report
IVL Swedish Environmental Research Institute

Proceedings from

Workshop on System Studies of Integrated Solid Waste Management in Stockholm 2 - 3 April 2001

Editors:
Jan-Olov Sundqvist, IVL Swedish Environmental Research Institute
Göran Finnveden, fms (Environmental Strategies Research Group),
FOI (Swedish Defense Research Agency)
Johan Sundberg, Chalmer's Institute of Technology; Dept of Energy System Technology
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IVL

Organisation/Organization
IVL Svenska Miljöinstitutet AB
IVL Swedish Environmental Research Institute Ltd.

Adress/address
PO Box 21060
SE-10031 Stockholm, Sweden

Telefonnr/Telephone
+ 46 8 598 563 00

RAPPORTSAMMANFATTNING
Report Summary

Projekttitel/Project title
Proceedings from Workshop on System Studies of Integrated Solid Waste Management in Stockholm 2- 3 April 2001

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Rapportförfattare/author
Jan-Olov Sundqvist, Göran Finnveden, Johan Sundberg (Editors)

Rapportens titel och undertitel/Title and subtitle of the report
Proceedings from Workshop on System Studies of Integrated Solid Waste Management in Stockholm 2- 3 April 2001

Sammanfattning/Summary
This international workshop was held to discuss results and experience from system studies of waste management system and methodological questions and issues based on case studies. The workshop gathered more than 40 participants. These proceedings document more than 20 presentations as well as six discussion sessions.

An overall aim of the workshop was to draw some general conclusions from the presented studies concerning
- waste strategies that generally seem to be favourable or not favourable
- methodological approaches and assumptions that can govern the results
- lack of knowledge.

Considering the environmental aspects, the presented studies indicated that the waste hierarchy seems to be valid:
- Paper and plastic: Material recycling < Incineration < Landfilling
- Biodegradable waste: Incineration ≈ Anaerobic digestion < Composting < Landfilling

A number of key aspects that can influence the results were identified:
- Avoided products (heat, electricity, material, fertiliser produced from waste).
- Efficiency in power plants, heating plants etc. and also recycling plants.
- Emissions and impacts from recycling plants
- Landfilling models, e.g. time frames.
- Final sinks: there should be a distinction between temporary sinks (landfills) and final sinks
- Local conditions and local impacts are often neglected.
- Electricity production
- Choice of alternatives to compare can have an influence on the conclusions drawn.
- Stakeholders' influence.
- Linear modelling.
- Data gaps. Especially data on toxic substances where identified as an important data gap.

Nyckelord samt ev. anknytning till geografiskt område eller näringsgren /Keywords
LCA, system analysis, waste management, ecology, economy, environmental impact, waste hierarchy

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Contents

Introduction

General

There are several universities and research institutes that have developed and implemented models and methods for system analyses of waste management systems. The purpose with this workshop was to gather some of those researchers to discuss results and methods.

This workshop can be seen as a continuation of the earlier workshops that we have held in Sweden: LCA and Solid Waste in Stockholm 1995[1], and System Engineering Models for Waste Management in Goteborg 1998[2]. The Swedish Waste Research Council financed those two workshops. However, since then the Swedish Waste Research Council has been discontinued. Since 1998, the Swedish National Energy Administration has a research program "Energy from Waste", which has included system analyses studies of waste management. This workshop has been financed from that research program.

Scope

The workshop discussed
- results and experience from system studies of waste management system
- methodological questions and issues based on case studies.

An overall aim of the workshop was to draw some general conclusions from the presented studies concerning
- waste strategies that generally seem to be favourable or not favourable
- methodological approaches and assumptions that can govern the results
- lack of knowledge.

The following subjects was given extra attention in discussions:

- Environmental and/or economic consequences of different waste management strategies. It is also of great interest to involve social parameters in environmental/economical studies.

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1 Finnveden, G., and Huppes,G. (1995): Proceedings from the International Workshop ”Life Cycle Assessment and treatment of Solid Waste”, September 28 - 29, 1995, Stockholm, Sweden, AFR Report 98

2 Sundberg J, Nybrant T, Sivertun Å. Seminar: System Engineering Models for Waste Management. Proceedings from the international workshop held in Gothenburg, Sweden 25-25 February 1998. AFR-Report 229

- Methodological issues that are of great importance for the result, e.g. system boundaries, time and space perspectives, choice of complementary systems
- Reaching the users: Models for whom and why? How to present the results.

Disposition of the workshop

Each participant was expected to make a presentation during the workshop. However, some "observers" were accepted. In all there were about 40 participants, and 25 presentations were made. The workshop was divided into six sessions – the first five sessions included presentations followed by a one-hour discussion. The last session was a general discussion where we tried to draw some general conclusions.

Location

The workshop (including accommodation) was held in a conference centre called Johannesberg Castle, which is situated about 25 km from the Stockholm airport (Arlanda).

Organisation

The workshop was organised by:

Jan-Olov Sundqvist, IVL, PO Box 21060, SE-100 31 Stockholm, Phone: + 46 8 598 563 74, Fax: + 46 8 598 563 90, E-mail**:** janolov.sundqvist@ivl.se**.**

Göran Finnveden, fms (Environmental Strategies Research Group), FOI, Box 2142, SE-103 14, Stockholm, Phone + 46 8 402 3827, Fax: + 46 8 402 3801, E-mail: finnveden@fms.ecology.su.se

Johan Sundberg, Chalmers University of Technology. Phone: +46 (0)31 720 8396, Fax: +46 (0)31, E-mail: johan.sundberg@profu.se or josu@entek.chalmers.se

Also involved in the organisation of the workshop was

- Swedish National Energy Administration, Eskilstuna, who financed the workshop.
- The International Expert Group on Life Cycle Assessments and Integrated Solid Waste Management, who had a meeting in Stockholm the day after the workshop.

Session 1

Chairman: Göran Svennsson; Secretary: Jessica Granath

Simon Aumônier
Identifying the Best Practicable Environmental Option: application of LCA and other decision-aiding tools

P.H.Brunner
Material Flow Analysis as a Decision Support Tool for Goal Oriented Waste Management

Roland Clift
CHAMP – A new approach to modelling material recovery, re-use, recycling and reverse logistics

Discussions

Identifying the Best Practicable environmental option assessment: Application of LCA and other decision-aiding tools

***Simon Aumônier*[3]**

Introduction

Waste Strategy 2000, the government's vision for sustainable waste management in England and Wales, published in May 2000, has, at its core, the key principle of the Best Practicable Environmental Option (BPEO). In *Chapter 4, Delivering Change*, the strategy states that *"decisions on waste management, including decisions on suitable sites and installations for treatment and disposal, should be based on a local assessment of the Best Practicable Environmental Option."*

The BPEO concept was first outlined in the 12th Report of the Royal Commission on Environmental Pollution, in 1988, as *"… the outcome of a systematic and consultative decision-making procedure which emphasises the protection and conservation of the environment across land, air and water. The BPEO procedure establishes, for a given set of objectives, the option that provides the most benefits or the least damage to the environment as a whole, at acceptable cost, in the long-term as well as the short-term."*

Identifying the BPEO is a complex task in the context of integrated waste management systems. It requires assessing the performance of options against a number of objectives, and resolving the conflicts between these objectives by making appropriate trade-offs. *Waste Strategy 2000* identifies the multi-criteria technique as providing a rational basis for balancing objectives, as well as setting out the step-by-step approach which simplifies the BPEO procedure. This is consistent with the UK HMIP/Environment Agency BPEO methodology introduced for integrated pollution control processes in the Environment Protection Act 1990, which has been widely applied.

BPEO has not routinely been applied to integrated waste management systems, which appears to be the government's intention, but the seven step approach is interpreted in this context of the proposed development as:

1. define the overall aim and performance criteria (objectives) to be addressed;
2. identify integrated waste management options to meet this aim;

[3] Environmental Resources Management, Eaton House, Wallbrook Court, North Hinksey Lane, Oxford, UK, OX2 0QS. E-mail: sxa@ermuk.com

3. assess the performance of the options against the criteria;
4. value performance;
5. balance the different criteria against one another;
6. evaluate and rank the different options; and
7. analyse the sensitivity of the results.

These case study application shows how each of the seven steps can be resolved in determining BPEO for an integrated waste management system.

Step 1 - Define the overall aim and performance criteria to be addressed

Aim

The overall aim of waste management arrangements with respect to this proposal is the management of municipal waste arising in an English County. Current arisings are approximately 260 000 tonnes per annum. However, since we are concerned with the BPEO over a period of time, the assessment was undertaken for waste arisings forecast for 2004/2005. *Waste Strategy 2000*'s recycling and recovery targets are due to be met in 2005, and new facilities could be expected to be commissioned by this point in time. We have used the 3% annual growth rate employed in the Strategy to predict arisings in 2004/2005. Our estimate is a total waste arising of approximately 310 000 tonnes.

Performance Criteria

The BPEO requires that a range of criteria or objectives are assessed in order for the option which performs best overall to be identified. The Environment Agency of England and Wales has, over a number of years, developed a software tool, *WISARD*, to help with the identification of BPEO. *WISARD* allows many environmental 'flows' and 'impacts' to be assessed, using a Life Cycle Assessment (LCA) approach, and some simplification of these is required to enable a clear analysis to be undertaken. Guidance from the International Standards Organisation (ISO14042) and the Agency's own application of *WISARD* suggest that the 'Problem Oriented Approach' to impact assessment is the most useful. This approach employs the following principal impacts:

- greenhouse effect;
- resource depletion;
- air acidification;
- eutrophication; and
- stratospheric ozone depletion.

In addition to these five impacts, we have added dioxins to the environmental criteria assessed, because of its perceived significance and as an analogue of human health impact. Dioxins are modelled in ***WISARD*** as an environmental flow, rather than related to a specific impact. These six criteria have been modelled using ***WISARD***.

The practicability of integrated waste management options is also captured alongside environmental impacts to ensure, in BPEO, that overall sustainability is addressed. In this assessment, we have employed two criteria to reflect practicability, *financial cost per tonne* and *reliability of delivery*. The former reflects the opportunity costs of more expensive options, and includes the costs of collection, transport and treatment/disposal. The latter reflects how proven is each option and how subject they might be to failure to secure processing sites public participation and materials markets, where required.

Financial costs have been assessed using generic data in ERM's in-house model for waste management and values used in the UK DETR's Regulatory Impact Assessment for *Waste Strategy 2000*. Reliability of delivery has been assessed on a qualitative basis using our expert judgement.

Step 2 - Identify Integrated Waste Management Options

We have developed five alternative integrated waste management options to manage municipal waste arisings in 2005. An example flow diagram is presented in Figure 1. The five options are 'led' by the following technologies:

Option 1: mass burn energy from waste facility;

Option 2: anaerobic digestion;

Option 3: enhanced recycling and composting, landfill in-county;

Option 4: enhanced recycling and composting, landfill out-of-county; and

Option 5: fluidised bed energy from waste facility.

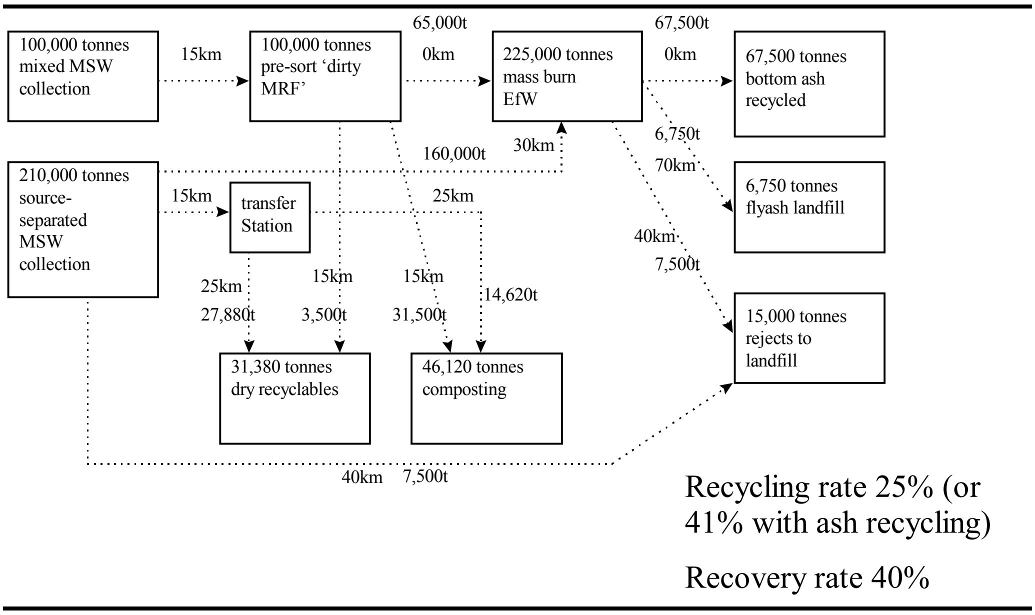

Figure 1. Flow Diagram for Option 1: Mass Burn Energy from Waste (2005)

Step 3 - Assess the Performance of the Options Against the Criteria

The modelled performance of the alternative waste management options against the criteria is presented in Table 1. Performance is shown graphically for the greenhouse effect in Figure 2. The performance matrix is a valuable aid to decision-making in itself. However, direct use of the results it contains is difficult because of the complexity of the matrix and the use of different units. The performance of each option for each criterion is also ranked, with the rank shown in brackets in Table 1. The principal conclusions that can be drawn are discussed briefly in Table 2.

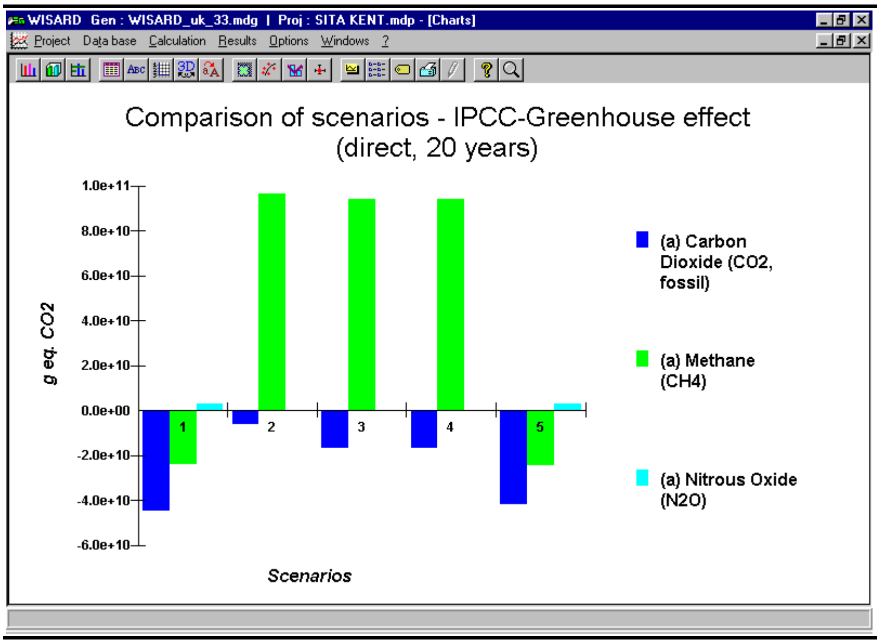

Figure 2. Option Impacts - Greenhouse Effect

Table 1. Performance of Waste Management Options (Rank in Brackets)

Criterion	**Option 1**	**Option 2**	**Option 3**	**Option 4**	**Option 5**
Greenhouse effect ($g\ CO_2$ equivalent)	-6.55 E+10 (1)	9.12E+10 (5)	7.83E+10 (3)	7.85E+10 (4)	-6.35E+10 (2)
Resource depletion (yr^{-1})	-2580000 (2)	-747000 (5)	-1620000 (3=)	-1620000 (3=)	-2660000 (1)
Air acidification ($g\ H^+$ equivalent)	-21000000 (2)	-3630000 (5)	-8950000 (3)	-8880000 (4)	-21300000 (1)
Eutrophication ($g\ PO_4$ equivalent)	-440000 (1)	505000000 (3=)	505000000 (3=)	505000000 (3=)	-376000 (2)
Stratospheric ozone depletion (g CFC-11 equivalent)	-2320 (2)	77900 (5)	74100 (3)	74300 (4)	-2840 (1)
Dioxin emissions (g)	0.0774 (5=)	0.0447 (1)	0.0450 (2=)	0.0450 (2=)	0.0774 (5=)
Financial costs per tonne (£)	72.66 (1)	93.66 (5)	82.20 (3)	85.20 (4)	76.68 (2)
Reliability of delivery (0-10 where 0 is the most reliable, 10 the least)	0 (1)	10 (5)	3 (3)	2 (2)	6 (4)

Table 2. Summary of Option Performance Against Assessment Criteria

Criterion	Performance Summary
Greenhouse Effect	The most significant releases of greenhouse gases are associated with emissions of methane to the atmosphere from wastes landfilled in options 2, 3 and 4. These are, to some extent, offset by the avoided burdens of carbon dioxide as a result of the recovery of materials. However, the most significant avoided burdens for both carbon dioxide and methane are for options 1 and 5 because of the recovery of both energy and materials.
Resource Depletion	The most significant non-renewable resource depletion burdens are for options 2, 3 and 4, principally associated with the depletion of natural gas and coal resources required for energy production because of that consumed in the recycling process. These are not repeated in options 1 and 2 because of the recovery of energy from waste and its substitution for fossil fuel, indeed these options actually show avoided resource depletion for natural gas and coal. All the options show avoided depletion for some resources, particularly copper, iron and bauxite ores, and oil reserves, due to recycling.
Air Acidification	All the options show a net benefit for this impact because of the recovery of energy and materials and their substitution for virgin resources. The most significant avoided burdens for sulphur oxides are for options 1 and 5 due to the significant proportion of the waste arisings which are treated by combustion and the recovery of energy (ie offsetting electricity generation from fossil fuels). These avoided burdens are balanced to a very limited extent by increased nitrogen oxide burdens from waste combustion.
Eutrophication	The most significant environmental burdens for ammonia emissions are for options 2, 3 and 4. This is because of the large proportion of wastes arising which are sent to landfill and the subsequent potential impacts of leachate. The impact of the energy from waste led options is negligible by comparison.
Stratospheric Ozone Depletion	The most significant environmental burdens for CFC-12 and HCFC-22 emissions are for options 2, 3 and 4 due to the larger proportion of waste arisings which are landfilled. The impact of the energy from waste options is, by comparison, relatively negligible.

Criterion	Performance Summary
Dioxin Emissions	The most significant environmental burdens for emissions of dioxins to the atmosphere are for options 1 and 5. The emissions for these two options and approximately twice those of the other three.
Financial Costs	For collection, treatment and disposal, options 1 and 5, employing energy from waste, are the cheapest options. Options 3 and 4 are next most expensive, with higher costs associated with the increased quantities of source separated wastes, but with some slight saving associated with the lower cost of landfill over energy from waste. This differential is likely to be eroded as the landfill tax increases beyond 2005. Option 2 is the most expensive option because of the increased costs of treating the biodegradable waste diverted by anaerobic digestion rather than composting.
Reliability of Delivery	Expert judgement suggests option 1 is the most likely to succeed, with many similar demonstrated plant in the UK, and the lowest reliance on increasing public awareness and participation in source separation. Options 3 and 4 are slightly less likely to succeed because of the need for greater public involvement in separation at source. Nevertheless, these levels have been demonstrated elsewhere in the UK. Option 5 is less reliable, due to the poor record of commissioning successful fluidised bed energy from waste plant. Option 2 is assessed the least reliable because of the failure to commission proposed plant in the UK for municipal wastes.

With the exception of dioxin emissions, option 1 or option 5 perform the best against all of the criteria assessed. Option 1 is ranked first for four of the criteria and second for three, whilst option 5 is ranked first for three criteria, and second for three. On the simple basis of assuming all criteria were equally important, and that the ranks are an acceptable substitute of the actual scores, one would expect option 1 to be the preferred integrated waste management solution, or BPEO.

Value Performance

The 'value' of each performance score can be assessed by converting actual scores into a scale of 0 - 1, where 0 is the worst performance and 1 the best. This simplifies the performance matrix in Table 1, retaining the cardinal nature of the data, whilst allowing performance against all criteria to be placed on a common scale. The valued performance data is presented in Table 3. Note that the units are now 'value' for each criterion.

Table 3. Valued Performance Data (2 decimal places)

Criterion	Option 1	Option 2	Option 3	Option 4	Option 5
Greenhouse effect	1.00	0.00	0.08	0.08	0.99
Resource depletion	0.96	0.00	0.46	0.45	1.00
Air acidification	0.98	0.00	0.30	0.30	1.00
Eutrophication	1.00	0.00	0.00	0.00	1.00
Stratospheric ozone depletion	0.99	0.00	0.05	0.04	1.00
Dioxin emissions	0.00	1.00	0.99	0.99	0.00
Financial costs per tonne	1.00	0.00	0.54	0.39	0.81
Reliaility of delivery	1.00	0.00	0.70	0.80	0.40

The values in Table 3 can be added together to give a total valued performance for each option. The results of this exercise are shown in Table 4. This provides a simple overall measure of how well the options perform against all the objectives, and again demonstrates that option 1, the proposed development, is preferred and indicated as being the BPEO. However, this approach implies the assumption that all the criteria, and the range of performance offered by the choice between options, are of equal significance. In practice, decision-makers are likely to give more weight to some criteria than others.

Table 4. Total Valued Performance (2 decimal places)

	Option 1	Option 2	Option 3	Option 4	Option 5
Total value	6.93	1.00	3.12	3.06	6.19

Step 5 - Balance the Criteria Against One Another

Decision analysis techniques, such as the multi-criteria assessment method suggested in *Waste Strategy 2000*, elicit and apply weights to reflect the relative significance of criteria, rather than assuming all criteria are equal. Appropriate weight sets are not widely published, but we have applied weights used by the Dutch Oil and Gas Exploration Association, derived using a Delphi Panel technique. These weights are shown in the second column of Table 5. Aquatic toxicology and photochemical oxidants are not considered in this assessment, and the weights applied have been scaled to reflect this, as shown in the third column of the table. The Dutch weight set was not derived for BPEO assessment, and does not, therefore, include the financial costs and reliability criteria. It considers environmental impacts only.

The results of applying the weight set in Table 5 to the valued performance in Table 3 are shown in Table 6. The final row in this table is a total weighted performance.

Step 6 - Evaluate and Rank the Options

Table 1 shows the relative performance of the five options across all the environmental criteria, and clearly shows that option 1 and 5 are preferred. All criteria except for dioxin emissions contribute to the margin between the energy from waste options and those led by other technologies. However, the strong weight given to the greenhouse effect is responsible for a significant proportion of the margin on its own.

Financial costs and reliability of delivery are not included in this total weighted performance. We have included these in the analysis by substituting them for the two criteria in the Dutch weight set which have not been considered, aquatic toxicology and photochemical oxidants. We have split the weight given to these criteria (0.208) between financial costs and reliability of delivery, giving the weight set shown in the fourth column of Table 5.

The results of applying this weight set is shown in Table 7. The total weighted performance clearly shows that option 1 is the preferred option or BPEO.

Table 5. Dutch Oil and Gas Exploration Association Weights and Modifications

Criterion (as used in this determination)	**Weight (as published)**	**Weight (scaled) excluding practicability criteria**	**Weight including practicability criteria**
Depletion (resource depletion)	0.125	**0.158**	**0.125**
Human toxicology (dioxin emissions)	0.113	**0.143**	**0.113**
Aquatic toxicology	0.125		
Acidification	0.128	**0.162**	**0.128**
Nutrification (eutrophication)	0.098	**0.124**	**0.098**
Ozone depletion	0.09	**0.114**	**0.09**
Greenhouse warming	0.24	**0.303**	**0.24**
Photochemical oxidants	0.083		
Financial costs			0.104
Reliability of delivery			0.104

Table 6. Weighted Performance Excluding Practicability Criteria (2 decimal places)

Criterion	**Option 1**	**Option 2**	**Option 3**	**Option 4**	**Option 5**
Greenhouse effect	0.30	0.00	0.03	0.02	0.30
Resource depletion	0.15	0.00	0.07	0.07	0.16
Air acidification	0.16	0.00	0.05	0.05	0.16
Eutrophication	0.12	0.00	0.00	0.00	0.12
Stratospheric ozone depletion	0.11	0.00	0.01	0.01	0.11
Dioxin emissions	0.00	0.14	0.14	0.14	0.00
Total weighted performance	**0.85**	**0.14**	**0.29**	**0.29**	**0.86**

Table 7. Weighted Performance Including Practicability Criteria (2 decimal places)

Criterion	**Option 1**	**Option 2**	**Option 3**	**Option 4**	**Option 5**
Greenhouse effect	0.24	0	0.02	0.02	0.24
Resource depletion	0.12	0	0.06	0.06	0.13
Air acidification	0.13	0	0.04	0.04	0.13
Eutrophication	0.10	0	0.00	0.00	0.10
Stratospheric ozone depletion	0.09	0	0.00	0.00	0.09
Dioxin emissions	0	0.11	0.11	0.11	0.00
Financial costs	0.1	0	0.06	0.04	0.08
Reliability of delivery	0.1	0	0.07	0.08	0.04
Total weighted performance	**0.88**	**0.11**	**0.36**	**0.35**	**0.80**

Step 7 - Analyse the Sensitivity of the Results

The results are strongly influenced by the weight set applied. However, the results are very robust, under reasonable variation of the weights, in showing that option 1 and option 2 are preferred over the other options. The weights applied to financial costs and reliability of delivery in Table 7 have not been derived by a Delphi panel, but are low compared with the other weights applied to environmental criteria. If anything, we believe these weights should be raised, reflecting the importance of the opportunity costs of waste management revenue costs and the imperative of reliable waste management arrangements. Should the weights for these two criteria be raised, option 1 becomes progressively more preferred as the BPEO.

Material Flow Analysis as a Decision Support Tool for Goal Oriented Waste Management

***Paul H. Brunner*[4]**

Abstract

The paper presents a new approach to evaluate and optimize goal oriented waste management. It is based on extensive experience with Material Flow Analysis (MFA) as a rigid mass balance method to analyse the flow of goods and substances through any given system. Combined with methods to assess the impact of material flows, MFA has shown to be a powerful method

- to early recognize problems and challenges in waste management,
- to evaluate and improve systems for the management of wastes and materials
- and to develop new concepts and strategies for the design of goods, processes and systems.

In the presentation, the application of MFA to assess, evaluate, optimize and design waste management systems on all levels (individual households, regional waste management, national materials management, waste derived nutrient management in the entire Danube basin) is discussed. Emphasis is laid on the linkage between the goals of waste management on one hand and the method and criteria for assessment of waste management systems on the other hand. In addition, new entropy based methods are presented for evaluation of MFA results.

The paper is based on more than 60 MFA-projects which have been carried out during the last ten years. Some of these projects are presented as case studies (assessment of plastic packaging waste management; regional evaluation of thermal and mechanical-biological waste management; alternative methods for cost effective routine waste analysis).

Key words: material flow analysis, analysis and control of goal oriented waste management, design of concepts and strategies for waste management.

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[4] Professor and Head, Institute for Water Quality and Wastes Management, Vienna University of Technology, Karlsplatz 13/226.4, A-1040 Vienna, Austria, Email: paul.h.brunner@awsnt.tuwien.ac.at, Phone: ++43 1 58801 226 40, Fax:++43 1 504 22 34

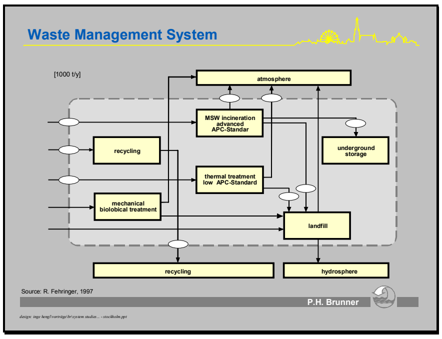
Methodology: MFA as a tool to assess waste management systems

1.1. Define Goals

The goals are *not* prevention and recycling!

1.2. Operationalise goals to a concrete level: E.g. Critical air volume, fuel equivalents, emission standards etc.

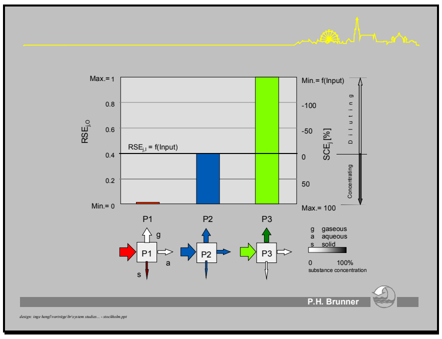
1.3. Define waste management system



MFA defines a system in a rigid, transparent way.

1.4. Assess flows and stocks of goods and substances

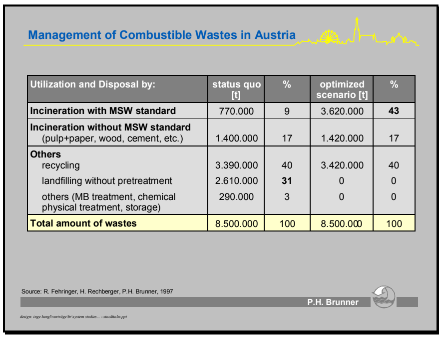
1.5. Evaluate flows and stocks in view of the goals: How to deal with those phenomena, which are not regulated yet (entropy)?



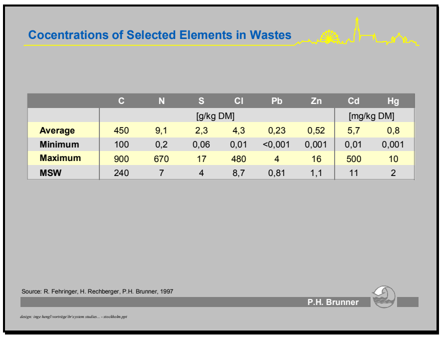
Substance concentration efficiency as a measure for waste treatment

Case studies

1.6. Case study 1: ASTRA - How to treat combustible wastes in Austria?

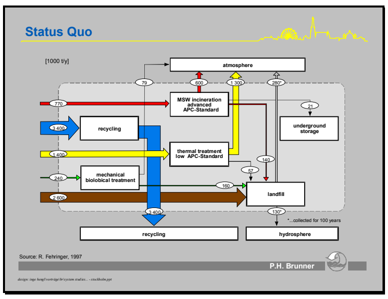


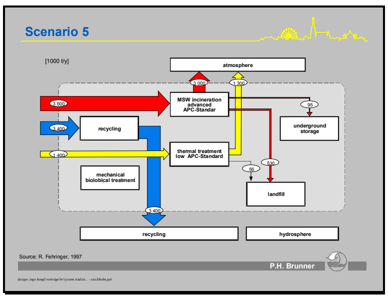
Utilization and Disposal by:	status quo [t]	%	optimized scenario [t]	%
Incineration with MSW standard	770.000	9	3.620.000	**43**
Incineration without MSW standard (pulp+paper, wood, cement, etc.)	1.400.000	17	1.420.000	17
Others				
recycling	3.390.000	40	3.420.000	40
landfilling without pretreatment	2.610.000	**31**	0	0
others (MB treatment, chemical physical treatment, storage)	290.000	3	0	0
Total amount of wastes	8.500.000	100	8.500.000	100

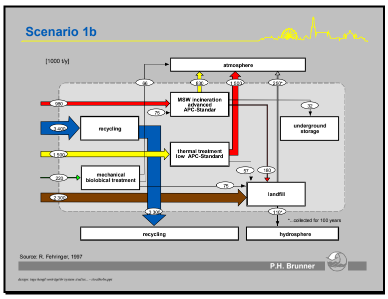


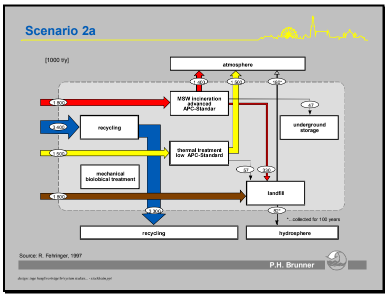
	C	N	S	Cl	Pb	Zn	Cd	Hg
	[g/kg DM]						[mg/kg DM]	
Average	450	9,1	2,3	4,3	0,23	0,52	5,7	0,8
Minimum	100	0,2	0,06	0,01	<0,001	0,001	0,01	0,001
Maximum	900	670	17	480	4	16	500	10
MSW	240	7	4	8,7	0,81	1,1	11	2

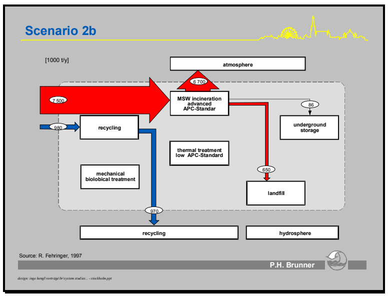
1.6.1. Define the systems and scenarios

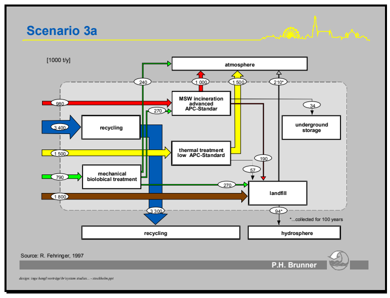


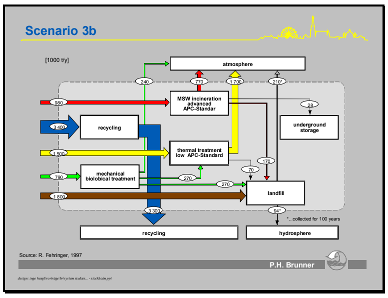


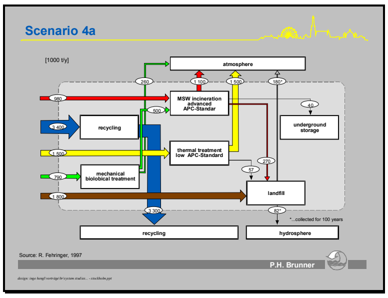


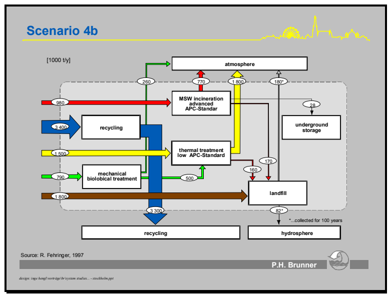




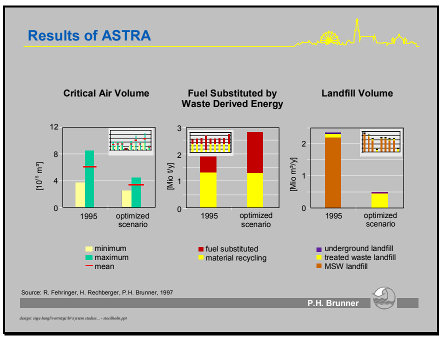


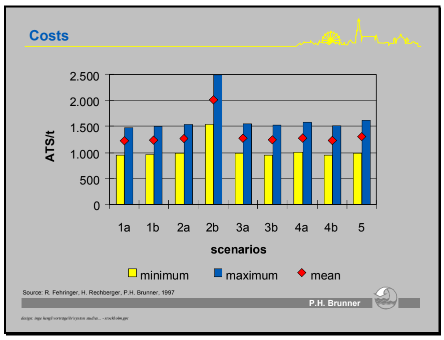






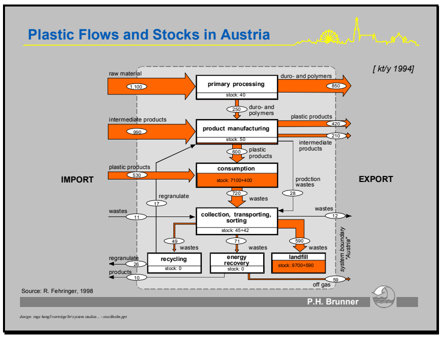
1.6.2. Results

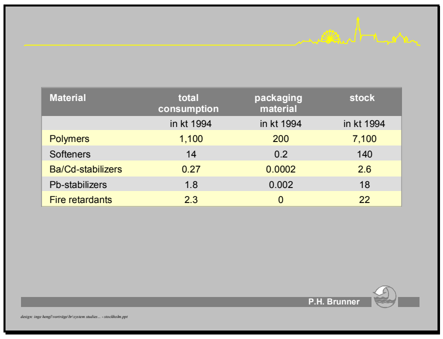




1.7. Case study 2: Plastic waste management – is the packaging ordinance effective in the management of plastic wastes?

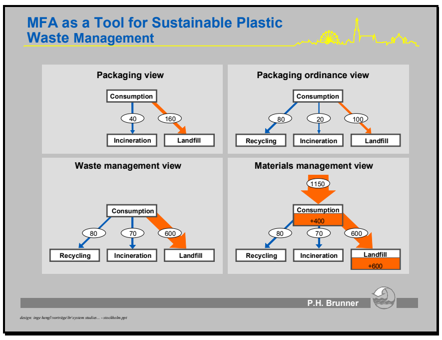
1.7.1. Flows of plastics and additives in Austria





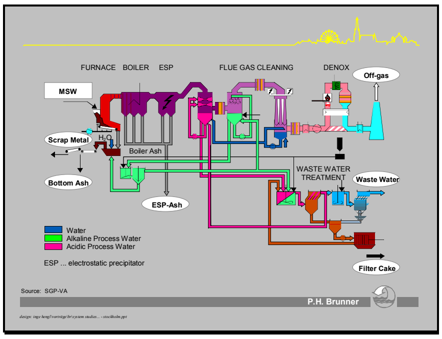
Material	total consumption	packaging material	stock
	in kt 1994	in kt 1994	in kt 1994
Polymers	1,100	200	7,100
Softeners	14	0.2	140
Ba/Cd-stabilizers	0.27	0.0002	2.6
Pb-stabilizers	1.8	0.002	18
Fire retardants	2.3	0	22

1.7.2. Plastic wastes represent both resources and hazardous materials. The management has to take both properties into account. MFA can quantify both and allows better waste management decisions through more comprehensive view of the total picture. The present packaging ordinance controls a minor amount of plastic wastes at high costs, and does not take hazardous materials into account. The major plastic waste problem is the large amount still going to landfills, and the toxic materials contained in long living plastic goods.



1.8. Case study 3: Monitoring waste composition

1.8.1. Analysing the products of MSW incineration allows to determine waste composition routinely, cost effectively and with comparatively high accuracy. MFA serves to measure the transfer coefficients of selected elements during incineration, and to determine waste composition continuously from a few inexpensive measurements.



Conclusions

If MFA is performed in a rigid, transparent and reproducible way, and if multiple evaluation criteria are applied to the MFA results, it serves well to:

- evaluate and improve waste management systems
- set priorities in waste management
- early recognize new challenges such as resource depletions or accumulations of hazardous materials
- design new systems, processes and goods for wastes *and* materials management.

CHAMP – a new approach to modelling material recovery, re-use, recycling and reverse logistics

Roland Clift[*a], Warren Mellor[a,b,c], Elizabeth Williams[a,b,c], Adisa Azapagic[b] and Gary Stevens[c]. [5]

Abstract

CHAMP – CHAin Management of Products – is a modelling approach which describes the successive use, re-use and reprocessing of materials and products, with the associated logistics, as they pass through a succession of uses in an Industrial Ecology. Technical performance characteristics, termed "utilities", are tracked as the basis for selection criteria which determine whether a recovered product or material is suitable for a particular application or whether it must be "cascaded down" to an application with lower specifications. Environmental impacts are calculated on a Life Cycle basis. CHAMP is developed to support decisions over material selection, recovery, re-use and recycling.

Key words: Industrial Ecology; material selection; Life Cycle Management

Introduction

Most system approaches to integrated waste management are concerned with treatment of a material stream accepted as waste. The approach outlined in this contribution takes a different approach, concentrating on single materials or specific products to compare alternative uses of the waste. It considers a material which can pass through several different uses, and enables materials and processing routes to be selected or designed for improved recyclability. The modelling approach – CHAMP : CHAin Management of Products – applies the ideas of Life Cycle Assessment (LCA) but goes further in describing possible different uses as a material passes through an Industrial Ecology. It combines Life Cycle Product Design (LCPD) with analysis of waste management systems.

[5] a. Centre for Environmental Strategy;
b. Department of Chemical and Process Engineering;
c. Polymer Research Centre;University of Surrey, Guildford, Surrey GU2 7XH, UK
* Corresponding author: e.mail: r.clift@surrey.ac.uk

The following account refers specifically to polymers or plastics, the class of materials for which CHAMP has so far been developed and applied [1]. The products considered all have short service lives, so that a steady-state system model can be used. The modelling approach is currently being extended to metals, where the possible uses have very different service lives; for example aluminium can be used in beverage containers (service life: a fraction of a year) or in automobile components (several years) or in buildings (several decades). These cases need a dynamic model allowing for stocks of material in use [2].

Industrial Ecology for Polymers

The full possible scope of an Industrial Ecology for polymers is shown in Figure 1. From primary resource (normally hydrocarbon reserves) the material passes through the steps of Extraction and Processing to produce a monomer; through Polymerisation to produce the basic material; through Blending and Forming, in which additives such as plasticisers, stabilisers, fillers and pigments are added, and the material is processed to produce an artefact, for example by extrusion or moulding; to make the product which is then put to use. Conventional LCA considers this sequence of operations to calculate the inventory and assess the environmental impacts associated with the product. System approaches to waste management consider ways of treating the product after use, either as a distinct waste or commingled with other wastes. CHAMP has been developed to include other possible subsequent uses.

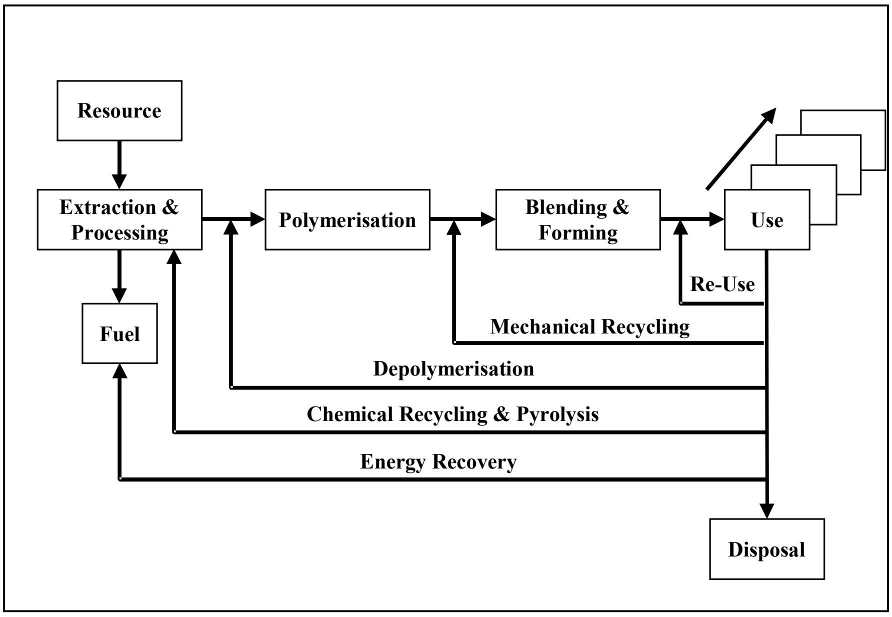


Figure 1. Industrial ecology for polymers [3].

The product might be re-used directly in the same use : *re-use* in Figure 1. To take the specific example of a container, it might be recovered and refilled. If the artefact has been damaged, re-use may not be an option. It might then be *recycled mechanically* (i.e. with no chemical treatment), for example by granulating the polymer and forming it into a repeat of the original artefact or into a different product. For example, post-consumer Low Density PolyEthylene (LDPE) first used as packaging may be recycled mechanically into larger items such as garden furniture. However, the demand for these products does not match the supply of waste, so that other recycling processes must be used. Similarly, if the material is too degraded or contaminated, mechanical recycling cannot be used. Some polymers, notably acrylics and some polyurethanes, can be *depolymerised* back to the monomer, and then formed back into the same polymer or a different copolymer [3-6]; this route usually requires the polymer to be kept uncontaminated. For other polymers or for mixed plastics, the next option is *chemical recycling or pyrolysis* which treats the material to produce a mixed feedstock which can be returned for processing, for example as a cracker feedstock to a refinery or petrochemical complex. Technologies for this recovery route have yet to be commercialised; they are likely to be capital-intensive, and therefore to require a large plant located adjacent to a refinery. The remaining option is *energy recovery* : using the polymer as a fuel, to displace direct use of hydrocarbons as fuels.

CHAMP is a framework for modelling the range of possible approaches to recovery, re-use and recycling, including transport as well as reprocessing or refilling. In general, successive "loops" in Chapter 1, from Re-use to Energy Recovery, involve progressively increasing environmental impact and economic cost. CHAMP has been developed to support decisions on the selection and processing of polymers to find the optimal route of materials through the Industrial Ecology.

Modelling Approach

Systems like that in Figure 1, with multiple recycle loops, are common in chemical process systems. The approach adopted in CHAMP derives from that commonly used in process system modelling [7]. The properties of each material or product are "tracked" as it passes through the sequence of operations or activities making up the Industrial Ecology system. Each activity is therefore represented by a model which quantifies its effect on the material properties, and which also evaluates the economic costs and the environmental burdens and impacts associated with the activity. Environmental effects are evaluated on a life cycle basis by considering the full supply chains of energy and ancillary materials as well as the burdens arising directly from each activity. To provide information to support decisions over selection of materials, applications, processes or options for recovery or disposal, multi-objective optimisation is used to generate the set of costs and impacts characterising each alternative, rather than reducing them to a single metric. This represents an application of an approach developed to apply LCA to process selection, design and operation [e.g. 8-11]. The CHAMP model has been set up so that it can automatically present alternative materials for any application or alternative applications for any material by generating possible routes through the Industrial Ecology.

Figure 2 shows the overall structure of the CHAMP model. The essential features of each of the main components will now be outlined.

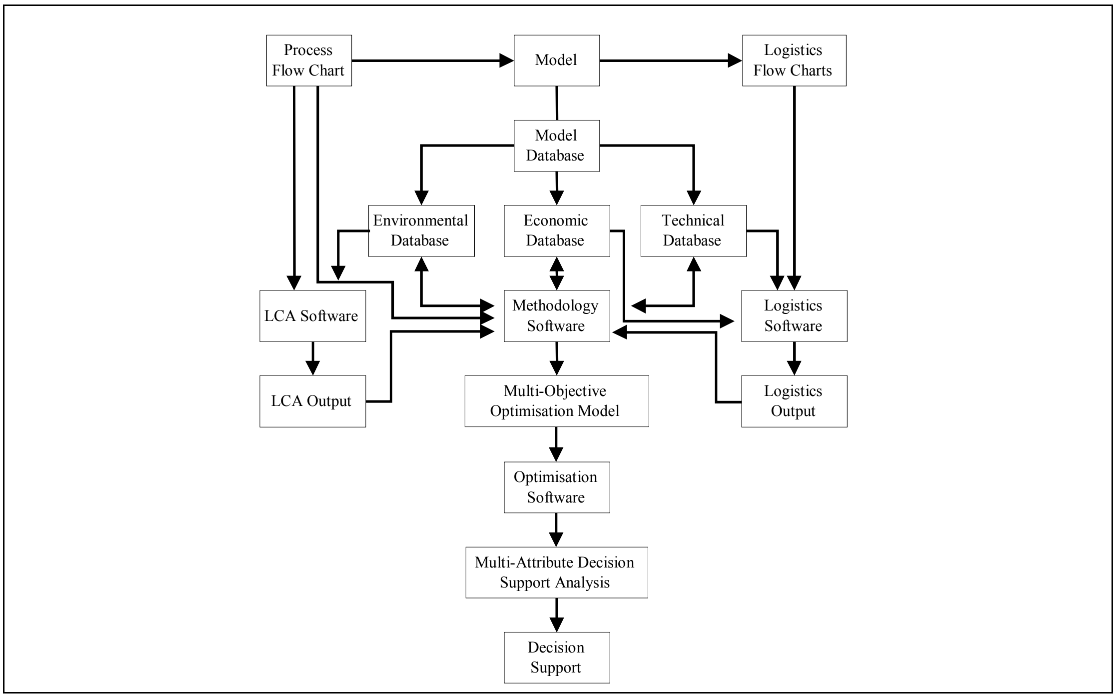


Figure 2. Overall CHAMP methodology

Material Properties

The CHAMP model is set up to describe *materials* (usually single polymers, with or without additives), *components* (which may be formed from more than one material) and *products* (i.e. assemblies of different materials and components). Assembly and disassembly of products and components can be included as part of the system being modelled, so that CHAMP can be applied to chain management of products as well as materials. As in conventional LCA, the model results are expressed per functional unit defined at some critical point in the system, such as first use or waste entering one of the outer loops in Figure 1.

A material at any point in the system is described by a set of technical characteristics which determine whether it can be used for any particular process or application. These characteristics are termed *utilities.* The utilities include material properties such as impact strength, tensile strength and hardness. The utility set also includes geographical location. This feature enables the same modelling framework to be used for processing, use and also for transport, both distribution and collection (i.e. "reverse logistics"). Each activity in the system is modelled by describing how the utility parameters are changed when a material or product passes through the activity. Mathematical details of the approach to modelling utility transformations are given elsewhere [1,7]. An individual activity changes only some of the parameters making up the utility set; thus a processing operation changes material properties but not location, while transport changes location but not other properties. Therefore although the CHAMP methodology describes both processing and logistics, it is convenient to separate the two kinds of activity within the model as shown in Figure 2.

Economic Costs and Environmental Impacts

The CHAMP model also calculates the costs associated with each of the activities making up the Industrial Ecology system, and accumulates them to give the total cost associated with any material, component or product at any point in the system.

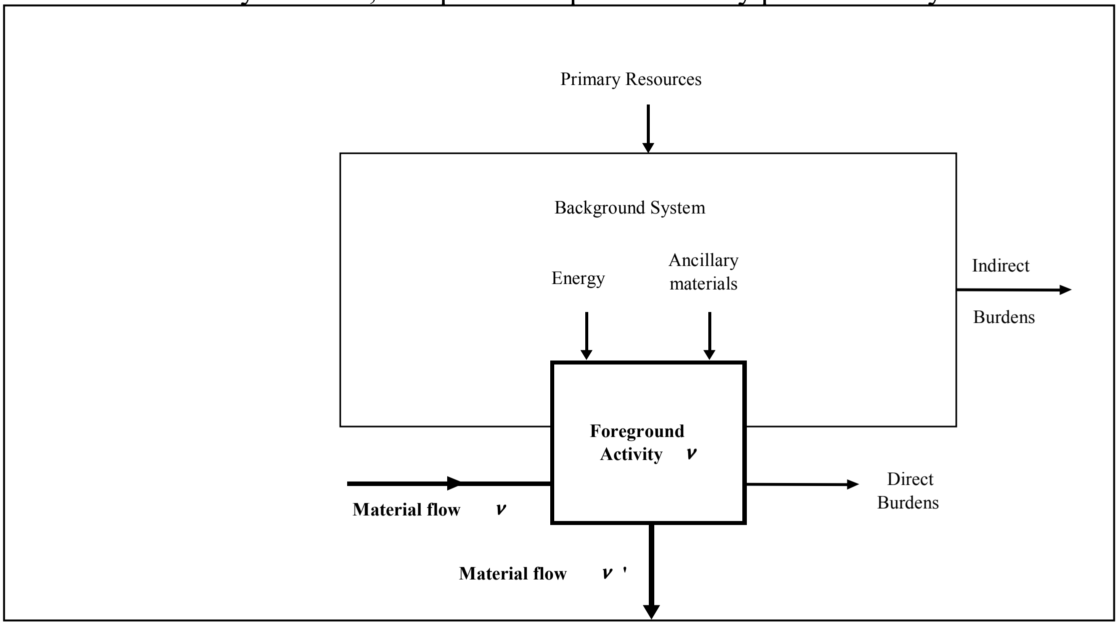


Figure 3. Extended system approach for environmental burdens and impacts

To complete the set of parameters characterising a material as it passes through the system, the environmental burdens associated with each activity are calculated and accumulated along the sequence of activities through which a material passes. The full set of burdens is then expressed in terms of their contributions to a set of environmental impact categories as in conventional Life Cycle Impact Assessment [e.g.12]. To ensure that the full life cycle impacts are included, an extended system approach [e.g.13,14] is taken as shown schematically in Figure 3. The burdens include not only direct emissions from the processing or transport activity itself but also the burdens from the background system supplying energy and ancillary materials to the foreground activity. The same extended system approach is used to assess the burdens avoided by recovering energy or materials from use in the background system [14]. In general, the environmental burdens are related to the utility change effected by the activity – for example distance transported or intensity of processing. The model of each activity must incorporate this relationship.

These elements are combined in the *methodology software* as indicated in Figure 2.

Multi-objective Optimisation

The CHAMP approach treats selection of materials, activities and applications as a multiple criteria decision problem, where the decision-maker selects between options on the basis of trading-off different performance objectives. It is an example of a *generating method*: it presents a set of options each of which is optimal in the Pareto sense; i.e. it is impossible to improve any one objective without worsening at least one other objective [14]. The environmental impacts and economic costs constitute the performance objectives, so that the trade-offs are between costs and different impacts. Specific activities, including transport, can be optimised individually within the overall multi-objective optimisation. Transport operations in particular are optimised using commercial route-scheduling logistics packages.

Selecting Materials and Activities

The CHAMP methodology includes a way of selecting materials which meet criteria defining acceptability for a use or a process and for routing a material to other possible activities if it does not meet the acceptance criteria. This feature is essential to enable the model to construct and explore possible industrial ecologies. The model describing any activity can include an *acceptance gate* which checks the properties of a material or product to ensure that it can pass through the activity; acceptance usually constitutes ensuring that the technical properties lie within specified ranges, but economic and environmental performance can also be subject to performance checks. The same approach lends itself to logistics activities, for example to ensure that a material or product can be collected by a particular vehicle on a specified overall route. Where a material is not acceptable, the model can automatically generate other options, for example routing the material to another process or application or blending with another material to bring the properties within the acceptable range.

Applications

The CHAMP methodology has been developed in collaboration with a range of industrial companies spanning the entire supply chain from polymer production, through several different applications, to material recovery and management. It has already been applied to a number of real applications : material selection for polymer interlayers of laminated glass windscreens and for the jackets of telecommunications cables; design of optical fibre cables; recycle or re-use of panels from used office equipment; optimal transport scheduling for delivery, collection and re-use of containers; and recovery of municipal and commercial waste [1]. All the partners report practical benefits; they include rethinking logistics operations and redesigning a product completely.

It was noted in the Introduction that CHAMP uses a steady-state model. However, some of the studies have highlighted the importance of developing a recyclate "pool" to store material pending re-use in the original or another application. The CHAMP approach is now being extended to metals, where a dynamic model is needed to account for "stocks" of material in use.

Acknowledgements

CHAMP has been developed as a LINK project with support from the Engineering and Physical Sciences Research Council and the Department of Trade and Industry in the UK. We would also like to thank our industrial partners for their involvement: Biffa, Brand-Rex, Corning Cables, European Vinyls Corporation, Mann, Pilkington and Xerox.

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Session 1: Summary of discussions

Summarised by Göran Svensson and Jessica Granath. Edited by Jan-Olov Sundqvist.

In this session three different model approaches were presented. Several issues were discussed and compared.

1. Implementation of the models
- Stakeholders have tested all three models.
- There is a gap between research and implementation.
- The use of models is often late in the decision phase.
- There is often conflict between different waste practitioners.
- There are large differences between countries how the models are implemented.

2. Methods and data
– Scenario choices: who defines the scenarios.
– Often simplifications are needed to make the models easier to understand. But there are risks connected with simplifications. It is important to not simplify too far.
– Final sinks were emphasised by Professor Brunner. Not only flows of heavy metals should be assessed, but also concentration and dilution processes.
- Data considering material balances are important. There is a general demand on data for transfer coefficients. The Vienna study was based on thoroughly measurements. It was also pointed out that transfer coefficients for cement kilns are very uncertain.
- Energy/electricity assumptions often play a large role for the result:
  . to use average or marginal source for electricity, district heating and other energy use for waste.
  . identification of the appropriate marginal source.
- Transparency is important.

3. Others
- Professor Brunner told that 14-15 % of the Hg input to Austria is found in Vienna's combustible waste.
- Some discussions were about how to encourage landfill operators to decrease landfilling. Today they have only restrictions against them, and they have in practice no incitements to avoid landfilling. Perhaps a system where they get money for avoiding landfilling could encourage them.

Session 2.

Chairman: Simon Aumônier; Secretary: Marcus Carlsson Reich

Göran Finnveden
Treatment of solid waste – what makes a difference?

Ola Ericsson
Energy recovery and material and nutrient recycling from a system perspective

Jürgen Giegrich
Reconsidering the German Dual System for Lightweight Packaging

Karl Vrancken
Evaluation of waste treatment processes for MSW rest fraction

Discussion

Life Cycle Assessment of Energy from Solid Waste – Total energy use and emissions of greenhouse gases

Göran Finnveden, *Jessica Johansson, Per Lind and Åsa Moberg*[6]

Abstract

The overall aim of the present study is to evaluate different strategies for treatment of solid waste based on a life-cycle perspective in Sweden. Important goals are to identify advantages and disadvantages of different methods for treatment of solid waste, and to identify critical factors in the systems, including the background systems, which may significantly influence the results. Included in the study are landfilling, incineration, recycling, digestion and composting. The waste fractions considered are the combustible and recyclable or compostable fractions of municipal solid waste. The methodology used is Life Cycle Assessment. The results can be used for policy decisions as well as strategic decisions on waste management systems.

Introduction

We live in a changing world. In many countries both energy systems and waste management systems are under change. The changes are largely driven by environmental considerations and one driving force is the threat of global climate change. When making new strategic decisions related to energy and waste management systems it is therefore of importance to consider the environmental implications.

A waste hierarchy is often suggested and used in waste policy making. Different versions of the hierarchy exist but in most cases it suggests the following order:

1. Reduce the amount of waste
2. Reuse
3. Recycle materials
4. Incinerate with heat recovery
5. Landfill

[6] Environmental Strategies Research Group (fms), Swedish Defence Research Agency and Department of Systems Ecology at Stockholm University, PO Box 2142, 103 14 Stockholm, Sweden

The first priority, to reduce the amount of waste, is in general accepted. However, the remaining waste needs to be taken care of as efficiently as possible. Different options for taking care of the remaining waste is the topic of this study. The hierarchy after the top priority is often contested and discussions on waste policy are in many countries intense. Especially the order between recycling and incineration is often discussed. Another question is where to place biological treatments such as anaerobic digestion and composting in the hierarchy. One of the aims of this study is to evaluate the waste hierarchy.

The overall aim of the study is to evaluate different strategies for treatment of solid waste based on a life-cycle perspective. Important goals are to identify advantages and disadvantages of different methods for treatment of solid waste, and to identify critical factors in the systems, including the background systems, which may significantly influence the results. Included in the study are landfilling, incineration, recycling, digestion and composting. The waste fractions considered are the combustible and recyclable or compostable fractions of municipal solid waste and for these fractions, the total amount of waste produced in Sweden during one year is considered. The study is presented in detail in a larger report (Finnveden et al. 2000) and other results are presented in another paper in these proceedings (Moberg et al. 2001).

Methodology

The methodology used is Life Cycle Assessment. An LCA studies the environmental aspects of a product or a service (in this case waste management) from "cradle to grave (i.e. from raw material acquisition through production, use and disposal). The methodology used is as far as possible based on established methods and practices for both the inventory analysis and the characterisation element of the life cycle impact assessment as described in for example (Lindfors et al. 1995; ISO 1997; ISO 1998; ISO 1999; Udo de Haes et al. 1999a and b) and for waste treatment processes in (Finnveden 1999; Clift et al. 2000). In the results presented here, emissions contributing to climate change are aggregated using Global Warming Potentials with a time perspective of 100 years (Albritton et al. 1996). In the weighting step two methods are used, a further development of Ecotax 98 (Johansson 1999) and Eco-indicator 99 (Goedkoop and Spriensma 1999). In the version of Ecotax 98 that is used here, several sets of weighting factors are developed using different characterisation methods and weighting factors. Here are only results for one of these sets presented. The focus is on overall energy use and emissions of gases contributing to global warming, but other impact categories such as acidification, eutrophication, photo-oxidant formation, human and ecotoxicological impacts are also included. In the study a base scenario is defined. In several alternative scenarios different assumptions are tested by changing them.

All waste treatment processes considered in this study produce some useful products: materials, fertilisers, fuels, heat or electricity, which can replace the same product produced in another way. This is taken into account in the studied systems. The environmental aspects of different waste treatment methods are therefore not only determined by the properties of the treatment method itself, but also by the environmental properties of the product that can be replaced and the environmental impacts associated with its life cycle.

Results and conclusions

In Figures 1-3, the results from several scenarios are shown compared to the base scenario for total energy use, emissions of greenhouse gases and weighted results using one of several sets of weighting factors described in (Finnveden et al. 2000). In the scenario "medium transports" the transport distances are increased compared to the base scenario. Passenger cars are also assumed to be used in one scenario, for recycling and incineration. In the scenario "long transports", transportation distances are further increased compared to the "medium transports" scenario. In the scenario "natural gas", it is assumed that the heat from incineration of waste and gas from digestion and landfilling replaces heat from incineration of natural gas. This is a change from the base scenario where it is assumed that the competing heat source is forest residues. In the scenario "saved wood used as fuel" it is assumed that the wood that is "saved" by recycling of paper materials is used as a fuel for heat production replacing natural gas. Such a scenario can correspond to a situation where there is increased competition for biomass. In the base scenario, emissions from landfills are considered for a hypothetical infinite time period. In the scenario "Short time perspective for landfills" this is changed. Also in the scenario "landfills as carbon sinks" only a short time perspective is considered and landfills are modelled as carbon traps for nondegraded biological materials. In the scenario, "Plastics replace impregnated wood" it is assumed that recycled plastics replace impregnated wood as palisades. This is a change from the base scenario where it is assumed that recycled paper and plastic materials replace the same materials produced from virgin raw materials.

To summarise some of the overall conclusions it can be noted that recycling of paper and plastic materials are in general favourable according to our study with regard to overall energy use, emissions of greenhouse gases and the total weighted results. These results are fairly robust. When looking at total energy use and emissions of greenhouse gases, recycling is the preferred strategy in all scenarios for the whole system, i.e. when all the studied waste fractions are included.

One exception to the general results is plastics when they are recycled and replace impregnated wood. In this case recycling of plastics is less favourable than incineration with respect to energy use and emissions of greenhouse gases although the difference is rather small. However, recycling may still be favourable with respect to toxicological impacts, and our results still show a benefit for recycling with regard to the total weighted results.

Incineration is in general favourable over landfilling according to our study with regard to overall energy use, emissions of gases contributing to global warming and the total weighted results. There are however some aspects which may influence this ranking. If longer transportation distances are demanded in the incineration case, especially by passenger cars, landfilling can become more favourable than incineration. The modelling of landfills can also have a decisive influence. If shorter time periods are used, in the order of a century, landfilling is favoured and may become a preferable option over incineration. This is further discussed in (Moberg et al. 2001).

LCAs can be used to test the waste hierarchy and identify situations where the hierarchy is not valid. Our results suggest however that the waste hierarchy is valid as a rule of thumb. The results presented here can be used as a basis for policy decisions as well as strategic decisions on waste management systems.

Acknowledgements

Financial support from the Swedish National Energy Administration is gratefully acknowledged. The full study (Finnveden et al. 2000) is available on www.fms.ecology.su.se.

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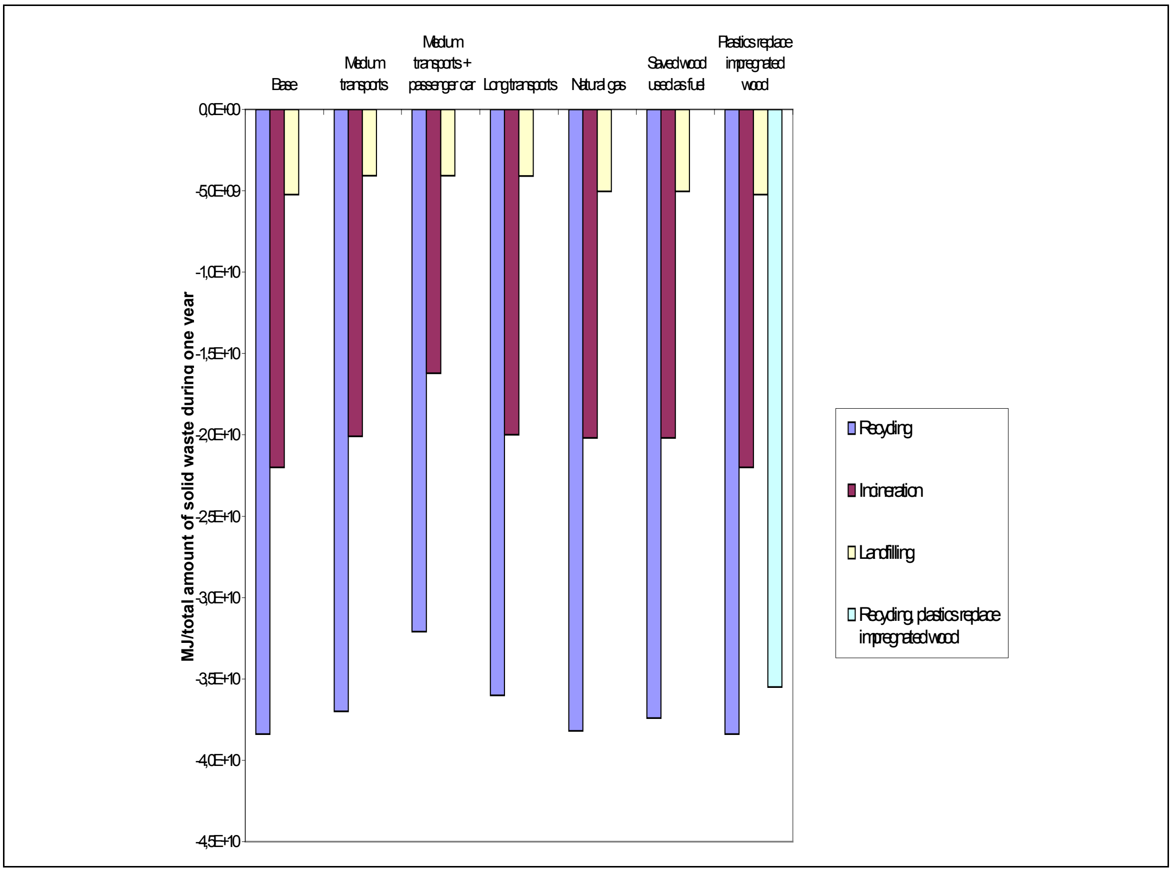


Figure 1. The total energy use for the whole system in a number of scenarios

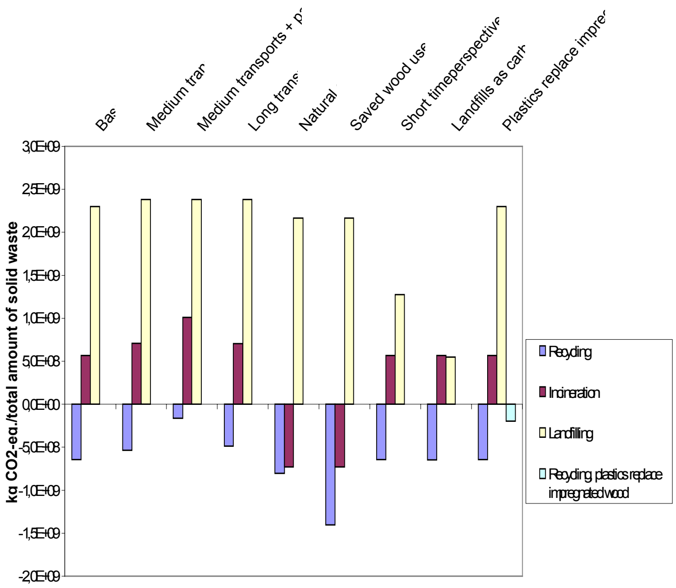


Figure 2. Contribution to global warming from the whole system in different scenarios.

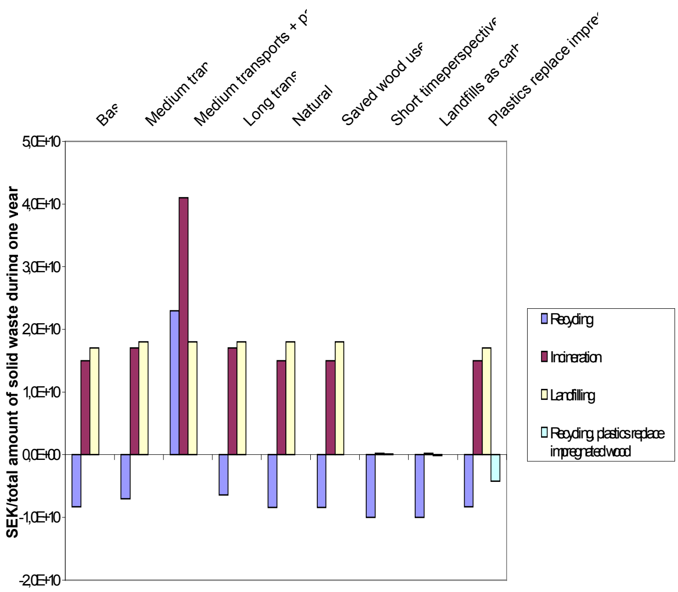


Figure 3. Total weighted results for the whole system using the Ecotax98/USESmax weighting method described in (Finnveden, Johansson et al. 2000). The results are presented per total amount of solid waste produced during a year in Sweden

Energy Recovery and Material and Nutrient Recycling from a Systems Perspective

Ola Eriksson[7], *B. Frostell*[8], *A. Björklund*[9], *G. Assefa*[10], *J. -O. Sundqvist*[11], *J. Granath*[12], *M. Carlsson Reich*[13], *A. Baky*[14], *L. Thyselius* [15]

Abstract

Consequences for energy turnover, environmental impact and economy of different management systems for municipal solid waste have been studied in a systems analysis. In the systems analysis, different combinations of incineration, materials recycling of separated plastic and cardboard containers and biological treatment (anaerobic digestion and composting) of easily degradable organic waste, were studied and also compared to landfilling. In the study a computer model (ORWARE) based on LCA methodology was used. Case studies were performed for three different municipalities: Uppsala, Stockholm, and Älvdalen. The following parameters were used for evaluating the different waste management options: consumption of energy resources, global warming potential, acidification, eutrophication, photooxidant formation, heavy metal flows, financial economy and welfare economy, where welfare economy is the sum of financial economy and environmental economy.

The study shows that reduced landfilling to the benefit of an increased use of energy and material from waste is positive from an environmental and energy as well as economic aspect. This is mainly due to the fact that the choice of waste management method affects processes outside the waste management

[7] Department of Industrial Ecology, Royal Institute of Technology (KTH), S-100 44 Stockholm, Sweden, E-mail: Olae@ima.kth.se

[8] Department of Industrial Ecology, Royal Institute of Technology (KTH), S-100 44 Stockholm, Sweden

[9] Department of Industrial Ecology, Royal Institute of Technology (KTH), S-100 44 Stockholm, Sweden

[10] Department of Industrial Ecology, Royal Institute of Technology (KTH), S-100 44 Stockholm, Sweden

[11] Swedish Environmental Research Institute (IVL), P.O. Box 21060, S-100 31 Stockholm, Sweden

[12] Swedish Environmental Research Institute (IVL), P.O. Box 21060, S-100 31 Stockholm, Sweden

[13] Swedish Environmental Research Institute (IVL), P.O. Box 21060, S-100 31 Stockholm, Sweden

[14] Swedish Institute of Agricultural and Environmental Engineering (JTI), P.O. Box 7033, S-750 07 Uppsala, Sweden

[15] Swedish Institute of Agricultural and Environmental Engineering (JTI), P.O. Box 7033, S-750 07 Uppsala, Sweden

system, such as production of district heating, electricity, vehicle fuel, plastic, cardboard, and fertiliser. This means that landfilling of energy-rich waste should be avoided as far as possible, both because of the environmental impact, and because of the low recovery of resources.

Incineration should constitute a basis in the waste management systems of the three municipalities studied, even if the waste has to be transported to a regional facility. Once the waste is collected, longer regional transports are of little significance, as long as the transports are carried out in an efficient manner. Comparing materials recycling and incineration, and biological treatment and incineration, no unambiguous conclusions can be drawn. There are benefits and drawbacks associated with all these waste management options.

Materials' recycling of plastic containers is comparable to incineration from a welfare economic aspect, but gives less environmental impact and lower energy use – on condition that the recycled plastic replaces virgin plastic. Materials' recycling of cardboard containers is comparable to incineration concerning welfare economy and energy, but has both environmental advantages and disadvantages. Anaerobic digestion of easily degradable waste gives a higher welfare economic cost than incineration, and has both environmental advantages and disadvantages. Conclusions regarding energy use depends upon how the biogas is used. Composting of easily degradable waste is comparable to anaerobic digestion from a welfare economic aspect, but gives higher energy use and environmental impact.

Introduction

Waste management in Sweden is rapidly changing. Due to political decisions more actions are taken towards more sustainable solutions to the waste problem. Producer's responsibility on i.e. paper, containers and tires has been introduced during the late 90's. From 2000 there is a tax on all waste to be landfilled. From 2002 all combustible waste should be sorted out and at the same time landfilling of combustible waste is prohibited. Three years later, 2005, there is a ban on landfilling of organic waste. On the European level new directives on landfilling (decided in 1999) and incineration of waste are introduced. All these actions will cause changes in the waste management now and in the future. As an example it could be mentioned that at the moment, Sweden has 22 incineration plants and about 20 more are now planned for the country around. A turn from landfilling into more incineration and different kinds of recycling (recovery of materials and nutrients) is to be awaited for.

Also, the energy system is in the position of many changes. One nuclear power reactor has been closed down and the governments aim is to close more reactors as renewable sources are introduced into the market. The use of fossil fuels is supposed to decline, which demands for other energy sources of which waste is one.

Using the energy in the waste can be done by incineration or by avoiding virgin production of materials or nutrients by recycling of different waste fractions. That means that the treatment capacity for incineration *and* biological treatment as well as material recycling has to increase in order to meet the new restrictions.

Objectives

The aim with the research project has been to “for a couple of municipalities - from a systems perspective - study how energy in waste is utilised at the best with respect to environment and economy”.

This means that in this study the consequences for three municipalities with respect to consumption of energy resources, different potential environmental effects and financial and environmental costs have been quantified by using systems analysis and mathematical modelling of waste management.

Method

Different solutions to waste management have been simulated with a computer-based model called ORWARE. The framework of the model has been developed during the past seven years in different research projects and describes the method used in this study.

ORWARE is a model for calculation of substance flows, environmental impacts, and costs of waste management. It was first developed for systems analysis of organic waste management, hence the acronym ORWARE (ORganic WAste Research), but now covers inorganic fractions in municipal waste as well.

ORWARE consists of a number of separate submodels, which may be combined to design a waste management system. Each submodel describes a process in a real waste management system, e.g. waste collection, waste transport, or a waste treatment facility (e.g. incineration).

Methods and general description of the model

All submodels in ORWARE calculate the turnover of materials, energy and financial resources in the process. Processes within the waste management system are e.g. waste collection, anaerobic digestion or landfill disposal. Materials turnover is characterised by (1) the supply of waste materials and process chemicals, (2) the output of products and secondary wastes, and (3) emissions to air, water and soil. Energy turnover is the use of different energy carriers such as electricity, coal, oil or heat, and recovery of e.g. heat, electricity, hydrogen, or biogas. The financial turnover is defined as costs and revenues of individual processes.

A number of submodels may be combined to a complete waste management system in any city or municipality (or other system boundary). Such a conceptual ORWARE model of a complete waste management system is shown in Figure 1.

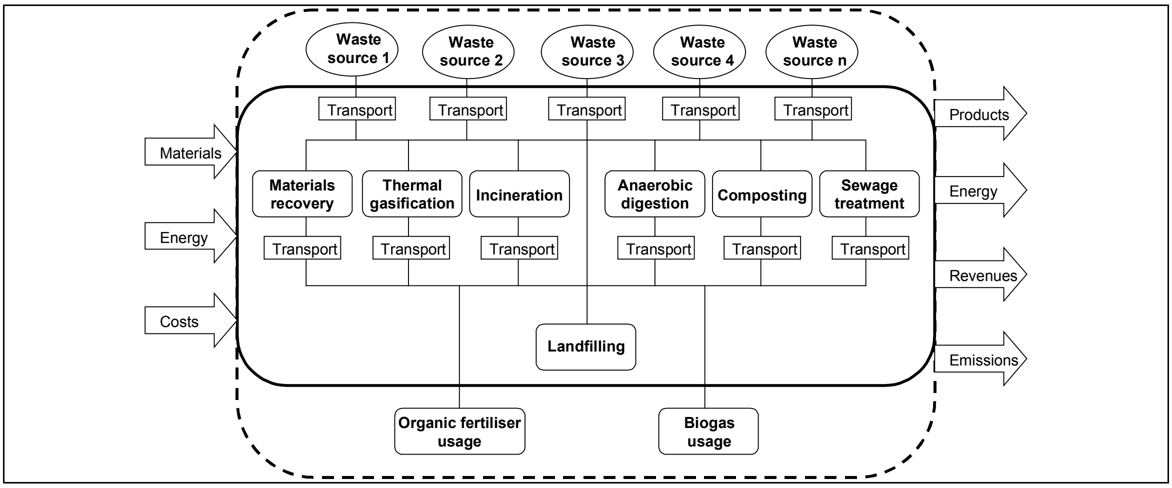


Figure 1. A conceptual model of a complete waste management system comprising a number of processes described by different submodels.

At the top of the conceptual model in Figure 2 there are different waste sources, followed by different transport and treatment processes. The solid line in Figure 3 encloses the waste management core system, where wastes are treated and different products are formed.

Li*fe* Cycle Assessment in ORWARE

The material flow analysis carried out in ORWARE generates data on emissions from the system, which is aggregated into different environmental impact categories. This makes it possible to compare the influence of different waste management system alternatives on e.g. the greenhouse effect, acidification, eutrophication and other impact categories.

The system boundaries are of three different types; time, space and function. In an analysis of a certain system, the temporal system boundaries vary between different studies (depends on scope) and also between different submodels. Most of the process data used are annual averages but for the landfill model and the arable land long-term effects are also included.

There is a geographical boundary delimiting the waste management system as shown in Figure 2, whereas emissions and resource depletion are included regardless of where they occur. The system boundaries in ORWARE are chosen with an LCA perspective, thus including in principle all processes that are connected to the life cycle of a product (in this case a waste management system). Our coverage of life cycle impacts covers raw material extraction, refinery, production and use. Construction, demolition and final disposal of capital equipment are not included regarding energy consumption and emissions but are included for economy.

The main function of a waste management system is to treat a certain amount of waste from the defined area. Today, many waste management systems provide energy supply in addition to waste treatment. In other cases, they provide fertiliser, or in most recent years recycled products or materials. The compensation of different functional units in ORWARE is achieved by expanding the system boundaries to include different so-called compensatory processes (cf. Figure 2). Either the waste management system or the compensatory system provides the functional units.

Compensatory systems also have up-stream and down-stream processes. Therefore, each treatment alternative in ORWARE has its own unique design of core system as well as different compensatory systems. This has been illustrated in Figure 2.

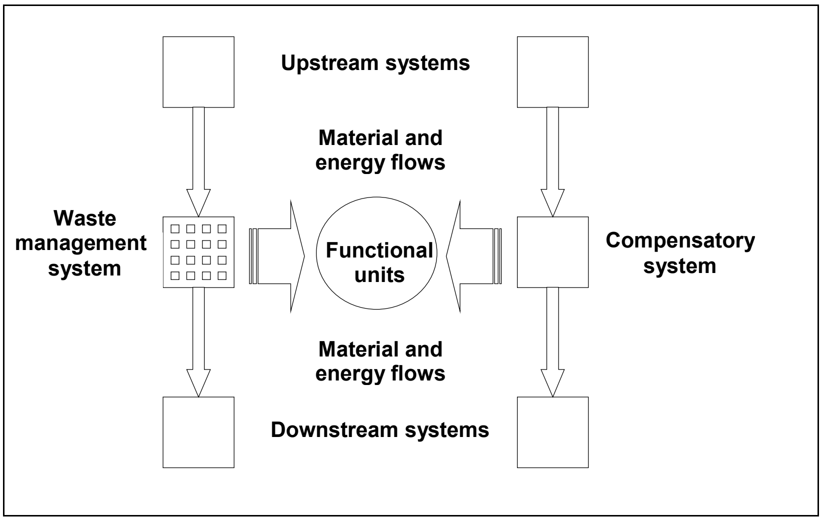


Figure 2. Conceptual model of the total system in ORWARE.

The total system comprises:

- the waste management system with different submodels i.e. the core system of the waste management system
- key flows of material and energy connected to up-stream and down-stream systems
- the compensatory system with core system as well as up- and downstream systems.

System boundaries in this study

The time frame of the study was one year. The space boundary was chosen to the three municipalities of Stockholm, Uppsala and Älvdalen.

- Stockholm is a big city with an incineration plant and system for district heating. There is no arable land within the municipality borders. Arable land is needed for spreading of the organic fertiliser produced from biological treatment of the organic waste.

- Uppsala is a relatively big municipality, also with an incineration plant and system for district heating. Arable land can be found close to the city area.

- Älvdalen is a small municipality and lacks of incineration plant and system for district heating. There is hardly any agricultural soil at all within the municipality. In Älvdalen some of the most famous ski-centres are located which means that during short time periods tourists produce large amounts of waste with a low degree of source separation.

Table 1. Statistical data for the three municipalities

	Älvdalen	Uppsala	Stockholm
Number of persons	8 100	186 000	496 000
Number of households	5 299	84 000	380 000
Number of detached houses in rural areas	Divided in	9 000	0
Number of detached houses in city areas	North 2 700 p.e.	19 000	40 000
Number of departments	South 5 400 p.e	56 000	340 000
Amount of easy biodegradable organic waste (tonnes/year)	1 388	23 155	93 121
Amount of plastic containers (tonnes/year)	172	2 616	21 056
Amount of paper containers (tonnes/year)	194	3 552	21 649
Total amount of waste (tonnes/year)	2 900	82 600	255 100

The parameters considered with respect to energy, environment and economy are:

Energy
- Consumption of primary energy carriers

Environmental effects
- Global Warming Potential
- Acidification Potential
- Eutrophication Potential
- Formation of photochemical oxidants
- Heavy Metals (input/output analysis)

Economy
- Financial costs
- Environmental costs (valuation of the emissions)

Important assumptions in these analyses are the choices of upstream and compensatory energy sources. In this study the electricity is supplied by power generation in Danish coal condense power stations. This assumption has a high implication on the results but it is hard to prove that this is always true. Small variations in the national electricity consumption are balanced for by making a change in the most expensive power supply. In many cases - due to that Sweden is linked to the power systems of the neighbouring countries - the most expensive power supply is coal condense power.

For district heating, there is no national grid. In each municipality the competing fuel could be peat, wood chips, oil or coal. It all depends on how much heat that is considered and when the question is raised. Biofuel was chosen to be the compensatory heat in all municipalities due to that biofuel is possible to combust in an incinerator, and a biomass fired heat plant is often the alternative if not building an incineration plant.

In a sensitivity analysis other options for upstream and compensatory energy has been studied. For electricity Swedish mix has been used and for district heating using coal in Uppsala and oil in Stockholm and Älvdalen.

Description of the scenarios

In all scenarios journal paper (75 %), glass (70 %) and metals (50 %) are sorted out and recycled outside the studied system. For the fractions organic waste, plastic containers and cardboard containers the upper limit of 70 % source separation in households has been chosen. For companies the corresponding figure is 80 % (including LDPE as well). The goals for material recycling in Sweden are far below this figure but 70 % has been chosen as a level possible to reach in the future, looking at the recycling levels for other waste fractions.

For the materials studied, following treatment options are available:

- Easy biodegradable organic waste: incineration, anaerobic digestion, composting (home composting and central composting), landfilling
- Cardboard containers: incineration, material recycling, landfilling
- Plastic containers: incineration, material recycling, landfilling
- Remaining combustible waste: incineration, landfilling

From this, following scenarios have been studied:

Incineration scenarios
A1 Incineration of all waste

A2 Incineration of 90 % of all waste, 10 % is landfilled during summertime. This is due to maintenance of the incineration plant and low demand for district heating leading to partial shutdown of the plant.

Biological treatment scenarios

Sorting out 70 % of the easy biodegradable organic waste. The rest of the waste is incinerated.

B1	Stockholm, Uppsala and Älvdalen:	Anaerobic digestion; the biogas is used for fuelling busses.
B2	Stockholm and Uppsala:	Anaerobic digestion; the biogas is combusted in a gas engine for generating heat and power
	Älvdalen:	70 % composted in central windrow compost and 30 % composted in households
B3	Stockholm:	Anaerobic digestion; the biogas is used for fuelling cars.
	Uppsala and Älvdalen:	Windrow composting.

Material recycling scenarios

C Sorting out 70 % of HDPE from households and 80 % of HDPE and LDPE from business. Material recycling. The rest of the waste is incinerated

D Sorting out 70 % of cardboard from households and 80 % of cardboard from business. Material recycling. The rest of the waste is being incinerated.

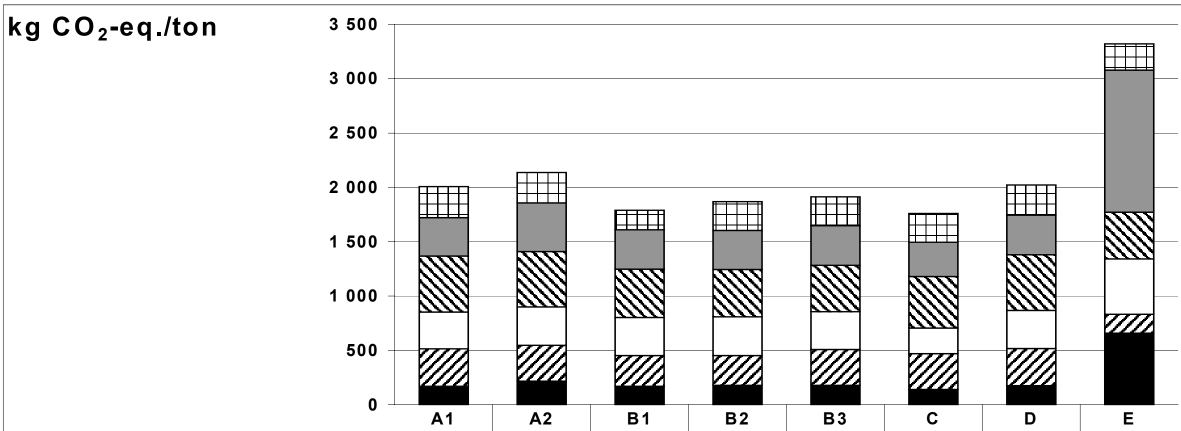
Landfill scenario

E Landfilling of all waste

Results

Results for Global Warming, Acidification, Eutrophication, Energy Consumption and Financial and Environmental Costs are displayed in diagrams covering all three municipalities. In order to capture the results in the same diagram, normalisation by the waste amount treated has been done. The results will only be discussed on a total level, not penetrating the three municipalities separately. Note that B2 and B3 are incomparable as these scenarios are designed in different ways in the three municipalities.

Global Warming

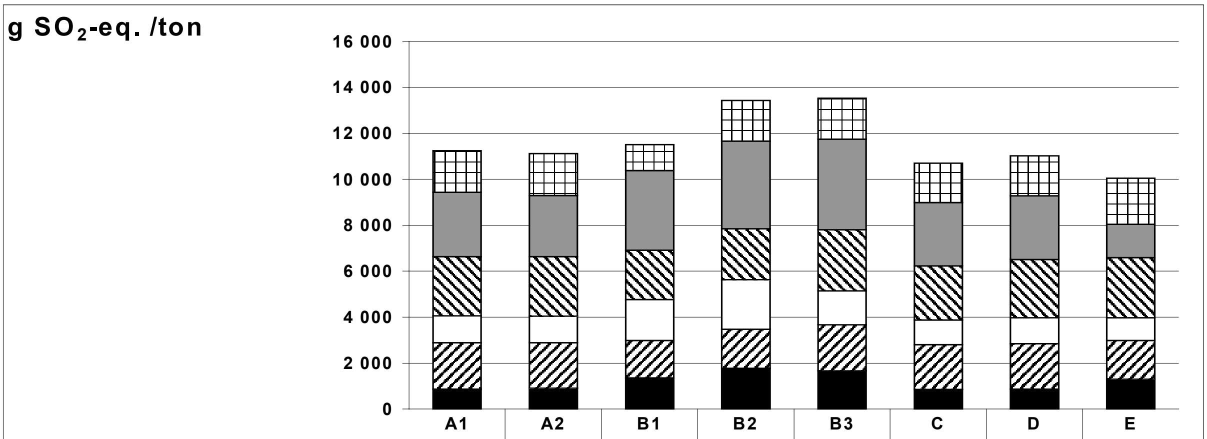


	A1	A2	B1	B2	B3	C	D	E
Compensatory System Älvdalen	284	280	179	267	267	266	278	243
Waste Management System Älvdalen	354	449	364	358	365	316	363	1 308
Compensatory System Stockholm	516	508	444	438	426	474	514	430
Waste Management System Stockholm	339	355	349	355	347	233	351	508
Compensatory System Uppsala	345	328	282	276	332	331	342	176
Waste Management System Uppsala	168	217	170	176	176	139	173	656

Figure 3. Global Warming Potential for the different scenarios.

The worst scenario is landfilling, especially landfilling in Älvdalen due to methane emissions. The landfill in Älvdalen does not have a system for collection of the methane gas. Compared to incineration recycling of materials and nutrients shows slightly lower impact. Changes are small but the lowest impact are found in scenarios B1 (anaerobic digestion using biogas in busses) and C (recycling of plastic containers).

Acidification

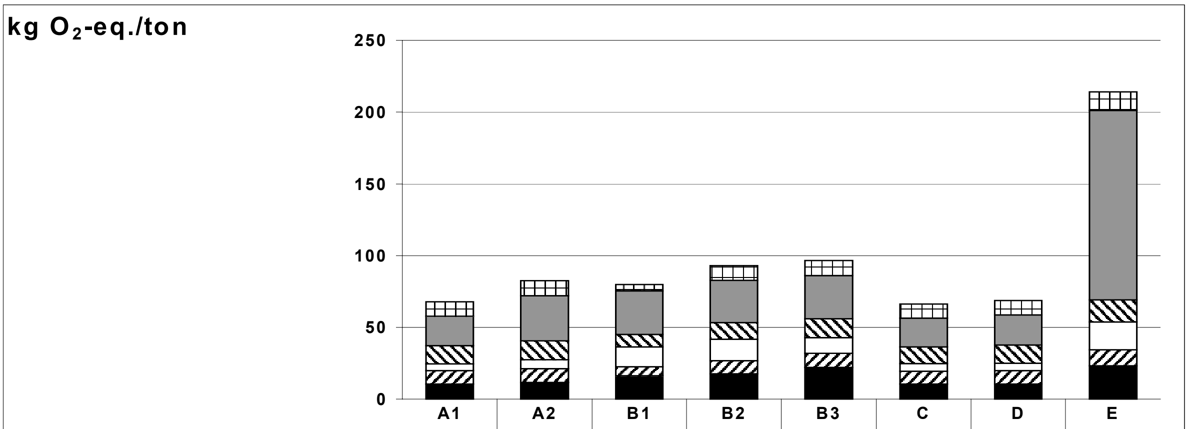


	A1	A2	B1	B2	B3	C	D	E
Compensatory System Älvdalen	1 803	1 824	1 115	1 775	1 775	1 704	1 732	2 009
Waste Management System Älvdalen	2 799	2 664	3 478	3 820	3 943	2 760	2 784	1 451
Compensatory System Stockholm	2 588	2 595	2 133	2 207	2 667	2 351	2 531	2 612
Waste Management System Stockholm	1 169	1 144	1 787	2 156	1 475	1 082	1 132	987
Compensatory System Uppsala	2 020	1 988	1 631	1 705	2 006	1 944	1 981	1 699
Waste Management System Uppsala	862	907	1 358	1 768	1 665	858	860	1 295

Figure 4. Acidification potential for the different scenarios.

Scenarios B2 and B3 are highest due to emissions of ammonia in the compost process and high NO_x-emissions from the internal combustion engine generating heat and power from biogas. All other scenarios are within the same range except for the landfilling scenario. The landfill in Älvdalen doesn't have a gas collection system why there are no emissions from combustion of landfill gas. In Uppsala and Stockholm landfill gas is being collected and combusted whereas the composition of the waste in Uppsala generates more gas than the waste in Stockholm.

Eutrophication

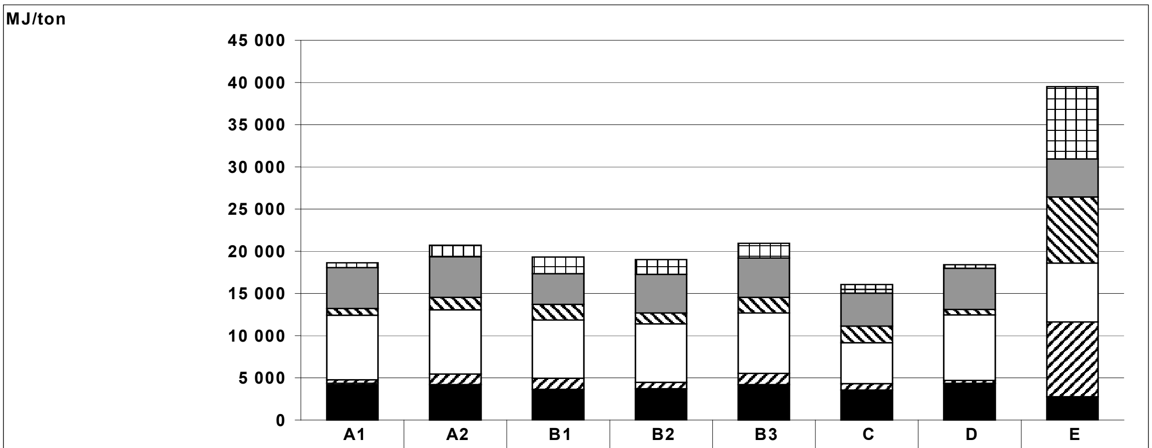


	A1	A2	B1	B2	B3	C	D	E
Compensatory System Älvdalen	10	10	5	10	10	10	10	13
Waste Management System Älvdalen	20	32	30	29	30	20	21	132
Compensatory System Stockholm	13	13	9	12	13	12	13	15
Waste Management System Stockholm	5	6	14	15	11	5	5	20
Compensatory System Uppsala	9	10	6	9	10	9	9	11
Waste Management System Uppsala	10	12	16	18	22	10	10	23

Figure 5. Eutrophication potential for the different scenarios.

As for GWP landfilling gives the highest impact. The landfill in Älvdalen lacks of leachate water treatment, which gives high emissions of Phosphorous, Nitrogen and COD. Recycling of nutrients causes emissions from spreading of the organic fertiliser. These emissions are higher than for spreading of mineral fertiliser. Recycling of materials gives just about the same impact as incineration.

Consumption of primary energy carriers



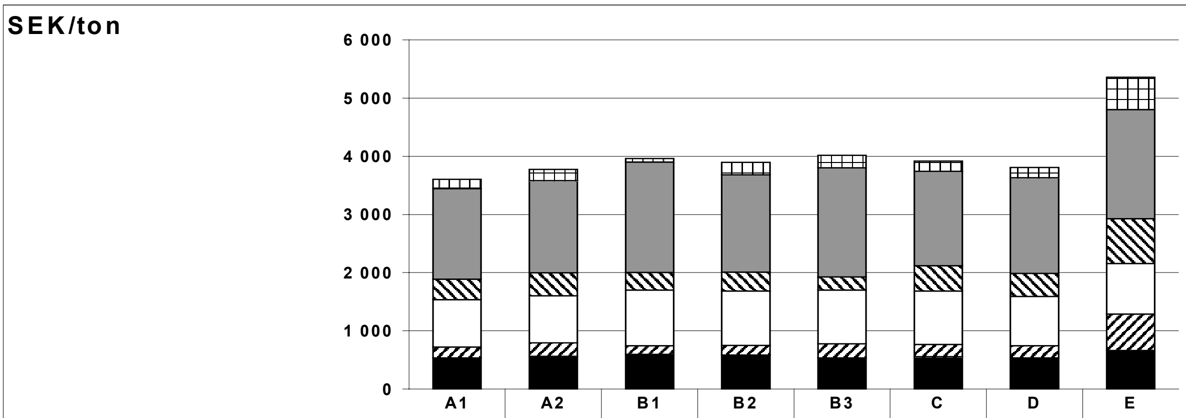
	A1	A2	B1	B2	B3	C	D	E
Renewable Älvdalen	564	1363	1963	1745	1742	1036	453	8556
Non Renewable Älvdalen	4839	4806	3630	4580	4676	3884	4890	4520
Renewable Stockholm	792	1500	1873	1286	1873	1975	607	7851
Non Renewable Stockholm	7674	7606	6874	6940	7135	4839	7775	6950
Renewable Uppsala	411	1257	1352	763	1306	767	348	8870
Non Renewable Uppsala	4346	4188	3628	3694	4226	3549	4356	2762

Figure 6. Consumption of primary energy carriers for the different scenarios.

In general the differences are small for all scenarios except for scenario E (landfilling) whose consumption of energy resources is much higher. The lowest consumption can be seen for recycling of plastic containers. By recycling of plastic, fossil carbon can be saved by

1. Not using oil for production of virgin plastic.
2. The emissions of fossil CO_2 from the incinerator are replaced by non-fossil CO_2 from combustion of biofuel.

Financial costs

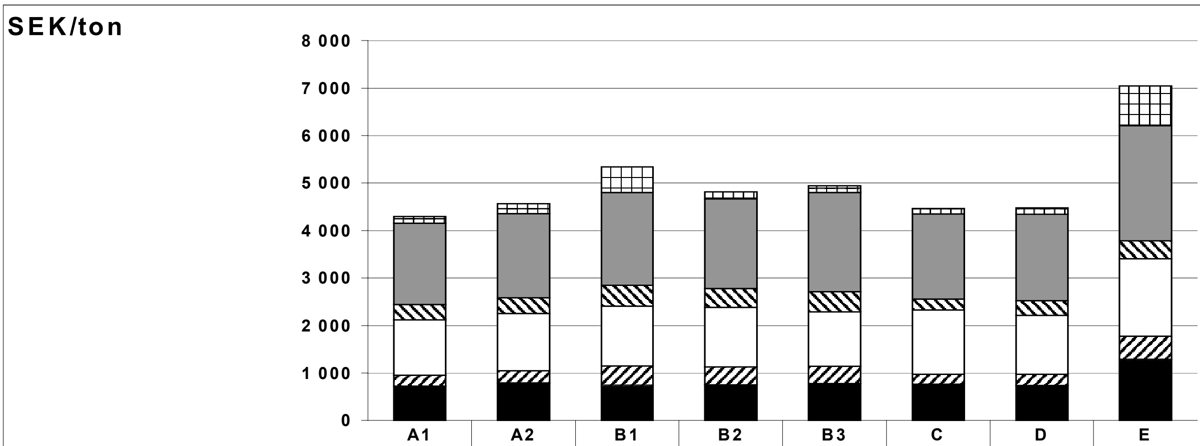


	A1	A2	B1	B2	B3	C	D	E
Compensatory System Älvdalen	155	195	58	214	213	179	177	549
Waste Management System Älvdalen	1 557	1 580	1 898	1 673	1 875	1 618	1 641	1 878
Compensatory System Stockholm	354	396	305	326	224	437	395	769
Waste Management System Stockholm	812	812	954	929	928	919	851	868
Compensatory System Uppsala	186	230	149	170	239	209	206	628
Waste Management System Uppsala	535	559	594	581	535	554	535	660

Figure 7. Financial costs for the different scenarios.

The total costs are slightly higher for the different recycling scenarios compared to incineration. Landfilling is the most expensive waste treatment due to the landfill tax. The most expensive waste treatment can be found in Älvdalen due to high transport costs. Uppsala reflects a fairly cheap waste treatment due to lower investment costs for the incineration plant (heat generation only) and cheaper collection.

Financial and Environmental costs



	A1	A2	B1	B2	B3	C	D	E
Environmental costs Älvdalen	139	209	536	147	142	110	137	835
Financial costs Älvdalen	1 712	1 775	1 956	1 886	2 088	1 797	1 818	2 427
Environmental costs Stockholm	322	328	438	403	423	226	309	375
Financial costs Stockholm	1 167	1 208	1 258	1 254	1 152	1 356	1 245	1 637
Environmental costs Uppsala	232	257	408	373	365	209	229	483
Financial costs Uppsala	721	790	743	751	773	763	741	1 288

Figure 8. Financial and environmental costs for the different scenarios.

Here the financial costs are adjusted in that way that all costs related to environment (taxes, fees) are subtracted. To compensate for this, the emissions from the system have been economically valued. In general the total cost is adjusted upwards compared to the last diagram, but approximately much as the same for all scenarios. Material recycling becomes relatively cheaper and landfilling more expensive related to incineration.

Conclusions

Despite the fact that the systems studied are designed with a high degree of source separation and well functioning facilities, the differences between energy recovery and materials' and nutrients' recycling are relatively small. Even with a high degree of source separation a large part of the waste has to be incinerated. A comparison between incineration and recycling of 1 kg of plastic will show a greater difference, but in this study the whole waste stream is being considered.

There are benefits and drawbacks associated with all waste management options.

- Materials' recycling of plastic containers is comparable to incineration from a welfare economic aspect, but gives less environmental impact and lower energy use – on condition that the recycled plastic replaces virgin plastic.

- Materials' recycling of cardboard containers is comparable to incineration concerning welfare economy and energy, but has both environmental advantages and disadvantages.

- Anaerobic digestion of easily degradable waste gives a higher welfare economic cost than incineration, and has both environmental advantages and disadvantages. Conclusions regarding energy use depends upon how the biogas is used.

- Composting of easily degradable waste is comparable to anaerobic digestion from a welfare economic aspect, but gives higher energy use and environmental impact.

It is however clear that direct landfilling of mixed household waste is not a good waste treatment option. Baling of waste during periods when incineration is impossible is thus a good measure. Landfilling plays an important role in the environmentally sustainable society as a sink for the residues from waste treatment that are sometimes hazardous and should be isolated from living creatures.

With respect to environment and consumption of energy resources transports are of minor importance. In sparsely populated areas collection and transports can be expensive, relatively speaking. In city areas transports may inflict on human health comprising impacts as i.e. noise. What is important to keep in mind is that waste management causes impacts on health that has not being evaluated due to difficulties in the assessment of ecotoxicology and human health.

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Reconsidering the German Dual System for Lightweight Packaging

Jürgen Giegrich, *Andreas Detzel*[16]

Abstract

Since 1993 the so called Duale System DSD had been established as a consequence of the packaging regulation in Germany. DSD organised a second (dual) private collection for packaging material with a annual budget of about 4.000 Mio DM. The high cost of the system lead to a decision of the German Bundesrat to re-evaluate the environmental and economical performance especially of the lightweight packaging fractions in order to perhaps change the packaging regulation. Total cost calculations and LCA were used in the study commissioned to HTP and IFEU to reach the objective.

The lightweight packaging (LVP) fraction includes all plastic packages, tin cans, aluminium cans, liquid packaging board containers and all other kinds of mixed material compounds. According to available statistics the LVP waste stream with a total amount of about 2.000.000 t per year had been subdivided into 17 different fractions not only using material characteristics but also packaging characteristics like bottles, boxes, films, etc.

With the help of the project partner HTP the material streams from the about 250 sorting plants for LVP in Germany the system had been modelled according to LCA technique and had been assessed. Three technology steps of sorting and material recovery (including feedstock recovery) had been applied which cover possible developments for the next 10 years:

- current state of the art (basis year 1999)
- optimised state of the art
- SORTEC technology

These material recovery options had been compared with waste disposal options like the current status (30% incineration/70% landfilling) 100% incineration and 100% incineration with optimised energy recovery.

The results which will be presented show a clear environmental advantage of the material recovery options for most of 17 packaging fractions for the current state of the art. Recycling of some fractions like some plastics and compound packaging however have a balanced result compared to the current disposal and even more to a 100 % incineration option. Then the higher cost of the DSD system in comparison to disposal is the main focus for the political discussion now. The future options show a clear environmental

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[16] ifeu - Institut für Energie- und Umweltforschung Heidelberg GmbH, Wilckensstr. 3, 69120 Heidelberg, Tel.: 06221/476721; Fax.: 06221/476719, e-mail: juergen.giegrich@ifeu.de

advantage for the material recovery options but leave some uncertainty because of the prognosis for future secondary material markets.

Key words: leightweigth packaging, Duales System DSD, sorting technologies, recovery options, plastic, compound packages

Evaluation of waste treatment processes for MSW rest fraction

Karl C. Vrancken*, *Rudi Torfs, Ann Van der Linden*[17]

Abstract

Treatment scenarios for grey waste were studied and compared for the Flemish (B) Ministry of the Environment. The aspects under study were: environmental impact, energy, recovery of materials, cost and operation. The choice of the scenarios was based on a selection and combination of processes that can be implemented in short term on the Flemish market and have been demonstrated for treatment of MSW at industrial or pilot scale. The evaluation was based on data from system suppliers and from literature. Own calculations and practical check-up controlled the data. The impact assessment was made using the Eco-Indicator 99 methodology. Grate incineration with energy recuperation, non-catalytic DeNOx, semi-wet flue gas cleaning, activated carbon injection and bottom ash treatment served as a reference scenario. The study involved two mechanical-biological processes, CFB-incineration, gasification and integrated pyrolysis.

For expansion of the Flemish waste treatment capacity on the short term, the following scenarios prove to have a better performance than the reference: (i)Grate incineration with selective catalytic reduction of NOx; (ii)Separation and digestion followed by circulating fluidised bed incineration of RDF, including the sludge cake; (iii) Biological drying and separation followed by circulating fluidised bed incineration of RDF. The differences in environmental performance between the above-mentioned systems are relatively small. The results of the study are used to set out and support the government policy concerning further expansion of the waste treatment capacity.

Introduction

In a study for the Flemish Minister of the Environment, various treatment scenarios for the rest fraction of MSW and non-specific category-II industrial waste were discussed and compared, concerning environmental impact, energy, materials recycling, costs and operation. For evaluation of the environmental impact, an LCA-based approach was applied, using the Eco-indicator 99 for impact assessment.

[17] Vito (Flemisch Institute for Technological Research), Boeretang 200 - B2400 Mol,
*tel. 00-32-14335647, fax. 00-32-14321186, mail: karl.vrancken@vito.be

The treatment scenarios involve the processing of the waste into heat and/or electricity. This treatment is performed in an integral or integrated process or by means of a combined mechanical-biological treatment and thermal valorisation of the refuse derived fuel (RDF).

The goal of the study was to make a comparative evaluation of various treatment scenarios that can be implemented in the current Flemish waste market on short term (max. 2 years) and in conformity with the Flemish legal framework.

Approach

The study had two phases. In phase one an inventory and technical evaluation was made for waste treatment processes. The technical feasibility was evaluated for the following techniques: mechanical-biological pre-treatment, single pyrolysis, integrated pyrolysis and thermal valorisation in incineration plants, in gasifiers, in small-scale CHP boilers and in industrial processes (cement). The techniques from the inventory were discussed and their feasibility was studied in view of the above-mentioned outset.

The study was followed up by a steering committee featuring representatives from the public and private sector: ministry, municipalities, environmentalist movement as well as owners and operators. This steering committee made a selection of techniques, based on the results of phase 1. The selected techniques are representative for the current Flemish market and may be extrapolated to a broader range of suppliers. The selected techniques are:

- Integral MSW treatment in a grate incinerator, as a reference scenario (GF)
- Separation and digestion of the MSW (SDig)
- Biological drying and separation of the MSW (SBioD)
- Incineration of the produced RDF in an external circulating fluidised bed (CFB)
- Gasification of the produced RDF in a slagging gasifier (Vgas)
- Thermal valorisation of the syngas in a gas engine (M)
- Thermal valorisation of the syngas in an IGCC (IGCC)
- Integrated pyrolysis of the MSW (pyro)

In Figure 1, the various scenarios are depicted schematically:

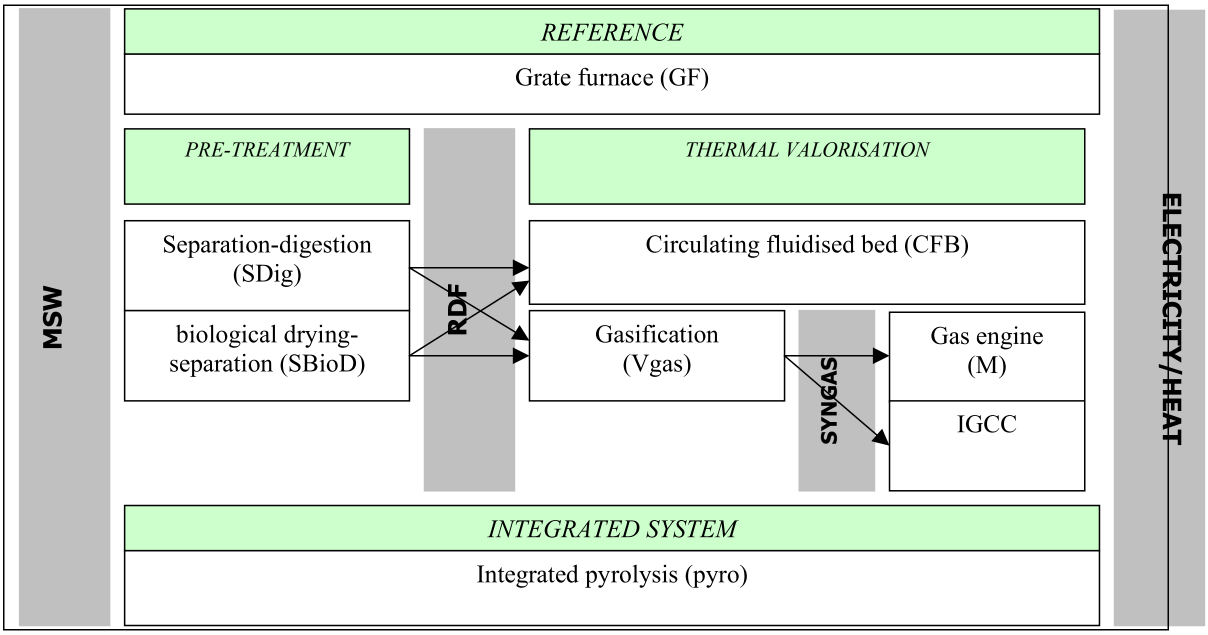

Figure 1. Waste treatment scenarios under study

The evaluation was based on data from system suppliers and from literature. Own calculations and practical check-up controlled the data. For set-up of the mass & energy-balance a defined waste composition was used. This waste has a heat of combustion of 8.53MJ/kg, a DS-content of 67.4% and an ash content of 26.9%DS. The systems were dimensioned to treat 150.000 ton/y of this waste. For evaluation of the different criteria, the impacts were calculated per ton of waste input.

Techniques

In the reference scenario the waste is processed integrally in a grate furnace with energy recovery, flue gas cleaning and bottom ash treatment. Two types of DeNOx-installation are considered, since both are in use in Flanders.

In the separation-digestion process, digestion is performed on a rest fraction, after separation of material (ferrous, non-ferrous and RDF) with a grading >40mm. The digestate is screened and washed, yielding inerts, sand, fibres and various residues. The end fraction is allowed to settle and processed in a belt-press to give a sludge cake.

Biological drying and separation aims at a maximal production of high-calorific fuel. The waste is processed in closed composting boxes, after size reduction and separation of the course ferrous fraction. After drying, a physico-mechanical separation yields fine ferrous fraction, non-ferrous, inert and the RDF.

Thermal valorisation of the RDF can be performed in a fluidised bed reactor. The material is incinerated in a turbulent sand bed. The RDF may also be gasified. The slagging gasifier was chosen as representing system, because this reactor has been optimised to treat municipal solid waste. The produced syngas is processed into electricity and heat through a gas engine or an IGCC.

Results

The various scenarios have been studied thoroughly. Each of the 5 aspects has been split up in subcriteria. The performance of each of the scenarios is compared to the grate furnace with SNCR (score 0). The targeted evolution is given in the second column (e.g. 'less environmental impact'). The scenarios are evaluated against this target: score '+' is the target is reached; score '-' if it is not reached. The various criteria are discussed below.

Table 1: Evaluation of waste treatment scenario's, score against reference GF-SNCR, '+' = better than GF-SNCR for the indicated criterion, '-' = worse than GF SNCR for the indicated criterion.

		GF SNCR	GF SCR	SDig-CFB	SBioD-CFB	SDig-Vgas	SBioD-Vgas	Pyro
Environment	less environmental impact direct + auxiliarries	**0**	**+**	**+**	**0**	**-**	**-**	**0**
	less environmental impact *incl. displaced emissions*	*0*	+	+	*0*	+	+	-
Energy	more recuperation of energy	**0**	**0**	**0**	**0**	**+**	**+**	**-**
Materials	less dsiposal	**0**	**0**	**-**	**+**	**+**	**+**	**+**
	more materials recovery	**0**	**0**	**+**	**0**	**+**	**+**	**+**
Economy	cheaper	**0**	**0**	**0**	**0**	**0**	**-**	**-**
Process	better process control	**0**	**0**	**0**	**0**	**-**	**-**	**-**
	higher flexibility	**0**	**0**	**0**	**0/+**	**-**	**-**	**0**

Environmental impact: less environmental impact

The environmental impact was assessed using the Eco-indicator 99-method. The impact is defined using 3 categories: damage to human health, disturbance of the ecosystem and exhaustion of natural goods. As an example the scores for 3 categories of damage to human health are given in Figure 2. It is not possible to set up a ranking for environmental impact, because various parameters are of importance. The scores in Table 1 are based on direct emissions and emissions caused by the use of raw and auxiliary materials on the one hand, on the other hand the scores including displaced emissions are given.

From evaluation of the data it is clear that various parameters and assumptions have impact on the result. The most important ones are:

- **the goal of the treatment** : The impact (per ton MSW input) is largely dependent on the amount of material for incineration. Scenario's with full thermal treatment (GF, pyro) or optimised RDF(fuel)-production show higher impacts. Contrarily, scenarios that bring carbon-containing fractions to disposal show lower impacts for this category.

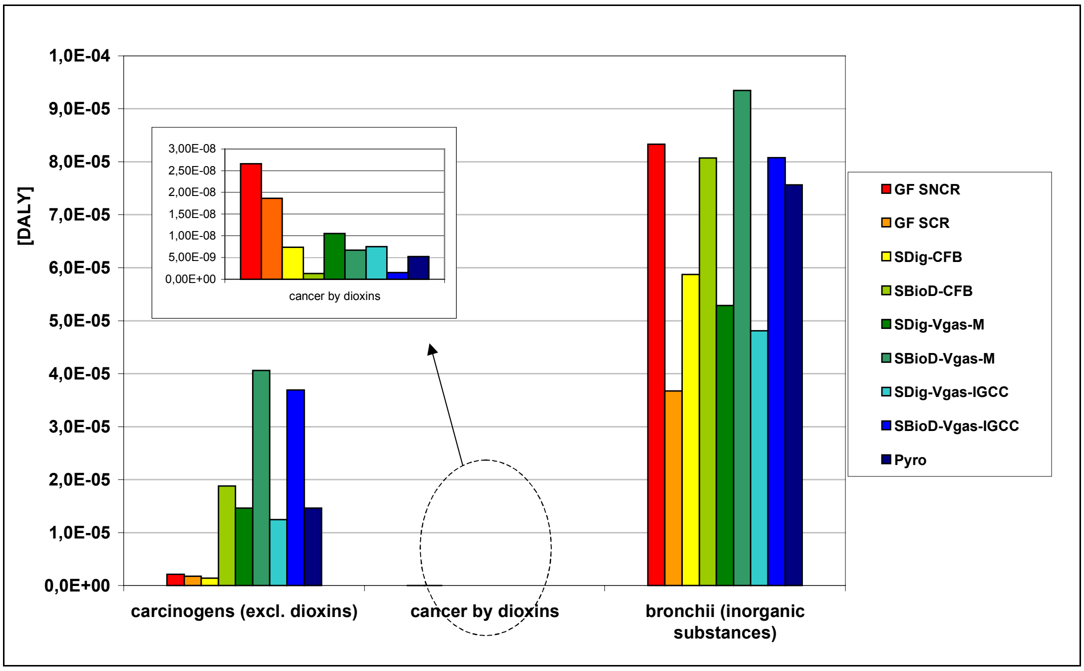

Figure 2: Impacts for damage to human health, in DALY (Eco-indicator 99)

- **the final destination of residual carbon**: The SDig rest fractions (sand, inerts, fibres, residue…) contain rest carbon. If we presume that all this carbon is turned into methane eventually, the SDig scenarios have a much larger impact in the category of greenhouse-related health effects compared to the other scenarios. The score in Table 1 would turn into '0' or '-'. In the basic analysis we presume that all carbon remains bound. In reality a partial (but not predictable) conversion into CO_2 and CH_4 will occur.
- **the production of electricity for own use** : Some processes need external electricity, fuels or oxygen. The latter are produced in an external energy-consuming process. Both external electricity productions cause a higher environmental impact. The score of these scenarios can be optimised compared to the GF if own electricity is used.
- **the basis for calculation of displaced emissions :** On basis of the produced energy, the displaced emissions (and thus effect on the environment) in other places can be calculated. The relative score against the reference changes if displaced emissions are allocated to an IGCC or fossil fuel combustion. However, the relative order of the impacts does not really change.

The choice of these parameters in the basic analysis will affect the relative scores and thus the final evaluation. The specific effect of each parameter can be assessed in a sensitivity analysis. In general the effect on the relative order of the scenarios is small. Nevertheless, the score against the reference scenario may change.

Energy: higher energetic efficiency

The energetic efficiency of the scenarios is evaluated as a second criterion. The net yield of electricity is the electricity produced, minus the internal use and the procured electricity. The electric efficiency is **the ratio between net electricity produced and the sum of the calorific values of wastes and fuels**.

Gasification reaches a clearly higher energy-gain than the other techniques. This results in a higher net efficiency for the scenarios with gasification, combined with a gas turbine (ca. 25%). The other combined scenarios have a comparable energetic efficiency of about 20%. The integrated pyrolysis has a low efficiency of 7%. The optimal combination of techniques concerning the energetic efficiency is dependent on the real-life syngasproduction. Based on the currently given syngasproduction, SBioD-Vgas-IGCC has got the highest efficiency.

Materials: less disposal - more recovery

The mechanical-biological treatment systems (SDig, SBioD) aim at producing a high calorific fuel (RDF) and materials for re-use. Also in the other techniques material for re-use is produced. The evaluation of the effective material recovery must be based on the amounts of products, their environmental and technical quality. The latter is of main importance for the inert fraction. In the evaluation, the following assumptions have impact on the final result:

- **destination of the inert fraction** : All techniques produce an inert fraction that complies with the Flemish regulation on secondary materials (VLAREA). However, this is no guarantee that all materials will find market introduction and effective valorisation. Practical large-scale application in road construction is only demonstrated for treated GF-bottom ash. In the final evaluation, we assumed that all material that complies with the regulation will find an application.
- **the goal of the treatment** : Optimisation of the treatment towards material recovery (SDig) causes the formation of residues for disposal. If the system aims at fuel production (SBioD) the amount of residue for disposal is minimal. The grate furnace produces a relatively high amount of residue for disposal. Only for the SDig-CFB-scenario, the amount is higher. The residue for disposal is minimal for the SBioD-Vgas scenario.
- **the final destination of non-inert fractions** : The residue for disposal from SDig contains sludge cake and residue. The sludge cake can be thermally treated, mixed with the RDF. This results in a change of score for less disposal from '-' to '+'. On the other hand, the amount RDF for incineration increases, while its calorific value decreases. This results in a higher environmental impact. The relative score ('+') however remains unchanged. The energetic efficiency of the SDig-Vgas-IGCC-route reaches a value greater than 25%.

Economy: cheaper treatment

Data concerning costs were collected mainly from the system suppliers and were completed with literature data. For optimal data quality, the calculated values were checked against data from existing facilities. It should be noted, however, that cost data are based on simulations and not on real tenders.

A distinction was made between capital costs and operating costs, corrected for revenues. On this basis a yearly cost and a cost per ton was calculated. Some additional elements like taxes, TVA, capital costs, government support etc. were not considered. The results could therefore not be regarded as a market price, but merely the **full operator costs**.

Treatment in a grate furnace (reference situation) appears to be the cheapest, having a cost of somewhat less (for SNCR) and somewhat more (for SCR) than 75 Euro/ton MSW input. Integrated pyrolysis is the most expensive scenario. The cost is more than 75% higher. Both investment and operating costs are higher and the expected revenues are smaller. Calculations for the scenarios with SDig and SBioD give results in between both extremes, albeit closer to the reference. For pre-treatment, SDig is more expensive than SBioD, but a smaller amount of RDF needs to be processed. This is reflected in the full cost of the entire treatment, mainly for Vgas-IGCC. If CFB is used as thermal treatment, the difference is smaller.

Compared to GF-SNCR, the scenarios SDig-CFB, SBioD-CFB and SDig-Vgas are within a range of 20%. The SBioD-Vgas is ca.40% more expensive. If GF-SCR is taken as a reference the ranges are resp. 15% and 30%.

Operation : better process control and higher flexibility of input and output.

Scores for evaluation of the operation are based on information from site visits, contacts with operators and literature data. This category inevitably has an **intrinsic subjective character**. However, the evaluation was discussed repeatedly in the steering committee. Here operators as well as other specialists could give their assessment of the various criteria. The final evaluation received a **consensus from the steering committee**. The full evaluation regarded process control, reliability, safety, maintenance and flexibility of input and output.

The grate furnace is the most well known and reliable treatment system for municipal solid waste. The system has a high flexibility to handle changes in waste composition. SDig, SBioD and CFB are techniques that have been initially developed for other waste types than MSW grey waste. All of them have been demonstrated recently to work on this grey waste (SDig, SBioD) or RDF (CFB) as well. The experience with long-term operation is still small. As compared to the grate furnace, mechanical-biological pre-treatment systems have a relatively simple process build-up and operation.

For SDig various rest fractions are produced. The performance of this system (and its environmental impact) is dependent on the application found for these flows. The flexibility of the SDig system is limited by the need for these application routes. This limit does not apply for the SBioD system. There is only a single inert fraction. The RDF is dry and stable. This gives the opportunity for stocking, without a loss of fuel quality.

CFB, Vgas and pyro systems have optimal operation with fuels that have been size-reduced. Additionally, the material needs to be homogenised. These systems are less appropriate to respond to changes in fuel quality. The flexibility of the Vgas is further hampered by the need of complementary fuels for optimal operation. The installation should not be operated with a single monostream. The fuel mix needs to be adapted to allow the proper operation of the reactor.

Integrated pyrolysis and gasification (in slagging gasifier) are relatively new processes that have not been demonstrated in continuous operation on MSW rest fraction. The production of synthesis gas leads to specific new process properties. The operation of reactors at elevated temperature and the coupling of the various process compounds cause possible problems with these techniques. A further demonstration of these techniques on industrial scale is needed in order to allow a full and good process control.

General conclusion

Treatment scenarios for grey waste were studied and compared. The evaluation was based on a multi-criteria approach. For assessment of the environmental impact the Eco-Indicator- 99 method was used. The quality of the end result and its applicability to support the waste treatment policy, is based on some specific characteristics of this study:

- The study was followed up by a steering committee featuring representatives from the public and private sector: ministry, municipalities, environmentalist movement as well as owners and operators. The committee was involved in setting out the boundaries for the methodology, the selection of techniques and the assumptions that had to be made.
- The study had a clear goal in that it had to be applicable in short term. This was reflected in a selection of techniques that are clearly present on the Flemish waste treatment scene. This resulted in good co-operation of the system suppliers.

Grate incineration with energy recuperation, non-catalytic DeNOx, semi-wet flue gas cleaning, activated carbon injection and bottom ash treatment served as a reference scenario. The evaluation resulted in the following final conclusions:

- The reference system is a reliable and high-performance system that complies with the current environmental legislation and the new European incineration directive.
- For expansion of the Flemish waste treatment capacity on the short term, the following scenarios prove to have a better performance than the reference:

  - Grate incineration with selective catalytic reduction of NOx
  - Separation and digestion followed by circulating fluidised bed incineration of RDF
  - Biological drying and separation followed by circulating fluidised bed incineration of RDF

  The differences in environmental performance between the above-mentioned systems are relatively small. In the combined scenarios, a reduction of the direct impacts is compensated by higher impacts from production of auxiliary materials and fuels. The SDig-scenario is characterised by a higher degree of material recovery. The practical application of the produced flows needs to be demonstrated. The SBioD-scenario is characterised by optimised (RDF) fuel characteristics and a maximal reduction of disposal of residues.
- Concerning energy, costs and process operation there are no great differences between the 3 above-mentioned scenarios and the reference.
- Integrated pyrolysis is characterised by a relatively high cost, low energetic efficiency and a limited demonstration. This technique holds no amelioration compared to the reference.
- Gasification in combination with IGCC is a promising scenario for thermal treatment of RDF. Today, this technique is insufficiently demonstrated to guarantee a reliable operation.

These results have been published on the internet and will be presented to a broad audience at a workshop. On this occasion there will also be a debate with representatives of the waste treatment sector, the communities responsible for waste management and the environmentalist movement. They will discuss their view on how this study can be implemented into a good waste management policy. The minister herself will base the issuing of working permits for new installations on this study. The Flemish waste treatment capacity will be expanded further on basis of mechanical-biological pretreatment and thermal valorisation of the RDF in a fluidised bed combustor.

Session 2: Summary of discussion

Summarised by Simon Aumônier and Marcus Carlsson Reich. Edited by Jan-Olov Sundqvist.

Four studies of different waste strategies for some waste fractions were presented.

A common denominator in the result was the evidence of the waste hierarchy (considering environmental aspects):

Material recycling > Incineration > Composting >> Landfilling
≀
Anaerobic digestion

Considering the environmental aspects, material recycling is more favourable than incineration, which is much more favourable than landfilling (considering environmental aspects). Anaerobic digestion is ”equal” to incineration, in some aspects it is better and in some aspects worse. Anaerobic digestion is better than composting, which is much better than landfilling.

The most important assumptions in the studies:
- External energy (e.g. electricity and energy)
- Markets for recycled materials.
- Efficiency in incinerators and combustors (coal, biofuel, etc.)

Another important question is what happens to the carbon in the rest waste.

The importance of final sinks (for metals) was discussed. In LCA there is problems how to deal with diluted flows. An example is when slag from waste incineration is used for road construction: what is the difference (in LCA) when slag is landfilled and slag is used for road construction.

The evaluation and weighing methods are important. There is a bunch of methods, which are used, but no method is perfect. All methods have disadvantages but they fill a purpose.

Session 3.

Chairman: Paul Brunner; Secretary: Stephanie Hellweg

Michael Eder
Long-Term Assessment of different waste management options – a new integrated and goal-oriented approach

Hannes Partl
Assessment of Kerbside Collection and Recycling Systems for Used Packaging Materials in Australia

Juha-Heikki Tanskanen
Integrated approach for formulating and comparing strategies of MSW management

Tomas Ekvall
Assessing external and indirect costs and benefits of recycling

Marcus Carlsson Reich
Economic assessment of waste management systems – case studies using the ORWARE model

Mattias Olofsson
A comparison of two different system engineering approaches for analysing waste-to energy options

Discussions

Long-Term Assessment of Waste Management Options in View of Final Storage Landfills

Michael Eder[18]

Remarks:

This paper is based on the study „Bewertung abfallwirtschaftlicher Maßnahmen mit dem Ziel der nachsorgefreien Deponie", (An Assessment of Waste Management Options for Creating a Long-Term Maintenance-Free Landfill), written by:

G. Döberl, R. Huber, P. H. Brunner, Institute for Water Quality and Waste Management, Vienna University of Technology, Vienna, Austria

M. Eder, R. Pierrard, W. Schönbäck, Institute for Public Finance and Infrastructure Policy, Vienna University of Technology, Vienna, Austria

W. Frühwirth, H. Hutterer, Corporation for Comprehensive Analyses Ltd., Vienna, Austria

Abstract

In a case study, different waste management options were compared with regard to the goals of the Austrian Waste Management Act, taking into account their long-term implications and their economic costs. Municipal solid waste and municipal sewage sludge have been defined as the system inputs. The selected options were compared to the status-quo by a material flow analysis and a corresponding economic evaluation. Both, a macro-economical cost-benefit analysis (CBA) and a "modified cost-effectiveness analysis" (MCEA) were used for this assessment. Unlike the classical CBA, the MCEA allows to include the long-term impacts of the landfilled residual material. The results obtained by the CBA correspond to those of the MCEA. The results confirm, that if long-term effects are taken into account, the goals of waste management can be reached more efficiently by thermal waste treatment than by mechanical-biological treatment or landfilling without pre-treatment.

[18] Institute for Water Quality and Waste Management, Vienna University of Technology, Karlsplatz 13/226.4, A-1040 Vienna, Austria, phone: +43 1 58801 226 56; fax:+43 1 504 22 34, e-mail: m.eder@awsnt.tuwien.ac.at

Keywords

Material Flow Analysis, Long-Term Emissions from Landfills, Assessment of Waste Management Options, Cost-Benefit Analysis, Modified Cost-Effectiveness Analysis

Introduction

The described report, “An Assessment of Waste Management Options for Creating a Long-Term Maintenance-Free Landfill” [1], compares and evaluates different waste management options with special consideration of their long-term implications. Multiple scenarios of these options were created and investigated as to which of them best fulfilled the goals of the Austrian Waste Management Act (Abfallwirtschaftgesetz, AWG [2]). The evaluation concentrated mainly on the macro-economic costs and the long-term stability of the options.

Serving as a basis for this study, an elaborated model of the Austrian MSW management system, as taken from the GUA & IFIP study [3], “Management of Household and Household-like Waste in Austria“ was used. Alterations to the GUA & IFIP study were done in order to best fulfill the needs of this assessment.

Methodology

The following steps were carried out in the study:

- System definition of Austrian waste management
- Definition of scenarios
- Determination of material, substance, and energy flows and business costs
- Modelling of short-, mid- and long-term landfill processes
- Assessment of ecological impacts and costs

System definition

For the comparison of different waste management options it was necessary to define the system exactly. An overall view of the defined system is given in Figur 1. The system consists of the system boundary (dotted line), processes (boxes), goods/substances (ellipses) and flows of goods/substances (arrows).

The spatial system boundary was defined as the borderline of the Republic of Austria. The system is also limited in time: 1 year for all processes, goods/substances and flows until landfilling and 10,000 years for landfill processes and underground waste disposal.

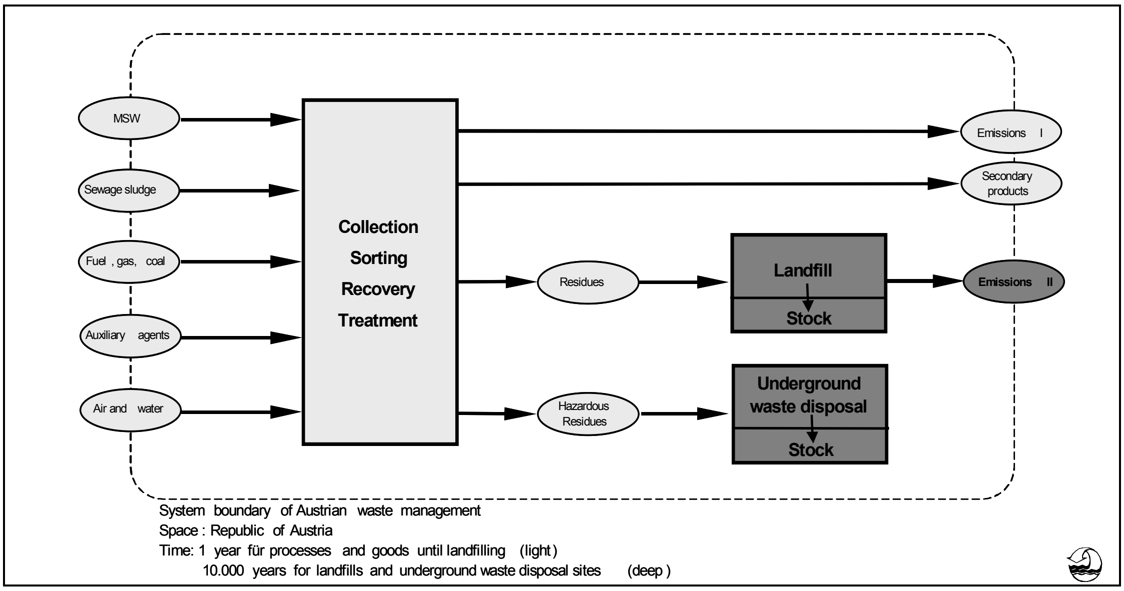

Figure 1: Overall view of the defined system of the Austrian Waste Management

The yearly amounts of MSW and municipal sewage sludge in the year 1996 as well as fuel, air, water and substances necessary for the different processes serve as system input. The amount of sewage sludge, which is deposited in agriculture has been excluded. Emissions and secondary products were defined as system output. Landfilled goods were kept within the system and were handled as stocks. The export of hazardous residuals in underground waste disposal sites outside from Austria is included in the system, other waste imports and exports were neglected due to their small amount and importance.

Definition of scenarios

To fulfil the Austrian Landfill Ordinance [4] MSW and sewage sludge have to be pre-treated before landfilling either by thermal or by mechanical-biological treatment. Thus the search for the best combination of MSW and municipal sewage sludge treatment and disposal options considering both – economic costs and fulfilment of the goals defined in Austria's AWG, has to concentrate on those two technologies mainly. Against this legal background following scenarios including different waste thermal and mechanical-biological treatment technologies were investigated. Each scenario varies in the effort for treatment and therefore differs from each other in the amount and the quality of the residues to be landfilled, too.

Table 1: Definiton of the scenarios investigated

scenario / group of scenarios	abbreviation
Up-dated status-quo (reference scenario)	P0
Maximum landfilling of untreated waste	M1
Maximum incineration without after-treatment of residues	M2a
Maximum incineration with cement stabilization of the residues	M2b
Maximum high temperature treatment	M2c
Maximum mechanical-biological treatment with the light fraction from sorting and splitting (LF) processed in a fluidized-bed furnace	M3a
Maximum mechanical-biological treatment with the light fraction from sorting and splitting (LF) processed in a cement kiln	M3b
Maximum mechanical-biological treatment with the heavy fraction of high calorific value (HF) processed in an incinerator and the LF in a fluidized-bed furnace	M3c
Maximum mechanical-biological treatment with the heavy fraction of high calorific value (HF) processed in an incinerator and the LF in a cement kiln	M3d

The input into the subsystem treatment consisted of the same amount and compound of waste for all scenarios. The subsystems collection (including transport), sorting and recovery have not been varied but adjusted to each scenario. The results of the scenarios' material and energy flow analyses are the input into the landfill model.

An updated status-quo (P0) serves as a reference scenario. This scenario describes the situation as it was in Austria in 1996: 17 % of MSW were incinerated, 15 % were treated in mechanical-biological plants and 68 % were landfilled without any pre-treatment. 34% of the municipal sewage sludge were treated in thermal plants, 13% were used in mechanical-biological plants and 31% were dehydrated and landfilled (BAWP [5]). The rest (22% used in agriculture) was not part of the system investigated.

Each of the scenarios was investigated and compared to P0. In order to oblige to the precautionary principle of the AWG comparision took place with regards to the short-, mid-, and long-term landfill behaviours of the deposited residual material.

Determination of material, substance, and energy flows and business costs

The case study is based on the system defined in Figur 2. Within the system all flows of goods, energy, money and selected emissions caused by the subsystems collection, sorting, recovery, treatment and landfilling of MSW and municipal sewage sludge were registered for one year. For that purpose a material flow analysis (MFA) for the substances C, N, S, Cl, Hg, Cd, Pb, Zn and their relevant chemical combinations as listed below was carried out using the methodology of Baccini & Brunner [6]. The amounts of the following emissions relevant to waste management have been calculated as loads [$kg \cdot a^{-1}$]: CO_2, CH_4, CO, C_xH_y (NMVOC), particulate matter, CFC, PCDD, PCDF, TOC, NO_x, NH_3, NO_3^-, NO_2^-, NH_4^+, SO_2, H_2S, SO_4^{2-}, HCl, Cl^-, Cd, Hg, Pb, Zn. Additionally the quality of the residues to be landfilled has been determined as concentrations [$mg \cdot kg^{-1}$].

For each process an energy balance was carried out. Among expenditures and returns, the substituted energy from avoiding primary production was calculated. Analogous to the handling of the substitution of energy sources the substitution of emissions was considered.

Modelling of short-, mid- and long-term landfill processes

The solid residues of the previous subsystems are assigned to different mono-landfill types. Landfill emissions were calculated for a period of 10,000 years. Each landfill is constructed according to the requirements of the Austrian Landfill Ordinance. The following simplifications were made:

1. The short-, mid-, and long-term amount of leachate is equal to the difference between the average annual precipitation rate and the average annual evapotranspiration rate for recultivated landfill surfaces.

2. The life time of landfill construction elements e.g. surface liner, base liner and man-made geological barrier is limited to 100 years. The ability of the geological barrier to bind heavy metals does not decrease during the whole period of 10,000 years.

3. The landfill is handled as a homogenous reacting block (monolith) without preferential leachate flow.

The calculation of the landfill emissions was based on schemes defining the key-reactions and changings of the physical-chemical conditions in the landfills as well as the relationship between the substances. Additionally the predominant species of the substances under different conditions was defined.

Gaseous (CH_4, CO_2) and liquid (TOC) carbon emissions from organic landfills were calculated applying the model introduced by Marticorena et al. [7]. During the intensive methane-production phase gaseous emissions of N, S, Cl and Hg-combinations were determined, too.

Emissions of other non-metals (N, S and Cl) and calcium were calculated using the model of Belevi & Baccini [8]. This model describes the discharge of these elements' combinations using a first-order kinetics. The resulting concentrations [$mg \cdot l^{-1}$] were transferred into loads [$kg \cdot a^{-1}$] by applying the simplified water balance described above. Liquid carbon-emissions (TOC) from inorganic landfills were calculated in the same way.

The emitted amount of heavy metals (Pb, Zn, Cd, Hg) was supposed to be constant in time depending on the predominant physical-chemical conditions only. Concentration values were estimated using data from operating landfills and natural analogues, such as peat deposits and mining waste tailings.

The dilution of leachate emissions in a hydrogeologically exactly defined aquifer is calculated. The accumulation of heavy metals in the landfill's sub soil after failure of the technical barrier (base liner) was considered as well. The results were used for evaluating remediation measures. Costs and time for the landfill remediation were estimated, as well as costs for design, construction, operation, and maintenance of the modelled landfills.

Assessment of ecological impacts and costs

Two assessment methods were used: cost-benefit analysis (CBA) and the within this study developed "modified cost-effectiveness analysis" (MCEA). Unlike the classical CBA, the MCEA enabled the long-term impacts of the deposited residual material to be evaluated in an appropiate way. The MCEA is based on a goal-hierarchy system, where the goals of the AWG (protection of humans and the environment, protection of resources[19] and maintenance-free landfills[20]) are situated at the top level. To consider the societal preferences, each one of the AWG goals awarded a specific weight by the clients of the study. Within the framework of the sensitivity analysis the influence of weighting was investigated.

To be able to assess the abstract goals of the AWG, it is necessary to find measurable goals. Therefore, on the lowest level of the hierarchy of targets are those, which are valuated using integrative target criteria. On this level a total amount of 110 targets were defined and valuated using an integrative target criterion, which is based on a scientifically measurable value. The integrative target criteria, which came into operation were: greenhouse-potential, ozone depletion potential, critical air, water and soil volumina, area used for landfills, statistical availability for raw materials, energy

[19] Including the resource „area"

[20] no endangering of future generations – precautionary principle

amounts and substance concentrating efficiency. For each integrative target criterion a desired value[21] was defined and the level of target achievement within each scenario was determined. Hence the target revenue relating to the reference scenario (P0) is calculated and then transformed into the effectiveness value (see Figur 3).

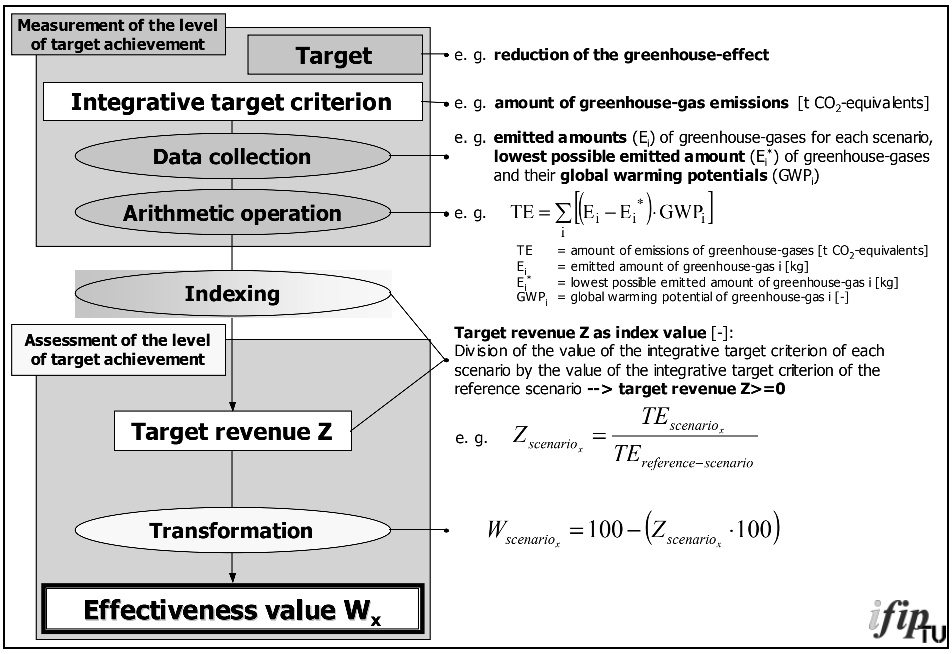

Figure 3: Methodology of measuring the efficiency in MCEA

The effectiveness values within a single scenario and the reference scenario are weighted with the correspondent average weighting factor provided by the study customers and then aggregated to form the weighted total effectiveness value. The costs of each scenario were related to the costs of the reference scenario and thus standardised. In a last step the standardised costs and the weighted total effectiveness values were used to calculate the total effectiveness-costs ratio for each scenario (Figur 4). This ratio finally was used to rank the different scenarios.

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[21] The value, which can be reached by orientation of the waste management system just to this one target

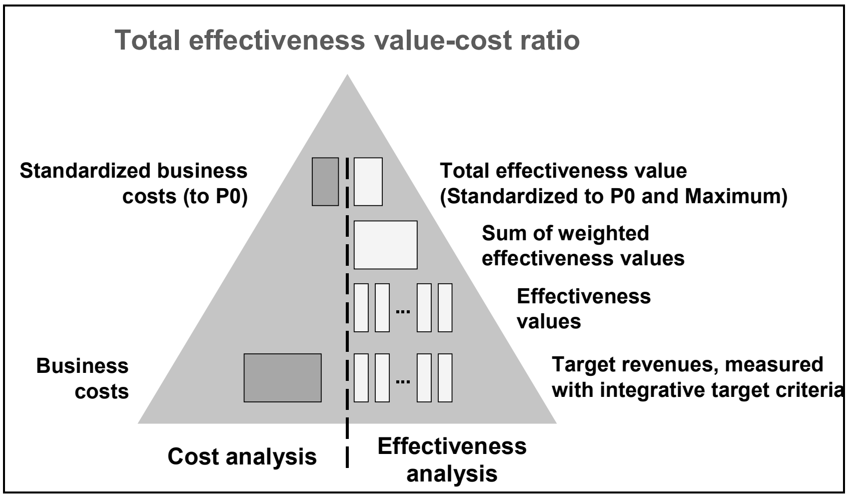

Figure 4: Developement of the total effectiveness value-cost ratio in MCEA

Results

The results of the MCEA articulated, independent of the weighting given by the AWG goals, the scenarios which belong to the incineration option (M2c before M2a and M2b) as the best waste management solutions, whereas the worst rating was given to direct landfilling (M1). In comparison to the reference scenario (P0), scenarios M3a and M3b were just a little better, while M3c and M3d were noticeably better.

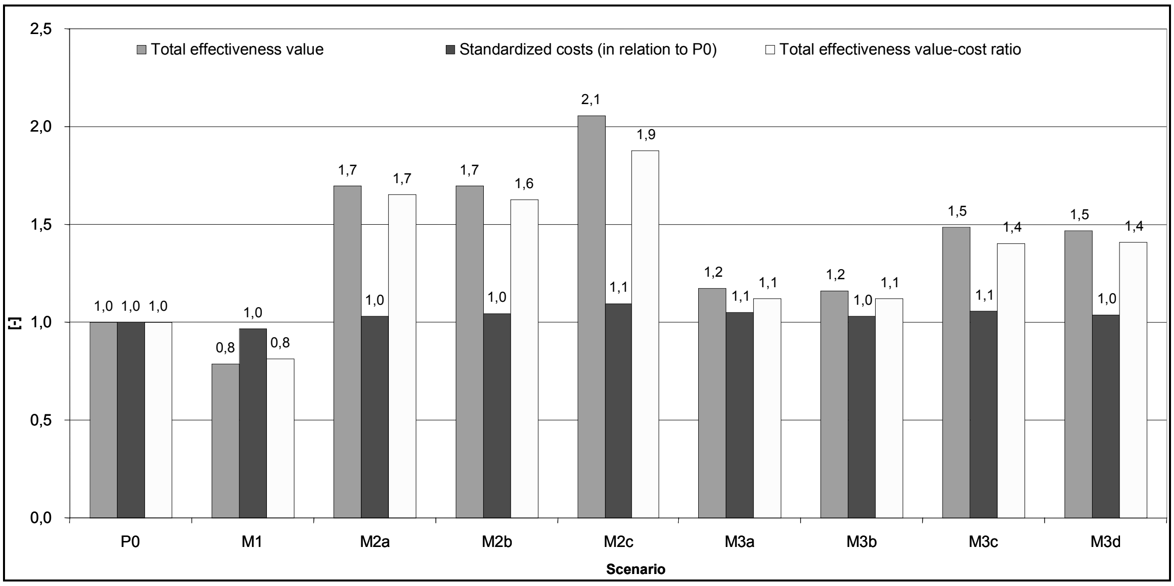

Diagram 1: Total effectiveness value, standardized costs and total effectiveness value cost-ratio of the scenarios investigated in comparison

The results obtained by use of the CBA concurred with those of the MCEA. The M2 scenario group was rated the best, followed by the M3 group, the reference (P0), and finally by M1. However, within each of the scenario groups, the internal ratings deviated compared to those of the MCEA. Overall, the scenario M2a showed the smallest macro-economical loss of all of the scenarios.

Table 2: Ranking of the scenarios, based on the results of MCEA and CBA

Scenario	**P0**	**M1**	**M2a**	**M2b**	**M2c**	**M3a**	**M3b**	**M3c**	**M3d**
Ranking based on MCEA	8	9	2	3	1	7	6	5	4
Ranking based on cost-benefit-balance	8	9	1	2	3	7	6	5	4
Ranking based on benefit-cost-ratio	7	9	1	2	3	8	6	5	4

Conclusions

Although the business costs for landfilling of untreated waste (M1) are lowest its performance in CBA and MCEA was poor in comparison to all other scenarios, which have been assessed. It can be demonstrated clearly that the closer landfilled goods come to final storage quality the better the scenario performs in total economic assessment. This is due to the low landfill emissions of these goods during mid and long-term periods. A reduction of reactivity of landfill material corresponds directly with a reduction of total economic costs. Therefore thermal treatment options, such as incineration and high-temperature treatment, can be rated better than mechanical-biological treatment options leaving behind landfill material with higher TOC-values and thus higher reactivity. The ranking of the scenarios does not depend on the time period investigated (years, centuries, millenniums) but is stable in all cases. The longer the period observed the more distinct the advantages of thermal treatment become.

In both economic assessments thermal treatment options show significant better results than mechanical-biological treatment of MSW and sewage sludge, which again was ranked better than the reference scenario. The more extensive thermal treatment of separated fractions during mechanical-biological treatment is done the better the scenario performs in the economic assessment (M3c and M3d better than M3a and M3b). The comparison of treating the separated light fraction in a fluidized-bed furnace (M3a and M3c) and in a rotary cement kiln (M3b and M3d) shows no significant difference. Because of considerations taking into account substance flow politics the fluidized-bed furnace option should be given priority over the cement kiln due to the dissipation of pollutants caused by insufficient offgas-cleaning technologies in cement producing plants and not controllable emissions from secondary cement products such as concrete.

Although both methods, CBA and MCEA, lead to the same main results, i. e. giving preference to thermal waste treatment, it cannot be suggested that both methods are equivalent. Due to the fact that CBA includes only internal and external effects, for which a monetary valuation is possible, some relevant (mostly negative external) effects remain excluded from the analysis.

The main purpose of CBA is to help selecting projects and policies, which are efficient in terms of their use of resources. Not only costs and benefits valued with market prices but also goods valued implicitly by individuals are included. If it can be achieved that all cost and benefit streams, which are induced by a project, are valued monetarily, CBA is a very suitable tool to give recommendations, whether a project should be realized. If the benefits induced by a project exceed its costs, the project is to be assessed positively.

As an alternative methods can be seen which are suitable to assess a project on a non-monetary base. The MCEA, which was developed from the classical CEA by the authors, is such a method. Based on the main targets, defined in the Austrian Waste Management Act, sub-targets were developed. These sub-targets can be measured according to scientific methods. In this way there is guaranteed a maximum of attainable objectivity on the lowest level of targets. The use of scientifically reasonable quantities being measured gives further the possibility of an aggregation of sub-targets on the superordinate level of targets. By using MCEA it could be succeeded that almost all effects, which remained intangible and thus excluded from CBA, especially those connected with influence to the environment, were assessed.

MCEA does not only enable the inclusion of monetarily not valuable effects. Furthermore it represents a better decision basis for decision makers than the „classical" CEA, which leaves the politicians alone with (at least in this study) about 110 efficiency values on the lowest target level.

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Independent Assessment of Kerbside Recycling in Australia

Hannes Partl *and Leanne Philpott* [22]

Keywords: Kerbside recycling, LCA, cost-benefit analysis, Australia.

Background

The National Packaging Covenant Council commissioned Nolan-ITU Pty Ltd in association with SKM Economics to undertake an *Independent Economic Assessment of Kerbside Collection and Recycling Systems for Used Packaging Materials in Australia*. The purpose of the study is to assess the net costs and benefits of kerbside collection and recycling systems and their viability, and to provide an improved framework for transparent decision making on sound financial, environmental and social bases.

The National Packaging Covenant was signed by government and industry representatives on 27 August 1999. The Covenant is heralded as a landmark agreement to foster efficient and environmentally sustainable systems for managing used packaging materials.

The need for the study is supported by Commonwealth, State and Local Governments as well as a wide range of industries in the packaging supply chain represented by bodies such as the Australian Food and Grocery Council, the Australian Supermarket Institute, the Beverage Industry Environment Council, the Packaging Council of Australia and the Plastics and Chemicals Industry Association.

Assessment and Approach

The net costs and benefits of kerbside collection and recycling systems have been assessed across the range of different collection systems for metropolitan and regional areas in each state and territory of Australia. In addition, a selection of alternatives to

[22] Nolan-ITU Pty Ltd, Suite 4, 11 Victoria Parade, Manly NSW 2095, Australia, Telephone: +61 2 9976 5411 Fax: +61 2 9976 5422, E-mail: hpartl@nolanitu.com.au, lphilpott@nolanitu.com.au

current collection and recycling systems for used packaging materials have been selected and assessed, on the basis of available technology and industry interest.

Information was collected and collated from about 200 Councils across the country, representing 12 million people, or two thirds of the Australian population.

The use of integrated financial, environmental and social cost benefit assessment in this study allows the actual costs and benefits to be assessed for:

- the cost implications of the systems in operation, with their varying yields.
- the different environmental impacts based on system yields.

Financial Assessment of Current Systems

The financial cost analysis has used an adapted form of the Australian Waste and Recycling Cost Model. The key findings from the financial analysis are:

- The net, or additional cost for both metropolitan and regional systems, compared to landfilling, varies between $17 and $38 per household per year, or 33 cents to 73 cents per household per week.
- For individual systems, the range in costs is much wider in regional areas than metropolitan areas.
- The financial benefit from avoided garbage costs attributable to recycling currently averages around $10 per household per year in metropolitan areas and $3 per household per year in regional areas. The variation is wider across individual systems due to the different system yields, and disposal costs.

Environmental Assessment

Methodology

The overarching methodology used for the study is Cost Benefit Analysis[23]. The environmental assessment component has sought to identify and value the environmental externalities (or non-financial costs) of collection and recycling systems so that they may be incorporated into the integrated economic assessment.

Life Cycle Assessment (LCA) and Environmental Economics have been applied as required within this framework. The study has only used those features of LCA and

[23] Commonwealth Department of Finance (1994) Cost Benefit Analysis – Guidelines and NSW Treasury (June 1997) NSW Government Guidelines for Economic Evaluation.

Environmental Economics which are necessary to meet the study requirements. The environmental assessment has included goal and scope definition, selection and application of life cycle assessment data and environmental economic valuation of impacts.

The Life Cycle Assessment component of the study included modelling of more than 50 substances – resource inputs and pollutant outputs for each aspect of the collection and recycling system. Consideration was given to both the kerbside system and the product system.

The commercial software tool, the Integrated Solid Waste Management Model[24], was used to apply Life Cycle Assessment (LCA) data to the systems studied. Once the LCA data was modelled for each system under study, it was aggregated into environmental impact categories and then valued by applying environmental economic benefit assessment techniques based on published Australian government references.

Results

The analysis indicates that the average national environmental benefit of current kerbside collection and recycling systems in metropolitan and regional centres is conservatively estimated to be $68 per household per year (between $41 and $119 depending on the system and location). Based on the analysis, the total national environmental benefit of kerbside recycling is estimated to be in the order of $424 million per year.

The environmental impact categories which contribute to the overall benefit of current collection and recycling systems are presented in Figure 3. The majority of the impact - 75 percent, comes from *air and water pollution* credits arising from the avoided product system associated with the avoided manufacture from virgin materials. The *natural resource* value of recycling is the next most influential factor at 21 percent of the benefit. This is followed by *global warming* credits, valued at 4 percent, and *landfill* savings at 1.6 percent. *Traffic (Noise and Traffic)* represents a net environmental cost to the system of 2 percent. All impact categories represent the balance of the marginal net collection and recycling system –ie: waste collection, transport, sorting, landfill and recycling. The main system components of the environmental benefit are detailed Table 1.

[24] White, P.R., Franke, M., Hindle, P., (1995) *Integrated Solid Waste Management – A Lifecycle Inventory*

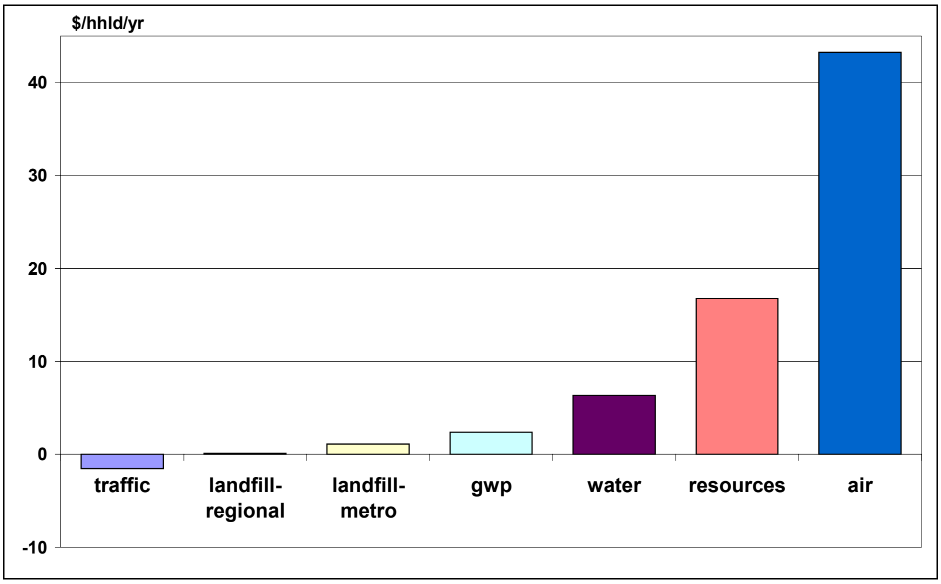

Figure 1. Environmental Costs and Benefits of Kerbside Recycling by Impact Category ($ per household per year - Population Weighted National Average)

Table 1. Environmental Valuation of Current Kerbside Recycling – National Averages

System	System Description	Net Value
Recycling	The *Recycling System* includes: Avoided Product Credits (raw materials extraction and manufacture); Avoided Landfill (non-chemical impacts only); and Transportation (based on average distances for each state and material).	+ $ 71
Collection	The *Collection System* is: The impact arising from the collection system including use of collection vehicles for waste and recyclables, transfer to landfill or waste facility and sorting and bulking.	- $ 3
Balance		+ $ 68
NB: All results are calculated for the marginal effect of recycling – garbage plus recycling less garbage only. To simplify the results, the environmental assessment is reported as an average monetary value. This approach is not appropriate for all decision making and caution should be taken in referencing the results for any other purpose. Similarly, the findings are dependant of variables which are specific to the system studied (such as processing yield and local waste practices). The results do not apply to all recycling programs as local performance variables may be crucial.		

The findings also shed new light on the relative significance of landfill and landfill savings as an environmental issue associated with recycling. The environmental benefit of the landfill savings[25] is valued at less than 2 percent of the overall system benefit. In the past, "landfill savings" have been heralded as a key environmental motive for recycling. The relatively low contribution of landfill savings to the net system benefit serves to highlight the magnitude of other environmental benefits which are not generally recognised – namely the avoided impacts associated with resource extraction, refining and manufacture for virgin materials.

To test the robustness of the outcomes, data sensitivity analyses have been conducted on the most sensitive and subjective variables. The main results are as follows:

- Between 92 and 95% of the net environmental benefit associated with air pollution comes from the pollutants for which the values are based directly on published Australian government cost–benefit valuations.
- Using the "low" values of a highly acclaimed overseas air pollutant valuation study, the net environmental benefit of current systems increased from \$68 to \$97 per household per year.
- The adoption of a *zero* value for forest resources reduces the net environmental benefit by 6% from \$68 to \$64 per household per year. Forest valuation is the least certain value used.
- The adoption of the only published Australian value for forest resources changes the national average value of recycling from \$68 to \$96 per household per year.

A key finding is that recycling yields are the single most important factor in the environmental performance of the system. The higher the yield, the higher the benefit to the recycling system.

Social Findings

Whilst there is a wealth of information on how communities regard recycling activities, the social impact assessment of kerbside systems in comparison with alternatives is severely constrained due to:

- Community survey information on economic and environmental impacts is currently perception-based rather than impact-based; and
- There is little social impact information on alternatives to kerbside recycling.

The social impact assessment is therefore restricted to the examination of the key issues which have been raised through available information rather than a detailed social impact analysis. There is also a lack of community knowledge about the true

[25] This value for *Landfill Savings* includes only the aesthetic and land impact values as the recyclables are defined for the purpose of this study as "inert" and held to have no impact on landfill emissions.

environmental benefits of recycling and alternatives to landfill disposal and kerbside collection and recycling.

The combining of financial and environmental costs and benefits in this study is an important step in providing the necessary information to the community to enable a more comprehensive community surveying program from which a detailed social impact assessment can be undertaken.

Occupational Health and Safety

Current and likely future occupational health and safety requirements are discussed and have been considered in the financial assessment of systems.

Materials & Markets

A total of over 800 000 tonnes of kerbside collected recyclate is reprocessed annually in Australia. 92% (by weight) is made up by paper, cardboard and glass. These materials contribute around three quarters of the revenue. Prices, material flows and trends have been documented.

Impact of Changes in Current Systems

The impact of changes in the current kerbside collection and recycling systems have been assessed. These are described below.

Higher Yields

Yields are by far the single most important factor in kerbside collection and recycling system performance with higher yields resulting in increased overall benefits.

Net costs per household fall slightly due to the increased material revenue and avoided garbage costs rising at a faster rate than the rise in system costs. This result does not hold for extremely high yield ranges. Where the gain from system efficiency slows, the net cost per household may begin to rise again.

High yield scenarios have been benchmarked against current recycling systems to highlight the potential environmental gains from increasing the yield of current systems. The average increase in environmental benefits for higher yields is in the order of $25 per household per year or 35% for the systems analysed.

Change in Mix – More PET, less Glass

A significant change of the packaging mix from glass to PET may range from cost neutral to slight savings for the average household. The environmental impacts have not been modelled.

Change in Mix – Paper and Glass Collection only

The net financial costs of providing a collection service for paper and glass only are around 36% lower than current collection systems which involve collection of a broad mix of material. The net environmental benefit of a paper and glass only collection are around 25% lower than a collection service for the full mix.

The overall benefits of a paper and glass only collection are around 22% ($16/hhld/yr) lower than a collection service for the full mix.

Energy Recovery from Source-Separated Paper and Plastics

The "Energy Recovery from Paper and Plastics" alternative involves 'conventional' kerbside collection with subsequent use of paper (including cardboard and liquidpaperboard) and all plastics as a fuel. The average net financial cost of the four kerbside systems modelled is approximately $5 per household per year higher than current reprocessing of these materials.

The environmental assessment of this option shows a reduction in the net environmental benefit of the system. The changes arise from new impacts associated with thermal processing of the material and from the reduced resource credits. The overall benefit of such a system is therefore significantly lower than for 'conventional' reprocessing.

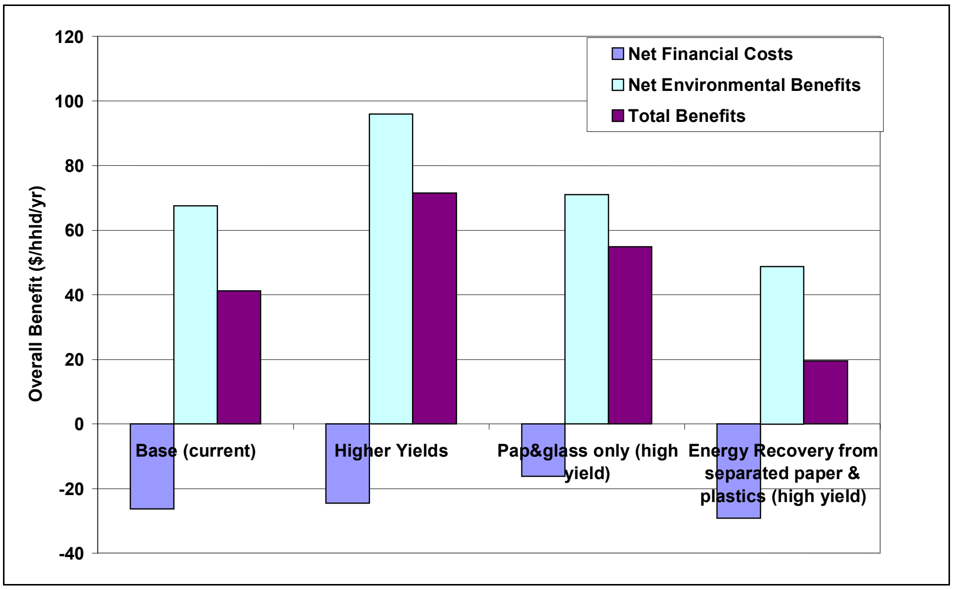

Figure 2. Comparison of Changes to Current Systems

Assessment of Alternative Recovery Methods

Mechanical-Biological Treatment (MBT) of Total Waste

This alternative involves collection of garbage with recyclables in a single bin with subsequent treatment of waste prior to disposal to reduce environmental impacts from the landfill. The financial costs for this alternative in metropolitan centres are on average $30 per household per year higher than for the average current waste and recycling system. This is largely dependant on the cost of landfill. Whilst there is no direct environmental benefit from MBT processing of recyclables in the combined stream, the environmental benefits of the MBT can result in landfill savings from garbage of between $50 and $250 per household per year. The net environmental benefits of MBT reduce as landfill standards improve.

Although MBT is not an *alternative* to kerbside recycling from a financial and environmental perspective, it offers benefits as an additional measure for improved management of the residual waste stream where the community is willing to pay for the extra service.

Waste to Energy (WtE) of Total Waste

This alternative involves collection of garbage with recyclables in a single bin with subsequent thermal treatment and energy recovery. Under the WtE alternative, the financial cost per household per year would increase by more than $70 from current system costs (garbage and recycling). The environmental performance of WTE technologies varies depending on the technology and configuration deployed. More work is required before a conclusion can be drawn to compare the net performance of total-waste to energy technologies with kerbside recycling, as part of an integrated waste management system.

Conclusions and Recommendations

The integrated cost-benefit assessment confirms a common perception that the current kerbside system in metropolitan and regional centres provides a total net benefit to Australian communities.

When combining the financial costs of kerbside systems with the environmental benefits (which have been estimated using conservative environmental values) it is clear that practically all current systems provide a significant net benefit to Australian communities. On average, net financial costs amount to $26 per household per year, environmental benefits to $68 per household per year, with an average overall benefit of around $42 per household per year.

Based on extrapolation, the national net financial cost for recyclables collection, sorting and delivery throughout Australia is estimated at $158 million per year or, if current collection practices (double siding) are included, $136 million. This represents the current cost over and above the base landfill option. The national net environmental benefit of kerbside recycling (over landfill) is $424 million dollars per year. The overall benefit is therefore an estimated $266 million per year.

The net benefit to society may be further improved by increasing yields and reducing contamination (within the capacity of the collection systems and the range of materials currently collected).

Obtaining the highest resource value utilisation is the key to economic and environmental sustainability. This is already present in current practice and should be maintained and possibly improved in the future.

Based on the findings of this study, there are a number of readily identifiable actions which would improve the financial, environmental and social performance of the current kerbside recycling systems. These include recommendations regarding preferred collection systems.

More importantly, a framework for sustainability is suggested on the basis of the report findings and the technical and socio-political context in which kerbside recycling systems in Australia operate. The recommendations seek to provide options for a more sustainable, viable market based recycling system as sought by the transitional arrangements of the National Packaging Covenant. The framework addresses the goal of the National Packaging Covenant “to minimise the environmental impacts of consumer packaging waste throughout the life cycle of packaging. In brief, these include:

- Integrating community values in decision making
- Addressing research needs
- Understanding the material flows of the net waste system
- Ongoing market development
- Incorporation of environmental externalities
- Product stewardship and product policy
- Reduction of direct financial subsidies to virgin material extraction

Integrated approach for formulating and comparing separation strategies of MSW

Juha-Heikki Tanskanen[26]

Abstract

An approach was developed and applied for the integrated analysis of recovery rates, waste streams, costs and emissions of municipal solid waste (MSW) management. The approach differs from most earlier models used in the strategic planning of MSW management because of a comprehensive analysis of on-site collection systems of waste materials separated at source for recovery. As a result, the recovery rates and sizes of waste streams can be calculated on the basis of the characteristics of separation strategies instead of giving them as input data. The approach was applied in three case studies where it proved to be a useful tool for strategic planning of MSW management. The method developed is generally applicable to all regions and municipalities.

Key words: municipal solid waste, separation, costs, emissions, models

Introduction

In the European Union, several regulations have been made during the recent years to promote prevention and recovery of wastes. In the management of municipal solid waste (MSW), the feasibility of high recovery levels depends on the approach applied. In countries in which incineration is an essential part of waste management systems (e.g. Belgium, Denmark, France, Germany, Luxembourg, the Netherlands and Sweden) comparatively high recovery levels can be reached with moderate separation strategies if the remaining mixed waste is incinerated for energy recovery. On the other hand, there are member states like Finland, Greece, Italy, Ireland, Portugal, Spain and the United Kingdom in which incineration is of minor importance. In these countries, high recovery levels are far more difficult to reach and implementation of highly efficient separation strategies is of vital importance.

In Finland, municipalities are trying to achieve high recovery levels of MSW mainly based on source separation and co-operation. According to Finland's National Waste Plan, 70 %wt of MSW should be recovered in the year 2005 (Ministry of the Environment, Finland [1]). Between the years 1994 and 1999 the recovery rate was

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[26] Finnish Environment Institute, P.O.Box 140, FIN-00251 Helsinki, Finland, phone: +358 9 4030 0421, fax: +358 9 4030 0491, e-mail: Juha-Heikki.Tanskanen@vyh.fi

raised from 30 %wt to 40 %wt. Thus, the share of MSW recovered should be raised by 30 %wt-units between the years 1999 and 2005.

Tanskanen [2] developed and tested an approach for formulating and comparing separation strategies of MSW management. The approach is of the same kind as many other strategic planning models developed in the 1990s including both cost and emissions of recovery-based MSW management. However, the approach differs from most earlier models on the basis of more detailed analysis of on-site collection systems applied for waste materials.

Materials and methods

Method and models developed

An approach for formulating and comparing waste management systems

The approach developed by Tanskanen [2] consists of six stages and includes formulation, analysis and comparison of MSW management systems (Fig. 1). A fundamental part of the approach is the analysis of the coverages[27] of on-site collection systems and the corresponding accumulations of waste materials at the properties (Fig. 2). The coverages are needed at the second stage of the approach to determine the recovery rates which can be reached with various separation strategies. The accumulations of waste materials are used at the fourth and fifth stages to calculate the unit costs and unit emissions of waste collection. Separation strategies can be modified after the second stage if the recovery levels are too low. After the final stage, alternative waste management systems can be created by modifying the collection systems and the separation strategies. The aim of these modifications may be the reduction of the total costs and emissions.

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[1]Coverage of on-site collection in an area is the ratio of (a) the amount of a material produced in those properties in which separate collection is available and (b) the amount of the material in question produced in all properties of the area.

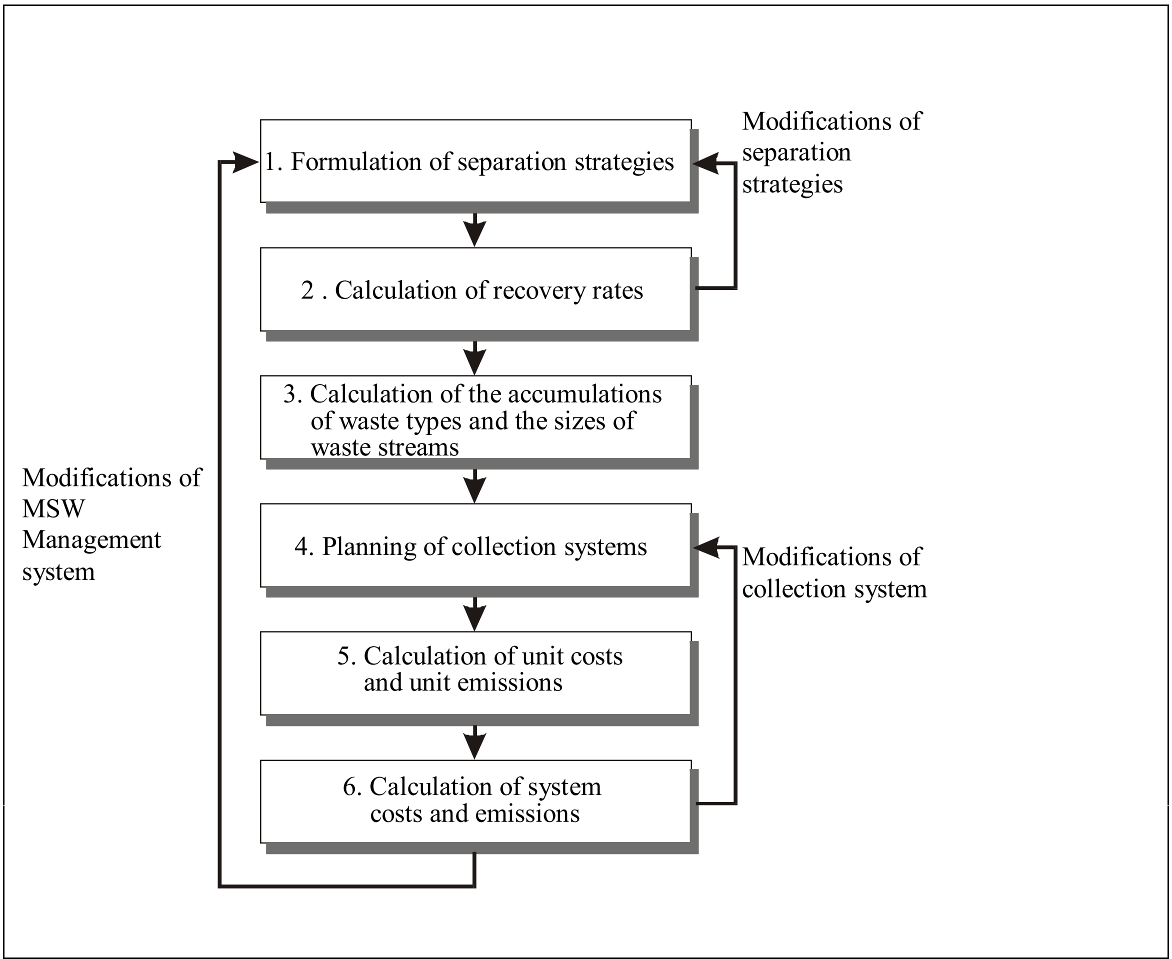

Figure 1. Stages of the modelling approach developed.

Calculation of the coverages is based on the fact that large properties are usually obliged to participate in on-site collection of recoverable materials before smaller ones. Thus, the coverages of on-site collection systems can be determined on the basis of the size distribution of properties. In Finland, the minimum size of a property obliged to participate in on-site collection of a material, termed on-site obligation limit, is determined on the basis of the number of households in residential properties and on the basis of the amount of a material produced in commercial establishments.

Applications of the approach

The approach was tested in three case studies by Tanskanen [2]. Firstly, the TASAR model was constructed to study Finland's national separation strategy (Fig. 3). The study included all the Finnish municipalities (452 in 1995). Secondly, the HMA model was constructed to analyse recovery rates, costs and emissions of MSW management in the Helsinki region (Table 1). Thirdly, the efficiency of alternative waste collection methods was compared in the Helsinki region. The models constructed for the case studies were static and linear simulation models in the format of Excel spreadsheets.

The main questions to be answered in the case studies were as follows: What kind of strategies are needed in Finland to reach the recovery target of 70 %wt? Is central sorting of mixed waste needed? Is incineration of mixed waste needed? How would the implementation of these strategies affect costs and emissions of MSW management?

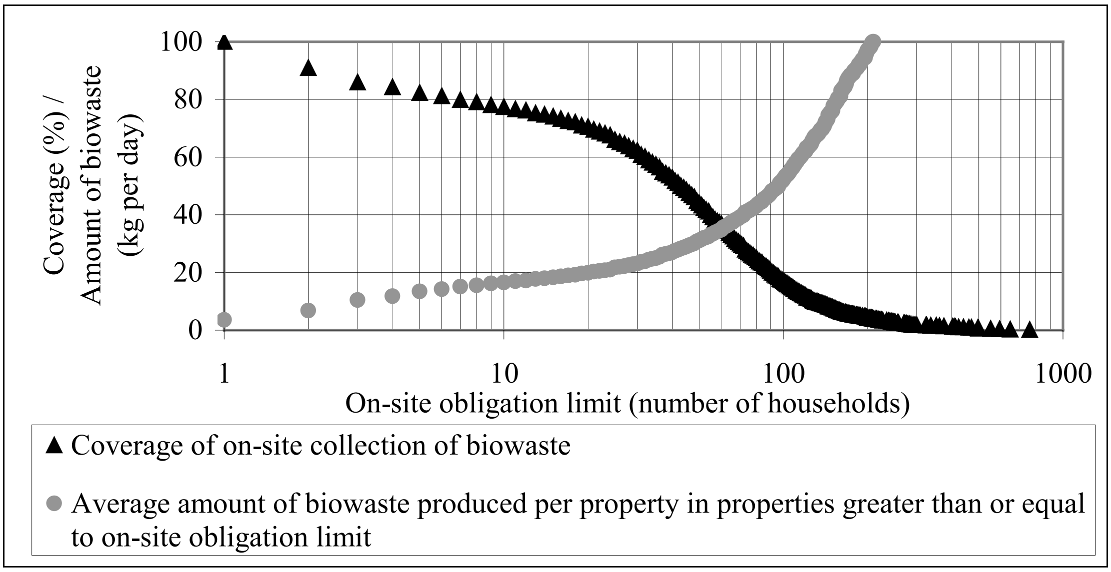

Figure 2. The coverage of on-site collection of biowaste and average amount of biowaste generated per property in residential properties greater than or equal to the on-site obligation limit in the Helsinki region. The analysis included 50 200 residential properties. On-site obligation limit is the minimum size of a property obliged to participate in on-site collection of a material in an area.

Input data

The input data used in the case studies were based mainly on statistical data, empirical data and earlier studies from the example areas (Tanskanen [2]). In addition, earlier studies from other areas and estimates made on the basis of the statements of experts were applied.

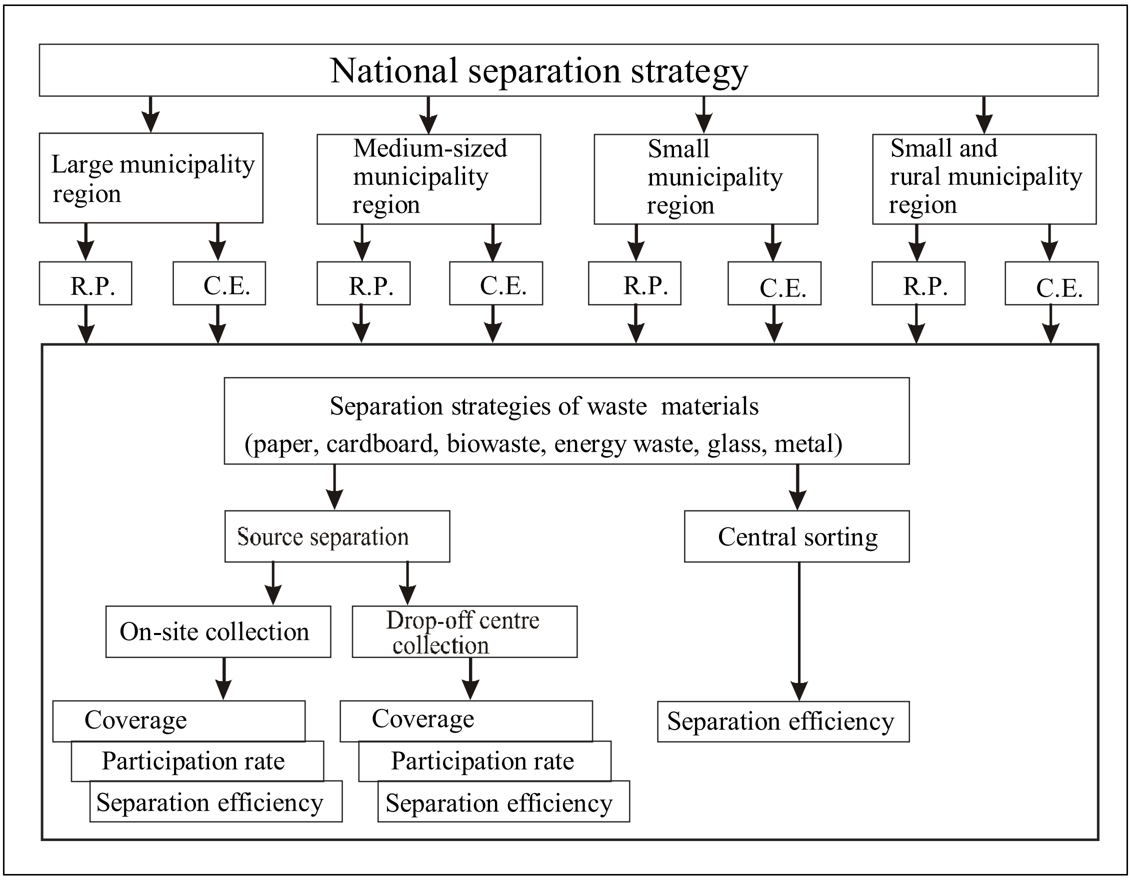

Figure 3. Elements from which a national separation strategy can be compiled in the TASAR model (R.P.=residential properties, C.E.=commercial establishments).

Results

Maximal recovery rates

Source separation strategies

In the national study, feasible source separation strategies resulted maximally in recovery rates around 35-40 %wt with the present separation activity of waste producers (i.e. 20-70 %wt). In these strategies, all recoverable materials were separated and collected on-site from residential properties bigger than 2-10 households and from commercial establishments producing recoverable materials at a rate of more than 20-50 kg per week. Drop-off centre collection was applied for detached houses and small terraced houses.

In the Helsinki study, the maximal feasible recovery rate of 52 %wt was reached with the present separation activity (i.e. 20-75 %wt) and the following separation strategy:

- Paper, biowaste and energy waste (energy waste includes e.g. plastics and packages made of mixed fibre and plastic materials) were collected on-site from all residential properties. In addition, glass and metal were collected as drop-off centre collection.

- Paper, cardboard, biowaste, energy waste, glass and metal were collected on-site from commercial establishments producing at least 20 kg per week of a given material.

The total recovery rate increased from 52 %wt to 66 %wt when the estimates of the highest reachable separation activities (i.e. 50 - 90 %wt) were used as input data.

Table 1. Functional elements, costs and emission components of MSW management included in the HMA model.

Functional element	Costs	Emission components
Waste collection	Yes	CO_2, NO_X, SO_2, VOCs
• bins and containers at the properties		
• containers at drop-off centres		
• structures of collection points		
• collection work at the collection area		
• transportation		
Transfer station	Yes	
Backyard composting	Yes	CO_2, CH_4, N_2O, NH_3, VOCs
Central composting	Yes	COD, CO_2, CH_4, N_2O, NH_3, NO_X[1], NH_4, SO_2[1], VOCs
Processing of source-separated energy waste	Yes	
Central sorting and processing of mixed waste	Yes	
Landfilling	Yes	
• decomposition of waste		COD, NH_4, CO_2, CH_4, VOCs
• landfill compactors		CO_2, NO_X, SO_2, VOCs
• recovery of landfill gas		CO_2, NO_X, SO_2,VOCs
Waste tax	Yes	
Revenues from recovered materials	Yes	

[1] Emissions from the production of energy needed in composting.

Source separation combined with central sorting

Both in the national study and in the Helsinki study a large share of remaining mixed waste consisted of recoverable materials despite extensive source separation strategies. As a result, source separation strategies were next complemented with central sorting of combustible components of mixed waste for energy recovery with the efficiency of 90 %wt. In the national study, the total recovery rate increased up to 65-80 %wt depending on the strategy in question. In the Helsinki study, the total recovery rate increased from 52 %wt to 74 %wt.

Costs

The total costs of MSM management increased by 41 % in the Helsinki study when the total recovery rate was increased from 27 %wt to 66 %wt based on source separation. By combining source separation with central sorting, the increase in total costs was 30 % and the total recovery rate achieved was 74 %wt. The most important reason for the growth of total costs was separate collection of waste materials from residential properties and commercial establishments. However, the increase in total costs clearly diminished when simultaneous collection was applied instead of separate collection. In the basic strategy, the total recovery rate was 27 %wt and the costs of MSW management were 41.4 million Euros (79.3 Euros per waste tonne and 46.5 Euros per inhabitant).

Emissions

In the Helsinki study, the separation strategies resulted in the following reductions in the combined emissions of collection, transportation, composting and landfilling: nutrient load 23-28 %, greenhouse gas load 37-53 % and ozone formation 17-33 %. On the other hand, the amount of acid load increased by 115-125 %. The reason for the reduction in the amount of emissions was decreased amount of waste disposed of to the landfill. The amount of emissions diminished despite the fact that the emissions of waste collection increased by 16-30 %. The total acid load increased because less landfill gas was available for energy production to replace fossil fuels.

Discussion

System boundaries

The approach developed by Tanskanen [2] differs from most earlier models used in the strategic planning of MSW management because of more detailed analysis of on-site collection systems. As a result, the recovery rates and the corresponding unit costs and unit emissions of waste collection can be calculated on the basis of the coverages of collection systems. In most earlier models, the amounts (or the ranges of the amounts) of materials separately collected and the corresponding unit costs and emissions of waste collection are treated as input data (Sundberg [3], Ljunggren [4]). In some models, recovery rates are calculated on the basis of participation rates and separation efficiencies but the analysis of the coverages of collection systems has been excluded from these models too (Anex et al. [5], Everett and Modak [6]).

A weak point in the approach presented by Tanskanen [2], as well as in several other models, is the description of emissions. The system boundaries used exclude emissions outside the waste management system, e.g. effects of recycling on raw material acquisition and production processes. To identify emissions comprehensively system boundaries should be extended to cover the life cycles of products in which waste materials are utilized.

Validation of the approach, results and input data

The approach adopted has been applied in four case studies with consistent results (Tanskanen [7], Tanskanen [2]). In addition, the MIMES/Waste Finland model was applied in the Tampere region with very similar results (Tanskanen et al. [8]). Thus, the approach would appear to function logically. Detailed verification of the results is difficult because there are no reliable data available on the total costs and total emissions of MSW management in the Helsinki region. The only reliable data available is the total recovery rate, which was 28 %wt in 1995 (Helsinki Metropolitan Area Council [9]). The recovery rate obtained for the year 1995 with the HMA model was 27 %wt. Uncertainty and sensitivity analyses were included in every case study in which the approach was applied. These analyses also proved that the approach developed is a reliable tool for formulating and comparing separation strategies of MSW management (Tanskanen [7], Tanskanen and Melanen [10], Tanskanen [11], Tanskanen and Kaila [12]).

Comparison between the results of various models

The results of individual case studies performed with various waste management models are difficult to compare. This is because the characteristics of study areas (e.g. waste management systems applied and size distribution of properties) and definitions of the case studies (e.g. study questions, system boundaries and strategies applied) vary greatly. For example, Sundberg [3] reported that composting would be a cost-effective alternative in the Gothenburg region in Sweden because it releases incineration capacity. Instead, Tanskanen [2], reported that separation measures would increase the costs of MSW management in the Helsinki region where incineration is not applied.

Conclusions

The approach developed by Tanskanen [2] proved to be a useful tool for the strategic planning of MSW management in various study areas. The most important advantage of the approach is that separation strategies can be formulated and compared with a reasonable amount of work. The users of the results (i.e. Helsinki Metropolitan Area

Council and the Ministry of the Environment, Finland) also found the approach a valuable tool in strategic planning.

A high recovery rate level (around 70 %wt) of MSW can be achieved in Finland without incineration of mixed waste. However, central sorting and co-incineration of centrally sorted waste components must be included in the national strategy. As a result, the costs of MSW management systems will increase around 30-40 %. The increase in total costs can be reduced by using simultaneous collection of several waste types instead of separate collection. Separation reduces most emissions caused by MSW management.

The results obtained by Tanskanen [2] do not reveal the effects of separation on the total amounts of emissions because the system boundaries were not broad enough to take emissions outside MSWM system into consideration. However, the Finnish Environment Institute is starting a new research project to integrate waste management modelling with life cycle assessment (LCA) of products and materials. The Helsinki Metropolitan Area will again serve as an example area and newspaper will be used as an example product. The study will be conducted between the years 2001 and 2004.

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Assessing external and indirect costs and benefits of recycling

Tomas Ekvall[28] *and Petra Bäckman*[29]

Abstract

Cost-benefit analyses of waste management options are integrated assessments of economic costs and environmental aspects. They often also include the time spent in the households on source separation. A methodology that was developed at CIT Ekologik, for assessing the costs of source-separation time, takes into account the indirect internal cost, in the form of lost production, as well as the external cost, which is estimated as the willingness to pay to avoid spending spare time on source separation. The external, environmental costs can be estimated through subtracting the estimated, internalised costs from the total environmental costs. These, in turn were estimated through a life cycle inventory analysis and the parallel use of three weighting methods. Our case studies confirm that both the costs of household spare time and the external, environmental costs can dominate the total cost-benefit results. The great uncertainties in the studies can probably be reduced through an extended study.

Key words: cost-benefit analysis, recycling, waste management, methodology

Introduction

During the past few years, several cost-benefit analyses (CBAs) have been performed to compare different waste management options in Sweden (e.g., [1-3]) as well as other countries (e.g., [4-7]). These studies are integrated assessments of various economic costs and environmental aspects. They often also take into account the time spent in the households on source separation. Two of the studies - by Bruvoll [5] and Radetzki [3] - received large attention in Sweden through, e.g., the national television. However, these studies have severe limitations. The Bruvoll study compared different waste management options in Norway, but it did not include the potential environmental benefits neither from incineration with energy recovery nor from recycling. These are

[28] Energy Technology, Chalmers University of Technology, SE-412 96 Göteborg, Sweden, phone: +46-31-772 14 45, fax: +46-31-772 35 92, e-mail: tomas.ekvall@entek.chalmers.se

[29] CIT Ekologik AB, Chalmers Teknikpark, SE-412 88 Göteborg, Sweden

key issues in the environmental assessment of waste management of combustible materials [8]. The costs of collecting material for recycling dominated the economic costs in the recycling case, but the data on collection costs were old and concerned collection in the US [5].

The Radetzki study was an assessment of the Swedish regulation for producer responsibility. It depended heavily on assumptions rather than data [9-10]. The dominant cost in the recycling case was the cost of the time spent in the households on source separation. However, the estimated time requirement was based on assumptions made before the producer responsibility was implemented. The second most important cost was the cost of source separation at companies. This cost was assumed to be more than 3500 Euro per ton. Consequently, Radetzki states that his study is based on an "extremely weak data foundation" [3].

The debate on the Bruvoll and Radetzki studies spurred an interest in improved CBAs. As a consequence, CIT Ekologik was given the task to carry through CBAs on different waste management options for newsprint, and various packaging materials. These studies involved certain developments in the CBA method. This paper presents the methodological developments in the assessment of indirect internal costs and benefits, and in the assessment of external costs and benefits. Here, internal costs are economic costs. Indirect internal costs are economic costs that occur outside the waste management system. External costs are costs that occur outside the economic systems. We expect the reports from the studies where these methodological developments were applied [10-11] to be available soon.

Scope of the studies

The CBAs in question compare different waste management options for the quantities of paper packaging, old corrugated containers, glass packaging, metals packaging, and plastic packaging that are currently being recycled in Sweden. This means that the studies are assessments of the Swedish recycling of these materials. The aim was to carry through screening CBAs that were based on easily available data but that, compared to the Bruvoll and Radetzki studies, were more complete and to a larger extent based on relevant data.

The waste management options in the studies are recycling, incineration with energy recovery, and disposal at landfill. The studies include an assessment of the economic effects and environmental effects. We also account for costs in terms of time and storage space required for source separation (Table 1). Part of the environmental effects are internalised into the economic system through environmental taxes and fees. To avoid double counting, the internalised environmental costs should be excluded from the environmental assessment. The cost of time includes indirect internal costs in the form of lost production, and external costs that are estimated as willingness to pay to avoid spending spare time on source separation. The internal cost of time is really part of the economic effects but it is estimated separately from the rest of the economic analysis. The reason is that the time-cost uncertainties are so large that, if integrated in the economic analysis, they would overshadow the other economic effects.

Table 1. Aspects included in the CBAs. Internalised environmental effects (indicated with a parenthesis) are included in the economic analysis and not as separate entities.

	Direct internal	Indirect internal	External
Economic effects	X	X	-
Environmental effects	(X)	(X)	X
Household time	-	X	X
Household space	-	-	X

Assessment of external environmental effects

The environmental burdens of the different waste management options are estimated using life cycle inventory analysis (LCI). The boundaries of the system investigated in the recycling case are expanded to include the avoided production of that material which is replaced through recycling. In the incineration case, the system boundaries are expanded to include the avoided combustion of the fuel that would be replaced if the currently recycled materials were incinerated instead.

The material replaced is assumed to be virgin material of the same type. The fuel replaced in the incineration case is assumed to be other waste flows that are displaced from the waste incinerators and end at landfills instead (Figure 1). The first of these assumptions is questionable and the second might not be valid in the long-term perspective [12].

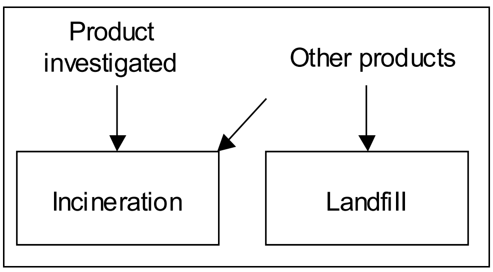

Figure 1. When waste incineration is restricted by incinerator capacity, the alternative fuel is other waste flows that are likely to end up at landfills when displaced from the waste incinerators [12].

The total environmental costs and benefits are estimated through a monetarisation – i.e., an assessment in monetary terms - of the LCI results. Three different sets of monetarisation factors were used in parallel:

- factors that were developed within the framework Environmental Priority Strategies in product design (EPS) [13],
- factors that were developed for a Swedish application of the ExternE approach [14], and
- factors based on Swedish environmental taxes and fees as compiled by Johansson [15].

We believe that the three sets of factors are sufficiently diverse to provide a good illustration of the large uncertainties involved in the monetarisation of environmental effects. While one set is based on taxes and fees on emissions and natural resources, the two other methods are based on different assessments of the costs of environmental damages. Resource depletion is emphasised by the EPS factors, but it is excluded from the ExternE factors.

The external environmental costs are obtained through subtracting the internalised costs from the total environmental costs. The internalised costs are estimated from the LCI results and the Swedish environmental taxes and fees. In reality, part of the emissions and resource demand will take place outside Sweden and, hence, be subject of different taxes. Different taxes are also applied on the same pollutant (e.g., CO2) from different activities in Sweden. These factors are not taken into account in the screening CBAs, which results in an error in the calculation in the external environmental costs. However, this error appears to be small compared to the uncertainties in the LCI and in the monetarisation.

Assessment of indirect and external time costs

Source separation requires time in the households, mainly for the rinsing of packagings and for the transport of the materials to pick-up points. Source separation is one of many duties in the household. If less time is required for household duties, we can choose either to spend more leisure time on other things, or to spend more time at work. The latter requires that we are prepared to work more hours and that we have an employer that is interested in paying for this extra work.

The indirect internal cost of the time is the production loss that follows from spending time at source separation instead of at work. No data were easily available on what share of the source separation time would be spent at work, but less than half of the Swedish population is currently employed - most of the remainder is either small children, students, or pensioners. In the light of the large uncertainty, we assumed that 3-30% of the time spent at source separation would otherwise be spent at work. The value of an additional work hour is approximately given by the salary costs, including taxes and social fees. The reason is that the salary costs reflects the marginal value of the work to the employer [16]. The average, Swedish salary cost was assumed to be approximately 20 Euro per hour. Based on these assumptions, the production loss and, hence, the indirect internal cost of source separation time was calculated to be 0.6-6 Euro per hour. The assumption regarding the salary cost can probably fairly easily be replaced by accurate data.

Given that 3-30% of source separation time would otherwise be spent at work, 70-97% of the time is a reduction in the leisure time available for other things. The external cost of the time stems from this reduction. It is estimated through the willingness-to-pay (WTP) approach. The willingness to pay to avoid spending time on a specific task depends heavily on the task in question [17, 10]. No data were easily available on the WTP to avoid spending time on source separation. Radetzki [3] assumes that it is similar to the WTP to avoid household duties in general. This, in turn, is set to nearly 7 Euro per hour, based on an official estimate of what a minority of the population pays for assistance in household duties. We believe that the WTP to avoid source separation in average is lower than this estimate for two reasons:

- the minority that pays for such assistance is likely to have less time and more money than the average population, and

- source separation is likely to be perceived as more meaningful than other household duties.

The current source separation takes place with no or little economic incentives to the households, which indicates that the WTP on the margin is nearly zero. Some consumers may even be prepared to spend money to contribute to the recycling of materials. However, one of the factors in the near-zero or sub-zero WTP is the expected environmental benefit of recycling. An estimate of the current, environmental costs and benefits of recycling is already included in the CBAs, through the environmental assessment. To avoid double counting, the expected, current environmental benefits should be excluded from the estimation of the WTP.

We do not know how important the expected, current environmental benefit are for the near-zero or sub-zero WTP. However, the arguments above indicate that the value of the leisure time lost is more than zero but less than 7 Euro per hour. Given that up to 97% of the source separation time is a reduction in leisure time for other things, the external cost of the time is 0-7 Euro per hour spent on source separation.

The uncertainty in the total cost of the time is very large. With the uncertainty ranges above, the minimum total cost is 0,6 Euro per hour. The maximum total cost is nearly 11 Euro per hour. The latter is obtained by assuming maximum work-time share (30%) and maximum specific leisure-time value (7 Euro per hour of leisure time lost). Extensive surveys are probably required to substantially reduce this large uncertainty.

Assessment of space costs

Source separation requires not only time but also space for storing the multiple waste fractions. We assume that source separation only takes place to the extent that there is room in the homes. In other words, we assume that reduced source separation would not result in smaller dwellings. This means that the space requirements of source separation do not entail any extra, internal cost. However, as a result of source separation, less space is available in the homes for other purposes. This means that the space requirement is an external cost. We assume that the value of the space is equal to the average cost of space in the homes, which is assumed to be 90 Euro per m^2 and year. The latter assumption can probably fairly easily be replaced by more accurate data.

Discussion

The studies that were carried through at CIT Ekologik are more complete than the studies by Bruvoll and Radetzki in the sense that they include several costs and benefits that were not included in the previous studies. Where the previous studies were based on assumptions or outdated, irrelevant data, our studies sometimes include recent data that are relevant for Swedish conditions. However, our studies also heavily depend on assumptions. For this reason, the uncertainties in the total results are very large. With such uncertainties it can, in most cases, not be expected to be possible to rank the waste management options. However, disaggregated results from our case studies confirm earlier findings that the costs of household spare time can dominate the total cost-benefit results. They also show that the external, environmental costs can dominate the results.

In a more extensive study, several of the large uncertainties can probably be reduced. Some of them can probably be significantly reduced through surveys. This can include the share of source separation time that would otherwise be spent on work, and the WTP to avoid spending leisure time on source separation. It can also include the quantities of time and space required for source separation. More accurate assumptions can probably be made, based on econometric models, concerning the materials that are replaced through recycling. Actual data can be collected concerning the average salary cost and the cost of space. The calculations of the internalised, environmental costs can also be improved.

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Economic assessment of waste management systems - case studies using the ORWARE model

Marcus Carlsson Reich[30]

Abstract

An LCA-based MFA system model for waste management, ORWARE, has been supplemented with tools for economic analysis. These tools consist of a financial LCC (parallel to the LCA-MFA model) and an environmental LCC (functioning as an additional, weighting tool). The basis for analysis is a municipal waste management system extended with functional units for external heat, electricity, nutrient (N, P), vehicle fuel, and materials (cardboard and plastics) production. Case studies have been made in three different Swedish municipalities: Uppsala, Stockholm and Älvdalen. Scenarios for incineration, biological treatment (anaerobic digestion and composting), materials recycling (cardboard and plastics), and landfilling have been analysed.

The financial LCC covers all the costs incurred by the extended waste management system, as though the LCA system was a single economic actor.

The environmental LCC puts an economic value, based on damage assessment, to all emissions to air, water, and soil, from the system.

The results show that both a financial and environmental LCC add knowledge about the system studied. By using the same system boundaries as the environmental analysis, the financial LCC adds knowledge about a vital decision aspect: economy. By adding an environmental LCC the results of the study can more easily be communicated, and some conclusions can be drawn concerning what are the hot spots for the studied system. The financial and environmental LCC's can be added up, with some adjustments, to form a welfare economic analysis of waste management, as long as one considers the relevant aspects of waste management are covered by economic and environmental issues. There are inherent difficulties in the methods used, namely using the same system boundaries for both the LCA and the financial LCC, and the difficulties in the entire field of environmental evaluation.

[30] IVL Swedish Environmental Research Institute, PO Box 21060, SE 100 31 Stockholm, Sweden, phone +46 8 598 56 331, fax +46 8 598 56 390, e-mail: marcus.carlsson@ivl.se

Fields of further research include actor-based analysis, household aspects of waste management, and more coherent environmental valuation studies.

Keywords: valuation, life cycle costing, LCC, waste management

Introduction

Decisions on strategies in municipal waste management are, not surprisingly, often taken at the municipal level. The decision makers should be acting in order to maximise the welfare of the inhabitants of the municipality. The municipal waste management affects the inhabitants in several ways, e.g. economically through the waste collection fee, and environmentally through emissions and indirect system effects, but also more diffuse effects such as the physical connection with waste management through the design of the collection system and the psychological effect of the localisation of waste management facilities.

The decision apparatus needs to consider all these aspects when making decisions. For reasons obvious to the audience of this workshop, LCA provides a good tool for environmental analysis of municipal waste management (i.e. systems perspective, comprehensiveness, accepted framework, etc.). However, as a very important decision factor for municipal waste management, apart from environmental issues, is economy, it is important to analyse this aspect as systematically as the environment with an LCA. Moreover, it is a great advantage if the systems studied with the economic analysis and the LCA have the same system boundaries in order to supplement each other in the decision process. The economic analysis used in combination with an LCA and with the same system boundaries is often called Life Cycle Costing, LCC.

Financial and Environmental LCC

As there is no standard or certification for an LCC, there are numerous examples and definitions of what it should and should not signify. Without discarding any other definition, I will here use the following definitions:

Financial LCC: a life-cycle perspective financial economic analysis of a product or function

Financial costs are here defined as all costs for the system studied, negative or positive. For example, if a financial LCC is done for an LCA system, all costs for fulfilling the functional unit (or units) should be included. As the environment is analysed in an LCA, no environmental effects are included in the financial LCC, other than when these environmental effects have an economic impact on the system studied (e.g. through taxation of emissions).

Environmental LCC: the valuation of environmental impacts of an LCA system in monetary terms

Thus, an environmental LCC is here defined as a weighting method for an LCA.

With these definitions, the financial LCC becomes a parallel analysis tool to an LCA, the LCA and the environmental LCC becomes consecutive tools, and the combination can be used as a welfare economic analysis, if the relevant aspects of the system studied can be narrowed down to environmental and economic aspects. See figure 1 for an illustration of this. The definitions and the illustration are based on CHAINET 2001 [1].

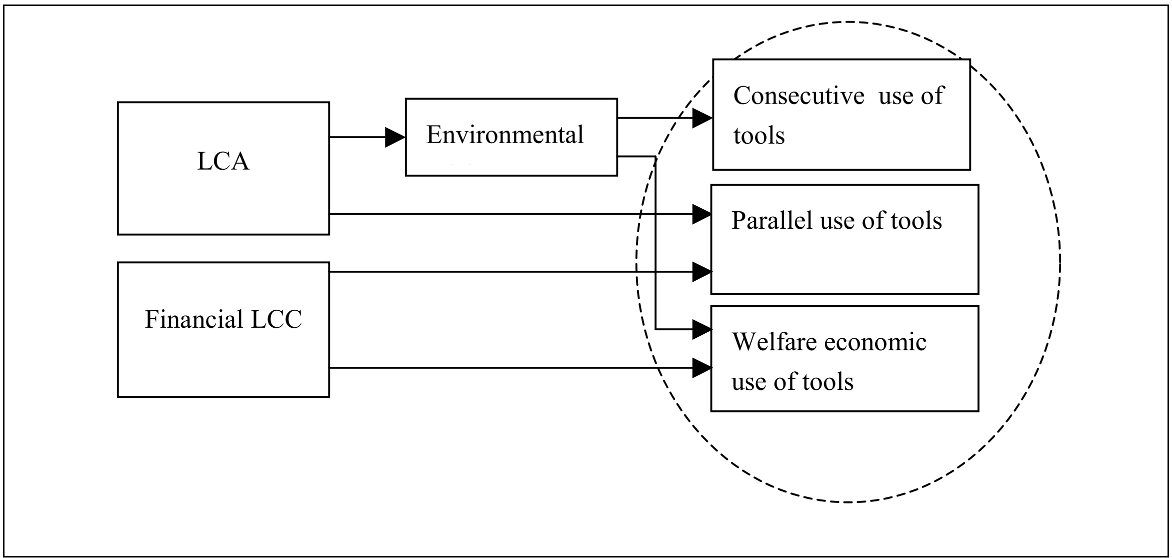

Figure 1. The use of LCA, Financial LCC, and Environmental LCC as decision support tools

Economic analyses of systems often use other terminologies. I will here explain why I do not use any of these.

Cost-Benefit Analysis or CBA aims to cover all positive and negative impacts from a policy or project in monetary terms. It is thus a very ambitious tool, but it also lacks in transparency, as the result is a single figure: whether the policy or project is a net benefit or a net cost to the system studied. A strict CBA could be seen as an overlapping tool to an LCA, but a CBA could also be “modified” and made to fit an LCA, and one such example would be the welfare economic analysis described in figure 1 above. I have here decided to use the term CBA in its stricter sense in order to avoid too much overlapping of tools and to keep the meaning of the term where it usually is used.

Cost-Effectiveness Analysis or CEA is usually a narrower tool than the ones described above. It aims to determine the least cost option of a predetermined target. Thus, there is no need to measure benefits

It is important to remember that there is no standard or widely accepted detailed specification for any of the above mentioned terms, so interpretations may vary quite a lot and make clarification as to what the terms actually imply difficult. All the terms can in theory be used for economic life-cycle analysis of a system or function: it is just a matter of defining the system boundaries so that they coincide with an LCA. In order to avoid confusion as to what is intended to be covered in each specific analysis, there is a need for a term for economic assessment of LCA systems, and LCC is the term which has a use which coincide the most with the intention in this study.

Financial LCC

Even though the exact methodology can vary depending on the studied subject, some general guidelines to follow and some common problems are presented below.

Method

In order to get the economic calculations to match the LCA calculations, using the same time frame becomes necessary. For example, if the functional unit for the LCA is taking care of the waste in one municipality during one year, the economic calculations should also be on an annual average basis. In order to be able to allocate costs accordingly, the use of standard economic tools is necessary, such as the time value of money (interest rate, discounting, present value), and annuity calculations (allocation of investments over time). In order to be able to structure all the costs adhered to a waste management system, it can also be useful to use a cost-breakdown scheme. For further information on these see for example Fabrycky, 1991 [2].

The general methodology for an LCC otherwise resembles an LCA and systems analysis. It is an iterative procedure with the following main steps:

Definition of object of analysis

As the aim is to combine the LCC with an LCA, this step is a joint step. As the system boundaries need to be the same, and the logical boundaries for an environmental and economic analysis sometimes differ, this can be difficult. An economic analysis is based on economic systems, such as a municipality, a corporation, a state, or the like. These economic systems rarely follow environmental life cycles for products or functions: the economic chain is often cut off by economic borders that should be ignored in a logical LCA system, and vice versa. Therefore it is important to realise this difficulty and define the object of analysis with both the environmental and economic analyses in mind. Often the system studied thus becomes a hypothetical system, which more or less diverges from reality.

Data collection, cost estimations

As a joint LCA-LCC most probably consists of some compromises in system boundaries, some costs are not readily available, but have to be theoretically constructed. This will be necessary if the transaction sought for does not take place in reality, e.g. the cost of time spent by households for source separation, or if the cost takes place within a company and is considered a trade secret, e.g. the production cost of virgin materials. Other costs are difficult to assess or even understand whether for the system studied it is a cost, revenues, or just a transaction within the system. An example of this is the waste reception fee at a waste incinerator: for the incineration company it is a revenue, for the municipality it is a cost, and for the entire waste system it is just a transaction within the system.

Calculations

Once costs and revenues are collected or estimated, calculations are straightforward and easy.

Analysis

As economic results come in only one unit, and as they readily are distributed over time through basic economic theory (as long as you accept the assumptions for economic theory), it is easy to interpret economic results. However, as we aim to combine the economic results with an environmental analysis through LCA data, the analysis becomes at least as complex as an LCA analysis, as we here add yet one impact category.

An LCC is subject to the same problems of uncertainty as an LCA: both for discrete choices and specific figures. It is recommended to use sensitivity analyses and cost spans with minimum and maximum estimates to deal with this.

Environmental LCC

The area of environmental valuation and also the entire field of the valuation step of an LCA are heavily discussed. There is no room in this paper for this discussion, but I rather refer to Perman et al 1996 [3] if you want to read more on the theory on environmental valuation, and to Finnveden 1999 [4] for a discussion on the valuation step in LCA's. I would merely want to point to that environmental valuation can be seen more as an attempt to bridge a gap in the field of environmental communication than an attempt to actually estimate the true value of our surroundings, and as such it may sometimes help differentiate between important and unimportant issues (in the same way as normalisation aims to do).

Merging of LCC's

As the financial and environmental LCC's use the same unit of account, it is possible to merge the two into an even more aggregated result, a welfare economic result (as long as the relevant aspects of the system can be narrowed down to economic and environmental issues). Theoretically, the main problem of this aggregation is to see to that no double-counting takes place (e.g. costs for environmental taxation in the financial LCC should be deducted as they are examples of environmental valuation covered by the environmental LCC). Care should be taken in the interpretation and presentation of these results, as the chain of assumptions by now has grown very long, and usually is not consistent at all times.

Case study

ORWARE

An LCA-based MFA system model for waste management, ORWARE, has been supplemented with tools for economic analysis. These tools consist of a financial LCC (parallel to the LCA-MFA model) and an environmental LCC (functioning as an additional, weighting tool). The basis for analysis is a municipal waste management system on a yearly basis extended with functional units (except the treatment of the waste) for external production of heat, electricity, nutrient (N, P), vehicle fuel, and materials (cardboard and plastics). Case studies have been made in three different Swedish municipalities: Uppsala, Stockholm and Älvdalen. Scenarios for incineration, biological treatment (anaerobic digestion and composting), materials recycling (cardboard and plastics), and landfilling have been analysed. In the scenarios presented, external heat is assumed produced from biofuels, and external electricity from coal. In the environmental analysis global warming potential, acidification, eutrophication, photo-oxidant formation, NO_X emissions, and heavy metal flows. The model and the results are fully reported in Sundqvist et al 1999 [5].

The financial LCC covers all the costs incurred by the extended waste management system, as though the LCA system was a single economic actor. As all the “goods” that can be produced by the system are considered functional units, no revenues are relevant for the system. Even though several recent studies have shown that the time and environmental effects of households heavily can influence the results, households have been left outside the system boundaries, mainly because of the large uncertainties entailed with the value households put on time.

The environmental LCC puts an economic value, based on damage assessment, to all emissions to air, water, and soil, from the system. These environmental values are based on studies on the marginal damage function of emissions, mainly collected from ECON, 1995 [6].

Here figures for the Uppsala case study will be presented. Uppsala is a municipality with some 180 000 inhabitants approximately 70 kilometres north of Stockholm, and it is the fourth largest city in Sweden. The other two case studies, even though carried out in what could be described as Swedish extremes (Stockholm municipality is the biggest rural settlement with some 700 000 inhabitants, and Älvdalen is a area-wise large but sparsely populated municipality with some 10 000 inhabitants), show basically the same results.

In figure 2 you will find a conceptual picture of the ORWARE system. In table 1 the waste amounts in Uppsala and their treatments in the scenarios studied are presented. Figures 3, 4, and 5 present the economic results from the case study.

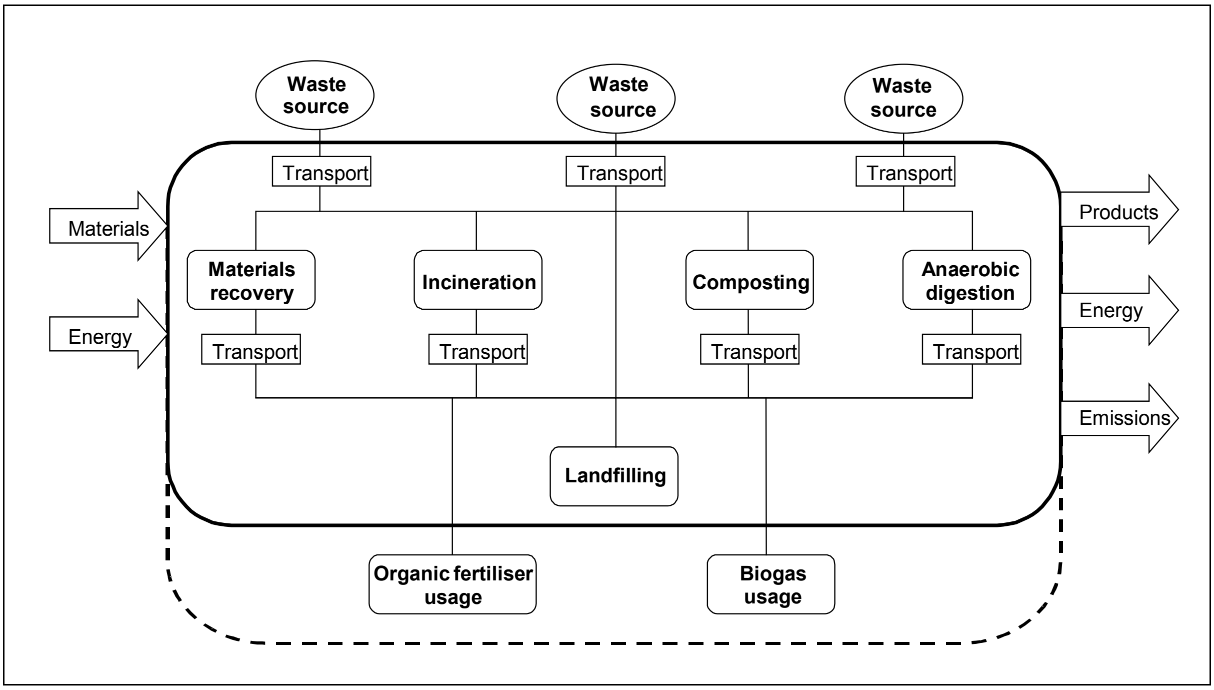

Figure 2. The conceptual ORWARE system

Table 1. Treated amount of waste in studied scenarios

Treatment (tons/yr)	Incineration	Inc + Lf	AD - bus	AD - heat+el.	Compost	Rec-Plastics	Rec-Cardboard	Landfill
Incineration	68 800	61 900	52 200	52 200	52 000	67 789	66 400	0
Anaerobic Digestion	11 500	11 500	28 400	28 400	11 500	11 500	11 500	11 500
Composting	2 266	2 300	2 200	2 200	19 000	2 300	2 300	2 300
Plastic recycling	0	0	0	0	0	1 200	0	0
Cardboard Recycling	0	0	0	0	0	0	2 000	0
Landfill of waste	20	6 900	20	20	200	20	20	68 800
Landfill of by-products	13 000	11 700	11 500	11 500	11 500	13 000	12 900	0

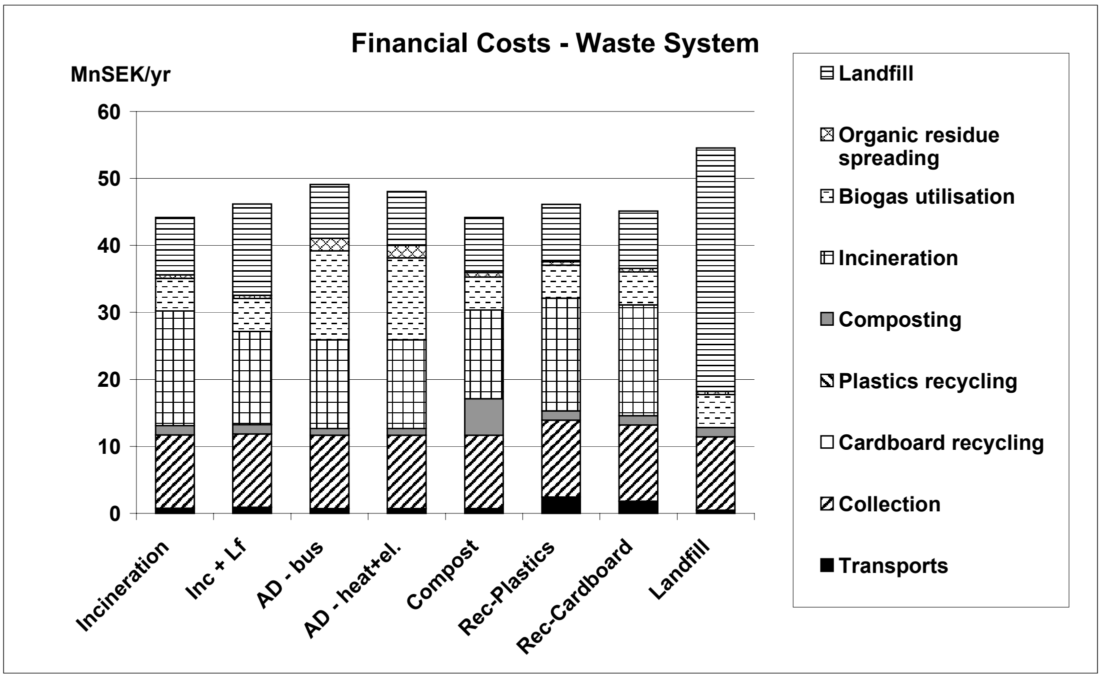

Figure 3. Financial costs for waste system – Uppsala case

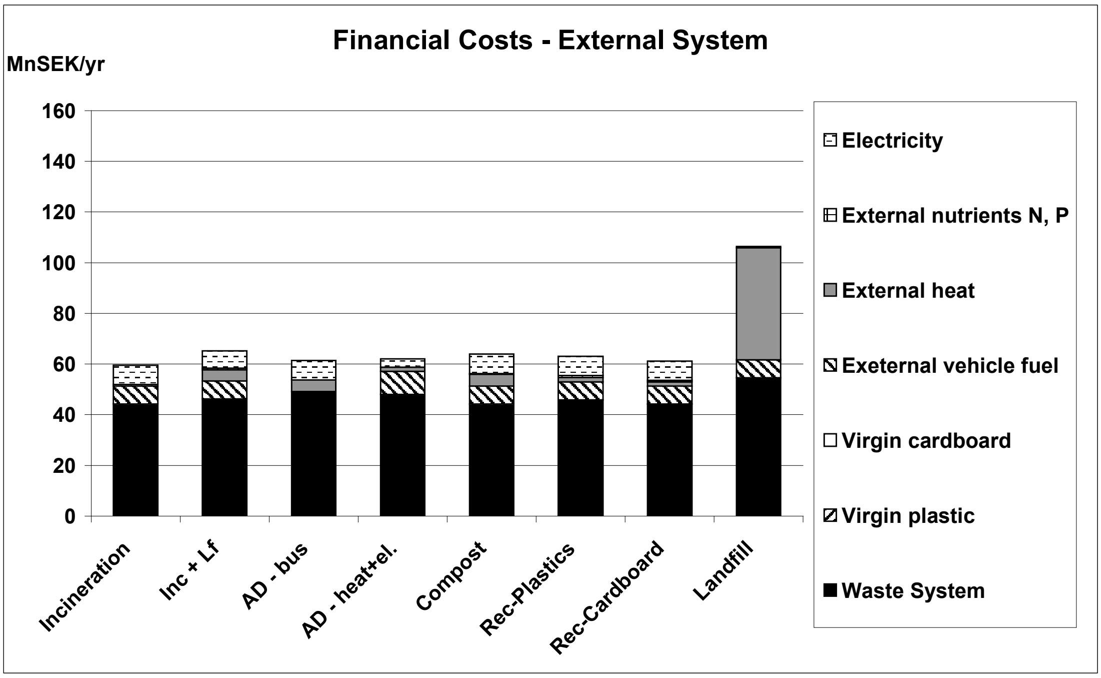

Figure 4. Financial costs for extended system – Uppsala case

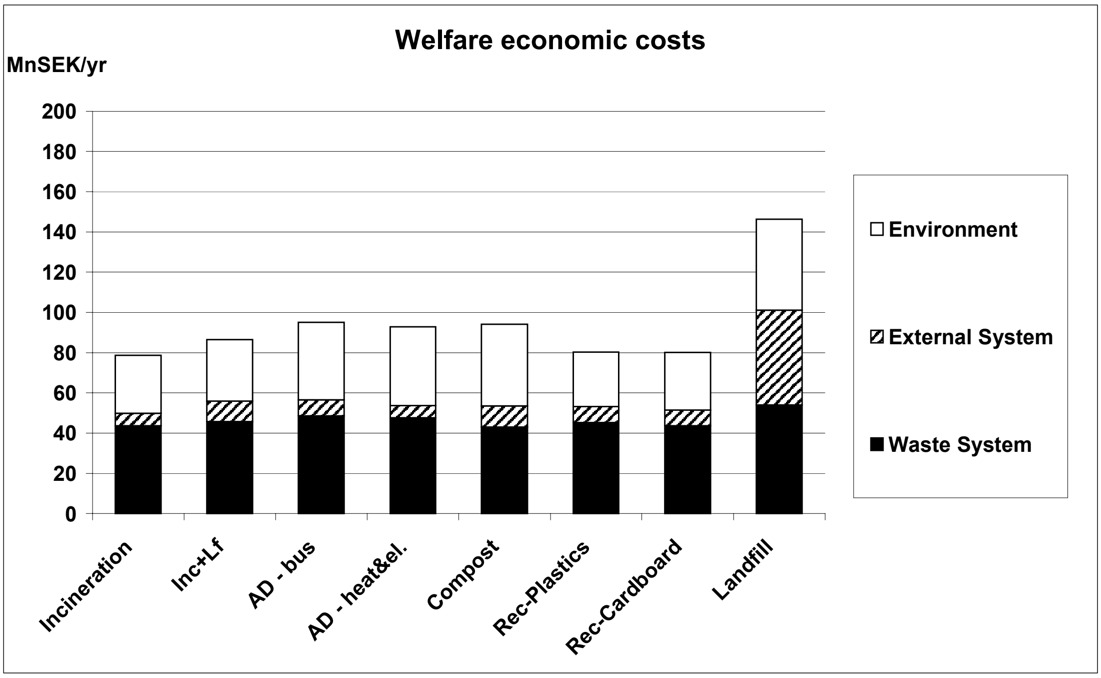

Figure 5. Welfare costs for waste system – Uppsala case

The case study shows that there is no major difference in total costs for the different waste treatment scenarios, except for landfilling, which is much more expensive because of the external production of heat. Transports and collection are important cost contributors, but do not differ noticeably from scenario to scenario. The environmental costs show the same tendency as the financial costs, although biological treatment also entails higher environmental costs. The major environmental cost contributors are CO_2, NO_X, and SO_2 emissions (and for landfilling also CH4), and lead on soil. The welfare economic results elucidate the tendency.

Other economic weighing methods

Because of the large uncertainty in valuation, three more valuation methods have been looked at briefly: EPS 2000 [7], ExternE [8], EcoTax '99 [9]. These are all methods used in Sweden in similar case studies. No attempt will be made to scrutinise the qualitative aspects of any of the methods, but they are merely presented as a form of benchmarking. In figure 6 the results from the environmental LCC of the case study on Uppsala with the four valuation methods are presented.

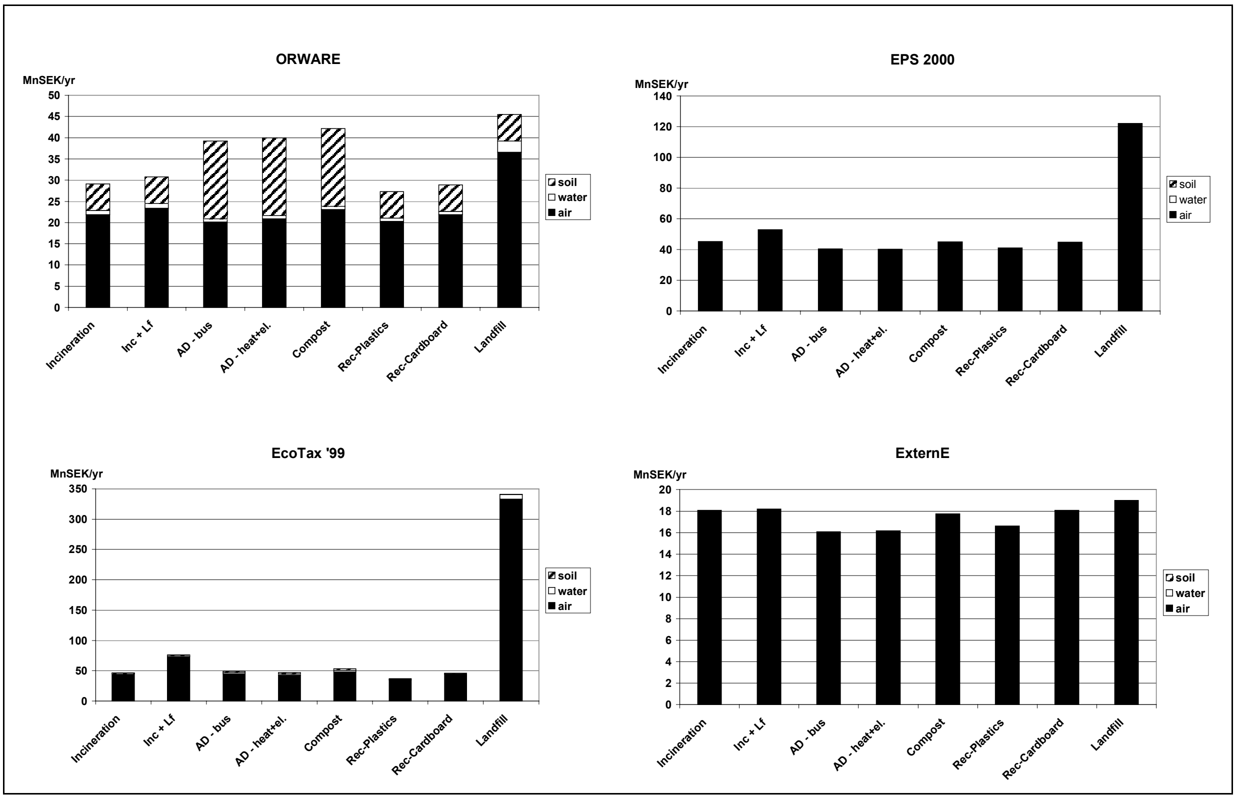

Figure 6. Four environmental valuation methods – Uppsala case

As can be seen from the figure, only in the ORWARE model anything other than air emissions is of noticeable importance. In the EPS 2000 model, VOC (for the landfill scenario), CH_4 (for the landfill scenario), CO_2, and to a smaller extent particles, are important contributors. In the EcoTax '99 model, CO_2, VOC, CH_4 (for the landfill scenario), and mercury are main contributors. In the ExternE model, CO_2, NO_X, and for the landfill CH_4, are important.

Results, Conclusions and Problems

The case study shows that for Uppsala it does not matter how the waste is treated as long as it is not landfilled, from an economic point of view. This is mainly due to that the energy in the waste in wasted in landfills to a greater extent than in the other treatment options. The exclusion of households from the system of course makes the results less certain. The environmental valuation points in the same direction, with the saving clause that the recirculation of compost and digestion residue may cause considerable environmental costs as this also entails a recirculation of heavy metals.

The use of several valuation methods shows that the conclusions are not that straightforward. The ExternE method gives all scenarios roughly the same environmental cost. EcoTax ’99 and EPS 2000 depict landfilling as the environmentally worst alternative, but do not give the recirculation of heavy metals the same importance.

The financial LCC is a fruitful complement to a decision support based on an LCA. Vital and not always obvious information can be achieved by a financial LCC, and the systems approach may also give a broader understanding than a regular economic analysis.

There is however a difficulty in communicating the results to actors in the waste management field. As the study, and thus the results, is based on a fictive system, the actors do not recognise their own contribution to the system, and therefore not the depiction of the system in the model results.

Overall it can be said that, as the welfare analysis consisting of a financial LCC and environmental LCC spans over a considerable amount of theories and assumption, there is a problem with comprehensiveness and consistency in both theory and data.

Further research

Further research should be directed into the possibility of elaborating a consistent methodology for the entire chain of economic assessment of LCA systems. Many of the areas that are uncertain in this study are however associated with genuine uncertainty, which no further research will change.

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Linking models for waste management systems and energy systems in the analysis of waste-to-energy technologies

Mattias Olofsson *and Johan Sundberg*[31]

Abstract

In this paper a method of linking models for waste management systems and energy systems in the analysis of waste-to-energy technologies is presented and discussed. The discussion is based on two case studies in the cities of Göteborg and Jönköping where the system engineering models MIMES/Waste and MARTES were linked. It was shown from the case studies that the linking procedure did improve the technical modelling results. Furthermore, the linking procedure was an effective way of getting the organisations responsible for the systems together and to capture the synergies from a coordinated planning of the overall system.

Keywords: Waste management system, energy system, model linking, waste-to-energy technologies

Introduction

Waste-to-energy technologies are characterised by the fact that the treatment of waste leads to recovery of energy, e.g. by incineration of waste heat can be recovered and used for production of electricity and district heating and by anaerobic digestion organic waste can yield biogas which can be used as a fuel for vehicles. The dual functionality of waste-to-energy technologies, i.e. both treating waste and producing energy, means that they can be analysed as options both in waste management system studies and in energy system studies. However, the objectives in these studies might be different which will influence the result of the study. In the waste management system, the main task is to handle and treat the waste. Energy recovered from the waste during the management is a by-product. In the energy system, the main task is to fulfil the energy demands. Energy technologies are chosen based on their economical, technical and

[31] Division of Energy Systems Technology, Chalmers University of Technology, SE 412 96 Göteborg, Sweden, phone: + 46 31 772 14 42, fax: + 46 31 772 35 92, e-mail*: olma@entek.chalmers.se

environmental performance. Waste-to-energy technologies are thereby evaluated in the same context as for example heat pumps and combined heat and power plants fired with natural gas.

During the last two decades many system engineering models have been developed to assist the analysis of waste management systems. Some examples on the local and regional level are the SWAP package [11], the MIMES/Waste model [15], the ORWARE model [2] and EUGENE [1]. Typically, these have been used for analysing questions of interest from a waste perspective, e.g. What are the environmental effects of increased source separation? Should the municipality invest in increased incineration capacity? How could the transports of waste be optimised? This is natural since the models often have been developed close to the actors dealing with waste, e.g. municipalities, private waste companies and local and regional authorities with the responsibility of developing policy instruments that minimise the amount and harm of waste. The needs and the objectives of these actors have been reflected in the models. The models have been used as decision support tools, thus improving the basis for decisions made regarding waste. However, models for strategic analysis are rarely used by the waste management organisations.

During the same time, system engineering models have been developed for analysing energy systems on different levels. One example is the MARKAL model [4], which has been used on local, regional, national and even international level for evaluating different strategies of solving problems in the energy system. Other examples are KRAM [7], MARTES [6] and MODEST [5]. In the same way as the waste management models are reflecting the ideas and the problems of the waste management actors, the energy system models originally stem from the ideas and the problems of the actors in the energy system. Typical actors are private and community owned energy companies and local and regional authorities responsible for energy supply. Energy system models have been used frequently as decision support tools.

The separate use of models for analysing waste management systems and energy systems, reflected by researchers and organisations with different objectives, and the dual functionality of waste-to-energy technologies might be a hinder for a consistent analysis of waste-to-energy technologies. Since waste and energy actors, as well as researchers in the field of waste and energy, tend to focus on their own system only, they might omit important features in the system environment for waste-to-energy technologies. This omission might strongly affect the result of the system studies.

In this paper a way of removing this omission and thus improving the analysis of waste-to-energy technologies is presented. By linking models, which represent the system in focus for waste and energy actors respectively, the idea is to bring together competence and relevant issues for the analysis. As a base for the paper two case studies in the cities of Göteborg and Jönköping are used [9,10]. In both these case studies the district heating system has been evaluated as the energy system.

In chapter 2 a brief description is made of the system engineering models and how they have been linked in the case studies. Similarities and differences between Jönköping and Göteborg regarding waste management and production of district heating are overviewed in chapter 3. In chapter 4 some of the results from the case studies are discussed and compared to results from earlier system studies in Jönköping and Göteborg. In chapter 5 the main conclusions of the paper are summarised.

Method and models

In the studies in Jönköping and Göteborg, several waste-to-energy options have been evaluated by a combined analysis of the waste management system and the district heating system. The systems engineering models MIMES/Waste and MARTES have been used to analyse the waste management system and the district heating system respectively.

MIMES/Waste is a static model that analyses waste streams in a municipality or a region during a year. The model is based on linear programming and includes both economic and environmental issues. The model handles the waste from collection via transportation and intermediate treatment to recovery or final disposal. Normally household waste, industrial waste and construction and demolition waste are included in the analysis, but more waste types can be included. The different waste types are further divided into fractions, such as paper, cardboard, plastics etc. Joint treatment of different waste types can be studied with the model. Energy recovery takes place within the system boundary when waste is treated by incineration, thermal gasification or anaerobic digestion. However, the energy recovered is assumed to be absorbed by a market sector, which is part of the system environment.

MARTES is a model for district heating systems with production of heat, steam and electricity. The model simulates the use of different plants to satisfy the demand of district heating during a year. The effects of the energy conversion in the district heating system on economy and emissions are calculated. The calculation is based on a load curve, which is divided into 730 periods, i.e. two periods for each day of the year. In every period the operation of the plants in the district heating system is simulated according to their production costs. Waste-to-energy plants that generate district heating, steam and electricity can be evaluated with the model. Waste is handled as a fuel by the model, i.e. collection and transports of waste and pre-treatment of the waste is part of the system environment. Waste enters the system when it is used as a fuel in the energy plant. MARTES is today the most used tool for strategic planning of district heating systems in Sweden (The model covers nearly 70 % of the produced heat).

In the studies of Jönköping and Göteborg, the models were linked in two slightly different manners. In Jönköping, where a new incineration plant was evaluated, the

MARTES model was first used to evaluate different heat capacities for the incineration plant. Every single heat capacity yielded a demand for waste as fuel. MIMES/Waste was after that used to minimise the costs of the waste management system given the demand of waste as fuel for an incineration plant. By adding the total costs for the waste management system and the district heating system, calculated by MIMES/Waste and MARTES respectively, the capacity leading to the least costs could be identified. The emissions of greenhouse gases in the waste management system and the district heating system were added to observe the total change of emissions.

In Göteborg, the waste-to-energy technologies (thermal gasification and incineration) were part of the waste management system and thus analysed with MIMES/Waste. The amount of district heating produced from waste was calculated with MIMES/Waste. In MARTES the total district heating demand was reduced with the amount that was produced from waste. Thereby the other plants available in the district heating system satisfied the rest of the demand. The costs and the emissions of greenhouse gases in the waste management system and the district heating system were added to observe the total change of costs and emissions.

Waste management and district heating in Jönköping and Göteborg

There are three common features about waste management and district heating production in Jönköping and Göteborg that make these cities interesting for an analysis on linking models for waste management systems and energy systems.

- Waste management and district heating production are handled by separate municipal organisations.
- System studies, with and without system engineering models, have earlier been performed on the waste management system and the district heating system separately. Some examples are [3, 8, 12-17].
- Waste-to-energy technologies have been options in the separate system studies.

The number of inhabitants in Jönköping is around 120 000. The municipal organisation Tekniska Kontoret has the responsibility for managing household waste and hazardous waste. Other waste types, e.g. industrial waste and construction and demolition waste, are mainly collected and treated by private entrepreneurs. The municipal landfill is used for final disposal of both household waste and other waste types. The district heating system in Jönköping is owned by the city and run by the city owned company Jönköping Energi. The main fuels in the district heating system today are pulverised wood, oil and waste heat from the wastewater treatment plant that is used in a heat pump.

In Göteborg, the number of inhabitants is around 450 000. The city holds 83,7 % of the shares in the regional waste management company Renova. The rest of the shares are divided among ten small municipalities close to Göteborg. Renova collects and treats the major part of household waste, industrial waste and construction and demolition waste in the region around Göteborg. The main treatment facility is an incineration plant, which was established 1972. 1998 approximately 380 000 tonnes of waste were incinerated, which yielded a net production of 945 GWh district heating and 84 GWh electricity. The district heating system is owned by the city and run by the city owned company Göteborg Energi. A major part of the production of district heating comes from oil refineries (waste heat) and from the incineration of waste done by Renova. Thus Göteborg Energi buys heat from the oil refineries and Renova. The rest of the production of district heating comes mainly from heat pumps and heating stations and combined heat and power plants fuelled with natural gas.

Results and discussion

At the start of the case study in Jönköping (autumn 1998) the municipal organisations respectively had separate solutions to the problems they were facing. In order to fulfil the coming bans on landfilling of combustible and organic waste in Sweden, Tekniska Kontoret wanted to introduce source separation of household waste into three fractions: one fraction for anaerobic digestion, one combustible fraction and one fraction for landfilling [3]. The combustible fraction was either intended to be sold to Jönköping Energi or incinerated outside Jönköping. Jönköping Energi was facing other problems. The base load plant in the district heating system, a combined heat and power plant that used pulverised wood and oil, was getting rather old and needed to be replaced in a near future. The MARTES model had been used several times for evaluating replacement alternatives to this old plant. The most interesting alternative seemed to be a bio-fuelled heating station.

The organisations had been working with parallel system studies with almost no interaction between each other. In the case study, representatives from both Tekniska Kontoret and Jönköping Energi were involved in a reference group. The separate solutions of the two organisations were compared with waste incineration. The waste incineration could either yield district heating or district heating and electricity. With waste incineration the main problems of the two organisations could be solved, the combustible and organic waste would not be landfilled and a new base load plant would be established. The combined study revealed that it would be cost efficient to invest in a waste incineration plant with production of district heating. In figure 1 the annual cost reduction can be observed. Depending on the heat capacity of the incineration plant, the annual cost reduction varies for the waste management system and the district heating system. The cost reduction for both systems is the sum of the cost reduction in the waste management system and in the district heating system. It is interesting to observe that the largest annual cost reduction for both systems together, approximately 25 MSEK/yr, is achieved with a heat capacity which is not optimal for either of the two subsystems. In earlier system studies of the waste management system performed 1993 [8] and 1994 [12] incineration was also an option. However, the size of the incineration in those studies was based on the needs of the waste management system only. In the combined study it was disclosed that a much larger plant would be of economical interest, which can be explained by figure 1. When also including the needs of the district heating system in the analysis the optimal size of the incineration plant was enlarged. This meant that the incineration plant would not only meet the waste treatment demands of Jönköping but also the demands of the smaller municipalities in the vicinity of Jönköping. Another new insight was the large effect on the size of the incineration plant when the material recovery of waste was assumed to be much higher in one scenario. The optimal size of the plant was lowered from 60 MW_{th} to 26 MW_{th} and the annual production of district heating from waste was lowered from 423 GWh to 212 GWh.

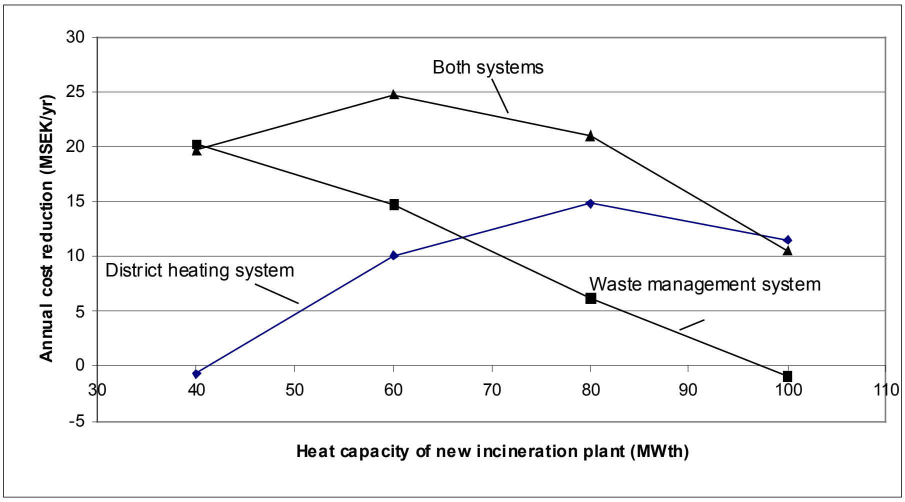

Figure 1 Annual cost reduction for the waste management system and the district heating system in Jönköping when investing in waste incineration with heat recovery (1 SEK approximately equals 0,1 US$).

In Göteborg, as well as in Jönköping, new knowledge on how the waste management system and the district heating system interact was gained through the study. At the start of the case study, the two municipal organisations Renova and Göteborg Energi were working on separate projects with no interaction between each other. The main question for Renova was whether to expand the incineration capacity due to increased demand for treatment of waste. For Göteborg Energi the big issue was whether to build a large combined heat and power plant fuelled with natural gas. Both the organisations were doubtful to the plans of the other organisation.

In the case study, representatives from both Renova and Göteborg Energi were involved in a reference group. Increased waste incineration and thermal gasification were evaluated as options to meet an increased demand for waste treatment. The analysis was made both with and without the eventual combined heat and power plant in the district heating system. The analysis showed that increased waste incineration was a cost efficient option. Furthermore, increased waste incineration could be combined with an investment in a new combined heat and power plant fuelled with natural gas and still result in a cost reduction compared to the present situation (see figure 2). The annual costs of both systems were lowered between 20 and 36 MSEK/year depending on the size of the plant and if the new combined heat and power plant was established.

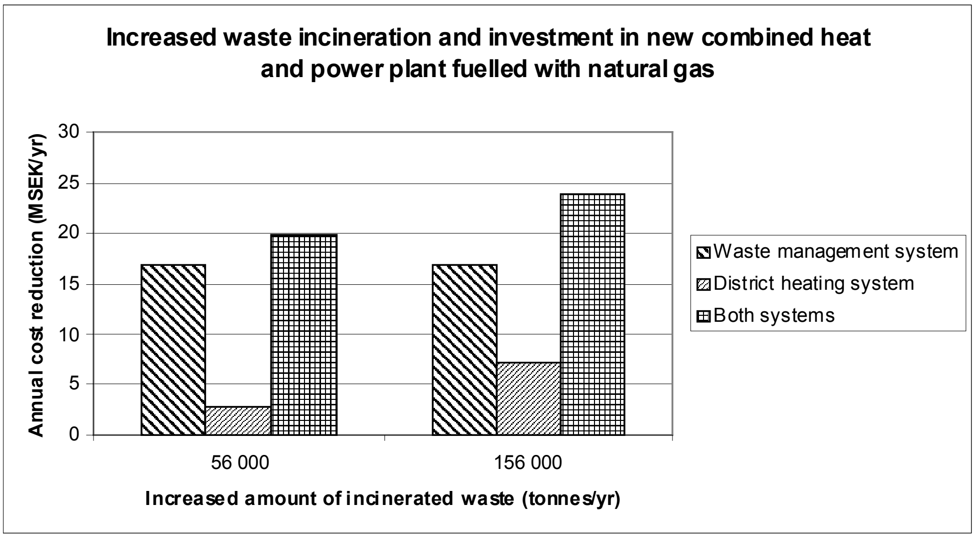

Figure 2. Annual cost reduction for the waste management system and the district heating system in Göteborg (1 SEK approximately equals 0,1 US$).

Another interesting result was how the net electricity production[32] could change in the district heating system as an effect of increased waste incineration. The increased district heating from the waste treatment could reduce the net electricity production, mainly because the production of district heating from the combined heat and power plants in the district heating system was reduced. Although some electricity was produced from the waste, the overall effect, i.e. from the waste management system and the district heating system, could be that increased waste treatment led to reduced net electricity production. Since the electricity produced in Göteborg was assumed to replace electricity production on the margin in the Nordic electricity system, which was assumed to be Danish coal condense power, the change of net electricity production had large effect on the emissions of greenhouse gases. Compared with a separate analysis of the waste management system only, where no effects on the net electricity production in

[32] Net electricity production = Production of electricity– Consumption of electricity

the district heating system were included, the emissions of greenhouse gases were increased with around 60 kton CO_2-eqvivalents when the incineration of waste increased with 156 000 tonnes. For comparison, the emissions of greenhouse gases in the waste management system of Göteborg were 52 kton CO2-eqvivalents.

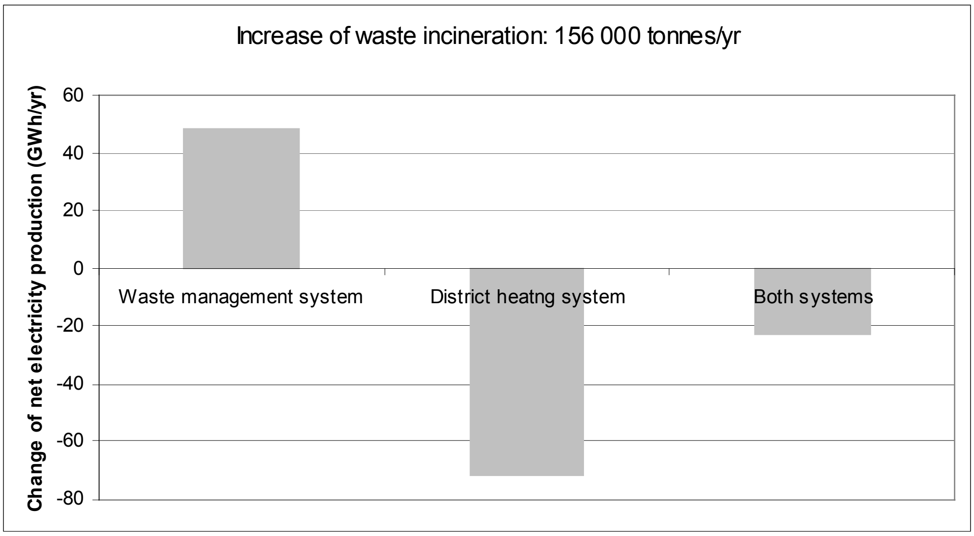

Figure 3. Change of net electricity production when the incineration of waste was increased with 156 000 tonnes/yr.

The procedure of linking models did not only improve the technical modelling results, it also had other advantages. It brought together the actors that were involved in the problem analysed. In the two case studies presented here the organisations in both cities brought validated models to the system studies, which they understood and believed in. The linking thus served as a way of communicating where the two linked models formed a neutral arena in which questions were put and answered. For all questions analysed there were three results, one for each organisation and one for the overall system. To produce and compare these three results proved, in both case studies, to be an effective way of getting the organisations together and to capture the synergies from a coordinated planning of the overall system.

Conclusions

In this paper results from studies on linking models for waste management systems and energy systems have been presented. Of special concern has been how the linking procedure affects the analysis of waste-to-energy technologies. The main conclusions from the studies of the waste management systems and the district heating systems in Jönköping and Göteborg are:

- Cost efficient joint strategies between the waste and the energy actors have been identified. In both Jönköping and Göteborg these strategies include waste incineration.
- A possible effect of increased waste incineration is decreased total net electricity production in the waste management system and the district heating system. This might strongly affect the effect of waste incineration on the emission of greenhouse gases.
- By the linking procedure the municipal organisations in the waste management and the district heating system have gained information and knowledge about the environment to their system.
- The procedure of linking models does not only improve the technical modelling results, it also brings together the actors that are involved in the problems analysed. The linking serves as a way of communicating where the two linked models forms a neutral arena in which questions are put and answered.

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Session 3: Summary of discussions

Summary by Paul Brunner; Edited by Jan-Olov Sundqvist.

Six different approaches were presented in this session.

Common denominators in the studies in this section were:

- Best environmental and economic solution were sought in the studies.
- There is a conflict between comprehensiveness and simplicity. Often there is a struggle between what is needed and what can we do.
- Waiting processes are needed. Standardisation is slow and costly, but is needed.
- All are eager to learn during the studies.

The general results in the studies were:

- Landfilling is the worst scenario in all six studies.
- Recycling of particulate fractions is beneficial.
- Incineration is an important process.

Importance of the methods applied:

- LCA (life cycle assessments) and MFA (material flow analyses).
- Systems definition in time and space are important.
- Assumptions can play a role for the result.

The evaluation methods were discussed:

- It is important to use more than one valuation method.
- There are several unsolved questions which can play role, e.g. the time spent by the households for waste sorting and waste transportation, and how that time is valued. Also the weighing of greenhouse gases is important.
- Local impact is not generally considered in LCA studies. One important example is smell.

Goals, methods and procedures are important to discuss.

A good study is not good enough. Important items in a good study is also:

- transparency
- communication with stakeholders and actors, and a user-friendly report.

Session 4.

Chairman: Johan Sundberg

Jan-Olov Sundqvist
Some methodological questions and issues that are of great interest for the result in a LCA

Åsa Moberg
Environmental effects of landfilling of solid waste compared to other options – assumptions and boundaries in life cycle assessment.

Stefanie Hellweg
Time- and site-dependent LCA of thermal waste treatment

Markku Pelkonen
Landfill emissions and their role in waste management system

Monica Salvia
Toward a sustainable waste management system: a comprehensive assessment of thermal and electric energy recovery from waste incineration

Jenny Sahlin
Waste incineration and electricity production

Anders G. Klang
Framework for sustainable waste management – examples from the building sector

Discussions

Some Methodological Questions and Issues that are of Great Interest for the Result in LCA

Jan-Olov Sundqvist[33]

Abstracts

In LCA and similar system analyses of waste management systems, the results are often depending on different choices and assumptions that are made during the goal definition and scoping stage. These choices and assumptions can be determining factors for the result. Examples of such choices are: electricity production, production of alternative heat, time aspects in landfilling, quality of recycled material, market for recycled material, choice of system boundaries, choice of allocation principles, and presentation of the result. These choices are often discrete choices (e.g. "shall the electricity be modelled as marginal or average") and not choices of data. If wrong choices have been made, the results can be misleading. There is an apparently risk that "wrong" choices are made, either by lack of knowledge or by full awareness (in order to "manipulate" the result).

It is of absolute importance that the quality of the study is guaranteed in order to avoid such misuse of LCA. That can be done if the following requirements are set up:

- An adequate interpretation of the result is necessary. The ISO 14040 Standard emphasises the importance of the interpretation of the result. The interpretation shall be based on the numerical results from the inventory, classification, characterisation and environmental impact stages, considering all choices and assumption that have been done in the "Goal definition and scoping stage or anywhere else in the study

- When presenting a LCA study all choices must be documented and motivated in the report. Transparency is a key word in this context.

- The LCA should include a sensitivity analysis were the consequences of all determining choices is studied.

- The LCA should be peer reviewed, to assure and guarantee the quality.

Key words: LCA, waste management, methodological choices, scenarios, sensitivity analysis

[33] IVL Swedish Environmental Research Institute, PO Box 21060, SE_100 31 Stockholm, SWEDEN, Phone: + 46 8 598 563 74, fax: + 46 8 598 563 90, E-mail: Janolov.Sundqvist@ivl.se

Introduction

In LCA and similar system analyses of waste management systems, the results are often depending on different choices and assumptions that are made during the goal definition and scoping stage. These choices and assumptions can be determining factors for the result.

Some examples

These choices occur for example when studying extended systems or spared systems, or when describing the interface between the waste system and the external system. Some examples are as follows.

Electricity production

There are two extreme cases that often are used for assuming how the electricity is produced:

a. Marginal electricity produced by for example coal condensation, which gives a very high environmental impact (acidification by SO_2 and emissions of fossil CO_2).

b. Average electricity production. In Sweden, for example, the average production is mainly hydropower and nuclear power, which both give very low environmental impact with the impact categories usually used in LCA.

It is possible to favour e.g. a scenario with high consumption of electricity, by choosing average electricity, or disfavour the same by choosing marginal electricity by coal condensation power. In the ORWARE study (12, 13, 14, 15, 16) we have made sensitivity analyses of electricity production, but in this case the same ranking order of waste strategies were obtained.

Heat production

The major product from waste incineration in Sweden is district heating. The district heating system in a city is usually based on a mixture of different energy sources: electricity (for heating pumps), oil, biofuel (wood), coal, peat and waste. A decrease of waste incineration leads to an increase of another energy source, and an increase of waste incineration leads to a decrease of other energy sources for district heating. When analysing waste incineration there are mainly three alternatives that can be considered:

a. The substitute fuel is a fossil fuel (oil, coal, and natural gas), which produces fossil CO_2.

b. The substitute fuel is a biofuel (wood) which does not produce fossil CO_2.

c. The substitute fuel is other waste that is landfilled if not incinerated.

The choice of substitute or supplementary fuel governs the assessment of waste incineration. If an increase of waste incineration is studied, the choice of supplementary fuel governs the result in the following way:

- Fossil fuel as supplementary fuel. An increase of waste incineration leads to a decrease of greenhouse gases since less fossil fuel is consumed.
- Biofuel as supplementary fuel. An increase of waste incineration leads to a slightly increase of fossil CO_2 (from plastics in the waste).
- Other waste as supplementary fuel (the waste is landfilled if not incinerated). An increase of waste incineration leads to the same consumption of fossil fuel and biofuels, and to a decreased amount of landfilled waste which gives a decrease of emissions of methane (a greenhouse gas).

Thus, if other waste or fossil fuel is the supplementary fuel, waste incineration is more favoured than if biofuel is the supplementary fuel.

In the ORWARE study (13, 14, 15, 16) different alternative fuels were studied. We found that the ranking order between incineration and anaerobic digestion were changed for the impact category Greenhouse gases when the supplementary fuel was changed.

Time aspects in landfilling

Wastes that are put into a landfill will cause emissions for a very long time in the future. Theoretically leaching from ashes can occur for 1 – 10 million years (17, 19). During the other stages in the lifetime of a product, all other emissions will occur more or less instantaneously or at least within a limited time period. Also emissions from other treatment methods such as incineration, composting, etc. causes emissions that occur more or less instantaneously. An important question is then how the future emissions shall be handled in LCA. At the workshop *LCA and Treatment of Solid Waste* (9) researchers working with LCA and waste were gathered and discussed different problems, for instance time aspects in landfilling. It was suggested that several of the time frame options could be relevant to use. The aims with the LCA, and decisions on system boundaries etc. during the phase ”goal definition and scooping” should govern which time frames that should be used. When time frames were discussed, there was a consensus that the emissions should be integrated over a special period, often called ”foreseeable” period. However, the time frame for the “foreseeable” period has varied from 15 years (e.g. 7) to 50 000 – 100 000 years in different studies (e.g. see paper by Hellweg in this proceedings).

Quality of recycled material

An important point is how recycled materials substitute virgin materials. A common assumption when assessing material recycling in LCA is that 1 kg of recycled material substitute 1 kg of virgin material. Due to different quality aspects, often more than 1 kg recycled material have to be used to substitute 1 kg of virgin material. Also common is that the recycled material is used for special ”low-quality” products that wouldn’t be produced of virgin material.

In the ORWARE study (13 – 16; see also papers by Ola Eriksson and et al and by Marcus Carlsson Reich in this proceedings), we found that when recycling cardboard, about 1.15 kg of recycled cardboard is used for substituting 1.00 kg of virgin cardboard.

In the ongoing ORWARE study also plastic recycling is under study. The present plastic recycling in Sweden indicates (4):

- LDPE is often used for production of sacks and bags (e.g. waste bags). A bag based on recycled LDPE usually consumes up to 30 % more plastic than a bag based on virgin PE.
- About 20 % of the recycled HDPE is used for substituting other material, e.g. wood palisades.
- The rest of the recycled HDPE is used for products where the quality is of lower importance and 1 kg recycled HDPE substitutes 1 kg of virgin HDPE)
- When comparing the first assumption (1 kg substitutes 1 kg), with the actual plastic recycling and substitution, it was found that the results differed some – the advantages by plastic recycling was not as obvious as first. However, the major results and major conclusions were consistent and material recycling was still more favourable than incineration.

Often it is assumed that the quality of compost and anaerobic digester residue is good, e.g. low metal content and no other ecological or “aesthetic” contaminants or impurities, and that it can be used for soil improvement and substituting chemical fertilisers. However, in the reality there are often composts and digester residues that have contaminants and impurities.

Market for recycled material

Another issue, with connections to the quality issue, is the market for recycled material. When we produce a product from waste, e.g. recycled plastic, recycled paper, compost, heat, electricity etc. we usually in the LCA assume that there is a 100 % market for the recycled material. However, there can be several factors limiting the real market:

- Bad quality of the recycled product can reduce the willingness to buy the product.
- The total market can be limited. E.g. when waste is incinerated for production of heat, we can only sell the heat needed for the actual heat distribution system. This can lead to that energy can not be used during the summer, when the heat need is low.

When interpreting the result from the LCA, the analyser should have the market in mind. It may be relevant to assume a 100 % market for the recycled product, if we want to study the potential of a recycled or recovered product in a “pre-study”. However, only limited conclusions can be drawn from such a study. If we want the actual effects of an implemented system, the LCA has to consider the real market.

System boundaries

System boundaries are sometimes chosen to avoid data lacks, and to exclude uncertain processes. In the case of LCA of waste management, upstream processes and downstream processes are often omitted from the studied system.

One example is the work done by households when source separating waste. If packages are washed, the washing will consume energy and will cause emissions to water. If the people use the car to take the source-separated waste to a collection place or drop-off centre; the car trip will consume energy and cause emissions to air. In the ORWARE study (12, 13, 14, 15, 16) we made some simple assumptions to see if the work done by households can be of importance. We found that the impact and energy consumption from the household can be noticeable, even if the ranking order of the studied alternatives did not changed.

Another example is spreading of compost and anaerobic digester residue to arable land. In the very first ORWARE study (5, 10) the system boundary when spreading the compost and digester residue was defined as “1 mm above the ground level”. The studied system included the energy for transporting and spreading the compost and digester residue, but excluded later emissions of e.g. ammonia to air and nutrients to ground and water. Later the system boundary was expanded (1, 6, 13, 16) to include the emissions from the compost and digester residue after spreading, which made the result changed remarkable.

Choice of allocation principle

A traditional problem in LCA is how to deal with processes or groups of processes with more than one input and/or output. The difficulties lie in how the emissions shall be shared between different input parameters. Waste treatment processes are examples of such processes. *Allocation* can, in LCA, be defined as *the act of partitioning in some proportionate shares the responsibility for environmental impacts caused by processes in a life cycle.* In LCA practice, when handling multi-input processes, there have been several allocation methods suggested, by weight, by volume, by price or cost or by causal physical-chemical parameters. Discussions about allocation in connection with waste treatment were given at the workshop LCA and Treatment of Solid Waste (9). Further discussion of allocation is also given in (17) and (19).

In the practical LCA work, emission data is often presented as emission factors. The emission factor can be expressed in different ways depending on the chosen allocation principle. The emission factor is usually defined as the quotient between the emission flow and some input flow expressed in e.g. mass, energy, monetary units, volume units, etc., depending on the chosen allocation principle. In causal allocation, the major problem is to find a relevant causal relationship between the emission and the material studied. Emissions can be divided into *product-related* and *process-related* to understand the formation of different emissions (8, 17, 18, 19). The product-related emissions are related to the chemical composition of the studied material, while process-related emissions are formed by the process and are difficult to relate to a specific element or compound in the waste. In this context the choice of allocation of process-related emissions are of interest to discuss.

One example of process-related emission is dioxin from incineration. The formation of dioxin is complex and it is difficult to relate the formation to one single parameter. E.g. it have been suggested by total weight, by heating value, by carbon content or by chlorine content, see several papers in (9), and (17). In the following Table (from [17]), the importance of the allocation principle is illustrated, with dioxin as example. For paper there is a factor 10 between weight allocation and chlorine allocation. For glass the emission is =0 for chlorine, carbon and heating value allocation, but 10^{-12} for weight allocation.

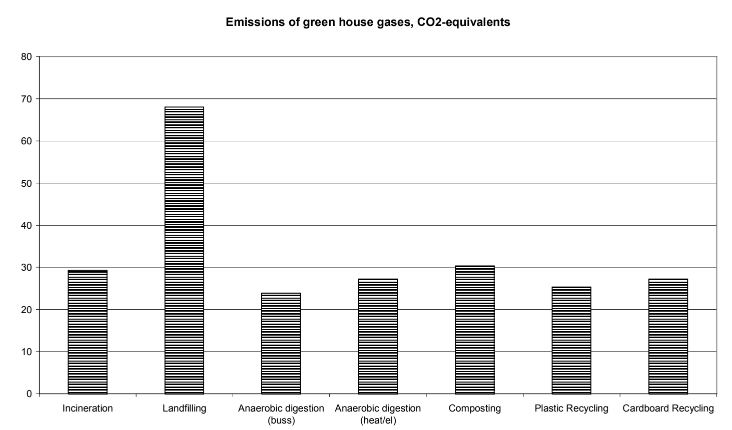
Parameter	*Unit*	**Emission factor**	Paper	PVC plastic	Polyethene plastic	Glass
Dioxins						
Weight allocation	*kg TCDD/kg*	$\mathbf{1{,}00 \cdot 10^{-12}}$	$1{,}0 \cdot 10^{-12}$	$1{,}0 \cdot 10^{-12}$	$1{,}0 \cdot 10^{-12}$	$1{,}0 \cdot 10^{-12}$
C allocation	*kg TCDD/kg*	**3,41** $\cdot 10^{-12}$	$1{,}1 \cdot 10^{-12}$	$1{,}3 \cdot 10^{-12}$	$2{,}3 \cdot 10^{-12}$	0
Cl allocation	*kg TCDD/kg*	**2,00** $\cdot 10^{-10}$	$1{,}0 \cdot 10^{-13}$	$8{,}0 \cdot 10^{-11}$	0	0
Heating value allocation	*kg TCDD/kg*	$\mathbf{9{,}13 \cdot 10^{-14}}$	$1{,}3 \cdot 10^{-12}$	$1{,}8 \cdot 10^{-12}$	$3{,}8 \cdot 10^{-12}$	0

Choice of impact categories

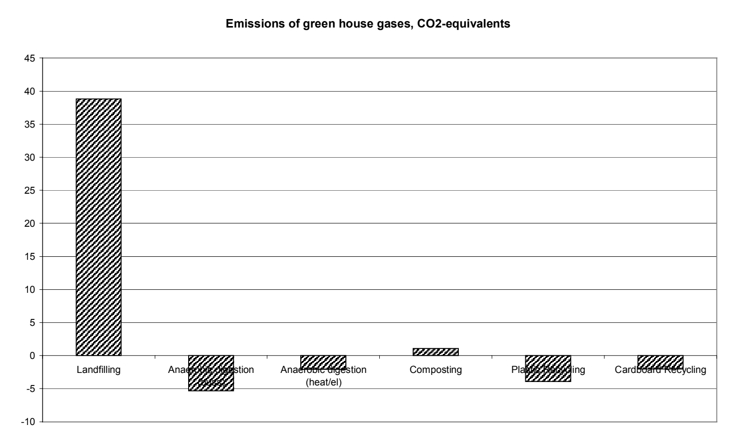
Sometimes impact categories are excluded in the impact assessment, due to difficulties to characterise some categories (e.g. eco-toxicity and human toxicity), or due to that a certain impact category can be considered ”unimportant” or ”negligible”.

Presentation of the result

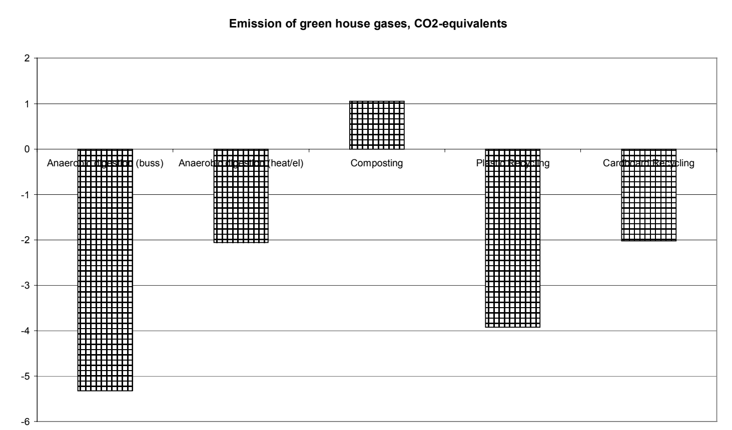
The same result can be presented in different ways, to emphasise different aspects. An illustration is given in the Figure 1. The Figure shows the emissions of greenhouse gases from the municipality waste management, including the emissions from the external or complementary system. The three diagrams are based on the same data set, but the second and third diagram shows the difference to the base scenario (Incineration). All data is from the ongoing ORWARE project (16).



The first diagram shows the total emissions from the waste system plus the complementary system. The diagram may be interpreted as “there are small differences between the alternatives, except landfilling”



The second diagram is based on the same data, but the difference to the base scenario is given (scenario Incineration is =0). This diagram may be interpreted as the first diagram “there are small differences between the alternatives, except landfilling”



The third diagram is identical to the second diagram, except that the landfill scenario is excluded. The difference to the base scenario is given (scenario Incineration is =0) When looking at this diagram it is easy to get the impression that the greenhouse gas emissions can be reduced

	significantly by anaerobic digestion, plastic recycling and cardboard recycling.

Figure 1. Illustration of different ways to present the result. (Data are preliminary, and taken from the ongoing ORWARE project (16))

Discussion and conclusion

As shown above, there are several choices that control the result. These choices are often discrete choice (e.g. "shall the electricity be modelled as marginal or average") and not choices of data. If wrong choices have been made, the results can be misleading. This gives rise to the following risks:

- If the LCA accomplisher has bad knowledge of the consequences of his/her choices, he/she can make wrong interpretation of the result and draw misleading conclusions.

- It is possible to manipulate a LCA! It is, in principle, possible to "order" a study with a wished result. A skilled LCA accomplisher can choose border conditions and scenarios and make choices to get a certain wanted result.

It is of absolute importance that the quality of the study is guaranteed in order to avoid such misuse of LCA. That can be done if the following requirements are set up:

- An adequate interpretation of the result is necessary. The ISO 14040 Standard emphasises the importance of the interpretation of the result. The interpretation shall be based on the numerical results from the inventory, classification, characterisation and environmental impact stages, considering all choices and assumption that have been done in the "Goal definition and scoping stage or anywhere else in the study

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Life Cycle Assessment of Energy from Solid Waste – Landfilling as a treatment method

Åsa Moberg, ***Göran Finnveden****, Jessica Johansson and Per Lind*[34]

Abstract

The validity of the waste hierarchy for treatment of solid waste is tested. This is done by using the tool Life Cycle Assessment on recycling, incineration with heat recovery and landfilling of recyclable waste for Swedish conditions. A waste hierarchy suggesting the environmental preference of recycling over incineration over landfilling is found to be valid as a rule of thumb. There are however assumptions and value choices that can be made which make landfilling more preferable. This is the case for some waste fractions and for some of the environmental impacts studied when only a limited time period is considered. When transportation of waste by passenger car from the households is assumed for the other treatment options but not for landfilling, landfilling also gains in preference in some cases. The paper concludes that assumptions made including value choices with ethical aspects are of importance when ranking waste treatment options.

Introduction

The present paper summarises some of the results from a study performed at the Environmental Strategies Research Group (fms) where different strategies for treatment of solid waste are evaluated based on a life-cycle perspective (Finnveden *et al.* 2000). The aim of this paper is to test the validity of the waste hierarchy, focusing on cases where the landfill option may be higher ranked. Assumptions and valuations leading to these cases are also discussed. Other results from the study are presented in another paper (Finnveden *et al.* 2001).

Methodology and assumptions

In general, Life Cycle Assessment (LCA) methodology based on standards and guidelines (ISO 1997; Lindfors *et al.* 1995) is used. This methodology is also applicable on LCAs on waste management (Clift *et al.* 2000; Finnveden 1999). The methodology used is described in more detail in Finnveden *et al.* (2000).

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[34] Environmental Strategies Research Group (fms), Swedish Defence Research Agency and Department of Systems Ecology at Stockholm University, PO Box 2142, 103 14 Stockholm, Sweden

In the characterisation a number of established methods are used including Houghton *et al.* (1996) for global warming . Two different methods are applied to characterise toxicological impacts, the toxicity parts of the Danish EDIP method (Hauschild *et al.* 1998a; Hauschild *et al.* 1998b) and the Dutch model USES-LCA (Huijbregts 1999). The results from the characterisation are further processed by weighting. For this a method based on Swedish taxes, Ecotax 98, is used (Johansson 1999). Details about the impact assessment are presented in Finnveden *et al.* (2000). The results are interpreted using the outcomes from all steps of the assessment.

The different waste management options studied are landfilling, incineration, recycling of paper and plastic fractions and digestion and composting of food waste (Finnveden *et al.* 2000). The household waste fractions used as input to the systems are the combustible and recyclable or compostable ones; food waste, newspaper, corrugated cardboard, mixed cardboard and five plastic fractions. In this paper the focus is on the paper and plastic fractions. The waste management options are studied in a base scenario, which is complemented with a range of "what-if" scenarios.

One important difference between landfilling and most other processes in an LCA is the time frame. Emissions from landfills may prevail for a very long time, often thousands of years or longer. The potential emissions from landfilling have to be integrated over a certain time-period. It is important to determine which time period is of interest. There is currently no international agreement on this question (Finnveden and Huppes 1995). Using the LCA definition as a starting point, it can be argued that emissions should be integrated until infinity. In practise however, a shorter time frame (decades and centuries) has usually been chosen (see Finnveden (1999) for a review). The choice of the time frame is clearly a value choice for the inventory analysis of an LCA. It is related to ethical views about impacts on future generations (Finnveden 1997). A similar situation may occur for different parts of the life cycle impact assessment. The choice made by the SETAC-Europe working group on Life cycle impact assessment is to consider first the infinite time period, then a short time period of 100 years and finally if wanted other time periods (Udo de Haes et al. 1999a, Udo de Haes et al. 1999b).

Here a hypothetical infinite time period is used when inventorying emissions, which is considered to be in line with the precautionary principle. This may be seen as a "worst case", assuming complete degradation and spreading of all landfilled material (Finnveden *et al.* 1995). To evaluate the effects of choosing another, shorter, time perspective this is also tried. A limit in time is then set after the so called surveyable time period. This is the period until the landfill has reached a pseudo steady state, a time period corresponding to approximately one century. For municipal solid waste landfills this is defined as the time it takes for the landfill to reach the later part of the methane phase when gas production is diminishing and this time is approximated to be one century (Finnveden *et al.* 1995). For landfilling of incineration ashes the surveyable time period is defined as the period during which the soluble chloride salts are leached out (Sundqvist *et al.* 1997).

Common practise in LCAs is to disregard biotic CO_2-emissions. This can be motivated from different perspectives (Dobson 1998). One includes an expansion of the system boundary to include also the uptake of the CO_2 in the growing tree. This expansion is often done as a thought experiment rather than an actual modelling. Another perspective can be the assumption that when biotic resources are harvested, new resources are planted which will take up an equivalent amount of CO_2. Again this modelling is normally not done explicitly. Yet another perspective is the assumption that if the biotic resources, e.g. trees, had not been harvested, they would have been left in the forest and degraded there. This degradation can however be quite slow, and the time frame has to be extended to several centuries before all biotic materials have been degraded (Zetterberg and Hansén 1998).

The biological carbon is thus seen as part of a cycle, where carbon is sequestered by and released from renewable sources continuously. However, if the surveyable time period is set as a boundary in time this cycle is interrupted. Then, one may consider landfills to be carbon sinks keeping carbon from being released to the atmosphere. With this perspective the landfilling option may be credited the avoidance of the global warming potential the trapped biological carbon would have had as carbon dioxide in the atmosphere. This is done by subtracting carbon dioxide emissions corresponding to the amount of biological carbon trapped (Finnveden *et al.* 2000). However, this concept is as noted above a value choice neglecting potential effects on future generations.

In the base scenario, the following major assumptions are made:

- distances for transportation of waste are moderate,
- heat production, which is credited waste treatment systems where heat is produced, is from incineration of forest felling residues,
- electricity is produced from hard coal,
- recycled material is credited waste treatment systems using data for production of virgin material of the same kind and
- the time perspective is a hypothetical infinite time period.

Several "what-if" scenarios are used to discover parameters of importance for the outcome of the study (Finnveden *et al.* 2000). The following are discussed here:

- *the surveyable time period scenario*, where a limit in time regarding landfill emissions is set after approximately one century,

- *the carbon sink scenario*, which has the same time limit as the surveyable time period scenario, but also credits landfills for the biological carbon which is not emitted – the landfill is regarded as a carbon sink
- *the increased transports scenario*, where longer transport distances by truck to incineration and recycling facilities are assumed, and
- *the passenger car scenario*, where waste for recycling and incineration is source separated and transported by car to collection points. This variant is tried both for recycling and incineration due to the possible development towards separate incineration of different waste fractions for better efficiency and also towards small-scale and co-incineration using specific fractions, even though these incineration techniques are not specifically modelled here.

Results and discussion

In the following presentation only a selection of results are shown, for a full presentation of results see Finnveden *et al.* (2000).

The results of the LCA of the base scenario indicate that landfilling is in general the least preferred option (Finnveden *et al.*, 2000 and 2001). In Figs 1-3 below, the results presented are for waste newspaper and PET (polyethylene terephtalate) representing paper and plastic waste.

When a shorter time perspective than in the base scenario is used concerning emissions from landfills the resulting rankings of the different waste management strategies may change. For newspaper and PET the categories where this happens are global warming and eco-toxicological impacts.

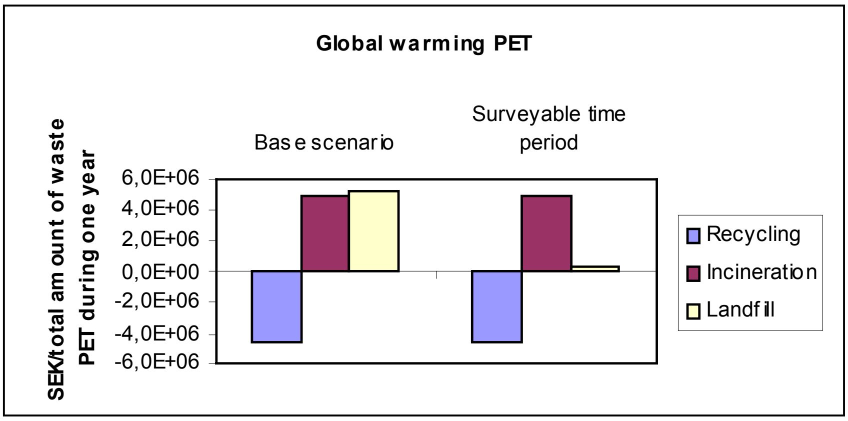

Figure 1. Results for the impact category global warming for waste PET. The results are shown for the base scenario and for the surveyable time period scenario.

For all the plastic fractions landfilling becomes preferable to incineration concerning global warming, although recycling is still ranked as the most preferable option when a short time perspective is used. This is illustrated with PET as an example in Figure 1. Emissions of carbon to air from landfilling of plastic waste mainly occur subsequent to the surveyable time period, and are thus omitted in this scenario. When incinerating plastic waste, all carbon is immediately emitted as carbon dioxide and thus landfilling of plastic waste is contributing less to global warming during the surveyable time period. In the case of landfilling newspaper, and other paper fractions, the major contribution to the global warming impact category is from emissions of methane during the surveyable time period and only a smaller difference is seen which, as shown in Figure 3, does not change the ranking of the treatment options.

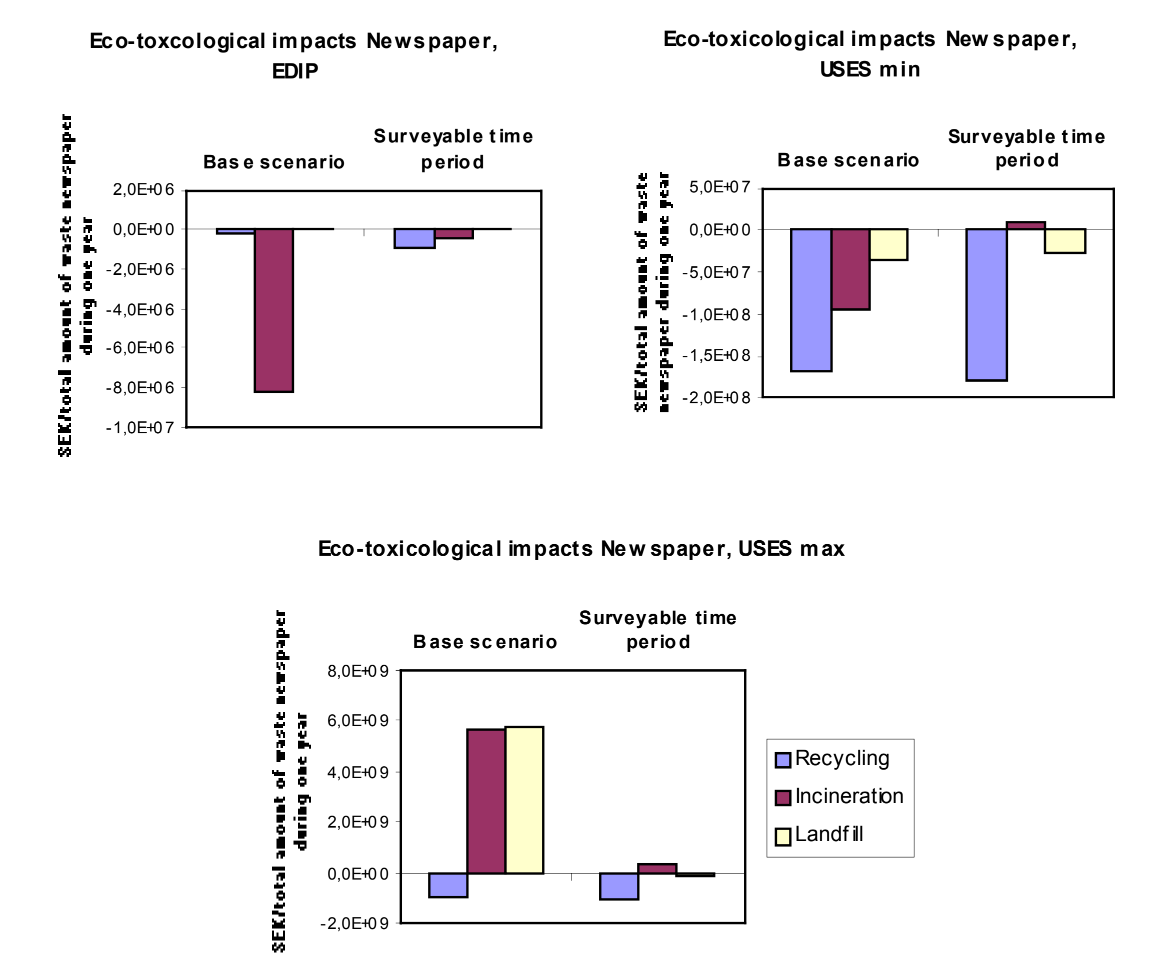

Figure 2. The results for the impact category eco-toxicological impacts for the waste newspaper fractions using three different impact assessment methods. The results are presented for the base scenario and for the surveyable time period scenario.

The other impact category affected by the change in time boundary is eco-toxicological impacts. Here, changes are dependent on how the eco-toxicological effects are modelled in the characterisation methods used. Since a large part of the metal content of the waste landfilled is modelled to leach out subsequent to the surveyable time period studied in this scenario the landfill option is better off here concerning eco-toxicological impacts. However, there are emissions of importance, using the toxicological impacts assessment methods presented earlier, which are counted also in this scenario. For example emissions from vehicles used and from landfill fires. The emissions from landfill fires which are most important are dioxins, but also to some extent PAHs. Ranking of waste treatment options concerning eco-toxicological impacts give landfill the ranking first, second and last depending on waste fraction and characterisation method used. In Figure 2, the results for the waste newspaper fraction is presented, changes for PET are similar.

If, with the limited time perspective, the landfill is considered to be a carbon sink additional advantage for the landfilling option is gained. The additional function of trapping biological carbon leads to more preferable results for the landfilling option concerning global warming for two of the waste paper fractions, newspaper and mixed cardboard. The resulting ranking is recycling before landfilling before incineration. As can be seen in Figure 3 the difference between landfilling and incineration is here small. The plastic fractions are not affected since their carbon content is of fossil origin.

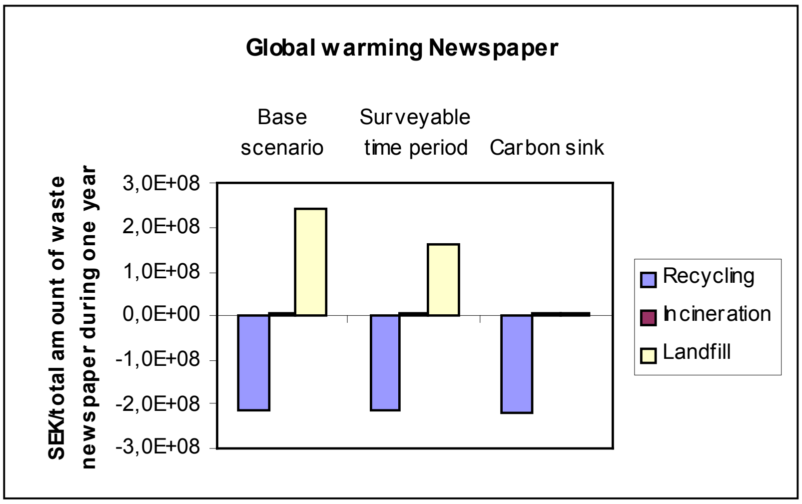

Figure 3. The results for the impact category global warming for waste newspaper. The results are presented for the base scenario, the surveyable time period scenario and for the scenario where the landfill is regarded as a carbon sink.

Different distances for transportation of waste by truck to treatment facilities does not influence the rankings of treatment options much (Finnveden *et al.* 2000 and 2001). However, transportation of waste from the household by passenger car to collection points may influence the results significantly. This can be seen in a scenario where passenger cars are assumed to be used for transportation of sorted waste for recycling and incineration. Major alterations in the resulting rankings are seen for the impact categories photo-chemical oxidant formation and for human and eco-toxicological impacts. In the toxicological categories landfilling is ranked as the most preferred alternative in several cases, when the other options are burdened with passenger car use. In Figure 4 the effects of transportation of waste newspaper is presented for the eco-toxicological impact category. It can clearly be seen that longer transportation by truck does not affect the results much, but when passenger car is used the ranking is altered. Similar changes appear for all waste fractions studied.

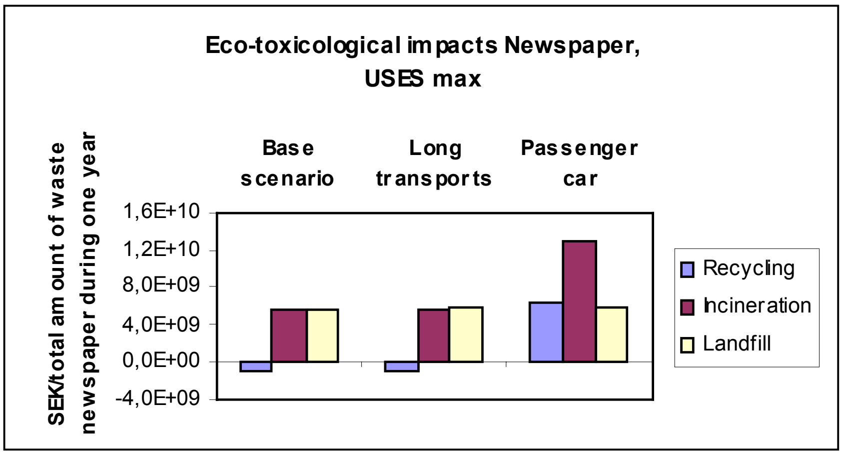

Figure 4. Results for the impact category Eco-toxicological impacts for Newspaper, using the USES characterisation method with maximum weighting according to the Ecotax 98 weighting method.

Many of the effects of altering assumptions and boundaries as described above are seen in the toxicological impact categories. It should be noted that any conclusions drawn from the results for these categories should be drawn with extra care. Uncertainties include data gaps, methods for comparing different toxicological impacts and also for estimating the impacts of different emissions, including cumulative and synergetic effects. Since emissions from landfills are also spread over large periods of time, actual emissions are not possible to measure and models and assumptions used include additional uncertainties.

Conclusions

One basic difference comparing landfilling of waste to other treatment strategies is that less co-functions are produced. Even though 50% of the landfill gas is assumed to be collected and combusted with energy recovery, this only makes out a part of the potential resource that the waste may constitute if treated by recycling or incineration. This is a draw back for the landfilling option.

Conclusions of the paper are that the waste hierarchy is valid as a rule of thumb. There are, however, certain assumptions and valuations that can lead to exceptions to this rule. Aspects of particular interest for the landfilling option are:

- Which time perspective is chosen. This concerns which emissions that are to be charged the landfilling option,
- If, with a limited time perspective, the landfill shall be credited for trapping biological carbon so far not emitted to the atmosphere,

- Transportation of waste, if this is substantially less in the case of landfilling compared to other waste treatment options. This is in particular relevant for transportation by passenger car.

A general conclusion is thus that assumptions made including value choices with ethical aspects are of importance when ranking waste treatment options.

Acknowledgements

Financial support from the Swedish National Energy Administration is gratefully acknowledged. This paper is a shortened and edited version of an earlier paper (Moberg et al, 2000) and relevant parts are reprinted with permission.

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Time- and Site-Dependent Life Cycle Assessment of Thermal Waste Treatment

Stefanie Hellweg, *Thomas B. Hofstetter, Konrad Hungerbühler*[35]

Key words: Time- and site-dependent assessment, landfill model, soil model, LCA, scenarios

Abstract

The high living standard of many industrial countries has directly lead to an increase in the amount of municipal solid waste generated. Parallel to this increase in waste, there has been a raising demand for environmentally benign waste treatment processes. In Switzerland, the predominant way of treatment is incineration. Since the environmental impact of waste incineration depends on the technology used, a comprehensive assessment of the different thermal processes is necessary. In order to determine the environmental impact, we propose a model that quantifies the emissions and resource use resulting from the incineration of waste using different technologies, the landfills for the incineration residues, the transport of waste, related infrastructure, as well as the production of ancillary products. Using the Life-Cycle Assessment (LCA) methodology, we performed a case study that compared the conventional grate technology to new high temperature processes recovering metals and vitrifying the incineration residues. The results show that if the plant is equipped with a modern gas purification system the incineration process itself is not a key environmental problem of the system considered. Using the energy gained from waste incineration as the functional unit, the environmental impacts of incineration plants are comparable to that of a conventional power plant. If long-term time horizons are considered, the critical aspect is the release of heavy metals from the landfilled incineration residues. Due to the better quality of the solid outputs new technologies have a lower potential for environmental impact than the conventional grate technology. This, however, depends on the time horizon considered. With a temporal system boundary of 100 years, the grate technology appears better, because new technologies generally use more energy and short-term emissions are of minor importance no matter what technology is used.

[35] Swiss Federal Institute of Technology Zurich (ETH), Chemical Engineering Department, Safety & Environmental Technology Group, Universitätsstr. 33, UNL, CH-8092 Zurich

Contact author, hellweg@tech.chem.ethz.ch, Fax: +41-1-632 12 83, T.: +41-1-632 33 39

The evaluation of waste incineration technologies largely depends on the assessment of heavy metal emissions from landfills and the weighting of the corresponding impacts at different points in time. Unfortunately, common LCA methods hardly consider spatial and temporal aspects. Several methodological innovations are suggested in this work. In order to quantify the impact of landfill leachates with respect to groundwater contamination, a simplified geochemical landfill model is proposed. The results indicate that slag landfills might release heavy metals over very long time periods ranging from a few thousand years in the case of Cd to more than 100`000 years in the case of Cu. The dissolved concentrations in the leachate exceed the quality goals set by the Swiss Water Protection Law (GSchV) by a factor of at least 50. The classification of the mobility of heavy metal cations in the subsoil of the landfill was performed with a generic guideline developed for this purpose. The method is easily applicable to individual landfill sites. The results indicate that the geological conditions below the landfills play an important role. Depending on these conditions, the retardation of the heavy metals ranged from a few days to many thousand years at different sites.

The long emission period of the heavy metals from landfills and the retardation of these pollutants in the subsoil raise the question whether impacts at different points in time should be weighted alike. For instance, the magnitude of damage might change as a consequence of a changing background contamination. It is concluded that possible future changes in the magnitude of damage should be considered in scenario analysis in the characterization phase of LCA.

The proposed methodological innovations for the assessment of heavy metal transport have been applied to the case study of Cd and Cu emissions from three slag landfills at different sites in Switzerland. The emissions of heavy metals to the subsoil as a function of time were calculated with the geochemical landfill model and their subsequent fate in the subsoil was assessed using the proposed soil guidelines. A scenario analysis was performed to consider possible changes in the background contamination of the groundwater influencing the magnitude of the potential damage caused by emissions of Cd and Cu. It was shown that landfills represent a significant risk for the groundwater in Switzerland since they accumulate big amounts of heavy metals that may be released over very long time periods. The ecological evaluation of Cd and Cu emissions from slag landfills per kilogram of incinerated waste largely depended on the development of the corresponding background contamination (method applied: *Swiss Ecopoints*). The site of the landfill also had a significant influence on the results. The impacts of Cu emissions from the slag landfill to the groundwater were assessed to be 2 to 18`000 times more important (depending on the site and the assumed background contamination) than the complete system of grate incineration in the previous analysis, where the landfill emissions had been supposed to enter the surface water. These results illustrate that a site- and time-dependent impact assessment of landfill emissions and the consideration of the groundwater compartment are crucial for a thorough assessment of different waste treatment technologies.

Slides from presentation

Outline

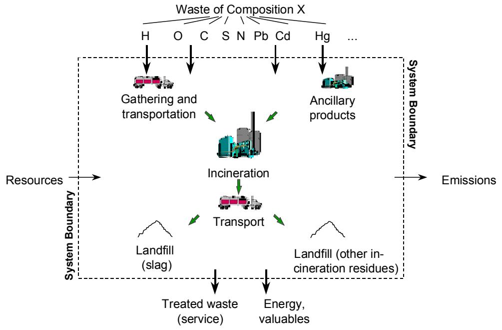
A. General LCA of thermal waste treatment processes

- Inventory model
- Case study results: Comparison of incineration technologies

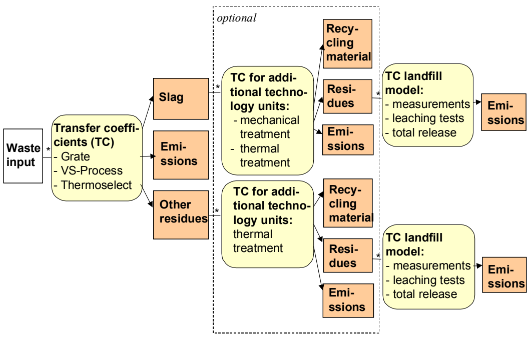
B. The specific problem of heavy metal releases from landfills

- Modeling the emissions of slag landfills as a function of time
- Site-dependent fate-assessment of heavy metals in the subsoil of the landfill
- Summarizing case study results
- Conclusions

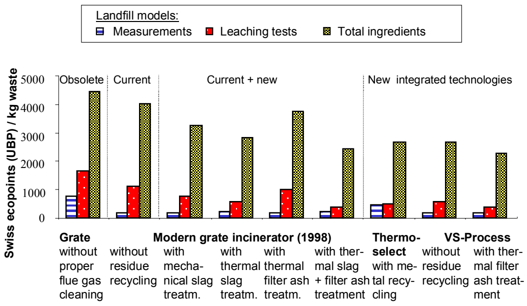
The incineration of waste, the landfills for incineration residues, transport, infrastructure, and the production of ancillary products lie within the system boundaries.



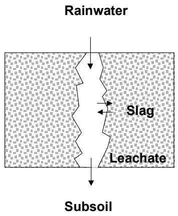
The product related output is calculated with transfer coefficients. Different technologies have been modeled. There are various ways to model the landfills.



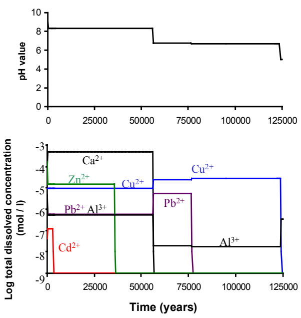
The choice of the landfill model largely determines the overall potential impacts of the system waste incineration.



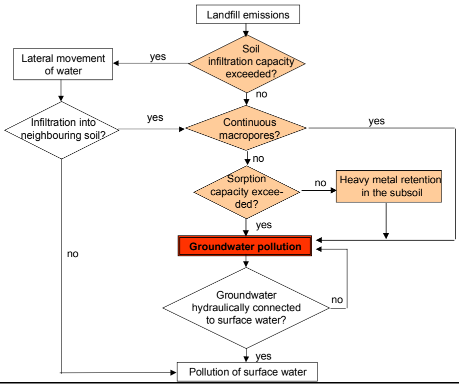
A one-dimensional transport and reaction model was used to simulate the pH development and the emission concentrations of a slag landfill as a function of time (slag from grate incineration).



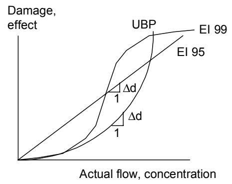
The dissolved concentrations of heavy metals in the leachate surpass the quality goals set by the Swiss Water Protection Law by a factor of at least 50.



The fate of heavy metals in the subsoil depends on spatial parameters.



The magnitude of an impact may change as a function of the background contamination level.



Swiss Ecopoints (UBP):

$$w_j = \Delta d = c * \frac{F_{a,j}}{(F_{crit,j})^2} = const * F_{a,j}$$

Where

w: Weighting- or ecofactor

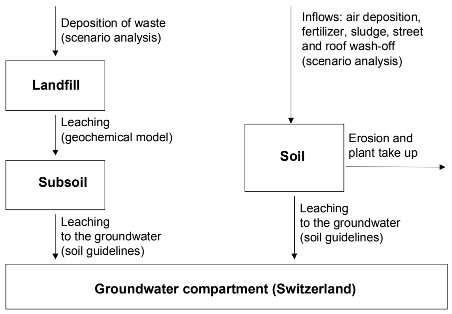
c: Constant

$F_{crit,j}$: Target value or critical flow of impact category j

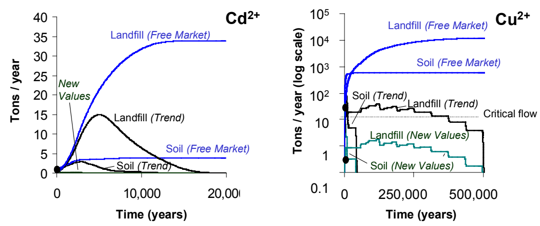
$F_{a,j}$: Actual flow / year in impact category j

The marginal damage and the ecofactors depend on the contamination level, if the LCIA methods *Swiss Ecopoints* or *Eco-indicator 99* are used. With respect to these methods, the modeling of the background contamination is important for the damage assessment.

The background contamination of the Swiss groundwater was modeled with scenario analysis and the proposed landfill and soil model.

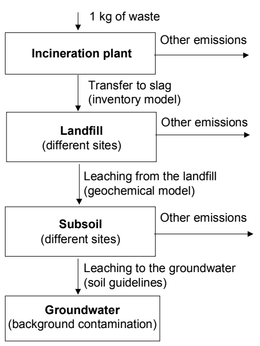


Landfills emit large amounts of Cd^{2+} and Cu^{2+} to the groundwater.

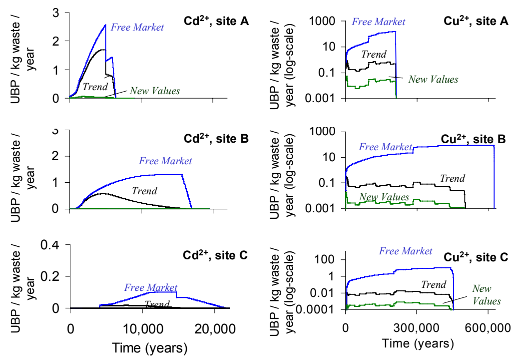


The modeling of the background contamination is used for the impact assessment of emissions from landfills.

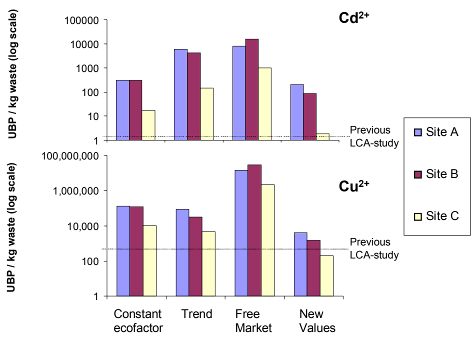
The Cd^{2+} and Cu^{2+} releases to the groundwater resulting from the incineration of 1 kg of waste were assessed taking account of the changing background contamination (only path via slag landfill).



The background contamination and the site of the landfill have a significant influence on the results (*Swiss Ecopoints*).



The potential impact of Cd^{2+} and Cu^{2+} releases to the groundwater is much higher than in LCA studies, where the emissions are assumed to enter the surface water.



Conclusions

- The landfills for incineration residues are the most relevant process steps in the system *waste incineration*, if long-term time horizons are considered.
- Landfills are significant potential pollution sources of the groundwater in Switzerland.
- Temporal and spatial factors play an important role in LCA of waste incineration.

Documentation

The full-length paper will be submitted to the *Journal of Cleaner Production*. The results of this paper were elaborated in a dissertation project

(http://www.dissertation.de/html/hellweg_stefanie.htm, for a printed version of the thesis contact hellweg@tech.chem.ethz.ch).

Acknowledgements

The funding of the project by the Swiss National Science Foundation within the Swiss Priority Program Environment (SPPE) is gratefully acknowledged.

Landfill emissions and their role in waste management system

Markku Pelkonen[36]

Abstracts

The emissions from landfills will have impact on total emissions of the waste management system for a long period and is often the dominating factor compared to other sources. Therefore information about the landfill emissions are of importance when the different waste management alternatives are compared.

In Nordic conditions the climate affects the degradation due to low temperature. Earlier (not Nordic) estimates have been that as a result of waste degradation the organic matter flow in liquid phase could be 1 – 2 % of total flow and the rest would emit in gas phase. Our estimates about organic matter transport in liquid phase are considerable higher, approximately 6 %, which is a result of a longer lasting acidogenic like condition. The estimation of the emission coefficients from landfills is based on two time frames. The first one is limited to 15 years, during which period a considerable fraction of organic matter has been emitted in the liquid phase and which can be estimated rather reliably. The second time frame is an ultimate, hypothetical one, during which all biodegradable organic matter is degraded. This leads to results that the emissions of COD in 15 years are 30 – 50 of the ultimate emissions depending on the waste fraction and of BOD_7 50 – 70 %. For nitrogen these emissions were considerable lower, appr. 4 – 6 % of the total emissions would emit in 15 years. This gives a very long nitrogen release, 200 – 400 years if linearly extrapolated. In gaseous phase the carbon emissions as methane are estimated 30 – 40 % in 15 years compared to the ultimate one.

When the eutrophication effect of COD and nitrogen are compared, in 15 years time frame COD is dominating in untreated leachate, but depending on technology applied in leachate treatment the role of nitrogen can be dominating in the effluent water even in this short time frame – in the long time frame it is clearly dominating. Therefore the behaviour of nitrogen in landfill management is of importance.

A comparison was made between landfills and composting of biowaste (especially small scale) showing that the eutrophication effect from composting can be around one third of short term landfill eutrophication effect, which does not support every house application of small scale composting. Also a system comparison was made about the role of landfills in respect to GWP. This showed that the emissions from landfills are the dominating even in 15 years perspective.

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[36] Helsinki University of Technology, Laboratory of Environmental Engineering, PO BOX 6100, FIN 02015 HUT (Espoo) Finland , E-mail: markku.pelkonen@hut.fi, fax +358-9-451 3856, phone +358-9-451 3847

Because landfill emissions are dominating, decrease of those emissions needs further system development, for example in the form of landfills as bioreactors including the 'accumulated old landfills'. In this way the emissions can be handled or utilised with possible new options and the long term costs can be managed.

Keywords: Landfilling, organic matter degradation, nitrogen, time frames, composting, emissions, system analysis

Toward a sustainable waste management system: a comprehensive assessment of thermal and electric energy recovery from waste incineration

Monica Salvia[a,b,*], *Carmelina Cosmi*[a,b], *Vincenzo Cuomo*[a,b,c], *Maria Macchiato*[b,4], *Lucia Mangiamele*[c], *Filomena Pietrapertosa*[a] [37]

Abstract

Energy-environmental planning must join normative, environmental and socio-economic features to obtain effective strategies aimed to a sustainable development. Therefore a comprehensive methodology for the analysis and the optimisation of the anthropogenic activities system configuration, can usefully support decision-makers in the definition of harmonised sector plans, joining waste management issues with resource use problems and exploiting energy and materials feedback among supply and demand sectors. In this paper we present an innovative application of the Advanced Local Energy Environmental Planning methodology (ALEP), aimed to the definition of optimal waste management strategies which comply with comprehensive as well as sectorial issues.

Keywords: Waste management, Integrated resource planning, Emissions control strategies, R-MarkAl model application.

Methodology

The Advanced Local Energy Environmental Planning methodology was developed under the aegis of the International Energy Agency in the Annex 33 "Energy Conservation in Buildings and Community Systems Programme" (IEA-BCS). According to the ALEP scheme, the planning process can be divided in three main phases [1]:

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37 [a]Istituto di Metodologie Avanzate di Analisi Ambientale, CNR, Tito Scalo (PZ), Italy
[b]Istituto Nazionale per la Fisica della Materia, Unità di Napoli, Napoli, Italy
[c]Dipartimento Ingegneria e Fisica dell'Ambiente, Università della Basilicata, Potenza, Italy
[4]Dipartimento di Scienze Fisiche, Università di Napoli Federico II, Napoli, Italy
*Corresponding author: Monica Salvia, IMAAA-CNR C.da S.Loya 85050 Tito Scalo (PZ), Italy,phone: +39 0971 427207; fax: +39 0971 427271, e-mail: salvia@imaaa.pz.cnr.it

1. Preparation: characterisation of the present energy and material system and definition of the objectives;

2. Project definition: evaluation and choice of measures and strategies;

3. Final strategy assessment: realisation, supervision and monitoring of the planning project.

The planning and the decision-making processes can be usefully supported by a comprehensive model capable of taking into account all the existing constraints and feedback among sectors, which must be based on the following features:

- ***optimising***, to find the most effective solutions either from an environmental and from an economical point of view;

- ***driven by the demand*** of goods and services, to preserve the actual life – style standards and to avoid an over – exploitation of resources;

- ***multi-period and dynamic***, to take into account technology development, depreciation times and costs, catching the main changes from supply market and from social background;

- ***technology and energy oriented***, to find for each considered time period processes and fuels that can satisfy the end uses demand with the lowest environmental impact;

- ***based on sensitivity analysis*** to investigate the steadiness of solutions to the variations of boundary conditions and other superimposed constraints. This feature is particularly important to analyse the structure of end-uses demand, the changes in resources availability and the effects of exogenous environmental constraints.

Such requirements are successfully met by the MARKAL models generator [2], which represents the focal tool of ALEP methodology. It was developed in the late 70's by a consortium of 14 countries under the aegis of an IEA committee (ETSAP, Energy Technology Systems Analysis Development Program). It allows the user to generate models suitable for the study objectives (that is different for time horizon, spatial scale, and technology detail). The original version has been subject to further implementation to take into account different purposes, and nowadays it is widely used by the most of OECD member countries to support energy - environmental planning at national and local scale [3,4,5].

In particular in this study the Regional version (R-MARKAL), up till now used for supra-national analysis, was applied to analyse the waste management system (WMS) and the energy uses of the Civil sector in the Basilicata Region (Southern Italy).

The local system and the REMS

Waste management in the Basilicata Region has been totally based on landfilling of untreated wastes. Such situation is not more sustainable by the light of the new Italian legislative framework (provided by the Ronchi law, n.22/97) which takes in the European normative (EEC/91/156, EEC/91/889 and EC/94/62) [6]. In this context Regional Plans assume a key-role in waste management providing the framework for the development of local scale strategies.

Moreover Waste Management Plans have to single out (Italian law n.22/97, subsection 22):

a) terms and technical criteria to be applied for localising waste processing technologies (except landfills) in industrial areas;

b) number and kind of waste processing and recovery technologies to be settled in the region, taking into account the industrial disposal and recovery supplies;

c) activities and technical requirements for assuring a cost-effective management of municipal waste inside of optimal autonomous areas, as well as for assuring the disposal of industrial waste close to their production (to reduce their movements);

d) an estimation of the value of collecting and disposal costs;

e) criteria to single out areas not suitable for localising waste processing and recovery plants (Province's duty);

f) initiatives for reducing waste production and for promoting its reuse, recycling and recovery as well as initiatives for promoting energy and materials recovery.

The waste management plan has to be co-ordinated with other regional plans (Energy, Transportation, Town planning, Industrial Settlement) exploiting the relationships and feedback among the variables involved.

Thus, an innovative application of R-MARKAL was developed to support the local authority in defining an appropriate configuration of the waste processing technologies which assures the full actuation of the new legislation.. In such an application two separate regions were modelled and jointly optimised. The first region represent the energy supply sector and the waste management system (WMS), whereas the second region represent the demand sectors (Agriculture, Industry, Civil, Transport), taking into account, in particular, energy demand and waste produced. Starting by previous studies on waste composition of MSW and taking into account the territorial features [7], the new waste management system for MSW has been based on an integrated system made up by separate collection, mechanical pre-treatment, incineration, aerobic stabilisation, composting and landfills for residuals [8].

The first step for modelling the local system and its possible evolution deals with the description of the network of technologies and physical flows which constitutes the Reference Energy and Materials System (REMS). In this phase it was necessary to characterise the processes by inputs/outputs of energy and materials, costs (investment, operating and maintenance), and environmental features (e.g. air pollutant emissions, land use). In our case study the REMS takes into account the linkages between the main macroeconomic sectors and the waste management system, showing in detail the chain of waste processing technologies and their feedback with the Civil sector (Figure 1).

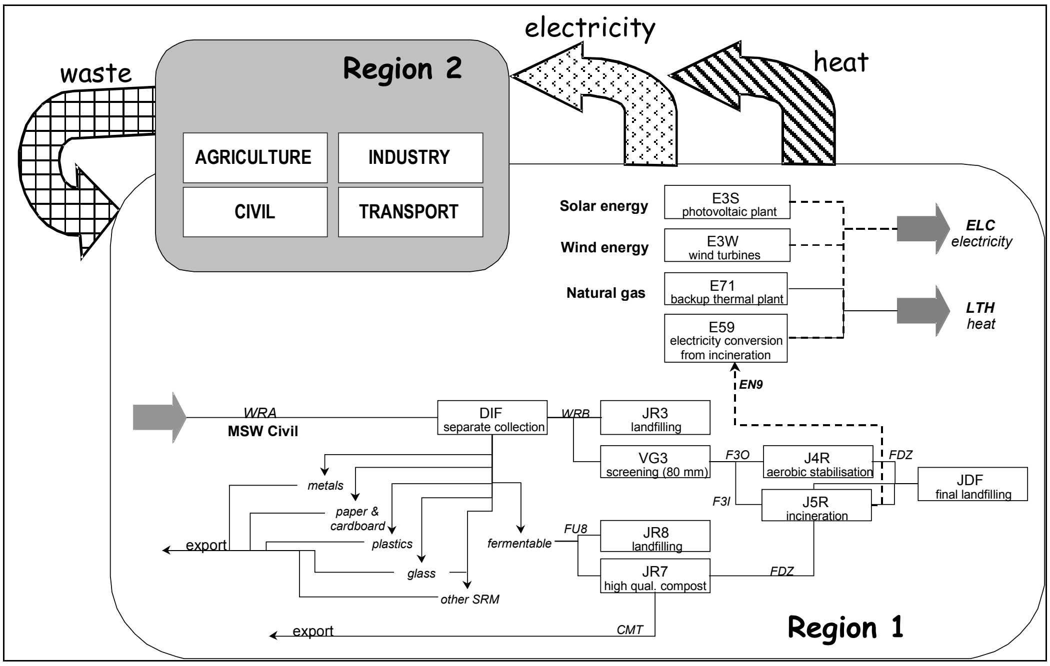

Figure 1: Simplified scheme of the Reference Energy and Materials System (REMS).

Work hypotheses

This study was aimed to evaluate the feasibility of thermal and electric energy recovery from waste incineration in terms of cost-effectiveness as well as environmental impact. Therefore some assumptions were made as regards the future development of the waste management system and the changes in the technological configuration of the Civil sector.

In order to take into account the technologies turnover, a 27-years time horizon was chosen and, following the MARKAL multi-period structure, it was divided into nine time periods of equal length (three years each one).

A scenario by scenario analysis was utilised to assess the impacts of technological changes on the anthropogenic activities system. In order to get a reference baseline, the BASE scenario was considered, taking into account standard technological improvements, insulation interventions in the Civil sector and an increase of separate collection of secondary materials from private households.

Two alternative scenarios were then formulated to model an integrated waste management system and to assess the most suitable options for thermal and electric energy recovery. In particular, the RONCHI1 scenario takes into account the recovery of both electricity and process heat from incineration (co-generation), allowing the fulfilment of dwelling heating demand in the areas surrounding the plant by means of a district heating grid. On the other hand, the RONCHI2 scenario assumes that only electricity is recovered from incineration.

Moreover, in these two scenarios, other renewable energy technologies were introduced and compared to traditional options. Table 2 summarises the main assumptions for the analysed scenarios.

Table 2: Overview of the analysed scenarios.

Scenario	**Features**			**Restrictions**
	CIVIL AND CONVERSION SECTORS	*WASTE MANAGEMENT SYSTEM*		
BASE	• Technological turnover for domestic electrical appliances and boilers • insulating interventions on existing buildings.	• increasing target for separate collection of secondary materials from private households (from 5% to 35%)	• Only landfilling	• Nothing (do nothing scenario)
RONCHI1	As for BASE *plus:* • Wind power and photovoltaic • Solar collectors • District heating		• Integrated WMS • Electricity and heat recovery	• No landfilling for untreated waste (from the III time period)
RONCHI2			• Integrated WMS • Only electricity recovery	• No landfilling for untreated waste (from the III time period)

Results

In absence of environmental restrictions the optimal model solutions point out the minimum cost options for each scenario, allowing the users to assess the effects of changes on resources use (materials and fuels mix, technologies).

Obviously without any constraint, landfilling is the minimum-cost option (the *marginal* technology) for waste management and it is possible to evaluate the variations of recoverable materials amounts due to the increase of the separate collection target from private households (Figure 2).

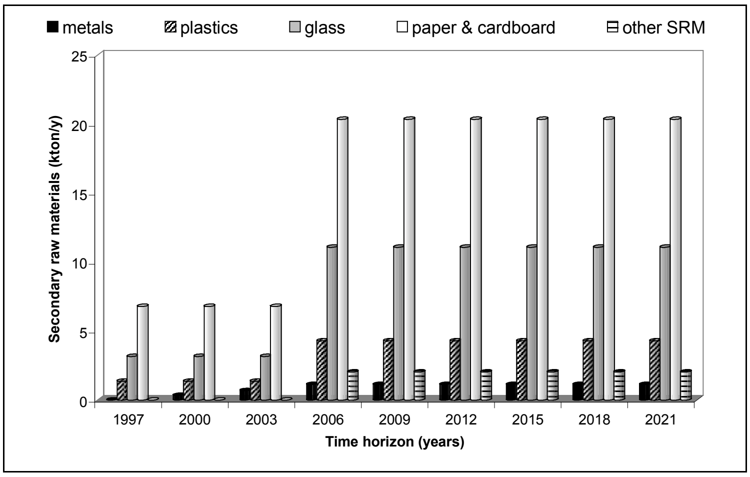

Figure 2: Annual amounts of secondary raw material (SRM) recoverable on the time horizon.

Moreover, the increase of the separate collection target causes also a 30% diminution of the annual volume required for landfilling, reaching about 250000 m^3/y as soon as the 35% target is achieved.

Costs analysis

An integrated waste management system implies higher costs than a landfilling–based system. Therefore to model the boundary conditions provided by the Ronchi Law, an exogenous constraint on landfilling of untreated waste was applied from 2003 (III time period), forcing the model to use the set of alternative technologies constituting the integrated system.

Figure 3 shows the increase of the total discounted system, that is about 19.3% for the RONCHI1 scenario and 18.6% for the RONCHI2 scenario on the whole time horizon.

The higher cost of RONCHI1 proves that, at the actual oil prices and in presence of a widespread natural gas grid, thermal energy recovery is not convenient: in fact, district heating is not activated, because of the huge investment costs of the distribution grid. However, the high costs of the waste management services are mitigated by the revenues from the selling of electric energy recovered: therefore, the larger contribution of the RONCHI2 scenario (11%) compared to the one of the RONCHI1 scenario (4%) allows to obtain a lower total discounted system cost.

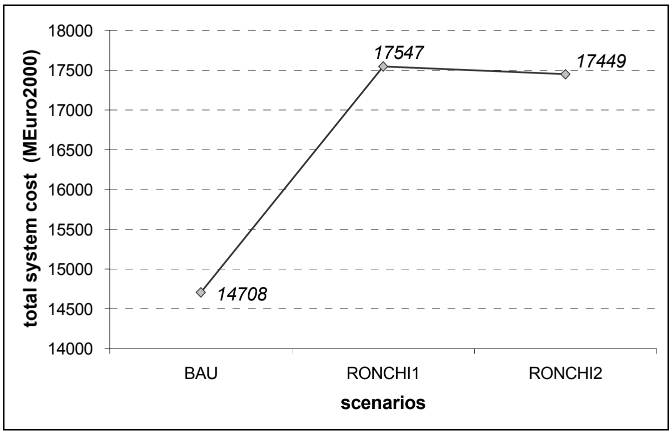

Figure 3: Total system cost in different scenarios.

It is interesting to evaluate the distance from marginality (the zero level, represented by the landfill for untreated waste) of the set of integrated waste processing technologies. Altogether, the integrated disposal cost is about 0.95 MEuro$_{2000}$/kton higher than the only landfilling-based one (Figure 4 a). By investigating the reduced costs of each technology (Figure 4 b) it can be seen that the largest contribution is given by the MSW incineration in terms of operating and maintenance as well as investment costs, whereas screening influences only the operating and maintenance costs and aerobic stabilisation contributes only to increase the investments.

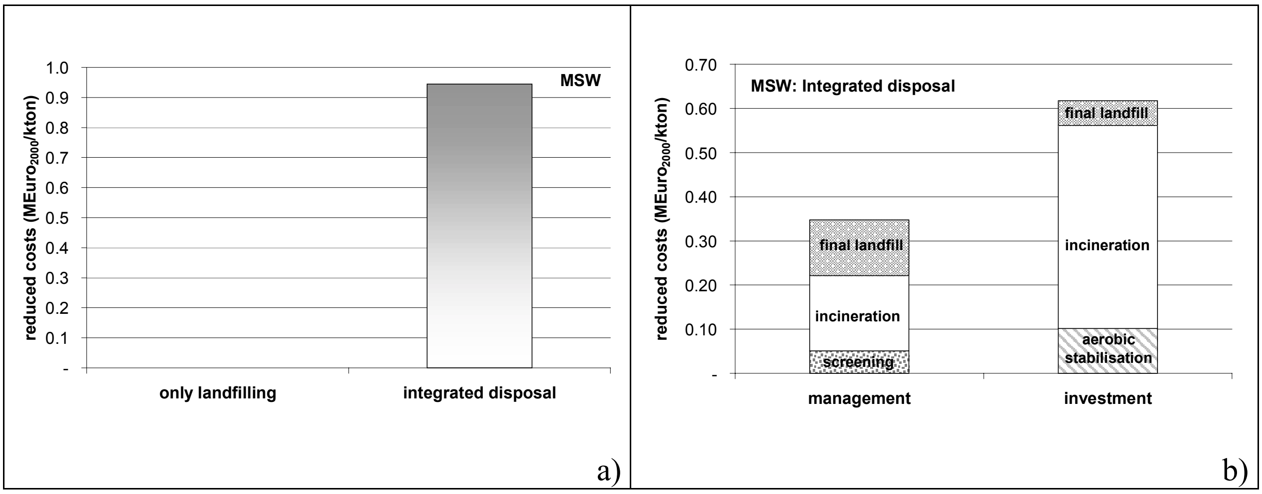

Figure 4: Reduced costs for MSW processing technologies: comparison among landfilling and integrated management (a) and a focus on the contribution of different options (b).

Environmental features

Besides the costs analysis, another crucial point in the choice of the optimal configuration of the integrated waste management system is represented by the environmental impact of each scenario. In our case study, the technological improvements and the optimisation of resources use induce a decrease of almost all the pollutants due to combustion processes, therefore the most interesting species for evaluating the impact of the new waste management system are carbon dioxide (CO_2) and methane (CH_4) [9]. In particular, CO_2 emissions take into account the variations which occur in the combustion process of Civil sector and in electricity production (thermoelectric plants and incineration) whereas CH_4 is a pollutant peculiar to the waste management system.

Table 3 summarises the annual average pollutants emissions estimated from 2003 to 2021 in all the scenarios, comparing them with the 1997 values.

With regard to carbon dioxide, it can be observed on the whole a general reduction which varies from 2.6% (BASE) to 5.9% and 9.1% (respectively for RONCHI1 and RONCHI2). This is due to different causes: the increase of efficiencies in the Civil sector (which causes a 17% reduction of CO_2 emissions, compared to the 1997 value), a lower use of thermoelectric plants (which contribution is respectively -7% and -15% for RONCHI1 and RONCHI2) because of the presence of incinerators, and a lower emission from the waste management system caused by the increase of separate collection (-6%, as in BASE) and by an use of landfilling limited to the inert residuals (-87% in RONCHI1 as well as in RONCHI2).

Concerning the methane emissions, a 5.5% reduction is achieved in BASE by increasing the separate collection target, whereas avoiding the landfilling of untreated wastes a considerable further decrease can be observed (-82%).

Table 3: Emission of air pollutants in different scenarios.

Pollutants	**Contribution**	**1997 [ton/y]**	***BASE***	***RONCHI1***	***RONCHI2***
			Average (2003-2021) [ton/y]		
CO_2	Thermoelectric power plants	1686520,00	1651948,57	1562340,00	1439581,43
	Waste Management System	79550,54	74970,65	37739,84	37739,84
	- Landfills	*79550.54*	*74970.65*	*1786,39*	*1786.39*
	- Incineration	*0,00*	*0,00*	*35953,45*	*35953,45*
	Civil sector	337552,89	278980,33	278980,33	278980,33
	TOTAL - All sectors	**3813303,70**	**3715582,38**	**3588744,41**	**3465985,84**
CH_4	Waste Management System	27987.80	26154.26	614.516	614.52
	TOTAL - All sectors	**33266.30**	**36647.29**	**5893.02**	**5893.02**

Conclusions

This study is a part of a broader research aimed to the definition of optimal resources management strategies at local scale based on a comprehensive analysis of the whole anthropogenic activities system. In this framework the ALEP (Advanced Local Energy Planning) methodology, developed during the IEA-Annex 33, constitutes an innovative tool which has been used here to single out a sustainable configuration of the waste management system for a local case study (Basilicata region, Southern Italy). A R-MARKAL based model was thus developed in order to analyse in great detail the waste processing technologies and their relations with macroeconomic sectors, pointing out energy and materials feedback and the effects of different choices on the whole system in terms of costs and environmental consequences.

The above mentioned methodology, based on a least-cost approach and on a scenario by scenario analysis, was used to assess the effectiveness of electric and thermal energy recovery from incineration in an integrated configuration of the waste management system which satisfies the legislative requirements.

The new configuration of the waste management system causes an increase of the total system cost, but environmental benefits (in terms of carbon dioxide and methane) can be observed due to the changes occurred in the electricity supply system as well as in the waste management configuration (thermoelectric production diminishes and uncontrolled atmospheric releases of biogas from landfills are avoided).

Moreover the comparison among RONCHI1 and RONCHI2 scenarios points out that, in our case study, the absence of a pre-existent district-heating grid and the availability of a widespread natural gas grid prevent the utilisation of heat recovered from incineration, whilst electric energy recovery contribute 11% to the fulfilment of electricity demand, reducing the endogenous production and, consequently, the emissions due to thermoelectric power plants.

These first results highlight the importance of comprehensive tools for approaching the Kyoto Protocol targets [10] at national as well as local scale. Therefore future developments of the study will deal with GHGs-constrained scenarios, and with a more detailed modelling of all the macroeconomic sectors.

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Swedish waste incineration and electricity production

Tomas Ekvall *and Jenny Sahlin*[38]

Abstract

Many life cycle assessments have been carried out to compare different waste management options for specific products or materials. A key issue in the environmental comparison between material recycling and incineration with energy recovery is what source of energy is displaced through waste incineration. In the short-term perspective, the displaced energy is often other waste flows that are deposited at landfills due to limitations in the incinerator capacity. In countries with a ban on landfill of combustible waste, the effect of an increased incineration of a specific product or material might be that the recycling of other materials increase or, in the long-term perspective, that the capacity for waste incineration is expanded. If the expansion is based on conventional technology for waste incineration, such an expansion in Sweden will primarily result in an increased production of district heat from the waste management sector. If the displaced sources of district heat to a large extent are combined production of heat and power based on fuels other than waste, the effect will be a lower electricity production in the district heating systems and, hence, an increased demand for separate electricity production. This would partly offset the long-term environmental benefits of waste incineration. Alternative technologies for waste incineration have been developed based on, e.g., pyrolysis or other types of gasification. These can result in an increased electricity production from waste, but it is unclear if they can compete economically with the conventional incineration technology and if they have reached the technical reliability desired for complete commercialisation.

Keywords: waste incineration, electricity production, technology development, waste management

Background

A large number of life cycle assessments (LCAs) have been carried through to compare the environmental aspects of different waste management options for specific products or materials (Ekvall & Finnveden 2000). A key issue in the environmental comparison between material recycling and incineration with energy recovery is what fuel is displaced through waste incineration (Ekvall 1999). The energy recovered from most Swedish plants for waste incineration is in the form of district heat with an efficiency of 85%. In addition, a relatively small quantity of electricity is produced in a few of the

[38] Energy Technology, Chalmers University of Technology, SE-412 96 Göteborg, Sweden

largest plants (RVF 2000a). The normal electrical efficiencies in these plants are 21% and the total efficiency 80%.

The energy technology displaced through waste incineration is, primarily, the alternative production of these quantities of district heat and electricity (see Figure 1). In early studies of Swedish waste management, the alternative production of electricity was often neglected because it was relatively small. The alternative heat production was assumed to be based on oil (e.g., Tillman et al. 1992) or renewable fuel (e.g., Baumann et al. 1993).

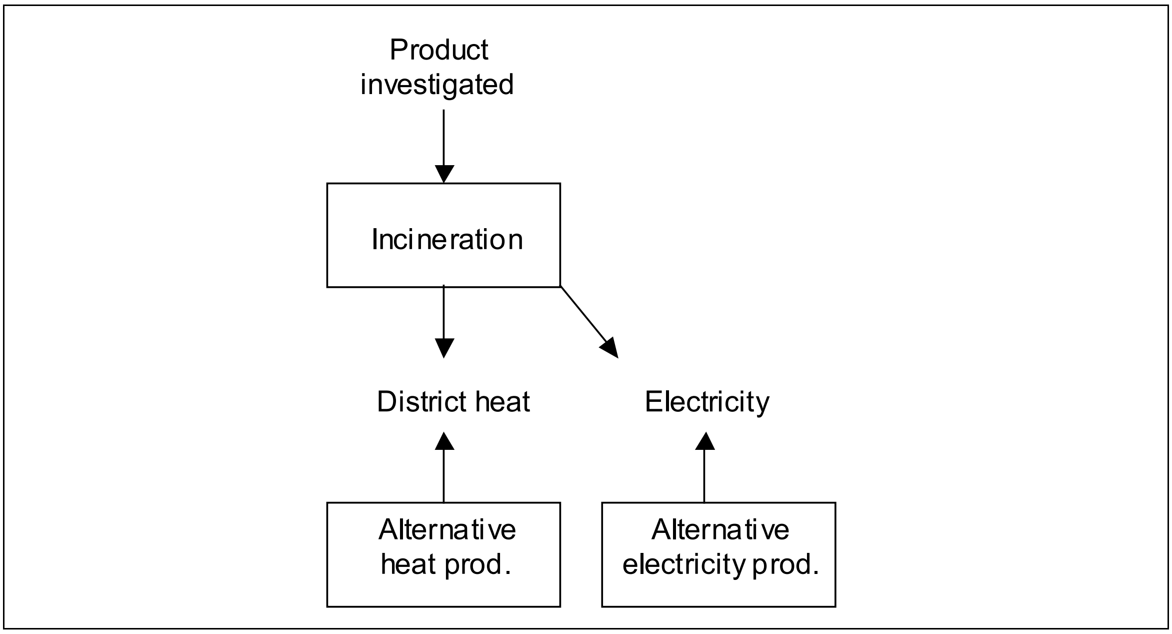

Figure 1. The alternative production of district heat and electricity is a key issue in the environmental comparison between waste incineration and recycling.

Existing conceptual model

Ekvall & Finnveden (2000) argued that, in most countries, the incineration capacity is likely to be much smaller than the total amount of municipal solid waste (MSW). Thus, there are large amounts of solid waste that are currently being deposited at landfills and that can replace the recycled paper in the incinerators. In most Swedish incineration facilities, a reduction in the quantity incinerated of waste paper packagings would, in the short-term perspective, be compensated through increased incineration of other types of waste that currently are being deposited at landfills (ÅF-IPK 1998). This means that the quantity of energy that is recovered through waste incineration is primarily constrained not by the amount of waste but by the incinerator capacity. In spite of existing plans to substantially expand the incinerator capacity, this situation is likely to remain for several years (Sundberg 2000). When the incinerators are constrained by the capacity to handle the energy flows, increased incineration of a specific product or material will not affect the quantities of heat and electricity recovered. Instead, the product investigated will displace other waste flows at the waste incinerators, and these waste flows are likely to end up at landfills (Figure 2). This conceptual model of the short-term effects of incineration has been used in recent LCAs (e.g., Ekvall et al. 2001).

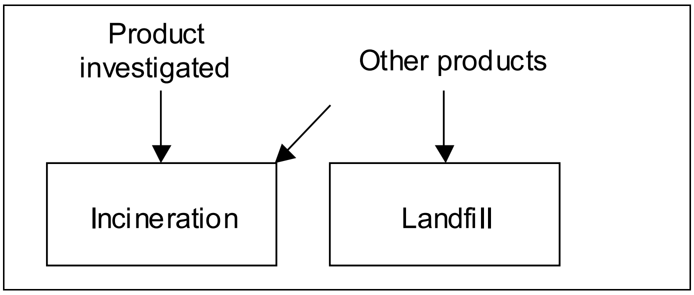

Figure 2. In the short run, the incineration is constrained by the incinerator capacity, and the product investigated will displace other waste flows.

From the year 2002, there will be a ban on depositing combustible waste at landfills in Sweden. Similar bans already exist in certain other countries. As a result of this ban, there are plans to expand the capacity in 16 of the current 23 plants for waste incineration in Sweden. In addition, 15 new plants are planned (Sundberg 2000). As stated above, the capacity will remain constrained for several years in spite of these plans. But, because of the landfill ban, the long-term effect of an increased incineration of a specific product or material in Sweden is likely to be an increased expansion of the incinerator capacity. The increased expansion of incinerator capacity will result in larger quantities of district heat and, to a lesser extent, electricity from the waste sector. This additional energy from waste is likely to replace other energy sources (Figure 3). This conceptual model of the expected long-term effects of incineration has also been used in recent LCAs (e.g., Ekvall et al. 2001).

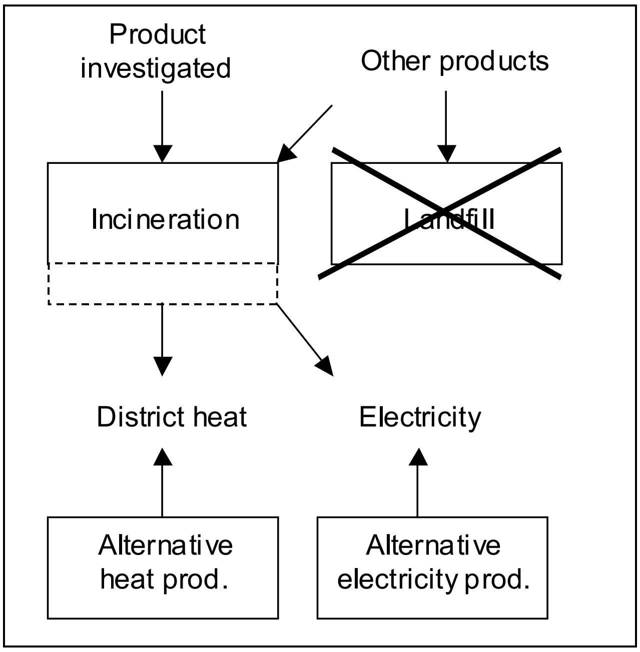

Figure 3. The long-term effect of incineration is likely to be increased incinerator capacity and, hence, increased quantities of energy from waste.

New additions to the model

In the long-term perspective, expansion of waste incinerator capacity competes with investments in other new plants for the production of district heat. For example, in Gothenburg, the expansion plans for the waste incinerator can affect the decision to invest in a combined heat-and-power (CHP) plant for natural gas. A small amount of electricity will be produced through the expansion in the waste incinerator, but a much larger quantity of electricity would be produced in the gas CHP plant (Olofsson 2001). Hence, if the waste incineration expansion replaces the gas CHP investment, the quantity of electricity produced in the Gothenburg district heat system will be smaller. As a result, the demand for separate electricity production will increase. Referring to Figure 3, the long-term effects of an increased incineration of the product investigated might well be that the alternative heat production is reduced but that the separate electricity production is increased.

In Sweden, this effect can be expected whenever heat from an expanded incineration of municipal solid waste with conventional technology replaces heat from a CHP plant. The waste incinerator expansion results in a relatively small increase in the quantity produced of electricity from waste. This is partly due to the fact that many waste incinerators are designed to produce district heat only. But even when the waste incinerator is a CHP plant, the ratio of electricity over heat – the α-value - is small when conventional technology is used, because of the properties of waste as fuel. A typical α-value is approximately 0.25 (see Background).

Alternative technologies for waste incineration have been developed based on, e.g., pyrolysis or other types of gasification. Some of these new technologies have a significantly higher α-value than the conventional technology. If such new technologies are used for the increased waste incinerator expansion in Figure 3, the incineration of the product investigated can result in a reduction in the separate electricity production.

The expansion of incinerator capacity is associated with significant costs. The Swedish government investigates the introduction of a tax on waste incineration, which would further increase the costs of waste incineration. As a result of these costs, it cannot be taken for granted that an increase in the incineration of the product or material investigated will result in a corresponding increase in the incinerator capacity. Instead, the situation can occur again, that the product investigated displaces other waste flows from the incinerator plants. The landfill of these, combustible waste flows will not be allowed, but the recycling or composting etc. can increase. This means, for example, that an increased incineration of paper can result in an increased composting of biological waste and/or an increased recycling of plastics.

If these thoughts are added to the conceptual model in Figure 3, we arrive at a new conceptual model, which is illustrated by Figure 4.

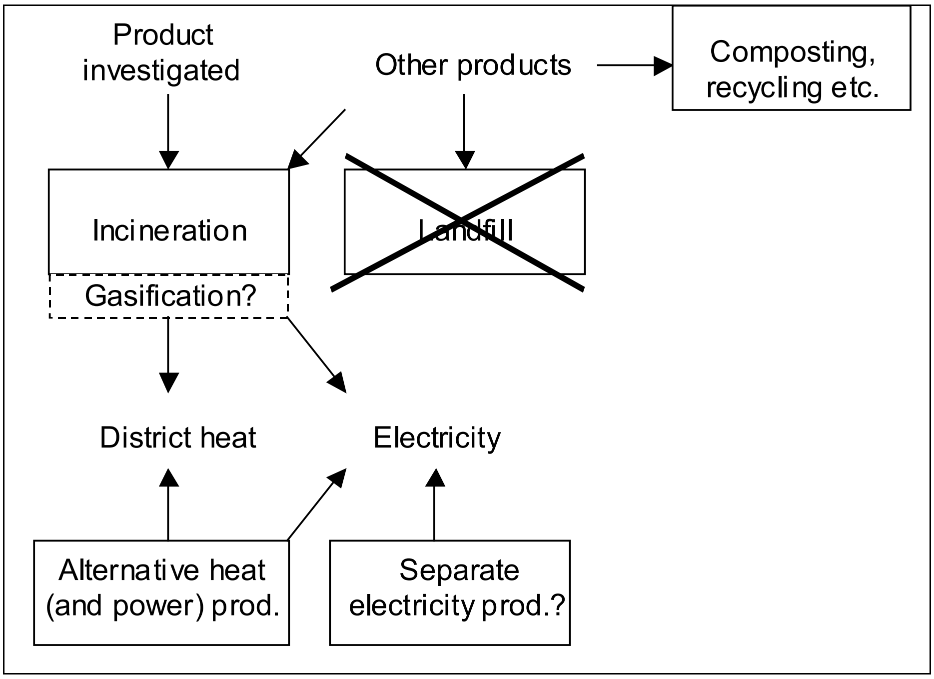

Figure 4. Modified conceptual model of the long-term effects of incineration in Sweden.

Waste gasification

The alternative waste-to-energy technology based on pyrolysis and gasification mentioned above has been applied as waste handling method since the 1970s. Tests with MSW gasification have been performed at e.g. the now closed-down pilot plant for biomass gasification in Värnamo owned by Sydkraft (Sydkraft 2000). However, even though a few plants have been operating commercially for several years, neither pyrolysis nor gasification methods are applied in other countries as fully commercial waste handling technologies on a broad scale. Hence, conventional incineration is more proven than pyrolysis and gasification as a MSW treatment method and pyrolysis and gasification still stand between research and commercialisation.

Gasification is based on heating of the fuel through partial incineration with deficiency of oxygen, preventing complete incineration. The carbonic substances form a fuel gas, the syngas, with a calorific value of about 4-6 MJ/Nm^3, if air is used as oxidation media, depending on the type of waste processed and the characteristics of the applied technology (Dahlroth 1998). In pyrolysis**,** chemical compounds are fractioned by heating at a lower temperature and at complete absence of oxygen, and the necessary temperature is achieved by external heating of the waste. The calorific value of the syngas is about 18 MJ/Nm^3 (Williams 1998). The formed syngas is both cases incinerated in e.g. gas engines, steam boilers or steam turbines, generating electrical energy and heat, at an electric efficiency up to 50% higher than at conventional waste incineration (Morris & Waldheim 1998). At combusting the cleaned syngas in an IGCC (Integrated Gasification and Combined Cycle) system, an electrical efficiency of more than 40% (Morris & Waldheim 1998) and a realistic total efficiency of 80% can be achieved (Rensfeldt 1997). This means that the α-value is approximately 1. Note that the efficiency values are all based on the usage of dried RDF[39] (Refuse Derived Fuel) and losses during the drying process are not considered.

Another main attraction with the pyrolysis and gasification technology is that many processes, instead of ash, produce a stable granulate, that can be more easily and safely utilized or disposed at landfills. Furthermore, additives to construction materials and metal recyclables are other possible ways of reusing the residual products, even though this type of recycling is connected to higher prices and have problems to compete economically under normal market conditions. The gases, oils and solid char from pyrolysis and gasification can not only be used as a fuel but also purified and used as a synthesis gas within e.g. the petrochemical industry.

As mentioned above, there are few existing data on full size, commercially operated plants on technical as well as economical aspects. Capital and operating costs reported by manufacturers differs a lot and it is often unclear how much technical equipment etc.

[39] RDF= Refuse Derived Fuel which is sorted, dried and chopped municipal solid waste in a pellet form.

is included. Furthermore, scale-effects, electricity prices and legal requirements are other factors that will impact on the facility's economy. One can also suspect lower initial prices because of an urge to prove the technology and getting the important first orders. However, a majority of the net operating costs is within the span of 50-100 US$/tonne, to compare with an average incineration cost of 100 US$/tonne in Europe (Juniper 2000). In Sweden, the average fee for incineration was during year 2000 20-50[40] US$/tonne (RVF 2000).

Capital costs vary widely as well, with a majority of the costs for mixed waste projects for pyrolysis and gasification within the span of 200-600 US$/ ton/year installed capacity for plants within the range of 50 000-200 000 tonnes/year (Juniper 2000). Note that the values are uncertain and should be considered guide values. Morris & Waldheim (1998) claim that the capital cost is comparable or even cheaper than costs for conventional incineration plants. In Sweden, a normal capital cost for a CHP waste incineration plant 350-600[41] US$/ ton/year installed capacity for plants within the range of 50 000-200 000 tonnes/year (Dahlroth 1998). All these facts depend heavily on the Swedish governments possible plans to introduce a tax on waste incineration. The judgement of the pyrolysis and gasification technology depends in such a case on among other things, the purpose of the new tax.

If the cost figures above are correct, the utilisation of pyrolysis and gasification technology in Sweden could contribute to an increased electricity production from the waste management system. In a market survey from 2000, waste gasification and pyrolysis are predicted a market share of 20% in Europe by 2008 (Juniper 2000). But the technology apparently needs more time to prove its capabilities and competitiveness before it is introduced on a broad, commercial scale.

Further research needs

The model described in Figure 4 is a conceptual model. Further information is required before it becomes operational in an LCA. The necessary investigations include, but are not limited to, the following:

- an investigation concerning to what extent gasification technology will be used in an increased expansion of waste incinerator capacity,
- an investigation concerning to what extent an increased expansion of waste incinerator capacity competes with investments in CHP plants for other fuels, and

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[40] =200-500 SEK/ton using a conversion ratio of 10

[41] Using the formulas in Dahlroth (1998), the cost is increased with 20% because of the harder demands on installation in flue gas cleaning systems due to the EC incineration directive.

- an investigation concerning to what extent an increase in the incineration of the product investigated can be expected to result in increased composting, recycling etc. of other materials.

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Framework for Sustainable Waste Management - Examples from the building sector

Anders G Klang*[42] *and Per-Åke Vikman*[43]

Abstract

The main focus of this paper is to present a sustainability evaluation framework which includes environmental, ecological and social aspects. The framework has been developed and tested in a case study within the construction and demolition sector. Groups of long term unemployed people were offered environmental education and manual labour, working with recovery and recycling of building and demolition wastes as a form of vocational advancement, within a project carried out in two Norwegian and one Swedish municipality. The paper presents result from a case study of the Swedish part of the project. Two groups of unemployed people have worked within the project for periods between six months and one year. A number of activities were studied in closer detail, and indicators of the different aspects were obtained empirically and through literature studies. Ratio-indicators, linking indicators from different aspects to one and other were calculated. For instance ecological and economical ratio indicators were calculated, resulting in eco-efficiency figures allowing for comparisons of different activities. As for reducing environmental impact, the most promising results were shown within the process of preparing bricks for re-using. This activity also proved to be economically sustainable, but concern for lacking sustainability from a health and work environment perspective is expressed. The discussion analyses the possibility to use the framework for sustainability analysis as intended, and some remaining questions that need to be addressed in further development of the framework. One conclusion is that the data collection to perform this kind of sustainability analysis is resource demanding, and that it therefore would be of interest to identify a smaller number of core indicators.

Keywords: Construction and demolition wastes, life cycle analysis, triple bottom line, sustainable jobs, unemployment

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[42] Mid Sweden University, Department of Natural and Environmental Sciences, Division of Ecotechnics, P O BOX 603, SE-832 23 FRÖSÖN, SWEDEN, +46-(0)63-16 57 60 (phone), +46-(0)63-16 54 50 (fax), anders.klang@ter.mh.se

[43] Mid Sweden University, Department of Natural and Environmental Sciences, Division of Ecotechnics, P O BOX 603, SE-832 23 FRÖSÖN, SWEDEN

Introduction

The concept of sustainability

Sustainability became an internationally wide spread concept through the work of the World Commission on Environment and Development and their report *Our Common Future* [1]. Their often cited definition of sustainability focuses on our obligation to ensure future generations abilities to fulfil their needs, but also to work towards a more equal distribution of wealth within the now living generation. These, and other aspects of sustainability, were also included in the declarations from the UN conference in Rio de Janeiro in 1992 [2]. Social equity, and the need to achieve sustainability through democratic methods were also stressed in Rio. The *Agenda 21* endorsed in Rio suggested that methods for monitoring trends of sustainability needed to be developed, and particularly emphasised the need to integrate environmental accounting with traditional national economics methods. A conclusion that can be drawn from the Rio definitions of sustainability is, that in order to claim that an activity is sustainable, or leads towards sustainability, one is obliged to take *environmental, economical and social* issues into account.

There is also a need to develop methods to assess sustainability and sustainable development on a smaller scale, such as businesses or projects [3]. Setting up targets in a limited system can be a more efficient way to influence behaviour and thereby also reaching effects on macro-scale [4]. Over the years, efforts have been made to find means for companies to integrate other aspects than strictly economical ones in their accounting systems. To do so, it is necessary to identify other, non-traditional, values, and other resource-bases, that are imperative for the operations. One example of such a method, is Sustainable Development Records [5], [6]. Several methods for development of indicators for sustainable development that can be used on projects or physical areas of different scales have been described [7], [8], [9].

Social aspects of sustainability

In this case-study, a number of social aspects of sustainability have been defined. It is argued that unemployment is a contributing factor to social inequities, which in turn leads away from sustainability. Especially unemployment among young people have been shown to be correlated to nervous and depressive symptoms [10], [11]. Other important social aspects taken into account in this study is the psycho-social working environment, as well as the physical working environment. Socially sustainable jobs must provide a healthy working environment.

Economical aspects of sustainability

There are, within the building sector, a number of materials and goods that are hazardous to health and/or environment. Examples of such materials are asbestos mats, joint compositions containing PCB, CFC-gases in cooling installations and many others [12]. Such materials must be taken properly care of regardless of costs. This is ensured through legislation, and through local governments regulations regarding demolition permits. But to convince businesses to reuse or recycle beyond the legislative demands, it is often necessary to point to economical benefits of such operations. In this case study, an activity is regarded as economically sustainable if it generates incomes, or leads to avoided costs, equal to or larger than the costs of the activity.

Environmental aspects of sustainability

There are many different kinds of environmental sustainability aspects. They can be divided into sub groups such as energy consumption, depletion of resources, emissions grouped in effect-categories [13] and loss of bio-diversity in micro and macro scale (=loss of eco-system resilience) [14]. In this case-study, energy consumption and emissions have been given special attention, since these aspects are most widely reported in available life-cycle analysis of building materials.

The case at hand

During two years time, the municipalities of Steinkjer and Trondheim in Norway, and Östersund in Sweden have been co-operating in a project aiming towards re-introducing long term unemployed persons on the labour market. A common interest was identified, to work with issues related to social competence and environmental knowledge as tools to achieve vocational advancement. Another mutual interest was to develop a tool to identify “green sustainable jobs”. That is jobs that have a beneficial impact on the environment, provide a physical and psycho-socially sound working environment and generate a large enough revenue to cover salaries and social fees. There were also suitable objects within the building and demolition sector, to work with on all three locations.

The building and demolition sector is a major source of solid waste in both Sweden and Norway, and a great many other European countries too [15]. In the mid-nineties Swedish contractors and building material producers therefore agreed on a voluntary extended producer responsibility. One of the objects was to decrease the amount of waste brought to landfills, from construction and demolition sites. Measures and time plan to achieve this, and other goals were stated in an action plan [16]. One part of the action plan deals with the subject of education on environmental issues and selective demolition. The Swedish part of the project therefore established a partnership with the local representatives of The Eco-cycle Council for the Building Sector regarding the professional education on selective demolition techniques etc. A programme for the environmental education was developed in co-operation with the Mid Sweden University. The work practise was then performed in periods of six months on a major construction site in Östersund. The main object of this paper is to present a suggested framework for sustainability evaluation, using one activity from this project, namely brick cleansing, as an illustrative example.

Methods and framework design

The suggested framework for sustainability evaluation is based on a “triple-bottom-line” concept [17]. Three corner stone aspects of sustainability are taken into account, and indicators to describe them were chosen, as well as methods for data collection. The three corner stones are environmental, economical and social aspects of sustainability. Figure 1 gives some examples of indicators chosen under each corner stone aspect.

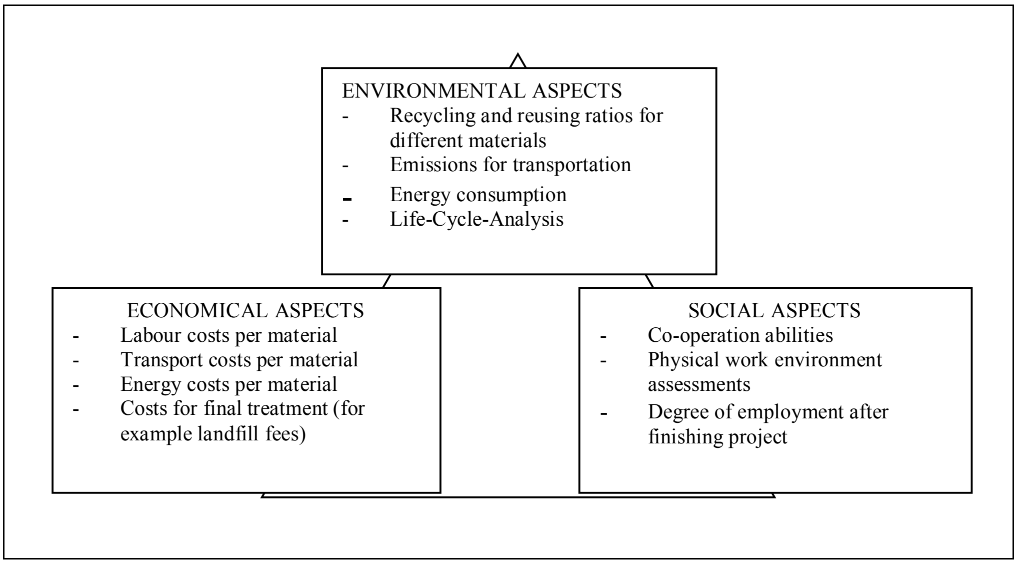

Figure 1. Examples of indicators used to evaluate the sustainability of different waste reusing/recycling activities

The methods for data collection varied from literature studies of life cycle analysis of building materials to questionnaires regarding physical and psycho-social work environment. Energy consumption of different tools and machinery was monitored, as well as transport work. Labour time and costs for dismantling and preparing different materials or products for recycling or reusing were measured by time studies and report cards filled in by the participants.

Ratio-indicators of the eco-efficiency kind

When indicators from the environmental aspects are linked to indicators of economical aspects, a form of eco-efficiency ratio-indicators are obtained [18]. Such indicators can either be formulated as key-ratios describing created economical values per unit of environmental load [19], or as avoided environmental load per unit of invested resource (for instance hours of labour). These indicators can be used to compare different activities with one another, and thereby determine how to allocate a labour resource to achieve largest possible environmental benefits.

Other forms of ratio-indicators

When relating economical aspects to social ones, socio-economic ratio-indicators can be calculated. What is for example, the societal costs of unemployment compared to societal costs of vocational advancement measures? Of importance for this case study is also to what extent there is a societal demand for environmentally beneficial, reused building materials. This can be said to be a ratio-indicator relating environmental aspects to social ones.

Results

The example of brick cleansing

There are a number of factors that have to be taken under consideration regarding the recycling of old bricks. First and foremost, not all bricks are recyclable. The old type of cement-free mortar must have been used. If not, the mortar will be harder than the stone, and the stone will most likely be damaged in the cleansing machine. In this case, nineteen century bricks with the old type of mortar were cleansed. Another important factor is where the reused brick is intended to be used. Old bricks don't have good thermal conductivity values, and are more sensitive to crack-formation due to freezing than newly produced ones. In this case, however, bricks were dismantled from inner-walls and then used to construct new inner-walls on other places but in the same buildings, so these issues are not of importance here.

Environmental aspects of brick cleansing

Mortar was removed from the dismantled bricks by an electrically powered machine. Emissions from electricity production of Swedish average electricity [20] was calculated and compared to a life-cycle analysis of production of new bricks [21]. The comparison shows that the potential environmental effects of brick cleansing only are a small fraction of the potential effects from emissions during new production (see figure 2). The main reason for this is that new production bricks consumes a lot of fossil energy. Through these comparisons, figures could be calculated, describing avoided environmental impact per square-metre of brick wall built using reused bricks instead of newly produced ones.

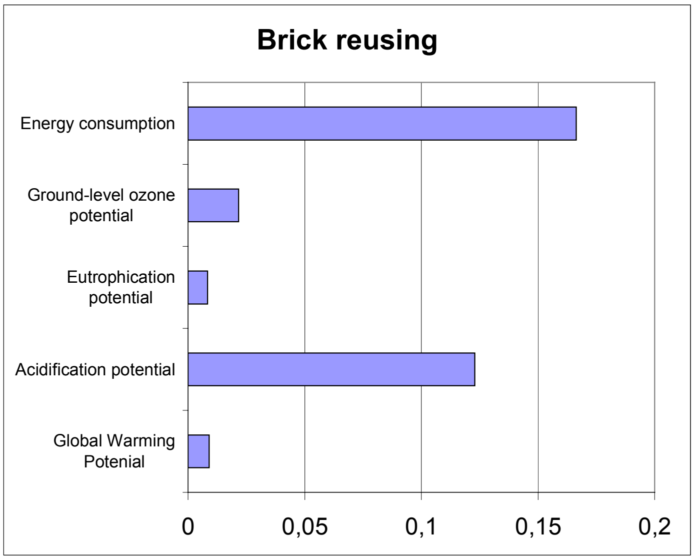

Figure 2: Potential environmental effects from cleansing bricks for reusing, in relation to producing new bricks from virgin raw material. The scale on the x-axis is given in percentage. That is the energy consumption when cleaning a brick, is only approximately 0,17% of the energy consumption of new production.

Economical aspects of brick reusing

The labour effort needed to clean bricks was measured, and the time needed to clean enough bricks for one square metre of brick wall could thereby be calculated. The per brick production cost, including salaries and social fees amounted to 1:75 SEK. The retail prize of newly produced bricks varies between 6 – 7 SEK (excluding VAT) [22]. In total, some 15 000 bricks were cleansed during this project, thereby generating a considerable cut in expenses for the constructor.

Social aspects of brick cleansing

The participants were asked to fill in questionnaires describing how they found the physical working environment during different activities within the project. As it turned out, a majority of the respondents found one or more of the stages of brick cleansing to be unsatisfactory, or highly unsatisfactory from an work environment perspective. The reason for this can be found in the ergonomics of the manual labour of handling quite heavy bricks (average weight approximately 3,7 kg). When handling a large number heavy objects, special attention must be brought to lifting and carrying techniques, and finding appropriate tools to prevent musculoskeletal disorders [23] .

Eco-efficiency ratio-indicators of brick cleansing

Some eco-efficiency indicators were calculated. For instance it was shown that for each labour hour invested in brick cleansing, 233 kg of CO_2-equivalents (Global Warming Potential - 100, [24]) were avoided. Corresponding values for other activities within this project, such as recycling of steel products and reuse of sanitary porcelain products indicate that brick cleansing is the most eco-efficient activity.

Other aspects of brick cleansing

There is a demand in the Swedish society today for reused bricks. Especially of the older kind, which sometimes is used for decorative purposes in interior design. This is shown by the fact that old nineteen century bricks actually sometimes fetch a higher retail price than do newly produced ones [25]. When old bricks are used like this it is not possible to calculate what the environmental effects of the reuse are, since it is unclear what alternative material production it should be compared to.

Conclusion regarding brick cleansing

Brick cleansing appears to be environmentally more sustainable than producing new bricks from clay raw material. The energy consumption of brick cleansing is much lower and, consequently, related emissions are only diminutive in comparison.

From an economical perspective, brick cleansing also seems to be sustainable. The "production" cost of a cleansed brick is roughly a quarter of the retail price of a new brick, leaving ample margin of profit for the interested entrepreneur.

The physical work environment during brick cleansing is not satisfactory, according to the respondents of the questionnaire. There are stages of especially the brick collection phase that need to be solved in another manner to ensure that occupational safety and health regulations are fully met [23].

The eco-efficiency indicators reveal that brick cleansing appears to be an effective activity to invest labour hours in, to reduce energy consumption and emissions of green house gases. In comparison to two other activities in the same project, namely dismantling of steel products for recycling, and reuse of sanitary porcelain ware, brick cleansing turns out to be the most eco-efficient one.

Other aspects also speak in favour of brick cleansing. It is comparatively labour intensive, that is it can provide more work opportunities per produced brick than a traditional new production of bricks, resulting in socio-economic benefits of increased employment. There is also an emerging market for reused building materials, that has been developing over the last few years, indicating both an environmental awareness and a demand for old materials to use in culturally sensitive buildings and surroundings [26].

The conclusion from this evaluation of brick cleansing is therefore that it has the potential of becoming a totally sustainable activity, but that measures to improve physical work environment conditions must be taken first.

Discussion

The object of this paper was to introduce a framework model for sustainability evaluation of waste management measures. The point of such a framework would be to guide policymakers both in municipalities and firms, to decide on appropriate allocation of resources for optimal effects in sustainability respect. It has been shown that the suggested framework can be used to perform comparisons between different activities and thereby draw such allocation conclusions. Another advantage of using the presented framework is that it ensures that a holistic view of sustainability, in the spirit of the Rio declarations, is maintained in the process.

The evaluation process helps in pinpointing weaknesses of different activities. In the example used in this paper, the activity of brick cleansing proved to have weak sustainability depending on physical work environment issues. As a result of this, suggestions on how to improve these conditions have been developed within the project in which the case-study was performed. Other activities could prove to be sustainable socially and environmentally, for instance, but weak in economic sustainability. A municipality authority could use such results of an assessment to provide guidelines for waste treatment or landfill fees for different categories of waste, and thereby enhance the economic sustainability of an operation. Construction and demolition firms would use sustainability evaluations in internal environmental guidance systems, to allocate given resources for optimum effects.

A crucial component in the comparison between different activities is the ratio-indicators of the eco-efficiency type. The presented eco-efficiency indicators are not constructed in the manner suggested by the World Business Council for Sustainable Development. According to their definition of eco-efficiency, it should be expressed as net sales or number of products or services sold per “creation of environmental influence” [27]. This definition of eco-efficiency is not appropriate for evaluations of the type requested by this framework. The type used is instead created by relating *avoided potential environmental impact* per *invested hour of labour*. If the WBSCD definition of eco-efficiency had been used instead, brick cleansing would still be considered eco-efficient, since the environmental impact from the operation of the mortar removing machine is very small, and the generated sales value of the cleansed bricks is quite high. In other cases, however, it is plausible that another definition of eco-efficiency would alter the outcome of the evaluation, something that will be investigated in the future development of the framework. It is likely that different users would prefer different types of eco-efficiency indicator. The WBCSD definition probably would suite a construction or demolition entrepreneur better than it would a municipal authority looking for guidance in establishing landfill fees.

Another conclusion is that data collection of indicators of many different types are resource demanding. One reason for this is that there still is a lack of up to date and reliable life-cycle analysis. In aftermath, it turns out that a relatively small number of indicators suffice to illustrate all types of sustainability aspects. It would be desirable to identify a number of such core indicators that could be used in all waste management sustainability evaluations, but it is uncertain if such indicators really exists.

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Session 4: Summary of discussions

Summarised by Johan Sundberg; Edited by Jan-Olov Sundqvist.

This session was characterised as a mixture of different approaches and themes, which makes it difficult to make any general summary. Several of the presentations considered landfilling or incineration, but also methodology and social aspects were considered.

It was pointed out that boundary assumptions play an important role. It is possible to ”manipulate” the result by making certain assumptions. This can be avoided by using transparency in the reports. Also critical reviews are important.

Landfilling can be modelled in several different ways. For example, the studies presented comprised time horizons from 15 years to 125.000 years.

Some studies also presented a widening approach, e.g. Monica Salvia presented an innovative application of the Advanced Local Energy Environmental Planning methodology (ALEP), and Anders Klang incorporated social aspects in the LCA.

Session 5.

Chairman: Stefanie Hellweg; Secretary: Mattias Olofsson

Jürgen Giegrich
Establishing the Waste Management Plan for Sewage Sludge in Northrhine-Westfalia with the Help of LCA

Sven Lundie
Life Cycle Assessment of Food Disposal Options in Sydney

Oliver Jolliet
Life Cycle Assessment of several processes applied to treat wastewater urban sludge

Patrick Wäger
A Dynamic Model for the Assessment of Plastics Waste Disposal options in Swiss Waste Management System

Discussions

Establishing the Waste Management Plan for Sewage Sludge in Northrhine-Westfalia with the Help of LCA (short presentation)

Horst Fehrenbach, Florian Knappe, ***Jürgen Giegrich****[44]

Key words: sewage sludge, agricultural application, incineration, co-incineration, waste management plan

Summary

With the German waste management law from 1996 the federal states (Länder) were due to present waste management plans for their territory until the end of 1999. Northrhine-Westfalia which is the most populated (18 Mio inhabitants) and most industrialised state decided to have separate plans for each waste category. The waste management plan for sewage sludge should be used to reconsider the different options and set a sign for the future policy towards this waste material.

For this purpose all current sewage sludge streams had been gathered from all authorities and all existing management options had been assessed. LCA was used to decide on the environmental performance of each management option and to find out the most influential system parameters. With this knowledge a county by county approach was used to connect the knowledge of the environmental aspects with the reality in the given region. An overall state-wide management plan had been proposed which is currently discussed in the authorities.

From the 600.000 t (33% dry substance) of sewage sludge in Northrhine Westfalia the major part is used for landspreading which includes agricultural applications and landforming activities e.g. in old open pit mining areas. For agricultural applications and due to the lack of appropriate nearby farmland the sludge is transported in some cases until the Polish boarder. The second largest part is incinerated in various plants reaching from specific sludge incinerators up to co-combustion in thermal power plants, municipal solid waste incinerators and gasification technologies. A small rest is still landfilled.

[44] ifeu - Institut für Energie- und Umweltforschung Heidelberg GmbH, Wilckensstr. 3, 69120 Heidelberg

*Tel.: 06221/476721; Fax.: 06221/476719, e-mail: juergen.giegrich@ifeu.de

The LCA encompassed the typical impact categories plus some calculations on the accumulation of heavy metals in soil. The results which will be presented showed clearly that incineration with the emission standards for waste incineration have an environmental advantage compared to agricultural use for the most of the sludges calculated with their measured contents of heavy metals. A problem for co-combustion in thermal power plants is given with the emission of mercury. An assessment of all power plants using sludge in Northrhine-Westfalia showed that those reducing the mercury emissions with specific cleaning facilities might be the best management solution.

The Federal Ministry of Environment decided to force sewage sludge – maybe defined with a limit of heavy metal content – to be incinerated. German Environment Agency and other federal states are now reconsidering their recommendations as well towards a more costly but more environmental friendly incineration.

Life Cycle Assessment of Food Disposal Options in Sydney

Dr Sven Lundie[45], *Dr Gregory Peters*[46]

Abstract

Food waste processor (FWP) units are mainly used to dispose of waste generated in the kitchen during the preparation of food. A limit or ban on their use has been sought by local council. In response, In-Sink-Erator (an international manufacturer of FWPs) has approached the Cooperative Research Centre for Waste Management and Pollution Control to investigate the environmental, technical, economic and social impacts of their product.

The environmental assessment has been based on Life Cycle Assessment (LCA) approach consisting of goal and scope definition, inventory analysis, impact assessment and interpretation.

The FWP option has been compared with alternative options of home composting, codisposal of food waste with municipal waste and centralised composting of green (food and garden) waste. For the comparison the functional unit was defined as the amount of food waste produced by a household in one year. The environmental assessment comprises energy consumption and contributions to climate change, eutrophication and acidification.

The impacts from one functional unit have been used to extrapolate the overall environmental impacts for greater Sydney area. Different scenarios have been analysed with regards to varying market penetrations of FWP (5%, 15%, 25% and 50% market penetration).

The results from the LCA have been combined with the economic, engineering and social investigation to support a holistic approach to ecologically sustainable decision making.

Keywords: Organic waste, Life Cycle Assessment, food waste disposer, centralised composting, decision making

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[45] Centre for Water and Waste Technology at University of New South Wales, Australia and

Cooperative Research Centre for Waste Management and Pollution Control

Contact: Centre for Water and Waste Technology, University of New South Wales, Randwick NSW 2031, Australia, Phone +61 – 2 – 9385 5097, Fax +61 – 2 – 9313 8426, E-mail S.Lundie@unsw.edu.au

[46] Centre for Water and Waste Technology at University of New South Wales, Australia

Introduction

In-Sink-Erator is the leading supplier of residential, sewer-based food waste disposal systems. Waverley Council has sought a Sydney Water Corporation limit or ban on in-sink food waste disposal (Davis, 1998). In-Sink-Erator approached the Cooperative Research Centre for Waste Management and Pollution Control for assistance regarding an environmental, technical, economic and social assessment of their product. Within this overall project, staff of the Centre for Water and Waste Technology at the University of NSW were asked to perform an environmental life cycle assessment (LCA) of the In-Sink-Erator technology. The aim of this project was to independently assess the environmental profile of the technology on the holistic basis of the ISO14040 standards. However, findings from previous studies have been used for this analysis, eg.: De Koning and van der Graaf (1996), Diggelman and Ham (1998), Griffith (1994), Hardin *et al.* (1999), NYC (1990), Partl *et al.* (1999), Sinclair Knight (1990), and Waste Board (2000). In order to reinforce the credentials of the study, and to obtain the necessary data, a steering committee for the project was constituted including representatives of the NSW EPA, Sydney Water Corporation, the NSW Waste Boards, Nature Conservation Council, Local Government and Shires Association and In-Sink-Erator. Thus, while the study was commissioned by In-Sink-Erator, the primary intended audience is the project's steering committee.

Goal and Scope Definition

Goal of the Study

The main aim of this study is to quantitatively evaluate the In-Sink-Erator food waste processor (FWP) system with the alternative options of:

- home composting;
- co-disposal of food waste with municipal waste; and
- centralised composting of green (food + garden) waste.

The main reasons for carrying out the study are to

- Quantify the overall potential environmental impacts from one functional unit under each of the four waste management options (see section 2.2.1);
- Obtain a detailed picture of potential environmental impacts of the four different waste management options and their (dis)advantages, ie.: energy consumption, climate change, human- and eco-toxicity, eutrophication and acidification;
- Focus on urban Australian conditions (in this study Sydney metropolitan area).

Scope of the Study

Functional Unit

The functional unit ("fu") definition is the disposal of the average amount of food waste produced by a household in one year. This amounts to 182 kg (wet) per annum (BIEC, 1998; CCWB 2000)[47].

System Boundaries

The foreground systems listed above are shown in Figure 11. The study was set in the context of medium to high density residential application of the different waste disposal options in the inner-urban environment of Waverley in Sydney.

Although in other LCA studies, the non-recurrent (construction) impacts associated with long-lived equipment are generally less important than recurrent impacts, the extent of the capital equipment requirement of each food waste disposal option varies considerably, so it was necessary to include the impact of the manufacture of the equipment or facilities in some way. Assuming data on the assembly or construction processes is not available, it is consistent to take into account the production of materials prior to assembly/construction, where the majority of the impacts generally occur (Clift *et al*, 1999).

As the construction of a green waste processing facility for food and garden compostable waste in Sydney would have to begin from scratch, rather than be an expansion of existing infrastructure, the 'proportional approach' was adopted that accounts for the impact of material acquisition for each entire process step and allocates the appropriate proportion of the total to the functional unit.

Some commonality of unit processes was encountered: co-disposal and centralised composting result in the production of leachates which are disposed to sewer as is the In-Sink-Erator liquor, causing additional incremental impacts due to additional volumes of effluent delivered to the sewage treatment plant. The use of the In-Sink-Erator, home composting and centralised composting systems result in a reduction of impacts associated with co-disposal of food waste with municipal waste. Apart from these issues of avoided impacts, no allocation issues were encountered.

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47 Food waste generation currently amounts to 210 kg/hh*a. A reduction to 170 kg/hh*a is expected by 2006 (CCWB 2000).

Assumptions in this LCA

For this LCA several assumptions had to be made based on initial research and decisions made during Steering Committee Meetings. Most important assumptions are listed below:

By-products: Detailed modelling of the beneficial use of by-products, such as compost and biosolids, is not part of the study due to the chemical complexity of these materials. Significant additional research would be required to examine their potential to replace artificial fertilisers. Therefore, in order to compare equivalent systems, it is assumed that the quantities of compost (54.6 dry kg/fu) and biosolids (37.4 dry kg/fu) produced replace the use of cow manure, dry tonne for dry tonne. Avoided transportation from the farms where cow manure is produced to the distribution systems where it could replace compost or biosolids is considered, ie.: 200 km for compost and 100 km for biosolids respectively.

Food waste processor: It is assumed that the FWP operates reliably and no maintenance is required over the lifespan of 12 years.

Home composting: The home composting unit is made of polyethylene. It is assumed that home composting is correctly operated. Therefore food waste degrades under aerobic conditions. The lifespan is assumed with 12 years.

Co-disposal: The disposal of food waste with municipal waste is common practice. It is assumed that degradation in landfill takes place under fully anaerobic conditions.

Centralised composting: Waverley Council currently collects garden waste at the kerb fortnightly (Fuller, 2000). It is assumed that: a) a centralised composting system for food and garden waste runs parallel to the existing MSW system; b) green waste is collected weekly; c) the same number of trucks is required for collecting the green waste as for collecting municipal solid waste; and d) the capacity of the centralised composting facility is 50,000 t/a.

Life Cycle Inventory

Data Collection

Data collection was based on site inspections (Eastern Creek Composting Facility, Malabar Sewage Treatment Plant), Sydney Water planning reports, LCA studies, Australian LCI data (electricity, gas, coal, transportation etc.) and the database of the LCA software GaBi 3v2.

Data is contained in planning documents produced for Sydney Water Corporation by consultants (Sydney Water, 1998) and public data (Sydney Water Corporation, 1999a, 1999b), supplemented by numerous communications with local government, waste managers and suppliers of waste management equipment. Data was also sourced from scientific literature (eg: Kogan and Torres, 1996; Tchobanoglous *et al.*, 1993). The output data from this LCA is considered prospective in nature, since it is based on estimates of future necessary plant and equipment using contemporary operational observations. As the supplier and literature data come from a wide range of sources, it is difficult to make generic statements about accuracy.

Process Tree and Definition of Options

Equipment is assumed to have a lifespan in accordance with the manufacturers' recommendations. For example, the annual impacts associated with construction of sewage treatment facilities are appropriately scaled down by a factor of 35 to take into account their lifespan (Sydney Water Corporation, 1999b). As the materials used in construction of the plant and equipment (primarily concrete and steel) are recyclable and their recycling is considered to reduce environmental impact in other product systems, the disposal of equipment is not considered in this LCA. Additionally, as we shall see, the material and energy flows associated with construction are considerably smaller than those associated with operation of the systems, and it is therefore to be expected that operational issues will dominate the total environmental impact of the system relative to construction and disposal.

In all systems except for the food waste processor option, food waste is treated with other wastes. The impacts of the construction and operation of the systems are allocated according to the proportion of the system load which the food waste represents.

Food waste processor (FWP) option

The system boundaries of the In-Sink-Erator option begin at the point of disposal of household food waste. The foreground system consists of a Model 75 In-Sink-Erator with the associated water supply and sewage treatment facilities. The systems ends with the delivery of biosolids at the application site.

Home compost option

This is the simplest system, connecting the kitchen with the garden via a standard polypropylene compost bin. Manufacture of the compost bin was considered as a background process.

Co-disposal option

This default route of food waste disposal was modelled with initial collection of household waste in an indoor 'kitchen tidy bin'. The waste is transported to a landfill via a transfer station.

Centralised compost option

This is the most complex system: the food waste is collected with green (garden) waste in household and communal bins separate from the general (inorganic) waste stream. The green waste is collected weekly and transported to a centralised composting facility. At the facility the green waste is further treated to compost and delivered to market place.

Life Cycle Impact Assessment

Selection of Impact Categories

The environmental indicator and impact categories chosen for this study are energy consumption, climate change, human toxicity potential, aquatic ecotoxicity potential, terrestrial ecotoxicity potential, acidification and eutrophication. These were chosen on the basis that they are most relevant to the systems undergoing comparison. Other categories have been developed, such as ozone depletion potential, but this is not considered relevant to this study.

Although not strictly an environmental impact category, energy consumption is useful as an indicator of the process intensity and the use of non-renewable resources. It can provide useful explanatory data for examining climate change, and is in any case, energy consumption is a prerequisite for the evaluation of the climate change of process systems. Climate change is obviously of international and local interest, given Australia's status as a major per capita emitter of greenhouse gases. Climate change is usually evaluated on a 20, 100 or 500 year timescale. For this study, the most commonly used timescale has been selected - 100 years. Human toxicity potential of airborne contaminants is of considerable interest in urban environments such as the Sydney region where this study was carried out, and has been studied in depth by Heijungs et al., (1992), Cowan et al., (1995), Lynch et al., (1995), Guinée et al., (1996a and 1996b), Udo de Haes (1996), Hauschild and Wenzel (1998a and 1998b), RIVM *et al*., (1998), and Huijbregts (1999). Aquatic ecotoxicity and eutrophication potential are considered highly relevant to an environmental comparison of these food waste disposal options, given the high moisture content of food and its capacity to generate high quantities of nutrient-enriched leachate on degradation. Since most of the options under study involve large amounts of coal-based electricity and diesel-powered trucks, terrestrial ecotoxicity and acidification potential are also considered necessary impact categories in this LCA.

Life Cycle Impact Assessment Results

Energy consumption and climate change

At 149 MJ/fu, the FWP option is an important user of energy although the centralised composting option uses much more. Some of the energy used in the FWP option is consumed by the unit itself (14% of the total energy consumption), but the biosolids trucking operation consumes more energy (31%). Additionally, the pumping of water to the unit (5%) and from it to the sewage treatment plant (4%) are important energy consuming operations. The balance of the energy is attributed primarily to materials production (see *Figure 6*).

Like the food processor option, the co-disposal option also requires recurrent energy input, in this case, 167 MJ/fu for truck-based transportation (57%) and site works (9%). The two options are not significantly different in terms of total energy consumption.

Running the centralised composting option involves intensive energy use: 546 MJ/fu. 18% of this is used in shredding and particle diminution, sorting shredded materials and turning windrows. As with the co-disposal option, however, this is less than the energy involved in collecting and trucking materials to the central composting facility (57%). While it might initially be expected that the codisposal and centralised composting options would have similar energy demands, the dictates of hygiene require weekly collection of the small amounts of compostable waste. Therefore, the energy consumption is higher on a food waste mass basis due to the diesel fuel consumed covering the distances involved. This cannot be partially allocated to the collection of a larger quantity of municipal waste as in codisposal. The energy required for the home composting option is just that required for the manufacture of the large outdoor composting bin: 14 MJ/fu. Energy savings due to avoided transportation of cow manure are rather small, ie.: -2.3 MJ for FWP and -6.8 MJ for home and centralised composting.

Perhaps surprisingly, the LCA results are markedly different for climate change. The key controlling variable here is the oxygen concentration during the breakdown processes - the metabolic processes are assumed to operate aerobically in home composting and central composting options.

The greenhouse gas emissions of the systems were calculated on two bases: including and excluding biogenic emissions. Home and centralised compost units, biosolids digesters and landfills all release significant quantities of CO_2 as food waste microbially degrades. However, this portion of the total CO_2 emissions is not derived from fossil sources, and in this respect is part of a natural cycle redirected through the human economy (USEPA, 1997). Therefore, Table 1 shows both types of result.

The savings due to avoided transportation of cow manure product are –0.4 kg CO_2-eq. for home and centralised composting and –0.1 kg CO_2-eq. for the FWP option.

Table 4: Contributions to climate change

Option	Greenhouse Emissions (kg CO_2-equivalents)	
	encluding biogenic CO_2	excluding biogenic CO_2
FWP	77	14
Home Composting	67[48]	0.1
Codisposal	172[49]	125
Centralised Composting	112[50]	45

Human toxicity and aquatic and terrestrial eco-toxicity

A comparison of the options in regard human toxicity, aquatic and terrestrial eco-toxicity provides a clear ranking of alternatives: home composting is by far the most environmentally sound alternative.[51] The next best alternative is co-disposal[52], followed by centralised composting[53] and FWP[54]. Home composting has a significant smaller contribution than the other options to all types of toxicity potentials (Figure 7 has a logarithmic scale).

The contribution of home composting to toxity is the result of the production polymer for the compost bin. During the operation of the bin, no contribution to human and eco-toxicity occurs. Savings occur for aquatic and terrestrial eco-toxicity because of avoided trucking operations.

The human toxicity potential of the co-disposal option is mainly caused by trucking operations during the collection of waste (42%) and diesel refining (31%). 9% of the total potential has its source in diesel emissions on site and 12% is emitted by the

48 83.4% anaerobic digestion at Bondi STP, 4.7% trucking and diesel refinery, 4.9% electricity production, 6.4% material production.

49 94.4% from degradation and flaring of organic material and 4.3% trucking.

50 59% breakdown of organic matter, 25% trucking and diesel refinery, 11% on site operation and 4% others.

51 HTP: 0.002 kg dichlorobenzene (DCB) - equivalents, AETP: -0.0001 kg DCB-eq. and TETP: -0.2009 kg DCB-eq. However, in Figure 7 very small positive values are used for AETP and TETP in order to a logarithmic scale.

52 HTP: 0.079 kg DCB-eq., AETP: 0.002 kg DCB-eq., TETP: 4.298 kg DCB-eq.

53 HTP: 0.271 kg DCB-eq., AETP: 0.006 kg DCB-eq., TETP: 16.72 kg DCB-eq.

54 HTP: 0.865 kg DCB-eq., AETP: 0.006 kg DCB-eq., TETP: 34.97 kg DCB-eq.

generation of the electricity required on site. Aquatic and terrestrial eco-toxicity are dominated by the rubbish collection operations (AETP: 53%, TETP 71%) and diesel refining (AETP: 9%, TETP: 12%), while the on-site use of the diesel fuel causes 8% and 10% of the aquatic and terrestrial ecotoxicity potential, respectively. Electricity generation contributes only 2% of the aquatic and 4% of the terrestrial ecotoxicity potential.

Centralised composting provides a similar picture to co-disposal: 41% of human toxicity potential is caused by trucking, 35% by diesel refining and 20% by operation on site. Aquatic and terrestrial eco-toxicity is fully determined by diesel production.

The majority of the toxicity potential caused by the FWP option is the result of the extraction and production of materials rather than the operation of the FWP itself. Fully 72% of the human toxicity potential stems from the production of copper and 22% from electricity generation. 76% of aquatic toxicity originates in material production (46% aluminium and 30% copper). Diesel refining causes 13% of the potential impact and electricity generation 12%. Terrestrial ecotoxicity is similar to aquatic ecotoxicity.[55]

Acidification and Eutrophication

A comparison of the options in terms of acidification potential reveals the centralised composting option has the highest environmental impact: 0.507 kg SO_2 eq. emitted. This is a consequence of the release of nitrous oxides in the combustion of diesel fuel. The FWP system is clearly better, emitting 0.104 kg SO_2 eq., which is better than the co-disposal option (0.124 kg SO_2 eq.). The home composting operation performs the best, with a contribution of 0.001 kg SO_2 eq. to the acidification issue.

The FWP system is the least favoured option in terms of eutrophication potential, emitting 0.176 kg P equivalent to water (river and seas) based on experimental data carried out during this project. This figure is controlled by the ability of the sewage treatment plant to remove nutrients from suspension and from the aqueous phase of sewage. Bondi STP is a “high rate primary” plant, so approximately 50% of the influent nitrogen and phosphorous are released in the treated effluent. Central composting is better in terms of eutrophication (0.104 kg P equivalent). The co-disposal option releases 0.051 kg P equivalent respectively. These figures are lower than the FWP due to the sequestration of nutrients in the landfill. The home composting operation performs well against all the technology-intensive options, releasing only 0.010 kg P equivalent per functional unit under aerobic conditions due to the emission of a weaker leachate.

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[55] 83% has its origin in material production (7% aluminium and 76% copper). Electricity generation for operation results in 12% of the terrestrial ecotoxicity potential impact compared with 5% from diesel refining.

Conclusions

This LCA study allows conclusions to be drawn with regards to the comparison of four different food waste disposal options (food waste processor (FWP), home composting, co-disposal and centralised composting), and additional general conclusions based on these results.

Comparison of four different food waste disposal options

Based on quantitative LCA results an overall assessment can be made with regards to the four options under consideration (see *Table 5*):

– Home composting has the smallest environmental impact on all impact categories. The environmental performance would be even better if recycled material were to be used instead of virgin material in the production of compost bins.

– The FWP unit is second best regarding energy consumption, climate change and acidification.

– Co-disposal is the second best performer in human toxicity, aquatic and terrestrial eco-toxicity and eutrophication potential.

– Centralised composting has a relatively poor environmental performance due to its energy intense collection activities (two collection systems for residual waste and green waste operating parallel on weekly basis). Other collection modes (weekly clearance of split bins or collection of green waste weekly and residual waste fortnightly) would reduce environmental impacts to all impact categories due to smaller energy consumption. These were not quantified in this study, and more research is certainly needed here

Normalisation: This report does not attempt to apply societal values to determine which of the four options is overall the most preferable in environmental terms. However, it should be stated that when normalised to annual per capita emissions, the data indicates the FWP's contribution to eutrophication produces the greatest relative potential impact during food waste disposal (1.2%), followed by the centralised composting unit (0.7%). Remaining normalised impacts contribute with less than 0.4%. The energy consumption and acidification potential of these four options is significantly smaller. This suggests the impacts of centralised composting in these categories should be of lesser concern to policy makers than other impacts made by all options in other categories.

Table 5: Ranking of FWP options based on quantitative LCA results

Rank	Energy	Climate change	Human toxicity	Aquatic ecotoxicity	Terrestrial ecotoxicity	Acidifi-cation	Eutrophi-cation
1	HC	HC	HC	HC	HC	HC	HC
2	FWP	FWP	CD	CD	CD	FWP	CD
3	CD	CC	CC	CC	CC	CD	CC
4	CC	CD	FWP	FWP	FWP	CC	FWP
FWP - food waste processor; HC - home composting; CD - codisposal; CC - centralised composting							

Odour: Impacts on odour could not be quantified due to the absence of uniform data. Sources of odour could be trucking of recycled waste and sewage treatment for the FWP option, operation of the home composting *only* in case of anaerobic digestion, trucking and landfill for co-disposal, and operation of centralised composting.

Human health effects from separate food waste collection: Separate food collection leads to higher concentrations of microbiological agents in households and during the collection of food waste. Effects on human health include respiratory disorders and discomfort of the stomach and intestine. However, reliable statistical information is not available regarding direct effects on human health due to higher concentrations of microbiological agents.

Influence of FWP market penetration

Water use: The usage of FWP leads to an additional water usage of 2.26 m^3 per household per year. A market penetration of FWPs of 5% consumes approximately 1.3 ML/a of additional water in the Waverley municipality.

Variation of total environmental impacts depending on different market penetrations of FWPs: An increase in market penetration from currently less than 5% to 50% would cause a reduction of greenhouse gases (-28%), energy consumption (-5%) and acidification (-7%). However, environmental impacts to other impact categories would rise dramatically due to the contribution made by the extraction and production of materials for the manufacture of FWP, and the additional aqueous nutrient loads emitted. At a market penetration of 50%, human toxicity would increase by factor of 6, aquatic eco-toxicity by a factor of 2, terrestrial eco-toxicity by a factor of 5 and eutrophication by a factor of 2.

Loads diverted from MSW collection: Food waste processors divert food waste from MSW collection in Waverley at the rate of 109 t/a based on 5% market penetration. At the same time 31 t/a of biosolids are captured at Bondi STP and are applied on land. However the use of FWPs increases the total transportation impacts by 2.9% due to the long transport distance from Bondi STP to the application on land.

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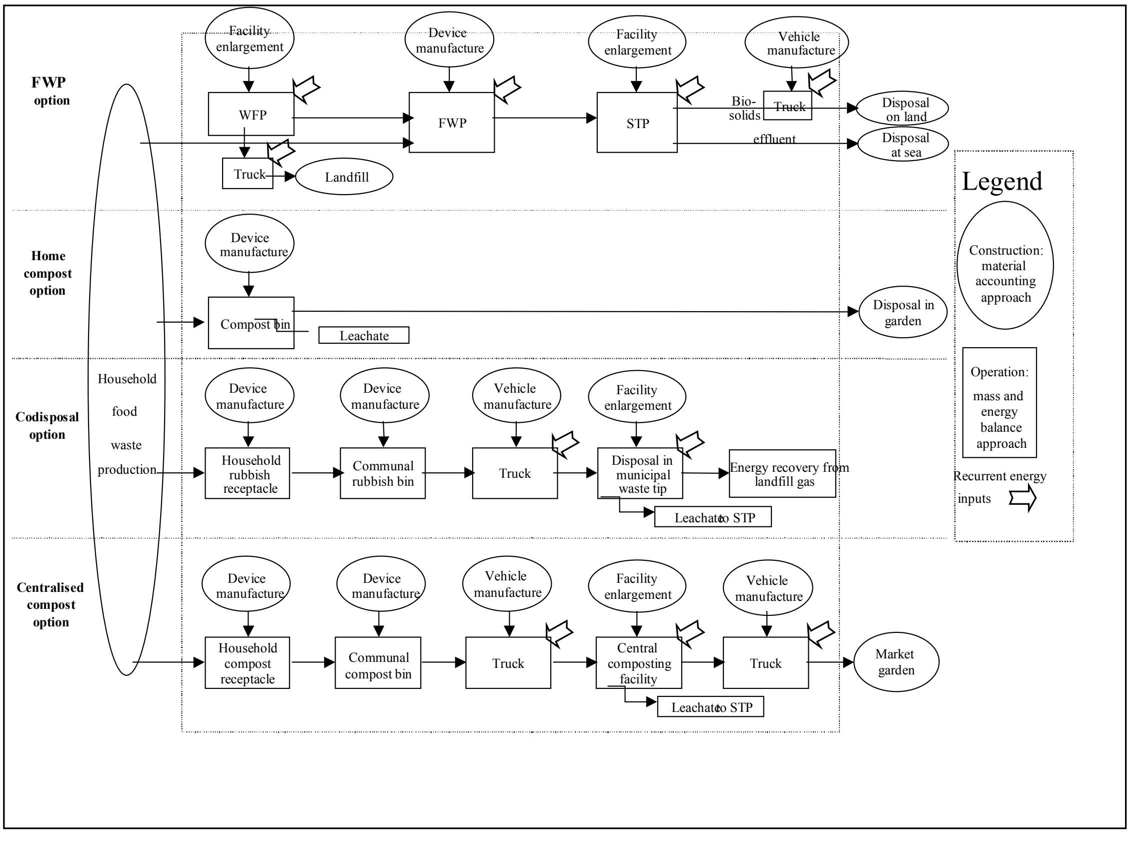

Figure 5: Alternative options for disposal of food waste - LCA system boundary

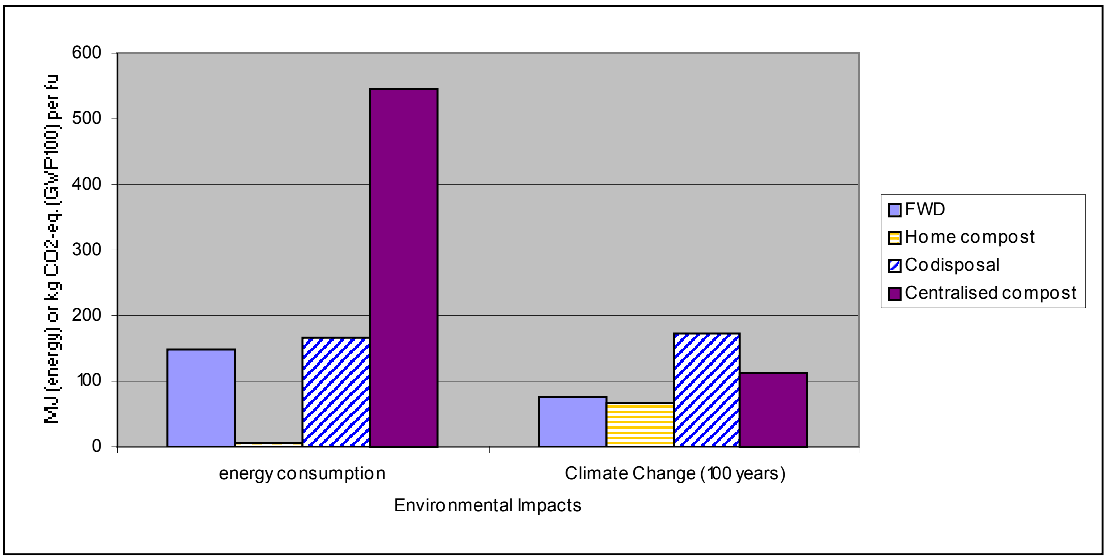

Figure 6: Energy consumption and climate change

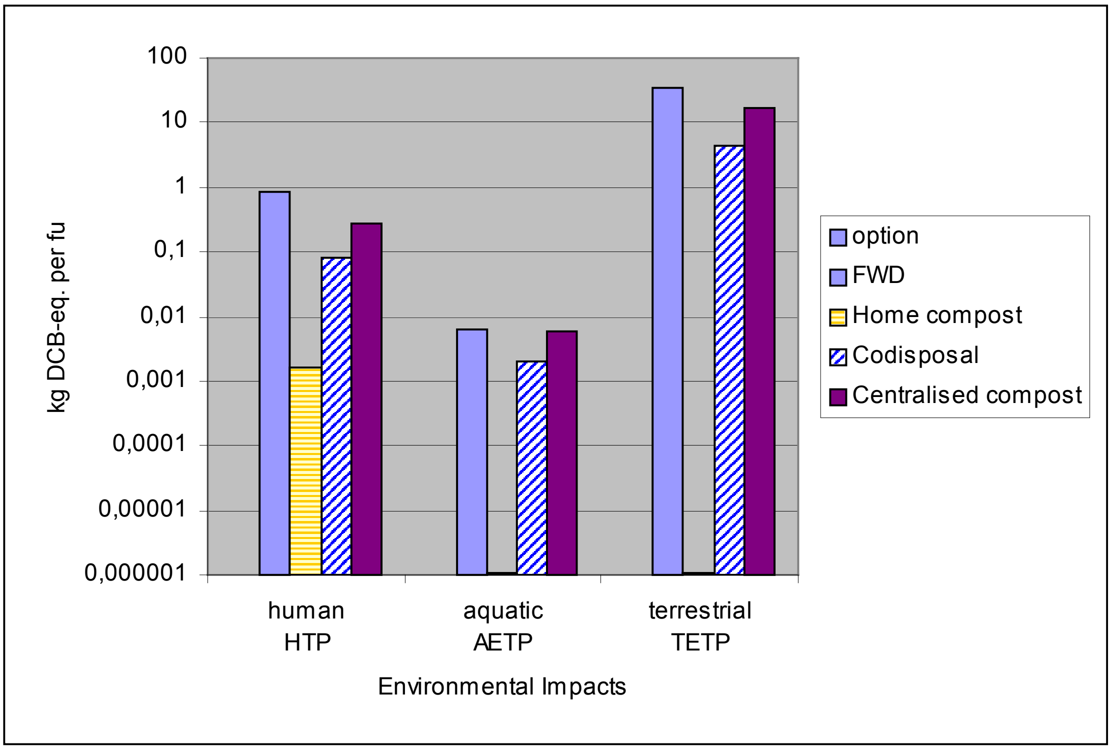

Figure 7: Human toxicity, aquatic and terrestrial eco-toxicity

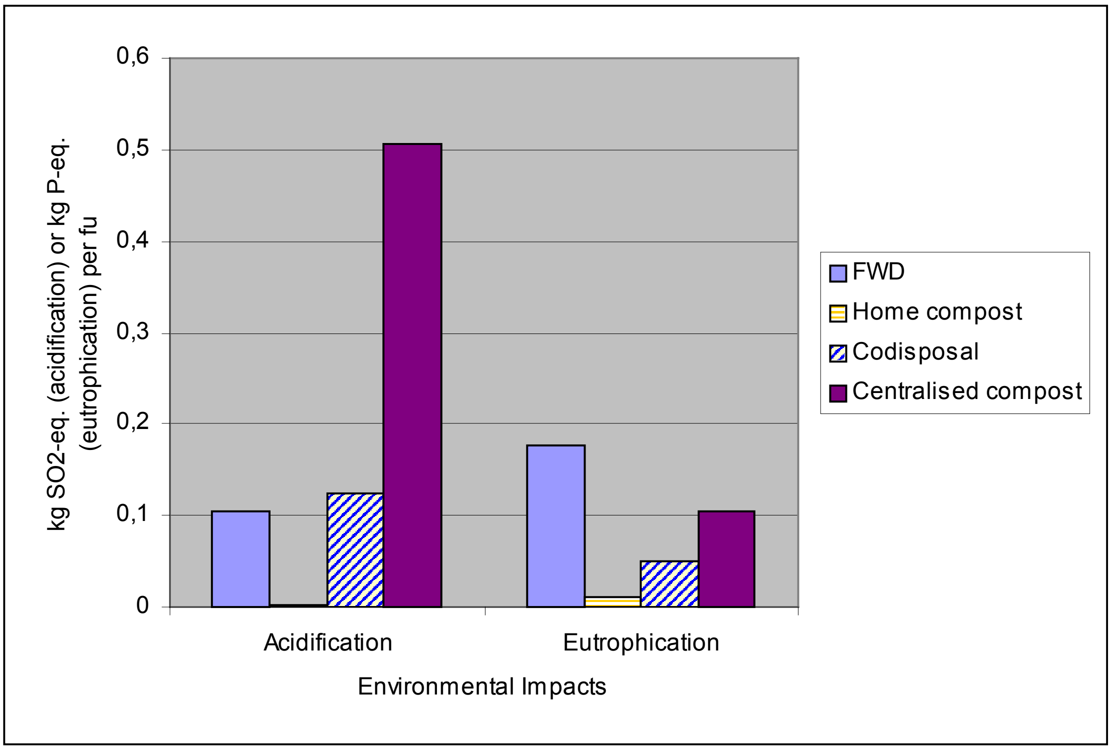

Figure 8: Acidification and eutrophication

A Dynamic Model for the Assessment of Plastics Waste Disposal Options in Swiss Waste Management System

Patrick Wäger[56]***,*** *Paul W. Gilgen*[57] *and Heinrich Widmer*[58]

Abstract

Commissioned by the Swiss Foundation for the Reintegration of Plastic Materials (SSK), an expert model has been developed, which allows to assess plastics waste management strategies by dynamic simulation of ecological and economical effects for time-periods up to 15 years. The key question to be answered was:What will happen, if up to 200'000 tons of plastics waste per year are taken out of the waste stream, which is incinerated in Swiss Municipal Solid Waste Incineration (MSWI) plants, and fed into thermal recovery or mechanical recyling? Simulations on a regional scale indicate that a diversion of industry plastics waste from the waste stream into MSWI plants makes sense from an ecological and an economical point of view, if the MSWI plant takes compensatory measures and investments into additional MSWI-capacities can be avoided.

Key Words: LCA, plastics, system dynamics, sustainable development, waste management

Introduction

The purpose of waste management is to ensure the disposal of waste according to law and under consideration of economical, ecological and social conditions. Waste management systems are strongly influenced by the interactions between the key players involved (public and private disposal organisations, authorities, associations, consumers) and the existing logistics and technical infrastructure.

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[56] Swiss Federal Laboratories for Materials Testing and Research (EMPA), Lerchenfeldstrasse 5, CH-9014 St. Gallen, phone: +41 71 274 78 45, fax: +41 71 274 78 62, E-mail: patrick.waeger@empa.ch

[56] Swiss Federal Laboratories for Materials Testing and Research (EMPA), Lerchenfeldstrasse 5, CH-9014 St. Gallen

[56] Rytec Inc., Alte Bahnhofstrasse 5, CH-3110 Münsingen, phone: +41 31 724 33 33, fax: +41 31 724 33 35, e-mail: widmer@rytec.com

Because of its high functional differentiation as well on the social level as on the te chnical level, which manifests itself in a great variety of specialised key players, processes, facilities and transportation systems, the waste management system shows typical characteristics of highly interrelated, complex systems. Some of these characteristics are [1]:

- their dynamism, which is due to the interaction between processes with different time-scales, feed-backs and the superposition of effects;
- a great number of possible actions, from which the most adequate has to be chosen in order to ensure the survival of the system;
- the emergence of conflicts about the question, which of the possible actions are to be preferred under the condition of limited resources;
- hardly foreseeable consequences of decisions.

In order to master the challenges of such complex systems, approaches are needed, which support the (public) discussion about their design and their regulation. These approaches should, in particular, help to

- integrate the different positions of the key players involved;
- structure the selection process for appropriate actions;
- show the interrelations and feed-backs in the system and reproduce the system behaviour;
- assess possible actions with respect to their effects and to their compatibility with principles of sustainable development.

The numerical method of system dynamics directs one's attention to the dynamic aspects of a system and has the potential to come up to these expectations [2,3,4]. Through application of system dynamics, the key players should in particular be supported to take on more system responsibility and to be better prepared to engage into processes, which are based on the principle of co-operation – e.g. within the scope of so called private-public-partnerships. These are essential prerequisites to solve present and future economical, ecological and social problems of our society.

Materials and methods

As a consequence of the increasing amounts of municipal solid waste in the last few years, additional Municipal Solid Waste Incineration (MSWI) capacities are planned to be installed in Switzerland. These intentions have led to controversial discussions, in which plastics waste plays an important role: Due to its amount (which was estimated at 570'000 tons in 1999), its heating value (which exceeds the heating value of typical MSWI waste by a factor of about 3) and its potential for recycling and thermal recovery in cement kilns, a diversion of plastics waste from MSWI plants into cement kilns and material recycling facilities could allow to avoid at least part of the planned investments for additional MSWI-capacities. At present, the main part of plastics waste - more than 80% - is incinerated in MSWI plants. Materials recycling amounts to less than 10%, thermal recovery in cement kilns to less than 5%.

To de-emotionalize this discussion, the Swiss Foundation for the Reintegration of Plastics Materials (SSK) has commissioned EMPA St. Gallen and Rytec AG, Münsingen, to developed a numerical model for the assessment of different plastics waste disposal options together with key players of the Swiss Waste Management System (i.a. the Swiss Agency for the Environment, Forests and Landscape (SAEFL), operators of MSWI plants, cement kilns and materials recycling plants). Starting point for the development of the model was the question: What will happen, if up to 200’000 tons of plastics waste per year are taken out of the waste stream, which is incinerated in Swiss MSWI plants, and fed into thermal recovery or mechanical recyling? The resulting expert model, called EcoSolver IP-SSK, was built with the software Powersim® Constructor [5], which supports the development of numerical models according to the system dynamics approach. System dynamic models are systems of non-linear ordinary differential equations, which generate simulation results through numerical integration.

Under consideration of the results of the project ‘Dynamics of Waste Treatment ‘ [6], EcoSolver IP-SSK was conceived as a model, which allows to simulate the ecological and economical effects of possible future developments (scenarios) in *regional* plastics waste management systems for time-periods up to 15 years. It consists of an input layer, the model construction layer and an output layer (see figure 1).

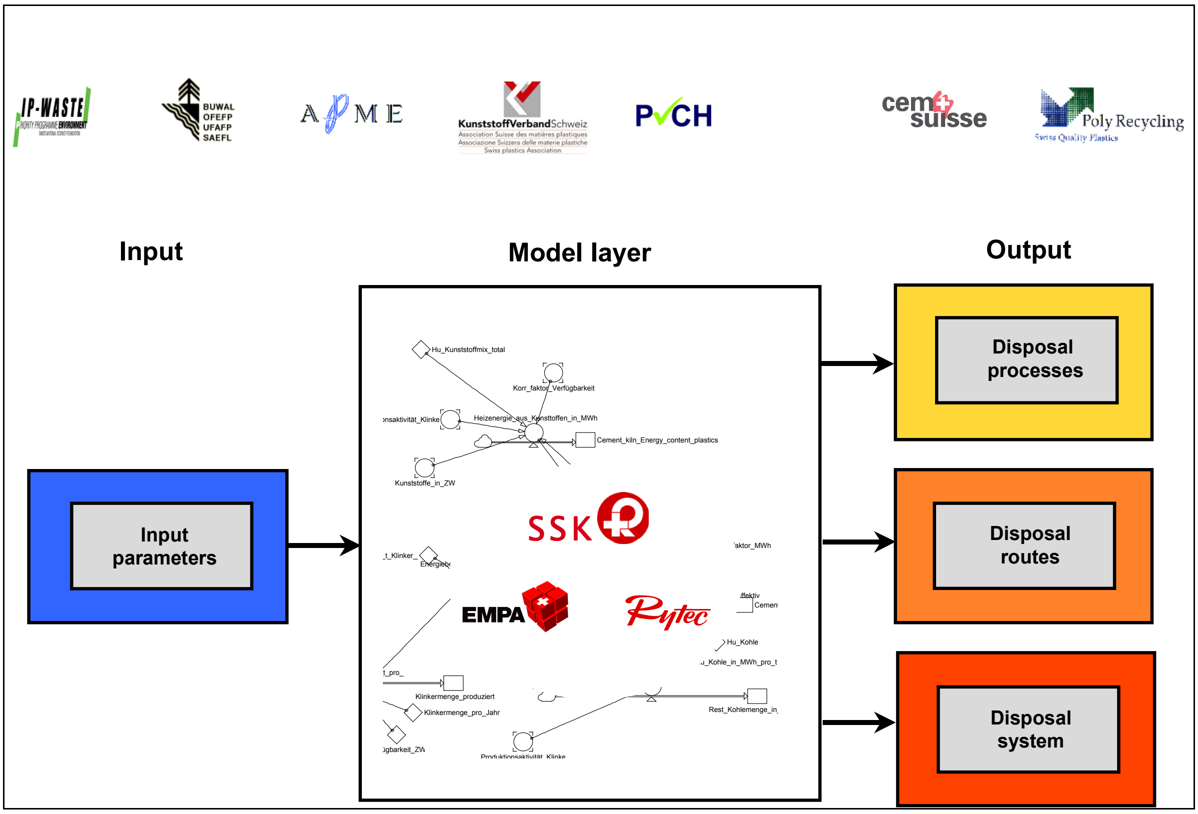

Figure 1. Structure of EcoSolver IP-SSK

On the *input layer*, input parameters are fixed according to the defined scenario. These are, among others, the expected development of the waste streams in the disposal system considered, the amounts of thermally recovered or recycled plastics waste and the transportation distances.

The *model construction layer* includes the core model and additional modules for the ecological and economical assessment of the disposal system looked at. In the core model, the transportation-, collection-, sorting- and treatment-processes related to the disposal routes considered (incineration in MSWI plants, thermal recovery in cement kilns and mechanical recycling) are represented. As a database, indicators (specific energy consumption and specific emissions of CO_2,NOx, Cd, Hg, COD, etc.) for processes and systems typically found in Switzerland have been used. Central element of the core model is the incineration process in MSWI plants, which has been modelled in detail.

The ecological assessment of the disposal system is based on the CML method and the 'basket of products' - principle, which allows a fair comparison of scenarios with different outputs [7,8,9]. In addition to the impact assessment categories (abiotic resource depletion, global warming, ozone layer depletion, etc.), environment indicators are calculated in order to consider important environmental aspects which are not addressed by the CML method (amount of waste materials, heavy metal distribution into different compartments, etc.; see figure 2). For the calculation of the inventories and the impact assessment categories, published (average) data have been used [9,10].

The economical assessment is based on process-specific, economical indicators. For the time being, it is limited to single processes and disposal routes. An approach, which is also based on the 'basket of products'-principle and allows the assessment of the entire system, has been developed [7].

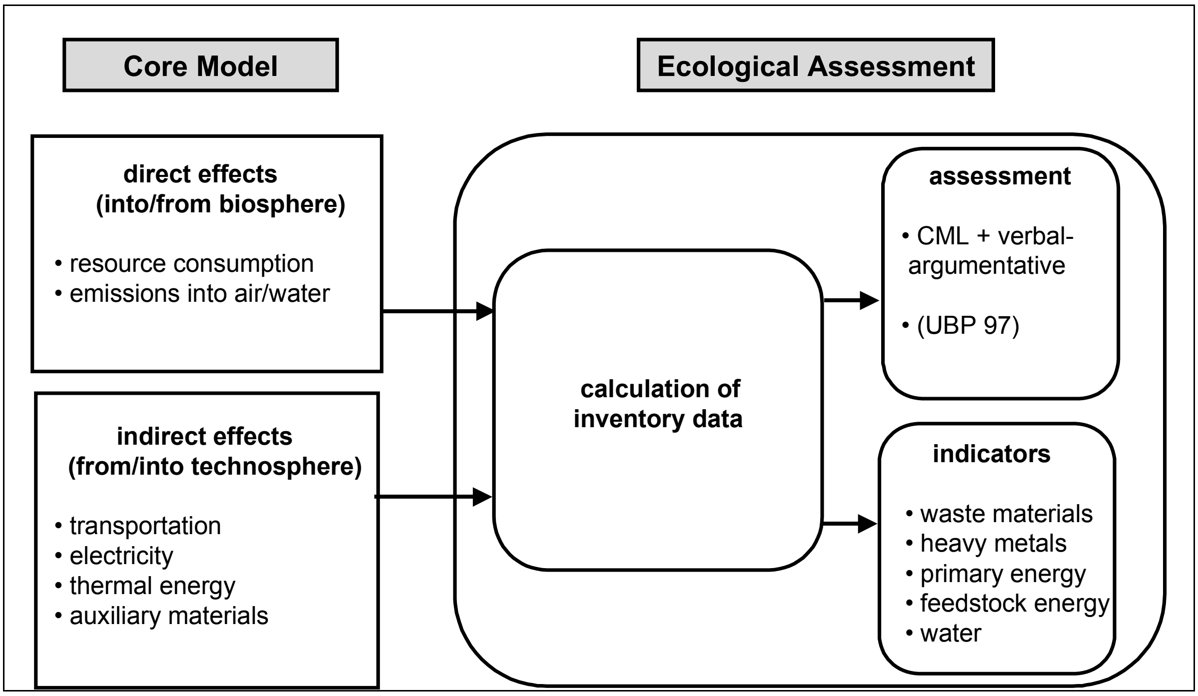

Figure 2. Ecological assessment concept of Ecosolver IP-SSK

The *output layer,* finally, shows the results of a simulation on three different levels:

the processes (e.g. the incineration process in an MSWI plant);

the disposal routes (i.e. the sum of the disposal processes for each disposal option);

the entire disposal system looked at.

In view of an assessment of the simulation results, the output of the simulation of a scenario has to be compared to the output of a corresponding reference scenario.

Results

In order to demonstrate the functionality of EcoSolver IP-SSK, scenarios have been defined, which describe thermal recovery and recycling of plastics waste in a defined model region of 207'000 inhabitants around a MSWI plant with different capacities

[11]. As MSWI plants differ from each other in technology and operation conditions, simulation results are specific for the model region considered. In the reference scenarios, all the waste was incinerated in an MSWI plant.

The scenario presented assumes that a regional equivalent of each 50'000 tons of industry plastics waste for whole Switzerland is diverted into cement kilns and mechanical recycling plants (see figure 3). On a national scale, such a diversion of 100'000 tons of industry plastics waste would - under consideration of the average heating value of plastics waste (9.8 MWh) and the typical heating value of municipal solid waste (3.5 MWh) - set free the capacities of more than two medium-size MSWI plants.

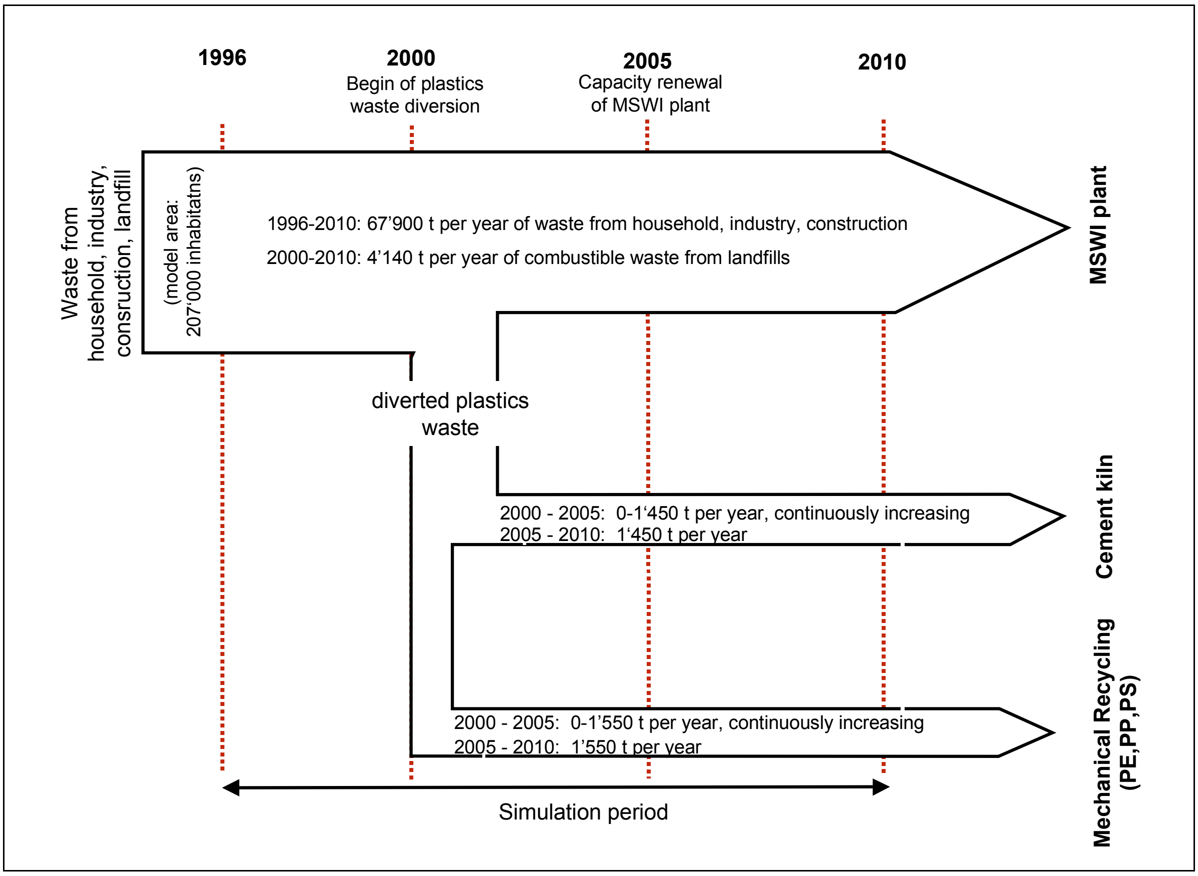

Figure 3. Scenario simulated with EcoSolver IP-SSK

The simulation results for this scenario show that a diversion of industry plastics waste from MSWI plants into cement kilns and mechanical recycling is ecologically beneficial. This is, among others, due to a lower consumption of non-renewable energy and less CO_2-emissions than in the reference-scenario, where the plastics waste is not diverted but incinerated in an MSWI plant. Altogether, the environmental burden of the scenario decreases in all impact assessment categories considered, when compared to the reference-scenario (see figure 4).

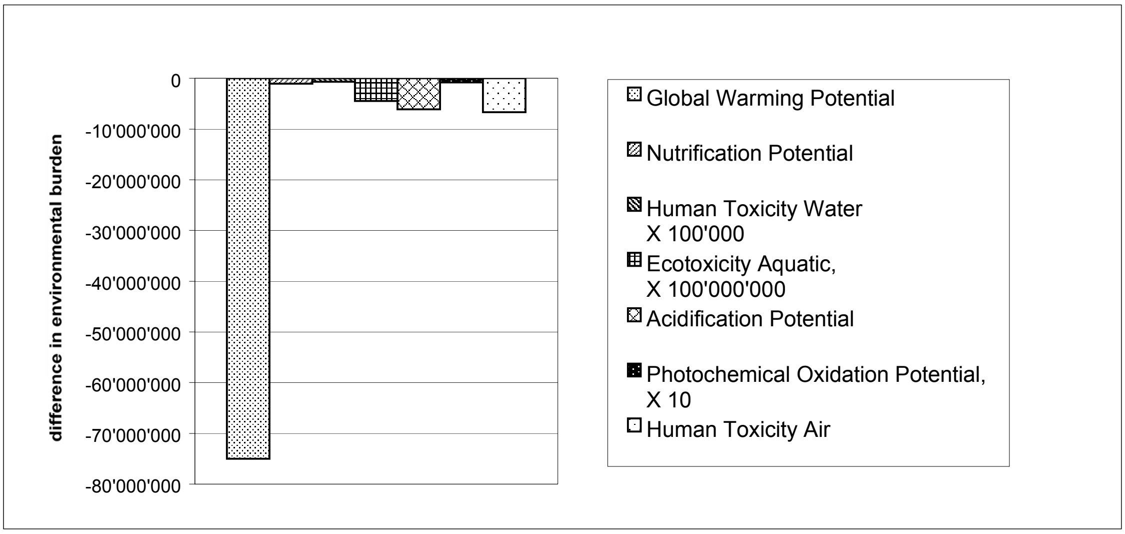

Figure 4. Environmental burden difference between each the scenario and the reference scenario (negative values stand for a lower environmental burden of the scenario, compared to the reference scenario)

From an economical point of view, the diversion of plastics waste has some beneficial effects with regard to the alternative disposal routes (thermal recovery in cement kilns and mechanical recycling). Disposal in the MSWI plant, on the other side, is negatively affected by a diversion of plastics waste, if the MSWI plant does not adapt its oven capacity or compensate the reduction of waste input by waste import from other regions (see figure 5).

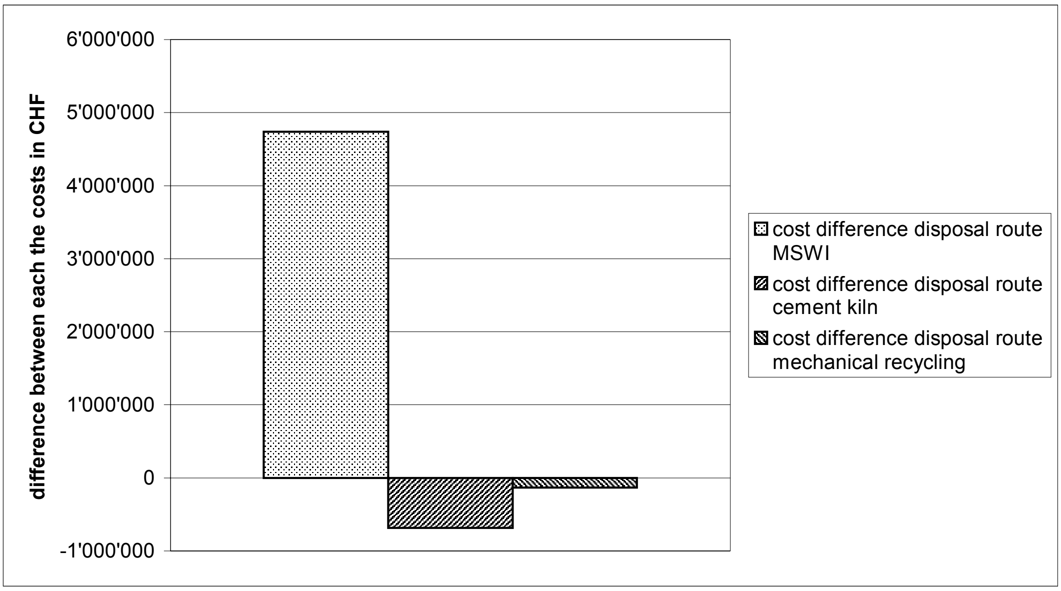

Figure 5. Cost difference between each the scenario and the reference scenario (negative values stand for lower costs of the scenario, compared to the reference scenario)

Conclusions

With ongoing differentiation and increasing interdependencies in the modern (information) society, computer-based numerical instruments to support (public) decision making processes will gain in importance. Prerequites for the application of such instruments are, among others, their user-friendliness as well as the readiness of the key players to get involved into joint modelling- and simulation processes, on the one hand, and to provide the necessary data, on the other hand.

Together with key players of the Swiss Waste Management System, an expert model, which is based on a system dynamics approach, has been developed. The model – called EcoSolver IP-SSK - allows to dynamically simulate plastics waste disposal options for time-periods up to 15 years and to quantify the resulting ecological and economical effects.

EcoSolver IP-SSK has been applied to simulate exemplary scenarios in a model region around an MSWI plant. Simulation results are plausible with respect to the effects caused by a diversion of plastics waste. They indicate that a diversion of industry plastics waste from the waste stream into MSWI plants makes sense from an ecological and an economical point of view, if the MSWI plant takes compensatory measures and if investments into additional MSWI-capacities can be avoided. Hence it seems appropriate to intensify the discussion about such a diversion, under consideration of the regional features of waste management systems and the optimisation potential of socio-technical systems. Further experience in the application of the instrument developed will show whether it can significantly contribute to the promotion of co-operative processes in search of sustainable waste management solutions.

Acknowledgements

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Session 5: Summary of discussions

Summary by Stephanie Hellweg and Mattias Olofsson; Edited by Jan-Olov Sundqvist.

All presentations of this session tried to find the optimal treatment option from an environmental and/or economical point of view for one type of waste (sludge, food, plastic) in a certain region (Northrine-Westfalia, Sydney, Paris, and Switzerland). Several important issues were discussed:

1. Influence of stakeholders on the outcome of the study

Stakeholders often manipulate the study according to their interests, which might lead to scientifically unsatisfying results. One example was that home composting was identified as the best option for food disposal in Sydney due to the non-realistic assumption that composting would be performed under aerobic conditions. The decision to consider home composting as an aerobic process was taken by the steering committee because the "green" groups pushed for it. Investigations in Austria showed that home composting is often performed under anaerobic conditions and, therefore, emissions of CH_4 are produced. Moreover, the nutrient balance was often not optimal since very often too much compost was applied on too little areas. These negative impacts of home composting were not considered in the Sydney-study because of the influence of interest groups.

There was a plea to separate ideology from environmental studies. One way to prevent such manipulation could be the involvement of many parties that control each other. For instance, in the study of sludge disposal in Paris two opposing companies were involved leading to a more balanced distribution of power. In the study of sludge disposal in Northrine-Westfalia, 13 authorities with differing interests were involved.

2. Heavy metals in sludge

All heavy metals in sludge were classified as important. Cd is one chief indicator as it is a very toxic metal that is easily taken up by plants. Zn has a high accumulation rate in soils.

3. Communication of results

It was emphasized that the results of any study need to be presented in a well understandable form. Since politicians usually have a limited time budget, a shortly summarized but clear message is very important.

Session 6. General Discussion – Summary

Chairman: Göran Finnveden
Secretary: Jan-Olov Sundqvist

The general discussion started with a summary of each session[59]. The chairman also declared that the expectations of the workshop (see Introduction) really had been accomplished. The discussion during the session focused on the central questions presented in the invitation to the workshop, i.e. to see if it is possible to draw some general conclusions from the presented studies on

- waste strategies that generally seem to be favourable or not favourable
- methodological approaches and assumptions that can govern the results
- lack of knowledge.

Environmental aspects

Considering the environmental aspects it was noted that based on the presentations by Roland Clift, Göran Finnveden, Ola Ericsson, Jûrgen Giegrich, Michael Eder, Hannes Partl, Monica Salvia, Anders Klang, Olivier Jolliet and Patrick Wäger, the waste hierarchy seems to be valid.

Material Recycling	<	***Incineration*** ***Anaerobic Digestion***	<	***Composting***	<	***Landfilling***
Small environmental impact						Large environmental impact

(the sign < should be interpreted as ”better than”)

Material recycling is in general more favourable than incineration, which in general is more favourable than landfilling (considering environmental aspects). Anaerobic digestion is difficult to compare with incineration, in some aspects it is better and in some aspects worse. Anaerobic digestion is in general better than composting, and also landfilling.

[59] The summaries are presented in the end of respectively session. *Editors note*.

Composting was specifically addressed in the study presented by Sven Lundie where home composting (using ideal data) was preferable over landfilling which in turn was preferable over central composting.

It was noted that incineration in many studies was treated as one single technique where in fact there are different incineration techniques available. Some were compared in the study by Karl Vrancken.

Sludges where specifically discussed in the studies presented by Jürgen Giegrich and Olivier Jolliet. Both concluded that incineration of sludges is preferable over agricultural applications.

Economic aspects

The presented economic studies didn't show consistency in the result:

- Marcus Carlsson Reich showed that
  incineration < *recycling* < *landfilling* (the sign < should be interpreted as "better than")
- Monica Salvia showed that
  landfilling < *incineration*
- Hannes Partl showed that
  recycling < *incineration*.

The difference between the results seems to depend on the system boundaries used ("which costs and for whom").

An aspect which was not considered in these studies was the time spent by households for source separation which often is required for recycling but not incineration or landfilling. This means that recycling often requires more time by households compared to incineration and landfilling and a controversial question is how this time shall be valued in cost-benefit analysis.

Key aspects

A number of key aspects that can influence the results were identified:

1) Avoided products (heat, electricity, material, fertiliser produced from waste). In order to have comparable systems, treatment methods that produce products (such as heat or materials) are typically credited for this by subtracting the environmental impacts from the avoided products. It is generally recognised that the results are very much influenced by the choice of the avoided products and their production techniques. For example, the results for incineration will change drastically if it is

assumed that electricity produced from the incineration of waste is replacing electricity from coal-fired power plants, or nuclear power plants or wind power. Important aspects to consider include: (a) what is the avoided material, (b) are there impurities in recycled and avoided material, and (c) can the recycled products reach a market

2) Efficiency in power plants, heating plants etc. and also recycling plants.
3) Emissions and impacts from recycling plants (there seems to lack of data from recycling plants; these are often assumed to work ideally, while especially incineration and landfilling is modelled from field data.
4) Landfilling models, e.g. time frames.
5) Final sinks: there should be a distinction between temporary sinks (landfills) and final sinks
6) Local conditions and local impacts are often neglected. Models should be more flexible to give possibilities for dynamic approaches.
7) Electricity is in several studies assumed to be produced by coal as a marginal electricity source. In long terms (and large changes) also other electricity sources can be of relevance, for example nuclear power or renewable sources.
8) Choice of alternatives to compare can have an influence on the conclusions drawn.
9) Stakeholders' influence. Stakeholders can influence the results for example by influencing which alternatives are studied. At the same time it is noted that the presence of stakeholders are necessary to make scenario choices that are applicable to decision-makers.
10) Linear modelling. The models used are typically linear and non-linear aspects are therefore difficult to capture. An example where non-linear aspects can be of importance is the difference in environmental impacts from recycling when moving from low to high recycling rates.
11) Data gaps. Especially data on toxic substances where identified as an important data gap.

The points identified as key aspects also correspond to major research needs.

APPENDIX 1. PROGRAM

Monday 2 April

Jan-Olov Sundqvist

Introduction to the workshop

Session 1.

Chairman: *Göran Svensson*

Simon Aumônier
Identifying the Best Practicable Environmental Option: application of LCA and other decision-aiding tools

P.H.Brunner
Material Flow Analysis as a Decision Support Tool for Goal Oriented Waste Management

Roland Clift
CHAMP – A new approach to modelling material recovery, re-use, recycling and reverse logistics

Discussions

Session 2.

Chairman: *Simon Aumônier*

Göran Finnveden
Treatment of solid waste – what makes a difference?

Ola Ericsson
Energy recovery and material and nutrient recycling from a system perspective

Jürgen Giegrich
Reconsidering the German Dual System for Lightweight Packaging

Karl Vrancken
Evaluation of waste treatment processes for MSW rest fraction

Discussion

Session 3.

Chairman: *Paul Brunner*

Michael Eder
Long-Term Assessment of different waste management options – a new integrated and goal-oriented approach

Hannes Partl
Assessment of Kerbside Collection and Recycling Systems for Used Packaging Materials in Australia

Juha-Heikki Tanskanen
Integrated approach for formulating and comparing strategies of MSW management

Tomas Ekvall
Assessing external and indirect costs and benefits of recycling

Marcus Carlsson Reich
Economic assessment of waste management systems – case studies using the ORWARE model

Mattias Olofsson
A comparison of two different system engineering approaches for analysing waste-to energy options

Discussions

Tuesday 3 April

Session 4.

Chairman: *Johan Sundberg*

Jan-Olov Sundqvist

Some methodological questions and issues that are of great interest for the result

Göran Finnveden

Environmental effects of landfilling of solid waste compared to other options – assumptions and boundaries in life cycle assessment.

Stefanie Hellweg

Time- and site-dependent LCA of thermal waste treatment

Markku Pelkonen

Landfill emissions and their role in waste management system

Monica.Salvia

Toward a sustainable waste management system: a comprehensive assessment of thermal and electric energy recovery from waste incineration

Jenny Sahlin

Waste incineration and electricity production

Anders G. Klang

Framework for sustainable waste management – examples from the building sector

Discussions

Session 5:

Chairman: *Stephanie Hellweg*

Jürgen Giegrich
Establishing the Waste Management Plan for Sewage Sludge in Northrhine-Westfalia with the Help of LCA (short presentation)

Sven Lundie
Life Cycle Assessment of Food Disposal Options in Sydney

Oliver Jolliet
Life Cycle Assessment of several processes applied to treat wastewater urban sludge

Patrick Wäger
A Dynamic Model for the Assessment of Plastics Waste Disposal options in Swiss Waste Management System

Session 6:

General Discussions. Conclusions

Chairman: *Göran Finnveden*

End of workshop

APPENDIX 2. PARTICIPANTS

Participants in workshop System Analyses of Integratyed Waste Management, 2-3 April 2001

Name	*Post adress*	*Title of paper*
Asefa, Getachew	Industrial Ecology, IMA, KTH (Royal Institute of Technology), 100 44 Stockholm, Sweden, Getachew@ima.kth.se	
Aumônier, Simon	Environmental Resources Management, Eaton House, Wallbrook Court, North Hinksey Lane, Oxford, OX3 7AQ, Great Britain, sxa@ermuk.com	Identifying the Best Practicable Environmental Option: application of LCA and other decision-aiding tools
Birgisdottir, Harpa	Milj¢ & Ressourcer Danmarks Tekniske Universitet, lca2@imt.dtu.dk	
Bjarnadóttir, Helga Jóhanna	Linuhonnun Consulting engineers Sudurlandsbraut 4a, 108 Reykjavík Iceland, helga@lh.is	
Brunner, Paul	Institute for Water Quality and Waste Management, Vienna University of Technology, Karlsplatz 13/226.4, A-1040 Vienna, Austria, Paul.h.brunner@awsnt.tuwien.ac.at	Material Flow Analysis of Integrated Waste Management
Bäckman, Petra	Chalmers Industriteknik Ekologik, Chalmers Teknikpark, 412 88 Göteborg, Sweden. Petra.backman@cit.chalmers.se	
Clift, Roland	Centre for Environmental Strategy University of Surrey, Guildford Surrey GU2 5XH UK r.clift@surrey.ac.uk	CHAMP – A new approach to modelling material recovery, re-use, recycling and reverse logistics
Dahlroth, Björn	STOSEB, Box 1023 101 38 Stockholm, Telefon: 08-402 21 90, Bjorn.Dahlroth@stoseb.se	

Eder, Michael	Institute for Water Quality and Waste Management, Vienna University of Technology, Karlsplatz 13/226.4, A-1040 Vienna, Austria, m.eder@awsnt.tuwien.ac.at	Long-Term Assessment of different waste management options – a new integrated and goal-oriented approach
Ekvall, Tomas	Energy Technology, Chalmers University of Technology, SE-412 96 Göteborg, Sweden, tomas.ekvall@entek.chalmers.se	Assessing external and indirect costs and benefits of recycling
Ericsson, Ola	KTH Royal Institute of Technology, Department of Industrial Ecology, SE-100 44 Stockholm, Sweden, Olae@ima.kth.se	ORWARE study – methodological discussion
Finnveden, Göran	fms Environmental Strategies Research Group, Box 2142, SE-103 14 Stockholm, Sweden, finnveden@fms.ecology.su.se	Treatment of solid waste – what makes a difference?
Giegrich, Jürgen	IFEU-Institut Heidelberg, juergen.giegrich@ifeu.de	1. Reconsidering the German Dual System for lightweight packaging. 2. Establishing the waste management plan for sewage sludge in Northrhine-Westfalia with help of LCA (short presentation)
Granath, Jessica	IVL Swedish Environmental Research Institute, Box 21060, SE 100 31 Stockholm, Sweden, Jessica.granath@ivl.se	
Hannerz, Nils	Miljö och Mediagruppen Södra Larmgatan 12 411 16 Göteborg, nils.hannerz@home.se	
Hellweg, Stefanie	Chemical Engineering Department Safety & Environmental Technology Group, H-Zentrum, Universitaetsstr. 33, UNL CH-8092 Zurich, Switzerland, hellweg@tech.chem.ethz.ch	Time- and site-dependent LCA of thermal waste treatment
Högberg, Sverker	Swedish Environmental Protection Agency, Stockholm, Sweden, Sverker.Hogberg@environ.se	

Johansson, Jessica	fms Environmental Strategies Research Group, Box 2142, SE-103 14 Stockholm, Sweden	
Jolliet, Olivier	EPFL-DGR-GECOS, olivier.jolliet@epfl.ch	Life Cycle Assessment of several processes applied to treat wastewater urban sludge
Klang, Anders	Mid Sweden University. Dep. of Natural and Environmental Sciences. Division of Ecotechnics, PO BOX 603 , SE-832 23 FRÖSÖN, anders.klang@ter.mh.se	Framework for sustainable waste management – examples from the building sector
Lind, Per	fms Environmental Strategies Research Group, Box 2142, SE-103 14 Stockholm, Sweden	
Lundborg, Anna	STEM, Swedish National Energy Administration, anna.lundborg@stem.se	
Lundie, Sven	Centre for Water and Waste Technology University of New South Wales Randwick NSW 2031 Australia, S.Lundie@unsw.edu.au	LCA of Food Disposal Options in Sydney
M¢ller, Jacob	Danish Forest and Landscape Research Institute , H¢rsholm Kongevej 11 DK-2970 H¢rsholm	
Olofsson, Mattias	Chalmers University of Technology, Energy Systems Technology Division, SE-412 96 Göteborg, Sweden, olma@entek.chalmers.se	A comparison of two different system engineering approaches for analysing waste-to-energy options
Partl, Hannes	Nolan-ITU, Australia, Hpartl@nolanitu.com.au, hannes.partl@tyrol.at	Independent Economic Assessment of Kerbside Collection and Recycling Systems for Used Packaging Materials in Australia
Pelkonen, Markku	Helsinki University of Technology, Laboratory of Environmental Engineering, PO BOX 6100, FIN 02015 HUT (Espoo) Finland, markku.pelkonen@hut.fi	Landfill emissions and their role in waste management system

Reeh, Ulrik	Danish Forest and Landscape Research Institute , H¢rsholm Kongevej 11, DK-2970 H¢rsholm, Tel: (+45) 45 17 82 63 , Fax: (+45) 45 76 32 33 , ULR@FSL.DK	
Carlsson-Reich, Marscus	IVL Swedish Environmental Research Institute, Box 21060, SE 100 31 Stockholm, Sweden, Marcus.carlsson@ivl.se	Economic assessment of waste management systems – case studies using the ORWARE model
Robertson, Kerstin	SP, Borås, Sweden, Kerstin.robertson@sp.se	
Sahlin, Jenny	Energy Technology, Chalmers University of Technology, SE-412 96 Göteborg, Sweden, jenny.sahlin@entek.chalmers.se	Waste incineration and electricity production
Salvia, Monica	Istituto di Metodologie Avanzate di Analisi Ambientale, CNR C.da S.Loya 85050 Tito Scalo (PZ), Italy, salvia@imaaa.pz.cnr.it	Toward a sustainable waste management system: a comprehensive assessment of thermal and electric energy recovery from waste incineration
Sundberg, Johan	Chalmers University of Technology, Energy Systems Technology Division, SE-412 96 Göteborg, Sweden	
Sundqvist, Jan-Olov	IVL Swedish Environmental Research Institute, Box 21060, SE 100 31 Stockholm, Sweden, johan.sundberg@profu.se or josu@entek.chalmers.se	Some methodological questions and issues that are of great interest for the result in LCA and system analyses
Svensson, Göran	Miljökompetens AB, miljokompetens@swipnet.se	

Tanskanen, Juha-Heikki	Finnish Environment Institute Tel. +358 9 4030 0421, Fax +358 9 4030 0491 P.O.BOX 140 * FIN - 00251 Helsinki Finland, Juha-Heikki.Tanskanen@vyh.fi	Integrated approach for formulating and comparing strategies of MSW management
Teller, Phillippe	Laboratoire de Chimie Industrielle, Belgium, Ph.teller@ulg.ac.be	
Watz-Wolf, Camilla	IVL Swedish Environmental Research Institute, Box 21060, SE 100 31 Stockholm, Sweden, Camilla.Wolf-Watz@ivl.se	
Winkler, Joerg	Technical University of Dresden, Joergwinkler@kpmg.com	
Wittgren, Hans B	Tema Vatten Linköpings Universitet 581 83 Linköping, hb.wittgren@tema.liu.se	
Vrancken, Karl	Vito (Flemisch Institute for Technological Research), Boeretang 200 - B2400 Mol, karl.vrancken@vito.be	Evaluation of waste treatment processes for MSW rest fraction
Wäger, Patrick	Sustainability in Information Society EMPA St. Gallen, Lerchenfeldstrasse 5 CH-9014 St. Gallen, Patrick.Waeger@empa.ch	A Dynamic Model for the Assessment of Plastics Waste Disposal options in Swiss Waste Management System

IVL Svenska Miljöinstitutet AB

IVL Swedish Environmental Research Institute Ltd

P.O.Box 210 60, SE-100 31 Stockholm
Hälsingegatan 43, Stockholm
Tel: +46 8 598 563 00
Fax: +46 8 598 563 90

P.O.Box 470 86, SE-402 58 Göteborg
Dagjämningsgatan 1, Göteborg
Tel: +46 31 725 62 00
Fax: +46 31 725 62 90

Aneboda, SE-360 30 Lammhult
Aneboda, Lammhult
Tel: +46 472 26 77 80
Fax: +46 472 26 77 90

www.ivl.se