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Environmental Assessment of a Waste Incineration Tax

Case Study and Evaluation of a Framework for Strategic Environmental Assessment

2003-09-25

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Preface

This report and the case study described here have been produced within the framework of three different projects:

1. Robust strategies for utilising energy from solid waste. This project is financed by the Swedish National Energy Administration. The aim of the project is to suggest robust and flexible strategies for utilising energy from solid waste. The project runs until the end of 2003 and includes several subprojects of which this case study, focusing on a waste incineration tax as a policy tool, is one. The project will be finalised with a synthesis report by the end of the year 2003.

2. Strategic Environmental Assessment (SEA) within the energy sector. This project is also financed by he Swedish National Energy Administration. The aim of the project is to develop methods for SEA within the energy sector. In this case study a framework for SEA is tested and evaluated. The project will be finalised with an introduction to SEA in the energy sector by the end of the year 2003.

3. Common techniques for environmental systems analysis. This project is financed by the Foundation for strategic environmental research. The overall aim of the project is to develop common techniques and components that can be used in several different environmental systems analysis tools. The development of a weighting method (Ecotax02) and site-dependent characterisation methods that are used and tested in this case study have been done within this project. The project includes several subprojects and will be finalised in 2006.

Project number one is performed at fms (Environmental Strategies Research Group) which is a cooperation between researchers from the Department of Environmental Strategies at FOI (Swedish Defence Research Agency), several departments at KTH (the Royal Institute of Technology) including the Division of Industrial Ecology and the Department of Infrastructure, and several departments at the Stockholm University. Project number 2 is done in cooperation between fms and Stockholm Environment Institute (SEI). Project number 3 is done in cooperation between fms, SEI, Environmental Economics at Göteborg University, Energy Systems Technology at Chalmers University of Technology, Environmental Technique and Management at Linköping University and Environmental Statistics at Statistics Sweden.
The results presented in this report are expected to be published also in several papers in scientific journals. The report has several appendices that are published as separate reports on our homepage www.fms.ecology.su.se. They are:
Appendix A: Current (1998-2000) waste flows
Appendix B: Chemical composition of waste fractions
Appendix C: Transport data
Appendix D: Process data records
Appendix E: LCI results

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Contents

Abstract

A framework for Strategic Environmental Assessment (SEA) is tested in a case study on a proposed waste incineration tax. Also included is testing of developed methods for valuation and site-dependent life cycle impact assessment. The results indicate that although a suggested waste incineration tax of 400 SEK/ton is likely to lead to environmental improvements, these are small compared to the potential improvements as shown in more visionary scenarios. In order to go in this direction a waste incineration tax based on the content of fossil carbon in the waste would be useful. The framework for SEA includes several different pathways. These have different advantages and disadvantages and provide different types of information. It is therefore suggested that they largely complement each other and that the choice of methods should be done in relation to the function of the SEA and the questions asked.

Summary

The background of this study is threefold; one is related to the development of Strategic Environmental Assessment (SEA), the second is the interest in evaluating different strategies for management of solid waste, and the third is to develop some methodological components.

The main purpose of SEA is to facilitate early and systematic consideration of potential environmental impacts in strategic decision-making. It is intended to be used on policies, plans and programmes. The growing significance of SEA as a form of support to decision-making is manifested by the recent EC directive (2001/42/EC) on the assessment of environmental effects from certain plans and programmes. However, a number of challenges need to be overcome for SEA to be an effective tool.

As a procedural tool, SEA can include a number of different analytical tools. However, appropriate methods need to be established. For instance, SEA guidance often refers to Environmental Impact Assessment (EIA) -type analyses but it is often difficult to use the methods associated with project EIA in SEA because they are adjusted for site-specific information and local impacts whereas SEA often is not site-specific and can often be primarily concerned with cumulative and indirect impacts. The lack of methodological guidance for SEA also acts as a barrier to the implementation of SEA in general and the European directive on SEA in particular. We have in an earlier paper suggested a framework for the integration of analytical tools in the SEA process with a special focus on the energy sector. One aim of the present study is to evaluate parts of this framework by testing it on a case study.

Waste management is undergoing changes in Sweden and in many other countries. Among the recent policy decisions in Sweden influencing waste management practises are the introduced tax on landfilled waste and bans on landfilling of combustible and organic waste. A tax on incineration of waste has also been discussed and is the subject of a recent governmental commission.

It is of interest to evaluate the environmental impacts of a tax on incineration of waste and we choose this as a case study for evaluation of the suggested framework for SEA. This case study is also a part of a larger study with the aim of suggesting robust and flexible strategies for waste management.

The governmental commission included some environmental assessments of the suggested tax levels compared to a no-action alternative. The environmental assessment was mainly based on previously published Life Cycle Assessments (LCAs) of waste management.

The framework for SEA that is tested here contains several alternative paths that can be chosen. As a part of the evaluation of the framework we are here testing several of these alternatives in order to see if they produce the same results and to test their feasibility. The environmental evaluation here is more comprehensive than previous LCA studies also by including more waste fractions. In addition some aspects of geographical differences within Sweden are tested in a site-adjusted assessment described below.

As an SEA case study it has several interesting features. It is more complex than a typical SEA within the energy sector since it involves also the waste management sector. This is also one reason why we are only testing parts of the framework. As noted above, SEA can be used to evaluate policies, plans and programmes. Most SEA cases in the literature are on a local or regional planning level. This case study is therefore of interest to demonstrate a study on a policy level.

The governmental commission evaluated three levels of an incineration tax as well as a no-action alternative. We choose to evaluate the no-action alternative, one of the tax-levels and two additional and more visionary alternatives for a more sustainable waste management. In the study we are also briefly discussing possible policy instruments for achieving these visionary alternatives.

There is also a third overall aim of the study to test some newly developed methods and data. Methods and data for a site-adjusted assessment are described as well as an update of a previously developed valuation method, the Ecotax method.

An introduction of a waste incineration tax is likely to result in environmental improvements. A waste incineration tax would be accompanied by an increased landfilling tax, in order to ensure that landfilling is avoided. The environmental improvements are partly accomplished by diverting materials from landfills to recycling as a result of the increased landfill tax, and partly by diverting materials from incineration to recycling.

Although the studied alternative of a waste incineration tax of 400 SEK/ton is likely to lead to environmental improvements, the improvements are small compared to the potential improvements as shown in the visionary scenarios developed here. In order to go in the direction of these visionary scenarios, a combination of a non-differentiated and a differentiated incineration tax would be useful. The differentiated incineration tax should be based on the content of fossil carbon in the waste. An incineration tax of approximately 2000 kr/ton of plastic materials would harmonise the waste incineration tax with the carbon dioxide tax. Such a tax would make recycling of plastic materials economically attractive.

A site-adjusted analysis indicates that the ranking between different alternatives are similar in different parts of the country. There is thus no indication that different waste policies should be used in different regions.

None of the methods used in this case study provide the tools necessary to estimate what may be called first order effects of an new strategy or policy. In our case, the first order effects would be the redistribution of waste flows as a result of introducing the waste incineration tax. In the case of a new energy policy, it might be changes in energy supply and use. Because we had access to the commission report (SOU 2002:9), which included estimates of waste flows in the no-action alternative and as a result of introducing the waste incineration tax, such tools were not necessary in our case. Methods for estimating first order effects include energy- and waste economic models and different types of scenario techniques, rather than the environmental methods of primary focus here.

The methodological pathways discussed here all have different advantages and disadvantages and provide different types of information. It is therefore suggested that they largely complement each other, rather than compete with each other.

A major advantage with a qualitative approach is that it can cover all relevant environmental issues. It can mainly be used for identifying aspects which are judged to be of importance. It is less useful for supporting choices between alternatives. Since the approach is less structured and quantitative, it is open for criticism concerning its subjectivity and lack of reproducibility.

A quantitative traditional LCA approach will avoid some of the drawbacks of a qualitative method. However, it can typically not cover all relevant types of environmental impacts and will have some datagaps also for impact categories which are included. It can therefore be useful to complement a traditional LCA with a structured qualitative approach in order to check that important issues are included in the overall assessment. A traditional LCA without weighting methods can be used to identify critical aspects, and also support choices between alternatives. However, if different impact categories point in different directions, no firm conclusions concerning the preference of different alternatives can be drawn.

Some type of weighting or valuation method is often useful when supporting choices between alternatives. It is also necessary in order to identify which environmental impacts are most important. In order to be useful, the valuation methods must however have some credibility among the stakeholders. The use of several valuation methods can sometimes increase the credibility of the results.

A drawback of a traditional LCA-approach is the lack of site-dependent information. Adding such information will add to the accuracy of the model. It can also allow other types of question being asked such as, Are there differences in different parts of the relevant region which warrants different policies?

A risk assessment approach is not used here. Risk assessments could be used for answering other types of questions. It could for example be used to study whether air quality norms are exceeded. This is particularly interesting when SEA is applied to energy and transportation planning. It is thus useful for identifying certain types of critical issues which can not be handled with other methods.

Since different methodological approaches can produce different types of answers, it is suggested that a careful consideration is given at the start to this issue: What are really the questions that the assessment should try to answer. It is also suggested that if any of the quantitative approaches is used, it is complemented by a structured qualitative approach with the aim of identifying critical issues which may go unnoticed in the quantitative approaches.

Site-dependent characterisation factors for Sweden have been developed for impacts on human health and ecosystems from emissions of NOx, SO_2 and particulates. For health impacts, the results are different characterisation factors for different parts of the country and from different stack heights. For ecosystem damages, the results are only small differences for the used definitions of ecosystem damages.

The developed characterisation factors are applied to the case study. It is interesting to note that although there are differences in characterisation factors, this did not affect the results and the conclusions.

The Ecotax weighting method as developed by Johansson (1999) has been updated and applied to the case study. The results can be compared with the results from other weighting methods. It is interesting to note that different weighting methods provide the same ranking between different alternatives, but for different reasons. Thus different methods identify different aspects as the most important ones. The Ecotax02 method generally point out resource use, emissions of gases contributing to climate change, and toxicological impact as the most important ones.

Different versions of the Ecotax method are used where minimum or maximum weighting factors are used in cases where there are large uncertainties, i.e. especially for the valuation of resources and toxicological impacts. These different sets produce significantly different results in absolute values. It is also interesting to note that the choices concerning the time frame of the study can have a significant influence on the results. If a cut-off is made after approximately one century, and emissions occurring after this time period are neglected, significantly lower results are produced **compared to the cases where also long-term emissions are taken into consideration.**

1 Introduction

The background of this study is twofold; one is related to the development of Strategic Environmental Assessment (SEA) and the second is the interest in evaluating different strategies for management of solid waste.

The main purpose of SEA is to facilitate early and systematic consideration of potential environmental impacts in strategic decision-making (Therivel and Partidario, 1996; Partidario, 1999). It is intended to be used on policies, plans and programmes. The growing significance of SEA as a form of support to decision-making is manifested by the recent EC directive (2001/42/EC) on the assessment of environmental effects from certain plans and programmes (Feldmann et al., 2001). However, a number of challenges need to be overcome for SEA to be an effective tool.

As a procedural tool, SEA can include a number of different analytical tools (Wrisberg et al., 2002). However, appropriate methods need to be established. For instance, SEA guidance often refers to Environmental Impact Assessment (EIA) -type analyses but it is often difficult to use the methods associated with project EIA in SEA because they are adjusted for site-specific information and local impacts whereas SEA often is not site-specific and can often be primarily concerned with cumulative and indirect impacts (e.g. Petts, 1999b). The lack of methodological guidance for SEA also acts as a barrier to the implementation of SEA in general and the European directive on SEA (European Parliament, 2001) in particular. We have in an earlier paper suggested a framework for the integration of analytical tools in the SEA process with a special focus on the energy sector (Finnveden et al, 2003). One aim of the present study is to evaluate parts of this framework by testing it on a case study. SEA and the framework are described in more detail in Chapter 2, Strategic Environmental Assessment and in Chapter 3, Approach and framevork.

Waste management is undergoing changes in Sweden and in many other countries. Among the recent policy decisions in Sweden influencing waste management practises are the introduced tax on landfilled waste and bans on landfilling of combustible and organic waste. A tax on incineration of waste has also been discussed and is the subject of a recent governmental commission (SOU 2002:9). Waste management policies in Sweden are discussed in more detail in Chapter 4, About the proposed waste incineration tax.

It is of interest to evaluate the environmental impacts of a tax on incineration of waste and we choose this as a case study for evaluation of the suggested framework for SEA. This case study is also a part of a larger study with the aim of suggesting robust and flexible strategies for waste management.

The governmental commission (SOU 2002:9) included some environmental assessments of the suggested tax levels compared to a no-action alternative. The environmental assessment was mainly based on previously published Life Cycle Assessments (LCAs) of waste management.

The framework for SEA that is tested here contains several alternative paths that can be chosen. As a part of the evaluation of the framework we are here testing several of these alternatives in order to see if they produce the same results and to test their feasibility. The environmental evaluation here is more comprehensive than previous LCA studies (e.g. Finnveden et al, 2000 and Sundqvist et al, 2002) also by including more waste fractions. In addition some aspects of geographical differences within Sweden are tested in a site-adjusted assessment described below.

As an SEA case study it has several interesting features. It is more complex than a typical SEA within the energy sector since it involves also the waste management sector. This is also one reason why we are only testing parts of the framework (cf. Chapter 3, Approach and framework). As noted above, SEA can be used to evaluate policies, plans and programmes. As noted below, most SEA cases in the literature are on a local or regional planning level. This case study is therefore of interest to demonstrate a study on a policy level.

The governmental commission evaluated three levels of an incineration tax as well as a no-action alternative. We choose to evaluate the no-action alternative, one of the tax-levels and two additional and more visionary alternatives for a more sustainable waste management. In the study we are also briefly discussing possible policy instruments for achieving these visionary alternatives.

There is also a third overall aim of the study to test some newly developed methods and data. In Chapter 5 Environmental analysis methods methods and data for a site-adjusted assessment is described as well as an update of a previously developed valuation method.

2 Strategic Environmental Assessment

The integration of environmental concerns into strategic decision-making and policy-making has been widely recognised as an essential feature for moving towards a more sustainable development in all policy sectors. Strategic Environmental Assessment (SEA) is a decision support tool that aims at integrating the environmental aspects of decisions in a structured manner. What makes the SEA strategic? The first connotation is that the object of assessment is policies, programmes and plans that have long-ranging implications on broader aspects of society. The second connotation is the up-stream focus, attempting to not just carry out an environmental analysis of decisions already made. A commonly used definition is: "a systematic process for evaluating the environmental consequences of a proposed policy, plan or programme initiatives in order to ensure they are fully included and appropriately addressed at the earliest appropriate stage of decision making on par with economic and social considerations" (Sadler and Verheem, 1996).

2.1 Components of SEA

SEA developed rapidly in the 1990s on the conceptual basis of environmental impact assessment (EIA) (Therivel et al., 1992). It was originally considered to be an application of EIA in strategic decisions, such as the formulation of policies, plans and programmes. The motivation was to bring the environmental issues up-stream in the decision-making hierarchies. By taking environmental considerations into account when preparing for strategic decisions, new possibilities are opened for influencing choices and directions towards a more sustainable society. More options for mitigation of harmful effects may be feasible the sooner these aspects are considered in the decision-making process.

The recent European Directive on environmental assessments of certain plans and programmes (Government of South Africa, 2000; European Parliament and Council of the European Union, 2001) provides a legislative framework for SEA that puts it high on the national political agendas as it is being implemented in national legislation all over the European Union in the coming years.

There are many varieties of SEA, but as a general template it can be said to include the following main components (Nilsson et al., 2001). These are not necessarily followed in a step-by-step process, although the general stream of activities is easy to follow.

Scoping: This component handles what to include in the SEA, the temporal and spatial boundaries, the identification of what environmental issues to cover as well as what type of delimitations are envisioned for the process in terms of participation of various stakeholders, experts and the public.

Situation Assessment: This component provides the baseline analysis to determine the environmental concerns, challenges and opportunities in the areas or sectors that are affected by the proposed plan. It leads to the formulation of objectives, criteria and indicators that informs the subsequent components.

Alternatives: This component makes the formulation and selection of decision alternatives for analysis in close deliberation with the decision-makers. In some cases, it is possible to introduce one or two 'sustainability alternatives' based on visions and objectives as part of the package.

Environmental analysis: This is in many cases the central part of the SEA, as it deals with the identification and analysis of environmental pressures and impacts of the various alternatives. Often, it works with checklists and matrices although more refined options include scenario analysis and environmental systems analysis to give substantive predictions. The environmental analysis will normally generate large amounts of qualitatively different data.

Valuation and information analysis: This component seeks to aid the decision process by making a transparent and deliberated weighting of impact information through, for instance multi-criteria analysis, economic valuation or other methods. It also includes analysis of uncertainties to account for the robustness of results.

Decision and documentation: This component is concerned with the structured presentation of the SEA process and results, in what ways the decision has taken environmental concerns into account, and the motivation for the choices made.

Mitigation and monitoring plan: This component provides a plan for measures to enhance the environmental performance as well as for monitoring of the performance in the implementation of the selected alternative, including time frames, performance indicators, allocation of resources and roles and responsibilities.

Successes of SEA implementation are mixed. Most evidence and cases of SEAs in literature have been performed in municipal and regional planning situations while SEA at the policy level is conspicuously missing (Therivel and Partidario, 1996). There seems to be two factors that explain this. The first is the poor fit between the SEA as currently constructed and the decision-making process. The second is the uncertainties regarding appropriate methods for SEA at higher levels of decision-making (Nilsson and Dalkmann, 2001).

2.2 SEA methods depending on context

SEA is a process tool or an approach within which a range of different analytical tools and methods can be applied. The question then becomes, which methods should we use in any particular context? Although the environmental analysis is central in most SEA frameworks, it is remarkable how little analytical and methodological guidance is given in literature and guidelines. Instead, the assessor is usually left with matrices and checklists that provide little help about what is needed to do the analysis. What, then, is needed?

To start off, the assessor should always ask: What decisions are we trying to influence and how can we best do that? (Therivel et al., 1992). Local, regional and national planning and policy levels each present their own particular institutional settings and particular information and analysis needs that the SEA process will have to adapt to. This means that also the analytical needs, the tools and methods will differ. As a result, several aspects of relevance to the selection of SEA tools might differ depending on the decision context, such as the environmental issues, the spatial and temporal scale and boundaries of the analysis, the participants in the decision, and the regulatory framework within which decision-making takes place (Dale and English, 1998). The assessor must also establish a clear role for the SEA in the process (Kørnøv and Thissen, 2000). Should the role of the SEA be to help find the best solution, or should it be there to contribute a particular perspective? Or is it to be used as a way to develop creative compromises of sustainable development character? The account below will focus on the implications of context-dependency on some of the central components of the SEA. In essence, the framing of the SEA and what goes into it takes place in the scoping stage.

A critical part of the scoping is the analysis of the decision-making process and institutional context. What is the decision we want to influence and who are the actors involved? Depending on political culture, planning traditions, rules and other institutional constraints, the decision-making process will differ. The possible role and focus of the SEA depends on this analysis. Also, different sectors and levels of governance will entail different groups of actors. The local planning level is likely to involve local businesses, energy companies, the municipal administration, the Agenda 21 office, and interested citizens. The national policy level, on the other hand, is likely to involve large national industries and industry organisations, renowned scientists, government experts, national NGOs and other interest organisations. To engage different groups in a meaningful deliberation around the SEA will require different communicative approaches and supporting analytical input. For instance, various audiences will have different perceptions of what types of indicators or information input is meaningful and adequate. This leads us the question of information needs.

What do we need to know from an environmental point-of-view? Information needs differ substantially from local to national levels. National environmental quality objectives, as has been established by the Parliament in Sweden, is an important input to establishing the information needs (Naturvårdsverket, 1999; Statens Offentliga Utredningar, 2000). Local-level planning might be framed by regionally or even locally disaggregated subsets of national environmental objectives. On the basis of these needs, analytical methods are selected that can be, for instance, qualitative or quantitative, generic or site-specific and pressure-, state- or impact-focused. The objectives might be presented in a different form depending on level, for instance as an impact target rather than a pressure target, in the PSIR indicator framework (OECD, 1994). When studying the proposed alternatives against a ”pressure” objective, a different tool might be required than if we study against a ”state” objective.

3 Approach and framework

As presented in Chapter 2, a typical SEA process contains a number of steps, including alternatives formulation, environmental analysis, and valuation. In each of these steps, a number of tools and methods can be used. For instance, the alternative formulation typically contains some kind of scenario or other future study method; environmental analysis might contain an LCA, or a risk assessment; and valuation might contain an economic valuation method or a multi-criteria analysis. The first phase of the project developed an SEA framework that incorporated a set of methodological components and their linkages within an SEA process (see **Fel! Hittar inte referenskälla.**). As stated earlier, the present study has been developed and carried out to test this methodological framework. The framework is tested through a case study on a current policy proposal in the waste-to-energy sector in Sweden. In addition to being a methodological exercise, it is our intention that results will also be instrumental in the evaluation and discussion on the actual policy proposals put forward.

Three alternatives, including the no-action alternative as a baseline, are examined within the methodological framework. The details of this case study and the alternatives examined are presented in Chapter 4, About the proposed waste incineration tax.

The study does not cover all components of the methodological framework. For instance, in our case study, the formulation of alternatives 0 and 1 are taken as given from the policy proposal, whereas alternative 2 is developed for goals achievement of greenhouse gas emissions reductions and energy recovery. In a full SEA, a future study component is applied to develop the set of alternatives to be considered and the description of them. Our departure point is the description of the technical system. From there, three principal analytical pathways in the framework are tested:

A qualitative pathway, based on qualitative information on environmental impacts, which keeps a multi-dimensional approach in valuating these qualitatively;

A traditional LCA pathway, based on quantitative information about environmental loads for each alternative, with traditional LCA characterisation.

A site-adjusted LCA pathway, based on quantitative information about environmental loads for each alternative with some site-dependent information. A site-adjusted LCA characterisation of certain air emissions (based on factors derived with a impact / risk assessment approach).

In these pathways, selected valuation methodologies are tested, including a qualitative methodology of environmental objectives, and Ecotax, Eco-indicator, and EPS, commonly applied in LCA systems. These are further described in Chapter 5.3 “Valuation”.

A risk assessment pathway is not tested in this case study. One reason for this is the resource requirements: the resources for this study did not allow a full test of this pathway, which is substantially more demanding in terms of modelling and expertise than the others. Secondly, questions were raised regarding the feasibility of this pathway when it comes to national policy-level SEAs. Assumptions need to be made concerning sites that would have an overwhelming influence on results. This is further discussed in Chapter 11.

Based on the empirical case testing, we are able to evaluate outcomes of the policy proposal and the alternatives within it, as well as evaluate how the different analytical methods worked in the context of an SEA. The full range of analytical methods in the framework is presented in Finnveden et al. (2003). The account below, of some of the components that this study is testing, is based on this article.

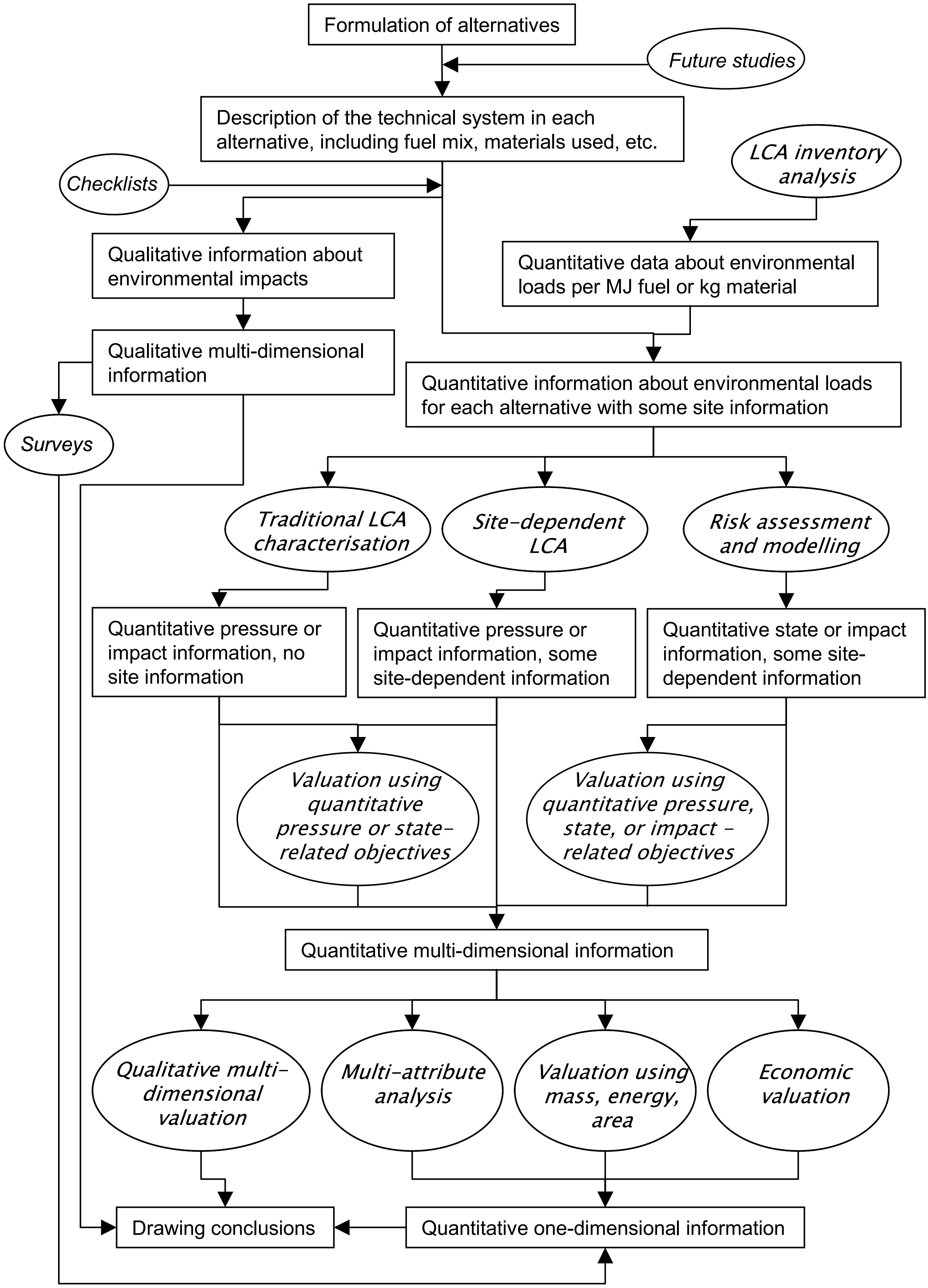

Figure 1. SEA Framework(Finnveden et al. 2003)

3.1 Future studies and describing the technical system

Before being able to say something about the environmental implications of a strategic decision, the assessor must develop an understanding of the systems changes in, for instance, a sector, a region or an infrastructure system. To understand what might happen as a result of a decision requires engaging in some type of future studies. There are several different approaches for studying the future. *Forecasts* indicate a probable future, often based on current and past trends. Forecasts are useful mainly for the shorter term and for well-defined areas (Höjer and Mattsson, 2000). *Scenarios* are helpful in the face of qualitative uncertainties. External scenarios focus on factors that cannot be controlled. They are useful to find strategies that are robust across a range of possible futures. *Policy scenarios* provide a template in which the planner can act, not only react. Finally, in *Back-casting* the future goal is first set and the ways to get there are described (Dreborg, 1996).

Just as there are many different approaches to future studies, there are varying needs in the SEA, depending on the decision context, the actors involved and the environmental issues at stake. The strategic decision might, for instance, have to do with supporting certain types of technologies or developing a large infrastructure. The implications for the environment of these decisions depend not only on the programme itself, but also on a range of other factors, such as external driving forces and exogenous policy variables. In this case, the use of policy scenarios in SEA has been recommended (Therivel and Partidario, 1996; Noble, 2000; Noble and Storey, 2001; Nilsson et al., 2002). A local planning process requires local environmental information. The future studies must follow suit, and provide localised scenarios. External factors could then constitute national-level decisions and local market and population conditions.

In many cases, the decision-makers make use of economic models to predict market impacts of a policy or an investment. The assessor needs to take this into account in devising the scenario analysis and might be able to use it for the SEA. It could be possible to develop a quantitative systems analysis by connecting environmental factors to the economic model already in use.

3.2 Environmental analysis

In the environmental analysis, which roughly covers the mid section of the framework, the analytical requirements regarding issues such as site-specificity, temporal scope, and system boundaries can also differ significantly when moving between sectors and institutional contexts and from the national to the local level of decision-making and planning. For instance, local planners need to know how their local energy production and consumption affect the environment in their municipality and, to some extent, in the places where the energy is supplied. National policy might not normally have to take such local considerations. Decisions at the municipal planning level needs at least partly localised and site-specific information about environmental implications. Decisions at national policy level might not need site-specific information but can rely exclusively on more generic environmental information.

Furthermore, different actors have different perspectives on what is a meaningful type of analysis. In certain contexts the use of quantitative information is considered crucial. Qualitative information about possible pros and cons from an environmental view-point will run the risk of going unnoticed in the decision-making process. In the case of chemicals, Tukker (1999) point out that industries often act within a “risk assessment frame” where risk assessments are considered useful and meaningful and management should be based on results from such studies, whereas environmental movements typically rely on a precautionary approach; a “phase-out frame”. Underlying these frames are often more fundamental values and beliefs regarding the role of scientific knowledge in policy-making (Andresen et al., 2000). Jasanoff (2000) demonstrates differences between countries in risk versus precautionary frames. These groups would then require different methods and types of information and different indicators will then carry meaning for different groups. Approaches that account for material flows are meaningful for the “phase-out frame”, whereas the “risk assessment frame” requires indicators on an impact level (Tukker, 1999).

In this case study we explore three alternative pathways for the environmental analysis within the framework; qualitative environmental analysis, life cycle assessment (LCA), and site-adjusted LCA.

Qualitative environmental analysis

A qualitative assessment can be carried out in a number of ways and it does not have one well-established method, like for instance LCA has. In a qualitative assessment the focus is on information available without having to engage in heavy calculation or modelling exercises. None the less it is a matter of gathering and structuring information. This information can for example be found in already published studies through literature surveys, in interviews with experts, in group discussions with experts or in a combination of those.

A way to structure the collection of information is to make a checklist over which environmental aspects to look for. This may also involve the formulation of indicators to be covered more in depth. The checklist and the indicators may in some respect be tied to different goals defined in the SEA. This could be societal goals such as the fifteen Swedish national environmental objectives. It could also be goals that are more specific to the decision situation at hand. The information gathered may then be organised in a simple matrix.

Life Cycle Assessment (LCA)

Life Cycle Assessment (LCA) is a tool to assess the environmental impacts and resources used throughout a product’s life from raw material acquisition through production use and disposal. The term “product” should be interpreted in a broad sense to include also services such as waste management.

An ISO standard has been developed for LCA providing a framework, terminology and some methodological choices (ISO, 1997, 1998, 1999). The basis for the calculations is the functional unit. This is a quantified measure of the functions of the studied product, for instance the production of 1 kWh electricity. All inputs and outputs are to be related to the functional unit. When different alternatives are compared, the functional unit is the basis for comparison and must be the same in all alternatives

According to the ISO-standard, an LCA is divided into four phases:

1. *Goal and scope definition*, in which the intended application, reason to perform the study, and intended audience should be determined. The functional unit should also be specified. An important aspect of the goal of an LCA study is whether it is retrospective or change-oriented, as it may influences the type of data applied.
2. *Inventory analysis*, where inputs and outputs to and from the systems are identified and quantified.
3. *Life cycle impact assessment*, aimed at understanding and evaluating the magnitude and significance of the potential environmental impacts of a product system. This phase is further divided into three mandatory elements:

3.1. Selection of impact categories, indicators for the categories and models to quantify the contributions of different inputs and emissions to the impact categories.

3.2. Assignment of the inventory data to the impact categories (classification).

3.3. Quantification of the contributions from the product system to the chosen impact categories (characterisation).

4. *Interpretation*, where the findings of either the inventory analysis or both the inventory analysis and the life cycle impact assessment phases are combined in line with the defined goal and scope of the study.

In principle, LCA is a comprehensive environmental assessment. In practice, not all types of environmental effects are equally well covered (Finnveden, 2000). Effects associated with land use are traditionally difficult to assess, although there has been a considerable methodological development during the last years (Lindeijer et al., 2003). Toxicological effects are often only included with data gaps. Effects associated with radiation, accidents and disamenities are typically not covered at all. The impacts typically best covered in a traditional LCA are environmental impacts from emissions to air, such as global warming and acidification, and use of energy resources.

Besides the mandatory elements within LCIA, there are also some elements described as optional (ISO, 1999), e.g. weighting which aims at converting and possibly aggregating indicator results across impact categories resulting in a single result. Methods for weighting include both economic valuation methods and multi-attribute approaches, further discussed below (Finnveden et al, 2003).

Site-adjusted LCA

LCA is traditionally a site- and time-independent tool. In a traditional LCA, no consideration is given to when and where emissions take place (Udo de Haes, 1996). Characterisation factors typically lack site-dependent information, resulting in a site-generic impact assessment. This is partly due to practical reasons; it is not practically possible to gather site-specific information for all sites included in an LCA. The reason is also partly theoretical. In LCA, not all emissions are considered, but only those that are allocated to the functional unit. There is however a trend towards making LCA more site-dependent if not site-specific (Huijbregts, 2001; Krewitt et al., 2001; Nigge, 2001a, b; Potting, 2000; Spadaro and Rabl, 1999). Introducing some typical environments and emissions situations does this. For example, emissions at low height in an urban environment are differentiated from emissions at high elevation in rural areas. Impact assessment factors are then calculated for these typical environments. Site-dependent characterisation factors can thus be calculated and used, resulting in a site-dependent characterisation. This has been done for some limited substances, which is described in detail in Chapter 5.2 “Site-adjusted LCA“.

3.3 Valuation

The valuation component demonstrates the trade-offs and weights of environmental values at stake as a result of the decision. The visualisation of environmental values can take different forms. Certain types of decisions use a monetary calculation, others use other types of decision criteria. Many SEA approaches suggest that an economic valuation approach (Bojö et al., 1992; ADB, 1996) is needed to evaluate the relative merits of the various decision alternatives. Through such a valuation we try to visualise the environmental costs and benefits of alternative decisions. This is an extensive research field, see e.g. (Markandya and Richardon, 1992; Viscusi, 1997), (Leksell, 1998) and (Commission for the European Communities, 1995). However, economic valuation is controversial and has been criticised both conceptually and in its applications. There are clearly decision situations where the economic single-dimensional valuation is inappropriate or lack acceptance among decision-makers. Like preferences on risk assessment, the preference for economic valuation is related to political culture and policy styles that differ between sectors and between countries (Peters, 2001). An alternative is to work with different types of multi-criteria analysis (MCA) methods (Peters, 2001). Using environmental objectives can offer an alternative valuation approach of multidimensional and qualitative nature (Finnveden et al., 2003). This probably resonates a lot better with the decision criteria in many institutional contexts than a MCA or a CBA.

Economic valuation in the SEA context addresses the divergence between social and private cost in the market as a result of a certain activity, i.e. a negative environmental externality (Bojö et al., 1992; Naturvårdsverket, 1997; Begg et al., 1987). There are different types of economic values and a first distinction can be made between use and non-use values. Use values are those values that are related to human production and consumption patterns and they can be direct, indirect or option values. Sometimes market prices exist for externalities relating to use values. For example, the effect of reduced crop productivity due to air pollution can be estimated by the market value of crops lost. For other such externalities no market prices exist, however, for example cough episodes and uneasiness as a result of urban air pollution (Commission for the European Communities, 1995). Non-use values are of a more philosophical kind and they can be either existence values or bequest values. Non-use values are usually only possible to estimate through artificial market methods. Valuation approaches can be grouped into those that use conventional markets, implicit markets or artificial markets (Commission for the European Communities, 1995). Conventional market valuations include analysis of changes in production, changes in earnings, replacement cost, and defensive expenditure. Implicit market valuations study the revealed preferences from actual consumer behaviour and choices. These include wage-risk approaches, travel cost approaches, land and property value or hedonic pricing approach. Artificial market valuations include measurements of consumer preferences in hypothetical situations, with Willingness-to-pay (WTP) or Willingness-to-accept (WTA) measures. These are sometimes referred to as direct methods (ADB, 1996).

4 About the proposed waste incineration tax

In 2000, a tax was introduced in Sweden on waste that is disposed of in landfills. Its purpose is to make landfilling less economically attractive, to the benefit of other waste treatment methods that are considered preferable from an environmental point of view. In 2001 a governmental commission was appointed to analyse and assess the economic and environmental impacts of introducing a similar tax on waste that is incinerated (SOU 2002:9). The commission report analyses the consequences of three different tax levels relative a no-action alternative. These alternative routes of action form the basis of this study.

4.1 Role in Swedish waste policy

Swedish waste strategies are developed in accordance with policies determined by the EU. At the core of the EU waste policy is the so-called waste hierarchy, according to which waste generation should in the first place be prevented. The hierarchy then states that waste should be treated in the following order of prioritisation: reuse, recycling (when environmentally motivated), energy recovery, and last landfilling. Another important principle in EU waste policy is that waste should be treated as close to its source as possible.

The proposed tax on waste incineration is one of several measures, which aim to influence waste management in direction of the waste hierarchy. Other important measures applied directly to reduce landfilling besides the landfill tax, are the ban on landfilling of sorted combustible waste from 2002, combined with an obligation to separate combustible and non-combustible waste, and the ban on landfilling of organic waste from 2005. To promote material recycling, a producer responsibility on packaging was introduced in 1994. It has been successively expanded to include more packaging materials, higher recovery targets, and other waste fractions than packaging (electronic waste, tyres).

4.2 Objectives

A waste incineration tax is not necessarily related only to waste policy. It may also be a tool in environmental, energy, or general tax policy. The commission mentions five conceivable reasons to introduce a waste incineration tax:

Harmonisation with energy and carbon dioxide taxation of fossil fuels.

Neutralise the advantages that waste incineration enjoys as a result of the current design of taxation of fossil fuels.

Improve the economic conditions for material recycling and biological treatment.

Reduce the current capacity shortfall for waste treatment, by increasing total capacity.

Eliminate the economic incentives for import of waste from other countries.

The background of the first two items is the design of taxation of fossil fuels in Sweden. Fossil fuels used for district heating are taxed in relation to the amount of distributed energy and the amount of CO_2 emitted (with certain exceptions made e.g. for industry and farming). Waste is incinerated mainly for heat production, but is considered a biofuel and is thereby exempt from both energy and CO_2 tax. Many waste fuels however contain a certain amount of fossil carbon. A waste incineration tax based on the fossil carbon content in waste could harmonise waste incineration with the taxation of fossil fuels. Because waste is exempt from both energy and CO_2 tax, the net revenues from producing heat from waste are higher than would otherwise be possible. A waste incineration tax would neutralise this relative advantage.

The third item views a waste incineration tax as a direct complement to the landfill tax, with the purpose to influence waste management in direction of the waste hierarchy. Material recycling and biological treatment are often more expensive and yield less revenue than waste incineration. A waste incineration tax would make these treatment methods more economically attractive by enabling them to charge higher reception fees. This may also stimulate the expansion of more expensive treatment methods, and thereby reduce the current shortfall of waste treatment capacity as a whole.

The last item relates to the principle of treating waste as near its source as possible. Currently, there are economic incentives for some European countries to export waste to Sweden. A tax on waste incineration could eliminate these incentives, thus reinforcing the subsidiarity principle in European waste management and ensuring that Swedish incineration capacity is used for Swedish waste.

4.3 Waste flows

The commission report contains data on waste flows in the no-action alternative and changes that are expected to occur as a result of the possible waste incineration tax at different tax levels (SOU, 2002), but data are very aggregated and not entirely comprehensive. For the purpose of the environmental analyses in this study a more detailed quantification is needed. A first step to reach this quantification is to identify how the case study should be delimited with respect to waste flows, so as to correspond to the considerations of the commission report. The next step is to make an inventory of waste amounts and composition. Last, the different alternatives are described in terms of anticipated degree of source separation and amounts of waste treated by different means.

Categorisation of waste flows

Sweden has adopted the EU definition of waste as being “every object, compound, or substance that is included in a waste category, that its possessor gets rid of, intends to or is obliged to get rid of” (Chapter 15, 1§ Swedish Environmental Code). The waste categories are defined in the European Waste Catalogue (EWC) (94/3/EC). However, this definition and the categories in the EWC have not yet penetrated the waste statistics, but available statistics are generally organised according to the older categorisation in the Swedish EPA regulations (SNFS 1991:3), which categorises the waste by source:

- household waste and similar waste
- park- and yard waste
- building and demolition waste
- waste from energy production
- waste from municipal waste water treatment
- waste from industrial waste water treatment
- waste from mining
- branch specific industrial waste
- non-branch specific industrial waste
- special waste

In principle, all waste categories and all waste incineration facilities should be covered by a waste incineration tax. In practice however, other priorities or practical obstacles may exclude certain waste streams and certain incineration facilities from the tax. The following delimitations were explicitly proposed in the commission report:

Wastes that can be regarded as biofuels: A large part of the branch specific waste generated in the pulp, paper, wood and graphical industries are also valuable as biofuels. Most of these fractions are incinerated by industry, not at regular waste treatment facilities. These waste fractions constitute the major part of wastes that are incinerated by industry. Since there is no aim to reduce incineration of waste that can be regarded as biofuels, the commission proposes that all waste incinerated by industry is excluded.

Hazardous waste: Thermal destruction is the most suitable treatment for many types of hazardous wastes. Because of very high temperature requirements, this is very expensive. In this case an incineration tax could have undesirable effects, and therefore hazardous wastes (categorised as special waste) should be excluded.

Biosolids: Biosolids from wastewater treatment are affected by the ban on landfilling of organic wastes. There are currently few alternative waste treatment options, and incineration may become a solution. Because of high moisture content, the cost of a weight based incineration tax on biosolids would be significant, especially on a per energy basis. The commission concludes that special rules would be necessary to include biosolids in the incineration tax. In the evaluation in the commission report, biosolids were not included at all.

Waste from energy production and mining: Not relevant in this case, because these waste types are under no circumstances treated by incineration. Indirectly however, the amount of waste from energy production is affected by how much waste is incinerated.

Using the categorisation of waste flows above, and taking into account the delimitations outlined in the commission report, the following waste categories were included in the SEA:

Household waste: Includes all waste that is generated by households, except hazardous waste, which is reported separately as special waste. It also includes any similar waste that is collected from trade, restaurants, schools etc. Rather pragmatically, this study divides household waste in three distinct subgroups - recyclables, rest waste, and bulky waste. Recyclables refers to packaging and paper that is covered by the Swedish producer responsibility, including amounts that are not source separated and collected for recycling today. Rest waste is household waste collected by “ordinary” waste collection, but excluding recyclables that may be co-mingled with this fraction. Bulky waste is a poorly defined fraction, the amount of which has been calculated as the difference between reported total amounts of household waste and the collected rest waste and recyclables.

Park and yard waste: In this study, this encompasses organic waste from parks and yards that is collected for centralised biological treatment. Some is also collected as household waste, and much is not collected at all, but these amounts are not included as park and yard waste.

Industrial waste: Industrial waste is sub-divided in branch specific and non-branch specific. Branch specific industrial waste is directly dependant on the production process (sometimes called production waste). Thus, its character is different for different types of industry. Branch specific industrial waste that can be regarded as biofuels is not included. This constitutes the majority of waste incinerated by industry. Non-branch specific industrial waste is not directly related to the production process (sometimes called consumption waste). It typically consists of packaging etc.

Building and demolition waste: Waste from all types of building, maintenance and demolition of buildings, infrastructure, and other constructions.

Quantification of current and future waste flows

All case study alternatives apply to estimated waste amounts in 2008. These estimates were reached by making inventories of current (1998 – 2000) waste amounts, which were then extrapolated to 2008 based on different growth rates for different waste categories. Inventories of current waste amounts and growth rates were as far as possible based on data presented in the commission report (SOU 2002:9), but data from other sources and some assumptions were necessary to complete the picture. References and assumptions on waste amounts, percentage composition of material fractions within each waste category, and current treatment are presented in detail in Appendix A.

Despite a seemingly systematic categorisation of waste categories, there is lack of comprehensive statistical data on waste flows. Overlaps, data gaps, and inconsistencies in terminology used by different data sources complicate the task of getting a comprehensive and quantified picture of the total flows of waste in Sweden.

Table 1 presents current (1998-2000) waste amounts and estimated amounts for 2008. The same percentage composition of material fractions within each waste category is assumed for both time periods. Different growth rates are used for different waste categories. Household waste is assumed to increase 2.2% per year, the reported growth rate for household waste in Sweden during 1992 – 2000 (Profu, 2001). Park and yard waste is assumed to increase 0.4% per year, which equals the population growth rate in Sweden. 2.2% increase per year is assumed for industrial waste, and 4% per year for building and demolition waste. This is based on Danish data, where industrial waste has been growing at the same rate as household waste, whereas building and demolition waste has been growing at 4.1% per year (Profu, 2001).

Table 1. Amount and composition of waste included in the study in 1998-2000 and 2008.

Waste fraction	Fraction (%)	1998-2000 (tonnes)	2008 (tonnes)
Household waste, total	100%	3 610 000	4 296 496
Household rest waste	24.0%	865 300	1 029 850
compost	17.5%	632 636	752 942
textile, rubber, leather	1.3%	47 056	56 004
mixed plastic	3.7%	133 324	158 678
mixed glass	0.7%	24 835	29 558
mixed metal	0.3%	10 457	12 445
combustibles	0.1%	3 921	4 667
non-combustibles	0.4%	13 071	15 557
Household packaging waste	46.9%	1 691 700	2 013 402
newspaper	15.8%	570 000	678 394
office paper	8.2%	295 000	351 099
corrugated cardboard	6.8%	246 000	292 781
cardboard	5.5%	198 000	235 653
PP	0.4%	14700	17 495
PS	0.3%	11760	13 996
PE	3.0%	109060	129 799
PVC	0.1%	4480	5 332
PET	0.2%	8 400	9 997
glass	4.6%	167 000	198 758

Waste fraction	Fraction (%)	1998-2000 (tonnes)	2008 (tonnes)
aluminium	0.2%	9 000	10 711
aluminium, refund	0.4%	15 000	17 852
steel	1.2%	43 300	51 534
Household bulky waste	29.2%	1 053 000	1 253 244
combustibles	14.6%	526 500	626 622
non-combustibles	14.6%	526 500	626 622
Park & yard waste, total	100%	203 000	241 603
Industrial waste, total	100%	1 767 000	2 103 022
paper & cardboard	32.2%	568 974	677 173
wood	27.6%	487 692	580 434
mixed plastic	9.5%	167 865	199 787
textile, rubber, leather	3.0%	53 010	63 091
glass	1.7%	30 039	35 751
mixed metals	11.0%	194 370	231 332
combustibles	3.3%	58 311	69 400
non-combustibles	11.7%	206 739	246 054
Building & demolition waste, total	100%	2 090 907	2 861 551
plaster board	6.1%	127673	174 729
mixed metals	6.2%	129766	177 594
concrete, tiles, brick	56.5%	1180452	1 615 530
asphalt	4.8%	100464	137 492
wood	21.3%	445809	610 120
paper	2.1%	43953	60 153
combustibles	3.0%	62790	85 932
ALL WASTE, TOTAL		7 670 907	9 502 671

Table 2 shows, at the most aggregate level, current treatment of the included waste categories as determined in Appendix A. These waste flows are used as a reference when estimating future waste treatment in the four case study alternatives described below. Composting of household waste includes both home composting and centralised large scale composting. Incineration includes both direct incineration and incineration of reject from material recycling. Material recycling only includes materials that have actually been recycled into new material, not reject material.

Table 2. Waste flows in 1998 – 2000 (tonnes/year) of waste categories included in this study.

Waste category	Amount	Material recycling	Compost	Anaerobic digestion	Landfill	Incineration
Household waste	3 610 000	1 051 601	149 916	46 992	853 254	1 508 237
Park & yard waste	203 000	0	150 000	0	53 000	0
Industrial waste	1 767 000	435 419	0	0	1 112 120	219 416
Building & demolition waste	2 090 907	442 042	0	0	1 375 258	273 607
TOTAL	7 670 907	1 929 062	299 916	46 992	3 393 632	2 001 260

4.4 Case study alternatives

Four alternatives were analysed in the SEA; the no-action alternative (Alt 0), one of the tax alternatives as described in the commission report (Alt 1), and two additional alternatives that explore the potential to achieve more far-reaching goals concerning optimisation of energy recovery (Alt 2a) and reduction of greenhouse gas emissions (Alt 2b). It is assumed that the different waste strategies represented by the case study alternatives have no influence on waste generation. Thus, the same amounts of waste with the same composition, corresponding to 2008, are treated in all alternatives. The alternatives only differ in the way waste is treated.

Alternative “No-action” (Alt 0)

The no-action alternatives described in the commission report and corresponds to waste management in Sweden in 2008 if no waste incineration tax is introduced, but all other means of control are introduced as planned. Waste treatment capacities have been estimated for 2008, based on current plans as described further below. The quantitative description of this alternative (Table 3) is as far as possible based on data presented in the commission report.

The commission report assumes that by 2009, the total waste incineration capacity will be 4.68 million tonnes per year, an increase of 109% compared to 1998 (2.25 million tonnes). As the majority, but not all combustible waste flows in Sweden are included in the SEA case study, we assume that the available capacity will be 109% of the incinerated amounts in Table 2. Thus, 4.18 million tonnes incineration capacity will be available in Alt 0.

Further, the commission report assumes that large scale biological treatment (composting and anaerobic digestion) will increase by 20 000 tonnes per year, from 392 000 tonnes in 2000 to 500 000 tonnes in 2008. This correspond to 27.6% increase. As for incineration, the biologically treated amounts in Table 2 are increased by this percentage, rather than assuming that all available capacity is used in the SEA case study. No increase of home composting is assumed.

We assume that rates of source separation for material recycling within all waste categories remain at the same level as today. That is, the percentage of source separation remains the same, leading to increasing amounts as waste amounts increase. This is a conservative assumption based on the fact that source separation rates have been increasing by 9% per year over the last years, but may be difficult to increase further as the most simple measures have already been taken (Profu, 2001).

Further, we assume that efficient sorting of combustible waste has been achieved at this point, and that the ban on landfilling of combustible waste is complied with. As a consequence, waste that is not source separated for recycling or biological treatment is perfectly separated in combustible and non-combustible fractions. 100% combustible waste is incinerated, while 100% non-combustible waste is landfilled. This is an ideal, but in reality not practically achievable situation.

Table 3. Waste flows in Alt 0.

Waste fraction	Total amount (tonnes)	Source sep. (%)	Material rec. (tonnes)	Incin. reject (tonnes)	Centr. comp. (tonnes)	Anaerobic dig. (tonnes)	Home comp. (tonnes)	Landfill (tonnes)	Incineration (tonnes)
Household rest waste, total	**1 029 850**		**0**	**0**	**102 106**	**59 967**	**69 908**	**57 560**	**740 309**
compost	752 942	30.8%	0	0	102 106	59 967	69 908	0	520 960
textile, rubber, leather	56 004	0.0%	0	0	0	0	0	0	56 004
mixed plastic	158 678	0.0%	0	0	0	0	0	0	158 678
mixed glass	29 558	0.0%	0	0	0	0	0	29 558	0
mixed metal	12 445	0.0%	0	0	0	0	0	12 445	0
combustibles	4 667	0.0%	0	0	0	0	0	0	4 667
non-combustibles	15 557	0.0%	0	0	0	0	0	15 557	0
Household PR waste, total	**2 013 402**		**1 251 579**	**65 933**	**0**	**0**	**0**	**58 282**	**637 607**
newspaper	678 394	79.65%	540335	0	0	0	0	0	138 059
office paper	351 099	37.00%	129907	0	0	0	0	0	221 192
corrugated cardboard	292 781	97.00%	247078	36920	0	0	0	0	8 783
cardboard	235 653	34.70%	80947	818	0	0	0	0	153 888
PP	17 495	31.93%	2625	2961	0	0	0	0	11 909
PS	13 996	31.93%	2100	2368	0	0	0	0	9 528
PE	129 799	31.93%	19478	21965	0	0	0	0	88 356
PVC	5 332	31.93%	800	902	0	0	0	0	3 630
PET	9 997	77.38%	7736	0	0	0	0	0	2 261
glass	198 758	86.11%	171146	0	0	0	0	27 612	0
aluminium	10 711	24.78%	2654	0	0	0	0	8 057	0
aluminium, refund	17 852	85.33%	15234	0	0	0	0	2 618	0
steel	51 534	61.20%	31539	0	0	0	0	19 995	0
Household bulky waste, total	**1 253 244**		**0**	**0**	**0**	**0**	**0**	**626 622**	**626 622**
combustibles	626 622	0.0%	0	0	0	0	0	0	626 622
non-combustibles	626 622	0.0%	0	0	0	0	0	626 622	0
Park & yard waste, total	**241 603**	**79.2%**	**0**	**0**	**191 398**	**0**	**0**	**0**	**0**
Industrial waste, total	**2 103 022**		**518 221**	**0**	**0**	**0**	**0**	**340 775**	**1 244 026**
paper & cardboard	677 173	41.7%	282069	0	0	0	0	0	395 104
wood	580 434	0.0%	0	0	0	0	0	0	580 434
mixed plastic	199 787	31.9%	63789	0	0	0	0	0	135 998
textile, rubber, leather	63 091	0.0%	0	0	0	0	0	0	63 091
glass	35 751	86.1%	30785	0	0	0	0	4 967	0
mixed metals	231 332	61.2%	141578	0	0	0	0	89 755	0
combustibles	69 400	0.0%	0	0	0	0	0	0	69 400
non-combustibles	246 054	0.0%	0	0	0	0	0	246 054	0
Building & demolition, total	**2 861 551**		**604 964**	**0**	**0**	**0**	**0**	**1 507 685**	**748 901**
plaster board	174 729	5.0%	8736	0	0	0	0	165 993	0
mixed metals	177 594	80.0%	142075	0	0	0	0	35 519	0
concrete, tiles, brick	1 615 530	20.0%	323106	0	0	0	0	1 292 424	0
asphalt	137 492	90.0%	123743	0	0	0	0	13 749	0
wood	610 120	0.0%	0	0	0	0	0	0	610 120
paper	60 153	5.0%	3008	0	0	0	0	0	57 145
combustibles	85 932	5.0%	4297	0	0	0	0	0	81 636
ALL WASTE, TOTAL	**9 502 671**		**2 374 764**	**65 933**	**293 504**	**59 967**	**69 908**	**2 590 924**	**3 997 465**

Alternative “Waste incineration tax SEK 400 per tonne” (Alt 1)

The commission report suggests a weight-based tax, motivated by administrative simplicity and the fact that many non-combustible waste fractions are heavy, thus giving a strong incentive to treat these by other means than incineration. The report analysed three levels of a weight-based incineration tax, which may be motivated for different reasons. At the lowest level, SEK 100 per tonne, the effects of energy taxation are neutralised. Setting the tax to SEK 400 per tonne is motivated by several reasons; it makes large scale composting economically competitive with incineration, the revenues from heat production from waste are about this high, it is the level of the waste incineration tax in Denmark, and it would eliminate the economic incentive to import waste from other European countries. The highest level, SEK 700 per tonne, was included mainly for the purpose of finding possible negative consequences of a high tax level. The commissioner presented SEK 400 per tonne as the most realistic and effective tax level, which was therefore chosen to be analysed in the SEA case study.

Estimates of treatment capacities in this alternative (Table 4) are made by the commission report, based on the assumption that the tax is introduced in 2004. Thus its effects should have made themselves felt by 2008. In parallel with introducing a waste incineration tax, the already existing landfill tax will be raised correspondingly. This will be a driver to reduce landfilling by increasing materials recycling and biological treatment.

The commission report assumes that source separation in households will increase by 70 000 tonnes compared to the no-action alternative (Alt 0). There are no estimates of how this increase may be distributed over different material fractions. We assume 6.8% increase for all recyclable material fractions in household waste, which gives an additional 70 000 tonnes of source separated materials. In lack of other data, we use the same assumption for recyclable materials in industrial waste. For building and demolition waste, the commission report assumes160 000 tonnes reduced landfilling. The following recycling rates were estimated as reasonable, while leading to the anticipated reduction of landfilled building and demolition waste:

<u>Plaster board:</u> We assume an increase from 5% today to 20% recycling. Recycled plaster board can be used in new plaster board, but also as for instance soil amendment or filling material (Hartlén et al., 1999). Little is recycled today because of unprofitability. There is large improvement potential in recycling of construction waste (Byggsektorns Kretsloppsråd, 2002).

<u>Metals:</u> We assume an increase from 80% to 85% recycling. While recycling rates are already, we assume that an increased landfill tax will lead to some further improvement.

<u>Concrete, tiles, and ceramics:</u> We assume an increase from 20% to 50% recycling. Concrete dominates this fraction. Crushed concrete can be used as filling material or ballast in road construction. While recycling is limited today because of unprofitability, better systems are being developed (Byggsektorns Kretsloppsråd, 2002). We assume that an increased landfill tax will lead to increased recycling.

<u>Other non-combustibles:</u> We assume no further increase above the current 90% recycling. This fraction is dominated by asphalt, with currently 90% recycling rates (Byggsektorns Kretsloppsråd, 2002).

<u>Wood:</u> We assume no material recycling of wood.

<u>Paper and Other combustibles:</u> We assume an increase from 5% to 10% recycling. This fraction mainly consists of paper, wood, and plastics. Source separation is required by the

producer responsibility, but hardly practised in the building industry (Byggsektorns Kretsloppsråd, 2002). We assume that an increased landfill tax will lead to a slight increase of materials recycling.

The commission report assumes 500 000 tonnes (90.9%) increase of total large scale biological treatment capacity in Alt 1 compared to Alt 0. This corresponds to realisation of existing expansion plans, and requires a large increase in source separation rate of organic waste. As explained for Alt 0, the percentage increase rather than the absolute increase is used to calculate the capacity in Alt 1. Home composting is not assumed to increase.

As a result of the waste incineration tax, the commission report assumes 140 000 tonnes (3%) reduction of total incineration capacity in Alt 1 compared to Alt 0. As explained for Alt 0, the percentage decrease rather than the absolute decrease is used to calculate the capacity in Alt 1.

Table 4. Waste flows in Alt 1.

Waste fraction	Total amount (tonnes)	Source sep. (%)	Material rec. (tonnes)	Incin. reject (tonnes)	Centr. comp. (tonnes)	Anaerobic dig. (tonnes)	Home comp. (tonnes)	Landfill (tonnes)	Incineration (tonnes)
Household rest waste, total	**1 029 850**		**0**	**0**	**194 895**	**114 462**	**69 899**	**57 560**	**593 034**
compost	752 942	50.4%	0	0	190 063	111 624	68 166	0	364 420
textile, rubber, leather	56 004	0.0%	0	0	0	0	0	0	56 004
mixed plastic	158 678	0.0%	0	0	0	0	0	0	158 678
mixed glass	29 558	0.0%	0	0	0	0	0	29 558	0
mixed metal	12 445	0.0%	0	0	0	0	0	12 445	0
combustibles	4 667	0.0%	0	0	0	0	0	0	4 667
non-combustibles	15 557	0.0%	0	0	0	0	0	15 557	0
Household PR waste, total	**2 013 402**		**1 319 885**	**67 906**	**0**	**0**	**0**	**43 283**	**582 327**
newspaper	678 394	85.1%	577078	0	0	0	0	0	101 316
office paper	351 099	39.5%	138740	0	0	0	0	0	212 359
corrugated cardboard	292 781	97.0%	247078	36920	0	0	0	0	8 783
cardboard	235 653	37.1%	86451	873	0	0	0	0	148 328
PP	17 495	34.1%	2804	3162	0	0	0	0	11 530
PS	13 996	34.1%	2243	2530	0	0	0	0	9 224
PE	129 799	34.1%	20803	23458	0	0	0	0	85 538
PVC	5 332	34.1%	855	964	0	0	0	0	3 514
PET	9 997	82.6%	8262	0	0	0	0	0	1 735
glass	198 758	92.0%	182784	0	0	0	0	15 974	0
aluminium	10 711	26.5%	2835	0	0	0	0	7 877	0
aluminium, refund	17 852	91.1%	16270	0	0	0	0	1 582	0
steel	51 534	65.4%	33684	0	0	0	0	17 850	0
Household bulky waste, total	**1 253 244**		**0**	**0**	**0**	**0**	**0**	**626 622**	**626 622**
combustibles	626 622	0.0%	0	0	0	0	0	0	626 622
non-combustibles	626 622	0.0%	0	0	0	0	0	626 622	0
Park & yard waste, total	**241 603**	**100.0%**			**241 603**			**0**	
Industrial waste, total	**2 103 022**		**553 459**	**0**	**0**	**0**	**0**	**329 054**	**1 220 508**
paper & cardboard	677 173	44.5%	301250	0	0	0	0	0	375 923
wood	580 434	0.0%	0	0	0	0	0	0	580 434
mixed plastic	199 787	34.1%	68127	0	0	0	0	0	131 660
textile, rubber, leather	63 091	0.0%	0	0	0	0	0	0	63 091
glass	35 751	92.0%	32878	0	0	0	0	2 873	0
mixed metals	231 332	65.4%	151205	0	0	0	0	80 128	0
combustibles	69 400	0.0%	0	0	0	0	0	0	69 400
non-combustibles	246 054	0.0%	0	0	0	0	0	246 054	0
Building & demolition, total	**2 861 551**		**1 123 280**	**0**	**0**	**0**	**0**	**1 347 089**	**741 597**
plaster board	174 729	10.0%	17 473	0	0	0	0	157 256	0
mixed metals	177 594	80.0%	142 075	0	0	0	0	35 519	0
concrete, tiles, brick	1 615 530	29.4%	474 966	0	0	0	0	1 140 564	0
asphalt	137 492	90.0%	123 743	0	0	0	0	13 749	0
wood	610 120	0.0%	0	0	0	0	0	0	610 120
paper	60 153	10.0%	6015	0	0	0	0	0	54 137
combustibles	85 932	10.0%	8593	0	0	0	0	0	77 339
ALL WASTE, TOTAL	**9 502 671**		**2 996 625**	**67 906**	**436 499**	**114 462**	**69 899**	**2 053 192**	**3 764 087**

Alternatives “Maximise energy recovery” (Alt 2a) and “Minimise greenhouse gas emissions” (Alt 2b)

Two case study alternatives explore the potential to achieve more far-reaching environmental goals than achieved by the tax alternative as described in the commission report. Alt 2a is designed to maximise energy recovery, and Alt 2b to minimise greenhouse gas emissions. The design of these two alternatives is based on experience from earlier environmental systems analyses of waste management systems.

The main difference between Alt 1 and these two alternatives is in increased source separations rates (90%), and as a consequence of that, reduced landfilling and incineration. 90% source separation may be practically achievable for some, but not all material fractions. As these are explorative scenarios, a high and equal source separation rate for all material fractions was however deemed reasonable.

The only difference between Alt 2a and Alt 2b is in the treatment of organic waste from households. In Alt 2a, which aims to maximise energy recovery, this fraction is incinerated (Table 5). Incineration has a very high efficiency of energy recovery. The recovered energy replaces virgin energy, in this case biofuels, with a 1:1 replacement ratio. In Alt 2b, which aims to minimise greenhouse gas emissions, this fraction is anaerobically digested (Table 6). Anaerobic digestion has a lower efficiency of energy recovery, but on the other hand the recovered gas can be used as vehicle fuel and thereby replace diesel. Assumptions concerning replacements are discussed in more detail in Chapter 5.

It is beyond the scope of this study to describe in detail how the alternatives 2a and 2b can be achieved. However, components of policy packages that would lead in this direction can easily be identified. The key policy instrument is probably a differentiated waste incineration tax.

One reason to introduce a waste incineration tax is to harmonise the taxation with energy and carbon dioxide taxation of fossil fuels (see section 4.2). Incineration of one ton of polyethelene (the most common plastic material in the studied waste fractions) produce 3.1 ton of CO_2. The CO_2- tax is 630 kr/ton (section 5.4). If the CO_2 tax and the waste incineration tax should harmonise, the waste incineration tax should be approximately 1950 kr/ton of plastic materials. This would make recyling much more economically attractive. The commission reports typical costs of incineration at 500 kr /ton and material recycling 1500 kr/ton. The difference (1000 kr/ton) is smaller than the CO_2-neutral incineration tax and an introduction of such a tax would thus lead to significantly increased recycling of plastic materials.

Introducing a differentiated waste incineration tax aiming at a tax which is harmonised with the carbon dioxide taxation of fossil fuels would thus be an effective policy instrument. This can be combined with a non-differentiated incineration tax to make sure that non- combustible materials are sorted out. Such a non-differentiated incineration tax can also make biological treatment methods compatible with incineration of organic wastes in line with scenario 2b. A combination of a differentiated and a non-differentiated waste incineration tax could thus effectively steer towards the scenarios outlined here.

Table 5. Waste flows in Alt 2a.

Waste fraction	Total amount (tonnes)	Source sep. (%)	Material rec. (tonnes)	Incin. reject (tonnes)	Centr. comp. (tonnes)	Anaerobic dig. (tonnes)	Home comp. (tonnes)	Landfill (tonnes)	Incineration (tonnes)
Household rest waste, total	**1 029 850**		**180 613**	**0**	**0**	**0**	**0**	**19 757**	**829 480**
Compost	752 942	0.0%	0	0	0	0	0	0	752 942
textile, rubber, leather	56 004	0.0%	0	0	0	0	0	0	56 004
mixed plastic	158 678	90.0%	142810	0	0	0	0	0	15 868
mixed glass	29 558	90.0%	26602	0	0	0	0	2 956	0
mixed metal	12 445	90.0%	11201	0	0	0	0	1 245	0
Combustibles	4 667	0.0%	0	0	0	0	0	0	4 667
non-combustibles	15 557	0.0%	0	0	0	0	0	15 557	0
Household PR waste, total	**2 013 402**		**1 714 037**	**118 520**	**0**	**0**	**0**	**27 886**	**152 960**
Newspaper	678 394	90.0%	610555	0	0	0	0	0	67 839
office paper	351 099	90.0%	315989	0	0	0	0	0	35 110
corrugated cardboard	292 781	97.0%	247078	36920	0	0	0	0	8 783
Cardboard	235 653	90.0%	209967	2121	0	0	0	0	23 565
PP	17 495	90.0%	7401	8345	0	0	0	0	1 750
PS	13 996	90.0%	5920	6676	0	0	0	0	1 400
PE	129 799	90.0%	54905	61914	0	0	0	0	12 980
PVC	5 332	90.0%	2255	2543	0	0	0	0	533
PET	9 997	90.0%	8998	0	0	0	0	0	1 000
Glass	198 758	90.0%	178882	0	0	0	0	19 876	0
Aluminium	10 711	90.0%	9640	0	0	0	0	1 071	0
aluminium, refund	17 852	90.0%	16067	0	0	0	0	1 785	0
Steel	51 534	90.0%	46381	0	0	0	0	5 153	0
Household bulky waste, total	**1 253 244**							**626 622**	**626 622**
Combustibles	626 622	0.0%	0	0	0	0	0	0	626 622
non-combustibles	626 622	0.0%	0	0	0	0	0	626 622	0
Park & yard waste, total	**241 603**	**0.0%**	**0**	**0**	**0**	**0**	**0**	**0**	**241 603**
Industrial waste, total	**2 103 022**		**1 029 639**	**0**	**0**	**0**	**0**	**272 762**	**800 620**
paper & cardboard	677 173	90.0%	609456	0	0	0	0	0	67 717
Wood	580 434	0.0%	0	0	0	0	0	0	580 434
mixed plastic	199 787	90.0%	179808	0	0	0	0	0	19 979
textile, rubber, leather	63 091	0.0%	0	0	0	0	0	0	63 091
Glass	35 751	90.0%	32176	0	0	0	0	3 575	0
mixed metals	231 332	90.0%	208199	0	0	0	0	23 133	0
Combustibles	69 400	0.0%	0	0	0	0	0	0	69 400
non-combustibles	246 054	0.0%	0	0	0	0	0	246 054	0
Building & demolition, total	**2 861 551**		**2 026 287**	**0**	**0**	**0**	**0**	**210 535**	**624 729**
plaster board	174 729	90.0%	157256	0	0	0	0	17 473	0
mixed metals	177 594	90.0%	159834	0	0	0	0	17 759	0
concrete, tiles, brick	1 615 530	90.0%	1453977	0	0	0	0	161 553	0
Asphalt	137 492	90.0%	123743	0	0	0	0	13 749	0
Wood	610 120	0.0%	0	0	0	0	0	0	610 120
Paper	60 153	90.0%	54137	0	0	0	0	0	6 015
Combustibles	85 932	90.0%	77339	0	0	0	0	0	8 593
ALL WASTE, TOTAL	**9 502 671**		**4 950 576**	**118 520**	**0**	**0**	**0**	**1 157 561**	**3 276 015**

Table 6. Waste flows in Alt 2b.

Waste fraction	Total amount (tonnes)	Source sep. (%)	Material rec. (tonnes)	Incin. reject (tonnes)	Centr. comp. (tonnes)	Anaerobic dig. (tonnes)	Home comp. (tonnes)	Landfill (tonnes)	Incineration (tonnes)
Household rest waste, total	**1 029 850**		**180 613**	**0**	**0**	**677 647**	**0**	**19 757**	**151 833**
compost	752 942	90.0%	0	0	0	677 647	0	0	75 294
textile, rubber, leather	56 004	0.0%	0	0	0	0	0	0	56 004
mixed plastic	158 678	90.0%	142810	0	0	0	0	0	15 868
mixed glass	29 558	90.0%	26602	0	0	0	0	2 956	0
mixed metal	12 445	90.0%	11201	0	0	0	0	1 245	0
combustibles	4 667	0.0%	0	0	0	0	0	0	4 667
non-combustibles	15 557	0.0%	0	0	0	0	0	15 557	0
Household PR waste, total	**2 013 402**		**1 714 037**	**118 520**	**0**	**0**	**0**	**27 886**	**152 960**
newspaper	678 394	90.0%	610555	0	0	0	0	0	67 839
office paper	351 099	90.0%	315989	0	0	0	0	0	35 110
corrugated cardboard	292 781	97.0%	247078	36920	0	0	0	0	8 783
cardboard	235 653	90.0%	209967	2121	0	0	0	0	23 565
PP	17 495	90.0%	7401	8345	0	0	0	0	1 750
PS	13 996	90.0%	5920	6676	0	0	0	0	1 400
PE	129 799	90.0%	54905	61914	0	0	0	0	12 980
PVC	5 332	90.0%	2255	2543	0	0	0	0	533
PET	9 997	90.0%	8998	0	0	0	0	0	1 000
glass	198 758	90.0%	178882	0	0	0	0	19 876	0
aluminium	10 711	90.0%	9640	0	0	0	0	1 071	0
aluminium, refund	17 852	90.0%	16067	0	0	0	0	1 785	0
steel	51 534	90.0%	46381	0	0	0	0	5 153	0
Household bulky waste, total	**1 253 244**		**0**	**0**	**0**	**0**	**0**	**626 622**	**626 622**
combustibles	626 622	0.0%	0	0	0	0	0	0	626 622
non-combustibles	626 622	0.0%	0	0	0	0	0	626 622	0
Park & yard waste, total	**241 603**	**0.0%**	**0**	**0**	**0**	**0**	**0**	**0**	**241 603**
Industrial waste, total	**2 103 022**		**1 029 639**	**0**	**0**	**0**	**0**	**272 762**	**800 620**
paper & cardboard	677 173	90.0%	609456	0	0	0	0	0	67 717
wood	580 434	0.0%	0	0	0	0	0	0	580 434
mixed plastic	199 787	90.0%	179808	0	0	0	0	0	19 979
textile, rubber, leather	63 091	0.0%	0	0	0	0	0	0	63 091
glass	35 751	90.0%	32176	0	0	0	0	3 575	0
mixed metals	231 332	90.0%	208199	0	0	0	0	23 133	0
combustibles	69 400	0.0%	0	0	0	0	0	0	69 400
non-combustibles	246 054	0.0%	0	0	0	0	0	246 054	0
Building & demolition, total	**2 861 551**		**2 026 287**	**0**	**0**	**0**	**0**	**210 535**	**624 729**
plaster board	174 729	90.0%	157256	0	0	0	0	17 473	0
mixed metals	177 594	90.0%	159834	0	0	0	0	17 759	0
concrete, tiles, brick	1 615 530	90.0%	1453977	0	0	0	0	161 553	0
asphalt	137 492	90.0%	123743	0	0	0	0	13 749	0
wood	610 120	0.0%	0	0	0	0	0	0	610 120
paper	60 153	90.0%	54137	0	0	0	0	0	6 015
combustibles	85 932	90.0%	77339	0	0	0	0	0	8 593
ALL WASTE, TOTAL	**9 502 671**		**4 950 576**	**118 520**	**0**	**677 647**	**0**	**1 157 561**	**2 598 367**

5 Environmental analysis methods

The environmental analysis methods that were applied in this case study were introduced in Chapter 0 “”. This chapter describes further how these methods were applied, and any necessary model development.

5.1 Qualitative assessment

The qualitative assessment is based on a checklist consisting of the Swedish national environmental objectives (Government Bill 2000/01:130). The environmental objectives are:

Reduced climate impact

Clean air

Natural acidification only

A non toxic environment

A protective ozone layer

A safe radiation environment

Zero eutrophication

Flourishing lakes and streams

Good-quality ground water

A balanced marine environment

Thriving wetlands

Healthy forests

A varied cultural landscape

A magnificent mountain landscape

A good built environment

A life cycle perspective is applied and the studied system is similar to that analysed in the traditional LCA (see section 5.2 for a description). This means that account is taken to avoided burdens and to upstream as well as downstream effects. The following assumptions are made:

Compost and digestion residues are used as fertiliser, replacing synthetic fertiliser.

Bio-gas produced in anaerobic digestion is used as bus fuel, replacing diesel fuel.

Heat recovered when waste is incinerated replaces heat production with forest residues as fuel.

Electricity is produced from hard coal.

The checklist is first filled in for each treatment option separately as well as for transports, heat production and electricity production. This is done in a group discussion. The resulting effects on each objective from the different alternatives are then compiled. The findings that are regarded as the most important for a decision on which strategy to choose, are then summarised.

A brief description of the Swedish environmental objectives as they are presented in the Governmental Bill (2000/01:130) is given below as a basis for how to fill in the checklist.

Reduced climate impact

The goal is to achieve a stabilisation of greenhouse gas concentration in the atmosphere at a level that would prevent dangerous anthropogenic interference with the climate system. The objective should be achieved in such a way and at such a pace as to ensure that biological diversity is preserved, food production is assured and other sustainable development objectives are not jeopardised. The objective can only be achieved in cooperation with other countries. Greenhouse gases include carbon dioxide, methane, nitrous oxide and fluoro-chloro carbons.

Clean air

Concentrations of air pollutants should not exceed low-risk concentrations for cancer or target values for protection against diseases or effects on plants, animals, materials and cultural objects. The target values are set with reference to persons who suffer from hypersensitivity or asthma. The objective mostly concerns urban outdoor air quality. Among substances affecting the objective are sulphur dioxide, nitrogen oxides, ground level ozone, non-methane volatile organic compounds, particles, polyaromatic hydrocarbons and other carcinogens.

Natural acidification only

The deposition of acidifying substances should not exceed the critical loads for water and land areas. Measures to prevent anthropogenic soil acidification and to preserve natural production capacity, archaeological objects and biological diversity should be taken. Forestry should be adapted to the sensitivity of each site to acidification, thus preventing the acidification of land and water due to land use. Important acidifying substances are sulphur dioxide and nitrogen oxides.

A non toxic environment

Humans and ecosystems shall not be exposed to substances and metals created or extracted by the society and that could threaten human health or the biological diversity. This objective concerns all environments, i.e. outdoor and indoor environment, including work environment.

A protective ozone layer

The use of ozone depleting substances should be phased out.

A safe radiation environment

Radiation doses should be limited as far as reasonably possible and the effects of UV-radiation should be limited as far as possible. Any risks associated with electromagnetic fields should be identified as far as possible and necessary measures should be taken as potential risks are identified.

Zero eutrophication

Nutrient inputs shall not cause adverse effects to human health and shall not be detrimental to biological diversity. The deposition of airborne compounds should not exceed the critical load for eutrophication of surface water anywhere in Sweden. Ground water should not contribute to eutrophication of surface water. The nutrient status of lakes and streams in forest and mountain areas should be the same as in nature. The nutrient status of lakes and streams in agricultural areas should not exceed natural concentrations, which means that the water may at most be nutrient-rich or moderately nutrient-rich. Nutrient concentrations in coastal waters and seas should essentially be the same as in the 1940s, and nutrient inputs into the sea should not cause eutrophication. Nutrient concentrations in forestland should be such as to promote preservation of the natural composition of species. The nutrient status of agricultural land should be such as to preserve biological diversity.

Flourishing lakes and streams

Nutrient inputs shall not have an adverse effect on biological diversity. Alien species and genetically modified organisms should not be introduced. The valuable natural and cultural assets shall be protected and maintained prudently and sustainably. There should be viable populations of fish and other aquatic species that are directly dependent on lakes and streams.

Good-quality groundwater

Groundwater quality shall not be adversely affected by human activities such as land use, extraction of natural gas, pollutant inputs etc. The quality of groundwater that leaks out should help to provide good habitats for plants and animals in lakes and streams. Consumption or other human impacts should not lower the groundwater level so as to jeopardise the supply and quality of the water. The level of anthropogenic pollution in groundwater should be so low that it its quality meets the requirements for good drinking water and good groundwater status under the EU Water Framework Directive (2000/60/EC).

A balanced marine environment and flourishing coastal areas and archipelagos

All Sweden's coastal waters shall have a good surface water status in terms of the composition of species and physical and chemical characteristics, as defined by the Water Framework Directive.

Thriving wetlands

There should be wetlands of various kinds all over the country with preserved biological diversity and cultural and historical assets. Endangered species should be able to spread to new habitats in their natural areas of distribution, thus ensuring viable populations. Alien species and genetically modified organisms should not be introduced. Peat extraction should only be carried out on sites that are suitable with regard to the natural and cultural environment and biological diversity. Wetlands should as far as possible be protected against drainage, peat extraction, road construction and other development operations. The recreational value of wetland is protected.

Healthy forests

The natural production capacity of the forestland should be preserved. The natural functions of and processes of forest ecosystems should be maintained. Natural regeneration should be practised wherever the land is suitable for this method. The forests natural hydrology should be protected. Importance should be attached to forests as sources of nature experiences and recreation should be taken into account. Endangered species and ecosystems should be protected. Endangered species should be able spread to near habitats in their natural areas of distribution, thus ensuring viable populations. Alien species and genetically modified organisms that may threaten the biological diversity shall not be introduced.

A varied cultural landscape

The nutrient status of arable land should be well-balanced, with a good soil structure and humus content, and pollution levels should be so low as not to affect the functioning of ecosystems and human health. Agricultural land should be cultivated in such a way as to minimise adverse environmental impacts and favour biological diversity. The land should be cultivated in such a way as to maintain its long-term productive capacity. The agricultural landscape should be open and varied, with plenty of small habitats and water environments.

A magnificent mountain landscape

The majestic mountain scenery with its pastures and continuous open spaces should be intact and the biological diversity in mountainous areas should be preserved. Measures should be taken to maintain low noise levels.

A good built environment

The built environment shall provide aesthetic experiences and wellbeing and offer a wide rage of housing, workplaces, services and culture that give everybody the opportunity to live a full and stimulating life, while reducing everyday transport needs. A sustainable urban structure should be developed both in connection with the location of new buildings, structures and industries and with the use management and conversion of existing buildings. The living and leisure environment, and wherever possible the work environment, should meet society's requirements in terms of design, freedom of noise and access to sunlight, clean water and clean air. Areas of unspoiled nature and green spaces close to built up areas, which are easily accessible, should be protected in order to meet the need of play, recreation, local farming and healthy local climate. Biological diversity should be preserved and enhanced. Transports and transport facilities should be located and designed in such a way as to limit interference with urban or natural environment and so as not to pose health or security risks or be otherwise detrimental to the environment. People should not be exposed to harmful air pollutants, noise nuisance, harmful radon levels or other unacceptable risks to health and safety. Land and water should be free from toxic and dangerous substances and other pollutants. The use of energy, water and other natural resources should be efficient, resource saving and environmentally sound. The preferred energy sources are renewable. Natural gas should only be used where it is not possible to use substitutes in specific applications. Deposits of gravel that are valuable for the drinking water supply and the natural and cultural landscape should be preserved. By 2010 the proportion of reused material should represent at least 15% of the ballast used. The quantity and dangerousness of waste should be decreasing. Waste and residues should be separated by categories and recycled on a co-operative basis by urban areas and the surrounding rural areas. The quantity of landfilled waste, excluding mining waste generated should be reduced at least 50% by 2005 compared to 1994, at the same time as the total quantity of waste generated does not increase. All landfill sites should conform to uniform standards by 2008 and should meet stringent environmental requirements in accordance with Council Directive 1999/31/EC on the landfill of waste.

5.1 LCA

This chapter describes the LCA model that was developed for the purpose of this case study, and briefly discusses certain methodological aspects of LCA that come into focus when modelling waste management systems. General reference is made to Finnveden et al (2000), in which methodological aspect of LCA specific to the modelling of waste management systems are more thoroughly discussed. The LCA model described in this chapter was also used for calculations in the site-adjusted analysis and in the valuation.

General description of the model

The LCA model represents national Swedish waste management. The functional unit is defined as the collection and treatment in 2008 of all waste fractions that are covered by the commission report about the waste incineration tax (SOU 2002:9) (c.f. Chapter 4). As the study is change-oriented, marginal data should ideally be applied to reflect changes. In practice, marginal data is only applied to model electricity use and avoided burdens of district heating, see below.

Figure 2 illustrates the system boundaries of the LCA model. Waste management, including collection and transports, treatment, and final disposal constitutes only part of the total system. The total system is expanded with a complementary system that includes processes for conventional production of resources that can be recovered from waste (explained further below). Raw materials and energy used by waste management and the complementary system are also included. The impact of waste sources is not included, only the waste as such.

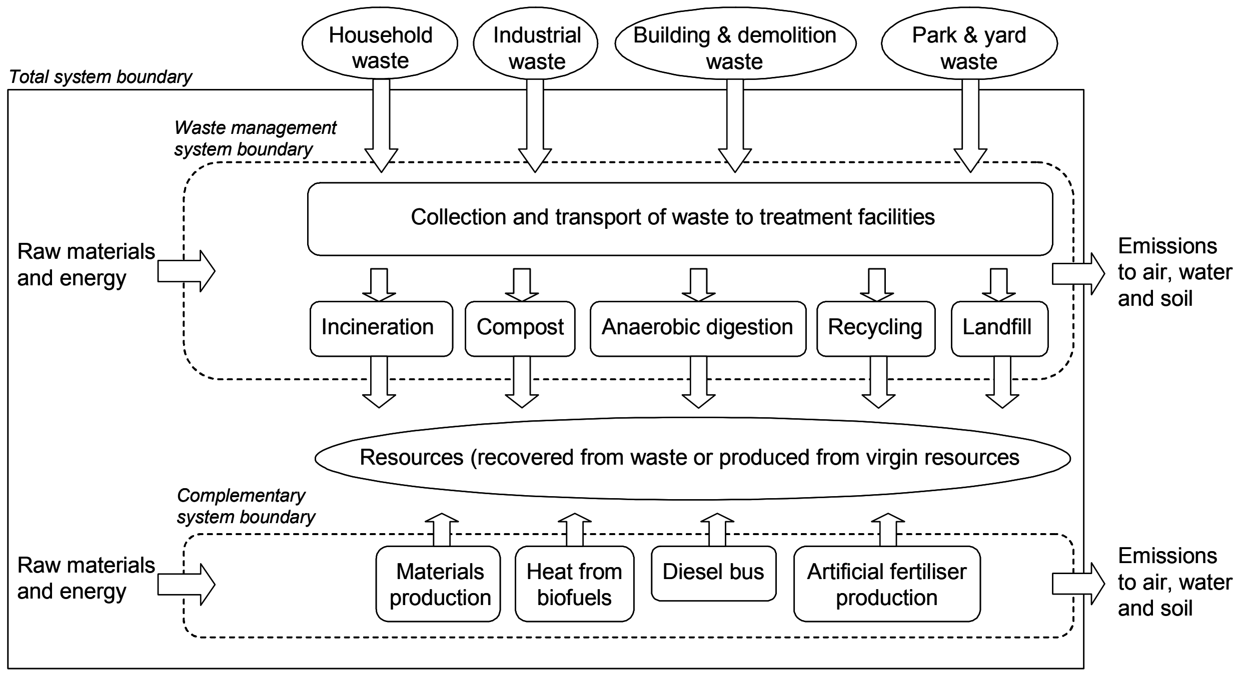

Figure 1. Simplified representation of the LCA model developed for this case study.

The model was implemented in the LCA software tool SimaPro (PRé Consultants 2001). It is in many parts based on an earlier model described in Finnveden et al. (2000) and process data available in the SimaPro database, but some new model development has taken place. Model data are briefly described below, and comprehensively documented in Appendix C (collection and transports) and Appendix D (full process data records).

Collection. The models for collection of recyclables and collection of rest waste are based on Swedish field data collected by Stenberg et al. (1999), complemented with some data on recyclables collection from Bäckman et al. (2001). A short, medium, or long collection route can be selected. The medium transport alternative is used in the traditional LCA model calculations, whereas the short and long transport alternatives are used in the site-adjusted LCA model calculations. No distinction is made between collection from different sources.

The same rest waste collection model is applied for all waste treated by incineration, landfill, or centralised biological treatment. Emissions and fuel consumption depend on collected weight and average distance travelled, with different average distances applied for the different treatment options. Emissions and fuel consumption depend on collected weight and average distance travelled, with different average distances applied for different material fractions. The same recyclables collection model is used for all waste fractions that are recycled, which probably over estimates the impacts from collection of high density materials.

Anaerobic digestion. Emission factors of the anaerobic digestion plant itself were derived from the Orware model (Dalemo et al. 1999; Sundqvist et al. 1999). Spreading of digester residue on agricultural land was modelled according to Nilsson (1997) as cited in Finnveden et al. (2000). Digester residue is used as organic fertiliser and is assumed to replace nitrogen and phosphorus in artificial fertiliser with a 1:1 replacement ratio. The avoided burdens of artificial fertiliser production were modelled using data from Weidema (1995). Biogas is used as bus fuel. It is assumed to replace diesel fuel with a 1:1 replacement ratio. The avoided burdens of diesel fuel production and use were modelled using data from Egebäck et al. (1999) as cited in Sundqvist et al. (1999). The anaerobic digestion model also includes transports of collected organic waste to the treatment plant, as described in Appendix C.

Composting. The model allows choosing between home composting or centralised windrow composting of household organic waste, or centralised windrow composting of park and yard waste. Home composting requires no transports. Emission factors of the different compost plants were derived from the Orware model (Sonesson 1997; Sundqvist et al. 1999). Spreading of compost on agricultural land was modelled according to Nilsson (1997) as cited in Finnveden et al. (2000). Compost is used as organic fertiliser, and is assumed to replace nitrogen and phosphorus in artificial fertiliser with a 1:1 replacement ratio. The avoided burdens of artificial fertiliser production were modelled using data from Weidema (1995). The compost model also includes transports of collected organic waste to the treatment plant, as described in Appendix C.

Recycling. The modelled recycling processes and main references used when developing each process model are listed in Table 7. Many recycled materials are assumed to replace an equivalent amount of virgin material. Some materials replace another type of material, and in some cases the replacement ratio recycled: virgin material is less than 1:1.

Table 7. Modelled recycling options and main references used when developing each process model.

Recycling process	Avoided material, replacement ratio (recycled: virgin)	Main reference of process model
Aluminum	Aluminium ingot (1:1)	EAA (1996) as included in IVAM2 database in SimaPro
Steel	ECCS steel sheet (1:1)	BUWAL 250 database as included in SimaPro
Cardboard	Cardboard (1:0.78)	Sundqvist et al. (1999) as described in Finnveden (2000)
Corrugated cardboard	Testliner, 52.2% (1: 0.82) Wellenstoff, 34.8% (1: 0.82)	FEFCO (1997) as described in Finnveden et al. (2000)
Glass	Glass, 65% (1:1) Mineral wool, 35% (1: 0.88)	BUWAL 250 database as included in SimaPro
Newspaper	Newspaper (1: 0.96)	Reforsk (1993) as described in Finnveden (2000)
Office paper	Paper towels (1:1)	Franklin USA (1998) database as included in SimaPro
PE	HDPE (1:1)	Sundqvist et al. (1999) as described in Finnveden (2000)
PET	PET (1:0.98)	Danish EPA (1998) as described in Finnveden (2000)
PP	PP (1:1)	CE (1994) as included Pre4 database in SimaPro
PS	PS (1:1)	CE (1994) as included Pre4 database in SimaPro
PVC	PVC (1:1)	CE (1994) as included Pre4 database in SimaPro
Plaster board	Fluewallboard (1:1)	Ullman (1991) as included in IVAM 2 database in SimaPro
Concrete	Gravel (1:1)	no process burdens modelled
Asphalt	Asphalt (1:1)	no process burdens modelled

The recycling models also includes transport of collected recyclables to a reloading point for long distance transport, further transport to the recycling facility, and in some cases transport of reject, as described in Appendix C.

Incineration. The model includes specific models for incineration of cardboard, corrugated cardboard, organic waste, mixed paper, mixed plastics, newspaper, office paper, combustibles, park and yard waste, PE, PET, PP, PS, PVC, textile and rubber, and wood. Emission factors for the incineration facility were derived using the Orware model (Björklund 1998; Sundqvist et al. 1999). The incineration models include transport of collected waste to the treatment facility, the incineration plant itself, transport of incineration ashes to landfill, and the ash landfill. It also includes the avoided burdens of heat generation from biofuels that is replaced by recovered energy from waste. The incineration model also includes transports of collected rest waste to the treatment plant, as described in Appendix C.

Landfill. The model allows to choose between landfilling of different inorganic materials; aluminium, glass, mixed metals, non-combustibles, steel, plaster board, asphalt, and concrete. Emission factors for the landfill were derived using the Orware model (Björklund 1998; Sundqvist et al. 1999). Due to the ban on landfilling of organic waste, landfilling of organic waste fractions was not modelled. Assuming that no organic waste will be landfilled lead to methodological problems, because existing data that were model landfill emission is representative of mixed waste landfills that are in use today, containing both organic and inorganic material. The microbial and chemical processes as well as the design of an inorganic waste landfill may be quite different, leading to other emissions. The landfill model also includes transports of collected rest waste to the treatment plant, as described in Appendix C.

Open-loop recycling

Open-loop recycling takes place when a product is recycled into another product, so that the system provides not one, but two or more functions. This causes problems, because impacts need to be allocated between the two functions - the original and the recycled product. Different means to handle this allocation problem have been proposed. The one most typically used in waste management LCA, which is also recommended by the ISO standards (ISO, 1998), is expanded system boundaries. With this approach, the burdens of virgin products that are avoided thanks to the recycled product are taken into account. This may be done by subtraction from the system that produces the recycled product. In a comparison of alternatives, adding the corresponding burdens to the system that does not produce the recycled product may also do it. This model handles the open-loop allocation problem by subtraction of avoided burdens.

When heat is recovered from incineration, we assume that the avoided burdens are those of district heating generated by combustion of biofuels (residue from timber felling). This most likely corresponds to the long-term base load marginal of heat production in Sweden (Finnveden et al. 2000; Sahlin, 2003). We do not assume any alternative use of saved biofuels, i.e. saved biofuels remain in the forest.

In most cases of materials recycling (PVC, PP, PS, PE, steel, Al, plaster board, asphalt) we assume that the avoided burdens are those of producing the corresponding virgin material, with a replacement ratio of 1:1. Exceptions are PET, corrugated cardboard, cardboard, and newspaper that replace virgin material at a slightly lower ratio, office paper that is recycled into paper tissue, concrete that replaces gravel, and glass that replaces glass (65%) and mineral wool (35%).

Biological waste treatment produces a residue that can be used as fertiliser or soil amendment. We assume that the avoided burdens are those of producing artificial nitrogen and phosphorous fertiliser, with a 1:1 replacement ratio. We further assume that there is no difference in leaching of nutrients from organic or artificial fertiliser, and thus no difference in contribution to eutrophication.

Anaerobic digestion produces a combustible gas that can be used as a fuel. We assume that the avoided burdens are those of producing and using diesel in buses with a 1:1 replacement ratio (per MJ).

Electricity

Electricity is used in several processes throughout the system. As the polluting potential of electricity is significant, the assumption of type of electricity generation is very important. When possible, we used electricity generated from hard coal, as argued for in Finnveden et al. (2000). It was not always possible to use this electricity source when process models from the SimaPro database were used. When electricity generation is included as an integral part of the total process, it can not be replaced by other type of electricity generation.

Time perspective for landfills and ashes

Emissions from landfills prevail for very long time, which is problematic when landfills are compared to processes with more immediate emissions. To include all potential emissions, one must in principle integrate landfill emissions over infinite time. This approach is in accordance with the LCA definition, but may be problematic both because future emissions are largely unknown, and because this perspective may simply be difficult to accept. It is therefore common in LCA of waste management to make a temporal cut-off at some much earlier point in time. With a temporal cut-off, a large part of the potential emissions from landfills are disregarded. The choice of time perspective for landfills is clearly a value choice (Finnveden 2000; Hellweg 2003).

To handle the temporal problem of landfill, we use the definitions surveyable time period (ST), which is around 100 years, and the remaining time period (RT), which is a hypothetical infinite time (Finnveden et al. 1995). During ST the emissions are relatively well known. When modelling RT we assume complete dispersion of landfilled material, corresponding to a worst case scenario. Both time perspectives are used in the model. ST is applied as a base case, and the emissions from RT are included in a sensitivity analysis.

A problem that is related to the time aspects of landfills is the fate of biotic carbon in landfills. It is common practice to disregard biotic CO_2 emissions in LCA, based on the cyclic nature of carbon in biomass and assumptions about sustainable forestry. In case a temporal cut-off is introduced, biotic carbon however remains trapped in the landfill, which thus acts as a carbon sink. To capture this aspect in the model, negative impact should be assigned to all biotic carbon that is trapped in the landfill after the cut-off. However, as this model includes no landfilling of organic waste, the carbon aspect of landfills is not an issue.

Closely related to the time-dependent emissions from landfills, is the handling of ashes from heat production from forest residue, which was used in the model of avoided heat production. We assume that the ashes are brought back and spread in the forest to retrieve nutrients and trace elements. This way, the ashes become a source of metal emissions. However, the metals are only assumed to have an impact after the surveyable time period, thus they are comparable to RT emissions from landfills.

Life cycle impact assessment

The life cycle model of waste management results in inventories of resource use and emissions from the different processes of the system. These inventories constitute the input to the life cycle impact assessment (LCIA), in which resource use and emissions are expressed in terms of potential environmental impact categories. The mandatory elements of LCIA are (ISO 1999):

Selection of impact categories to include in the study.

Classification, assigning inventory data to the different impact categories.

Characterisation, quantification of the contributions to the different impact categories.

The last step is done by means of certain characterisation methods. A number of such methods are available. In this study we use the CML 2000 baseline characterisation method (Guinée, 2002) as included in the SimaPro software, which includes the following impact categories:

global warming

ozone layer depletion

human toxicity

fresh water aquatic ecotoxicity

marine water aquatic ecotoxicity

terrestrial ecotoxicity

human toxicity

photochemical oxidation

acidification

eutrophication

abiotic depletion (not used in this study)

Because the characterisation method does not include all possible synonyms of substances, we have complemented the method with such synonyms, so as to make the method as far as possible compatible with the process sub models in SimaPro. Thus, while we have made an effort not to alter the actual method, the version of CML 2000 that we use is not identical to the original method.

We also use two additional impact categories that measure exergies of natural resource use (Finnveden and Östlund 1997):

exergy of abiotic resource use

exergy of biotic resource use

In addition, total energy use is presented as well as total use of renewable and non-renewable energy.

5.2 Site-adjusted LCA

A site-adjusted LCA model was developed for emissions of SO_2, NO_x and PM. Site-adjusted in this context means a model with some, but not complete, site-dependent information on emission sources and characterisation factors. The site-adjusted model was used to compare the same waste management strategy in Skåne in the south of Sweden, with high population density and short transport distances, versus Norrland in the north of Sweden, with low population density and long transport distances.

The site-adjusted model consisted of the traditional LCA model with minor adjustments, and specifically derived site-dependent characterisation factors.

Site-adjustment of LCA model

The site-adjusted LCA model was based on the traditional LCA model, complemented with some geographical information on transport distances and emission sources to represent Skåne and Norrland. A summary of these adjustments is presented below.

Transport distances for all waste related transports (collection, transport to treatment, transport of reject etc.) were modelled as being either long (in Norrland) or short (in Skåne). The total waste flow in each alternative is modelled as being treated in either Skåne or Norrland, rather than distributed over the country depending on population density and industrial activities. While this is an unrealistic representation of waste flows in Sweden, it serves the purpose of methodological simplicity. As a consequence, the total impact of each region is not relevant. Still, the impact per amount of treated waste, and the relative difference between the two regions allows us to evaluate the impact of the same waste management strategy in a regional context.

All process models were defined as being situated either locally in the analysed region (Skåne or Norrland), or at an unknown location. The potential environmental impact of local processes was calculated with characterisation factors developed for that specific region (see below). The impact of processes with unknown location was calculated using average characterisation factors. Local process models were defined as being all waste treatment processes except recycling processes, waste transports, and the avoided use of diesel when biogas vehicles use biogas from anaerobic digestion, as well as the avoided use of biofuels for district heating when heat is recovered from waste incineration. All other processes were defined as having unknown location.

Further, every process model was defined as being either a ground level or high level emission source, based on Potting (2000). All energy conversion facilities, both waste incineration and other fuels, were defined as being high level. All other processes were defined as being ground level.

Development of site-dependent characterisation factors

Site-dependent characterisation factors were developed for emissions of SO_2, NO_x and PM by using an integrated impact assessment model, EcoSense 2.0 (IER, 2000). This is an impact pathway model that follows the sequence from emissions of pollutants, through to impacts on receptors, as depicted in Figure 2.

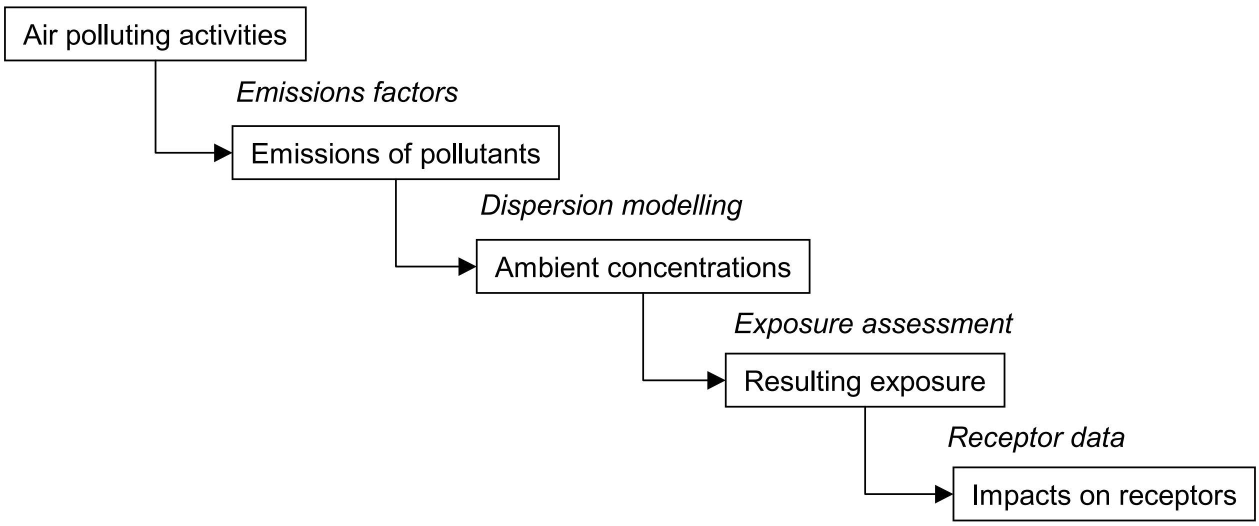

Figure 2. Impact pathway model in Ecosense, as illustrated in Finnveden et al. (2003).

The characterisation factors were derived by calculating the environmental impact caused by emissions from a well-defined emission source. By using impact models with linear dependence on amount of emitted substance, the resulting impact could be divided by amount of emission to derive the characterisation factors.

Model runs were done in which a hypothetical electricity production plant was located at four different locations in Sweden.

In Skåne in the south, close to continental Europe and with a relatively high population density in the surrounding region.

In Västergötland, a bit further north-west, with relatively low population density.

In Stockholm, further north-east but with a relatively high population density.

In Storuman in the north, ca 400 km north of Östersund, with very low population density.

The dispersion of emissions may vary depending on the height of the emission source. To derive characterisation factors for ground level and high level emission sources, model runs were made for each site with low and high stack heights. The characteristics of the model plant are listed in Table 8.

Table 8. Characteristics of incineration facility in Ecosense model runs.

Parameter	Value
Capacity	138.9 [MW]
Electricity sent out	138.9 [MW]
Full load hours per year	4413 [h]
SO2 Emissions	50.0 [mg/Nm3]
NOx Emissions	50.0 [mg/Nm3]
TSP Emissions	50.0 [mg/Nm3]
Stack height	165.0 [m] / 45 [m]
Stack diameter	3.8 [m]
Flue gas volume stream	570000.0 [Nm3/h]
Flue gas temperature	353.0 [K]
Surface elevation at site	0.0 [m]
Anemometer height	10.0 [m]

The EcoSense model contains two atmospheric emission dispersion models, the local and the regional level. Ecosystem impacts are calculated only at the regional level, whereas human health impacts are calculated at both the regional and the local level.

At the local level (corresponding to an area of 100*100 km around the emission source), the Industrial Source Complex Model (ISC) is used to calculate dispersion of SO_2, NO_x and PM.

At the regional level (covering Europe), the Windrose Trajectory Model (WTM) is used to calculate dispersion of SO_2, NO_x and PM, and the atmospheric transformation into nitrate and sulphate particles as a result of reaction with regional background concentrations of NH_3.

The resulting concentrations are linked to exposure response functions on human health impacts and ecosystem impacts. Table 9 lists the exposure functions that were selected in Ecosense to derive the site-dependent characterisation factors.

Table 9. Exposure functions selected to derive site-dependent characterisation factors.

Receptor	Impact	Pollutant	Reference function
total population	'acute' YOLL	so2	Anderson/Touloumi (1996)
adults	'chronic' YOLL	tsp	Pope(PM10)
adults	'chronic' YOLL	Nit	Pope(Nit.)
adults	'chronic' YOLL	Sul	Pope(Sul.)
total ecosystem	RDW N ecosystem area	nde	UN-ECE, 1995 (TOT)
total ecosystem	RCW SO2 ecosystem area	so2	UN-ECE, 1993
total	RCW NOx ecosystem area	nox	UN-ECE, 1993

Recent advances in health impact methodologies relate mortality impacts to years of life lost (‘YOLL’) rather than the increased probability of death. YOLL refers to the expected number of life years lost as a result of the increased pollution. This is due to the consideration that both chronic and acute effects causing premature deaths do not, on average, occur on ‘prime age’ individuals, but rather affecting persons with lower remaining life expectancy.

Ecosystem impacts are based on the critical loads concept as developed in UNECE convention. The indicator used is the change in actual exposure in an ecosystem area divided by the critical load or level of exposure in this area. This indicator is chosen instead of change in exceedence area because it allows us to account for the pollution that affects areas that are already above critical loads, which was necessary to derive characterisation factors that are linearly related to emissions.

Human health impacts are based on risk assessment approach and the functions used are the ones recommended by the ExternE project. Following the recommendations, we assume a linear relationship between a change in mortality and concentration of pollutants.

The regional analysis shows no great differences in ecosystems impacts (Table 10). Västergötland and Norrland display 20-30% greater impacts, which is probably attributable to being inland emissions rather than coastal emissions.

Table 10. Site-dependent characterisation factors for ecosystem impact of NOx and SOx emissions.

Emis-sion	Unit	Västerg (high)	Västerg (low)	Skåne (high)	Skåne (low)	Stockh. (high)	Stockh. (low)	Norrl. (high)	Norrl. (low)
N (RDW)	km^2/TWh	1.6E+01	1.6E+01	1.4E+01	1.4E+01	1.3E+01	1.3E+01	2.0E+01	2.0E+01
	km^2/kg	7.7E-05	7.7E-05	6.7E-05	6.7E-05	6.4E-05	6.4E-05	9.7E-05	9.7E-05
SOx (RCW)	km^2/TWh	1.3E+01	1.2E+01	9.7E+00	9.1E+00	8.6E+00	8.1E+00	1.5E+01	1.4E+01
	km^2/kg	6.2E-05	5.8E-05	4.7E-05	4.4E-05	4.2E-05	3.9E-05	7.3E-05	6.8E-05
NOx (RCW)	km^2/TWh	6.2E+00	6.2E+00	4.3E+00	4.3E+00	3.2E+00	3.2E+00	6.5E+00	6.5E+00
	km^2/kg	3.0E-05	3.0E-05	2.1E-05	2.1E-05	1.6E-05	1.6E-05	3.2E-05	3.2E-05

Relative deposition weighted (RDW) area; relative deposition = nitrogen deposition divided by critical load of nutrient nitrogen; calculation: ecosystem area times relative deposition

Relative concentration weighted (RCW) area; relative concentration = SO2 concentration divided by critical level of SO2; calculation: ecosystem area times relative concentration. (Source: EcoSense model)

Results for human health impacts are presented in Table 4. These are aggregated results for both local ad regional impacts. The difference in impacts between different regions in Sweden is approximately one order of magnitude. The difference between low and high stacks is approximately a factor of2 for densely populated areas and zero for sparsely populated areas.

For local impacts the differences are larger. The analysis shows that the height of emissions can play a significant role for health impacts at the local level, particularly in densely populated areas. The local impacts of high level emissions in densely populated regions are about 10-20% of impacts of ground level emissions (data not shown). In sparsely populated regions the corresponding figure is 70-80%. At the regional level, the stack height does not matter. In total, the effect is about 50% in high population density regions and 10% in low-density population regions.

Table 11. Site-dependent characterisation factors for human health impact of NOx, SOx, and particulate emissions.

Emission	Unit	Västerg (high)	Västerg (low)	Skåne (high)	Skåne (low)	Stockh. (high)	Stockh. (low)	Norrl. (high)	Norrl. (low)
SOx	YOLL/kg	1.3E-05	1.2E-05	2.0E-05	1.9E-05	8.7E-06	9.1E-06	3.5E-06	3.3E-06
	mECU/kg	5.1E-05	4.8E-05	7.7E-05	7.8E-05	3.4E-05	3.8E-05	1.3E-05	1.3E-05
part	YOLL/kg	1.7E-05	1.8E-05	2.9E-05	5.1E-05	1.3E-05	3.8E-05	3.0E-06	3.0E-06
	mECU/kg	6.6E-05	7.0E-05	1.1E-04	1.9E-04	5.0E-05	1.4E-04	1.1E-05	1.1E-05
NOx	YOLL/kg	1.1E-05	1.1E-05	1.7E-05	1.7E-05	6.7E-06	6.7E-06	2.5E-06	2.5E-06
	mECU/kg	4.1E-05	4.1E-05	6.3E-05	6.4E-05	2.5E-05	2.6E-05	9.4E-06	9.4E-06

Comparing low and high population density, the local analysis further reveals that in low population density regions areas, the impacts are 10 or 100 times lower than in high population density regions, depending on low or high emissions height (data not shown). In total, the effects are 10 times lower and 15 times lower respectively.

Comparing the regional location, there is a clear tendency for lower impacts further north, close to a factor 10 between Skåne and Norrland, and Stockholm and Västergötland in between.

5.3 Valuation methods

Valuation as a further step within Life Cycle Impact Assessment is explained in Guinée et al (2002) as an optional step of Life Cycle Impact Assessment, in which the indicator results for each impact categories assessed are assigned numerical factors according to their relative importance, multiplied by these factors and possible aggregated. Valuation can be based on different choices; it could for example be monetary values, standards or expert panel. Since there is no best method available a set of weighting methods are preferable. This does not disqualify the fact that some methods are more flexible and capable than others. Aspects that could be of significance when choosing valuation methods include number of included impact categories and substances, choice of characterisation method and choice of valuation base.

In some literature and particular older ones, it is common to use the expression valuation method describing weighting methods in Life Cycle Assessment (LCA). However the term ”valuation method” is used in Strategic Environmental Assessment (SEA) and valuation and weighting will be used as a synonym in this document.

In this study we are using three weighting methods: Ecotax02, Eco-indicator 99 and EPS 2000, all described below. Data for the latter two methods are taken from the used software programme SimaPro version 5.0.9.

Ecotax 02

The monetary weighting method Ecotax 98 developed by Johansson (1999) is based on environmental taxes and fees in Sweden 1998. The method links a tax or a fee to a relevant impact category. This means that when the tax or fee is expressed for one substance, characterisation factor conversion make it possible to get a reference equivalent weight. Ecotax 02 is an updated version of both characterisation methods and tax bases and it is described in detail in Eldh (2003). The tax base is updated until November 2002. Weighting factors used in Ecotax 02 are listed in Table 12.

Table 12. Weighting factors for Ecotax derived from environmental taxes and fees in Sweden 2002.

Intervention	Weighting factor	Tax or fee base
Extraction		
Fossil energy	0-0.15 SEK / MJ	Tax on fossil energy
Biotic energy	0-0.069 SEK/MJ	Tax on biotic energy
Emission		
CO2	0.63 SEK/kg	Tax on carbon content in fossil fuel
Ozone depleting substances	1200 SEK/kg	Fee for using prohibited ozone depleting substances
Nitrogen	12 SEK/kg	Tax on nitrogen content of fertiliser recalculated due to leakage of 15% (tax 1.80 SEK/kg)
HC	20-200 SEK/kg	Emission fee for air traffic
Sulphur	30 SEK/kg	Tax on sulphur content in fossil fuels
Toluene	17.65-36.07 SEK/kg	Tax differentiation on petrol qualities (unleaded petrol vs. alkylate petrol)
Cadmium	30 000 SEK/kg	Tax on content of cadmium exceeding 5 g/1000kg phosphor in fertiliser
Pesticides / Copper	20 SEK/kg	Tax on active substance in pesticides

The weighting factors in Table 12 are combined with different impact categories in Table 13. Minimum and maximum values are used for some impact categories indicating uncertainties in the methods. The weights of reference in Table 13 indicate the value of the reference substance used in the different impact categories.

Table 13. Weights used in minimum and maximum combinations.

Impact category	Combi-nation	Weighting factor	Reference of the characterisation method (eq)	Weight of reference
Abiotic resources	Min	0 SEK / MJ	MJ	0 SEK/MJ
	Max	0.15 SEK / MJ	MJ	0.15 SEK/MJ
Biotic resources	Min	0 SEK / MJ	MJ	0 SEK / MJ
	Max	0.069 SEK / MJ	MJ	0.069 SEK / MJ
Global warming	Min	0 SEK / kg CO_2	CO_2	0 SEK / kg CO_2
	Max	0.63 SEK / kg CO_2	CO_2	0.63 SEK/kg
Depletion of stratospheric ozone	Min/Max	1200 SEK / kg ozone depleting substance	CFC-11	1200 SEK/kg
Photochemical oxidation	Min	20 SEK / kg HC	C_2H_2	48 SEK/kg
	Max	200 SEK / kg HC	C_2H_2	480 SEK/kg
Acidification	Min/Max	30 SEK / kg Sulphur	1.2 SO_2	18 SEK/kg
Eutrophication	Min/Max	12 SEK / kg N	PO_4	28.57 SEK/kg
Fresh water aquatic ecotoxicity	Min	17.65 SEK/kg Toluene	1,4-dichlorobenzen emitted to freshwater	60.86 SEK/kg
	Max	36.07 SEK/kg Toluene		124.37 SEK/kg
Marine aquatic ecotoxicity	Min	20 SEK/kg Copper	1,4-dichlorobenzen emitted to seawater	1.333×10^{-5} SEK/kg
	Max	20 SEK/kg Glyphosate		0.606 SEK/kg
Terrestrial ecotoxicity	Min/Max	30 000SEK/kg Cd	1,4-dichlorobenzen emitted to agr. Soil	176.47 SEK/kg
Human toxicity	Min/Max	30 000SEK/kg Cd	1,4-dichlorobenzen emitted to agr. Soil	1.50 SEK/kg

Eco-indicator 99

The Eco-indicator 99 is described by Goedkoop and Spriensma (2000) and there are three versions of the Eco-indicator method with different "perspectives" available; Egalitarian, Individualist and the Hierarchist allowing different assumptions on time horizon, manageability etc. The default hierarchist is used in this study. The Eco-indicator 99 developers have prioritised two things. The number of subjects should be as few as possible to the panel. Subjects to be weighted should be easily explained to the panel. From these requirements the panel should only weight three types of environmental damages (endpoints); *Human health, Ecosystem quality and Resources*.

Linking the damage categories to inventory results have been done with help of damage models. Damage to *Human health* is expressed as DALY (Disability Adjusted Life Years). Climate change, ozone layer depletion, ionic radiation, respiratory and carcinogenic effects have been modelling in different ways to serve this purpose. Damages to *Ecosystem Quality* are expressed as the percentage of species that have disappeared in a certain area due to the environmental load. Here ecotoxicity is expressed as the percentage of all species present living under toxic stress (PAF). Acidification and eutrophication are treated as a single impact category. Land-use and land transformation are also included in this damage category. *Resource* extraction is related to a parameter that indicates the quality of the remaining mineral and fossil resources.

EPS 2000

EPS 2000 evaluate impacts on the environment via its impact on one or several safeguards subjects. These have according to Steen (1999) been chosen from those who were included in the Rio protocol, although not necessarily explicitly formulated there: human health, resources, ecosystem production capacity, bio-diversity and esthetical values. Esthetical value is today extended and named “cultural and recreational” values and resources are specified as “abiotic stock resources”. Default weighting indicator is willingness to pay (WTP) to restore impacts on the safeguards subjects. For some category indicators, the market price may be used to estimate WTP. It could also happen that no monetary value at all is found, in that case some other estimation is made.

ExternE

The European research programme ExternE has established the external environmental cost per unit of energy for different energy types, as well as the cost per tonne of various emissions (Table 14). It used an impact pathway approach with exposure-response functions to estimate the impacts and damages of SO_2, NO_x and PM10 on the most important receptors, that is human health, crops, materials, and ecosystems (the same approach as used for calculating site-dependent characterisation factors above). Furthermore, it used results from the most updated environmental economic research studies for applying Euros per impact case.[1] In the case of health impacts the values are based on willingness-to-pay studies for various health cases. For crop production, the market value of the production loss was used, and for material damage, the costs of restoring corroded materials were used as a basis. For non-air pollution impacts, direct willingness-to-pay studies were made for issues such as cultural landscape effects, visual intrusion and noise. For climate change, special modelling was made with the FUND model and the Open Framework model (European Commissions, 1999). While the project did not calculate any average damage cost estimates for SO_2, NO_x and PM10, we have used the average of the calculations made for Sweden (Nilsson and Gullberg, 1998). The CO_2 estimate of 2.4 Euros/tonne is recommended to be used with an indicative low-high range from 0.1 to 16.4 (Krewitt, 2002).

Table 14. external environmental costs as determined by the ExternE programme.

		CO_2	NO_X	SO_2	PM10
ExternE	Euros/tonne	2.4	2065	2500	3150
	SEK/kg	0.02	18.5	22.5	28.5

Modifications in methods

When results from the inventory analysis are compared with list for the used characterisation and weighting methods, it becomes evident that some inventory parameters are undefined, i.e. they are not included in the impact assessment data sets. Three different cases can be found. In some cases it is only a matter of the same substance but with another name, then it can easily be added to the impact assessment data sets. In some cases there are no equivalent substances to insert, we must then leave them out. And finally some substances are similar and can be approximated with the related substance.

[1] The most updated dose-response functions and monetary values used as in-data in the project are available in Rainer Friedrich and Peter Bickel, Stuttgart, Germany (Eds.) Environmental External Costs or Transport. 2001. Springer.

6 Results of the qualitative analysis

In this section the results from filling in the checklists in the qualitative assessment are presented. This can be seen as a decomposed description of the different alternatives focusing on the main environmental aspects of the different treatment methods. How the different waste fractions are treated in each alternative is summarised in table 15. For a more detailed and disaggregate description of the waste flows, please see tables 3-6. For simplicity only those objectives under which entries have been made are shown here. For the complete list of environmental objectives, please see section 5.1. First the results for transports, coal power and heat from timber felling residues are presented separately. After that the results for each waste management option is shown. Information for filling in the checklists for transports, coal power and heat from timber felling residues have been taken from a compilation of data for electricity and fuels (Wolf-Watz et al, 2000). For the waste management alternatives information were discussed and agreed on within the project team based on previous studies and knowledge. For the waste management options a distinction has been made between direct and indirect effects, where effects connected to the waste handling itself are considered direct, and effects of using by-products of the waste handling are considered indirect. For example, the emissions in connection to collection and composting of park and garden waste are direct, while effects of using the compost residues as fertiliser are indirect. Among the indirect effects in this case is that production of synthetic fertiliser, and its environmental burden, can be avoided. Towards the end of the section a summary is made where the waste fractions are commented with respect to the treatment options that are considered for each material in the different alternatives. Based on this, conclusions can be drawn on the merits of the alternatives.

Table 15. An overview of which waste fractions goes to which treatment in the different alternatives.

	Digestion (kton/year)			Composting (kton/year)			Recycling (kton/year)			Incineration (kton/year)			Landfill (kton/year)		
Alternative	**0**	**1**	**2a/b**	**0**	**1**	**2a/b**	**0**	**1**	**2a/b**	**0**	**1**	**2a/b**	**0**	**1**	**2a/b**
Compost	60	114	0/678	172	265					521	374	753/75			
Park and garden waste				191	242					50		242			
Wood										1191	1191	1191			
Textile, rubber and leather										119	119	119			
Newsprint							540	577	611	138	101	68			
Other paper and cardboard							743	780	1436	874	838	180			
Plastics							89	95	393	436	430	90			
Glass							202	216	238				62	48	26
Aluminium							18	19	26				11	9	3
Steel containers							32	34	46				20	18	5
Mic. metals							284	302	379				138	119	42
Gypsum							9	26	157				166	149	17
Concrete, tiles and bricks							323	808	1454				1292	808	162
Asphalt							124	124	124				14	14	14
Other combustibles								9	77	782	778	709			
Other non-combustibles													902	902	902
Total	**60**	**114**	**0/678**	**363**	**507**	**0**	**2240**	**2990**	**4941**	**4111**	**3831**	**3352/ 2674**	**2605**	**2067**	**1171**

6.1 Transports

The waste is assumed to be transported by trucks or other specially designed vehicles, using diesel for fuel.

Reduced climate impact

Carbon dioxide is emitted from the combustion of the fuel.

Carbon dioxide is also emitted from extraction and production of oil products.

Clean air

Hydrocarbons, nitrogen oxides, and particles are emitted on combustion of the fuel. These substances, along with sulphur oxides, are also emitted during production and distribution of the fuel, although in lower amounts.

Natural acidification only

Nitrogen oxides are emitted during production, distribution and combustion of diesel fuel.

Sulphur oxides are emitted during production and distribution of fuel.

A non-toxic environment

Hydrocarbons are released when combusting the diesel fuel.

Harmful substances from tyres, breaks and asphalt are released to the environment as these materials wear during use.

Zero eutrophication

Emissions of nitrogen oxides occur from production, distribution and use of diesel.

Good-quality groundwater

When distributing the fuel there may be risks of accidents, with subsequent contamination of ground water.

A balanced marine environment and sustainable coasts and archipelagos

When transporting oil, there is a risk for oil-spills, due to either accidents or careless handling.

A good built environment

Transports often cause problematic intrusion into the built environment, in various ways disturbing people's wellbeing including health and aesthetic experiences.

6.2 Electricity production from coal

Where electricity is bought from the grid or saved, in comparison to other alternatives, it is assumed to be produced in coal fired power plants, as described in section 5.2.3.

Reduced climate impact

Carbon dioxide, as well as some methane is emitted from the combustion of coal. These substances are also released during production and distribution of the fuel.

Clean air

Combustion of coal gives rise to emissions of sulphur oxides, nitrogen oxides, particles, various hydrocarbons and heavy metals.

Natural acidification only

Emissions of sulphur and nitrogen oxides occur form the combustion of coal.

A non toxic environment

Emissions of toxic substances, such as heavy metals and hydrocarbons will take place from the combustion of coal.

Toxic substances may leak from mining processes and mining waste.

A safe radiation environment

Some radiation may be released from coal mining.

Good-quality groundwater

Leakage from coal mining may contaminate groundwater.

No eutrophication

Nitrogen oxides are emitted from coal combustion.

A good built environment

The power plant requires some space. It will probably not be appreciated as a positive component of the built environment. Coal will have to be supplied to the facility in quite large amounts, generating a lot of transports. Mining produces a lot of residues and the mines require a lot of space.

6.3 Heat production from timber felling residues

Reduced climate impact

Carbon dioxide is emitted when combusting timber felling residues, but since they are of recent biological origin these emissions can be disregarded. Some carbon dioxide emissions however occur from the collection and transport of the residues from the forest.

Clean air

Emissions of nitrogen and sulphur oxides, hydrocarbons, heavy metals and particles are made from the combustion.

Natural acidification only

Sulphur and nitrogen oxides are emitted.

Taking out the forest felling residues may lead to acidification, especially if the ashes are not brought back to the forest.

A non toxic environment

The slag and ashes formed during the combustion process have to be disposed of. Sustainable harvesting of forest felling residues probably requires the ashes to be redistributed in the forest. The slag and ashes may contain metals and other harmful substances that could leak out in toxic concentrations.

Emissions of heavy metals and hydrocarbons from the combustion of the residues and from the collection and transport of the residues.

Sustainable forests

Despite a lot of research in this area, the long term consequences of harvesting timber felling residues for the micro and macro organisms of the forest community are uncertain. A sustainable harvest probably requires slag and ashes to be brought back to the forest in order not to disturb the nutrient balance.

A good built environment

The heating facilities are placed close to residential areas. Forest felling residues are rather voluminous and thus quite a lot of transport will be generated. There will also be a need to transport the slag and ashes from the facility.

6.4 Digestion

Anaerobic digestion is an alternative for compostable household waste in scenarios 0, 1 and 2b. Useful by-products are biogas that is assumed to be recovered and used to replace diesel as bus fuel, and a residue that is used as fertiliser, replacing synthetic fertiliser.

Transports: Transports of the waste to the digestion facility and of the residues to farmland are required. Digesters do not have to be run in large scale in order to be efficient and can thus be placed at a close distance from each other. The household waste can also be digested together with farming residues. Because of this the transports may be relatively short.

Reduced climate impact

Direct effects:

A small amount of methane may leak out.

Indirect effects:

Replacement of diesel fuel in busses will affect the objective positively since emissions of fossil CO_2 is replaced with CO_2 of biological origin.

If the fertiliser replaced is synthetic, energy for the production of that fertiliser is offset, potentially saving CO_2-emissions. On the other hand increased N_2O emissions from the digestion-derived fertiliser can be expected in the field, since it is difficult to control the dosage of nitrogen applied.

Clean air

Indirect effects:

When the gas recovered is used to replace diesel fuel there may be less emissions of NO_x and SO_x and there will be less particles emitted.

Air-polluting emissions from electricity production from coal can be avoided when synthetic fertiliser is replaced.

Natural acidification only

Indirect effects:

There will probably be more leakage and air emissions of acidifying nitrogen compounds from the digestion residues when applied to farmland than if synthetic fertiliser was used.

Energy savings when synthetic fertiliser does not have to be manufactured leads to less emission of nitrogen and sulphur oxides from coal combustion.

Less nitrogen and sulphur oxides are probably emitted from the biogas than from diesel.

A non toxic environment

Indirect effects:

Via the distribution of digestion residues to farmland a diffuse spreading of various contaminants and harmful substances present in the residues may occur. They could come from objects that are not intended for digestion but has ended up together with the compostables anyway, such as batteries. It could also be harmful substances, such as heavy metals, present in the waste itself.

Energy savings when synthetic fertiliser does not have to be manufactured leads to less emission of toxic compounds from coal combustion.

Less toxic emissions are made when biogas is used as a fuel compared to diesel.

Distribution of cadmium and other heavy metals present in synthetic fertiliser can be avoided.

No eutrophication

Indirect effects:

More nutrients probably leak out from the soil after application of digestion residues compared to if synthetic fertiliser was used.

Less emission of nitrogen oxides probably occurs from combustion of biogas than from combustion of diesel.

Flourishing lakes and streams

See A non toxic environment and No eutrophication.

Good-quality groundwater

Se A non toxic environment and No eutrophication.

Thriving wetlands

Indirect effects:

By the application of digestion residues the humus content in the soil is improved. This could limit the demand for peat to enhance agricultural soil and in turn leave more peat bogs undisturbed.

A balanced marine environment and sustainable coasts and archipelagos

Se A non toxic environment and No eutrophication.

A varied cultural landscape

Indirect effects:

The use of digestion residues increases the humus content, improves the soil structure and adds nutrients. This will promote the long-term productivity of the agricultural soil.

Se also A non toxic environment and No eutrophication.

A good built environment

Direct effects:

The digestion process requires a rather large facility.

There may also be a need to stock the waste for some time before it can be digested. This stock could give rise to some odour.

Indirect effects:

Using the residues as fertiliser can be regarded as a kind of recycling.

Energy and natural resources can be saved.

Using the biogas instead of diesel means that a renewable one can replace a fossil energy source.

Natural resources can be saved.

6.5 Composting

Composting is a treatment alternative for compost able household waste and garden and park waste in scenarios 0 and 1. Most of the combustible waste is taken care of in large-scale composts but some of the composting takes place in small composts in individual households. The compost is assumed to be used as fertiliser, replacing synthetic fertiliser.

Transport: In the case of large-scale composts transportation of the waste to the facility is required, as well as transportation of the residues to farmland. In the case of household composting no transports are needed, instead transport demand may be reduced.

Reduced climate impact

Direct effects:

Biological CO_2 is released from the compost directly into the atmosphere. The composting process is supposed to take place under aerobic conditions. However, if the compost is ill-maintained anaerobic conditions may occur, causing methane to be emitted as well. There is no major difference between larger scale composting and household composting other than that it may be easier to control the conditions in the large scale compost.

Indirect effects:

Residues from large scale composting may replace synthetic fertiliser. Energy will be saved but more N_2O may be emitted from the fields.

Clean air

Indirect effects:

Energy form coal is saved when synthetic fertiliser can be replaced, hence air-polluting emissions from coal combustion can be avoided.

Natural acidification only

Direct effects:

Nitrogen compounds are emitted from the compost.

Indirect effects:

As for digestion there will probably be more leakage and air emissions of acidifying nitrogen compounds from the compost residues when applied to farmland than if synthetic fertiliser was used.

Energy form coal is saved when synthetic fertiliser can be replaced, and emissions of nitrogen and sulphur oxides can be avoided.

A non toxic environment

Direct effects:

There may be a leakage of toxic substances trough the drainage water, especially if rainwater is allowed to percolate through the compost, as is often the case in large scale open string composts.

Indirect effects:

The distribution of compost to farmland may contribute to the diffuse spreading of harmful substances in the environment.

Energy savings when synthetic fertiliser does not have to be manufactured leads to less emission of toxic compounds from coal combustion.

Distribution of cadmium and other heavy metals in present in synthetic fertiliser can be avoided.

Zero eutrophication

Direct effects:

There may be a leakage of nutrients from the compost.

There may be emissions of nitrogen compounds to air from the compost.

Indirect effects:

Compared to synthetic fertiliser more emissions of substances contributing to eutrophication can be expected from the compost residues.

Energy savings when synthetic fertiliser does not have to be manufactured leads to less emission of nitrogen oxides from coal combustion.

Flourishing lakes and streams

See A non toxic environment and No eutrophication.

Good-quality groundwater

See A non toxic environment and No eutrophication.

A balanced marine environment and sustainable coasts and archipelagos

See A non toxic environment and No eutrophication.

Thriving wetlands

Indirect effects:

Application of compost residues increases the humus content of the soil, and this may decrease the demand for peat to enhance agricultural soil. Exploitation of peat bogs may thus be avoided.

A varied cultural landscape

Indirect effects:

The use of compost residues increases the humus content, improves the soil structure and adds nutrients. This will promote the long-term productivity of the agricultural soil.

See also A non toxic environment and No eutrophication.

A good built environment

Direct effects:

For large scale composting a facility is required. For home composting no special building is required.

The compost may give rise to odour and attract vermin such as rats. Home composts are often closed and do not have the same problem with vermin and odour.

Indirect effects:

Using the residues as fertiliser can be regarded as a kind of recycling.

Energy and natural resources can be saved.

6.6 Incineration

Incineration is considered for compostable household waste, garden and park waste, wood, textile, rubber and leather, paper and cardboard, plastics and the rest fraction ‘other combustibles’ in nearly all scenarios. The by-product, heat used for district heating, is assumed to replace heat produced from the combustion of timber felling residues.

Transport: Waste incinerators are generally built with large capacity. This means that the uptake area from which waste is transported to the incineration facility is large and the transport distances are thus quite long. The slag and ashes formed have to be transported to a landfill site.

Reduced climate impact

Direct effects:

Carbon dioxide is emitted from the combustion. The carbon in the different fractions is of biological or fossil origins, where CO_2 formed from biological sources are sometimes considered not to affect the objective. Fossil sources of CO_2 in the waste are different kinds of plastics. Note that some of the textile, rubber and leather are made from fossil raw materials, e.g. polyester, and synthetic rubber and leather.

Clean air

Direct effects:

Incineration of waste causes emissions of nitrogen and sulphur oxides, and a wide range of organic and inorganic compounds.

Indirect effects:

Heat production from timber felling residues can be avoided.

Natural acidification only

Direct effects:

Sulphur and nitrogen oxides are emitted from the combustion.

Indirect effects:

Heat production from timber felling residues can be avoided.

A non toxic environment

Direct effects:

From the combustion air emissions occur as described under 'clean air' above.

The slag and ash formed during combustion may either be disposed of in a landfill or used as ballast material in e.g. road constructions. In both cases toxic substances, such as heavy metals, may leak out into surrounding ecosystems.

Indirect effects:

Heat production from timber felling residues can be avoided.

Zero eutrophication

Direct effects:

Nitrogen oxides are formed and emitted from the combustion process, as stated above.

Indirect effects:

Heat production from timber felling residues can be avoided.

Good-quality groundwater

Indirect effects:

Toxic substances, such as heavy metals, may leak out from the slag and ash into the ground water.

Heat production from timber felling residues can be avoided.

Sustainable lakes and watercourses

Indirect effects:

Toxic substances, such as heavy metals, may leak out from the slag and ash into the ground water.

Heat production from timber felling residues can be avoided.

Healthy forests

Indirect effects:

Heat production from timber felling residues can be avoided.

A good built environment

Direct effects:

The incineration facilities are rather large, occupy a lot of space and make a visual impact.

The facilities have quite a bad reputation and people are reluctant to have them in the vicinity of their homes, partly due to fear of unhealthy emissions.

The facilities often generate a lot of traffic, since large volumes are incinerated daily.

Indirect effects:

Natural resources are saved when timber felling residues are replaced.

Parts of the waste can be considered to be of renewable origin.

Natural gravel can be saved if replaced by slag as a ballast material.

6.7 Recycling

Recycling is a treatment alternative for paper and cardboard, plastics, glass, metals, gypsum, asphalt, concrete, bricks and tiles and ceramics in nearly all scenarios. The recycled product is assumed to replace a similar product produced from virgin raw-material.

Transports: The waste has to be transported from its source to the recycling facility. In the case of household waste, the waste will have to be collected from the households or from certain collection points. If the waste is transported from the household to these collection points by car, considerable emissions may occur. The recycling facilities are typically few for each fraction and thus it can be expected that the transport distances will be fairly long. However, transportation of virgin materials for production of replaced materials can be reduced.

Reduced climate impact

Direct effects:

The recycling processes require energy. Electrical energy from coal gives rise to CO_2 emissions.

Indirect effects:

Production from virgin raw materials can be avoided. Generally recycling tends to use less energy than virgin production processes, both for processes and for the extraction of raw materials. For paper made from mechanical pulp, e.g. newsprint, substantial energy savings can be made. This means a lower consumption of electrical energy from coal. For recycling of paper made from chemical pulp it is uncertain whether energy is saved compared to virgin production. In both cases less timber has to be transported from the forests to the pulp mills.

Production of recycled plastics is less energy demanding than virgin plastic production, and thus less energy from coal will be used. The same holds for glass.

For metals mining and processing of ores can be avoided by recycling, both of which can be very energy intensive.

For gypsum the gain is more uncertain, since much of the raw material for gypsum products are made from gypsum recovered as a by-product from the purification of flue gas in incinerators.

To crush concrete, ceramics and bricks and tiles for use as ballast material in e.g. road constructions is probably less energy intensive than to crush rock for the same purpose. The crushing will probably be carried out with machines powered by diesel.

Clean air

Direct effects:

The emissions from the production processes themselves should not differ too much between virgin production and recycling.

Indirect effect:

In recycling emissions connected to the extraction and processing of raw materials can be avoided to a large extent.

Differences in energy use between recycling and virgin production also determine how the objective is affected (see above).

Natural acidification only

NOx and SOx emissions will occur from the energy processes. Less energy is often used in recycling than in virgin production.

A non toxic environment

Direct effects:

Paper intended for printing-grade products must be de-inked, after pulping and for some uses the stock is bleached before pressing into sheets. Chemicals may be emitted from the de-inking and bleaching processes. Water emissions of organic substances from pulp production, reaction products from bleaching and metals from wood should be less from recycling than from virgin production.

Chemical additives present in plastics may be spread in a diffuse way when plastics are recycled into new products. They may be emitted to air, process water or pass on to the new product and in general little is known about this.

Mining and extraction of metals causes a lot of emissions of toxic substances. These emissions can be avoided when recycling.

Zero eutrophication

NOx from energy processes.

Nitrogen, phosphorus and other nutrients from wood leak out from pulp production. A large part of this can be avoided when recycling.

When food containers are to be collected for recycling they have to be rinsed. With the cleaning water some emissions of nutrients are made.

Sustainable forests

When paper is recycled some wood can be saved. This may ease the intensity of the logging in the forests.

A good built environment

Recycling means that natural resources can be used more efficiently and often energy is saved. An increased recycling rate is a goal in itself within this objective.

Natural gravel can be saved as concrete, bricks and tiles and ceramics are reused as ballast material.

Recycling often requires the incoming waste to be source separated. The collection points are sometimes not adding to aesthetic experiences and wellbeing, due to their design or to how they are maintained. They sometimes also attract vermin such as rats. These are not inherent problems but require some efforts to be solved.

Cleaning and transporting the waste to collection points may increase energy use on the household level significantly, especially if the waste is transported by means of passenger cars. This may also increase the transport needs.

6.8 Landfill

Landfill is an alternative for glass, metals, gypsum, concrete, bricks and tiles and ceramics, and ‘other non-combustible waste’ in all scenarios. For landfills we have not assumed any useful by-products and thus all effects are considered direct effects.

Transports: Landfill sites are often situated fairly close to cities and towns, allowing for comparably short transport distances.

Reduced climate impact

The materials here considered for landfill have a low carbon and nitrogen content and will probably not have a major effect on the objective ‘limitation of climate change’. An exception may be the fraction other non-combustible waste, where the composition is unknown.

Clean air

Limited impact.

Natural acidification only

Limited impact.

A non toxic environment

Various toxic substances are known to leak out of landfills. It could for instance be different chemical additives and their decomposition products, and metals.

Zero eutrophication

Limited impact.

Flourishing lakes and watercourses

There have been toxic effects on water organisms, which are suspected to have their origin in emissions from landfills. What substance it is or which material it comes from is not exactly known.

Good-quality groundwater

Toxic substances may leak out from the landfill and reach the groundwater.

A good built environment

Landfilling is supposed to be cut by half between 1994 and 2005. Choosing this option will make it harder to reach that goal.

Landfilling can generally not be considered as an efficient way to use natural resources.

6.9 Summary and conclusions

A summary of the major advantages and disadvantages of the different treatment options are listed in table 16. With these pros and cons in mind we can evaluate how the different waste fractions turn out in the different alternatives.

Compost is treated with digestion, composting or incineration in the different alternatives. It is difficult to determine which of these methods is environmentally preferable. Digestion and composting share many features, but from an energy perspective, considering that energetic gas can be recovered from the digestion process, digestion may be the preferable option of the two. Choosing between digestion and incineration is less straightforward. The choice involves, on the part of digestion, a fertiliser improving the soil quality which is available with little energy input, but which is potentially toxic and that may cause increased nutrient leakage. On the part of incineration the choice involves an easily available bio fuel where the major down sides are air pollution and the formation of slag and ash.

If composting is judged to be the least preferable option alternatives 2a/b are the best, since no composting occur in these alternatives. In alternative 1 a larger fraction is composted than in alternative 0. In alternative 2a all of the compost is incinerated, while in 2b nearly all of it is digested. In the 0-alterantive more is incinerated than in alternative 1.

Park and garden waste is either composted or incinerated, with the above mentioned implications valid here as well. In alternatives 1 and 2a/b all of the park and garden waste is *incinerated, while in the 0-alternative most of it is composted.*

Wood and *textile, rubber and leather* are incinerated in all alternatives.

Paper and cardboard is either recycled or incinerated. Recycling is in general the most energy effective option and it also saves raw material. The most recycling occurs in alternatives 2a/b. In alternative 1 there is slightly more recycling than in the 0-alternative.

Plastics are recycled or incinerated. If the plastics can be separated into suitable fractions recycling saves energy and raw material. This separation is however difficult to achieve. This often means the recycling is made from mixed plastics. The product then holds a lower quality than plastics made form virgin materials, and therefore it also has a weak market. Recycled plastics may also carry contaminants that are difficult to keep track of. These contaminants may have been present in the incoming waste or may have been formed in the re-melting process. Incineration on the other hand gives rise to air pollution and the CO_2

emitted is of fossil origin. In alternative 0 and 1 most of the plastic waste is incinerated and a smaller fraction is recycled. In alternative 2a/b it is the other way around.

Glass is either recycled or put on landfill. The preferable option is recycling since energy and material can be saved this way. The alternatives are quite similar although alternatives 2a/b have the highest recycling rates. Alternative 1 has slightly more recycling than alternative 0.

Table 16. A summary of some of the major advantages and disadvantages with different waste treatment options.

	+	-
Digestion	Biogas is produced that can replace fossil fuels. The digestion residues can be used as fertiliser and may then replace synthetic fertiliser. Energy can be saved when less synthetic fertiliser has to be produced. When applied to farmland the residues also improve the soil structure. The residues also contain more micronutrients than synthetic fertiliser does.	Increased leakage of nutrients and other substances that may be present in the residues from farmland when residues are used as fertiliser. If toxic substances are present in the residues they may enter into food crops. Requires technically advanced facilities.
Composting	The compost residues can be used as fertiliser and may then replace synthetic fertiliser. Energy can be saved when less synthetic fertiliser has to be produced. When applied to farmland the residues improve the soil structure. They also contain more micronutrients than synthetic fertiliser does. Fairly simple facilities are sufficient.	Greenhouse gases may be emitted from the compost. Increased leakage of nutrients and other substances that may be present in the residues from farmland when residues are used as fertiliser. If toxic substances are present in the residues they may enter into food crops. There may be problems with odour and vermin.
Incineration	Heat can be recovered and thus it may replace other fuels in district heating production. Harmful substances present in the waste may be destroyed in the combustion process.	Fossil greenhouse gases are released when waste of fossil origin is incinerated. Sulphur and nitrogen compounds are emitted. It is however unclear if the amounts involved are greater than those from incineration of other fuels, such as forest felling residues. Toxic substances are formed and released. A lot of slag and ash is formed. Requires large and technically advanced facilities.

		A lot of transports are generated.
Recycling	Natural resources are saved. Energy is generally saved compared to virgin production. Environmental effects form extraction, production and transports of raw materials can be avoided.	Generally requires careful source separation. This also makes waste collection more problematic. It may sometimes be hard to achieve sufficient quality in the recycled products. Toxic substances in the waste materials may enter unnoticed into the recycled products and new toxicants may be formed in the recycling processes.
Landfill		Fossil greenhouse gases are released over time as waste of fossil origin degrades. Leakage of various substances. Requires a lot of space. Often considered bad management of natural resources.

Metals are recycled or put on landfill. For metals there are obvious benefits with recycling. Mining operations and various production processes can be avoided. Recycling rates are highest in alternatives 2a/b followed quite closely by alternative 1. The 0-alternative has the lowest recycling rates.

Gypsum is recycled or landfilled. Recycling probably saves energy and resources. The absolutely highest recycling rates occur in alternatives 2a/b. The difference between alternative 1 and 0 is small, however, alternative 1 has the highest recycling rate.

Concrete, tiles and ceramics are recycled or put on landfill. Recycling probably saves energy and resources. The highest recycling rate is present in alternative 2a/b, followed by alternative 1. The absolutely lowest rate can be found in the 0-alternative.

Asphalt is recycled or put on landfill. In all of the alternatives most of the asphalt is recycled.

A large undefined waste fraction is termed *other combustibles*. Some of this fraction is thought to be recyclable. In all of the alternatives most of this fraction is incinerated, but in alternatives 2a/b some of it is recycled. A small part is also recycled in alternative 1. Another large undefined fraction is called other *non-combustibles*. In all alternatives this fraction is put on landfill.

A high recycling rate is generally considered desirable. If this is determining the choice alternatives 2a and 2b are the best. The preference for how compost and park and garden waste is to be treated can be another determinant. It may, however, be wise to consider these fractions separate from the other ones, especially since they may require other policy measures. If incineration is found to be the most desirable, then an exemption of these fractions from the general tax may be for example be advisable.

7 Results of the traditional LCA

This chapter analyses the characterised results of the LCA. Characterisation is done using the CML 2000 method with additions as explained in Chapter 0 Complete results from the inventory analysis are presented in Appendix E.

First the results of the base case are presented, in which the LCA model runs were performed with the surveyable time perspective for landfill emissions (c.f. Chapter 5). This is followed by a brief analysis of the effect of applying an infinite time perspective for landfills.

Energy turnover is presented first, because it has a large influence on many environmental impact categories and thus gives a good overview of the system as a whole. This is followed by a summary of all environmental impact categories, in which major trends are pointed out. Then for each environmental impact category, the no-action alternative (Alt 0) is described in terms of what processes and what emissions dominate that particular impact. Alternatives 1, 2a, and 2b are then compared to the no-action alternative to identify the most significant improvements and detoriations.

There are both contributions to and savings of each impact categories. Savings are a result of avoided burdens when resources are recovered from waste. If resource recovery (material recycling, nutrient recycling, or energy recovery) results in net savings, this means that resource recovery from waste (including transports, energy, and processing) causes less impact than producing the same product from virgin resources. It is important though to remember that the waste input to the system is viewed as zero-burden, i.e. the production and use phases of the materials constituting the waste is not included in the inventory. Likewise, the energy content in waste is not considered an energy resource input to the system, although it may be recovered as an output from the system. If the waste input was not zero-burden, there would never be savings of an impact category.

7.1 Energy

Table 17 summarises the net energy turnover in terms of total, non-renewable, and renewable energy of the different alternatives. It also shows the relative changes compared to Alt 0. The relative changes are primarily meant for comparison among Alt 1, Alt 2a, and Alt 2b, as they say nothing about the significance of the absolute changes.

Table 17. Summary of net energy turn-over [MJ] and relative change [%] compared to Alt 0.

	Alt 0	Alt 1	Alt 2a	Alt 2b
total energy	-1.04E11	-1.07E11 (-3%)	-1.38E11 (-33%)	-1.37E11 (-32%)
non-renewable energy	-1.3E10	-1.54E10 (-19%)	-4.38E10 (-237%)	-4.6E10 (-246%)
renewable energy	-9.11E10	-9.11E10 (0%)	-9.86E10 (-8%)	-9.53E10 (-5%)

All alternative waste management strategies achieve net savings of total energy use in the order of 10^5 TJ. This shows that material and energy recovery from waste as a whole saves more virgin energy resources than what is expended by waste handling and recovery processes. With the specified system design, savings of renewable and non-renewable energy are in the same order of magnitude, although total energy savings are dominated by renewable energy.

The main savings of renewable energy in Alt 0 are achieved by energy recovery from waste incineration, which replaces biofuels in heat production, and to some extent by newspaper recycling, which avoids use of biomass in virgin newspaper production. The main savings of non-renewable energy in Alt 0 are achieved by recycling of newspaper, metals, and plastics in industrial waste. These savings are attributable to the avoided use of fossil fuels, and electricity from fossil fuels, in virgin material production.

The difference in total energy use between Alt 0 and Alt 1 is small, although Alt 1 appears to bring a slight improvement by a net reduction of non-renewable energy use. Less waste incineration in Alt 1 reduces the savings of biofuels in district heating, but this is cancelled out by increased savings of renewable energy achieved by recycling of newspaper, office paper, and corrugated cardboard. A minor increase of non-renewable energy use in Alt 1 is caused by corrugated cardboard recycling, but the increased savings achieved by recycling of steel, newspaper, plastics in industrial waste and aluminium are much larger.

The potential energy savings in Alt 2a and Alt 2b are considerably larger than in Alt 1. The largest savings are in non-renewable energy use, dominated by recycling of steel, PE, and newspaper. Even less waste is incinerated than in Alt 1, leading to further reduced savings of biofuels for district heating. Despite this, renewable energy use is reduced as a whole, mainly due to increased recycling of office paper, cardboard, and newspaper.

Contribution by waste category

The contribution by waste category to the overall energy turnover in Alt 0 is displayed in Figure 3. The pattern is similar in the other alternatives.

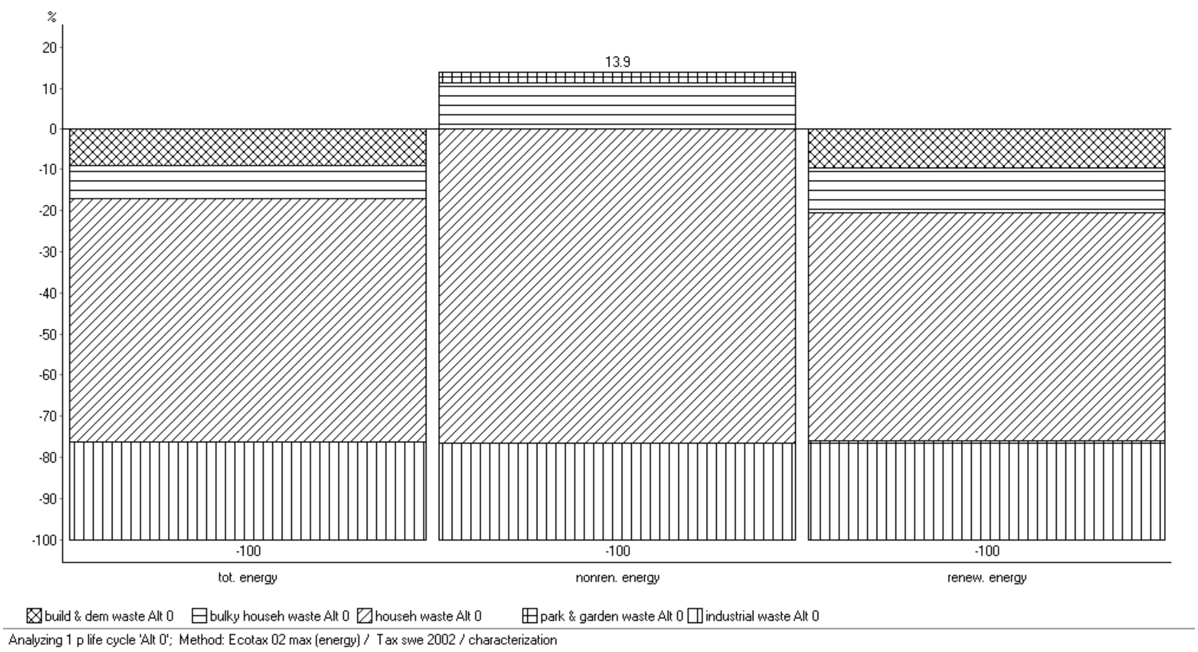

Figure 3. Energy turn over in Alt 0. Contribution by waste category.

The diagram shows that the net energy savings are dominated by resource recovery from household waste, and to a lesser extent by industrial waste. Building and demolition waste and bulky household waste are much less significant, whereas the role of park and yard waste is insignificant compared to the other waste categories.

The difference in contribution by waste category is partly explained by the amounts of the different waste categories that are included in the study. However, household waste has a larger relative share of the total energy savings, than its share of the total waste flow. On the contrary, building and demolition waste has a smaller relative share of the total energy savings, than of the total waste flows. This reflects the treatment of the different waste categories, with relatively more resource recovery (materials recycling and incineration) and less landfilling of household waste than of the other waste categories. It is also partly explained by the large amounts of concrete, asphalt, and plaster board in building and demolition waste. The LCA model assumes that recycling and the corresponding avoided production of these fractions have relatively similar impacts, thus no large savings are achieved although recycling rates are high.

7.2 Environmental impact categories

Figure 4 displays a relative comparison of environmental impacts in all alternatives. The largest net impact (positive or negative) is scaled to 100%. It is important to note that a large relative change between two alternatives does not imply that the change is large in absolute numbers.

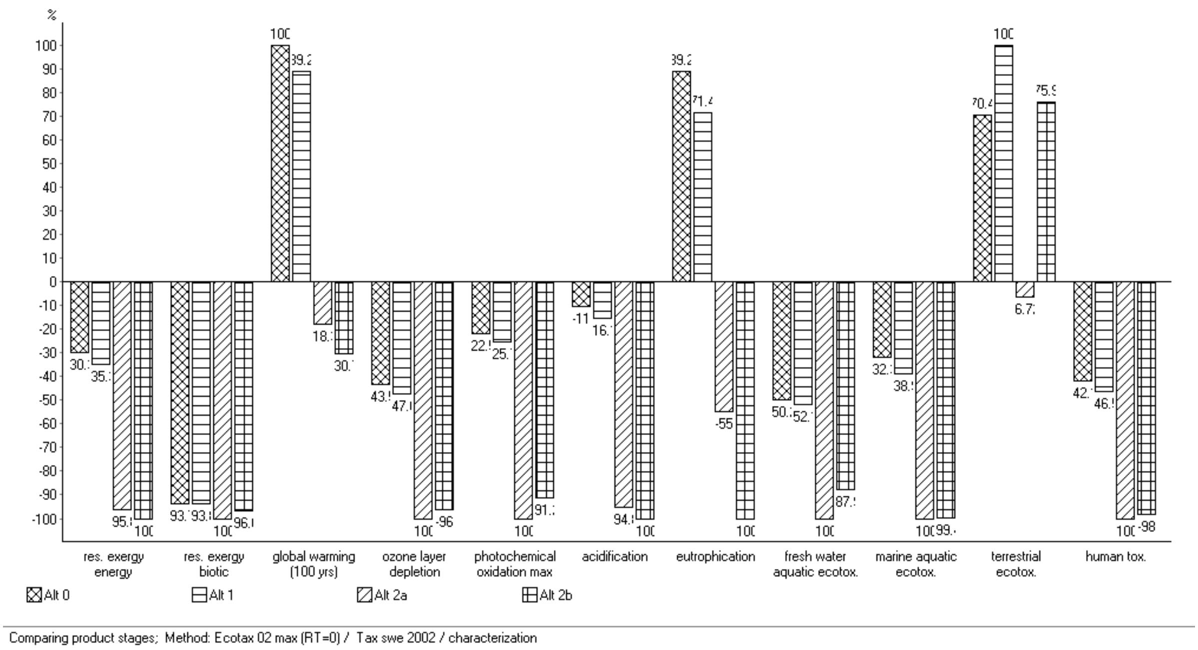

Figure 4. Relative comparison of environmental impact categories in all alternatives.

Waste treatment as a whole has a net negative environmental impact for most impact categories in the analysed waste management alternatives. Alt 1 leads to small relative improvement of environmental impact compared to Alt 0, except for terrestrial ecotoxicity. Alt 2a and Alt 2b lead to potentially much larger improvement compared to Alt 0. Most impact categories are reduced in Alt 2a and 2b, indicating the strong potential of the strategies represented by these alternatives to reduce environmental impact. Terrestrial ecotoxicity is however reduced in Alt 2a but not in Alt 2b.

Table 18. Net contributions to potential environmental impact, and relative change (%) compared to Alt 0.

Impact category	Units	Net potential environmental impact			
		Alt 0	Alt 1	Alt 2a	Alt 2b
Global warming	kg CO_2 eq.	1.65E9	1.48E9 (-10%)	-3.03E8 (-118%)	-5.07E8 (-131%)
Fresh water aquatic ecotox.	kg 1,4-DB eq.	-2.85E7	-2.96E7 (-4%)	-5.68E7 (-99%)	-4.99E7 (-75%)
Marine water aquatic ecotox.	kg 1,4-DB eq.	-2.93E10	-3.53E10 (-20%)	-9.07E10 (-210%)	-9.01E10 (-208%)
Terrestrial ecotoxicity	kg 1,4-DB eq.	1.18E7	1.67E7 (42%)	-1.13E6 (-110%)	1.27E7 (8%)
Human toxicity	kg 1,4-DB eq.	-1.92E9	-2.12E9 (-10%)	-4.56E9 (-138%)	-4.46E9 (-132%)
Acidification	kg SO_2 eq.	-1.2E6	-1.76E6 (-47%)	-1.03E7 (-758%)	-1.09E7 (-808%)
Eutrophication	kg PO_4^{3-} eq.	4.09E5	3.27E5 (-20%)	-2.52E5 (-162%)	-4.59E5 (-212%)
Ozone layer depletion	kg CFC-11 eq.	-56.9	-62.3 (-9%)	-131 (-130%)	-126 (-121%)
Photochemical oxidation	kg C_2H_2 eq.	-5.05E5	-5.78E5 (-14%)	-2.25E6 (-346%)	-2.05E6 (-306%)
Exergy (resource use)	MJ eq.	-1.64E10	-1.91E10 (-16%)	-5.19E10 (-216%)	-5.42E10 (-230%)
Exergy (biotic resource use)	MJ eq.	-9.01E10	-9.01E10 (0%)	-9.61E10 (-7%)	-9.28E10 (-3%)

Contribution by waste category

The contribution by waste category to each environmental impact category is important in understanding the role of different waste flows. Figure 5 displays the contribution by waste category in Alt 0, which is very similar to Alt 1 (not displayed). The correlation between waste amounts and net impact is even less apparent than for energy turn-over (Figure 3), although most impact categories are dominated by net reductions achieved by treatment of household waste. The explanation to this is the same as for the contribution by waste type to energy turnover. That is, relatively more resources are recovered from household waste. More net savings are also achieved per amount of resources recovered. Because of this, the contribution of household waste increases as resource recovery increases in Alt 2a and 2b and Figure 7).

(

Exergy, which is closely related to energy, shows a pattern similar to that of energy use in Figure 3. While global warming is also closely related to energy, it is not affected by non-fossil energy use, but on the other hand by incineration of waste plastics, and thus shows a different pattern. Other factors than waste amounts and energy turnover may be determining for the other impact categories.

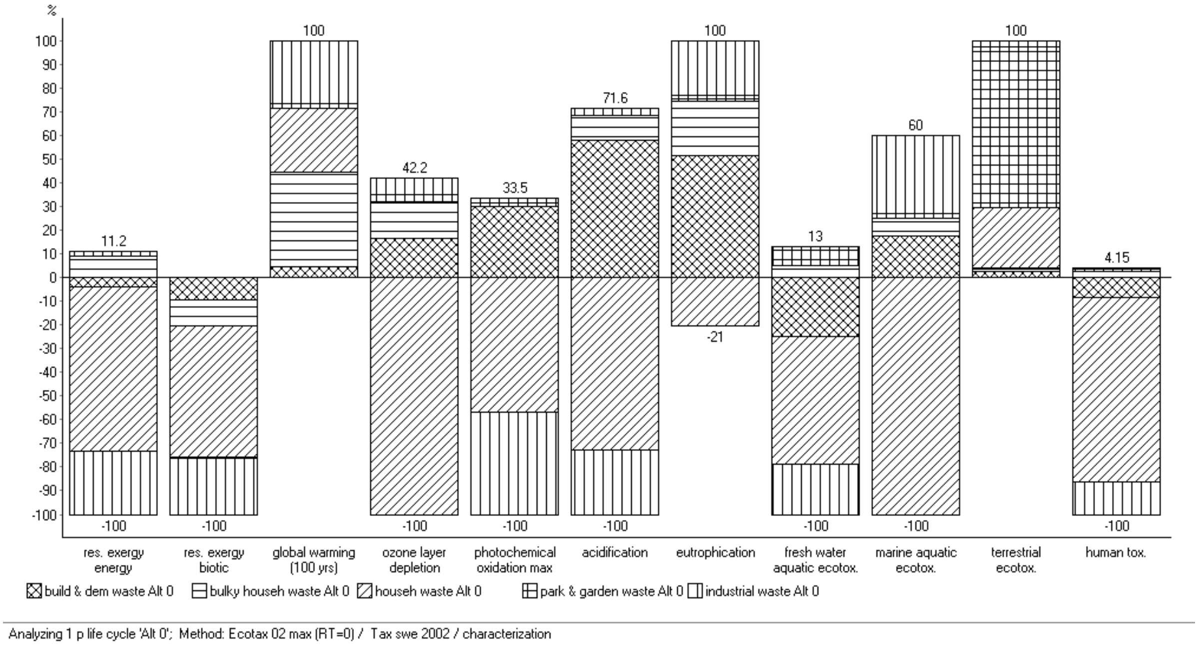

Figure 5. All impacts in Alt 0. Contribution by waste category.

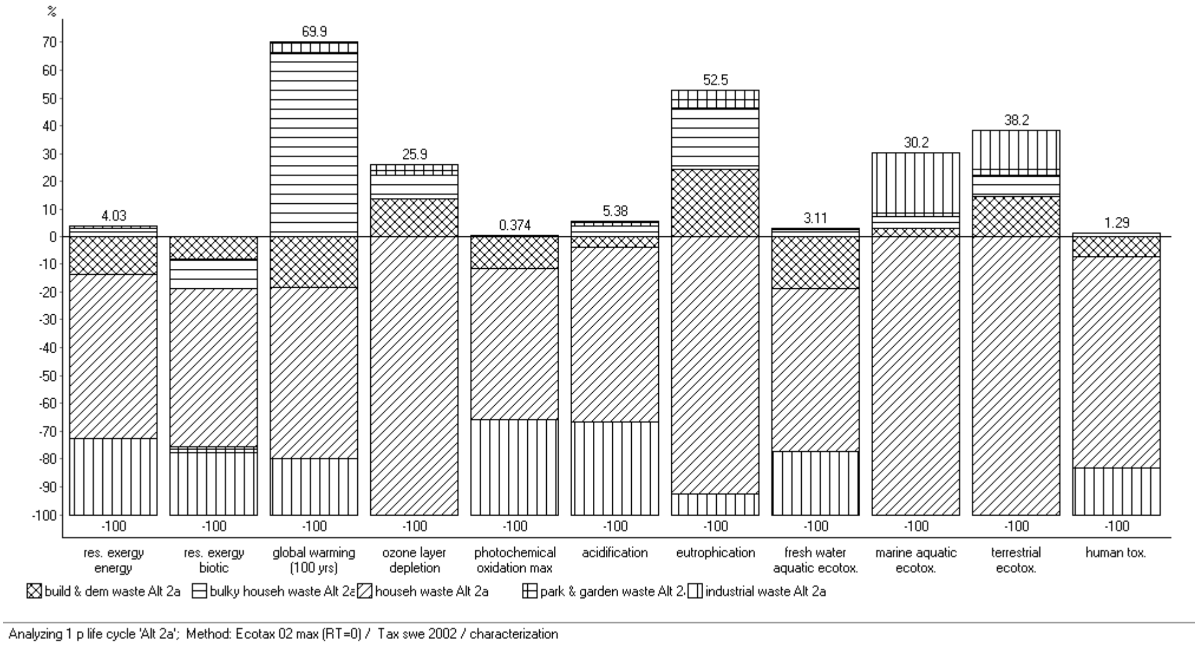

Figure 6. All impacts in Alt 2a. Contribution by waste category.

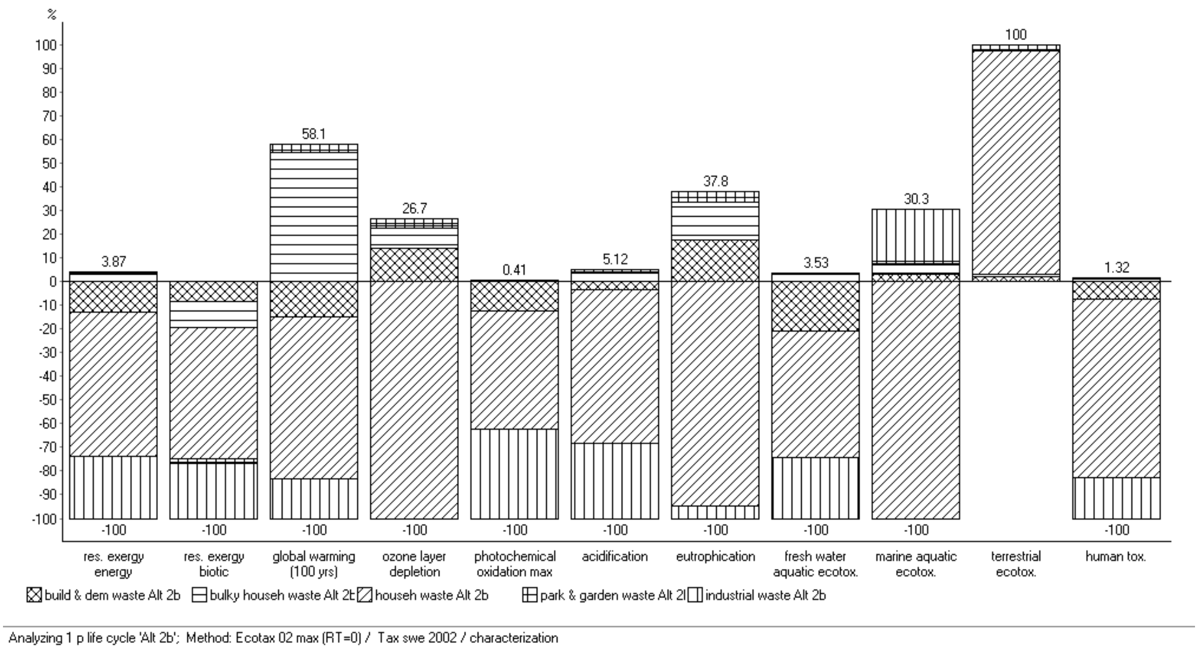

Figure 7. All impacts in Alt 2b. Contribution by waste category.

Global warming potential

Both Alt 0 and Alt 1 give net contributions to global warming, whereas Alt 2a and 2b have the potential to achieve net reductions in global warming potential.

In Alt 0 there is a net contribution to global warming potential, and handling of all waste categories contribute to this. The impact category is dominated by fossil CO2 emissions from incineration of household and industrial waste. Some savings are achieved by recycling of newspaper, steel scrap and aluminium, thereby avoiding fossil energy use.

Alt 1 achieves a small reduction in global warming potential compared to Alt 0, but as in Alt 0 all waste categories cause net contribution. The reduction is mainly attributable to increased recycling of steel scrap and newspaper, thereby avoiding fossil energy use, and to reduced waste incineration.

Both Alt 2a and Alt 2b reduce the global warming potential compared to Alt 0, and results in net savings. Handling of bulky household waste and park and yard waste cause net contribution to global warming, while the other waste categories achieve net savings. The largest savings of greenhouse gas emissions are due to recycling of PE, newspaper, and steel.

It may be worth noting that landfills, which are often pointed out as important sources of greenhouse gas emissions from waste management, do not stand out in the results. This is explained by the underlying assumptions, that source separation of combustible waste is efficient and that the ban on landfilling of source separated combustible waste is complied with. Thus, very little biodegradable material is landfilled, and little methane is generated.

Fresh water aquatic ecotoxicity

All alternatives achieve net savings of fresh water aquatic ecotoxicity potential. The net savings in Alt 0 and Alt 1 are of the same magnitude, but are only about half of the savings in Alt 2a and Alt 2b.

In Alt 0, there is a net saving of fresh water aquatic ecotoxicity potential. Handling of bulky household waste and park and yard waste causes net contributions, while all other waste categories achieve net savings. These savings are dominated by avoiding heavy metal emissions through steel recycling, and emissions of PAH and heavy metals avoided by aluminium recycling. Because the content of heavy metals is much higher in organic fertiliser than in artificial fertiliser, biological treatment if organic waste causes a net contribution.

Fresh water aquatic ecotoxicity potential does not change in Alt 1 compared to Alt 0, and the contributions or savings by different waste categories are about the same as in Alt 0. Increased recycling of steel and aluminium results in a small improvement compared to Alt 0, but this is offset by increased biological treatment that leads to increased contributions to fresh water aquatic ecotoxicity potential.

In Alt 2a the savings of fresh water aquatic ecotoxicity potential are about twice those achieved by Alt 0. Still, handling of bulky household waste and park and yard waste causes net contributions, while the other waste categories achieve net savings. Important savings compared to Alt 0 are achieved because there is no biological treatment of organic waste in this alternative, and consequently no spreading of heavy metals in organic fertilisers. The improvement compared to Alt 0 is mainly attributable to increased steel and aluminium recycling.

The savings of fresh water aquatic ecotoxicity potential are slightly less in Alt 2b than in Alt 2a, although with about the same contributions or savings by different waste categories. The only difference compared to Alt 2a lies in the handling of organic household waste, which is anaerobically digested, not incinerated. Because of this, heavy metals in organic fertiliser are spread in the environment.

Marine water aquatic ecotoxicity

All alternatives achieve net savings of marine water aquatic ecotoxicity potential. The potential savings in Alt 2a and Alt 2b are about twice as large as in Alt 0 and Alt 1.

In Alt 0 there is net saving of marine water aquatic ecotoxicity potential. Handling of household waste achieves net savings, while all other waste categories cause net contributions. The savings are dominated entirely by emissions that are avoided when aluminium is recycled, mainly air emissions of HF. Net contributions are caused by air emissions of HF from glass recycling, and air emissions of HF from steel recycling.

Alt 1 achieves a slight reduction in marine water aquatic ecotoxicity potential compared to Alt 0, i.e. larger savings. As in Alt 0, handling of household waste alone is responsible for these savings. The improvement is largely due to increased recycling of aluminium.

In Alt 2a the savings of marine water aquatic ecotoxicity potential are about twice those achieved by Alt 0. In this alternative, only handling of household waste and park and yard waste achieve net savings, while all other waste categories cause net contributions to this impact category. As in Alt 0, the largest savings are due to aluminium recycling.

There are no major differences between Alt 2b and Alt 2a.

Terrestrial ecotoxicity

Alt 0, Alt 1, and Alt 2b all cause net contribution to terrestrial ecotoxicity potential of about the same magnitude. Alt 2a differs by achieving net savings of this impact category.

In Alt 0, handling of all waste categories cause net contributions to terrestrial ecotoxicity potential. The impact is entirely dominated by heavy metals in residue from biological treatment, when the residue is used as fertiliser.

In Alt 1 the contribution to terrestrial ecotoxicity potential increases compared to Alt 0, and handling of all waste categories cause net contribution. As in Alt 0, the impact category is entirely dominated by heavy metals in residue from biological treatment that is used as fertiliser.

In contrast to all other alternatives, Alt 2a achieves net savings of terrestrial ecotoxicity potential. In this alternative, handling of household waste achieves net savings, while all other waste categories cause net contributions. This improvement compared to all other alternatives is mainly a result of the treatment of organic waste, which in Alt 2a is incinerated to a large extent, but biologically treated in all other alternatives. Consequently, there is no organic residue to be used as fertiliser in Alt 2a. As organic fertiliser contains more pollutants than artificial fertiliser, there is less contribution to terrestrial ecotoxicity potential. Increased recycling of newspaper, aluminium and corrugated cardboard also brings a slight improvement by avoiding air emissions of heavy metals.

The contribution to terrestrial ecotoxicity potential in Alt 2b is about the same as in Alt 0, and handling of all waste categories cause net contribution. As in Alt 0, the impact category is entirely dominated by heavy metals in residue from biological treatment that is used as fertiliser. However, this negative effect is limited by the fact that park and yard waste is incinerated, not composted. As in Alt 2a, increased recycling of newspaper, aluminium, and corrugated cardboard also brings a slight improvement compared to Alt 0.

Human toxicity

All alternatives achieve net savings of human toxicity potential. The savings in Alt 2a and Alt 2b are about twice as large as in Alt 0 and Alt 1.

Alt 0 achieves net savings of human toxicity potential. Handling of household, building and demolition, and industrial waste achieve net savings, while the other waste categories cause net contribution. The savings are dominated by hydrocarbon emissions that are avoided due to recycling of aluminium, PET, and steel. Some contribution to human toxicity potential comes from hydrocarbon emissions from corrugated cardboard recycling.

Alt 1 achieves net savings of human toxicity potential of about the same magnitude as Alt 0. As in Alt 0, handling of household, building and demolition, and industrial waste achieve net savings, while the other waste categories cause net contribution. Increased recycling of aluminium, PET, and steel achieves a small improvement compared to Alt 0.

The savings in Alt 2 human toxicity potential are more than twice as large as in Alt 0. Still, only handling of household, building and demolition, and industrial wastes achieve net savings, while the other waste categories cause net contribution.

There are no major differences in human toxicity potential between Alt 2b and Alt 2a.

Acidification

Savings of acidification potential are achieved by all alternatives. Alt 1 achieves some reduction compared to Alt 0, while Alt 2a and 2b have the potential to reduce the impact further by almost an order of magnitude.

Alt 0 achieves net savings of acidification potential. Handling of household and industrial waste results in net savings, while all other waste categories cause net contributions to this impact category. The most important savings are avoided SO_x and NO_x emissions from newspaper, aluminium, and steel recycling. Large negative contribution is caused by SO_x from landfilling of plasterboard. These emissions originate from sulphate in plaster. The model assumes that sulphate is to some extent microbiologically reduced to H_2S, which is collected in landfill gas and combusted to SOx. This process may not be representative of a landfill containing mainly inorganic waste (c.f. Chapter 5 "LCA").

Savings of acidification potential are larger in Alt 1 than in Alt 0. As in Alt 0, handling of household and industrial waste results in net savings, while all other waste categories cause net contributions to this impact category. The improvement is mainly due to reduced landfilling of plasterboard, but also due to increased recycling of newspaper, aluminium, and steel.

Alt 2a achieves net savings of acidification potential that are almost an order of magnitude larger than in Alt 0. Handling of bulky household waste and park and yard waste cause net contributions, while all other waste categories achieve net savings. The reduced impact compared to Alt 0 is mainly attributable to increased recycling of newspaper, cardboard, corrugated cardboard, PE, and steel.

Savings of acidification potential are of the same magnitude in Alt 2b as in Alt 2a.

Eutrophication

Alt 0 and Alt 1 cause net contributions to eutrophication potential, while Alt 2a and 2b achieve net savings of about the same order of magnitude.

Alt 0 causes net contribution to eutrophication potential. Only handling of household waste achieves a net saving, while all other waste categories cause net contribution. The main contribution is by NOx from combustion processes, both from waste incineration, transports, and work equipment at landfills. Recycling of newspaper, cardboard, PE, aluminium, and steel also achieve some savings. This is mainly due to avoided NOx emissions, but in the case of newspaper, cardboard, and aluminium also avoided water emissions of nitrogen, phosphorous, and organic material.

A small reduction of eutrophication potential is achieved by Alt 1 compared to Alt 0. Still, only household waste achieves net savings. The improvement is achieved mainly by increased recycling of newspaper, cardboard, PE, aluminium, and steel.

Eutrophication potential is significantly reduced in Alt 2a compared to Alt 0, and net savings are achieved in this alternative. Still only handling of household waste and industrial waste achieves net savings, while the other waste categories cause net contribution. The improvement is achieved primarily by reduced landfilling and incineration, and by increased recycling of newspaper, cardboard, PE, aluminium, and steel

Eutrophication potential is reduced somewhat more in Alt 2b than in Alt 2b. The improvement is due to the treatment of organic household waste. Incineration of this fraction in Alt 2a gives a net contribution of NO_x emissions and nutrient leaching from landfilling of ashes, while anaerobic digestion gives net savings when the biogas is used to replace diesel fuel.

It may be worth noting that the model assumes no difference in eutrophication potential between artificial fertiliser and organic fertiliser from composting or anaerobic digestion. If a difference was assumed, this would have influence on the difference between Alt 2a and Alt 2b.

Ozone layer depletion

All alternatives show net savings of ozone layer depletion potential. Alt 2a and Alt 2b achieve about twice as large savings as Alt 0 and Alt 1.

Alt 0 achieves a small net saving of ozone layer depletion potential. Handling of household waste alone achieves a net saving, while all other waste categories cause net contribution. Savings are entirely dominated by aluminium recycling, by which HALON-1301 and hard CFC emissions from virgin aluminium production are avoided. Some savings are also achieved by recycling of steel and cardboard.

There are no major differences in ozone layer depletion potential in Alt 1 compared to Alt 0. Slightly increased recycling results slightly increased savings.

Savings in ozone layer depletion potential are about twice as high in Alt 2a compared to Alt 0. Still, only handling of household waste achieves net savings, while all other waste categories cause net contribution. The main reductions are due to increased recycling of aluminium and cardboard.

There are no major differences in ozone layer depletion potential in Alt 2b compared to Alt 2a.

It is important to note that the limited emissions of ozone layer depleting substances in this model may be the result of data gaps. Refrigerators, freezers and other products containing CFC were not identified as specific product groups in the inventory. Thus, the specific handling of these products, where careful recycling can make a large difference for the emissions, was not modelled. Thus, total emissions of CFC are probably higher in all alternatives, and higher recycling rates probably result in reduced emissions.

Photochemical oxidation

All alternatives achieve net savings of photochemical oxidation potential. The potential savings in Alt 2a and Alt 2b are two to three times larger than in Alt 0 and Alt 1.

Alt 0 achieves net savings of photochemical oxidation potential. Handling of household, bulky household, and industrial waste achieves net savings, while the other waste categories cause net contribution. Main savings are achieved by recycling of PE, newspaper, steel, aluminium, and PET, which leads to avoided emissions of CO, SO_x, VOC, and NO_x. Large contribution comes from VOC and NO_x from landfilling of plasterboard and concrete.

Alt 1 achieves a slight reduction of photochemical oxidation potential compared to Alt 0. As in Alt 0, handling of household, bulky household, and industrial waste achieves net savings, while the other waste categories cause net contribution. Reduced landfilling of plasterboard and concrete, and increased recycling of PE and steel mainly achieve the improvement.

Savings of photochemical oxidation potential in Alt 2a are more than three times as large as in Alt 0. Handling of household, bulky household, industrial, and building and demolition waste achieves net savings; only park and yard waste causes net contribution. The improvements are achieved mainly by reduced landfilling of plasterboard and concrete and increased recycling of PE.

Somewhat less savings of photochemical oxidation potential are achieved in Alt 2b than in Alt 2a. This is because the contribution from anaerobic digestion of organic household waste is higher than from incineration of this waste fraction. The contribution from anaerobic digestion is dominated by VOC emissions.

Exergy

Net savings of exergy are achieved by all alternatives. When exergy of all resources is considered, the savings in Alt 2a and Alt 2b are about twice the savings in Alt 0 and Alt 1. When only biotic resources are considered, there are no major differences between the alternatives, and this category is not further commented.

Alt 0 achieves net savings of exergy use. Handling of building and demolition, household, and industrial waste achieve net savings, while the other waste categories cause net contribution. The major savings are achieved by recycling of newspaper, steel, aluminium and PE.

A slight improvement is achieved in Alt 1 compared to Alt 0, but the contribution by different waste categories is about the same as in Alt 0. The improvements are achieved by increased recycling of newspaper, steel, aluminium and PE.

The savings of exergy in Alt 2a are about twice those achieved in Alt 0. Handling of building and demolition, household, and industrial waste achieve net savings, while the other waste categories cause net contribution. The improvements are mainly due to increased recycling of PE, newspaper, and steel.

There are no major differences between Alt 2a and Alt 2b.

7.2 Short vs. long time horizon for landfill emissions

In the base case, the model only includes emissions from landfills and ashes from incineration of forest residue for the surveyable time period, which corresponds to about 100 years. With this temporal cut-off, large amounts of heavy metals (and some other substances) are not released and thus cause no environmental impact. To assess the importance of these substances as a potential future emission source, the model was also run including the remaining time period. This is a worst-case scenario, which allows all substances in landfills and ashes to be released to the environment. It is nevertheless an important scenario to consider. Although the fate and impact of these pollutants are not very well known, the possibility can not be disregarded that they will actually be released and cause harm to future generations.

Figure 8 shows the relative comparison of environmental impact categories when the remaining time period is included. When compared to Figure 4, in which the remaining time period is not included, the ranking between alternatives is changed only for eutrophication, fresh water aquatic ecotoxicity, and terrestrial ecotoxicity.

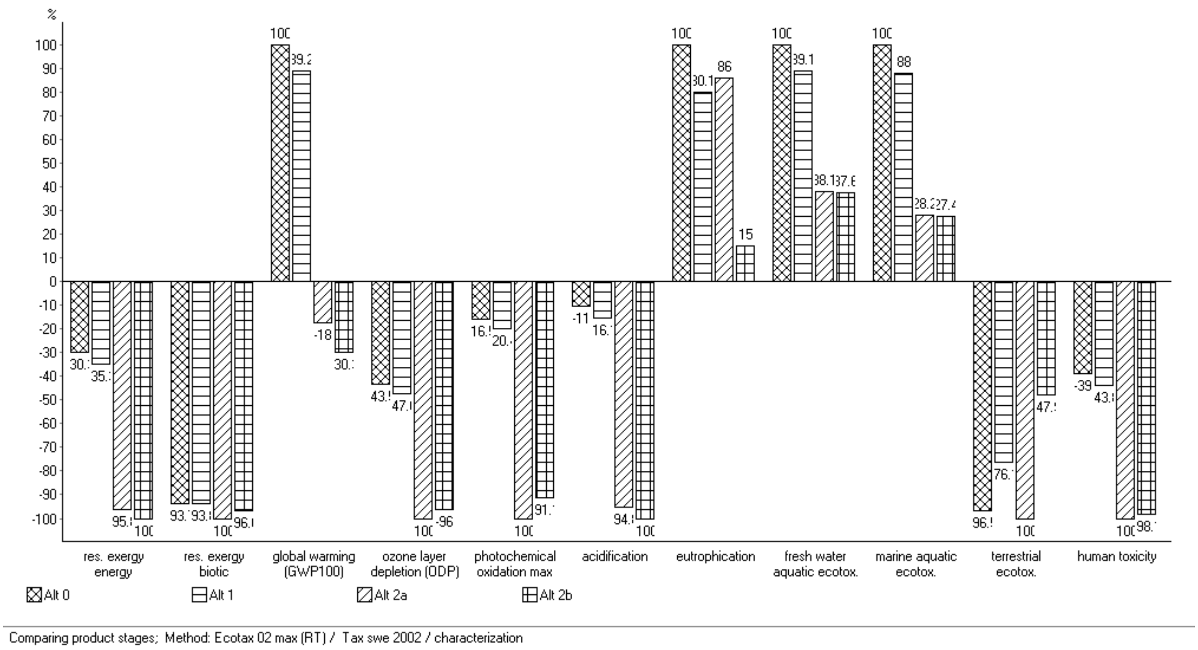

Figure 8. Relative comparisons of environmental impact categories in all alternatives when the remaining time period is included for landfills and incineration ashes.

However, the absolute values for several impact categories are strongly affected by including the remaining time period, as are seen in Table 19. Long-term leaching of nutrients from slag and ash landfills leads to much higher eutrophication potential, so that all alternatives in this case cause a net contribution. Long-term leaching of heavy metals and organic pollutants from slag and ash landfills leads to high increase of fresh water and marine water aquatic ecotoxicity, so that all alternatives increase from net savings to net contribution. Long-term leaching of heavy metals and organic pollutants from slag and ash landfills also affects human toxicity, but to a much lesser extent. Terrestrial ecotoxicity on the other hand is reduced when the remaining time perspective is included. When long-term emissions are included, this results in avoided leaching of heavy metals from ashes from burning forest residue for district heating.

Table 19. Net contributions to environmental impact categories when the remaining time period is included, and relative change (%) compared to the base case, in which the remaining time period is not included.

Impact category	Units	Net potential environmental impact			
		Alt 0	Alt 1	Alt 2a	Alt 2b
Global warming	kg CO_2 eq.	1.67E9 (1%)	1.49E9 (1%)	-3.01E8 (1%)	-5.05E8 (0)
Fresh water aquatic ecotox.	kg 1,4-DB eq.	1.66E9 (5925%)	1.48E9 (5100%)	6.33E8 (1214%)	6.24E8 (1351%)
Marine water aquatic ecotox.	kg 1,4-DB eq.	4.79E11 (1735%)	4.22E11 (1295%)	1.35E11 (249%)	1.31E11 (245%)
Terrestrial ecotoxicity	kg 1,4-DB eq.	-3.05E7 (-358%)	-2.4E7 (-244%)	-3.16E7 (-2696%)	-1.51E7 (-219%)
Human toxicity	kg 1,4-DB eq.	-1.73E9 (10%)	-1.94E9 (8%)	-4.44E9 (3%)	-4.35E9 (2%)
Acidification	kg SO_2 eq.	-1.2E6 (0%)	-1.75E6 (1%)	-1.03E7 (0%)	-1.09E7 (0%)
Eutrophication	kg PO_4^{3-} eq.	3.65E6 (792%)	2.93E6 (796%)	3.14E6 (1346%)	5.49E5 (220%)
Ozone layer depletion	kg CFC-11 eq.	-56.9 (0%)	-62.3 (0%)	-131 (0%)	-126 (0%)
Photochemical oxidation	kg C_2H_2 eq.	-3.67E5 (27%)	-4.55E5 (21%)	-2.23E6 (1%)	-2.03E6 (1%)
Exergy (energy resources)	MJ eq.	-1.64E10 (0%)	-1.91E10 (0%)	-5.19E10 (0%)	-5.42E10 (0%)
Exergy (biotic resources)	MJ eq.	-9.01E10 (0%)	-9.01E10 (0%)	-9.61E10 (0%)	-9.28E10 (0%)

7.3 Conclusions of the traditional LCA

Without going into details of the different waste categories and treatment options, some conclusions can be drawn from Figure 4. The effect of the proposed waste incineration tax, as represented by Alt 1, reduces several environmental impact categories included in this LCA, some remain unchanged, while only terrestrial ecotoxicity appears to increase. This result holds also in the case of including long-terms impacts of leaching from landfills and ash from incineration of forest residue. But it is also clear from Figure 4 that the improvements achieved as a result of the waste incineration tax are relatively small. By analysing the different impact categories in detail, it is also seen that the largest improvements are achieved by reducing landfilling to the benefit of increased recycling. This shift in waste flows is caused by the landfill tax raise that is done in parallel with introducing the incineration tax. Thus, the improvements are more of an indirect result of the waste incineration tax.

The two more explorative strategies represented by Alt 2a and Alt 2b have the potential to achieve more far-reaching environmental improvements than the waste incineration tax as represented by Alt 1.

8 Results of the site-adjusted LCA

This chapter analyses the site-adjusted LCA results, which compare emissions of NO_x, SO_x, and particulates, and the corresponding environmental impact categories, for two different regions in Sweden, Skåne and Norrland. The model is based on the traditional LCA model, but with some important differences. First, the model applies other transport distances than were used in the traditional LCA. A set of shorter transport distances are used to represent Skåne, while a set of longer transport distances are used to represent Norrland. This is the only actual difference introduced in the model of the waste management system that results in different emissions from the two regions. Second, different characterisation factors are applied depending on whether an emission occurs locally or at an undefined location, and whether an emission occurs at ground level or from a high stack. See Chapter 5.2 "Site-adjusted LCA" for detailed description of the site-adjusted LCA model.

Emissions of NO_x, SO_x, and particulates are higher in the sparsely populated Norrland than in the densely populated Skåne (Figure 9, Figure 10, and Figure 11). This, of course, is because of the much longer transport distances in Norrland. For SO_x and particulates, the net emissions are negative in both regions, whereas for NO_x, the net emissions are negative in Skåne but positive in Norrland. However, the ranking between the different alternatives remains the same in both regions, with Alt 2a and Alt 2b having lower potential environmental impact than Alt 0 and Alt 1.

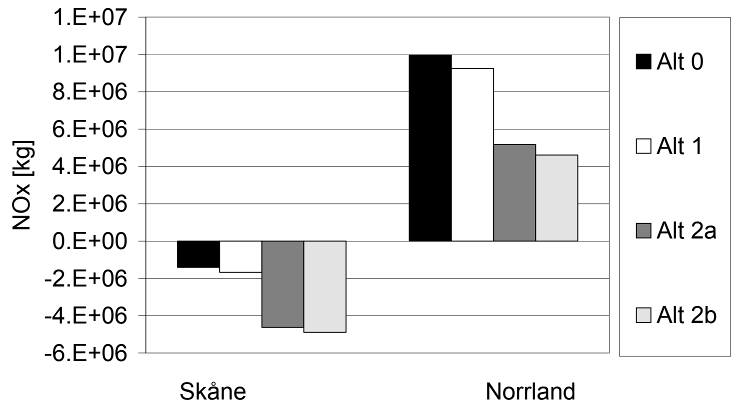

Figure 9. NO_x emissions (kg) in Skåne and Norrland.

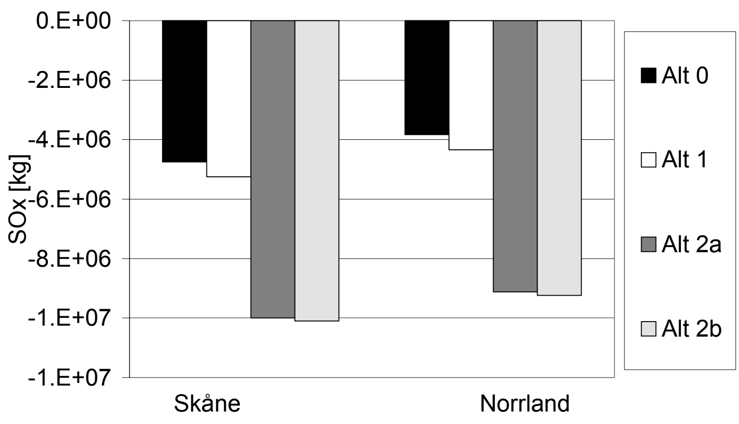

Figure 10. SO_x emissions (kg) in Skåne and Norrland.

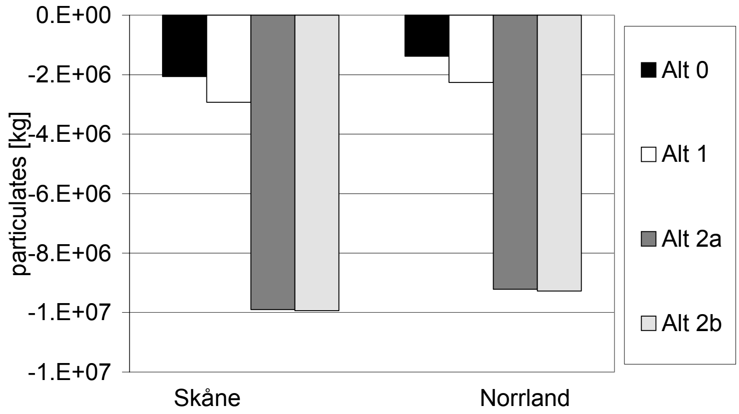

Figure 11. Particulates (kg) in Skåne and Norrland.

Despite the fact that the modelled emissions are higher in Norrland than in Skåne, the net human health impact is actually lower in Norrland (Figure 12, Figure 13, and Figure 14). This is because the site-adjusted human health characterisation factors are lower for emissions in Norrland, only about 6 to 18% of those for Skåne (cf. Chapter 5.2 "Site-adjusted LCA"). As a consequence, the health impact of all emissions occurring in Norrland is given a lower value, leading to a net reduction of the impact as a whole.

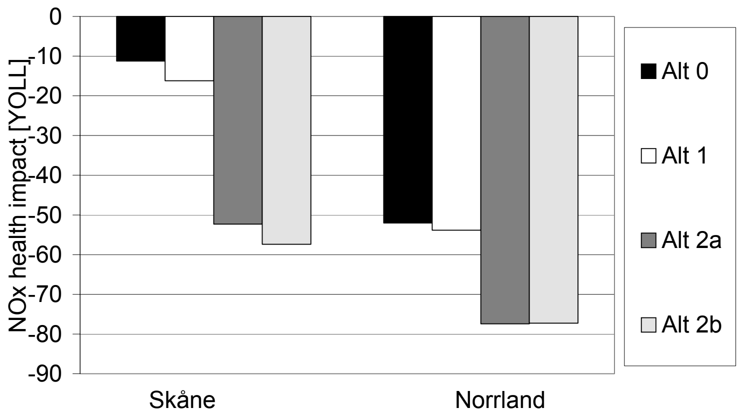

Figure 12. NO_x health impact (YOLL) in Skåne and Norrland.

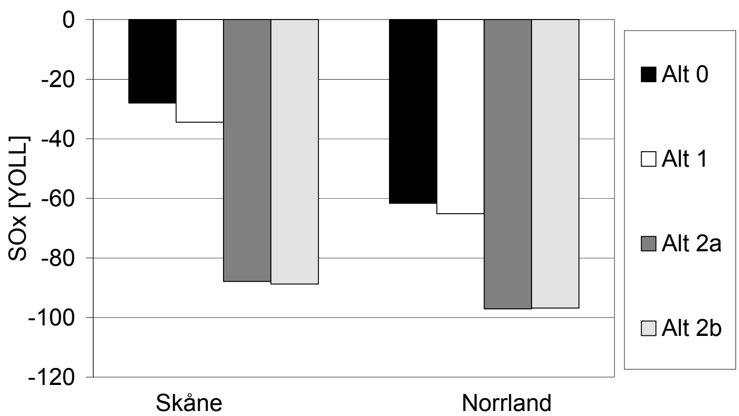

Figure 13. SO_x health impact (YOLL) in Skåne and Norrland.

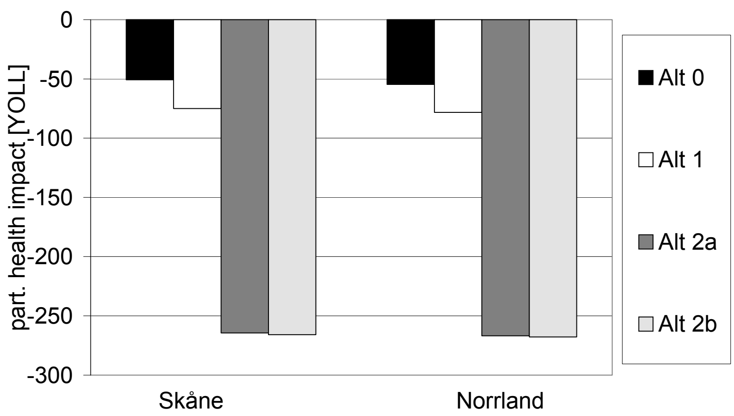

Figure 14. Particulates health impact (YOLL) in Skåne and Norrland.

Ecosystem impact correlates better with emissions, that is, the impact is higher in Norrland where emissions are higher, than in Skåne.

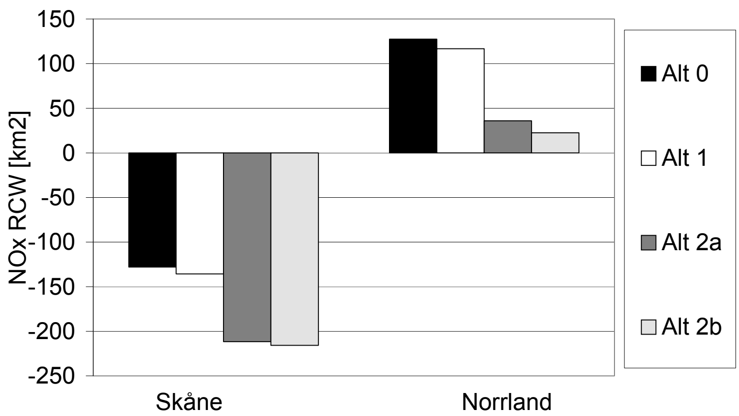

Figure 15. NO_x ecosystem impact (RCW) in Skåne and Norrland.

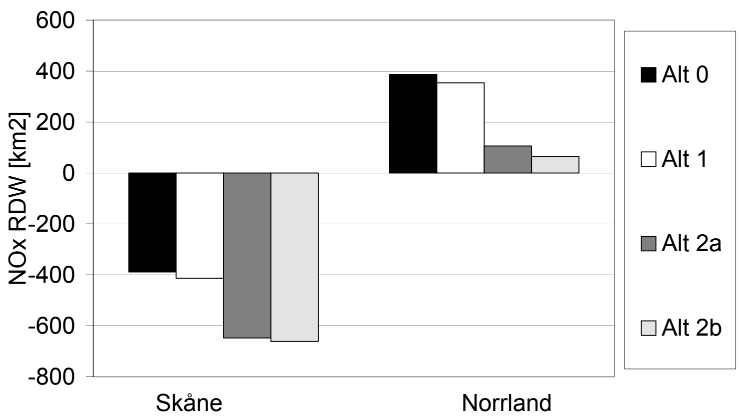

Figure 16. NO_x ecosystem impact (RDW) in Skåne and Norrland.

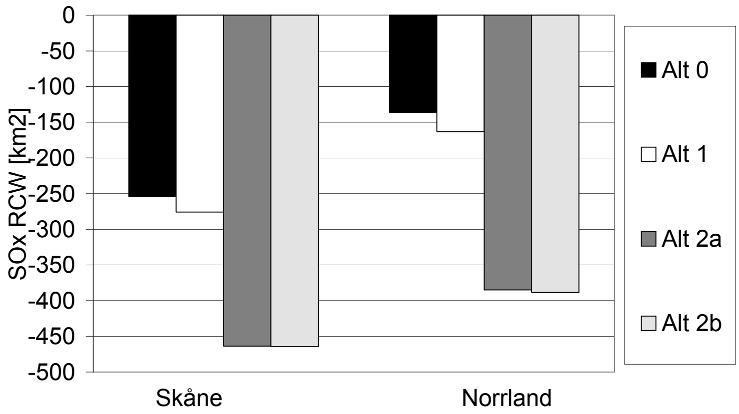

Figure 17. SO_x ecosystem impact (RCW) in Skåne and Norrland.

8.1 Conclusions of the site-adjusted LCA

There are two kinds of site-related differences between the two regions that are quite significant. First, the human health impact per amount of emitted substance is lower in Norrland than in Skåne. Second, the total emissions are higher in Norrland than in Skåne. Despite this, it is interesting to note that the results regarding the ranking between alternatives remain the same for the two regions, and that this ranking is the same as in the traditional, not site-adjusted LCA.

However, as a measure of human health impact, emissions of SO_x, NO_x, and particulates is not entirely representative. In the traditional LCA, this impact category is dominated by emissions of other substances, mainly hydrocarbons (C_xH_y).

9 Results of the LCA valuation

Results for the different valuation methods are showed for the four different waste alternative scenarios in this section. The base scenario is Alt 0 and most calculations are made with Ecotax02, meaning that EPS and Eco-indicator 99 are added as references to the discussion.

Presentations of results are mainly structured either by the five different waste categories: *Household waste, Industrial waste, Building and demolition waste, Bulky household waste and Park and garden waste* or structured by impact categories. The evaluation focus is embedded in the question: What are the largest environmentally related problems with Alt 0? This also includes sub questions like, what impact categories play the most important role? and what kinds of interventions or waste categories are important to that impact category?

Describing the different scenarios either through a single score perspective (i.e. summary of all scores of all categories compared) or comparing one impact / waste category to another are the two possibilities given by weighted values. This enhances the possibilities to compare the results to each other that would not be possible without weighting methods.

Weighted results of impact categories for each alternative are listed in Table 20. The table provides here relevant information from three perspectives. The first vital information from the table is the internal ranking of all impact categories within each alternative. This makes it possible to see which impact category is the largest. In this case marine aquatic ecotox is for Alt 0 ranked as number one followed by res. exergy biotic and fresh water aquatic ecotox. At the end of the table ozone layer depletion is positioned. Thus in terms of impact categories that matters the most, we find that; ecotox categories, resources exergy categories and global warming category are the most important in Alt 0. We can also see that Acidification and Eutrophication are of less importance. The second thing that the table shows is that the ranking order between the alternatives is different. In this case the photochemical ox is ranked higher than global warming for Alt 2a and Alt 2b than it was for Alt 0 and Alt1. The third thing that can be read from the table are the single score values for the alternatives. This means that a direct comparison can be performed between the alternatives and their impact categories, and a ranking is possible between them showing that all alternatives give a net saving compared to Alt 0. Furthermore it can be noted that Alt 2a and Alt 2b gives significant savings in relation to Alt 0 and Alt 1.

Table 20. Weighted contributions by impact category (Ecotax02 max RT=0).

Impact category	Alt 0	Alt 1	Alt 2a	Alt 2b
res. exergy energy	-2,77E+09	-3,23E+09	-8,77E+09	-9,15E+09
res. exergy biotic	-6,22E+09	-6,22E+09	-6,63E+09	-6,40E+09
global warming (100 yrs)	1,04E+09	9,30E+08	-1,91E+08	-3,20E+08
ozone layer depletion	-6,82E+04	-7,48E+04	-1,57E+05	-1,51E+05
photochemical oxidation max	-2,42E+08	-2,78E+08	-1,08E+09	-9,84E+08
acidification	-2,16E+07	-3,16E+07	-1,86E+08	-1,96E+08
eutrophication	1,17E+07	9,35E+06	-7,21E+06	-1,31E+07
fresh water aquatic ecotox.	-3,55E+09	-3,68E+09	-7,07E+09	-6,21E+09
marine aquatic ecotox.	-1,78E+10	-2,14E+10	-5,50E+10	-5,46E+10
terrestrial ecotox.	2,08E+09	2,95E+09	-1,99E+08	2,24E+09
human tox.	-2,88E+09	-3,18E+09	-6,84E+09	-6,70E+09
Total	**-3,03E+10**	**-3,41E+10**	**-8,59E+10**	**-8,24E+10**

Base scenario (Alt 0)

As shown in Figure 18, savings in total environmental impacts are made only for *household waste* in Alt 0 in this case. This is primarily due to recycling of aluminium and savings in making new aluminium (data not shown). *Industrial waste* stands for the largest part of the remaining four waste types. *Building and demolition waste* contributes with costs from especially two particular processes: land filling of concrete and incineration of wood. The last two waste types *Bulky household waste* and *Park and garden waste* are building their cost on incineration of other combustible waste and central composting of Park & garden waste respectively.

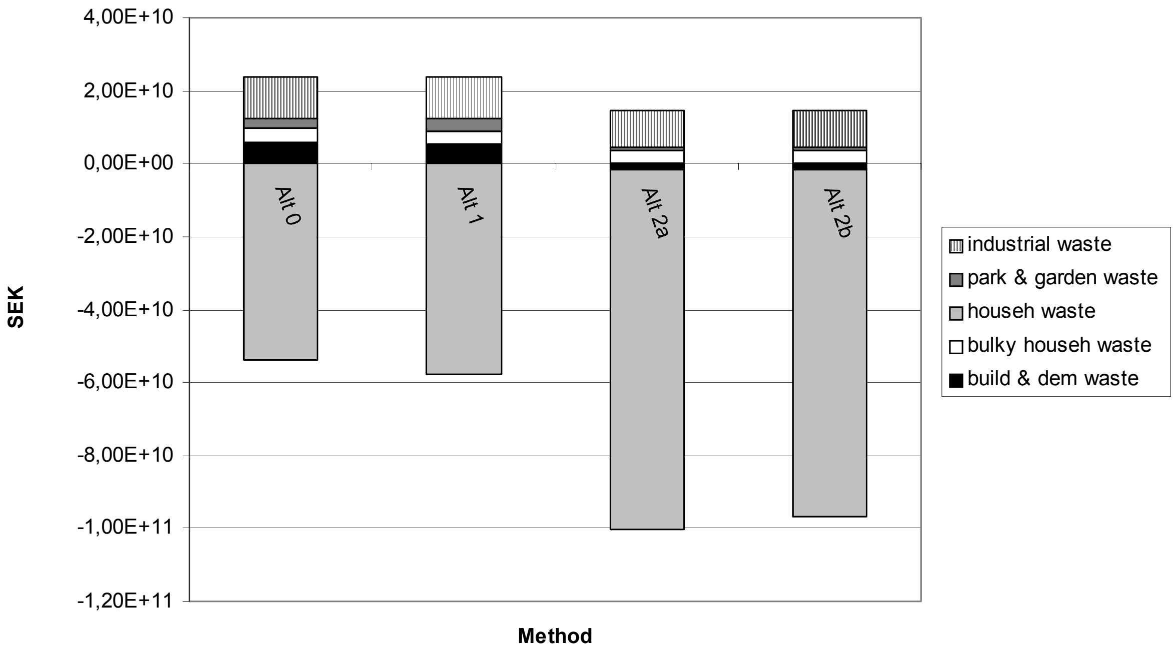

Figure 18. Single score contribution of waste category Ecotax02 max (RT=0) to all alternatives.

Interventions with highest numerical values (largest seven positive and seven negative values for each method) for Alt 0 are listed in Table 21. Interesting information to extract from table is what kind of substance matters? How much does remaining time (RT) influence on the results? Differences between weighting methods?

In Ecotax02 max (RT=0) HF gives the largest net cost, followed by Cr, CO_2, Ni and use of crude oil. This goes hand in hand with the impact categories already pointed out as important in the previous section. Notable is also that these are allocated in many different compartments like air, soil, water and Raw materials. Unlike this Ecotax02 max (RT) put three substances released during the remaining time (RT) (i.e. after the surveyable time period) from water compartment in the top, Cu RT, Ni RT and Zn RT. HF follows these and CO_2 is not found within the seven largest. In this method six substances of seven in the top are RT substances. In contrast to the max method with RT=0 in Ecotax02, the Ecotax min, RT=0 method put HF outside the seven highest ranked and Cr, CO_2 plus CxHy_te are among the three in the top. Ecotax02 min RT does not show similar results and Ecotax02 max RT.

When looking at the results from method EPS 2000 RT=0 it becomes quit clear that this is a different weighting method comparing to Ecotax02. Only air emissions and use of raw material are highly ranked and except for Pb there are no metals within this table. The highest ranked substance due to negative environmental effect is CO_2 (fossil) followed by natural gas ETH and crude oil. Finally the weighting method Eco-indicator 99 H shows a wider diversity in compartments than EPS 2000; substances in water, raw materials, soil and air are represented in the table. Cr RT, Ni RT and crude oil are found in top, CO_2 are not found within the seven largest.

Table 21. Largest interventions listed for all valuation methods for Alt 0.

Substance	Compartm	max (RT=0)	max (RT)	min (RT=0)	min (RT)	EPS 2000 RT=0	Eco-ind 99 RT
HF	Air	1,67E+10	1,67E+10	3,48E+06	3,48E+06	1,39E+03	-
Cr	Soil	2,09E+09	2,09E+09	2,08E+09	2,08E+09	-	1,04E+07
CO2 (fossil)	Air	1,35E+09	1,35E+09	1,35E+09	1,35E+09	2,35E+08	8,82E+06
Ni	Soil	1,09E+09	1,09E+09	1,65E+08	1,65E+08	-	7,15E+05
crude oil	Raw	9,97E+08	9,97E+08	0	0	7,11E+07	2,96E+07
Cu ST	Water	9,03E+08	9,03E+08	2,28E+08	2,28E+08	-	3,59E+04
Ni ST	Water	7,97E+08	7,97E+08	8,98E+07	8,98E+07	-	2,85E+05
natural gas ETH	Raw	6,88E+08	6,88E+08	0	0	8,22E+07	1,96E+07
CxHy_te	Air	6,28E+08	6,28E+08	6,08E+08	6,08E+08	-	5,31E+03
Cu	Soil	6,16E+08	6,16E+08	1,59E+08	1,59E+08	-	4,84E+05
natural gas (vol)	Raw	2,70E+08	2,70E+08	0	0	3,23E+07	8,06E+06
CxHy_fe	Air	2,09E+08	2,09E+08	2,02E+08	2,02E+08	-	1,76E+03
crude oil ETH	Raw	1,97E+08	1,97E+08	0	0	1,40E+07	6,06E+06
N2O	Air	1,47E+08	1,47E+08	1,47E+08	1,47E+08	2,88E+07	1,01E+06
Pb	Air	4,45E+07	4,45E+07	2,03E+07	2,03E+07	1,61E+07	1,09E+06
Cd RT	Water	0,00E+00	4,78E+09	0	1,37E+09	-	2,13E+07
Cr RT	Water	0,00E+00	2,37E+09	0	7,29E+08	-	2,86E+09
Cr RT	Soil	0,00E+00	-6,16E+09	0	-6,15E+09	-	-3,08E+07
Cu RT	Water	0,00E+00	2,47E+11	0	6,25E+10	-	9,80E+06
Ni RT	Water	0,00E+00	2,22E+11	0	2,50E+10	-	7,92E+07
Ni RT	Soil	0,00E+00	-2,69E+09	0	-4,10E+08	-	-1,77E+06
PAH RT	Water	0,00E+00	3,79E+09	0	2,06E+09	-	4,93E+04
Zn RT	Soil	0	-2,76E+09	0	-1,38E+09	-	-4,32E+07
Zn RT	Water	0,00E+00	4,33E+10	0	1,22E+10	-	2,76E+06
Dust	Air	-2,40E+06	-2,40E+06	-2,40E+06	-2,40E+06	-2,06E+07	-4,20E+06
iron (ore)	Raw	-5,37E+07	-5,37E+07	0	0	-4,15E+08	-7,83E+05
metallic ions	Water	-1,86E+08	-1,86E+08	-1,42E+07	-1,42E+07	-	-8,32E+07
natural gas	Raw	-3,24E+08	-3,24E+08	0	0	-3,87E+07	-9,23E+06
crude oil (42.6	Raw	-3,36E+08	-3,36E+08	0	0	-	-9,84E+06
crude oil	Raw	-3,53E+08	-3,53E+08	0	0	-2,51E+07	-1,05E+07
natural gas (35.2	Raw	-4,31E+08	-4,31E+08	0	0	-	-1,42E+07
CO2	Air	-4,51E+08	-4,51E+08	-4,51E+08	-4,51E+08	-7,81E+07	-2,93E+06
CxHy	Air	-4,85E+08	-4,85E+08	-4,69E+08	-4,69E+08	-	-4,10E+03
PAH's	Air	-7,80E+08	-7,80E+08	-7,68E+08	-7,68E+08	-5,70E+07	-2,95E+03
CxHy (non	Air	-9,95E+08	-9,95E+08	-9,63E+08	-9,63E+08	-	-8,41E+03
Hydrocarbons	Air	-1,30E+09	-1,30E+09	-1,25E+09	-1,25E+09	-	-1,10E+04
PAH's	Water	-1,46E+09	-1,46E+09	-7,95E+08	-7,95E+08	-	-1,90E+04
V	Water	-1,83E+09	-1,83E+09	-1,59E+08	-1,59E+08	-	-
coal (29.3 MJ/kg)	Raw	-2,07E+09	-2,07E+09	0	0	-2,08E+07	-3,76E+06
biomass (50%	Raw	-3,77E+09	-3,77E+09	0	0	-	-
Ni	Water	-5,00E+09	-5,00E+09	-5,63E+08	-5,63E+08	-	-1,78E+06
Se	Water	-5,00E+09	-5,00E+09	-8,37E+07	-8,37E+07	-	-
Ba	Water	-2,51E+10	-2,51E+10	-7,06E+08	-7,06E+08	-	-
Total	All included	-3,03E+10	4,81E+11	-1,49E+09	9,42E+10	-2,04E+08	2,83E+09

Results of Alt1, Alt 2a and Alt 2b compared to Alt 0

In order to see the relative change compared to Alt 0, Table 22 shows the relative change in percent for each waste category. This is done because identifications of changes are important to see for all waste categories. All waste categories have changes except for waste category *bulky household waste* that is unaffected. Almost all other changes are net savings for all alternatives with some exceptions for a few impact categories.

Information given in relative changes does not say anything about its magnitude to other categories. Therefore data in Table 23 are presented as SEK. *Household waste* is the largest waste category and marine aquatic ecotox is the dominating impact category. As the second largest waste category is *industrial waste* and also here marine aquatic ecotox is the most dominating impact category. *Park & garden waste* has very little impact on the total results of

Alt 2b (as well for all alternatives). As mentioned before the internal order of impact categories are not the same in Alt 2a and Alt 2b compared to Alt 0 and Alt 1. This is primarily because of the larger savings in global warming. The result of this is that photochemical oxidation has a larger role and hereby has larger effect to the total results than before. Due to its magnitude, ozone layer depletion has almost no effect to the total results.

Table 22. Relative change for impact category relatively waste category for all alternatives due to Alt 0 with Ecotax02 max RT=0.

Ecotax02 max RT=0	build & dem waste			bulky househ waste			househ waste			park & garden waste			industrial waste			Total		
impact / waste category	Alt 1	Alt 2a	Alt 2b	Alt 1	Alt 2a	Alt 2b	Alt 1	Alt 2a	Alt 2b	Alt 1	Alt 2a	Alt 2b	Alt 1	Alt 2a	Alt 2b	Alt 1	Alt 2a	Alt 2b
res. exergy energy	-79%	-806%	-806%	0%	0%	0%	-12%	-150%	-168%	-8%	29%	29%	-12%	-198%	-198%	-17%	-217%	-230%
res. exergy biotic	0%	7%	7%	0%	0%	0%	-1%	-10%	-4%	102%	-387%	-387%	0%	0%	0%	0%	-7%	-3%
global warming (100 yrs)	-43%	-349%	-349%	0%	0%	0%	-23%	-237%	-283%	-7%	26%	26%	-9%	-147%	-147%	-11%	-118%	-131%
ozone layer depletion	29%	47%	47%	0%	0%	0%	-9%	-80%	-75%	3%	-13%	-13%	-18%	-43%	-43%	-10%	-130%	-121%
photochemical oxidation	-20%	-215%	-215%	0%	0%	0%	-2%	-186%	-139%	17%	-67%	-67%	-6%	-135%	-135%	-15%	-346%	-307%
acidification	-14%	-118%	-118%	0%	0%	0%	-2%	-122%	-140%	-3%	14%	14%	-13%	-217%	-217%	-46%	-761%	-807%
eutrophication	-7%	-52%	-52%	0%	0%	0%	-43%	-350%	-540%	-40%	152%	152%	-7%	-134%	-134%	-20%	-162%	-212%
fresh water aquatic ecot.	-9%	-33%	-33%	0%	0%	0%	-1%	-96%	-56%	24%	-91%	-91%	-12%	-90%	-90%	-4%	-99%	-75%
marine aquatic ecotox.	-3%	-69%	-69%	0%	0%	0%	-9%	-77%	-77%	3%	-13%	-13%	3%	19%	19%	-20%	-209%	-207%
terrestrial ecotox.	7%	-18%	-18%	0%	0%	0%	94%	-160%	295%	26%	-99%	-99%	0%	-15%	-15%	42%	-110%	8%
human tox.	-32%	-97%	-97%	0%	0%	0%	-8%	-125%	-119%	22%	-82%	-82%	-9%	-182%	-182%	-10%	-138%	-133%
Total	-9%	-125%	-125%	0%	0%	0%	-7%	-83%	-77%	17%	-67%	-67%	2%	-11%	-11%	-12%	-183%	-171%

Table 23. Weighted Impact categories listed for each waste category in Alt 2b for Ecotax02 max (RT=0).

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impact / waste category	build & dem waste	bulky househ waste	househ waste	park & garden waste	industrial waste	Total
res. exergy energy	-1,26E+09	2,87E+08	-5,77E+09	8,15E+07	-2,49E+09	-9,15E+09
res. exergy biotic	-5,64E+08	-6,88E+08	-3,54E+09	-1,31E+08	-1,48E+09	-6,40E+09
global warming (100 yrs)	-1,18E+08	4,17E+08	-5,16E+08	2,70E+07	-1,29E+08	-3,20E+08
ozone layer depletion	2,87E+04	1,80E+04	-2,06E+05	3,53E+03	4,73E+03	-1,51E+05
photochemical oxidation max	-1,26E+08	-2,22E+06	-4,88E+08	4,05E+06	-3,72E+08	-9,84E+08
acidification	-7,90E+06	7,86E+06	-1,33E+08	2,73E+06	-6,57E+07	-1,96E+08
eutrophication	3,67E+06	3,34E+06	-1,99E+07	9,44E+05	-1,16E+06	-1,31E+07
fresh water aquatic ecotox.	-1,37E+09	1,98E+08	-3,39E+09	2,93E+07	-1,67E+09	-6,21E+09
marine aquatic ecotox.	2,44E+09	3,20E+09	-7,84E+10	9,41E+08	1,72E+10	-5,46E+10
terrestrial ecotox.	4,59E+07	2,38E+07	2,12E+09	7,92E+06	4,53E+07	2,24E+09
human tox.	-5,33E+08	8,16E+07	-5,09E+09	7,70E+06	-1,16E+09	-6,70E+09
Total	-1,49E+09	3,53E+09	-9,52E+10	9,71E+08	9,88E+09	-8,23E+10

Results summary

It is interesting to know if all weighting methods show similar results. This is also one way of verifying that the results actually are rigid. Illuminating this is done in figure 20 with all weighting methods in the same graph showing relative changes.

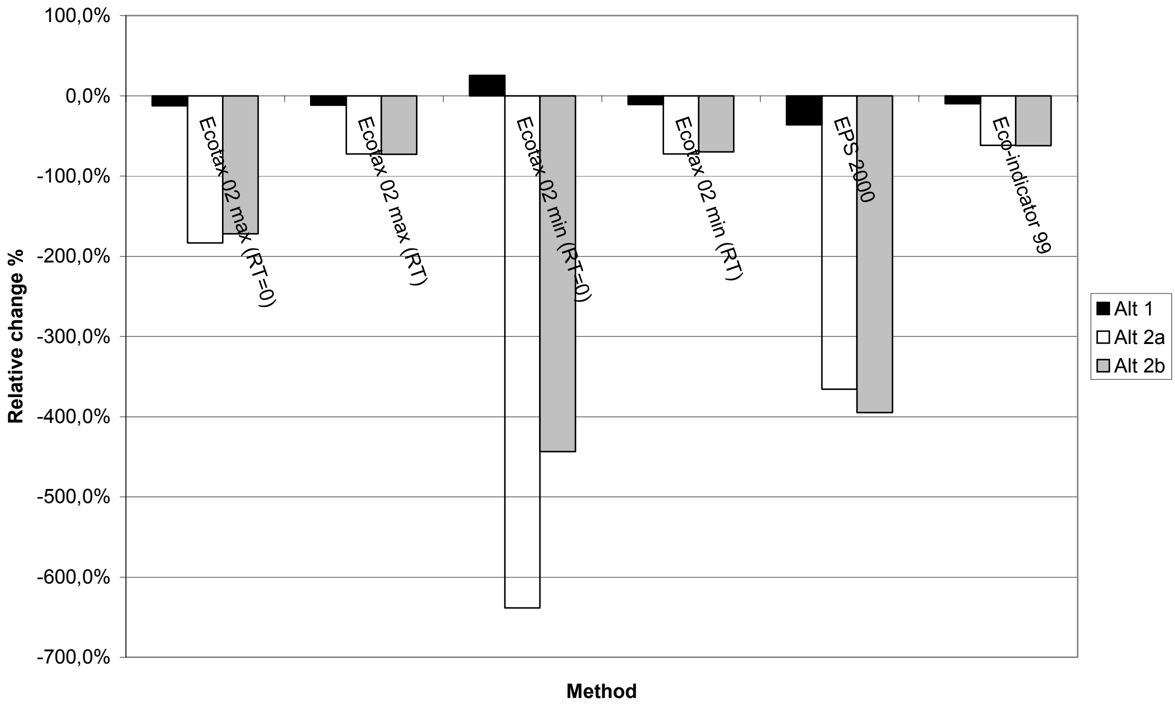

Figure 19. Relative change for alternatives due to Alt 0 for all methods.

Differences between Alt 0 and Alt 1 are small as well between Alt 2a and Alt 2b, but there is a significant difference between these groups with all methods. The pattern in Figure 19 shows that Alt 2a and Alt 2b are the alternatives that make most net reductions in environmental impacts. Alt 1 is generally favourable compared to Alt 0; a small exemption to that pattern is in Ecotax02 min (RT=0) with increasing costs for Alt 1. This is because less net savings in *household waste* and larger net costs in *park & garden waste* for Alt 1 and that terrestrial ecotox weighting results increase much more than the other impact categories. For Ecotax02 max (RT=0) there is also a peek for Alt 1 for terrestrial ecotox for these two waste categories but here other savings will make the summation to a net saving anyhow. Table 24 shows that all weighting methods used here indicate that Alt 2a and Alt 2b are better alternatives than Alt 0 and Alt 1. Figure 18 gives a good overview of how these are allocated with respect to waste category and total magnitude.

Table 24. Total single scores of all methods and alternatives.

Weighting method / Alternatives	Alt 0	Alt 1	Alt 2a	Alt 2b
Ecotax 02 max (RT=0)	-3,03E+10	-3,41E+10	-8,59E+10	-8,24E+10
Ecotax 02 max (RT)	4,81E+11	4,24E+11	1,32E+11	1,31E+11
Ecotax 02 min (RT=0)	-1,49E+09	-1,11E+09	-1,10E+10	-8,10E+09
Ecotax 02 min (RT)	9,42E+10	8,38E+10	2,59E+10	2,82E+10
EPS 2000	-2,04E+08	-2,78E+08	-9,50E+08	-1,01E+09
Eco-indicator 99	2,83E+09	2,55E+09	1,08E+09	1,07E+09

Different valuation methods have different data gaps and that is why it is important to use a set of valuation methods. In this study the results are pointing towards the same directions but when the methods are examined it is clear that the reasons why they do this is not the same (see results in Table 20).

10 Evaluation of the framework

As stated in the introduction, one purpose of this case study exercise is to test and evaluate parts of the methodological framework, and to test some newly developed methods and data with the framework. This section presents our evaluation of the framework. We also discuss whether the finding concerning methodologies can be generalised to other applications and sectors or if they are specific to the waste incineration case that we have studied. Specifically, the discussion will focus on the following aspects.

The types of questions addressed by the analysis. Different decisions have different information needs and might also need different analyses. In some contexts, the different pathways in the framework might be seen as complementary. Two different functions of an SEA were discussed by Finnveden et al (2003): To support a choice between two or several alternatives and to identify critical aspects of studied alternatives. Different methods may be more or less appropriate for the different functions.

The coverage of the analysis in terms of environmental objectives. There is often a trade-off between coverage and precision. A qualitative analysis might be possible to make more comprehensive than the quantitative pathways, and risk assessments are typically only possible for some limited substances and issues.

Whether the analysis can be usefully reproduced by other teams and applied in other segments of the energy sector. The framework is supposed to cover all aspects of the energy sector. Are some pathways more relevant for certain issues?

The resource requirements of the analysis. The different pathways put different demands on the SEA in terms of expertise needed, time requirements and financial resources. The transparency of the analysis. To what extent is it possible for an independent assessor or the user of the SEA to check the analysis in detail?

The results of the assessments. Did the different approaches produce the same or different results?

Below we discuss each of the analytical pathways in relation to these issues.

10.1 Qualitative environmental assessment

Questions addressed

In the qualitative assessment any information perceived relevant may be included. If needed impacts on a local level as well as different risks can be analysed. A qualitative analysis is especially suitable for identifying issues which are judged to be of importance. Since different alternatives may have different advantages and disadvantages, it may be difficult to support choices based on a purely qualitative assessment.

Coverage

The qualitative assessment is open to the inclusion of all well-known environmental problems. But the depth at which different topics are handled is very much up to the knowledge and resources available to the analysts.

Reproducibility and general applicability

A qualitative assessment is hard to reproduce. The result is very much dependent on which information can be easily obtained or which experts are consulted. This is especially true if the studied system is a complex one and if it the conclusions are sensitive to different assumptions. Using a life cycle perspective, as was done in this study, significantly adds to the complexity of the studied system. If there is a multitude of pros and cons of very different natures and of different degrees of certainty it is difficult to keep them all in mind at the same time. It is easier then to pick out some bits and pieces and focus on them. Which of them that gets picked out probably depends on the background, interests and prior knowledge of the assessors and this may affect the result greatly. This is less of a problem if there is little ambiguity concerning which environmental problems that are to be expected and how severe they are. To deal with the complexity a more structured approach such as MCA could be used.

Qualitative assessment is generally applicable and can easily be tailored to suit different situations and needs.

Resource requirements

The level of ambition determines the amount of resources needed. A qualitative assessment can be everything from a simple group discussion summarised in a short PM to exhaustive literature surveys and a broad involvement of different experts and stakeholders. If a lot is known through previous studies in the areas of interest less effort is of course needed to compile sufficient information for the decision.

Results of the case study.

In the assessment main environmental aspects of the different treatment methods were described. Based on this, a summary of major advantages and disadvantages of the different treatment options were listed. Overall conclusions are then discussed. However, since there are advantages and disadvantages of all treatment methods, it was difficult to draw firm conclusions concerning the overall preference of the studied alternatives. It is also interesting to note that results from previous quantitative methods were useful when drawing conclusions.

10.2 Traditional LCA

The strengths and weaknesses associated with LCA in the context of SEA are basically the same as for any other use of LCA, although some aspects may be accentuated while others are of less importance. The issues addressed in this section are in principle valid also for site-adjusted LCA and the evaluation, as both of these are based on the traditional LCA.

Questions addressed:

A traditional LCA quantifies the potential life cycle environmental impacts of the individual processes within a system, as well as of the system as a whole. LCA does this without site-related information. Therefore, questions related to for instance risk can not be included. The results can be used to analyse the system at different levels of aggregation. In our case, at a lower level for instance the pros and cons of treating a certain waste fraction in a certain way can be assessed, or the dominating contributions and savings in terms of processes or emissions of a certain impact category may be identified. At the most aggregated level, the net burdens of the system for each impact category are assessed.

Traditional LCA may be difficult to use for priority setting in SEA. The LCA will quantify each environmental impact category, but it will not show what impacts are more important and therefore should be emphasised. The importance of absolute changes in impacts can only be assessed by some sort of normalisation or weighting. As a result, the evaluation of results is difficult to optimise when all impact categories must be given equal importance. In our case, we could however to some extent rely on earlier results of LCA of waste management to prioritise between impact categories.

Coverage:

LCA addresses many, but not all impacts that are part of the national environmental objectives. More impact categories may however be included than those that were included in this study. Typically, impacts on bio-diversity, land use and the built environment are not well covered in LCA. Also for impact categories included, there may be significant data gaps, limiting the coverage. Typically toxicological impacts and emissions to water have significant data gaps (Finnveden, 2000). Because LCA is not site-specific, it is very difficult to fully assess impacts that depend on site and concentrations. Local impacts such as noise and odour are typically not included.

Reproducibility and general applicability:

LCA relies on a standardised methodology and is based on natural science. Ideally, it should therefore be reproducible. However, although standardised, it includes many elements that are more or less subjective. Therefore, in practice there is always a risk that different analysts will come to different conclusions. To minimise this risk and to reveal any subjective choices, the standardised procedure requires peer review and full transparency in the results presentation.

LCA methodology is applicable to other segments of the energy sector. Because it has limitations in assessing risk, local concentrations and impact, and radiation, it is not suitable for analysis of all types of energy systems.

Resource requirements:

LCA is a resource intense methodology. In the first place, it requires LCA expertise, but then of course it is also necessary to have access to expertise within the specific field of the study. Data requirements are also significant (see below), which may be both time consuming and expensive. Data collection is facilitated to some extent by the use of available LCA databases.

Transparency is a requirement. Therefore, all model data should be published along with the results. It however requires LCA expertise to be able to interpret and make use of this large amount of information.

Results of the case study.

As noted above, there are limitations in coverage. However, for those impact categories that are included, quantitative results are provided. The assessment can be used to identify critical aspects. It can also be used to support choices between the studied alternatives. It can for example be concluded that for most impact categories (terrestrial ecotoxicity is the exception) does the suggested waste incineration tax provide improvements compared to the zero-alternative. It can however, also be noted that these improvements are small, compared to the potential improvements that the alternatives 2a and 2b represent. The alternative 2a (minimise energy use) is better than the zero alternative with respect to all included impact categories. These conclusions are not sensitive to the chosen time perspective. It can however be noted that it is not possible to draw any firm conclusions concerning the overall preference between alternatives 0 and 1, since it is possible that the impact category Terrestrial ecotoxicity is of such an importance that this category can outweigh the results of the other impact categories. A weighting is thus required to reach an overall conclusion.

Other aspects, data:

Large amounts of data are involved in LCA; both inventory data and generated results. This is actually both a strength and a weakness of LCA. Collecting inventory data may be difficult, time consuming, and expensive. But once collected, this data enables quantification of a complex system like few other methods. The amount of output data may be overwhelming and difficult to interpret, but with clever and careful analysis, many aspects of the system may be revealed that could not be intuitively understood.

In our specific case of analysing several dimensions of waste management (many different waste flows and fractions, treatment options, avoided processes etc.) at the national level, the problems with data collection and evaluation were accentuated. Waste management is seldom analysed at this scale and with the level of detail required by an LCA. Consequently, it was difficult and time consuming to find inventory data. Undoubtedly this also leads to a greater risk of introducing unnoticed data gaps, which in turn may lead to under estimated burdens associated with process for which emission data is lacking, as well as over estimated importance of processes for which better inventory data is available.

Evaluation of results also became a complicated task because of the many dimensions of the system. The results contain information on every aspect of the system, but because of the size of the system, it was difficult to realise small, but possibly important changes between alternatives. Evaluating the LCA results also raises questions such as what is the net impact of recycling a certain material, how may a certain recycling system be improved, what is the total burdens of transports within the system, etc. Although the LCA results contain this information, the answers to such question may be difficult to extract. For practical reasons, the evaluation had to be done at a very aggregate level. The same problems would not necessarily arise in LCAs of other types of policies if fewer material end energy flows were affected.

One may argue though, that looking into details is not necessary to evaluate a strategy of this scale. If the strategy as a whole leads to general improvement, trade-offs within the system may not be important. However, if one is to better understand the strengths and weaknesses of the strategy, when it works efficiently and when it does not, the details of the system need to be understood. Only then can one find possible ways to improve the strategy. By being modular, the LCA model does allow the kind of disintegrated analysis that this would require. But using the model in that manner would be much more time-consuming and actually may not be within the scope SEA.

Another problem related to collection of inventory data aroused because of the uncertainty of the design of the different alternatives. All alternatives were placed in the future, which is uncertain in itself, so that even the design of the no-action alternative was uncertain. Estimating the effect on waste flows of introducing the waste incineration tax was an even more delicate task, although this to some extent was already done by the commission report. Besides, all process models are based on historic data, or at best, rather current data. As it is one of the basic ideas of SEA to make assessments at an early stage in a planning process, these problems will arise in any type of LCA made for this purpose.

Because of the many processes involved in the system, using process data from different databases was a practical necessity. This caused a number of problems. Different databases are not consistent in the way inventories are made. Thus, some may have very complete emission inventories, while other suffer from data gaps. Also, the same compounds may be called different things in different inventories, which in theory should not be a problem, but in practice may be difficult to handle. Further, it may be difficult to get a complete picture of for instance the impact of electricity production, as different electricity models are used in different databases. These problems arise in most LCAs, but become more difficult to handle the larger the system becomes.

10.3 Site-adjusted LCA

Questions addressed:

A site-adjusted LCA can in principle address the same issues as a traditional LCA. However, it can be used to provide a more accurate model, by bringing in some site-dependent information into the assessment. In addition, a site-adjusted LCA can address other questions which may be of relevance especially in an SEA context. For example, it can address the issue whether there are reasons to have different policies in different parts of a country.

Coverage:

In this study, only some air emissions were addressed in the site-adjusted assessment. In principle, other emissions could have been included. It is in general more information available for air emissions than water emissions increasing the possibilities for a site-adjusted assessment.

Reproducibility and general applicability:

In this particular study, we have developed characterisation factors for some air pollutants using a specific model (Ecosense 2.0) with a certain data set and assumptions. If others are using the same characterisation factors, the results should be perfectly reproducible. Also, if the same model, with the same data is used, the same characterisation factors should result. Whether the same results would have be given, if a different model had been used and or slightly different input data had been used is not tested here. There are ongoing discussions in the scientific community concerning differences in results from different models (e.g. Krewitt et al 2001).

Resource requirements:

To develop models for calculating site–dependent characterisations factors is very resource demanding. If models are available, the calculations do require some resources but are feasible. Using the developed characterisation factors is in principle straight forward and as resource demanding as a traditional LCA. However, in practise, the work can be significantly increased since all inventory data has to be attributed to the relevant site-dependent categories. Depending on the amount of inventory data, this can be a rather time consuming exercise.

Results from the case study:

In this case study, only air emissions of NOx, SO2 and particulates were studied in the site-dependent assessment with respect to ecosystem damages and health impacts. An interesting result was that although there are differences in the characterisation factors for different parts of the country, the ranking between the studied alternatives did not change. This result suggests that there are no reasons to have different policies in different parts of the country.

10.4 Weighted LCA

Use of weighting methods may raise a lot of questions attached to methods rather than results. This takes down some of the integrity in the methods in common and is no exception when used in a SEA. A set of weighting methods is preferred as discussed earlier since a lot of the existing methods use different approaches to calculate weighting factors.

Questions addressed:

Weighting methods is used primary for two reasons. One of them is the weighted single score value. This is useful when comparing several alternatives and a favourable is to be chosen, or in studies where the main goal is to "put a price" on the environmental activity. The second reason is the possibility to compare between different impact categories and see their specific contribution to the total single score result or to find out what impact categories matters the most. This comparison is only possible because of the use of a specific weight of reference applied to all impact categories. Therefore comparisons and priority settings are doable between impact categories. For SEA the focus is the same as a traditional LCA when using weighting methods. Questions like; Do weighting method results points out something else that was not discovered in the results of characterised LCA? is to be kept in mind. In a complex system it could be of great help to be able to add one impact category to another to see the relationship between different systems or subsystems. Especially in systems with many flows of unlike proportions and type.

Coverage:

Since every weighting method are based on one or possible a few classified approaches like for example monetisation, standards or panels, the specific method only reflect the values from the point of view it is built upon. Limitations are also the same as for the chosen impact categories for respectively methods used, depending on choice of impact categories some are not included, for example in Ecotax02 land use and bio-diversity are not included, but a variety of toxicity groups are included. In other weighting methods it could be the opposite as for the other weighting methods used in this study. There are also large variations in compartment groups for each method. Some methods use for example air, water, soil and air frequently in their impact categories while others more consequently use only one of these.

Reproducibility and general applicability:

Ecotax02 may be reproducible according to Finnveden et al (2002) but only if the procedure is clearly described. It is only when a change is done to a method as for example the weight of reference in Ecotax02, it become less reproducible i.e. if same weighting method always is used it gives a good reproducibility. Applicability is for Ecotax02 limited to Sweden because of the link to Swedish fees and taxes. Depending on the availability on local fees and taxes Ecotax02 could bee used elsewhere, but it is clearly that these kinds of fees and taxes not are available in some countries or areas. Since weighting methods in common are built upon a base of valid parameters it is not suitable to use them for some system that contains parameters not included in method, such as for example radiation etc.

Resource requirements:

Ecotax02 among with other weighting methods do not require a lot of efforts, that is if only executing existing generic weighting factors. This is also discussed in Finnveden et al (2002) and therefore conclusion is that the critical aspect lies within establishing new weighting factors. Executing weighting methods within SEA does not demand more resources than within a traditional LCA.

Other aspects, data:

Transparencies in weighting methods are connected to how well procedures and calculations are documented rather than the type of methods, therefore all steps should be properly documented. Examine input related data Ecotax02 as a method of willingness to pay based upon fees and taxes is heavily depending on the availability of these data and the diversity of them. High amount of specific fees and taxes connected to specific emissions or substances give a higher rate of accuracy.

Output data is in general depending on many steps such as inventory data, correctness of models of processes etc. and will in a “worst” case be a product of these factors with an inappropriate result. In this study it is also showed that the use of weighting methods with RT data and methods without it, differ in output values. Thou this is a fact the pattern of all weighting methods is still quit homogeneous in direction even if not in magnitude.

Results from the case study:

The results from the weighting can be used to support the choice between different alternatives. In this case the results from all used weighting methods were consistent in some aspects. Alternative 1 gives some environmental improvements compared to the zero-alternative. However, the improvement is limited compared to potential improvements in scenarios 2a and 2b. The applied weighting methods were however not consistent concerning the ranking between alternatives 2a and 2b. Weighting methods can also be used to identify critical aspects of the studied systems, for example which environmental impacts are most important for the studied systems. It is interesting to note that although the used weighting methods give consistent results concerning the ranking between the studied alternatives, the reasons for this is very different. Thus, different methods point out different aspects as the most important ones. It can also be noted that those methods which presents results in monetary units (Ecotax, EPS and ExternE, do provide magnitudes in their results.

11 Final discussion and conclusions

This study has several aims which are discussed separately.

11.1 Assessment of a waste incineration tax

An introduction of a waste incineration tax is likely to result in environmental improvements. A waste incineration tax would be accompanied by an increased landfilling tax, in order to ensure that landfilling is avoided. The environmental improvements are partly accomplished by diverting materials from landfills to recycling as a result of the increased landfill tax, and partly by diverting materials from incineration to recycling.

Although the studied alternative of a waste incineration tax of 400 SEK/ton is likely to lead to environmental improvements, the improvements are small compared to the potential improvements as shown in the visionary scenarios developed here. In order to go in the direction of these visionary scenarios, a combination of a non-differentiated and a differentiated incineration tax would be useful. The differentiated incineration tax should be based on the content of fossil carbon in the waste. An incineration tax of approximately 2000 kr/ton of plastic materials would harmonise the waste incineration tax with the carbon dioxide tax. Such a tax would make recycling of plastic materials economically attractive.

A site-adjusted analysis indicates that the ranking between different alternatives are similar in different parts of the country. There is thus no indication that different waste policies should be used in different regions.

11.2 Evaluation of the SEA framework

None of the methods used in this case study provide the tools necessary to estimate what may be called first order effects of an new strategy or policy. In our case, the first order effects would be the redistribution of waste flows as a result of introducing the waste incineration tax. In the case of a new energy policy, it might be changes in energy supply and use. Because we had access to the commission report (SOU 2002:9), which included estimates of waste flows in the no-action alternative and as a result of introducing the waste incineration tax, such tools were not necessary in our case. Methods for estimating first order effects include energy- and waste economic models and different types of scenario techniques, rather than the environmental methods of primary focus here.

The methodological pathways discussed here all have different advantages and disadvantages and provide different types of information. It is therefore suggested that they largely complement each other, rather than compete with each other.

A major advantage with a qualitative approach is that it can cover all relevant environmental issues. It can mainly be used for identifying aspects which are judged to be of importance. It is less useful for supporting choices between alternatives. Since the approach is less structured and quantitative, it is open for criticism concerning its subjectivity and lack of reproducibility.

A quantitative traditional LCA approach will avoid some of the drawbacks of a qualitative method. However, it can typically not cover all relevant types of environmental impacts and will have some data gaps also for impact categories which are included. It can therefore be useful to complement a traditional LCA with a structured qualitative approach in order to check that important issues are included in the overall assessment. A traditional LCA without weighting methods can be used to identify critical aspects, and also support choices between alternatives. However, if different impact categories point in different directions, no firm conclusions concerning the preference of different alternatives can be drawn.

Some type of weighting or valuation method is often useful when supporting choices between alternatives. It is also necessary in order to identify which environmental impacts are most important. In order to be useful, the valuation methods must however have some credibility among the stakeholders. The use of several valuation methods can sometimes increase the credibility of the results.

A drawback of a traditional LCA-approach is the lack of site-dependent information. Adding such information will add to the accuracy of the model. It can also allow other types of question being asked such as, Are there differences in different parts of the relevant region which warrants different policies?

A risk assessment approach is not used here. Risk assessments could be used for answering other types of questions. It could for example be used to study whether air quality norms are exceeded. This is particularly interesting when SEA is applied to energy and transportation planning. It is thus useful for identifying certain types of critical issues which can not be handled with other methods.

Since different methodological approaches can produce different types of answers, it is suggested that a careful consideration is given at the start to this issue: What are really the questions that the assessment should try to answer. It is also suggested that if any of the quantitative approaches is used, it is complemented by a structured qualitative approach with the aim of identifying critical issues which may go unnoticed in the quantitative approaches.

11.3 Development of a site-adjusted approach

Site-dependent characterisation factors for Sweden have been developed for impacts on human health and ecosystems from emissions of NOx, SO_2 and particulates. For health impacts, the results are different characterisation factors for different parts of the country and from different stack heights. For ecosystem damages, the results are only small differences for the used definitions of ecosystem damages.

The developed characterisation factors are applied to the case study. It is interesting to note that although there are differences in characterisation factors, this did not affect the results and the conclusions.

11.4 Update of the Ecotax weighting method

The Ecotax weighting method as developed by Johansson (1999) has been updated and applied to the case study. The results can be compared with the results from other weighting methods. It is interesting to note that different weighting methods provide the same ranking between different alternatives, but for different reasons. Thus different methods identify different aspects as the most important ones. The Ecotax02 method generally point out resource use, emissions of gases contributing to climate change, and toxicological impact as the most important ones.

Different versions of the Ecotax method are used where minimum or maximum weighting factors are used in cases where there are large uncertainties, i.e. especially for the valuation of resources and toxicological impacts. These different sets produce significantly different results in absolute values. It is also interesting to note that the choices concerning the time frame of the study can have a significant influence on the results. If a cut-off is made after approximately one century, and emissions occurring after this time period are neglected, significantly lower results are produced compared to the cases where also long-term emissions are taken into consideration.

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