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Environmental systems analysis tools for decision-making

LCA and Swedish waste management as an example

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Licentiate thesis

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Environmental Strategies Research

Stockholm 2006

Titel:
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Cover page photo: Marianne Lockner

TRITA-SOM 06-002
ISSN 1653-6126
ISRN KTH/SOM/--06/002--SE
ISBN 91-7178-304-0

Printed in Sweden by US AB, Stockholm, 2006

Abstract

Decisions are made based on information of different kinds. Several tools have been developed to facilitate the inclusion of environmental aspects in decision-making on different levels. Which tool to use in a specific decision-making situation depends on the decision context. This thesis discusses the choice between different environmental systems analysis (ESA) tools and suggests that key factors influencing the choice of ESA tool are object of study, impacts considered and information type regarding site-specificity and according to the DPSIR-framework.

Waste management in Sweden is used as an example to illustrate decision-making situations, but discussions concerning choice of tools are also thought to be of general concern. It is suggested that there is a need for a number of ESA tools in waste management decision-making. Procedural tools like Environmental Impact Assessment (EIA) and Strategic Environmental Assessment (SEA) should be used e.g. by companies applying for development of waste management facilities and by public authorities preparing plans and programmes. Within these procedural tools analytical tools providing relevant information could be used, e.g. Risk Assessment (RA), Life Cycle Assessment (LCA) or Substance Flow Analysis (SFA). Analytical tools may also be used separately. If the decision-making situation concerns a choice between different waste management options, such as recycling, incineration and landfilling, environmental aspects could be assessed using LCA or Material Input Per unit Service (MIPS). To study certain substances within the waste system, RA or SFA could be used.

An LCA of different strategies for treatment of municipal solid waste was made. A conclusion from this study is that the waste hierarchy is valid as a rule of thumb. Suggestions resulting from this study are that decisions promoting recycling of paper and plastics should be pursued, preferably in combination with decisions promoting the use of plastics replacing plastics made from virgin sources. The study further identifies a need for limiting transportation by private car for options requiring source separation of waste. When recycling is not an alternative, incineration is in general preferable to landfilling. Key issues that may affect the ranking of the waste treatment options include alternative energy sources, the material the recycled material replaces and the time perspective chosen.

It is suggested that LCA may be a useful tool in waste management, both on its own and as a part of an SEA. Results from LCAs can provide advice on ranking of alternatives. More importantly, key assumptions and value choices that may influence the rankings can be highlighted and thus made clear to the decision-makers. In general, LCA results are not site-specific and provide information in the form of potential environmental impacts, and thus could be combined with other tools if other type of information is needed.

Sammanfattning

Beslut kan fattas baserat på olika slags underlag. Ett stort antal verktyg har utvecklats för att underlätta att hänsyn tas till miljöaspekter i beslutsfattande på olika nivåer. Vilket verktyg som är lämpligt att använda i en specifik beslutssituation beror på sammanhanget. I avhandlingen diskuteras val av miljösystemanalytiska (MSA) verktyg för beslutsunderlag. Studiens objekt, vilka aspekter som hanteras samt vilken sorts information som efterfrågas, med avseende på platsspecificitet och utifrån DPSIR-ramverket, föreslås vara viktiga faktorer som påverkar val av verktyg.

Avfallshantering i Sverige används som exempelområde för beslutssituationer, men resultat och diskussion kan troligen även vara av mer generellt intresse. Besluts-situationer inom avfallsområdet kan vara av olika slag och det finns ett behov av olika MSA verktyg. Processverktyg som miljökonsekvensbedömning (MKB) och strategisk miljöbedömning (SMB) bör användas av företag som t. ex. ansöker om byggnadslov för avfallshanteringsanläggningar och av myndigheter som förbereder planer eller program som kan antas ge upphov till betydande miljöpåverkan. Inom dessa processverktyg kan analysverktyg som riskbedömning eller livscykelanalys (LCA) användas för att ge relevant information. Analytiska MSA verktyg kan även användas separat. Om beslutssituationen rör val mellan olika avfallshanterings-strategier, som återvinning, förbränning och deponering, kan miljöbedömning göras med hjälp av LCA eller MIPS (Material Intensity Per unit Service). För att studera specifika substanser inom systemet kan riskbedömning vara lämpligt.

En LCA av olika avfallshanteringsstrategier för brännbart och komposterbart hushållsavfall har utförts. En av slutsatserna från studien är att avfallshierarkin gäller som tumregel. Resultat från studien ger underlag som stödjer fortsatt främjande av materialåtervinning av plast- och pappersavfall. Helst i kombination med främjande av användning av återvunnen plast i produkter där jungfrulig plast annars skulle ha använts. Studien påvisar också vikten av att begränsa hushållens transport av avfall med personbil. I de fall där material-återvinning inte är ett alternativ visar studien att förbränning, i de flesta fall, är mer fördelaktigt än deponering ur miljösynpunkt. Antaganden av stor vikt som kan påverka utfallet mellan de olika avfallshanteringsstrategierna gäller alternativa energikällor, vilket material återvunnen plast respektive papper ersätter samt tidsperspektiv.

LCA föreslås som ett användbart verktyg för att ge beslutsunderlag inom avfallsområdet, dels i sig självt och dels inom processverktyg. Resultat från livscykelanalyser kan visa på för- och nackdelar med olika alternativ. Dessutom kan viktiga antaganden och val som kan påverka bedömning av olika alternativ synliggöras för beslutsfattare, vilket kanske är ännu viktigare. LCA-resultat är inte platsspecifika och beskriver potentiell miljöpåverkan. LCA kan kombineras med andra MSA verktyg för att ge ett bredare miljömässigt beslutsunderlag.

Acknowledgements

I would like to thank my supervisor Göran Finnveden for encouraging me to actually do this and for support and discussions during the work. Thanks to colleagues at fms, it has been great to be with you again. Thanks also to colleagues at Sustainability & Foresight, STFI-Packforsk. Maria, thanks for encouragement and interesting discussions, there will be more of them to come. Martin, thanks for all your help with "the other work".

Jessica, thanks for good co-operation, inspiring discussions and support especially during the early work.

Financial support from MISTRA, the Swedish Energy Agency and the Swedish Environmental Protection Agency is much appreciated.

Finally, thanks Petter for admirable patience and Märta and Hedvig for making me realise what's really important.

List of papers

This thesis is based on the following papers that are referred to by their roman numerals. The published papers are reprinted with the kind permission of the copyright holder.

Paper I

Finnveden G. and Moberg Å., 2005. Environmental systems analysis tools - an overview. *Journal of Cleaner Production* 13:1165-1173.

Paper II

Finnveden G., Johansson J., Lind P. and Moberg Å., 2005. Life cycle assessment of energy from solid waste – part 1: general methodology and results. *Journal of Cleaner Production* 13:213-229.

Paper III

Moberg Å., Finnveden G., Johansson J. and Lind P., 2005. Life cycle assessment of energy from solid waste – part 2: landfilling compared to other treatment methods. *Journal of Cleaner Production* 13:231-240.

Paper IV

Finnveden G., Nilsson M., Johansson J., Persson Å., Moberg Å. and Carlsson T. 2003. Strategic environmental assessment methodologies – applications within the energy sector. *Environmental Impact Assessment Review* 23:91-123.

Contents

1. Introduction

1.1. Background

Decisions are made all the time. There are usually two or more alternatives to choose between. Individuals make personal decisions and decisions are made in companies and organisations and on a societal level. Decisions stretch from operational to strategic, e.g. from which kind of printer paper to buy to setting the strategy for product development. When decisions are made, we tend to weigh different criteria against each other. When choices are made in the supermarket, the criteria may be taste and price and for some also the origin of the product and the possible environmental label. In more strategic decision-making too, different individuals include different criteria in their evaluation of the alternatives and different individuals also weight the criteria differently. The choice of criteria and the weighting of them may be carried out more or less consciously. The information underlying the decisions made may be quantitative or qualitative and more or less well-founded. In the case of decisions to be made outside the individual sphere, the criteria and the possible weighting of them should be made clear to avoid misunderstandings and conflicts late in the decision-making process. The collected information on which to base a decision can be substantial, but most of all it should be relevant for improving the decision-making. However, the definition of what is relevant information and what is the appropriate way to provide it might differ widely among different actors.

There are a large amount of tools available to provide decision support. Some of these tools focus on giving information from an environmental or sustainability perspective, e.g. environmental or fair trade labels, life cycle assessments and risk assessments. Tools are continuously being developed, some of which are much used by decision-makers themselves, others more frequently used by consultants or researchers providing the decision-makers with information to facilitate better (more informed) decisions. It is debatable whether more information leads to better decisions. Wrisberg et al. (2002) believe that agreement on criteria for evaluation should be reached before detailed and resource-demanding assessments are made. Sexton et al. (1999 p3) state that 'good decisions require more than good analyses and good intentions. They require judicious blending of facts and values to make informed judgements about critical trade-offs'.

In this licentiate thesis, the focus is on tools facilitating the inclusion of environmental aspects in decision-making. As environmental aspects are increasingly being asked for by legislation, in procurement, for communication and marketing etc., there is a demand for tools to handle these issues. Many tools have been developed over recent decades. Economic tools have been developed to incorporate environmental issues (e.g. Cost-Benefit Analysis) and tools explicitly focusing on environmental aspects have been developed (e.g. Life Cycle Assessment and Environmental Impact Assessment). Overviews of environmental concepts, methods and tools have been presented in e.g. Baumann and Cowell (1999), Moberg et al. (1999), Petts (1999), Wrisberg et al. (2002), EEA (2003) and Morrissey and Browne (2004). Today environmental tools are being developed to also incorporate economic and social aspects in order to provide tools for sustainable development.

1.2. Environmental decision-making

It is not quite clear what distinguishes environmental decision-making from other decision-making, if it differs at all. It could be argued that the environment is in fact affected by all kinds of decision-making. As Baumann (2004) discusses in her presentation of the new concept and field of environmental assessment of organising (EAO), management and strategies that are not intentionally environmental may nevertheless have a great influence on environmental impacts from an activity. Sexton et al. (1999, p 2) define environmental decisions as 'choices that individuals, groups, organizations, and societies make about the environment or affecting the environment'. Environmental decision-making is here not distinguished from decision-making generally.

Decision-making occurs within a context. Wrisberg et al. (2002) suggest a framework for environmental decision-making, including core and context characteristics (Table 1). The aim is for their framework to be used to describe business decision-making processes and to define the need for environmental information. There are also other suggestions for how to characterise environmental decision-making and decisions (e.g. Sexton et al. 1999).

Table 1 Framework for environmental decision-making as suggested by Wrisberg et al. (2002)

Core characteristics	Context characteristics
Decision object	Level of improvement
Temporal & spatial characteristics	Importance of subsystem
Question types	Complexity of system change
	Aspiration of decision-maker
	Level of chain control
	Decision types
	Decision steps
	Cultural context

English (1999) states that understanding of the boundaries (ramifications) of the issue is an essential first step in the decision-making process. In this step 'conceptual tools can help, but ultimately this understanding depends on individual reflection and the exchange of ideas among people' (English 1999, p 60). English further states that understanding the social setting of the decision-making is necessary to be able to choose the right tools. This is in line with Tukker (2000), who discusses different value frames (precautionary, strict control and risk assessment) and points out that different approaches to decision support are needed depending on the frames of the stakeholders. Social setting, or cultural context, is also emphasised by Wrisberg et al. (2002). They distinguish between decision-making situations where criteria are agreed or not agreed, and decisions without agreed criteria are further divided into cases without specified criteria and cases with differing criteria. For the latter, they state that the use of analytical tools to facilitate decision-making may be wasted. Thus the cultural context of decision-making should be considered when choosing tools for decision support.

Weighting is a crucial part of decision-making. Analytical ESA tools may provide information about different alternatives, but the information provided will often not simply produce <u>one</u> answer. For example, different alternatives may be preferable depending on which environmental aspect that is being considered; acidification, land use, climate change, etc. In order to make a decision, the different environmental aspects must be weighted. This may be done more or less unconsciously by the decision-maker or through the use of some kind of weighting method. The latter may either be done by the analyst who performed the assessment, the decision-maker or by both together. As environmental aspects are often not the sole aspects to be considered in making the decision, a weighting must also be done between the environmental dimension and other dimensions, such as the economic and the social. There are methods for weighting different environmental aspects against each other today, but they are not generally accepted. For a discussion on weighting see for example Finnveden (1997) and Bengtsson (2001).

Decisions may be defined as strategic or operational, or as single, unique decisions as opposed to regular routine decisions (Wrisberg et al. 2002). In strategic decision-making, effects of the decision to be made may be hard to foresee, as the future surrounding systems are not known. Robért et al. (2002) emphasise that the goal should be clearly stated in strategic decision-making and that agreement on criteria (objectives) and direction of the planning process are essential. They also point out the necessity of selecting and designing tools that are relevant for the strategic approach, to avoid solutions that are not robust.

Some researchers advocate more participation in decision-making (see e.g. Petts 2003, Morrissey and Browne 2004). This would lead to an emphasis on providing more easily understandable and transparent decision support. Petts (2003) suggests that the possibility for the public to discuss the influence of different future scenarios could enhance the understanding and provide public engagement in solutions.

1.3. Environmental systems analysis (ESA) tools

Systems analysis is, as the name suggests, analysis of a system. Thus, instead of considering separate parts of a larger system, a more holistic approach is taken. Of course there is always a larger system of which the studied system forms part, but still the systems analysis approach aims at widening the perspective and thereby avoiding sub-optimisation and unwanted effects. The systems approach has, according to Olsson and Sjöstedt (2004), gained importance during the last 40-50 years, a result of increasing awareness of the interdependence of different aspects of our society and environment. An essential part of systems analysis is the setting of system boundaries, which must be relevant and transparent to make possible interpretation of the outcome of the analysis.

In a description of the course 'Applied Environmental Systems Analysis' (KTH 2006), it is stated that '*Environmental Systems Analysis treats analysis and assessment of the interaction between anthropogenic (human-made) systems and their environment(s). It aims at providing a basis for decisions and planning for a more sustainable behaviour at an individual, organisational and societal level'.* The Environmental Systems Analysis group at Wageningen University in the Netherlands describe it as '*a quantitative and multidisciplinary research field aimed at combining, interpreting and communicating knowledge from the natural and social sciences and technology'* (Wageningen University 2006). At Chalmers University of Technology, the research department of Environmental Systems Analysis describes ESA as *'including methods and tools for the environmental assessment of technical systems of different kinds'* (Chalmers 2006, my translation). Burström and Frostell (2000) discuss the definition of environmental systems analysis and conclude that it is not possible to give an unambiguous answer to what ESA is, but that it is related to some kind of decision-making situation where even the definition of a decision-making situation may be discussed.

The term ESA tool is used in the present study to describe methods and tools for the environmental assessment of human-made systems using a systems perspective.

Environmental information gained by using ESA tools can be used for learning purposes or communication, or to facilitate more informed decision-making. Some of the ESA tools studied also consider other aspects, economic and/or social.

1.3.1. ESA tools

There are a considerable number of ESA tools available. They may be divided into procedural and analytical tools as suggested by Wrisberg et al. (2002). Procedural tools focus on improving the procedures leading to decision-making, while analytical tools provide information that may be used for communication, optimisation of the studied system, comparing different alternatives, etc. Analytical tools may be used within procedural tools.

Analytical tools may be accounting or change-orientated, i.e. they describe a state or consequences of a choice (e.g. Tillman 2000). Many of the tools may be used for both these purposes and differences in methodology should be made, e.g. regarding use of marginal or average data as will be discussed in Section 2.2.3. Procedural tools include tools for the operational management of companies (Environmental Management Systems) as well as tools to be used in strategic decision-making (Strategic Environmental Assessment).

Below, the tools that are most frequently discussed in this thesis are briefly described. For a more thorough description of these and other tools and further references, see Paper I, Moberg et al. (1999) and Wrisberg et al. (2002).

LCA (Life Cycle Assessment) is an analytical tool assessing potential impacts from products or services using a life cycle perspective, including impacts from raw material acquisition, production, use and waste management as well as transportation.

LCC (Life Cycle Costing) can be used to assess the costs of products or services using a life cycle perspective. Social and environmental costs may be included.

MIPS (Material Input Per unit Service) is similar to LCA but only include material inputs throughout the life cycle of a product or service.

Bulk-MFA (Material Flow Analysis) handles the input of bulk material in terms of physical measures to a system.

RA (Risk Assessment) is a broad term and includes both the risk assessment of chemical substances and of accidents. The latter concerns unplanned incidents, whereas the former concerns the dispersion of chemicals, which is often part of the use of the chemicals. RA of chemicals includes exposure assessment and effect assessment, while RA of accidents includes the analysis of probability and possible consequences.

SEA (Strategic Environmental Assessment) is a procedural tool for handling environmental (and sustainability) aspects in strategic decision-making (policies, programmes and plans). It is required by law for certain programmes and plans (SFS 1998:808).

EIA (Environmental Impact Assessment) is a procedural tool required by law in some situations (SFS 1998:808). This tool describes the environmental impact of a suggested project and its alternatives (e.g. the construction and localisation of a waste incineration plant). How the assessment of environmental impact should be made is not predefined and analytical ESA tools can be used within EIA.

Because they all focus on environmental impacts and share the systems analysis perspective, it is of interest to characterise the different tools in order to better understand their interrelationships and the appropriateness of different tools in different applications.
Wrisberg et al. (2002) have provided an overview of supply and demand relating to environmental information and the link between them, focusing on business decision-making. Wrisberg et al. (p. 78) state that there is a lack of analytical and procedural tools to be used in strategic planning and for decisions concerning capital investments. They also identify the need for future-orientated tools like back-casting and scenario analysis to combine with other tools in decision-making concerning strategic planning and capital investments. This is in line with the suggestions for SEA methodologies presented in Paper IV.

1.4. Waste management

1.4.1. Handling of waste

In this thesis, waste management of household waste was chosen as a sample object for decision-making.

Waste is defined as every object, material and substance part of a waste category that the owner is getting rid of, plans to get rid of, or is obliged to get rid of. Household waste is defined as waste from households and comparable waste from other activities (SFS 1998:808).

From throwing waste out of the window into the street to organised collection of sorted waste for treatment in central facilities through recycling, incineration, digestion, composting or landfilling, waste management of household waste has made amazing progress. In Sweden, 90 million tonnes of waste were generated in 2002 and 5% of the waste generated was household waste (Naturvårdsverket 2005a). In 2004, the main proportion of the household waste (47%) was taken care of through incineration with energy recovery, 33% was handled through material recycling, 10% through biological treatment and 9% through landfilling (RVF 2005).

The main purpose of the different waste management strategies is to take care of the waste generated, but in most cases other services are simultaneously provided; material or energy is recovered and food waste may be converted into fertiliser, and the waste is therefore sometimes also described as a resource. These extra functions make waste management more complex.

Major concerns when trying to achieve environmentally sustainable waste management are greenhouse gases and toxic substances, as the Swedish national environmental objectives where the waste system has the relatively largest contribution are *Reduced Climate Impact* and *A Non-toxic Environment* (Naturvårdsverket 2005a).

1.4.2. Swedish regulation

Goals for Swedish waste management are formulated within the national environmental objectives (Naturvårdsverket 2005a). The interim target for waste is that '*the total quantity of waste generated will not increase and maximum use will be made of its resource potential while minimising health and environmental effects and associated risks'* (Government Bill 2000/01:130) This interim target is divided into sub goals including:

'By 2010, at least 50% of all household waste will be recycled through materials recovery, including biological treatment.'

In Sweden, since 1991 all municipalities are required to have a waste plan (SFS 1998:808). This plan must include information on waste within the community and also on the community's actions to decrease the amount of waste and to decrease its danger. Swedish municipal authorities currently have responsibility for the waste from households but not waste from SMEs (small- and medium-sized enterprises), building and demolition waste or industrial waste. On the other hand, the municipal authorities are responsible for the planning of all categories of waste within the community (Naturvårdsverket 2005b). In the proposal for new regulation, it is suggested that local goals based on the national goals for environmental quality should be included in the plan, as well as a statement on how the SEA of the plan has been performed (Naturvårds-verket 2005b, 2005c). Other tools could also be used in connection with the waste plan to facilitate the performance of the SEA and in monitoring the fulfilment of the goals specified. A bill regarding the role of municipal authorities within waste management is planned to be given in March 2006 (Statsrådsberedning 2006).

Since Producer Responsibility came into force in 1994, all producers are responsible for their products even after their use. In Sweden, Producer Responsibility is required for packaging, tyres, waste paper, motor vehicles and electr(on)ic products. The long-term aim of Producer Responsibility is that it will lead to more environmentally responsible product development (Naturvårdsverket 2006). However, the annual monitoring performed by the Swedish EPA focuses on the recovery rates compared to recovery targets set (Naturvårdsverket 2005d).

Assessments of impacts of certain projects (EIA), plans and programmes (SEA) on the environment are required by law (SFS 1998:808). There are specified requirements for what should be included in the reports documenting the assessment. However, the way in which significant environmental impacts are to be identified and assessed is not regulated by law. Waste management activities are affected by these regulations, e.g. for public plans and programmes regarding waste management on national, regional and local level that are assumed to result in significant environmental impacts and also for projects, as the construction and location of waste management facilities.

Waste management in Sweden is also affected by the landfill tax and the decision to stop landfilling of organic waste after the year 2005. A bill regarding tax on waste for incineration, as suggested in SOU (2005:23), is planned to be given in April 2006 (Statsrådsberedningen 2006).

1.4.3. Waste hierarchy

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

- Reduce the amount of waste
- Reuse
- Recycle materials
- Incinerate with heat recovery
- Landfill

The first priorities, to reduce the amount of waste and to reuse, are in general accepted. However, the remaining waste needs to be taken care of as efficiently as possible. In regulation, the above hierarchy is suggested when recycling of material is expected to result in lower environmental impact than incineration (e.g. Government Bill 2002/03:117). The hierarchy after the top priorities is often contested and discussions on waste policy are intense in many countries. The order regarding recycling and incineration in particular is often discussed. Another question is where to place biological treatments such as anaerobic digestion and composting in the hierarchy.

Even though the top priority is in general accepted, strategies leading towards the reduction of waste generated are often lacking. Jacqueline McGlade, Executive Director of the European Environmental Agency (McGlade 2005), acknowledges that the EU Directive targets for packaging waste have helped in achieving higher recycling rates across Europe. However, she states that 'the need to tackle the ever increasing generation of waste has remained in the form of an objective rather than a specific, quantitative target, and there has been little progress on the issue'.

1.4.4. Waste management decision-making

Waste management decision-making is performed by authorities at different levels, waste management companies, companies in general, organisations, the public, etc. Decisions in this field may be operational or strategic, influence few or several, and have implications on the environment as well as social and economic implications. In summary, decision-making in the field of waste management is diverse, as it probably is in most other fields too.

Decision-making regarding waste management by Swedish authorities on different levels should in a longer perspective strive towards national environmental goals, such as the national environmental objectives (Government Bill 2000/01:130) and goals set in the Ecoefficient Society Bill (Government Bill 2002/03:117). In the Swedish national waste plan it is stated that an important prerequisite when aiming towards sustainable waste management is that there is a well-functioning planning process with clear goals and strategies that are well-founded and approved by stakeholders (Naturvårdsverket 2005a).

1.5. The use of ESA tools in waste management decision-making

Some ESA tools have to be used, since they are required by law, e.g. EIA and SEA. The effectiveness of these tools in providing relevant decision support is not unquestioned. In a Master's thesis on the decision-making procedure for construction of waste incineration plants (Lindberg 2004, p. 21), a comment given by a representative from a company in an interview was that EIA was mainly a tool for the authorities. On the other hand, another company representative said in the same study that EIA could be called a tool for helping them. Cashmore et al. (2004) present some potential interpretations of EIA by different stakeholders. The potential interpretation of EIA by the developer is there presented as 'an unnecessary, additional bureaucratic hurdle undertaken at the developer's expense, for reasons of political expediency'. Cashmore et al. (2004) suggest that EIA has been developed without considering the decision-makers needs. Thus the information provided by the EIA may not be what is most useful for the decision-makers and stakeholders (ibid).

SEA is also demanded by law for certain programmes and plans. This regulation is quite new and therefore the effectiveness of this tool is hard to judge as yet. There are, however, studies on SEA and waste management (e.g. Poulsen and Hansen 2003, Brooke et al. 2004). Two different interpretations of SEA methodology are made (Thérivel and Partidário 1996). SEA can be seen as more or less an EIA, performed on more strategic objects to assess the potential environmental impacts of proposed policies, plans or programmes. It can also be regarded as a tool influencing strategic decision-making processes along the way, being part of it from brain-storming to monitoring of implementation. Thus, there can be some confusion about the concept of SEA. Poulsen and Hansen (2003), for example, more or less equate SEA to a strategic LCA. This also accentuates the need for clear guidelines on the different tools, as suggested by Petts (2003).

Apart from the procedural tools that are required by regulation, there are other procedural tools suggested for use in waste management decision-making (e.g. Niutanen and Korhonen 2003, Morrissey and Browne 2004). The Regional Environmental Management System (REMS) suggested by Niutanen and Korhonen (2003) is, as the name implies, a tool for managing a regional system. Their vision is that this tool should be used to manage all issues within a region, but in their study they chose waste management as an example of what can be gained using this tool. Conventional Environmental Management Systems (EMS) is another procedural tool that may be used in companies and organisations involved in waste management or may include aspects related to waste management when used in other companies and organisations.

Analytical tools are also used in waste management decision-making. There are a number of studies where LCA has been used to assess the environmental impacts of different waste management options (e.g. Arena et al. 2003, Eriksson et al. 2005). LCA has also been used in combination with LCC for municipal waste management (Carlsson Reich 2005).

The European Environment Agency (EEA) in a report from 2003 aims at 'providing an overview and assessment of existing/available assessment tools relevant for waste and material flows suitable for EEA reporting requirements', including tools for a range of purposes from state of the art concerning waste streams to more strategic questions like effects of planned policy measures. The conclusions in the report are that there is first of all an immediate need for better statistics to be able to provide relevant information for use in indicators and assessments and that EEA assessment will largely be based on indicators. Modelling is regarded as useful for prospective analysis, but it is not stated how the information to be included in the models should be gained. However, LCA is recommended as a tool for EEA if there is a need to undertake product- or process-specific assessments (EEA 2003). This could be a way to provide input to the models. GIS and remote sensing are also recommended if more detailed spatial information should be needed by the EEA. It is not quite clear why other tools that are part of the overview are not recommended (e.g. Risk Assessment and Cost-Benefit Analysis).

On the website of the European Topic Centre on Resource and Waste Management (ETC/RWM) (a topic centre of EEA) there is a special section for LCA, which illustrates the EEA opinion that LCA is a relevant tool for waste management (ETC/RWM 2006).

Other ESA tools used within the field of waste management include Multiple Criteria Decision Analysis (MCDA) (e.g. Cheng et al. 2002), Modified Cost-Effectiveness Analysis (MCEA) (Döberl et al. 2002), ecological footprint of materials and waste (e.g. Chambers et al. 2004) and CBA (e.g. Bruvoll 1998, Strömberg and Ringström 2004).

Combinations of tools may be used and results from previous studies may also be used as information sources in new decision-making situations. One example is the proposal for a tax on incineration of waste. A commission has in this case provided the Swedish Government with information on effects of different policy options (SOU 2005:23). The focus has been on economic consequences and environmental impacts. Tools used to assess the environmental impacts of the suggested policy included different futures studies concerning the effects of a waste incineration tax on waste flows, waste producers and waste managers. Rough environmental assessments were made, mainly based on results from previous studies not made explicitly for the purpose of this decision-making situation, including LCA, Energy analysis and SEA.

1.6. Aims of the thesis

The overall aim of this thesis was to *discuss the choice of different environmental systems analysis tools for decision support in decision-making using waste management as an example and to apply LCA for municipal solid waste management as a case showing the type of conclusions that can be drawn*. The activities pursued in order to reach this aim were the following. Firstly, a comparison between different ESA tools was made to facilitate a better understanding of their interrelationships and of the appropriateness of different tools in different applications. Secondly, one of the environmental systems analysis tools, Life Cycle Assessment, was used to assess the potential environmental impacts of different strategies for treatment of municipal solid waste in Sweden, providing decision support on a strategic level within the waste management sector. Assumptions and valuations leading to changes in the resulting rankings of waste management strategies were discussed. This is an important aspect, as strategic decision support should be robust and transparent. Thirdly, the use of tools within other tools was studied, using Strategic Environmental Assessment within the energy sector as an example.

What tools should be/could be/are actually used in a defined decision-making context are relevant questions. In this thesis, the question considered is *What methods could (or even should) be used…?*. The tools actually used and how the choices of tools are actually made in practice are also relevant questions, but these were not covered in this thesis.

2. Methodology

2.1. Different environmental systems analysis tools (Papers I and IV)

2.1.1. Selection of ESA tools

Environmental systems analysis tools are used to facilitate decision-making, providing support concerning relevant environmental aspects, or to provide information for learning and communication. To overview different ESA tools a literature study was made. This was first done in a Master's thesis and a report (Moberg 1999, Moberg et al. 1999). Knowledge on different tools was thereafter increased through further literature studies.

ESA tools incorporate both tools that are analytical and tools that are procedural. As can be seen in Table 2, the majority of the tools covered in Papers I and IV are of analytical character. The tools providing a possible basis for evaluation are not comprehensively covered here.

Table 2 List of tools covered in Papers I and IV with a short comment on which type of decision support they provide

Tool	Paper		Type of decision support
Cost-Benefit Analysis, CBA	I		Analytical
Direct Material Consumption, DMC	I	IV	Analytical
Direct Material Input, DMI	I	IV	Analytical
Ecological Footprint, EF	I	IV	Analytical
Environmental Impact Assessment, EIA	I		Procedural
Economic Valuation		IV	Evaluation
Environmental Management System, EMS	I		Procedural
Energy analysis, En	I	IV	Analytical
Futures studies		IV	Analytical and procedural
Input-Output Analysis, IOA	I	IV	Analytical
Impact Pathway Approach, IPA		IV	Analytical
Life Cycle Assessment, LCA	I	IV	Analytical
Life Cycle Costing, LCC	I		Analytical
Multiple Attribute Analysis, MAA		IV	Analytical
Material Intensity Per Unit Service, MIPS	I	IV	Analytical
Material Flow Analysis, MFA	I		Analytical
Risk Assessment, RA	I	IV	Analytical
Strategic Environmental Assessment, SEA	I	IV	Procedural
System of Economic and Environmental Accounts, SEEA	I		Analytical
Substance Flow Analysis, SFA	I		Analytical
Surveys		IV	Evaluation[1]
Total Material Requirement, TMR	I	IV	Analytical

The tools studied in Paper I were chosen after discussions with a reference group containing members from the Swedish Environmental Protection Agency and industries. The main criterion for the choices was to some extent assumed frequency of use, but the aim was also to get a good mix of tools capturing different angles and values. Quantitative analytical tools and procedural tools were selected, excluding qualitative analytical tools. Comparisons of some qualitative tools can be found in e.g. Johansson et al. (2001) and Byggeth and Hochschorner (2005). In Paper IV, some tools that may be relevant to include within a Strategic Environmental Assessment (SEA) are suggested. The tools considered in Paper IV were selected because they are

[1] Surveys may provide decision support other than evaluation, which was the focus for this tool when considered in Paper IV.

established methodologies that are well-known, they have been developed, tested and applied in the energy sector (which is the focus of the paper) at policy, plan and programme levels, and there are data available from these tools for testing the methods in a pilot study (see Nilsson et al. 2005). There are additional quantitative and qualitative tools that could be useful in an SEA context that were not considered in Paper IV. The tools in Table 2 are briefly described in Papers I and IV, further descriptions and references of the tools used in Paper I can be found in Moberg (1999) and Moberg et al. (1999).

The ESA tools studied in Papers I and IV may not be explicit for environmental studies. Some tools can be modified for use (e.g. environmentally extended IOA) and some may be used to facilitate environmental systems analysis (e.g. futures studies).

2.1.2. Characteristics for description

Characteristics used to describe tools in Paper I were:

- Procedural or analytical
- Impacts considered
- Object of study
- Accounting or change-orientated

Originally these characteristics were part of a larger set used in Moberg (1999). The basis for the characteristics was modified from a framework presented in SETAC (1997) and further developed by Baumann and Cowell (1999). The selection of the four characteristics used in Paper I was based on earlier studies, but other aspects and characteristics may also be used to structure tools (see e.g. Wrisberg et al. 2002).

The impacts considered by the tools were divided into: Natural resources; environmental impacts; natural resources and environmental impacts; and economic aspects including natural resources and environmental impacts. Objects included: Policy, plan, programme and project; region and nation; organisation (including company); product/function; and substance.

The characteristics considered for the tools in Paper IV were:

- Degree of site-specificity,
- Degree of time-specificity,
- Type of comparison,
- Degree of quantification,
- System boundaries (largely determined by object of study),
- Type of impacts and effects considered,
- Information provided by the different tools according to the DPSIR model[2].

[2] DPSIR is a model describing different levels of environmental information: Driving forces, Pressure, State, Impact and Response (see Smeets and Weterings 1999).

These are the characteristics, as suggested in Paper I, for consideration when tools are to be used in combination. Thus they are important when considering tools to be used within an SEA.

In Paper IV, the procedural or analytical aspect was not considered, as the aim was to suggest tools to use within a procedural tool, be it analytical or of other kind. Type of impacts considered and system boundaries, which are largely determined by the object of study, are characteristics included in both papers. The aspects of accounting or change-orientated and degree of site- and time-specificity could also have been included in both papers. Type of comparison concerns whether the possible comparison is made between different alternatives, within a studied system or against a reference

In Paper IV the information provided by the different tools was also categorised using the DPSIR model. This model is based on the need from policy-makers for 'clear and specific information on driving forces and the resulting environmental pressures on the state of the environment and impacts resulting from changes in environmental quality and on the societal response to these changes in the environment' (Smeets and Weterings 1999 p. 6).

2.2. Life-cycle assessment (Papers II and III)

2.2.1. General framework

LCA is a tool to assess resources used and potential environmental impacts (inputs and outputs) throughout a product's life from raw material acquisition through production use and disposal (i.e. from cradle to grave). The term 'product' can include not only the product systems but also service systems. An ISO standard has been developed for LCA providing a framework, terminology and some methodological choices (ISO 1997, 1998, 1999).

The framework according to the ISO-standard (ISO 1997) consists of four iterative steps (Figure 1):

1. *Goal and scope definition*, where the goal of the study is defined, system boundaries are set and the functional unit is defined.
2. *Inventory analysis*, where all relevant data are identified and quantified.
3. *Impact assessment*, aimed at understanding and evaluating the magnitude and significance of the potential environmental impacts. Including:
    a. Selection of impact categories, indicators for the categories and models to quantify the contributions of the different inputs and emissions to the selected impact categories.
    b. Classification, assignment of the inventory data to the impact categories.
    c. Characterisation, quantification of the contributions to the chosen impact categories.
4. *Interpretation*, where inputs from all three previous steps should be considered.

There are also some optional elements that may be included in an LCIA (Life Cycle Impact Assessment). Weighting may be included to convert and possibly aggregate indicator results across impact categories, resulting in a single result. Normalisation is another optional element whereby the magnitudes of the impacts are related to reference values, e.g. total contribution to an impact category by a nation.

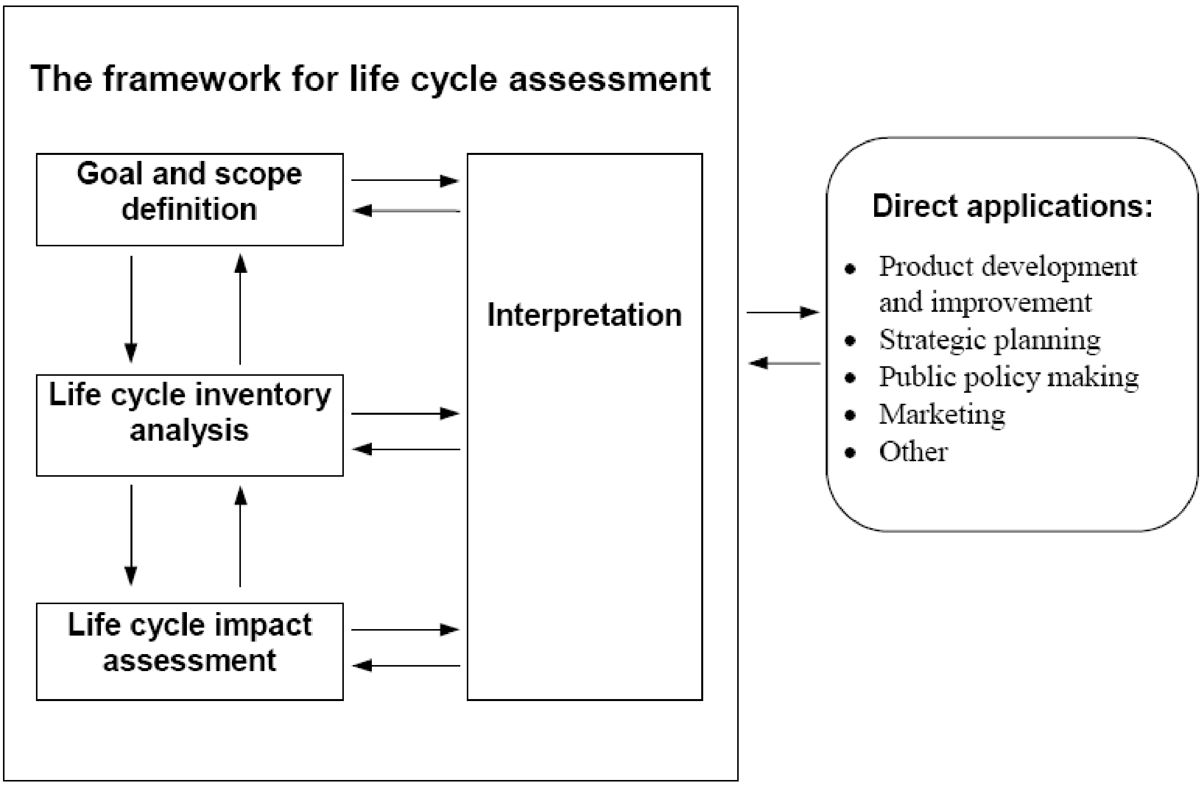

Figure 1 The phases of a Life Cycle Assessment (modified from ISO 1997).

LCA may be accounting or change-orientated, depending on the goal of the study (e.g. Tillman 2000). In the former, what is or was actually happening in a system should be inventoried, or if the accounting study concerns a future system, what will actually happen in the studied system under assumptions made. Change-orientated LCA on the other hand studies the consequences of a choice. For a change-orientated study, the ideal data are marginal data reflecting the actual change (e.g. Weidema et al. 1999). Weidema et al. (1999) suggest a procedure for identification of marginal data. According to this procedure if the time perspective is long (decades), it is in general changes in the base-load marginal that are of relevance. The long-term base-load marginal is determined by e.g. whether the total market is increasing or decreasing and if there are any constraints for using the respective technique (Weidema et al. 1999). If the total market increases new investments will be made, while if the market is decreasing production capacity will be decreased.

2.2.2. LCA methodology and integrated solid waste management

The description here of methodological aspects of LCAs of waste management is based on the methodological description in the report underlying Papers II and III (Finnveden et al. 2000). The description in the report is in its turn based mainly on Tillman et al. (1994), Finnveden (1999), Clift et al. (2000), Ekvall and Finnveden (2001). Many of the methodological aspects of LCA and waste management are general and should also be useful when using other ESA tools.

<u>System boundaries</u>
In LCAs of waste management systems, an exception to the rule of LCA that inputs and outputs should be covered from 'the cradle to the grave' is usually made. The input to the system is usually solid waste, e.g. from households. This is, however, still compatible with the LCA definition, if the same inflow appears in all systems that are to be compared. This is because identical parts of systems to be compared can be disregarded. If one of the systems e.g. produces more or less waste than the other, this is no longer true and the system boundaries should be moved upstream. This may not be done in practice, but should then be carefully noted and handled in the interpretation of the results.

Downstream system boundaries may also be complicated in LCAs of waste management systems. Products from the recycling of waste material are usually not followed to the grave. Furthermore, the products that are replaced by the products from recycling are also usually not followed to the grave. This may also be compatible with the LCA definition if the products are 'identical', i.e. providing the same function and having the same environmental interventions. If not, then the system boundary should be moved downstream.

Recycling

When a product is recycled into another product, this is called open-loop recycling. In LCAs the setting of system boundaries can be handled in two ways for open-loop recycling; either by setting the boundary between the two products and allocating environmental interventions between them or by expanding the system to include both products (functions). If system A has the products 'treatment of solid waste' and 'heat' and system B has the product 'treatment of solid waste' only (as in Figure 2a), the expanded system solution may either be used by adding the production of the equivalent amount of heat to system B (Figure 2b) or by subtracting the production of the equivalent amount of heat from system A (Figure 2c). The use of expanded systems results in the need for assumptions regarding which production system should be used for the expansion, e.g. which is the alternative heat source, alternative material used, etc. The use of expanded system boundaries to avoid the allocation problem is often recommended (e.g. Lindfors et al. 1995, ISO 1998). The recommendations of the ISO standard have been critically reviewed, but systems expansion is still generally recommended for change-orientated studies (Tillman 2000, Ekvall and Finnveden 2001).

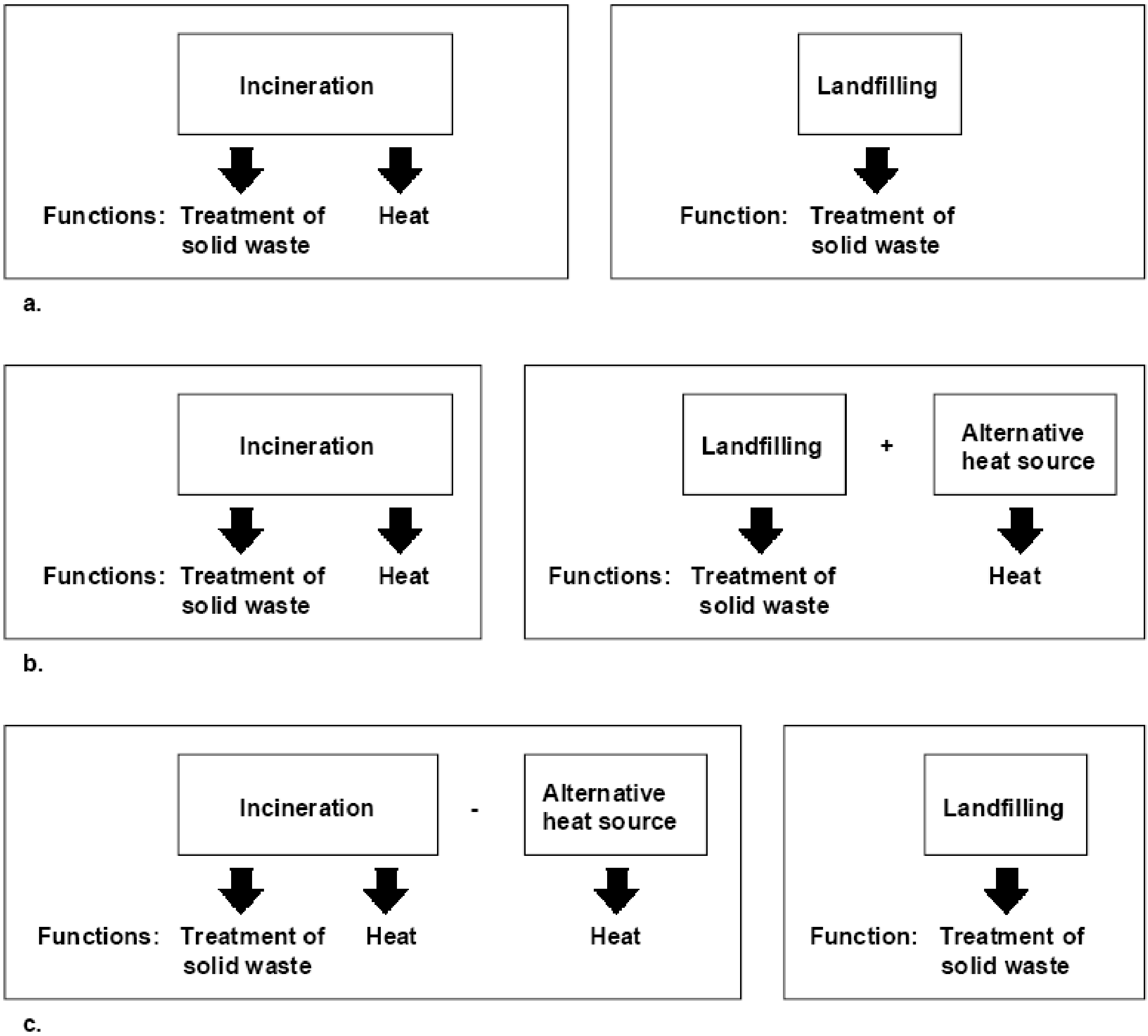

Figure 2 a) An example of an allocation problem. b) A possible way of avoiding the allocation problem by expanding the system boundary. c) An alternative way of presenting the expanded system boundary described in b) (after Finnveden and Ekvall 1998).

Time aspects

The system boundary in time is very relevant when studying waste management if the landfill option is included. As landfilling is generally included as treatment for residues from other waste management options, the time aspect is relevant in most studies of environmental impacts from waste management. Emissions from landfills may prevail for a very long time, often thousands of years or longer. It is important to define the time frame of the study. Using the LCA definition as a starting point, it may be argued that emissions should be integrated until infinity. In practice, a shorter time frame is usually used, often 100 years, but shorter time frames are also commonly used (see Finnveden 1999 for examples). The choice of time frame is important when studying landfilling of waste, since materials like plastics, which are persistent, and substances like metals, which only slowly leach out, will be significantly differently perceived within systems with different time frames. The choice of time frame is a value choice related to ethical views about impacts on future generations (Finnveden 1997).

In the study underlying Papers II and III, two time perspectives were used (Finnveden et al. 1995):

- 'The surveyable time (ST) period', which was defined as the time period it takes to reach a pseudo steady state in the landfill. The surveyable time period should correspond to approximately one century.

- 'The hypothetical infinite time period' was defined by total degradation and emission of the landfilled materials. This time period was introduced to get the maximum potential impacts.

Carbon sinks

When carbon flows are modelled, a distinction is often made between fossil and non-fossil flows. Non-fossil carbon dioxide is often disregarded. This may be motivated by an expansion of the system to include the uptake of carbon dioxide by the growing tree. Another way of thinking is that the harvested tree will be replaced through planting a new tree that will take up an equivalent amount of carbon dioxide. These expansions are not done explicitly. Another perspective is that if the biotic resources had not been harvested they would have been left in the forest and eventually degraded there. With this perspective, the time frame needs to be several centuries for all biotic material to be degraded (Zetterberg and Hansén 1998).

Biological carbon is seen as part of a cycle. When using a limited time perspective, this cycle is interrupted. Then, one may consider landfills to be carbon sinks keeping carbon from being released to the atmosphere within the time frame used. With this perspective the landfilling option can be credited for avoidance of global warming potential. However, the limited time perspective is, as mentioned above, a value choice with ethical aspects.

The carbon sink is only an issue for renewable materials where carbon dioxide emissions are normally not considered and when a limited time perspective is used.

2.2.3. Scope and methodology in the LCA study (Papers II and III)

Goal and scope

Results from a change-orientated LCA of different strategies for treatment of municipal solid waste in Sweden (Finnveden et al. 2000) are presented in Papers II and III. The aim was to evaluate different strategies for treatment of solid waste based on a life cycle perspective, rank the options and identify critical factors in the systems which may significantly influence the results. The study was aimed at providing decision support to public decision-makers ranging from local level to national or even international, and also to decision-makers in companies, organisations, etc. The results were intended to be usable for strategic decisions, including decisions on investments and policies. The information was on a general level, without any site-specific data. Fractions of municipal solid waste that are combustible and recyclable or compostable were included in the study. The composition of this waste was estimated from an analysis of household waste by REFORSK (Olsson and Retzner 1998). The treatment strategies considered were incineration (all fractions), landfilling (all fractions), recycling (excluding food waste), anaerobic digestion (food waste) and composting (food waste). Here the focus was on the three first. When the total amount of waste was considered, the recycling option also included the treatment of food waste.

Treatment of the amount of the included waste fractions collected in Sweden during one year was the functional unit of the study. System expansion through subtraction was used to handle the multi-output of the systems studied. The calculations were made for the unrealistic situation that all waste included was treated with the same strategy.

<u>Scenarios and major assumptions</u>
In the LCA study assumptions were made and some of these assumptions were tested through analysis of different scenarios where the assumptions were changed.

In the base scenario, the following major assumptions and system boundaries were used (the choices are discussed in Paper II and in Finnveden et al. 2000):

- Electricity is produced from hard coal.
- Heat production, which is credited to waste treatment systems where heat is produced. is from incineration of forest felling residues.
- Recycled material is credited to waste treatment systems using data for production of virgin material of the same kind.
- The time perspective is a hypothetical infinite time period.
- Distances for transportation of waste are moderate.

Several 'what-if'-scenarios were used to identify parameters of importance for the outcome of the study. Examples included:

- *The natural gas scenario*, where avoided heat production is assumed to be from natural gas.
- *The saved forest scenario*, where the forest 'saved' through the recycling of paper is assumed to be used as fuel replacing natural gas for heat production.
- *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 plastic palisade scenario*, where mixed plastics are assumed to be recycled into palisades which replace wooden palisades impregnated with copper compounds.
- *The increased transport scenarios*, where longer transport distances by truck to incineration and recycling facilities are assumed.
- *The passenger car scenario*, where waste for recycling and incineration is source separated and transported by car to collection points. This variant was tested 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 were not specifically modelled here.

One major assumption that was not tested in any 'what-if' scenario was the assumption that coal is the marginal source of electricity.

<u>Inventory of data</u>

Data for the life cycle inventory were partly obtained by modelling the treatment of the waste fractions by incineration and landfilling using models as presented in Björklund (1998) and partly by using data from literature and databases.

Although this study was strategic and change-orientated, we mainly used data for the current situation due to lack of relevant data. Some aspects, for example concerning the surrounding energy systems in the future, were inherently uncertain. Efforts were focused on trying different assumptions in order to find aspects that were critical for the results, instead of devoting resources towards finding better data for the current situation. The quality and representativity of the data used varied.

Since the study was change-orientated, the ideal data for background systems would be data for the processes that were actually affected, which in general would correspond to a base load marginal type of data. For electricity production and heat generation, it is known that the environmental impacts vary significantly between different energy sources and also that the choice can have a significant influence on the final results for a waste management LCA (Finnveden and Ekvall 1998). In this study, we therefore tried to assess which energy sources were relevant from this perspective. For other areas, average types of data that were accessible in LCA databases were mainly used.

The electricity chosen was electricity generated from hard coal. This decision can be explained by the suggestion that hard coal is the unconstrained technology with the lowest long-term production cost in the EU and the assumption that electricity use in Europe is increasing (Weidema et al. 1999). Another way of handling this is to assume that the aim is towards a sustainable way of life. In this case the electricity use would decrease and the least preferred technology from a sustainability perspective would be phased out first, and it could be argued that this would be electricity from coal.

Heat produced through waste management was assumed to be transferred to the district heating system. The district heating system is expanding in Sweden and this trend is likely to continue. Forest fuels, including forest felling residues and by-products from industries, now contribute the largest share of the fuels used for district heating and the amount has increased considerably over the years (Energimyndigheten 2004). Assuming that these trends continue, it is likely that forest fuels will continue to be the long-term base load marginal. It may, however, be argued that the long-term capacity is limited and that other fuels are thus also of relevance. The use of natural gas is currently limited in Sweden but may increase if new pipelines are built (Nilsson et al. 2001). Both forest residues and natural gas were used as the marginal district heating sources in different scenarios.

Impact assessment

For the impact assessment, established methods were used to classify and characterise the data into several impact categories. The impact categories included are listed in Paper II. To express the uncertainties in methods to characterise toxicological impacts, two different methods were used: USES-LCA (Huijbregts 1999) and EDIP (Hauschild et al. 1998a, Hauschild et al. 1998b). The weighting method used was EcoTax 98 (Johansson 1999), which is based on fees and taxes. Different weighting factors can be combined with different impact categories and characterisation methods in different ways. Thus a large number of possible combinations are possible. We used three different combinations of characterisation and weighting factors, one minimum combination and two maximum combinations, in order to trace the largest spans. The two maximum combinations were identical except for the characterisation methods used for toxicological effects (EDIP or USES). The resulting sets of weighting factors were called USESmin, USESmax and EDIPmax.

Another impact assessment method, Eco-indicator 99 (Goedkoop and Spriensma 1999), was used to check the robustness of the results, which in the characterisation step models damaged human health, ecosystem quality and resource stock. The weighting step in Eco-indicator 99 is based on a panel procedure.

The results are presented both on impact category level (selected categories in Paper II, all categories in Finnveden et al. 2000) and as a single score for the waste management options studied.

Data quality and uncertainty

An estimation regarding the data gaps and uncertainties of the respective impact categories has been made (Finnveden et al. 2000). Impact categories with no significant data gaps and with estimated uncertainties of a factor 10 or less are: total energy, non-renewable energy, abiotic resources, global warming, SO_x and NO_x.

3. Results

3.1. Environmental systems analysis tools – an overview (Paper I)

The results of Paper I can be summarised as an overview of some 15 tools to provide basic knowledge of their characteristics and a comparison of these tools regarding relevant aspects. The tools and their characteristics are illustrated in Figure 3, where objects of study and impacts considered by the respective tools are used to structure them.

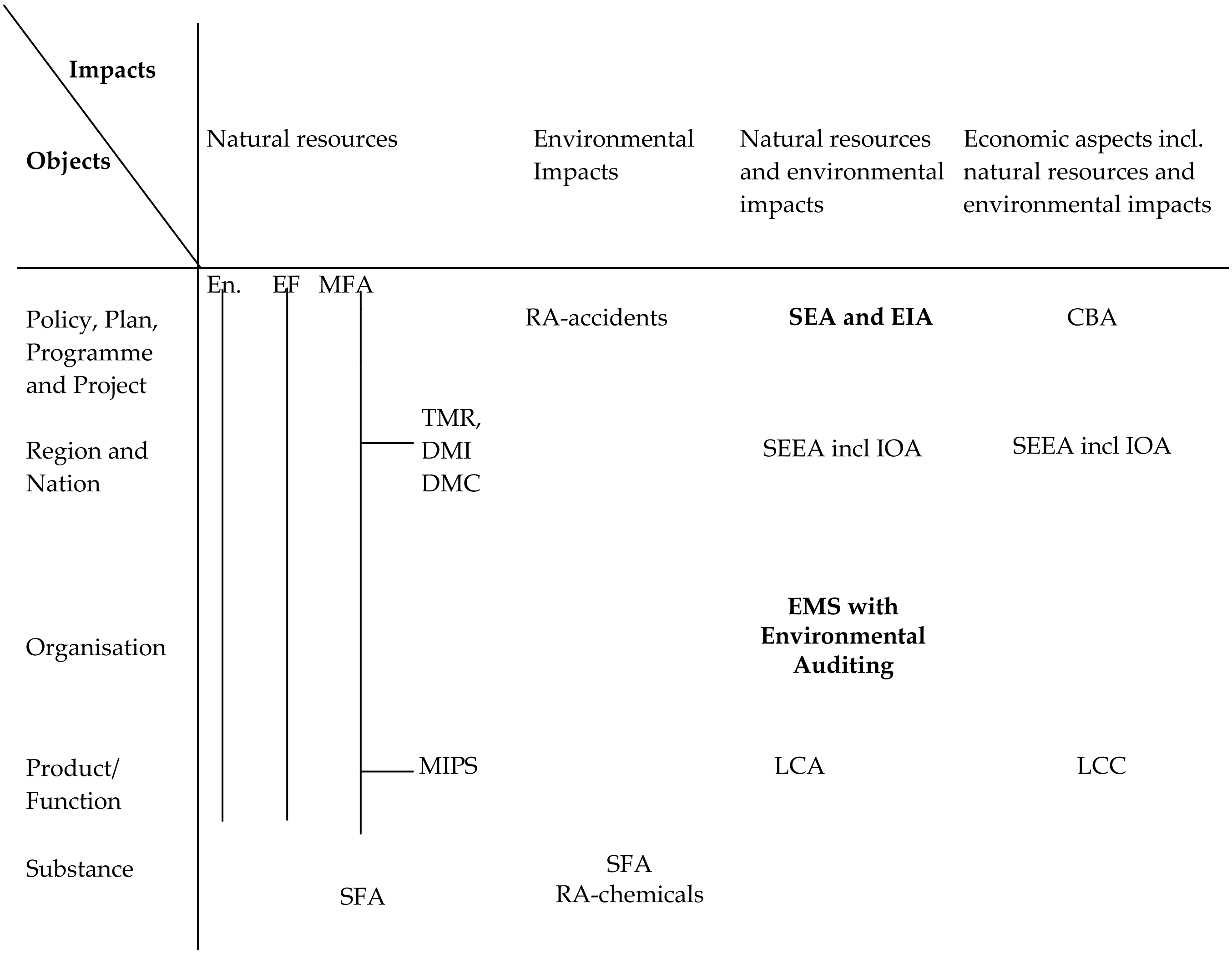

Figure 3 The tools used shown in relation to their focus, i.e. the object to which the impacts are related and the impacts included in the study. Procedural tools are shown in bold type.

In Figure 3 it can be noted that there is a spread of the tools considered, suggesting that the two chosen characteristics, impacts included and object studied, are suitable for describing and characterising different tools. Some of the tools focus on natural resources, either energy, materials or space, and can be used as assessment or evaluation methods for a wide range of objects. For tools that focus on both the use of natural resources and environmental impacts, it seems that for each object there is a tool that is the most suitable. For example, if the object under study is a project, EIA is the most suitable tool and cannot easily be replaced by other tools if the focus is on the use of natural resources and environmental impacts. This suggests that there is indeed a need for a large number of different assessment tools. Some of the tools are developed for the same object of study, but then they often differ regarding impacts included, e.g. CBA is also a tool that can be used to assess projects, but then the extra dimension of economics is added.

Figure 3 suggests that there are tools available for most objects discussed here. EMS with environmental auditing is suggested as a tool for organisations and companies including natural resources use and environmental impacts, while other management systems focusing on quality, working environment, etc. could also add the economic and social aspects. In Figure 3, procedural tools are written in bold type and it can be seen that there are procedural tools available for some objects but not all. This could suggest a need for developing such tools. However, the lack of tools could also be because there is no need for such tools, or because such tools exist but are developed and used internally and not presented in scientific papers, etc.

Different tools have been developed and used mainly for accounting purposes, and others for change-orientated studies. Since both accounting and change-orientated studies are useful depending on the question to be answered, it is of interest to discuss whether the same types of methodologies and data can be used for both types of studies, or whether adaptations should be made. Within the LCA world, it is now generally accepted that there may be a need for different sets of data and methodology depending on whether the tool is being used for accounting or change-orientated studies. For the other tools studied here, this may also be relevant, even though some of them are purely accounting or change-orientated.

In Paper I, a suggestion is given on aspects to consider if combinations of tools are to be used. Differences between tools with regard to these aspects can determine if and how different tools can be used in combination. If two tools are identical with respect to all these aspects, they may be competing. If there are differences they answer different questions, which means that they can complement each other by providing different types of results. If tools are to be used sequentially or are encompassing, it is important that they are compatible when information is to be transferred between them. Aspects include degree of site- and time-specificity, degree of quantification, impacts included and system boundaries/object.

3.2. LCA of waste management options (Papers II and III)

3.2.1. A case study (Paper II)

In Paper II, results from the LCA of municipal solid waste management options are presented as *energy use*, *emissions of greenhouse gases* and *total weighted results*. The total weighted values were obtained using the EcoTax 98 method described in Paper II, Finnveden et al. (2000) and Johansson (1999). In the full study presented in Finnveden et al. (2000), results for other impact categories are also presented, as well as weighted results using another weighting method.

Some of the results presented below are negative, which means that environmental impacts can be avoided through avoided alternative heat and material production, etc. The reason for this is that the waste comes into the systems studied without any environmental burdens associated with it. If the upstream processes had been included as well, there would probably not have been any negative results at all.

When discussing 'the whole system' the total amount of waste was considered and in the recycling option recycling of the paper and plastics fractions was then combined with digestion of the food waste using the biogas produced for production of heat and electricity. The rankings for the fraction of food waste are discussed separately. The results for this waste fraction were different from the others and included different recycling options.

For *total energy use* the ranking of recycling before incineration before landfilling was robust. The order of preference for the waste management options remained the same in the different scenarios and for different waste fractions.

The results for *non-renewable energy use* were sensitive to assumptions made concerning the avoided heat production and the use of wood saved when recycling waste paper, as can be noted in Figure 4. When the avoided heat was assumed to come from natural gas instead of biofuels, incineration saved more non-renewable energy than recycling. However, the order was again reversed when it was assumed that 'saved' biomass could be used as a fuel replacing natural gas.

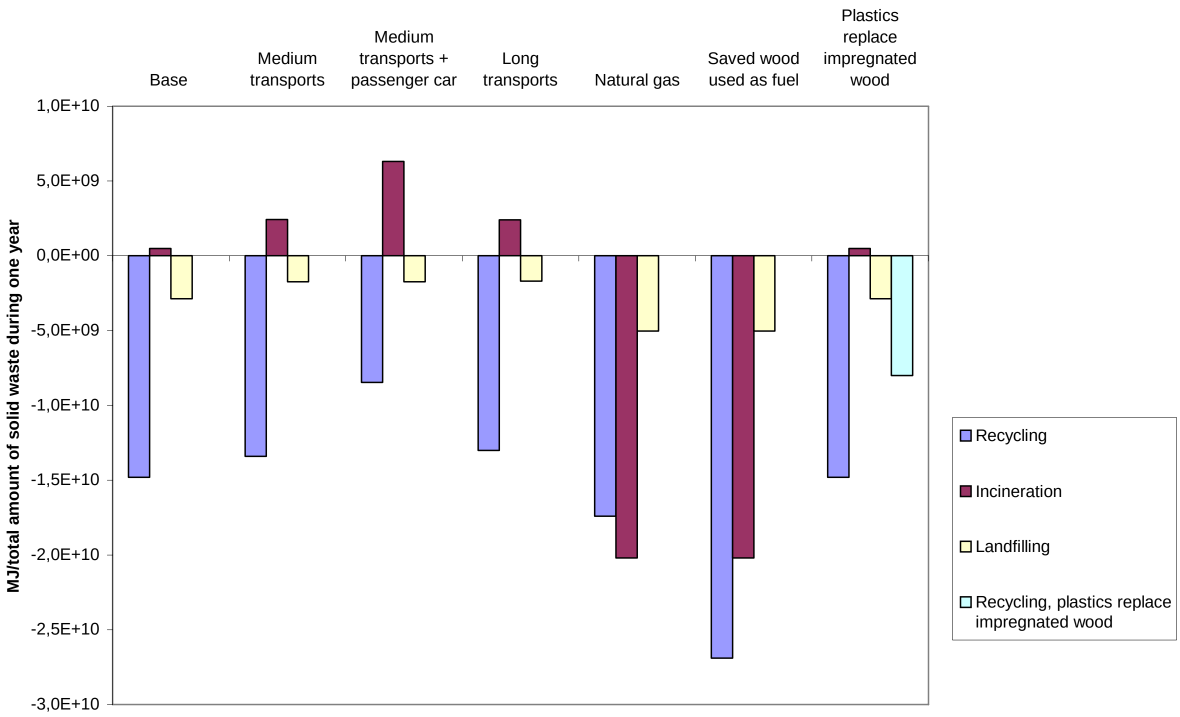

Figure 4 The use of non-renewable energy for the whole system including all waste fractions studied in different scenarios. The scenarios are explained in Section 2.2.3.

For *global warming* the ranking was that recycling was preferable to incineration, which was preferable to landfilling. This result was fairly robust and was valid for all plastic and paper materials in the base scenario. However for some materials, the order of preference for incineration and landfilling was sensitive to the modelling of landfills (when only a shorter time period was considered the results for landfilling were improved). However, in all these cases, recycling was still the most preferable option. When recycled plastics replaced impregnated wood, recycling of plastics was less favourable than incineration with respect to energy use and emissions of greenhouse gases, although the difference was rather small.

For *total weighted results* the order of preference was again recycling before incineration before landfilling. This result was fairly robust and was valid for the whole system and e.g. newsprint and PET in most scenarios. The ranking for incineration and landfilling can be changed by the modelling of landfills, as well as by assumptions regarding transport distances. Recycling was not preferable to incineration when passenger cars were used in the recycling case or the incineration case.

An interesting result was that in general, the ranking order of the waste management options was not altered when assumptions on long-distance transportation by truck were changed. Results were influenced by the assumption of use of passenger car for transportation of waste when waste was source-separated, but the ranking often, but not always, remained the same.

For the fraction *food waste* it can be said that generally among the recycling options, anaerobic digestion was preferable over composting and landfilling regarding energy use, global warming and total weighted results. The ranking between digestion and incineration varied, as well as the ranking between composting and landfilling. The results for waste management of food waste should be carefully used, since key assumptions were made, including the assumption that the residues could be used as fertilisers.

There are uncertainties and gaps within the results of the LCA of waste management options. For example, impacts from land use were not included in this study at all. This is of relevance for example when discussing forestry. In addition, impacts in work environment, casualties, noise and odour were not included at all, nor was biodiversity considered explicitly. The toxicological impact categories were plagued with data gaps, which indicate that conclusions on an overall level should be drawn with caution.

3.2.2. Assumptions and value choices concerning landfilling (Paper III)

Paper III was also based on the LCA of waste management options described above. In this paper, the validity of the waste hierarchy was tested, mainly for cases where the ranking of the landfilling option might be altered depending on assumptions and value choices.

As results presented above for Paper II indicate, the waste hierarchy was suggested to be valid as a rule of thumb. This meant that landfilling was in general the least preferred option from an environmental point of view. Assumptions that influenced the results for the landfilling option in the study are discussed below.

<u>Source of the heat production avoided</u>
The non-renewable energy balance was dependent on assumptions concerning the origin of the heat production avoided when heat was produced from waste incinerators or landfill gas combustion. Landfilling was better off when the source of the alternative heat production was renewable. This was a consequence of more energy being recovered through incineration than through landfilling.

<u>Characterisation methods and weighting values used for ecotoxicological impacts</u>
There were large uncertainties related to the toxicological impact categories. The method used for characterisation and weighting was in some cases decisive for the ranking of landfilling compared to the other management options. This was a way of illuminating the large uncertainties related to the toxicological impact categories.

Time perspective chosen

The base scenario applied a hypothetical infinite time perspective where 'everything' was assumed to be emitted from landfills at some time in the future. In the surveyable time period scenario, a limit in time regarding landfill emissions was set. The main effect of this cut-off was that a large proportion of the metals, but also most of the fossil carbon, e.g. in waste plastics, was not emitted from the landfill at all. The incineration option was also affected by the assumption of a shorter time perspective, since ashes were landfilled and e.g. the major proportion of the metals within the ashes was not emitted in this scenario.

For the plastic fractions, landfilling became preferable to incineration concerning global warming when a short time perspective was used, but recycling was still ranked as the most preferable option. This is illustrated with PET as an example in Figure 5. Expected emissions of carbon to air from landfilling of plastic waste mainly occur subsequent to the surveyable time period, and were thus omitted in this scenario. When plastic waste is incinerated, all carbon is immediately emitted as carbon dioxide and thus landfilling of plastic waste contributed less to global warming during the surveyable time period. In the case of landfilling newsprint and other paper fractions, the major contribution to the global warming impact category was from emissions of methane during the surveyable time period and only a smaller difference was seen, which did not change the ranking of the treatment options.

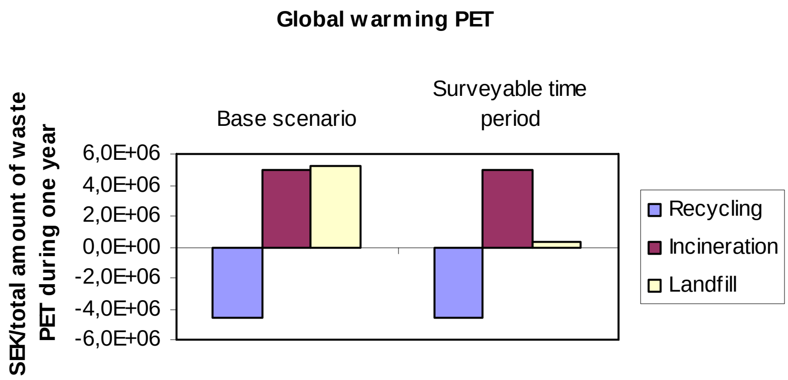

Figure 5 Results for the impact category global warming for waste PET, in the base scenario and for the surveyable time period scenario. Figures are presented in SEK. This is because weighted figures were used, but this does not affect the relationships within the impact category.

Another impact category affected by a change in time boundary was ecotoxicological impacts. In this case, changes in results and ranking were dependent on how the ecotoxicological effects were modelled in the characterisation methods used.

Considering the landfill as a carbon sink

When, with the limited time perspective, the landfill was considered to be a carbon sink, an additional advantage for the landfilling option was gained. The additional function of trapping biological carbon led to more preferable results for the landfilling option concerning global warming for paper waste. The plastic fractions were not affected, since their carbon content was of fossil origin.

Transportation of sorted waste by passenger car

When passenger cars were assumed to be used for transportation of sorted waste from the households for the recycling and incineration options, major alterations in the resulting rankings were obtained for the impact categories photo-chemical oxidant formation and for human and ecotoxicological impacts. In the toxicological categories, landfilling was ranked as the most preferred alternative in several cases, when the other options were burdened with passenger car use.

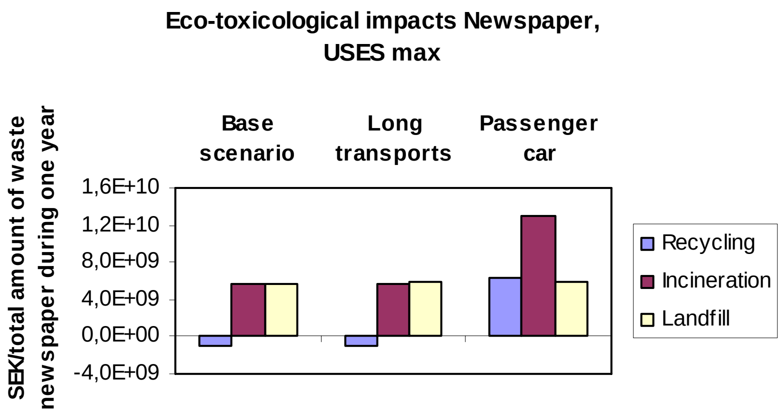

Figure 6 Results for the impact category eco-toxicological impacts for newsprint, using the USES characterisation method with maximum weighting according to the Ecotax 98 weighting method. Figures are presented in SEK. This is because weighted figures were used, but this does not affect the relationships within the impact category.

In Figure 6, the effects are illustrated for the newsprint waste fraction. It can be seen that longer transportation distances by truck did not influence the results greatly, but passenger car transportation did. In the passenger car scenario, transportation by car of waste from the households was assumed for the recycling and incineration options. To compare the recycling with passenger car transport option with the incineration without passenger car transport option, the result for incineration in the base scenario can be compared to the result for recycling in the passenger car scenario (the impacts are similar).

3.3. Use of tools within SEA (Paper IV)

Paper IV showed that several existing tools can contribute to the procedural tool SEA. The SEA process functions as a framework where different tools can be used to support the steps of the process. It was suggested that tools can contribute either by focusing mainly on the identification of interventions, analysis of environmental change (e.g. LCA, RA, future studies) or by focusing mainly on the valuation stage (e.g. MAA methods, economic valuation methods, surveys).

Future studies can be used at several stages in the SEA process. The outcome of future studies in SEA can either be a set of alternatives that all lead towards the predefined goal (as in back-casting studies) or a set of future scenarios in which the different alternatives can be placed. Back-casting can be used in the formulation of alternatives. Other types of future studies are suggested for use in the scenario analysis phase of the SEA.

Futures studies may also be used within the environmental analysis step of the SEA. The consequences of assessed alternatives depend on future environmental baseline situations. Future studies could also be of relevance in the valuation step. This is because future societal values may be different from current values.

4. Discussion and conclusions

4.1. ESA tools for decision-making

Several aspects affect what tool may be 'appropriate' for a given decision-making situation. English (1999) states that it is difficult to specify universal criteria for choosing among tools for environmental decision-making. However, the same author suggests some criteria, divided into essential and desirable. Suggested essential criteria are honesty, freedom from bias and 'tunnel vision' and intelligibility. It can be argued that these suggestions are rather a wish for how the tools should be used and that they do not address inherent properties of tools. The desirable criteria presented by English (1999), complementarity, proportionality and flexibility, are also hard to explicitly see as decisive when choosing tools. This can illustrate the difficulty in setting criteria for the choice of ESA tool in a given situation.

Paper I concludes that the two key aspects of the decision context determining the choice of tools are the object of the study and the impacts of interest. Other aspects either influence how the chosen tools are used, or have an indirect influence via the key aspects.

In Paper IV, characteristics employed to describe analytical tools to be used within an SEA were: 1) Definition of system boundaries; 2) type of environmental changes included in the assessment; 3) degree of site-specificity desired; 4) degree of quantification desired; 5) degree of aggregation of results desired; and 6) preference of information type according to the DPSIR-model (the first five based on suggestions in Paper I).

The suggestions provided in Paper I and IV may be narrowed down to three key factors (substantive issues[3]) directly influencing the choice of tools:

- Object of study
- Impacts of interest
- Desired information type regarding site-specificity and according to the DPSIR-model

Other aspects of the decision context may influence how chosen tools are used rather than the choice of tool. Examples are degree of aggregation and level of detail. The definition of system boundaries is affected by the object of study, but also influences

[3] English (1999) splits the context of environmental decision-making into substantive issues and social settings.

how tools are used through defining the system under study and determining the scope of the inventory of data (geographically and in time).

The possibility to use quantitative data is influenced by the impacts that are of interest, since certain impacts are hard to quantify, e.g. biodiversity. The degree of quantification desired may also affect the choice of tool if qualitative studies are wanted, but since quantitative tools were the focus here this is not discussed in any detail.

Cultural context may also influence the choice of tool through knowledge and experience of, and access to, different ESA tools. The opinion of a tool may depend on various reasons. Concerning credibility, it is of value to standardise the use of specific tools. The standardisation of LCA (ISO 1997, 1998, 1999) gives an advantage as regards credibility over tools like Ecological footprint and MIPS. Olsson and Sjöstedt (2004) see a risk of the knowledge and availability of tools framing the research issue. The cultural context, through values and practice, also influences the impacts that are considered to be of interest and the kind of information that is regarded as relevant. As suggested by English (1999), Tukker (2000) and Wrisberg et al. (2002), the cultural context must be understood in order to define which information that is needed and relevant. The cultural context in this case indirectly affects the impacts of interest and the information type desired and may also affect the definition of object. Wrisberg et al. (2002) state that 'deeper, structured analysis will only be appropriate if there is agreement on the criteria to be used for evaluation'. If there are different opinions regarding the criteria the decision should be based upon, then results of detailed studies based on data of high quality might be of limited use as decision support.

Wrisberg et al. (2002) also suggest other context characteristics that put different requirements on the use of tools. These characteristics include e.g. level of improvement, complexity of system change, aspiration of decision-maker and chain control. These characteristics could be argued to mainly affect how tools are used, or indirectly affect the choice of tool through influencing which impacts to consider, etc. They can also have an influence through being part of different cultural contexts.

The amounts of resources available would influence how the tools are used (screening study, detailed study or no study). Comprehensive, detailed studies may be used to develop indicators or checklists which can be used in subsequent decision-making situations. Studies already performed on similar issues may be used as information input in a new decision-making situation. This emphasises the need for transparent studies and for users of results from earlier studies to take their time in interpreting and applying the results in their own case. In the Swedish proposal for a tax on waste for incineration (SOU 2005:23), as an example, several earlier studies on e.g. environmental aspects of different waste management options were used as information sources (Finnveden et al. 2000, Björklund and Finnveden 2005, Eriksson et al. 2005).

Starting from the key factors affecting the choice of tool as suggested above, differences among the tools are obvious. There are certain tools for certain objects, but there are also tools that may be used for several different objects (as can be seen in Figure 3). Thus the object is not a totally exclusive parameter in the choice of tool. Some tools are demanded by law for certain objects, e.g. EIA and SEA. However, there is no regulation of how the assessments and other activities within the EIA or SEA should be performed and what tools could or should be used within these procedural tools.

Request regarding which impacts to include make the range of alternative tools narrower. Impacts of interest may be the inflows to the studied system if inflows are considered of higher strategic value than outflows (Robért 2000). Decision-makers may have different frames wherein decision support should fit. Different types of decision support are requested with the risk perspective than with the precautionary perspective[4] (Tukker 2000). When focusing on inflows, tools like TMR, Energy analysis or MIPS may be used depending on object. In the case of different perspectives or frames, it may be harder to decide which impacts or which information is desired. Risk Assessment serves well as decision support for decision-makers with a risk perspective, but not for those with a precautionary perspective. If the focus is on land use, an Ecological footprint may be appropriate. The wish for a simultaneous assessment of environmental and economic issues could lead to the choice of CBA or LCC.

In decision-making, both site-specific and site-independent information may be of interest. The interest in site-specific information may be partly influenced by the object. In cases of locating facilities, site-specific information is relevant and feasible, but when discussing national policies more general information may be sufficient. Some tools have been developed for site-specific analysis (e.g. IPA and RA). Others can be modified to become more site-dependent. LCA-methodology is being developed to facilitate site-dependent analysis (e.g. Potting et al. 1998, Krewitt et al. 2001). In LCA as it is generally performed, there is no consideration regarding when or where emissions take place. Site-dependent approaches aim at incorporating differences between for example emissions made in densely populated areas and in rural areas or in the middle of the ocean and in sensitive lakes. The problem of using site-specific data in LCA lies in the concept of the life cycle perspective. It would in most cases be unfeasible, if not impossible, to inventory the whole life cycle of a product using specific data in inventory and impact assessment. If the decision-maker is very concerned about local effects, e.g. when deciding on the location of

[4] With a precautionary perspective a focus on inflows may be preferred.

some facility, site-specific assessments should be made and could be combined with a site-independent study, e.g. with a life cycle perspective giving additional information, as suggested for EIA and LCA in Tillman et al. (1997).

The preference of information type according to the DPSIR-model also influences the choice of tool. In the thesis this have not been handled explicitly. Most of the tools considered provide information on pressure and impact, e.g. RA provides information on impact, TMR on pressure[5] and LCA may provide both.

Factors that affect the definition of object of study, the choice of impacts to be considered and information type desired are interesting but are not covered in this thesis.

Concerning the degree of aggregation, there is always the possibility of using some ad-hoc valuation method(s) to transform a multi-dimensional result into a one-dimensional if that is preferred. In some of the ESA tools covered here, weighting is part of the tool. CBA and LCC convert all interventions into monetary terms. In a way Ecological footprints, Material Flow and Energy analyses weight the impacts through measuring effects in area, mass and energy units respectively. MAA also includes the weighting of different impacts. In LCA, weighting is an optional element. Bengtsson (2001) notes that the cultural context affects whether a single result is wished for and found reliable. He argues that the presentation of a single result using only one method for weighting is not satisfactory if the opinion among the decision-makers is that there are multiple values and attitudes that affect the modelling and evaluation of environmental impacts. On the other hand, if there is a belief that science can judge which alternative is the best, a single result may be asked for (Bengtsson 2001). Thus the issue of weighting is also dependent on the cultural context of the decision.

Transparently describing underlying assumptions and making possible interpretation of the results is a common challenge of importance when using ESA tools.

It must not be forgotten that more than one tool may be used to get a broader, more robust, result. Combinations of tools may be made up of tools that differ regarding object, impacts considered, site-specificity, quantification, etc.

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[5] TMR provides pressure information if natural resources are the focus, but rather driving force information if environmental impacts are focus of the study.

4.2. Waste management decision-making

Several different ESA tools may be used for supporting waste management decision-making. There is of course a large variation regarding decision context that needs to be considered here. The object of waste management decision-making may vary - it may be a waste policy, plan or programme; it may be an activity or project such as the location of a waste treatment facility; it may be a service such as the treatment of waste of some kind; etc. According to Figure 3, there are different tools that are applicable for these different objects. Of course, there is also variation regarding the cultural context which should be considered in each specific decision situation and the use of tools, especially resource-intensive types, should be preceded by an assessment of the cultural context and a definition of agreed criteria for evaluation.

Most of the ESA tools discussed in this thesis can be said to have more or less relevance in waste management. Some of them are mentioned here.

EIA should be used in waste management by companies requesting the development of waste management facilities and SEA should be used by public authorities when making waste programmes or plans that are expected to give rise to significant environmental impacts. These procedural tools could include analytical tools providing relevant information, e.g. RA, LCA or IOA. For projects and plans, CBA could also be used. In strategic decision-making, such as planning, future studies may also be useful. The System of Economic and Environmental Accounts (SEEA) is an accounting tool that provides information on a national level, including information on waste per sector and per waste management option (SCB 2006). These data may also be used as reference in other studies.

Regarding policies, Nilsson et al. (2005) tested the use of analytical tools within an SEA on a waste incineration tax proposal. They tested different paths, including a qualitative pathway, an LCA pathway and a site-dependent pathway and found that these provided complementary information. The authors concluded that the choice of analytical tools within an SEA depends on the decision context, e.g. the actors involved and underlying frames and preferences. The choice between the three pathways tried by Nilsson et al. (2005) would be a choice of impacts to include, quantification and level of detail. It is not always the case that actors involved are part of the choice of tool, and then it is extra important to try to consider all actors and their underlying frames, preferences, etc. to choose a tool that provides relevant information.

EMS is used to structure the management of environmental issues in a company or an organisation and could of course also be used in the waste management sector.

If the focus of the decision is waste management services, environmental aspects could be analysed using LCA. If the impacts of interest are natural resources used (inflows), Energy analysis or MIPS could be used. An Ecological footprint could also be made to illustrate the impact of waste management, but is probably more relevant as part of the larger footprint of a region, product, etc. To study certain substances within the waste system, RA or SFA could be used.

An example of an ESA tool being used to assess different municipal waste management systems is the LCA study presented in Papers II and III. This kind of assessment could be performed within an SEA or separately. Some overall conclusions from the study, which considered waste management options for combustible and recyclable or compostable municipal solid waste, were that recycling of paper and plastic material was in general favourable regarding energy use, emissions of greenhouse gases and total weighted results. One important exception was when recycled plastics replace impregnated wood. In this case, recycling of plastics was less favourable than incineration with respect to energy use and emissions of greenhouse gases, although the difference was rather small.

According to the LCA study, incineration was in general favourable over landfilling with regard to overall energy use, emissions of greenhouse gases and the total weighted results. However, there were some aspects that might influence this ranking, e.g. the modelling of landfills. When shorter time periods were used (in the order of a century), landfilling was favoured and in some cases a preferable option over incineration.

Our results suggest that the current waste hierarchy is valid as a rule of thumb.

Other LCA studies have given similar results. Eriksson et al. (2005) concluded that there are only small differences between recycling and incineration but that there is in general an advantage for recycling of plastics. Finnveden and Ekvall (1998) and Ekvall and Finnveden (2000) reviewed several studies on recycling versus incineration of paper packaging materials and noted that the use of total energy and emissions of gases contributing to global warming was lower in cases where biofuel was the competing energy source, thus getting similar results to those in our LCA study. Björklund and Finnveden (2005) also reviewed several studies of waste management strategies using a life cycle perspective and compared results for emissions of greenhouse gases and total energy use. They found that the ranking of recycling before incineration was rather robust for non-renewable materials, the key factor that could influence the results being what material the recycled material replaced. For renewable materials, they found that there were a number of factors that could influence the resulting ranking of waste management strategies, including paper quality, energy source avoided by incineration, and energy source at the paper mill. An LCA of using different fuels for district heating has also considered waste incineration (Eriksson et al. 2006). In this study the results again show that incineration of waste is preferable if landfilling is the alternative waste management. If materials recycling is the alternative waste management option, however, incineration of waste is not preferable. In most cases waste incineration avoiding materials recycling proved to be a less environmentally preferable alternative than incineration using other fuels (incineration of biofuel or natural gas).

The policy implications of the results of our LCA study depend on the aim of the policy. The discussion here takes as a starting point that the aim of the policy is to reduce the total use of energy and emissions of gases contributing to global warming (normally the aim of a policy is a mixture of different environmental, social and economic aspects).

The results from our LCA study suggest that a policy promoting recycling of paper and plastic materials should be pursued, preferably combined with policies promoting the use of recycled plastics replacing plastics made from virgin materials. The study also suggests that if biomass saved from increased recycling can be used as a fuel replacing fossil fuels, this can reduce emissions of greenhouse gases. Although increased long-distance transportation by truck in general did not affect the ranking for recycling and other treatment methods, transportation of waste materials should be minimised and it is of importance to design source separation systems so that transportation by passenger cars can be avoided, since such transport can reduce the benefits from recycling.

In the system studied, heat from incineration of waste replaced either heat from forest residues or natural gas. If the waste can replace oil or coal as energy sources, and neither biofuel nor natural gas is an alternative, a policy promoting incineration may be successful for paper materials regarding emissions of greenhouse gases (Finnveden and Ekvall 1998, Ekvall and Finnveden 2000).

In situations where recycling is not an alternative, a policy promoting incineration is generally better than a policy promoting landfilling. However in a short time-perspective, incineration may lead to increased emissions of greenhouse gases compared to landfilling of materials that are not easily degradable, such as plastics and some constituents of paper materials.

For food waste, the present study did not provide clear answers to the comparison between incineration and anaerobic digestion. However, neither landfilling nor composting seems in general to be an attractive strategy if the aim is to reduce the use of energy and emissions of greenhouse gases. An exception may be if large transport distances can be avoided. In such situations, home or small-scale composting may be attractive, although this was not studied here.

The most important impact categories in our LCA study, according to the Ecotax 98 weighting method, are in general abiotic resources, global warming, toxicological impacts and photo-oxidant formation (Finnveden et al. 2000). This is in line with the statement that the Swedish national objectives where the waste system has the relatively largest contribution are *Reduced Climate Impact* and *A Non-toxic Environment* (Naturvårdsverket 2005d). It is of interest to note this, since toxicological impacts are often excluded from LCAs and similar studies of waste management systems. The use of different impact assessment methods here makes the uncertainty more visible and thus enables decision-makers to include this important aspect.

Results from an LCA study (or another ESA study) could be used as decision support in decision-making where no case specific study is made, or it could be used as additional information combined with more case-specific studies. The report underlying Papers II and III is referred to in SOU 2005:23, and is thus used in decision support. For national level strategic decisions, more specific data may not be asked for, but may still be preferable if the decision-maker and stakeholders can be part of, or can influence, at least the goal and scope setting of the study. In the interpretation phase, collaboration between the performer of the LCA (or other method) and the stakeholders/users of the results is also preferable. If previous studies are used in a new decision-making situation, it is important that the system boundaries, value choices and assumptions leading to the results are fully understood.

Paper III highlights the importance of value choices such as boundaries in time. For strategic decision-making this is of the utmost importance. Showing the effect of different underlying value choices and assumptions also underscores that there is a clear need for transparency in environmental assessments, in order to make it possible for decision-makers and other stakeholders to understand the results and the factors on which they depend.

4.3. Conclusions

Since there is a choice between several ESA tools for the assessment of environmental impacts using a systems perspective, this thesis attempted to identify key aspects for deciding which tool to use in a given decision-making situation. The key factors identified were the *object*, *impacts* of interest and the preference of information type regarding *site-specificity* and according to the *DPSIR*-model. The cultural context was also considered to influence the choice. As there is a need for tools for different objects and providing different kinds of information, there is indeed a need for a number of ESA tools in decision-making. These tools may be used in combination to give more comprehensive and robust decision support.

Waste management decision-making was used here as an example. Several ESA tools may be used in this field, for example Environmental Impact Assessment (EIA), Risk Assessment (RA) or Cost-Benefit Analysis (CBA) e.g. for location of waste management facilities; Strategic Environmental Assessment (SEA) for policies, plans and programmes; Life Cycle Assessment (LCA), Energy analysis, Material Intensity Per unit Service (MIPS) or Life Cycle Costing (LCC) e.g. for comparing waste management options, RA e.g. for potential effects of substances in waste reused, Substance Flow Analysis (SFA) e.g. to follow substances in the waste; and Environmental Management System (EMS) for waste management companies.

LCA may be used for comparing different waste management options on its own or within an SEA. An LCA study was performed and the results showed that recycling of paper and plastic materials was in general favourable over incineration with regard to energy use, global warming and total weighted results. Incineration was in general favourable over landfilling with regard to the same impact categories. Key aspects influencing the results were the alternative material and the alternative energy source that were avoided and also the time perspective chosen. The study further suggested that if biomass saved from increased recycling could be used as a fuel replacing fossil fuels, this could reduce emissions of greenhouse gases. It was shown that the possible need for transportation of sorted waste by private car in a management option might influence the results. Toxicological impacts were shown to be important in this study (using the weighting method Ecotax 98). This is of interest since this impact category is often excluded in studies of this kind.

Based on results like these, decision-makers may get general advice on choice of waste management options, but more importantly they also gain knowledge regarding the aspects that may have a significant influence on the choice of alternative, thus making more robust decisions possible. The use of other ESA tools may complement the information provided. Since a product or a service is the object of an LCA, the study is performed and results are presented from that view. LCA covers potential impacts throughout the life cycle and does not include local and specific impacts. Impacts included are natural resource use and environmental impacts. LCA is a resource-intensive tool and results from earlier studies could be used as information sources in subsequent decision-making situations.

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