanders (Figure 13), and avoided emissions from an average electricity mix in Belgium. A detailed sensitivity analyses examining for instance other avoided emissions, and higher damages due to metals are given in section 6.14. For the application of these results in other countries note that the impacts of PM_{10}, NO_x and SO_2 depend on the emission site and that the emissions are likely to be different.

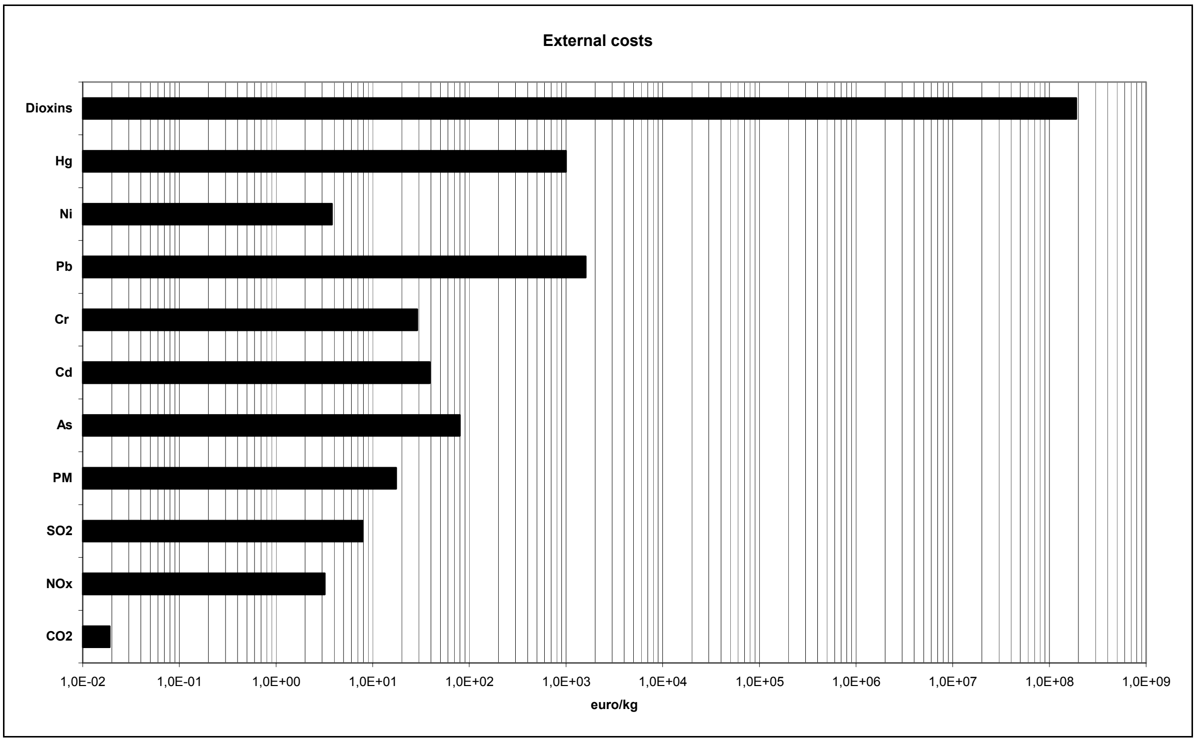
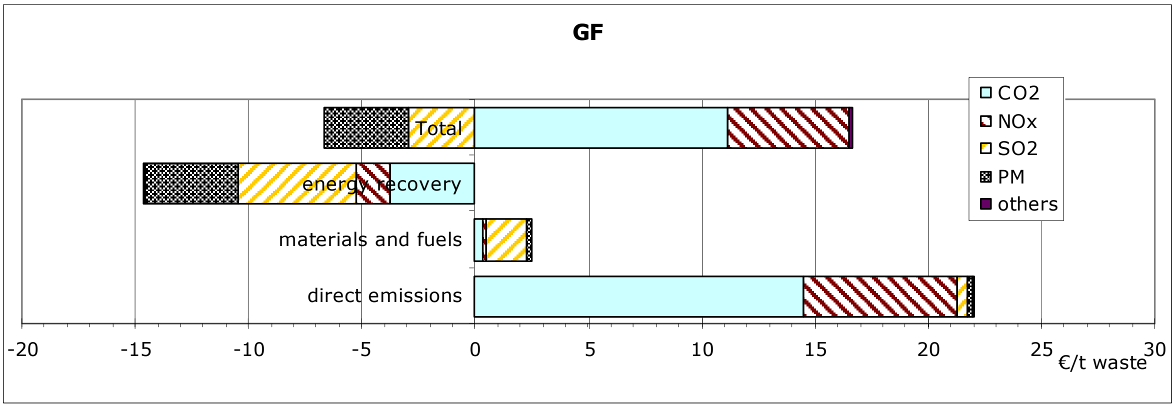
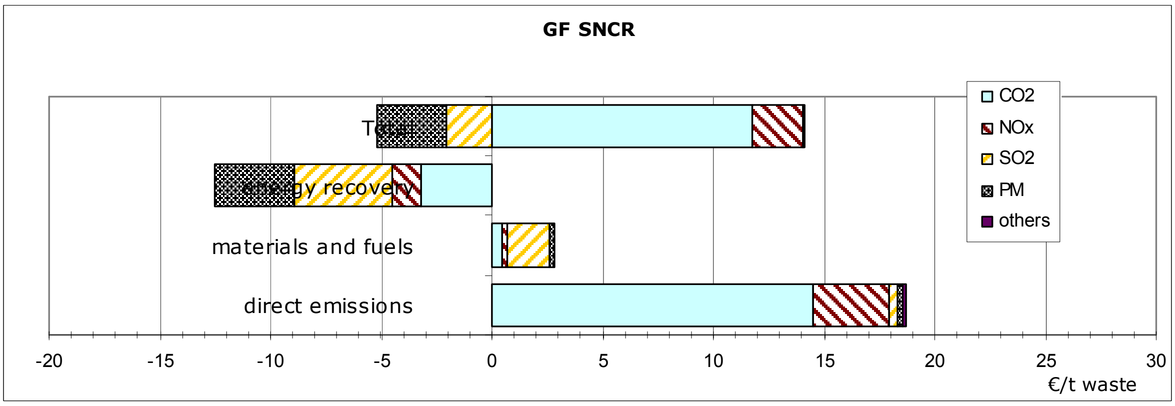


Figure 13: Damage costs per kg of pollutant emitted by incinerators in typical locationsin Flanders (based on ExternE)





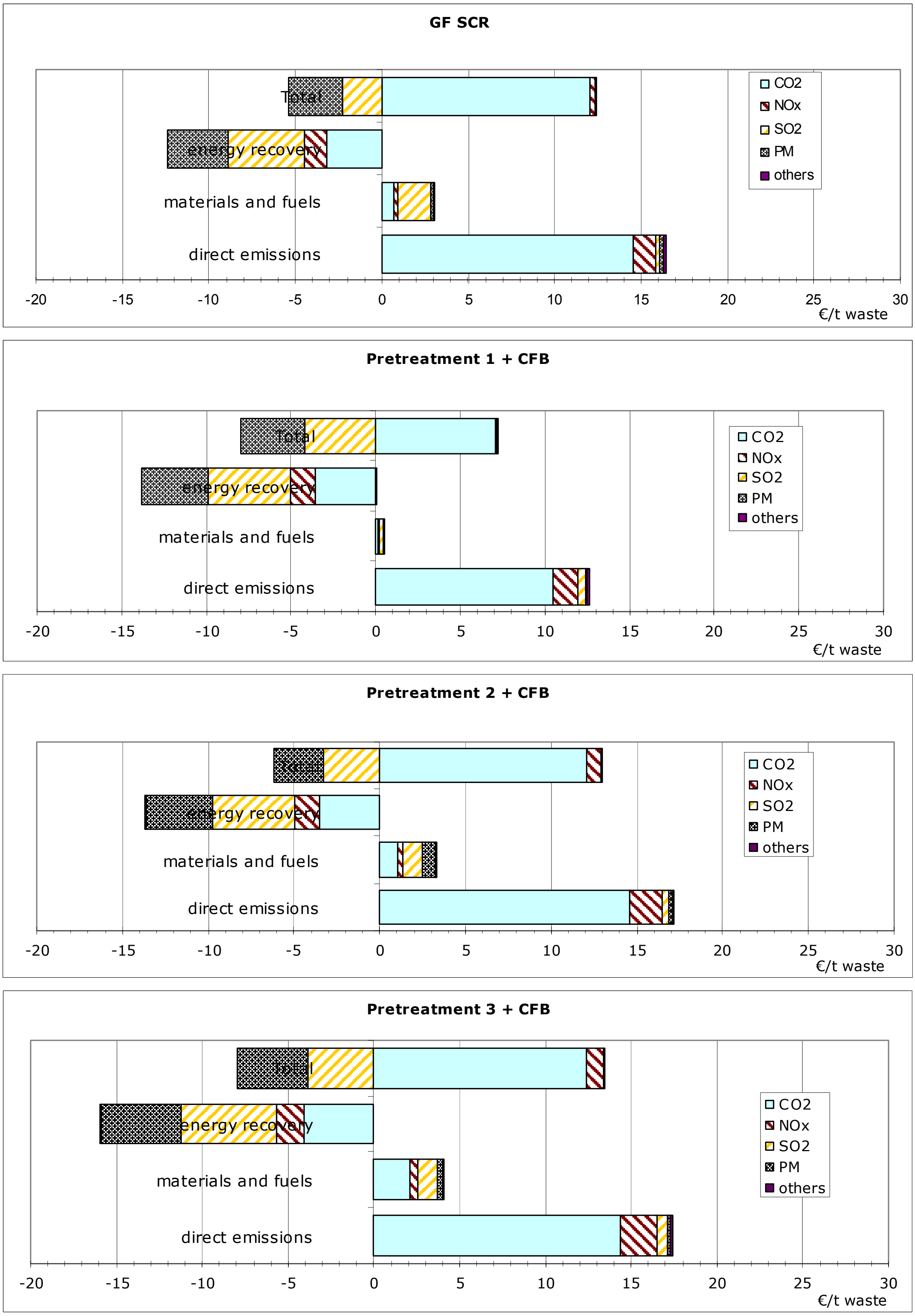


Figure 14: Damage costs from waste treatment, base case (€/t waste).

Damage costs of auxiliaries and fuels are small: Figure 14 shows that used auxiliaries and fuels are responsible for only a small amount of damage costs for all treatment scenarios. In the grate furnaces the damage is mainly due to SO_2 emissions originating from the use of oil. In the pre-treatment scenarios the damage cost are mainly caused by CO_2 emissions, from the use of electricity in pre-treatment 2, and from the combination of the use of natural gas and electricity in pre-treatment 3. Pre-treatment 1 uses no external energy, and this is reflected in this damage category.

Total damages are dominated by direct impacts and avoided impacts.
Damage from direct emissions is mainly caused by CO_2. The amount emitted is similar for all scenarios except pre-treatment 1 + CFB. This is due to the fact that the carbon content of the organic fraction that is suitable for reuse is not taken into account.

Damage avoided by the production of electricity is similar for the GF, pre-treatment 1 + CFB and pre-treatment 2 + CFB. Both grate furnaces with deNOx have a lower net electricity output, and therefore less avoided emissions. Pre-treatment 3 + CFB produces the largest amount of electricity, and therefore avoids the most emissions. It has to be stressed however that in the scenario pre-treatment 3 + CFB much more fuel is added to the system than in the other scenarios. To evaluate the net electricity production on an equal basis this additional fuel use should be taken into account. The net electricity production can be compared to the total input of energy of the waste and of the used fuels. These calculations are based on the assumption that natural gas is used to raise the heat for the process. Use of waste heat and CHP or use of part of the fibre are potential options to optimise the energy consumption and to decrease the use of fossil fuel. These options are not taken into account is this case study.

From the sensitivity analyses in section 6.14 it can be learnt that the choice of avoided energy mix can influence the ranking of technologies, and that emission factors for metals and dioxins do not influence the results. Moreover, transportation by truck or inland shipping does not change the overall ranking. The external costs are 2 orders of magnitude lower then those from incineration, even with congestion taken into account for trucks.

6.12.2 Full costs

Not only can we compare options based on their damage costs, but a comparison of treatment options based on full (i.e. external + private costs) costs is possible. In general conflicts between the different criteria that we chose in section 6.11 are to be expected. And private and external costs are a typical example: the solution with the lowest private cost has the highest external cost and vice versa, as can be seen in figure 6. Including avoided externalities can change the overall ranking in terms of total costs, depending on the choice of electricity mix that is assumed being avoided.

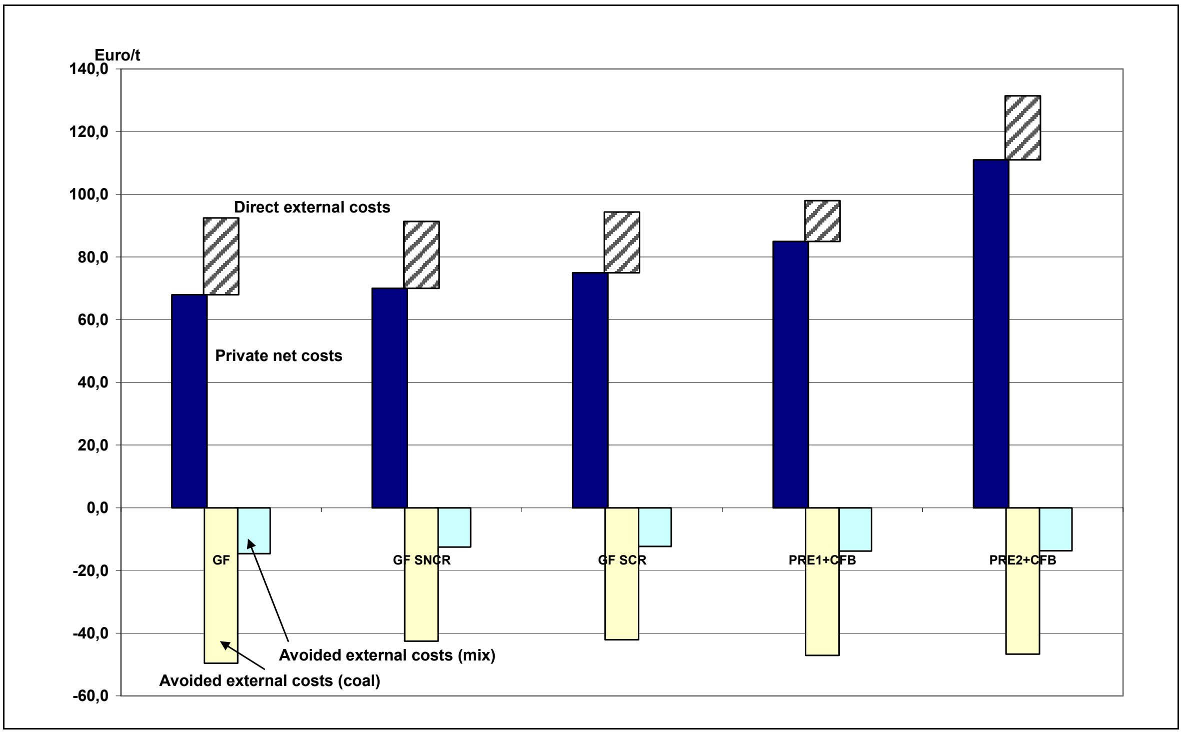


Figure 15: External costs and private costs for the Flanders case study.

6.13 Values for the criteria

In Table 34 the values for the direct emissions for the criteria defined above are given for the base scenario in Flanders. These values are the result of LCI and IPA and will be used to enable the MCDA. It is obvious that even an informed stakeholder will have difficulties interpreting these values and ranking the treatment options. Therefore a formal and structured multi-criteria analysis is necessary.

Table 35 gives some extra information on how good the different combustion scenarios keep their emissions below mandatory limit values. For stakeholders operating or controlling these installations, this information is useful in deciding between waste treatment options, keeping in mind possible future developments (lower limit values for example). 100% below limit values means that less than 1% of what is allowed in the limit values is actually emitted.

Table 34: Overview of values for criteria (in damage costs or other physical units) for the six scenarios.

	Grate Furnace	Grate Furnace with SNCR	Grate Furnace SCR	pre-treatment 1 + CFB	pre-treatment 2 + CFB	pre-treatment 3 + CFB	
Global warming	764	764	764	552	765	756	kg CO2eq/t waste input
Global warming	15	15	15	10	15	14	€/ton waste input
Impacts on health	872,4	468,9	206,8	229,7	291,3	334	YOLL/t waste input
Impacts on health	7,4	4	1,8	2	2,5	3	€/ton waste input
Impacts on ecosystem (acidification)	0,0477	0,0247	0,01	0,0116	0,0145	0,017	mAeQ*
Impacts on ecosystem (acidification)	7,2	3,8107	1,5662	1,9033	2,2877	3	€/ton waste input
Final residue to landfill	132,7	150,11	150,11	139,33	69,42	78	kg/t waste input
Ferro/non-ferro recovered	34,4	34,4	34,4	45	48,7	46	kg/t waste input
Minerals recovered	80,8	80,8	80,8	159,6	100,6	92	kg/t waste input
Organic material recovered	n/a	n/a	n/a	69,9	n/a	n/a	kg/t waste input
Impacts from fuels and auxiliaries (life cycle impacts	2,5	2,8	3,1	0,5	2,9	4	€/t waste input
Avoided impacts from heat and electricity recovery	-12,6	-10,8	-10,7	-12	-11,8	-14	€/t waste input
Amenity impacts (odour, noise)	0	0	0	-	-	-	
Net costs	68	70	75	85	111	67	€/t waste input
Operational risks	low/proven technology	low/proven technology	low/proven technology	medium/ partial proven technology	low/proven technology	high/no full scale available in Europe	

*: 10^{-3} acidifying equivalents
YOLL : years of life lost

Table 34 clearly shows that the damage caused by *global warming* is similar for all scenarios except pre-treatment 1 + CFB (see section 6.12). The *impact on health* is caused by emissions of NO_x, SO_x and PM. The positive influence of a deNOx installation is clearly shown is this table. The same conclusion can be drawn by the *impacts on the ecosystem* (acidification). These impacts are caused by NO_x and SO_x emissions. The installation of a deNOx system has a great influence on the impacts on acidification.

Final residue to landfill are mainly inert fractions that are not suitable for reuse. This inert fraction is formed during the incineration (different types of ashes) or separated during the pre-treatment of the waste. Besides the inert fraction this residue contains also flue gas cleaning residue. In Flanders inerts can be reused if the environmental quality meets the VLAREA-legislation. According to this legislation the leachability of heavy metals and the total concentration of organic components needs to be determined and compared with the limit values. Besides this environmental quality, the technical applicability of this fraction has to be demonstrated before this fraction can be reused. Inert fractions, that comply with the Flemish regulation on secondary raw materials are marketed as 'secondary building materials'. This fraction is discussed in the section 'minerals recovered'. In the scenario pre-treatment 1 + CFB almost the same amount of residue needs to be land filled as in the scenarios with a grate furnace. This leads to the following conclusion regarding pre-treatment technologies: the more fractions that are separated from the waste, the more residue is formed. Both other pre-treatment techniques produce only half the amount of residue that has to be land filled.

The scenarios with pre-treatment of the waste recover about the same amount of *ferro and non ferro*. This is due tot the fact that higher recovery rates are used for the pre-treatment techniques than for the grate furnace (approx. 90% vs. 75%).

Pre-treatment 1 + CFB recovers by far the most *minerals* that can be reused. The other pre-treatment scenarios recover smaller amounts, but still more than the grate furnace.

Organic material is only recovered in pre-treatment 1, that is designed to separate as much different fractions as possible.

The most impacts from *fuels and auxiliaries* are formed by the pre-treatment 3 + CFB scenario. This is because this scenario uses large amounts of external energy (gas and electricity). For the three grate furnace scenarios and pre-treatment 2 + CFB these impacts are similar. The impact from pre-treatment 1 + CFB is small compared to the other scenarios.

Pre-treatment 3 + CFB avoids the most *electricity*. The grate furnace with SCR avoids the least electricity. But overall the avoided impacts are similar for all scenarios.

In this evaluation the amenity impact (odour, noise) of the grate furnaces is given a 0. The impacts for the pre-treatment scenarios a -. It is very difficult to make a proper estimation regarding this impact because there are no full scale installations in Flanders.

The net costs for the grate furnace, the grate furnace with SNCR and pre-treatment 3 with CFB are similar. The cost for treatment in a grate furnace with SCR is slightly higher. Pre-treatment 1 + CFB is 20 % more expensive than the cheapest treatment options. Pre-treatment 2 + CFB more than 50 % more expensive. However, all cost calculations have to be taken with the greatest caution. Crucial data on investment costs and operational costs have been provided by the technology suppliers who all reported in their own way. Costs do not include taxes, levies or subsidies. Neither do they include the cost of capital or a risk premium. The calculated cost is not a commercial market price.

Operational risks are estimated to be low for the grate furnaces and pre-treatment 3 + CFB because full scale installations are available in Europe. Because pre-treatment 1 + CFB has no full scale installation of the SORDISEP process on MSW the operational risks are estimated to be medium. Pre-treatment 3 + CFB has no full scale installation what so ever and therefore this scenario is given a high operational risk.

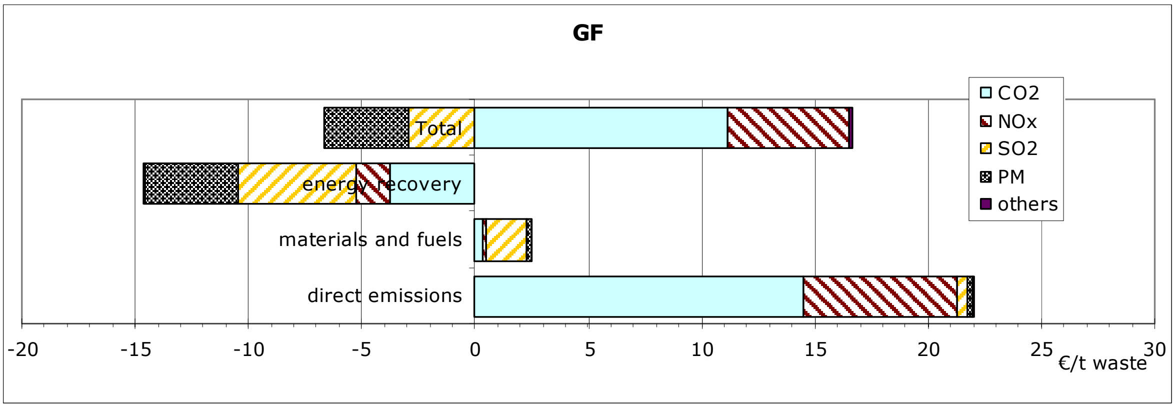
Table 35: Adherence to legislation, expressed as percentage below limit values of the incineration directive.

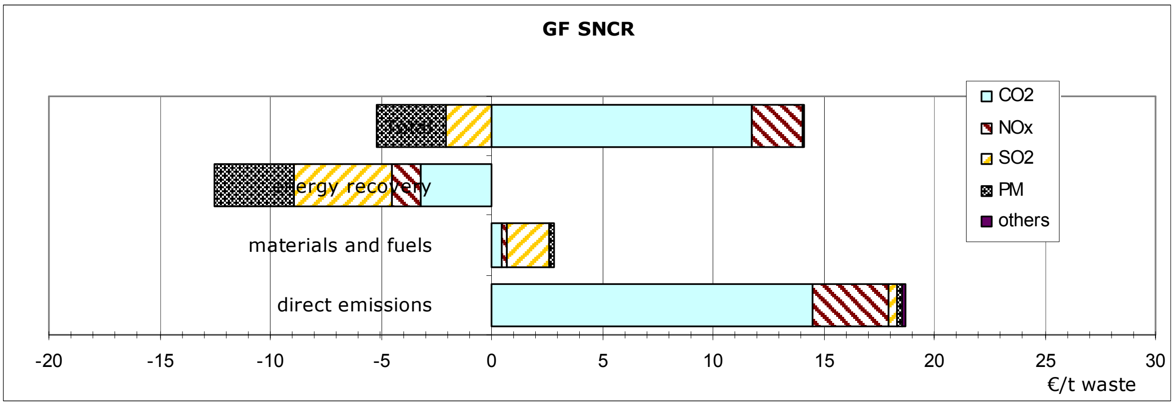
		Grate Furnace	Grate Furnace	Grate Furnace	pre-treatment 1 + CFB	pre-treatment 2 + CFB	pre-treatment 3 + CFB
Legislative aspects	**limit value (mg/Nm³)**		**with SNCR**	**with SCR**			
NOx	200	-100%	0%	60%	35%	35%	31%
SO2	50	80%	80%	90%	80%	80%	80%
PM	10	77%	77%	77%	77%	77%	77%
Cd+Tl	0,05	100%	100%	100%	100%	100%	100%
Hg	0,05	100%	100%	100%	100%	99%	99%
As+Pb+Cr+Co+Cu+Mn+Ni+V	0,5	64%	64%	64%	76%	90%	73%
dioxin	0,0000001	60%	60%	66%	60%	60%	60%

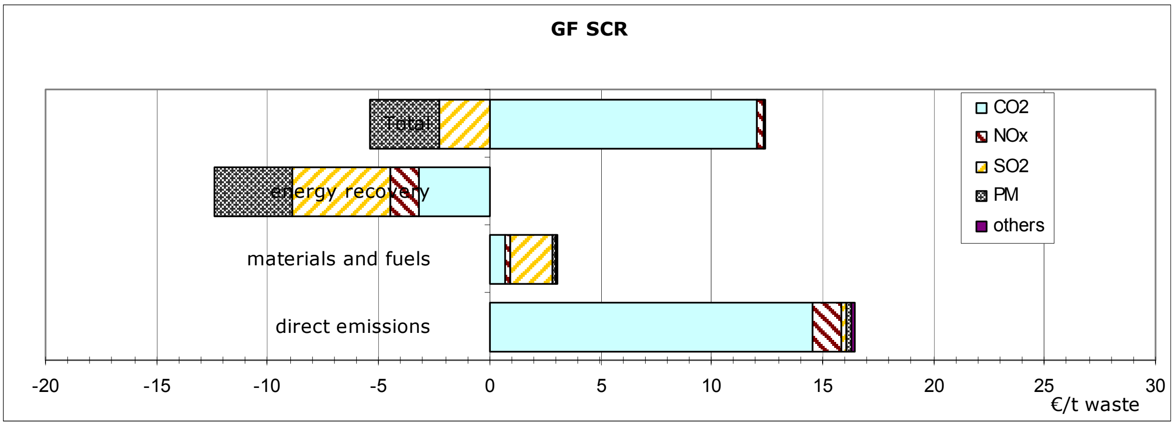
6.14 Damage costs of waste treatment : sensitivity analysis

6.14.1 Avoided electricity

Following figures give an overview of the influence of electricity that is avoided in different types of installations. In the base case avoided electricity was originating from the Belgian mix. The average Belgian mix for electricity production consists mainly of nuclear power (58%).







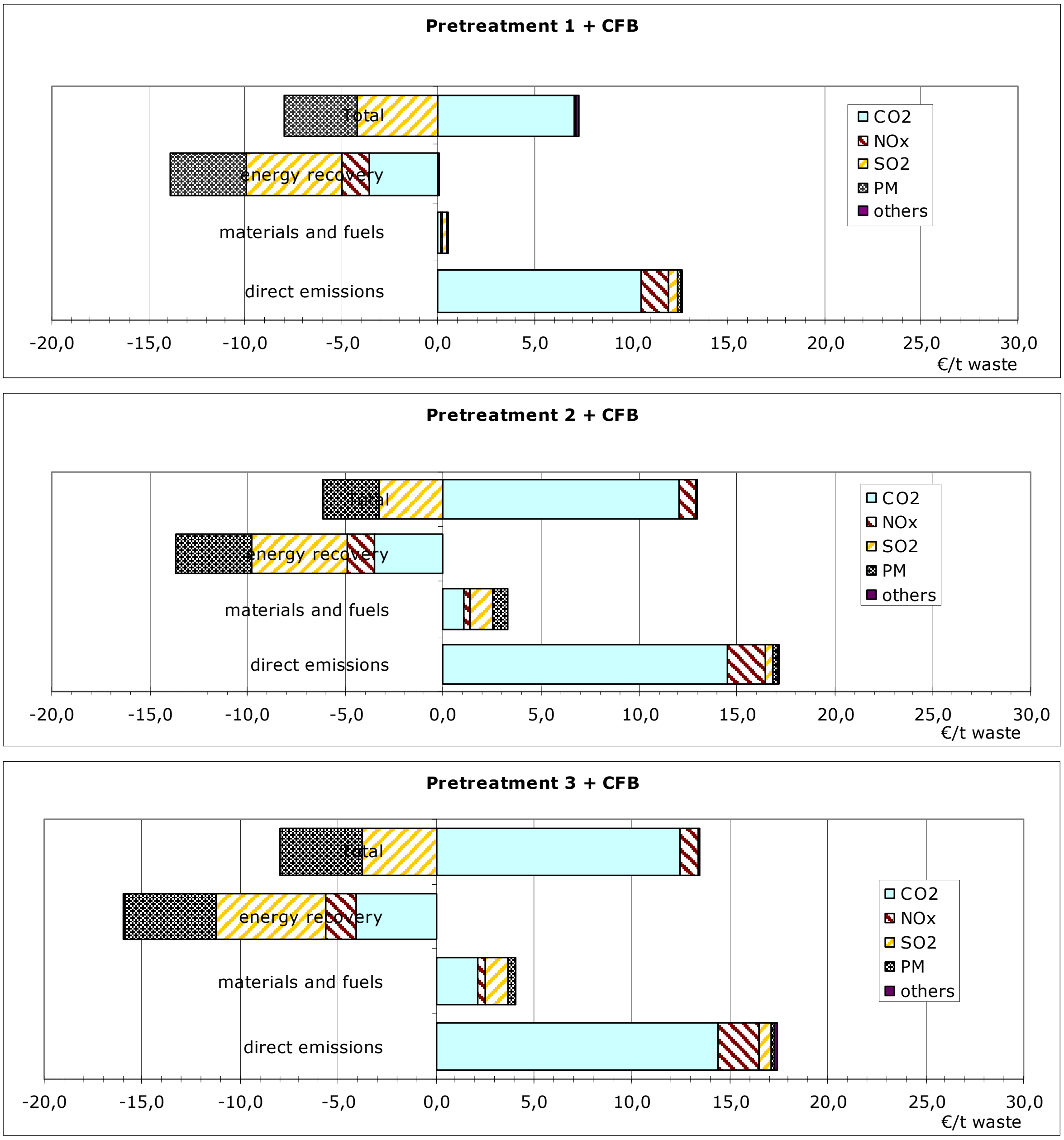
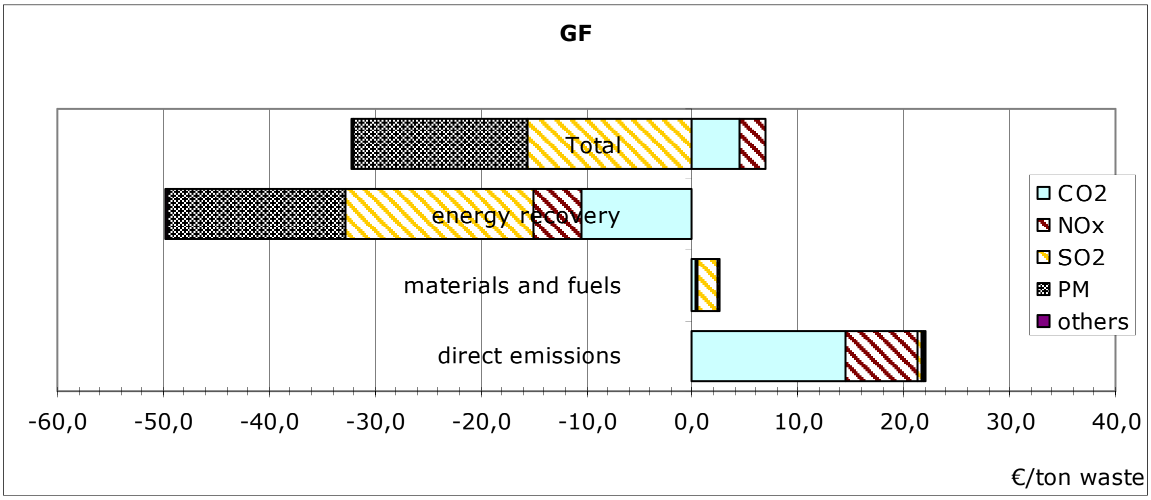
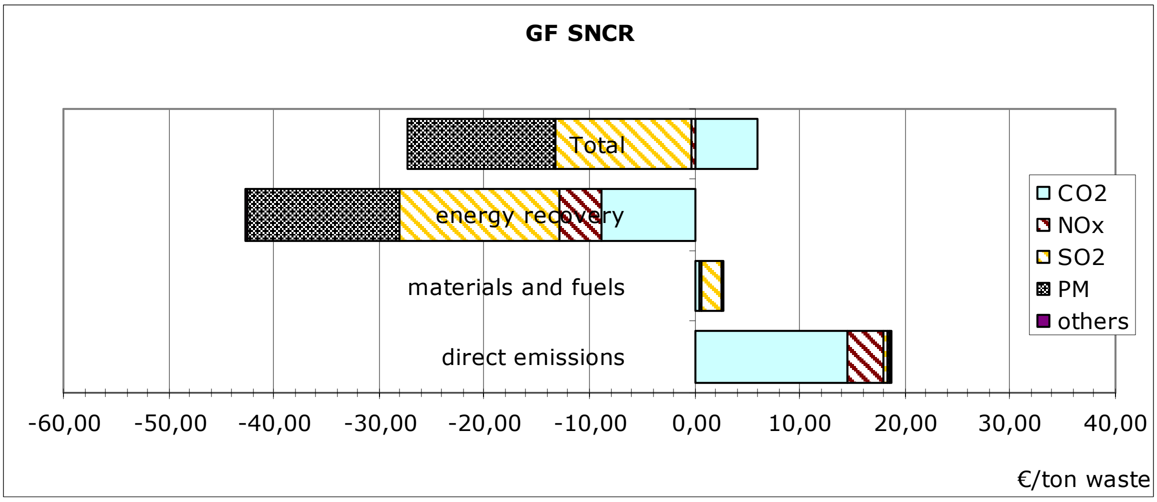
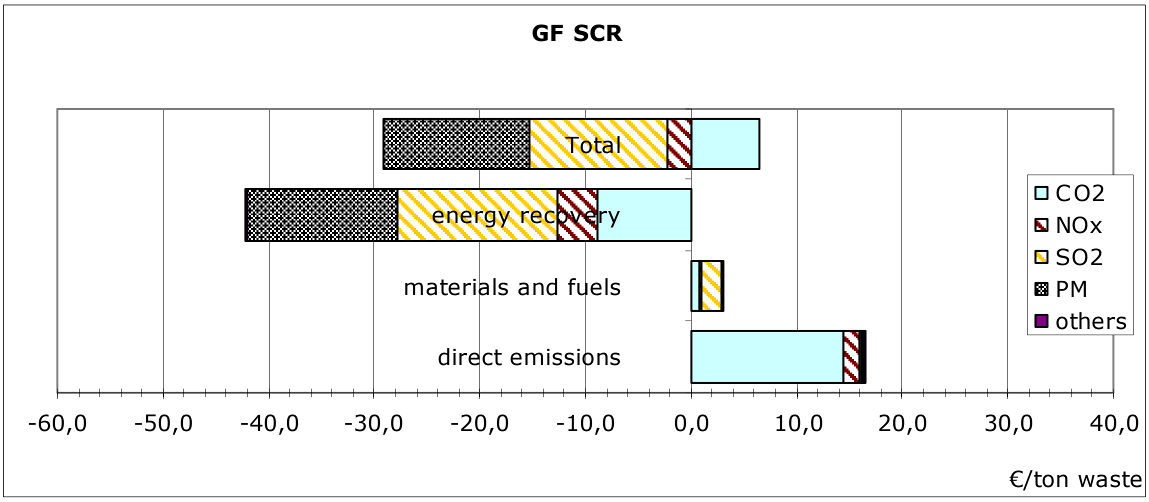


Figure 16:Breakdown of external costs for waste treatment in Flanders.

We tested the effect of avoided emissions from a coal-fired power plant, a gas-fired power plant and a nuclear power plant. In Figure 17 it is seen that avoiding electricity production from coal generates negative damage costs for all treatment scenarios. Avoiding electricity from gas has a smaller effect. On the other end of the spectrum the effect of avoiding electricity from a nuclear power plant is negligible (Figure 18). The overall damage costs stay positive for all scenarios.


GF
Total
energy recovery
materials and fuels
direct emissions
CO2
NOx
SO2
PM
others
-60,0
-50,0
-40,0
-30,0
-20,0
-10,0
0,0
10,0
20,0
30,0
40,0
€/ton waste


GF SNCR
Total
energy recovery
materials and fuels
direct emissions
CO2
NOx
SO2
PM
others
-60,00
-50,00
-40,00
-30,00
-20,00
-10,00
0,00
10,00
20,00
30,00
40,00
€/ton waste


GF SCR
Total
energy recovery
materials and fuels
direct emissions
CO2
NOx
SO2
PM
others
-60,0
-50,0
-40,0
-30,0
-20,0
-10,0
0,0
10,0
20,0
30,0
40,0
€/ton waste

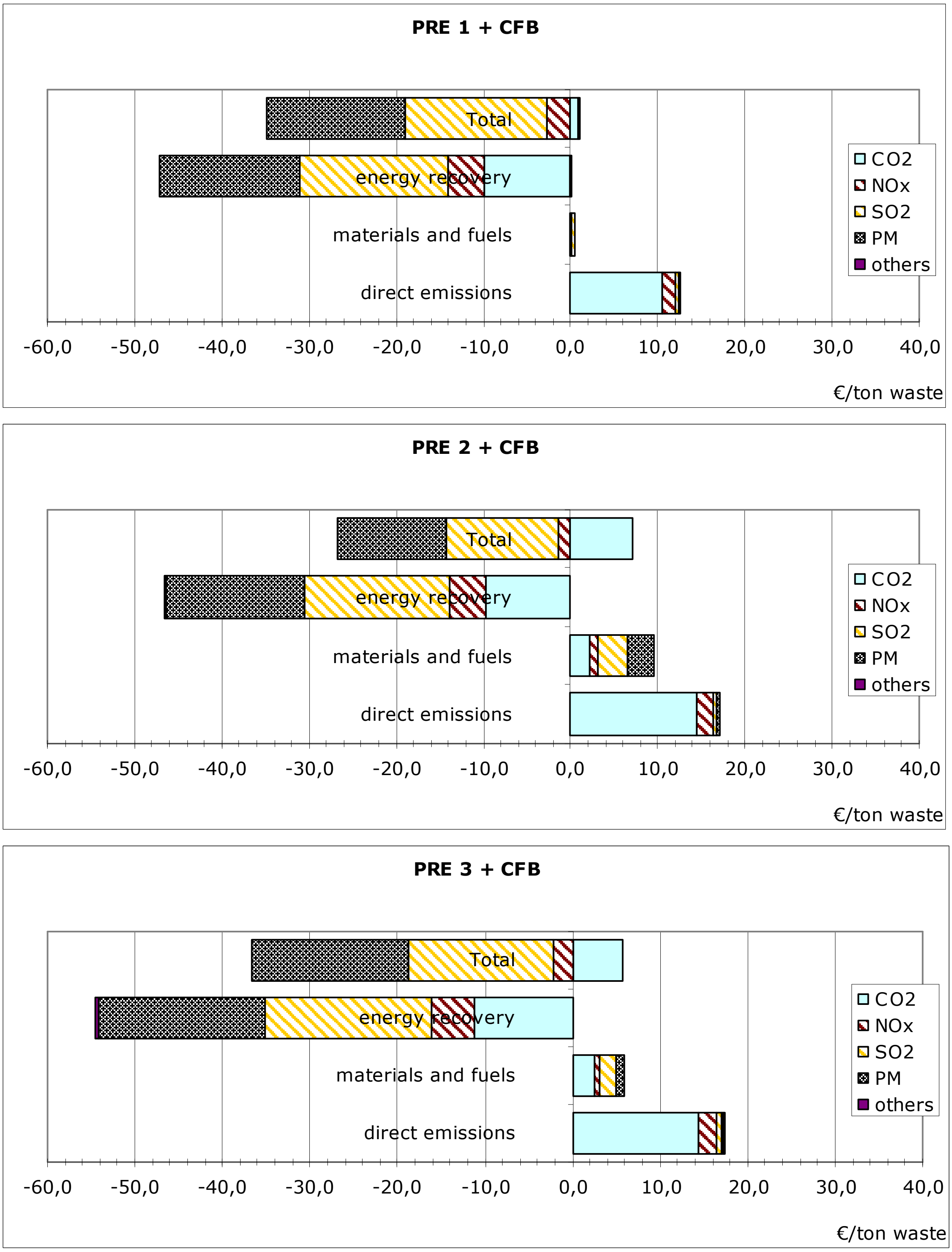
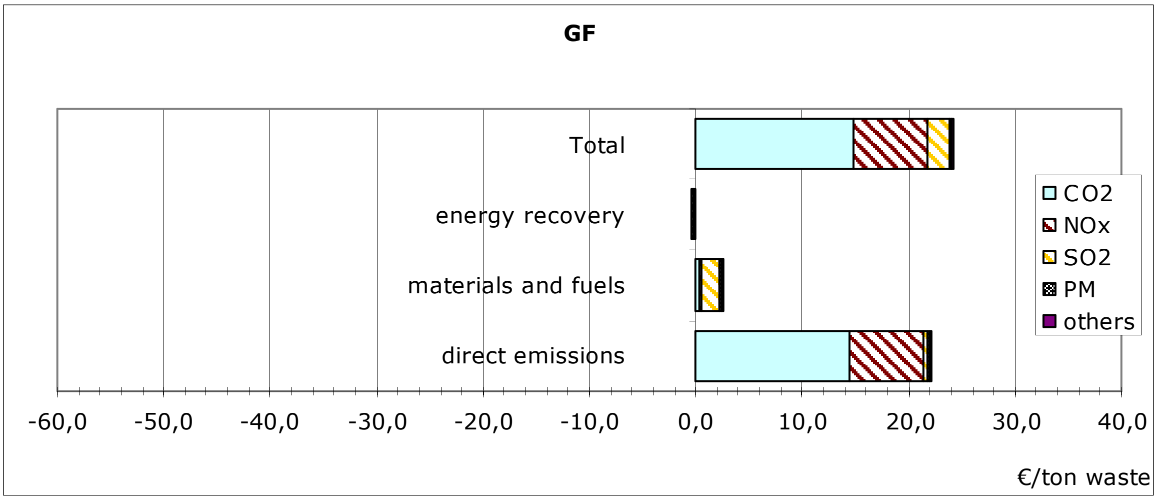
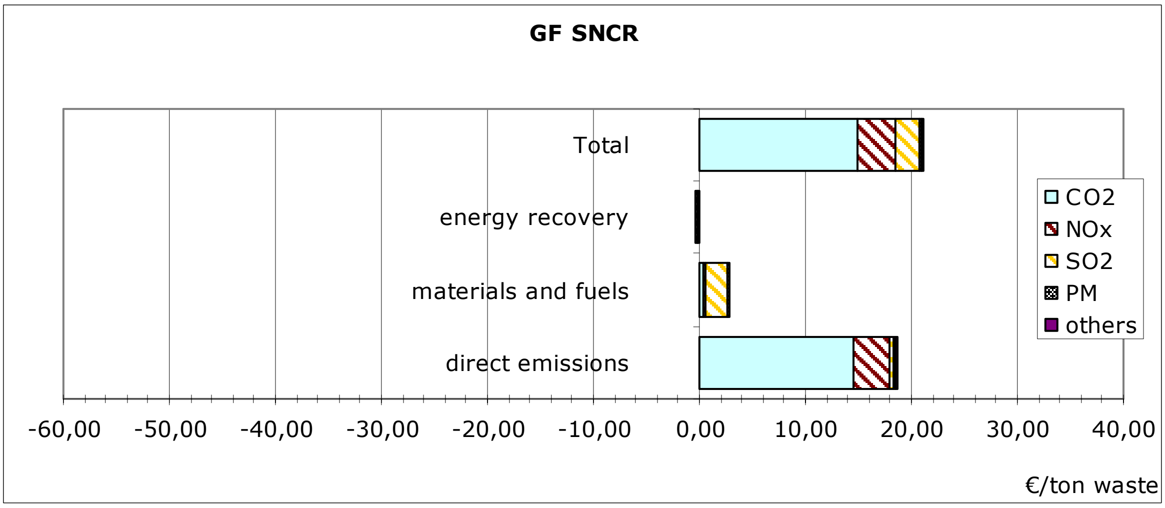
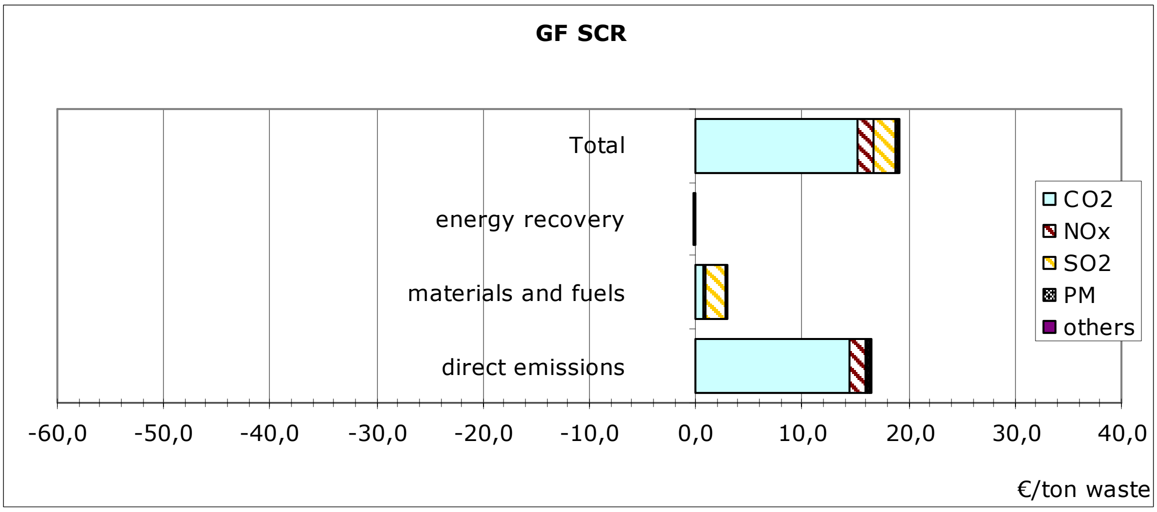


Figure 17: Damage costs from waste treatment in Flanders, avoiding electricity from coal.


GF
Total
energy recovery
materials and fuels
direct emissions
CO2
NOx
SO2
PM
others
-60,0
-50,0
-40,0
-30,0
-20,0
-10,0
0,0
10,0
20,0
30,0
40,0
€/ton waste


GF SNCR
Total
energy recovery
materials and fuels
direct emissions
CO2
NOx
SO2
PM
others
-60,00
-50,00
-40,00
-30,00
-20,00
-10,00
0,00
10,00
20,00
30,00
40,00
€/ton waste


GF SCR
Total
energy recovery
materials and fuels
direct emissions
CO2
NOx
SO2
PM
others
-60,0
-50,0
-40,0
-30,0
-20,0
-10,0
0,0
10,0
20,0
30,0
40,0
€/ton waste

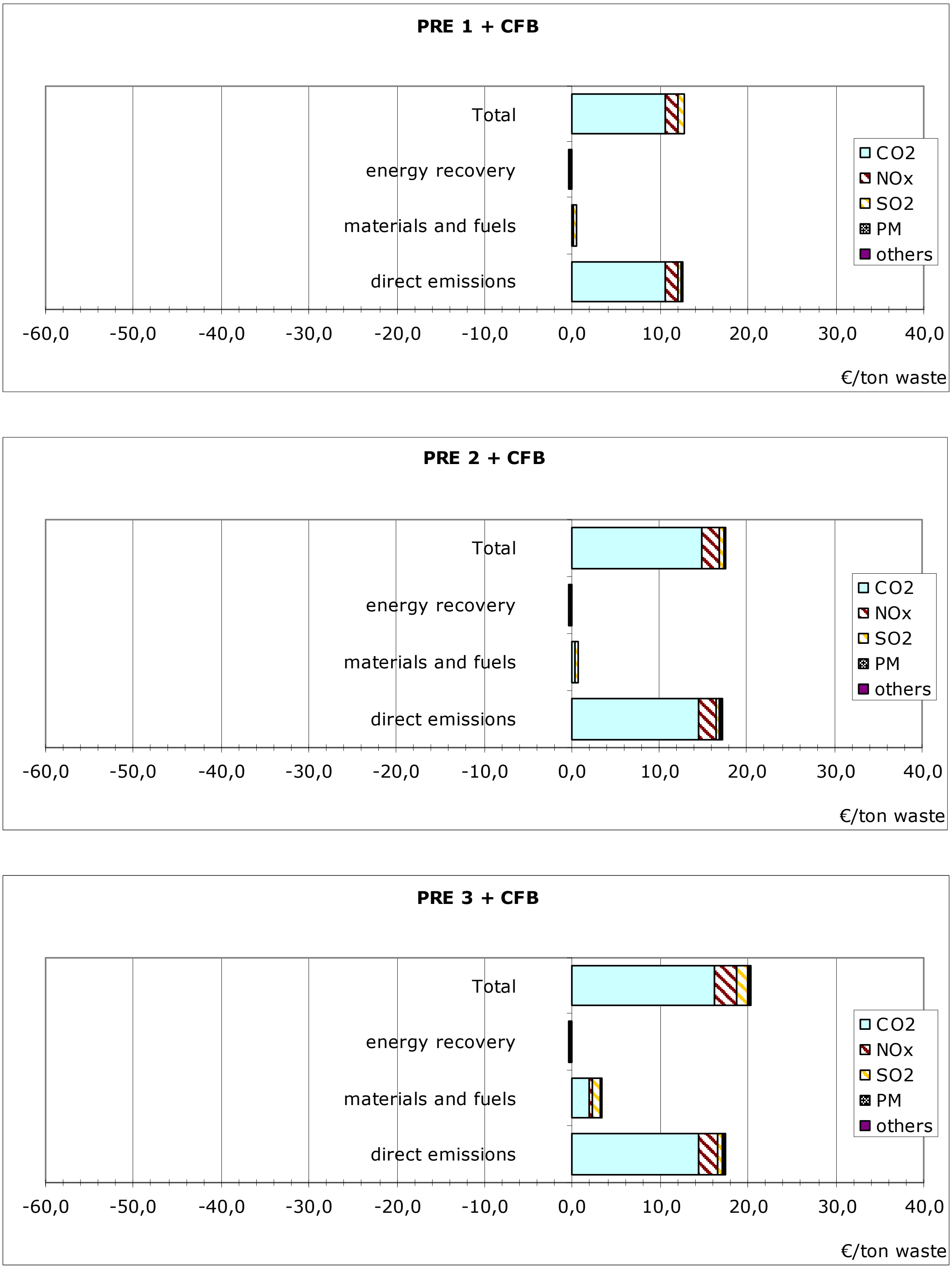


Figure 18: Damage costs from waste treatment in Flanders, avoiding nuclear electricity.

More important, the choice of avoided emissions can influence the ranking of technologies, as illustrated by following figures.

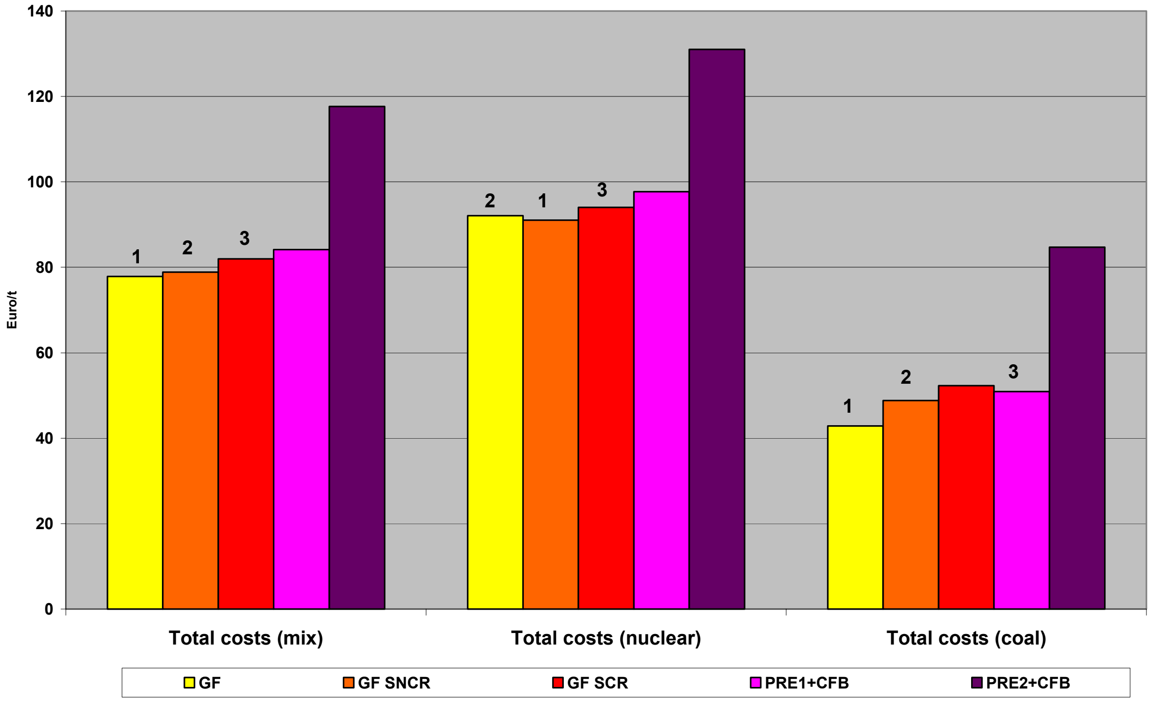


Figure 19: influence of choice of avoided emissions

Using limit values for coal-fired or gas-fired power plants give very high avoided impacts, of the order of 200-300 € per tonne of waste. Reasons are:

- Real emissions are in general lower than limit values;
- In particular:
  - particle emissions from CCGT (gas-fired power plants) are almost zero in reality, whereas a limit value of 5 mg/Nm³ is still allowed in the LCP directive;
  - SO_2 emissions from CCGT are also almost zero, whereas LCP allows for 35 mg/Nm³
  - nuclear power plants provide 58% of the average electricity production in Belgium. Including or excluding nuclear energy has a 60% effect (decrease or increase) on emissions, because the emissions of nuclear power plants are nearly zero.

6.14.2 Increased metal and dioxin emissions

We tested the effect of increased metal and dioxin emissions of waste incineration on the damage costs. We increased the level of emissions up to the limit values for metals, which implied that metal emissions were sometimes unrealistically high. None of the increases had an impact greater than 0,1 euro per tonne of waste. Only lead could have a noticeable effect. When increasing the lead emissions to the limit value (0,5 mg/Nm³), damage costs increased for the grate furnace scenarios with 4 euro per tonne of waste.

From this we can conclude that differences in waste compositions (within certain boundaries) for the different scenarios will also have no influence on the external costs in this study. This does not mean that differences in waste compositions have no influence at all. The emissions itself are directly influenced by the composition of the waste as described in section 6.5. And changes in the waste composition will still influence the results as described in section 6.3.

6.15 Summary

The extensive knowledge and data on waste treatment in Flanders has been used to demonstrate the application of the SusTools framework. The case study deals with the policy question of how to minimize environmental impacts of waste treatment, while optimizing the energy and material recovery and minimizing the costs. LCA methods have been used to define the system boundaries, and to develop a database of emissions and burdens. The life cycle inventory gives partial answers. CO_2 emissions for example are significantly lower in pre-treatment 1 + CFB, but this treatment options has a higher amount of final residue that needs landfilling. Moreover the costs are also higher for this pretreatment option. LCI can be considered the basis for the evaluation but is not the final tool on which to base a decision. An impact pathway approach to calculate external costs gives another perspective. The direct external costs are not very different (between 19 and 24 €/tonne of waste), except for the pre-treatment 1 +CFB option that is lower (13 €/tonne of waste). Taking into account the recovery of energy and avoided emissions decreases the difference between technologies, changes the ranking, but still leaves pre-treatment 1 as the best option. Combining the tools IPA and LCI gives already a better insight in the results. We have included costs in the evaluation, and were able to make a full cost assessment op the treatment options. Changing the assumptions on energy recovery and avoided emissions can change the overall ranking based on full costs. Now the conclusion that pre-treatment 1 + CFB is better no longer holds. All the information is used to built a set of values for different decision criteria. This matrix of values is then used in a formal multi-criteria decision analysis with stakeholders on a workshop in Brussels (October 14th, 2004). The results are presented in the MCDA report.

7 CONCLUSIONS

The choice between municipal solid waste management is often taken at a local or regional level, on the basis of national or EU objectives. The Sustools project has developed case studies of local decisions on incineration and landfill in France and on choosing between technologies in Flanders, Belgium. The case studies are not designed to be applied for Europe-wide analysis, because of the localised impacts of emissions and because of differing political environments. However, these case studies do illustrate the use of the Sustools methodology in integrating different techniques, enabling a balanced assessment of different policy options. The results of the different tools can than be used within an MCDA framework to provide for an assessment of policy decisions.

8 GLOSSARY

MCDA:	Multi-criteria decision analysis
LCA:	Life cycle assessment
LCI:	Life cycle inventory
IPA:	Impact pathway approach
Aeq:	Acidifying equivalents
RDF:	Refuse derived fuels
SCR:	Selective catalytic reduction
SNCR:	Selective non-catalytic reduction
GF:	Grate furnace
CH_4:	Methane
CBA:	Cost benefit analysis

9 REFERENCES

ADEME 2000. "Analyse environnementale de systèmes de gestion de déchets ménagers", Phase 1: analyse des parameters determinants pour les impacts environnementaux des différents modules". Study by BIO Intelligence and Ecobilan for ADEME and Eco-Emballages.

AEA Technology (2001) *Waste Management Options and Climate Change*. Report for DG Enviroment, European Commission.

Allsopp, M., P. Costner and P. Johnston (2001) *Incineration and Human Health: State of Knowledge of the Impacts of Waste Incinerators on Human Health.* Greenpeace Research Laboratories, Exeter University. Available online at http://archive.greenpeace.org/toxics/reports/euincin.pdf Accessed 17.06.04

Atcha, S.Y. and V.T. Van Son (2002) *Corporate Guide to Green Power Markets: Opportunities with Landfill Gas.* World Resources Institute. Available online at http://pdf.wri.org/gpmdg_corporate_guide_02.pdf Accessed 16.12.03

Bridges, O., J.W. Bridges and J.F. Potter (2001) "A generic comparison of the airborne risks to human health from landfill and incinerator disposal of municipal solid waste." *The Environmentalist* **20** pp.235-344

Cambridge Econometrics, in association with EFTEC and WRc (2003) *A study to estimate the disamenity costs of landfill in Britain: Final Report.* Defra.

Coopers and Lybrand (1996) *Cost-benefit Analysis of the Different Mubicipal Solid Waste Management Systems: Objectives and Instruments for the Year 2000.* Report for the European Commission DGXI.

Cottica A & Kaulard A (1995): *The Costs, Environmental Benefits and Direct Employment Implication of Greening Municipal Waste Management in Europe: An Engineering Estimation,* NOMISA, Bologna.

COWI (2000) *A study on the economic valuation of environmental externalities from landfill disposal and incineration of waste: Final Main Report.* European Commission and DG Environment.

CSERGE, Warren Spring Laboratory, EFTEC (1993) *Externalities from Landfilland Incineration*, Report to the UK Department of the Environment, London: HMSO..

Directive 2000/76/EC of the European Parliament and of the Council of 4 December 2000 on the incineration of waste *Official Journal L 332 , 28/12/2000 P. 0091 – 0111*

Directive 2001/80/EC of the European Parliament and of the Council of 23 October 2001 on the limitation of emissions of certain pollutants into the air from large combustion plants *Official Journal L 309 , 27/11/2001 P. 0001 – 0021*

Dolk, H,. M. Vrijheid, B. Armstrong, L. Abramsky, F. Bianchi, E. Garne, V. Nelen, E. Robert, J.E.S. Scott, D. Stone, R. Tenconi (1998) Risk of congenital anomalies near hazardous-waste landfill sites in Europe: the EUROHAZCON study. *The Lancet,* **352** 423-427.

EC (1999). ExternE: Externalities of Energy. Vol.7: Methodology 1998 Update (EUR 19083); Vol.8: Global Warming (EUR 18836); Vol.9: Fuel Cycles for Emerging and End-Use Technologies, Transport and Waste (EUR 18887); Vol.10: National Implementation (EUR 18528). Published by European Commission, Directorate-General XII, Science Research and Development. Office for Official Publications of the European Communities, L-2920 Luxembourg. Results are also available at http://ExternE.jrc.es/publica.html.

EC (2001) *Employment effects of waste management policies.* Risk and Policy Analysts Ltd.

EC (2001) *Employment effects of waste management policies.* Risk and Policy Analysts Ltd.

ECON Senter for økonomisk analyse (1995): *Environmental costs of different*

ECON Senter for økonomisk analyse (1995): *Environmental costs of different*

Elliott, P., D. Briggs, S. Morris, C. de Hoogh, C. Hurt, T.K. Jensen, I. Maitland, S. Richardson, J. Wakefield and L. Jarup. (2001) "Risk of adverse birth outcomes in populations living near lanfill sites." *British Medical Journal* **323** pp363-368.

Elliott, P., D. Briggs, S. Morris, C. de Hoogh, C. Hurt, T.K. Jensen, I. Maitland, S. Richardson, J. Wakefield and L. Jarup. (2001) “Risk of adverse birth outcomes in populations living near lanfill sites.” *British Medical Journal* **323** pp363-368.

Environment Agency (2002) *Solid Residues from MunicipalWaste Incinerators in England and Wales.* Available online at http://www.environment-agency.gov.uk/commondata/105385/ash_pdf4.pdf

Environment Agency (2002) *Solid Residues from MunicipalWaste Incinerators in England and Wales.* Available online at http://www.environment-agency.gov.uk/commondata/105385/ash_pdf4.pdf

Enviros Consulting Ltd, University of Birmingham with Risk and Policy Analysts Ltd, Open University and Maggie Thurgood. (2004) *Review of Environmental and Health Effects of Waste Management: Municipal Solid Waste and Similar Wastes.* Defra.

Enviros Consulting Ltd, University of Birmingham with Risk and Policy Analysts Ltd, Open University and Maggie Thurgood. (2004) *Review of Environmental and Health Effects of Waste Management: Municipal Solid Waste and Similar Wastes.* Defra.

EPA 1998. “Human Health Risk Assessment Protocol for Hazardous Waste Combustion Facilities, Support Materials. U.S. EPA Region 6 U.S. EPA Multimedia Planning and Permitting Division Office of Solid Waste, Center for Combustion Science and Engineering. Available at http://www.epa.gov/epaoswer/hazwaste/combust/riskvol.htm#volume1

EPA 2000. “Exposure and Human Health Reassessment of 2,3,7,8-Tetrachlorodibenzo-p-Dioxin (TCDD) and Related Compounds: Part III: Integrated Summary and Risk Characterization for 2,3,7,8-Tetrachlorodibenzo-p-Dioxin (TCDD) and Related Compounds”. Report EPA/600/P-00/001Bg, September 2000. United States Environmental Protection Agency. Washington, DC 20460. Available at www.epa.gov/ncea. ETSU 1996. "Economic Evaluation of the Draft Incineration Directive". Report for the European Commission DG11. ETSU, Harwell Laboratory, Didcot, Oxfordshire OX11 0RA, UK.

Eschenroeder, A.Q., S. Wolff, A. Taylor and D. Burmaster (1990) *Air and Waste Manangement Association Annual Meeting, Pittsburgh, PA.*

Eschenroeder, A.Q., S. Wolff, A. Taylor and D. Burmaster (1990) *Air and Waste Manangement Association Annual Meeting, Pittsburgh, PA.*

ETSU 1996. "Economic Evaluation of the Draft Incineration Directive". Report for the European Commission DG11. ETSU, Harwell Laboratory, Didcot, Oxfordshire OX11 0RA, UK.

ExternE (1999) *ExternE Externalities of Energy, Vol. 9: Fuel cycles for Emerging and End-Use Technologies, Transport and Waste.*Prepared by AEA Technologies (eds.) for European Communities. Part III (Waste) by Ascari, S. and M. Fontana of IEFE, Italy.

Finnveden, G., A-C. Albertsson, J. Beredson. E. Erikson, L.O. Höglund, S. Karlsson and J-O. Sundqvist, (1995) "Solid waste treatment within the framework of life-cycle assessment." *Journal of Cleaner Production* **3**(4) pp 189-199.

Friedrich R. & Bickel P. (2001) Environmental External Costs of Transport. Springer Verlag Heidelberg 2001.

Garrod, G. & K. Willis (1998), ‘Estimating lost amenity due to landfill waste disposal’, Resources, Conservation and Recycling, 22 (1–2), pp 83–95.

Gendebien, A., Pauwels, M., Constant, M., Ledrup-Damanet, M.J., Nyns, E.J., Willumsen, H.-C., Batson, J., Fabry, R. and Ferrero, G.-L. (1992) *Landfill gas from environment to energy*. Eur 14017r1 EN CEC. Luxembourg.

Hickman, H.L. 1988 Regulating municipal solid waste management in the United States. In *Proceedings of Conference on Landfill Gas and Anaerobic Digestion in Solid Waste*, Chester.

IEA [International Energy Agency] (2000) *International Perspective on Energy Recovery from Landfill Gas.* IEA Bioenergy Programme and IEA CADDET Renewable Energy Technologies Programme. Available Online at http://www.caddet-ee.org/public/uploads/pdfs/Report/ar06.pdf

ILO, (1998) *Encyclopaedia of Occupational Health and Safety*, 4th Ed., Geneva, 1998, Vol.3, p. 101.22. http://www.ilo.org/encyclopaedia/

Görner K. Waste Incineration - European State of the Art and New Developments. IFRF Combustion Journal. Article Number 200303, July 2003. http://www.journal.ifrf.net/library/july2003/2003-03-2303Goerner.pdf . accessed 08/12/04.

Kiel, K.A. and K. T. McClain (1995) House prices during siting decision stages: the case of an incinerator from rumor through operation. *Journal of Environmental Economics and Management* **28**, p.241-255.

Malkin, R, P Brandt-Rauf, J Graziano, and M Parides. 1992. "Blood lead levels in incinerator workers." *Environ Res* 59(1):265-270.

Miranda, M.L. & Hale, B. (1997) 'Waste not, want not: the private and social costs of waste-to-energy production', *Energy Policy*, **25**(6), 587-600.

Rabl A & JV Spadaro. 1999. "Environmental Damages and Costs: an Analysis of Uncertainties". *Environment International*, Vol.25, 29-46 (1999)

Rabl A, JV Spadaro & PD McGavran 1998. "Health Risks of Air Pollution from Incinerators: a Perspective". Waste Management & Research, vol.16, pp.365-388. This paper received the 1998 ISWA (International Solid Waste Association) Publication Award.

Schilling, CJ, IP Tams, RS Schilling, A Nevitt, CE Rossiter, and B Wilkinson. 1988. "A survey into the respiratory effects of prolonged exposure to pulverised fuel ash." *Br J Ind Med* 45(12):810-817.

Spadaro JV & A Rabl 1999. "Estimates of real damage from air pollution: site dependence and simple impact indices for LCA". *International J. of Life Cycle Assessment*, Vol.4 (4), 229-243.

SWPHO (2002) *Waste management and public health: the state of the evidence.* South West Public Health Observatory.

Taylor, T., S. Arnold and A. Markandya, (2004) *Financial and External Costs of Waste Management Options in the European Union - An Overview of the Literature.* Working Paper, University of Bath.

Theunis J., Van der Linden A., Int Panis L., Claeys P., Torfs R., 2003. Actualisatie van de Vito studie rond verwerking van restafval toegespast op de verwerking van restafval in de Provincie Limburg. Vito report. (Comparison of waste treatment options, an update for the Province of Limburg).

types of waste (Miljøkostnader knyttet til ulike typer avfall). Final Report (only available in Norwegian).

types of waste (Miljøkostnader knyttet til ulike typer avfall). Final Report (only available in Norwegian).

USEPA (1988) *Preliminary Draft EIS Air Emissions from MSW Landfills - Background Information for Proposed Standards and Guidelines*. Research Triangle Park, NC.

Valenti, M. (1999) "Trash, Heat and Ash" in *Mechanical Engineering Magazine* August 1999. Available online at http://www.memagazine.org/backissues/august99/features/trash/trash.html

Vrancken K., Torfs R., Van Der Linden A., 2000. Vergelijking van verwerkingsscenario's voor restfractie van HHA en niet-specifieke categorie II bedrijfsafval. Vito report. (Comparison of waste treatment options for the Flemish waste authorities(OVAM)).

Vrijheid M., H. Dolk, B. Armstrong, L. Abramsky, F. Bianchi, I. Fazarinc, E. Garne, R. Ide, V. Nelen, E. Robert, J.E.S. Scott, D. Stone, R. Tenconi (2002) "Chomosomal congenital birth abnormalities and residence near hazardous waste landfill sites" *The Lancet,* **359** pp320-322

Waste Online (2004) *Incineration* [www] accessed 05.04.04 http://www.wasteonline.org.uk/resources/Wasteguide/mn_wmo_thermtreatment_incin.html

PART 2 :
STAKEHOLDER WORKSHOP AND MULTICRITERIA ANALYSIS

1 INTRODUCTION

The final stage of the proposed integrated framework involves stakeholders concerned with the two policy areas. They are invited to a workshop, one for each case study, where the policy options and their performances in a number of economic and environmental criteria are presented. Economic criteria refer to the costs induced by each policy option, while environmental criteria include both, monetized and non-monetized impacts. The essence of the problem faced by the stakeholders is that there is no option that is superior in all considered criteria. This means that the best solution can only result by designating the relative importance of criteria for solving the emerging conflict.

Multi-Criteria Analysis (MCA) (often called Multi-Criteria Decision Analysis -MCDA) provides the methodological tools to handle this type of conflict in a systematic way. It is a formal approach helping decision makers to effectively handle complex decision situations in which the level of conflict between criteria is such that intuitive solutions cannot be satisfactory. In addition, MCA can help in resolving disagreement if stakeholders have different views on the relative importance of the considered criteria. It is important to stress that MCA is not a tool providing the right solution in a decision problem, simply because no such solution exists. Instead, it is an aid to decision making that helps stakeholders organize the available information, think on the consequences, explore their own wishes and tolerances and minimize the possibility for a post-decision disappointment (Belton and Stewart, 2002).

In the last 30 years, MCA methods have known a remarkable progress in the framework of Operational Research and Decision Sciences. This progress is manifested not only in the impressive number of communications in scientific journals and conferences, but merely in the increasing use of relevant approaches in real-life problems in the public or private sector. Because of its capacity to handle conflicting decision situations, MCA is particularly suited for sustainability problems where the aim is the reconciliation of economic, environmental and social values.

MCA relies on preference elicitation methods aiming at gradually constructing values for unfamiliar goods, rather than revealing some 'true' values hidden in human mind. More specifically, the following advantages of MCDA methods are emphasized in the literature:

- MCA methods can **consider a large variety of criteria** independently of the type of data (quantitative or qualitative) and the measurement scale. Hence, it allows for a comprehensive analysis including all various aspects of sustainability and not only marketed goods or monetized costs and benefits [Omann, 2000].
- MCA is **directly involving the stakeholders** facing a particular decision problem in order to detect their own preferences and values regarding the decision criteria. Hence, the extracted values better reflect the concerns and priorities of the people concerned.
- MCA is acting as an **interactive learning procedure** that motivates stakeholders to think harder about the conflicts addressed by taking into account other points of view and opposing arguments [Martinez-Alier et al., 1998, Omann, 2000]. Such a transparent and constructive procedure enables stakeholders to better understand the problem at hand and eventually arrive at a better and commonly accepted solution [Faucheux and Froger, 1995; Lahdelma et al. 2000].

- MCA is a **multi-disciplinary approach** that is capable of better capturing the complexity of natural systems, the plurality of values associated with environmental goods and the variant perceptions of sustainable development [Toman, 1997]. The stakeholders participating in a MCDA procedure have the possibility and the responsibility to go beyond their own discipline and to take into account perspectives and information that are possibly fields from other disciplines.

This chapter is organized as follows: section 6.2 presents the multicriteria method and the weighting technique providing the framework for the participation of stakeholders in the workshops, section 6.3 gives the organizational details for the two workshops and the input to MCA provided to the stakeholders, section 6.4 presents the obtained results and section 6.5 summarizes the main findings and draws conclusions.

1.1 Methodology for MCA

A multiplicity of MCA methods is currently available for use in a wide variety of situations. Furthermore, several weighting techniques have been developed to help stakeholders involved in a MCA procedure understand and articulate their preferences concerning the relative importance of the examined criteria. The following paragraphs present the tools selected in the present applications.

1.1.1 Selection of MCDA method

Depending on the theoretical background and the keyassumptions adopted in the modeling procedure and in the aggregation of the performances and/or preferences regarding the decision criteria, MCDA methods can be divided into two broad categories, as follows:

➔ **Multi-Attribute Value Theory methods** (MAVT) trying to associate a unique number (‘value’) representing the overall strength of each alternative if all criteria are taken into account.

➔ **Outranking methods** trying to associate a preference index to each pair of alternatives that is further exploited to rank alternatives in a descending order of preference.

Between these two categories, MAVT methods present the advantage of greater simplicity and transparency. In addition, they are more compatible with CBA because they use a similar utilitarian background where decisions result from explicit or implicit trade-offs between conflicting interests or points of view. Although trade-offs are not acceptable under a strong sustainability perspective, they represent the most realistic approach for implementing the sustainability concept in practice.
The starting point in all MAVT models is the definition of partial value (or utility) functions in each criterion for reducing performances in the [0-1] interval. Value functions differ according to the attitude of stakeholders against risk. One can distinguish risk prone, risk neutral and risk averse attitude as shown in Figure 6.1. It is accepted that for the type of sustainability criteria under consideration, decisions should adopt a risk averse attitude, meaning that even a small improvement from the worst impact level is valued higher than the same improvement at already reduced impact levels.

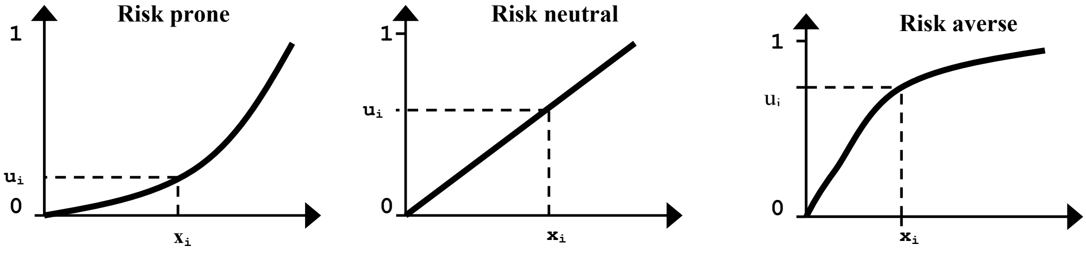


Figure 20: The three types of utility/value functions

Partial values are then aggregated for deriving total values and constructing a complete preorder of the examined alternatives. The transition from partial to global value functions (taking into account the whole set of criteria) implies the use of an aggregation formula together with the inter-criterion preferences provided by the decision maker. The simplest and most commonly used aggregation model is the additive one:

$$V(a) = \sum_i w_i \bullet v_i(a)$$

where, ***V(a)*** is the total value associated with each alternative ***a***, and $\boldsymbol{w_i}$ is the weight attached to each criterion ***i*** by the stakeholder. Policy options are ranked according to *V(a)* from the highest to the lowest value.

A commonly applied MAVT model is the method of *"Displaced Ideal"* (Zeleny, 1982) which has been selected for the present applications. It is based on an additive aggregation model measuring the distance of each alternative option from a virtual *'ideal solution', I.* The solution I does not belong to the set of examined solutions and is used as a reference point to evaluate their overall performance. The solution I is defined on the basis of the best score recorded in each criterion, as follows:

$$I = (g_1^*, g_2^*, \ldots, g_m^*)$$

where,

$g_j^* = \text{Max } \{g_j(1), g_j(2), \ldots, g_j(n)\}$ for maximization criteria
$g_j^* = \text{Min } \{g_j(1), g_j(2), \ldots, g_j(n)\}$ for minimization criteria
m: the number of criteria, n: the number of alternatives

It is clear that by reducing performances into the [0-1] interval, the ideal solution is defined by the unity vector: I = (1, 1, ..., 1).

The distance of an alternative α from I depends on the relative deviations from the best scores in each criterion. Figure 6.2 shows the distance from the 'ideal solution' in the case of a bi-criterion problem. It can also be seen that the choice is practically confined to the set of alternatives α_i, since alternatives b_i are dominated by at least one of the efficient solutions α_i, and can therefore excluded from the analysis.

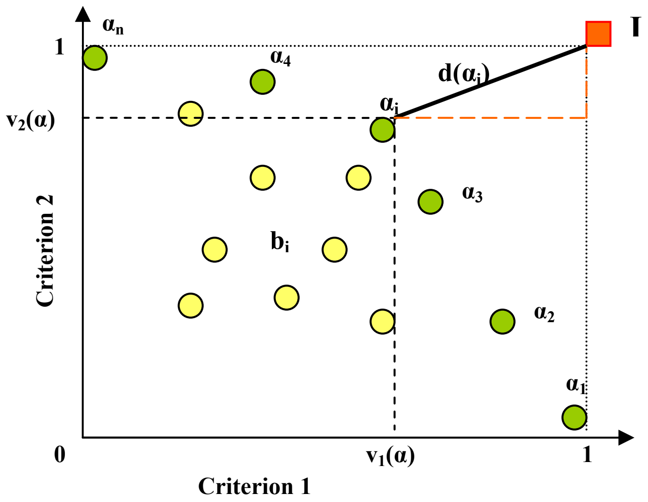


Figure 21: The distance of alternative α_i from the ideal solution I

The most commonly applied aggregation model is based on the concept of Euclidean distance. The distance metrics d(α) results as the weighted average of the partial deviations $v_j(\alpha)$ from the best score in each of the m evaluation criteria, weighted according to the relative importance of the considered criteria.

$$d(a) = \sqrt{w_1^2(1-v_1(a))^2 + w_2^2(1-v_2(a))^2 + ...w_m^2(1-v_m(a))^2}$$

Because of the algebraic definition of 'distance', solutions with large deviations in one criterion are disfavored since such large deviations are raised to square and cannot easily be compensated by good scores in other criteria. This is the most appreciated advantage of the method of *"Displaced Ideal"* if stakeholders try to avoid unbalanced solutions.

1.1.2 Selection of weighting technique

Weights in MAVT methods play the role of scaling factors in the sense that they relate scores in one criterion, to the scores of all other criteria. This means that by assigning weights of relative importance, stakeholders implicitly determine how much units in one criterion they are willing to give up, in order to improve the performance of another criterion by one unit. So, if the weight of criterion ***i*** is double the weight of criterion ***j***, then the stakeholder values 10 units on criterion ***i***, the same as 20 units on criterion ***j***. In order for the stakeholders to more clearly realize their preferences in terms of the necessary trade-offs between criteria, weights are defined on the initial natural scales and by taking into account the absolute level of performances and absolute differences in scores.
From a literature review different weighting methods have been identified. These can be distinguished in compensatory and non-compensatory methods. Since the additive aggregation model of MAVT assumes strong compensation between criteria, a compensatory weighting method should be used here. Among the most known compensatory weighting techniques the SWING method [von Winterfeldt & Edwards, 1986] was selected because of the following advantages:

- Simplicity and transparency in the elicitation of preferences;

- Sensitivity to impact range (thus enhancing the reliability of the weights for a specific decision context);
- Ability to handle problems regardless of the number of alternatives or criteria;
- Avoidance of direct questions on trade-offs (usually awkward for the respondents).

The SWING method highlights the hidden dilemmas behind a number of mutually exclusive options evaluated across multiple criteria by making stakeholders aware of the potential gains and losses implied by their choice. For this purpose, two extreme hypothetical Scenarios W and B are constructed, the former presenting the worst performance in all criteria (worst score of the examined alternative options) and the latter the corresponding best performance. It is assumed that the current state for the stakeholder is Scenario W. The preferences are elicited by asking the stakeholders to carefully look at the potential gains of moving from W to B and then to decide which of the criteria they want to first shift to Scenario B. Assuming that this first swing is valued with 100 units on a hypothetical value scale, the stakeholders are asked to assign a value (<100) to the second criterion moved to B, then to the third and so on until the last criterion is moved to Scenario B.

1.1.3 Indirect monetization of environmental impacts

Furthermore an attempt was made to derive implied monetary equivalents for some of the non-monetized impacts. That is possible if at least one of the criteria is stated in monetary terms (for instance the cost of pollution abatement for the Flemish case study on waste). The procedure for this was developed during the ExternE-Pol project [2004]. The scope was to relate the weight of each non-monetized criterion with the weight of a cost criterion, both reduced to the corresponding impact scale:

$$\frac{w_i}{IR_i} = \frac{w_c}{IR_c}$$

where IR_i and IR_c denote the impact range (i.e. the maximum potential gain) of the physical impact i and of the cost criterion, respectively. The underlying assumption is that the value functions for all impact categories are linear, e.g. improvements in the impact level are valued the same, independently from the absolute impact level. The linearity assumption reflects a neutral behaviour towards risk, which is a reasonable hypothesis, for the limited range of impact levels under consideration. Hence, the per unit monetary value of the physical impact i, m_i is calculated as

$$m_i = \frac{IR_c \cdot w_i}{IR_i \cdot w_c}$$

It can be seen that for each pair i/c the higher the ratio of weights and/or the lower the ratio of impact ranges, the higher is the unit monetary value implicitly assigned to the physical impact i.

1.2 The Workshops

During the last month of the SusTools project (October 2004) a workshop was held at the EC in Brussels. For this purpose, the details of the communication procedure have been established as follows:

- Select stakeholders to be invited to the workshop;
- Prepare summary of the case studies to be sent with the invitation;
- Formulate questionnair to be completed by stakeholders, to obtain their input for the MCA;
- Hold the workshop and perform the MCA in an interactive way.
- Define how the outcome of the MCA will be presented to and discussed with the stakeholders.

Invitations were sent to about 20 individuals or organizations (including NGOs, industry, national governments and the EC), together with a brief summary of the policy options and results. Most of the invitees expressed their interest, even though many had already other commitments for the proposed date. Finally 13 individuals attended the workshop on waste treatment.
During the workshops members of the SusTools team presented an overview of the tools used (LCA, IPA, CBA, MCA), the policy options under consideration and the results obtained so far. The discussions highlighted the importance of feedback from stakeholders. Several important comments concerned the assumptions and data for the analysis; these comments have been taken into account in our final reports. Following this discussion, the MCA framework was explained and instructions were given for the completion of questionnaires.
Tables 6.1, 6.2 and 6.3 show the forms for eliciting the weights of the criteria in each policy problem. The dilemma is represented by the two extreme scores recorded in each criterion (best and worst), and stakeholders were asked to focus on their difference illustrating the degree each impact level can be improved by the decision taken. Moreover, this difference was put into perspective by means of general national figures in order to facilitate stakeholders to express their preferences.

Table 36: The SWING questionnaire for Waste Treatment (generic problem, France)

Criterion	Worst score	Best score	Difference = potential gain	Putting difference into perspective	VALUE
Net Cost *(€/t)*	95	52	**43** *€/t*	*14 € per capita per year*	
Global warming *(CO2 kgeq/t)*	650	420	**230** *kg CO2 eq/t*	*< 0.8% of the per capita CO2 emission per year*	
Health Impacts *(€/t)*	1.2	-1.9	**3.1** *€/t*	*1 € per capita per year*	
Residue to landfill *(kg/t)*	1000	50	**950** *kg/t*	*decrease by 95% of MSW going to landfill*	
Recovery Fe, non-Fe *(kg/ t)*	0	22	**22** *kg/t*	*about 20 kg steel and 2 kg Al. You can make 300 steel/Al cans out of it, or 75 Al cans*	

Table 37: The SWING questionnaire for Waste Treatment (site-specific problem, Flanders)

Criterion	Worst score	Best score	Difference = potential gain	Putting difference into perspective	VALUE
Net Cost *(€/t)*	111	68	**43** *€/t*	*7 € per capita per year*	
Global warming *(CO2 kgeq/t)*	765	552	**213** *kg CO2 eq/t*	*<0.5% of the per capita CO2 emission per year*	
Health Impacts *(€/t)*	7.4	1.8	**5.6** *€/t*	*0.8 € per capita per year*	
Impacts on ecosystems *(milli- Acid. Potentials)*	0.05	0.01	**0.04** ***milli- Acid. Potentials***	*less than 0.001% of the per capita acidifying emission per year in Flanders*	
Residue to landfill *(kg/t)*	150	70	**80** *kg/t*	*decrease of 50 % of MSW going to landfill*	
Upstream impacts *(€/t)*	3.1	0.6	**2.5** *€/t*	0.4 *€ per capita per year*	
Recovery Fe, Non-Fe *(kg/ t)*	34.4	48.7	**14.3** *kg/t*	*≈13kg steel and 0.8 kg Al. You can make 200 steel/Al cans out of it, or 50 Al cans*	
Recovery of Minerals *(kg/ t)*	81	160	**80** *kg/t*	*12 kg/capita/year*	

In addition the stakeholders were also asked about their intuitive ranking of the policy options. The completed questionnaires were analysed during lunch break and then the results were presented to stakeholders.
The participants found the MCA procedure very helpful and most agreed that the resulting solution from MCA represents a realistic compromise between the conflicting criteria. It is worth noticing that in all cases the formal MCA results were very similar with the intuitive ranking.

1.3 Results of the MCA

1.3.1 MCA of Waste Treatment options - Generic Problem, France

The policy options examined in this case study include different incineration and landfill technologies. The cost of each method is considered along with the 4 environmental criteria shown in Table 36. Environmental impacts have been calculated for France.
The weights calculated on the basis of the values provided by the 13 stakeholders present significant differences (Figure 22). However, the average weights show a smoothing of these differences with *'Cost'*, *'Human health'* and *'Residue to Landfill'* being assigned with slightly higher weights than the other two criteria.

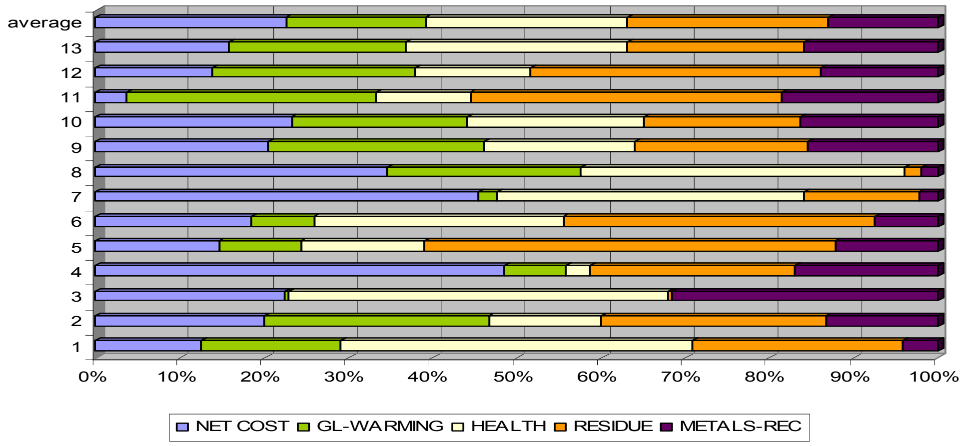


Figure 22: The weights of individual stakeholders (waste treatment, France)

From Figure 23, showing the dispersion of values in each criterion. it can be seen that the biggest divergence in stakeholders' opinion is observed in these three criteria, with weights ranging from very low values (<5%) to approximately 50%.

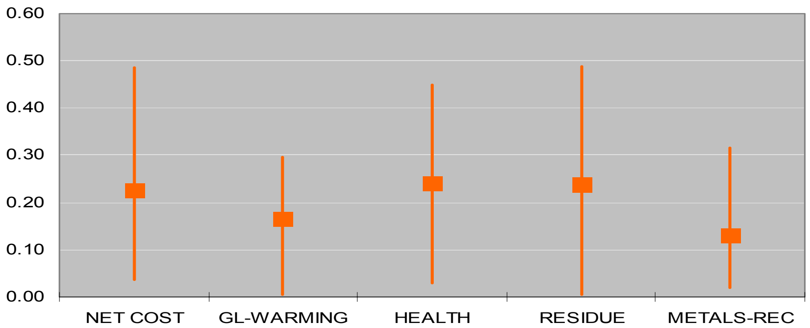


Figure 23: Dispersion of weights (waste treatment, France)

The ranking provided by the average weights shows that *'Incineration - Electricity/Heat'* is by far closer to the ideal solution with the two landfill options following in the second place (Figure 24).

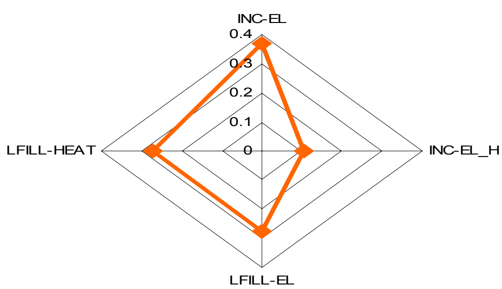


Figure 24: The distance from ideal solution (waste treatment, France)

Despite weighting differences, Fig.6.6 shows that no significant differences exist between individual rankings. *'Incineration - Electricity/Heat'* appears as the most preferred option for all participants, except for No 7 and 8 who rank *'Landfill- Heat'* first because they attribute higher weight to the cost criterion.

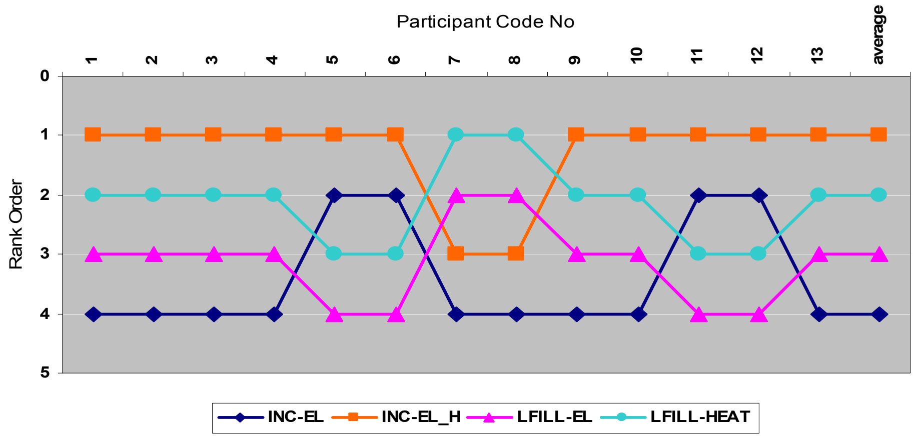


Figure 25: Rank order of policy options by individual stakeholders (waste treatment, France)

'Landfill- Heat' is placed by most participants in the second place, although for 4 participants the 2nd best choice is *'Incineration-Electricity'* and *'Landfill-Electricity'* for participants No 7 and 8.
The dominance of the *'Incineration - Electricity/Heat'* option is confirmed also by the intuitive ranking provided by stakeholders. As shown in Figure 26 this option is ranked in the first two places by 11 out of the 13 stakeholders in both MCA and intuitive ranking.

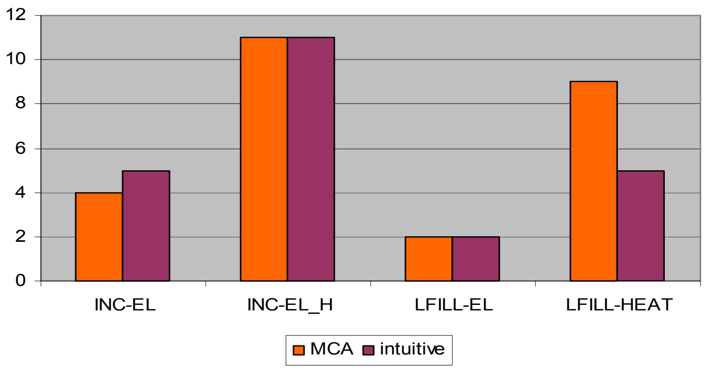


Figure 26: Number of appearances in the first two places (waste treatment, France)

1.3.2 MCA of Incineration options – Site-specific Problem, Flanders

The policy options examined in this casestudy include different incineration technologies. The cost of each method is considered along with 7 environmental criteria shown in Table 37: The SWING questionnaire for Waste Treatment (site-specific problem, Flanders)Table 37. Environmental impacts have been calculated for a specific site in Flanders.
The weights for the 13 stakeholders present significant differences (Figure 27). *'Cost'* and *'Human health'*, and *'Residue to Landfill'* are again considered as important criteria by the

majority of participants, with *'Residue to Landfill'* and *'Global Warming'* appearing also as very important criteria.

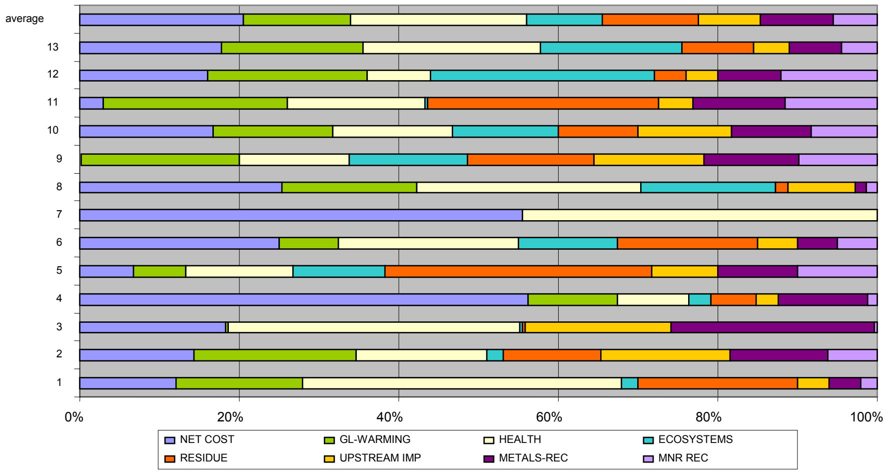


Figure 27: The weights of individual stakeholders (waste treatment, Flanders)

From Figure 28 showing the dispersion of values in each criterion it can be seen that the biggest divergence in stakeholders' opinion is again observed in the most important criteria, especially in *'Cost'*, but also in *'Health Impacts'* and *'Global Warming'*.

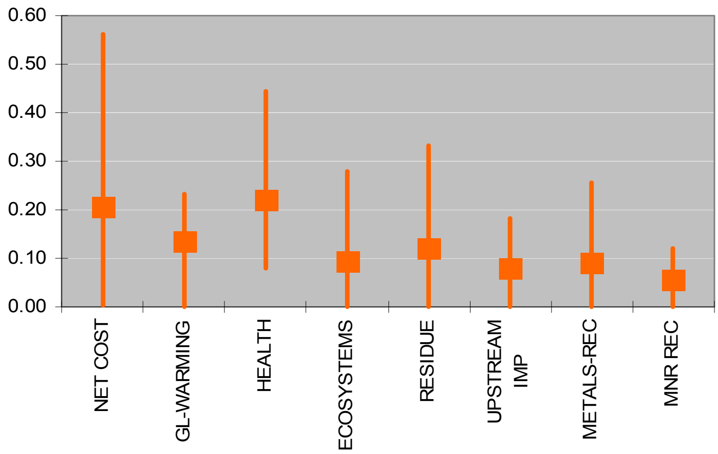


Figure 28: Dispersion of weights (waste treatment, Flanders).

The ranking provided by the average weights shows that *'Pretreatment-1-CFB'* is by far closer to the ideal solution, while the most distant is the simple *'Grate Furnace'* technology (Figure 29).

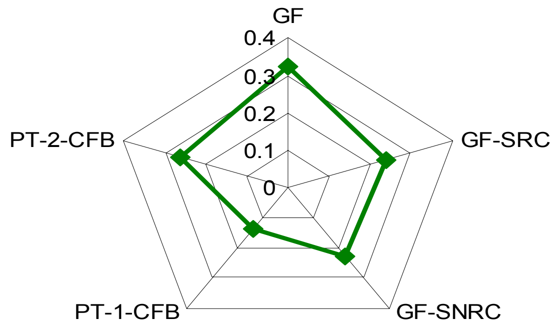


Figure 29: The distance from ideal solution (waste treatment, Flanders)

Despite weighting differences, Figure 30 shows that no significant differences exist between individual rankings. *'Pretreatment-1-CFB'* appears as the most preferred option for all participants, except for participant 5 who assigns low weight to cost and thus prefers *'Pretreatment-2-CFB'* and for participant 7, who from a different point of view (high weight to cost) prefers the grate furnace technology with selective catalytic reduction (GF-SCR). For the intermediate places there is a disagreement, while most of them converge to the conclusion that grate furnace without NOx reduction is the least preferred option.

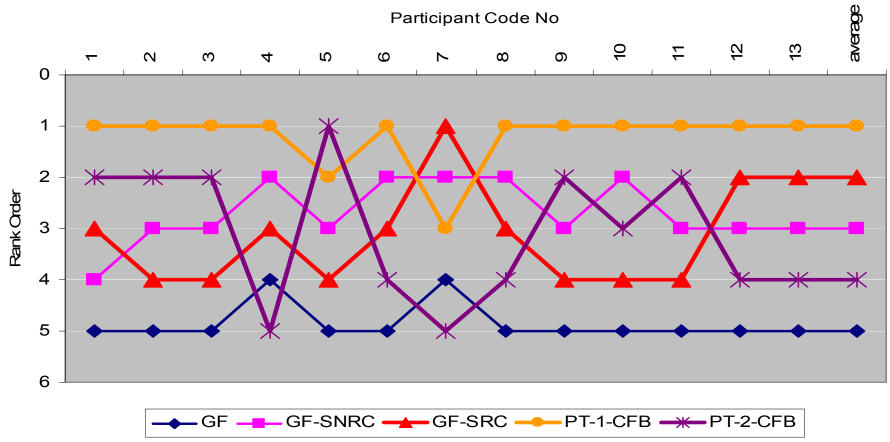


Figure 30: Rank order of policy options by individual stakeholders (waste treatment, Flanders)

The comparison between multicriteria and intuitive opinions demonstrates that there is no complete concurrence between the two rankings. As shown in Figure 31, intuitive preferences of stakeholders are shared beween *'Pretreatment-1-CFB'* and *'Grate Furnace – SNCR'*, while considerable acceptance is received also by *'Grate Furnace-SCR'* and *'Pretreatment-2-CFB'*.

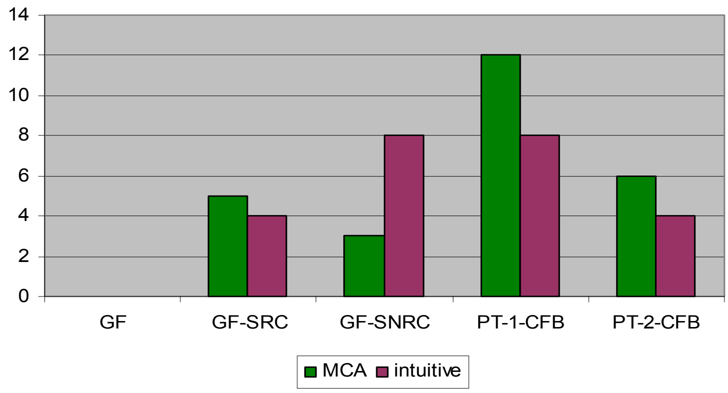


Figure 31: Number of appearances in the first two places (waste treatment, Flanders)

1.3.3 Monetary equivalents of stakeholders' preferences

The MCA also revealed the monetary equivalents for non-monetized impacts that were implicit in the stakeholders' preferences. It has been found that these monetary equivalents are highly dependent on the decision context, e.g. the type of problem, the type and number of criteria and the range of impact values in each criterion. For example, the global warming effect – expressed in kg of CO_{2qu}.- was valued at 13 €/t_{CO2eq} in the nitrogen fertilizer problem, but at around 170-190 €/t_{CO2eq} in the two case studies on waste treatment.
A general observation is that most environmental impacts were overestimated in the two waste treatment problems in comparison with either market values or ExternE estimates. This leads us to the recommendation that the preference elicitation could be assisted by merging MCA with monetary valuation. Implied monetary values could serve as a yardstick making the necessary trade-offs more tangible and leading preferences towards more realistic ranges. This has been already tested in [Diakoulaki and Grafakos, 2004] and indicates a promising direction for further research.

1.3.4 Conclusions About Stakeholder Involvement

The concept of sustainable development as an evolutionary process encompassing economic, environmental and social dimensions by definition gives a multicriteria character to the task of tracing policies for sustainability. Any policy choice can possibly satisfy one dimension by at the same time being in contradiction with the others. Conflicts also exist within the same dimension. It is very common that by effectively coping with one environmental aspect, other aspects become worse.
Therefore, the proposed integrated approach developed in this project for dealing with sustainability problems incorporates a MCA method as a final stage, one capable of assisting policy makers to better understand and consciously resolve the emerging conflicts. In fact, MCA is an open and flexible assessment framework that can be easily adapted to the particularities of the problem under consideration. Moreover, MCA provides a platform on which different points of view can be expressed and converge on a unique solution that is more probable to be widely adopted and implemented in practice.

Following an electronic communication with a large list of potentially interested individuals or organizations, two workshops have been organized in which the proposed methodological approach was presented together with its implementation in the respective policy problem. The participation of the stakeholders has offered the opportunity to communicate the consequences of alternative policy options for waste treatment and use of nitrogen fertilizer. The stakeholder involvement was critical in highlighting different aspects of the problems and to identify aspects needing further research. Moreover, the stakeholders were confronted with the dilemmas raised and tried to solve them through a structured preference elicitation procedure that helped balancing the pros and cons and reach the most satisfactory compromise. The transparency of the procedure and the open discussion on critical points led to the common approval of this compromise solution (which for all case studies turned out the same as the one chosen intuitively by most participants).

In any case, an interactive-iterative procedure of stakeholders' involvement is necessary for the assessment of policy options, and the workshops have demonstrated the value of the framework proposed in the SusTools project.

2 REFERENCES

Belton, V. and Stewart, T. J., (2002), Multiple Criteria Decision Analysis: An integrated approach. Kluwer Academic Publishers, Dordrecht.

Diakoulaki D. and Grafakos S. (2004), Treatment of uncertainty in weights elicitation through the disclosure of the hidden monetary values assigned to sustainability criteria. 15th Mini-EURO Conference, 'Managing Uncertainty in Decision Support Models'. Coimbra, Portugal.

EC (2004), Externe-Pol: Externalities of Energy: Extension of accounting framework and policy applications, Contract No: ENG1-CT-2002-00609.

Faucheux, S., and Froger, G., (1995), Decision making under environmental uncertainty. Ecological Economics, 15, 29-42

Lahdelma, R., Salminen, P., Hokkanen, J., (2000), Using Multicriteria Methods in Environmental Planning and Management. Environmental Management, 26 (6), 595-605.

Martinez–Alier J., Munda G., O'Neill J., (1998), Weak comparability of values as a foundation for ecological economics. Ecological Economics, 26, 277-286.

Omann, I., (2000), How can Multi-criteria Decision Analysis contribute to environmental policy making? A case study on macro-sustainability in Germany. paper for the Third International Conference of the European Society for Ecological Economics, Vienna.

Toman, M., (1998), Sustainable Decision Making: The state of the art from an economic perspective. Discussion Paper 98–39, Resources for the Future, Washington DC.

von Winterfeldt D, Edwards W, (1986) Decision analysis and Behavioural research. Cambridge University Press, New York.

Zeleny M., (1982), Multiple Criteria Decision Making, McGraw-Hill, NY.

APPENDIX A

Proceedings of the Workshop
“Evaluation of treatment options for municipal solid waste”
Thursday 14 October 2004
European Commission in Brussels

Invitations to the workshop had been sent to about 20 individuals in a variety of organizations, including NGOs, industry, national governments and the European Commission. The invitees were also sent a summary of the results to be discussed at the workshop. The time for preparing the workshop had unfortunately been very short because of the summer vacation and the Olympic games that prevented an earlier coordination meeting with the MCA team of the National Technical University of Athens. The invitations were not ready for mailing until about a month before the workshop, and many of the invitees had already other commitments. The individuals who were able to attend are listed in Table 6.1. Several invitees who were unable to come expressed great interest and asked to be informed about the outcome of the workshop and about further research on this problem.

Table B.1. Participants at workshop on treatment options for municipal solid waste (in addition to members of SusTools team: Ari Rabl, Assaad Zhougaib, Mithra Moezzi, Mike Holland, Danae Diakoulaki, George Mavrotas, Rudi Torfs, Tim Taylor, Dimcho Boyadjiev).

Name	**Organization**	**e-mail**
Marialuisa Tamborra	EC DG RTD	marialuisa.tamborra@cec.eu.int
Luk Umans	OVAM	luk.umans@ovam.be
Peter van Acker	OVAM	peter.van.acker@ovam.be
David Pennington	JRC	david.pennington@jrc.it
Viorel Vulturescu	EC DG Research	viorel.vulturescu@cec.eu.int
Steven Geirnaert	Bond Beter Leefmilieu	steven.geirnaert@bblv.be
Gar Crawford	ONYX	gcrawford@cgea.fr
Francis Angotti	ONYX	FANGOTTI@cgea.fr
Hubert de Chefdebien	CNIM	hdechefdebien@cnim.com
Emmanuelle Aoustin	Véolia Environnement	Emmanuelle.AOUSTIN@groupve.com
Guido Wauters	INDAVER MV	guido.wauters@indaver.be
Philippe Bajeat	ADEME	philippe.bajeat@ademe.fr
Thomas Gaudin	ADEME	Thomas.Gaudin@ademe.fr

Agenda of workshop on treatment options for municipal solid waste

9:30 Marialuisa Tamborra, followed by Ari Rabl: introduction
9:50 Mike Holland: tools, followed by discussion
10:20 Ari Rabl: French case study, followed by discussion
11:15 coffee
11:30 Rudi Torfs: Flemish case study, followed by discussion
12:15 Danae Diakoulaki: multicriteria analysis (MCA), followed by discussion
12:40 Stakeholders: fill in forms for MCA
13:00 lunch
14:00 interpretation of results and discussion
15:00 coffee and end

The discussions highlighted the importance of feedback from stakeholders. Some of the discussions concerned clarifications of the results and of the methodology. For example David Pennington pointed out that the methodology of LCA impact assessment has been evolving and now even site-specific effects can be taken into account. In fact, the quantification of impacts and damage costs (Impact Pathway Analysis) should ideally be part of LCA, although in practice there are major differences between the IPA of ExternE and the impact assessment of most LCA practitioners. In particular, ExternE uses dose-response functions and proceeds all the way to the monetary valuation of damages, by contrast to the refusal of monetary valuation by most of the LCA community.

Several comments concerned the boundary of the analysis. For example, a participant asked how a cost-benefit analysis (CBA) would take into account the impacts of increased energy use if NOx emissions are reduced by selective catalytic reduction. The answer is that it is straightforward in principle to include such induced effects but that in practice the necessary data may not be readily available or the analyst may overlook the need to include them.

Quite generally the need to include upstream or downstream impacts arises from the lack of complete internalization of the external costs. If all external costs that arise upstream or downstream were internalized by an optimal pollution tax (i.e. a tax equal to the marginal damage) or by tradable permits that are auctioned by the government, there would be no longer any need for LCA – otherwise there would be double counting of upstream costs in a CBA. Of course, that is not the case at the present time at least for most pollutants and in most countries. If, on the other hand, external costs are internalized by tradable permits that are free, the residual damage will not have been paid by the polluters and should be included in the analysis.

Another problem with CBA is the risk that impacts for which monetary evaluation was not possible may be simply forgotten, i.e. effectively set equal to zero, instead of being considered in the MCA.

Some recommendations were made for the presentation of results. The importance of showing uncertainties and sensitivity to assumptions was emphasized, e.g. assumptions about % of CH4 that escapes from a landfill to the atmosphere. This particular sensitivity is actually quite easy to show by indicating in the caption what percentage of the greenhouse gas emission is due to this CH4, so the reader can mentally change the length of the respective bar in the graph. This should be done in the final report and in future studies.

Concerning the value of energy recovery, one participant argued that the benefits of reducing the risks of nuclear power and the scarcity of fossil fuels should be taken into account. However, there are no reliable cost estimates for these items, and in any case the amount of energy recovered from wastes is to small that such benefits would be negligible compared to the effects that have been quantified. These items could of course be included in the MCA.

One participant objected that the CO2 emissions from landfill should be counted only to the extent that they are due to inorganic waste. However, this objection is not correct as can be seen by comparing two ways of treating organic waste: one where the waste is permanently sealed and one where the CO2 is allowed to escape. Clearly the damage cost is higher for the latter, even though the CO2 is of biogenic origin.

As to recovery of Fe and nonferrous metals, one of the participants pointed out the need to distinguish the quality of the recovered materials between different treatments. Metals recovered after the incinerator are of higher quality than those recovered by pre-treatment. This difference has not been accounted for in the analysis although it could and should be integrated into the MCA. As for emissions from pre-treatment of waste the difficulty lies in the availability of data.

Another impact that has not been considered is the occupational health of workers in the different waste treatment facilities.

Following the discussion on methodological issues, data and assumptions, the MCA framework was presented by giving emphasis to the decision dilemma raised in the particular case study and the way this dilemma was going to be solved with MCA. Explanations were given on the completion of questionnaires by the participants. In fact, stakeholders were asked to provide their intuitive ranking of the considered policy options and to determine weights by means of a compensatory weighting technique (SWING method). Filled questionnaires have been elaborated during the break and the obtained results were presented to the participants.

The weights calculated on the basis of the values provided by the 13 stakeholders present significant differences. *'Human health'*, and *'Residue to Landfill'* are considered as the most important criteria by the majority of participants, followed by *'Net cost'*.

Despite weighting differences, the obtained MCA rankings were similar for most participants. In the generic problem it was found that *'Incineration – Electricity/Heat'* appears as the most preferred policy option for all but 2 participants. In the site-specific problem examining alternative incineration technologies, it is the Fluidized Bed Combustion combined with a simple pre-treatment of waste that was ranked at the 1st place. It is worth noticing that for all stakeholders, MCA results were very similar to the intuitive rank order of the examined policy options.

All participants have found the MCA procedure very helpful and most agreed that the resulting solution from MCA represents a realistic compromise if multiple conflicting criteria are taken into account.

APPENDIX B
Questionnair