NORDTEST Report
TR 517
Approved 2002-11

Guidelines for the use of LCA in the waste management sector

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NT TECHN REPORT 517
Approved 2002-11

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Title (English):

Title (Original): Guidelines for the use of LCA in the waste management sector

Abstract:

This report contains guidelines to the application of life cycle assessment (LCA) in the waste management sector. Focus is put on the most common municipal waste management scenarios in the Nordic countries and the guidelines are supported with case studies in the appendices. In an LCA study the environmental aspects and potential impacts throughout a product's life, from raw material extraction through production, use and disposal are analysed. Provided all upstream and downstream impacts are equal, the life cycle of waste starts when products are disposed of in the trash bin and ends when the waste material is degraded or brought back to the technological system through recycling and replaces other products. LCA in the waste management sector can be applied in order to compare the environmental performance of alternative waste treatment systems and identify focus areas for system performance improvement. It can also help to improve product development, e.g. eco-design, environmental labelling and declarations and introduce regulations that promote better alternatives.

This report consist also of 3 appendices as separate reports in pdf format, which all can be downloaded from Nordtest web site. Addresses to these documents in internet can be found on page 6 of this report.

Technical Group: Expert Group Environment and Natural Resources

ISSN: 0283-7234	**Language:** English	**Pages:** 96
Class (UDC): 628.4	**Key words:** Life cycle assessment, LCA, landfill, recycling, waste, environmental impact, Nordic	
Distributed by: NORDTEST Tekniikantie 12 FIN-02150 ESPOO Finland	**Publication code:**	
	Report Internet address: http://www.nordtest.org/register/techn/tlibrary/tec517/tec517.pdf	

SUMMARY

This report contains guidelines to the application of life cycle assessment (LCA) in the waste management sector. Focus is put on the most common municipal waste management scenarios in the Nordic countries and the guidelines are supported with case studies in the appendices. In an LCA study the environmental aspects and potential impacts throughout a product's life, from raw material extraction through production, use and disposal are analysed. Provided all upstream and downstream impacts are equal, the life cycle of waste starts when products are disposed of in the trash bin and ends when the waste material is degraded or brought back to the technological system through recycling and replaces other products. LCA in the waste management sector can be applied in order to compare the environmental performance of alternative waste treatment systems and identify focus areas for system performance improvement. It can also help to improve product development, e.g. eco-design, environmental labelling and declarations and introduce regulations that promote better alternatives.

The guidelines follow the general methodological structure of LCA described in the ISO 14040 series. Prioritised issues are system boundaries, inventory data, allocation and impact assessment. The focus is on mixed municipal waste and less focus is put on pure material recycling processes.

How to define the functional unit in an LCA for waste management and what life cycle stages should be taken into consideration when defining the system boundaries of a study are defined. Cut-off criteria that are common to use to limit life cycle systems both within a defined system and also with respect to start and end of the waste life cycle are listed and discussed. Guidelines to questions such as how far do we follow products from recycling, how far do we follow products replaced by products from recycling and how long do we take into account emission and resource consumption related to a landfill are provided.

If an LCA study involves specific waste treatment processes, attempts should be made to collect and apply data that are as specific as possible for the process in question. In the case of more generic studies, such as e.g. a basis for political decisions, generic data should be applied. In the guidelines, parameters and to a certain extent, data that are commonly applied in inventory of LCA for waste management are presented. The treatment alternatives: Incineration, landfilling, aerobic composting, anaerobic digestion and biocell are discussed separately. Inventory data of interest related to these treatment alternatives are listed. Typical process flow charts are drawn and critical issues related to emission and resource consumption are discussed and guidelines given. If energy is recovered from incineration plants or when incinerating collected landfill gas or biogas, energy sources in other systems are substituted with the recovered energy. Steps that should be followed when identifying substituted energy sources, are listed and guidelines given on how to credit the waste management system by avoided impact of the energy source substituted. Anaerobic digestion and aerobic composting produce products that can be used as fertilisers. The products can replace artificial fertiliser, although there are great uncertainties related to what extent the artificial fertilisers are replaced. Guidelines are given for calculations

on how much artificial fertiliser is substituted, avoided impact related to production of artificial fertiliser and data for pollutants in sludge and compost. Problems related to allocation in LCA for waste management (i.e. in multi input recycling and open loop recycling) are discussed and guidelines given.

The general methodology on how to perform quantitative life cycle impact assessment is described in numerous methodology reports. Hence it is not described in any details in this report. LCA applied for municipal waste management usually includes the same environmental impacts as LCA studies in general. Based on the Danish UMIP study and the Nordic Guidelines on Life-Cycle Assessment applicable impact categories are listed. The toxicity impact category is an important category for LCA applied in the waste management sector but needs further development within that sector /5/. Characterisation models for the toxicity impact category are listed and recommendations made in the guidelines. Normalisation and weighting are optional elements in the life cycle impact assessment. Many weighting methods exist, but no methods have been identified that are particularly developed for application in the waste management sector. The newest and most commonly used weighting methods applied in the Nordic countries are listed in the guidelines. It is emphasised that weighting is a controversial issue and there is no consensus within the Nordic countries or other international forums on recommended weighting methods.

The interpretation phase of an LCA requires an analysis of the results of the LCA, conclusions and recommendations according to the ISO standard. In the guidelines questions are listed related to the waste management sector, to assist fulfilling these requirements.

Several groups are working on LCA in the waste management sector, developing new models and performing LCA studies. Some of these groups and projects are listed and described in the report and references given for where to seek further information. Findings of LCA in the waste management sector are discussed. The results of these studies can however, in most cases, not be generalised as results of LCA studies are site dependent and depend on assumptions and choices made. The discussion however provides ideas about what kind of conclusions can be drawn from LCA studies in the waste management sector.

In appendices to the guidelines are case studies of LCA for waste management in Iceland and Norway. Appendix 1 contains a comparative LCA screening study for waste management in Reykjavik, Iceland, where landfilling with gas collection, composting in containers and waste treatment in biocell were compared. Appendix 2 contains a descriptive LCA case study for waste management in South Iceland where the land is sparsely inhabited. In appendix 3 are summaries of three Norwegian LCA case studies for municipal waste and sludge treatment.

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THE GUIDELINES INCLUDE THE FOLLOWING APPENDICES AS SEPARATE REPORTS:

APPENDIX 1 Icelandic comparative case study – landfill, biocell, compost.
http://www.nordtest.org/register/techn/tlibrary/tec517/tec517_ap1.pdf

APPENDIX 2 Icelandic descriptive case study – landfill without gas collection.
http://www.nordtest.org/register/techn/tlibrary/tec517/tec517_ap2.pdf

APPENDIX 3 Summary of Norwegian case studies.
http://www.nordtest.org/register/techn/tlibrary/tec517/tec517_ap3.pdf

1 INTRODUCTION

1.1 Background

Solid waste management is currently the subject of many debates in the Nordic countries. This debate is driven by consumer and legislative pressure. Legislators have been active in establishing a legislative framework aiming to exploit the inherent resources (energy and materials) in waste.

European Community (EC) directives and documents such as the Packaging and Packing Waste directive, the Landfill directive, Incineration directive and working document on the Treatment of Biowaste, all have in common the waste management hierarchy. That is waste minimisation at source, reuse, recycling, incineration with energy recovery and landfill and limitation of environmental impacts of the waste treatment alternatives. To demonstrate the performance of management alternatives in the decision-making process, authorities, communities, industry and waste management companies should use environmental assessments in addition to the evaluation of technical and economical aspects.

The application of life cycle assessment (LCA) to products and services has become a useful tool in decision-making processes and system performance documentation. The interest of using LCA in the waste management sector is increasing. In the Nordic countries several projects have been performed that gives methodological solutions on how to apply LCA to waste treatment practices. This is briefly documented in the Nordtest state-of-the-art report on LCA in the waste management sector /5/. However, the report also highlights the need for a consensus in the Nordic countries on a range of important issues. Although the LCA methodology is standardised through the ISO 14040 series /1/-/4/, there are several issues that make LCA in the waste management sector complicated. In order for LCA to assist in decision-making, it is important that the challenging issues are solved within a common framework. A guideline on LCA in the waste management sector will contribute to guide LCA practitioners in such a way that important topics are taken into account.

This guideline document has been developed by Linuhonnun Consulting Engineers (Iceland) and Det Norske Veritas (Norway). The project team was composed of Helga J. Bjarnadóttir (LH), Guðmundur B. Friðriksson (LH), Tommy Johnsen (DNV) and Helge Sletsen (DNV).

The project was financed by Nordtest, Linuhonnun Consulting Engineers, Det Norske Veritas, Orio (Norway) and Fenur (Iceland). In order to incorporate expertise from other Nordic countries and in order to spread the results, it was decided to include in the project an independent critical review group. Specialists from Sweden, Finland and Denmark were contacted and active members of the critical group were Göran Finnveden and Anna Björklund, (FMS, Sweden) and Michael Hauschild (IPU, Denmark). They gave valuable comments on both the guidelines and the appendixes.

1.2 Objective

The objective of the project is to develop a guideline that can help decision-makers in the municipal waste management sector to perform LCAs and use the results in the decision-making process. The geographical scope is the Nordic countries.

It must however be noted that the data given on waste composition in this report must be regarded as a snap-shot at a given time in any Nordic country. In the mean time the waste management policy might have changed and subsequently waste composition. Therefore, given data should be regarded more as a guide on format and potential references rather than the numbers themselves.

1.3 Scope

As a basis this study refers to the ISO 14040 series, relevant studies identified in the state-of-art report /5/ and other studies that are relevant in an LCA in a waste management perspective. LCAs applied to Nordic waste management scenarios will be used to make examples practical. The following main tasks will be performed and reported:

- Short introduction to the general LCA methodology and applications.
- A literature survey will be made as an updated complementary input to the identification of relevant studies in the Nordtest state-of-the-art report.
- A guideline will be written focusing on how to carry out the most critical parts of an LCA study for the most common waste management scenarios, including examples from case studies.
- Examples will be established based on case studies carried out before and during (Icelandic case study) the project period. In addition to be used as practical examples, the case studies also has the purpose to build and transfer LCA competence to Iceland, where no such studies have been performed to date in the waste management sector.

The following limitations are valid for the study:

- Experience will mainly be drawn from Nordic studies. Studies from countries outside Scandinavia will only be mentioned for reference purposes. This is due to both time restrictions and because applied technology is regarded to be on similar levels.
- This guideline document is written for studies with the objective to identify and assess the environmental key issues of the treatment processes of municipal waste from the waste collection system and to the point where the waste ceases to exist through decomposing at landfills, composting or bio-reactor plants, incineration or recycling into a new product system. However, many of the recommendations can be transferred to other waste streams as well. In any case, one should be careful to apply data from the guideline directly without first checking the relevance for the specific waste stream and treatment technology under study. Product specific LCAs and related waste streams will not be treated.
- The application of LCA to municipal waste streams will be limited to a given amount of mixed waste. This means that the functional unit is given as weight (ton) or volume (m^3) waste, and that the pre-disposal life cycle stages of the products generating the waste is not included. Further, this means that product

design, production efficiency and degree of consumption will not influence the results.
- Work environment is not treated as impacts in the study.
- Only the operational stage of the life cycle of waste treatment plants is considered. Construction and the end of life stages are not included.

1.4 Use of the guideline

The guidelines might be used as a guide when performing an LCA or as a checklist/baseline when validating an LCA. It should be used in combination with more general LCA guidelines or standards such as the ISO 14040 series in combination with recent general methodological development documents.

The guideline will provide assistance in the following:

- It will give a brief introduction to managers in the waste management sector on what LCA is and how it can be applied to the benefit of the decision-making process.
- It will work as a checklist for LCA practitioners on the most central issues of LCA in the waste management sector.
- Provides baseline data and information for critical review or validation purposes.

The guideline does not address the planning stage of an LCA as this is properly treated in more general guidelines and standards.

2 LIFE CYCLE ASSESSMENT AND APPLICATIONS

2.1 The concept of life cycle assessment

Generally, life cycle assessment (LCA) can be defined as a method that studies the environmental aspects and potential impacts of a product or system from raw material extraction through production, use and disposal. The general categories of environmental impacts to be considered include resource use, human health and ecological consequences.

The result of an LCA is an environmental profile that expresses the performance of the total system life cycle and single life cycle stages. It has become a recognised tool in decision-making within industry and public administration. As a consequence, several international, national, industry branch and company specific LCA databases are established to provide data efficiently. Data on waste treatment processes still tend to be missing or are of low quality. However, the situation is improving due to projects carried out among other in the Nordic countries that latest 2-4 years. Figure 2-1 shows a general life cycle system.

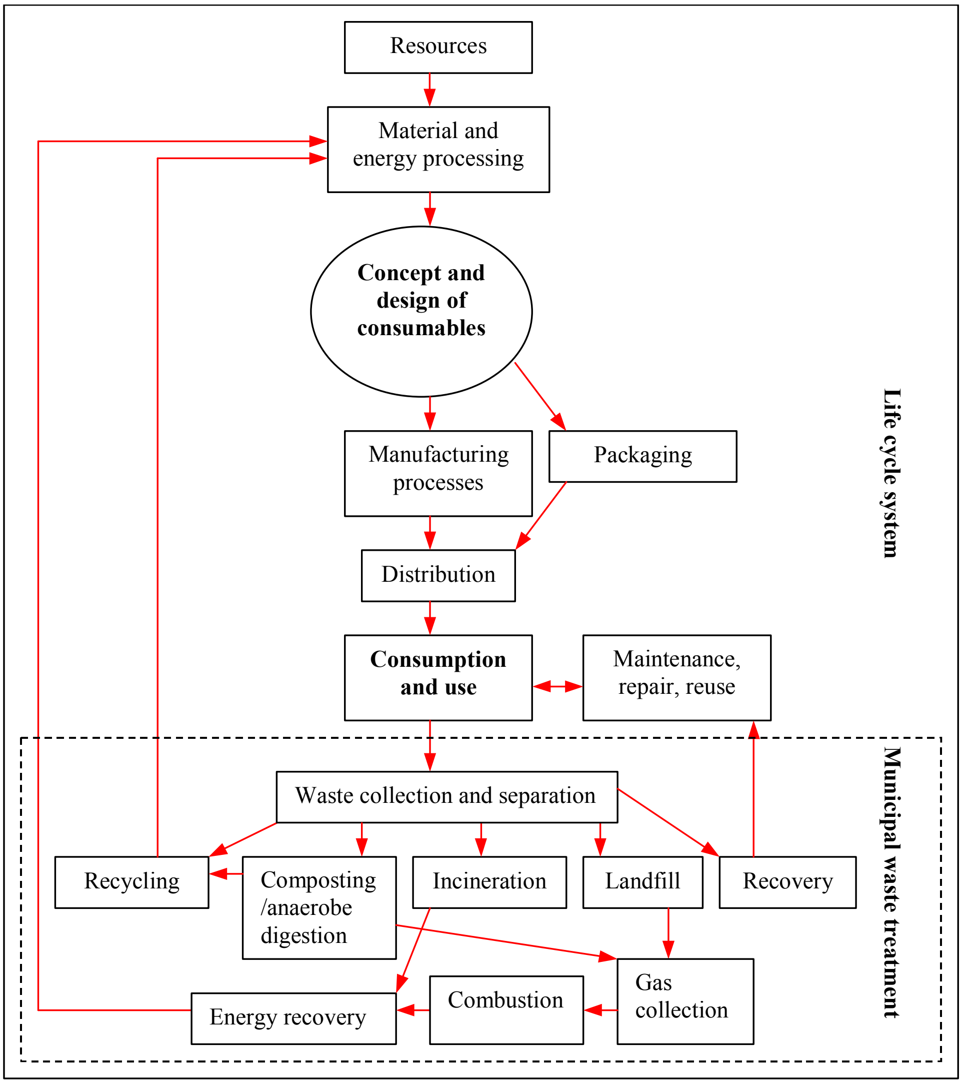

Figure 2-1 Schematic system life cycle

As can be seen from Figure 2-1, the environmental impacts of the municipal waste system depend on three different system characteristics:

1. **The concept and design of the products** that end up as waste have influence on the type and amount of material that the products consist of, life-time of the products, to what degree the products are recyclable and non-hazardous, and to what degree they can be dismantled into recyclable fractions.
2. **Consumption patterns** influence the municipal waste flow because it is the consumers who buy the consumables that flow though the system and who partly decides the lifetime of the consumables.
3. **The municipal waste treatment** decides to what extent waste shall be distributed between the treatment alternatives and the technology and efficiency of the treatment options.

LCA that focuses on waste gives different system boundaries depending on the goal and scope of the study. Figure 2-2 shows examples of three different levels of system boundaries. The foreground system illustrates the main processes to be analysed, while the background system is other processes that are influenced by the foreground processes.

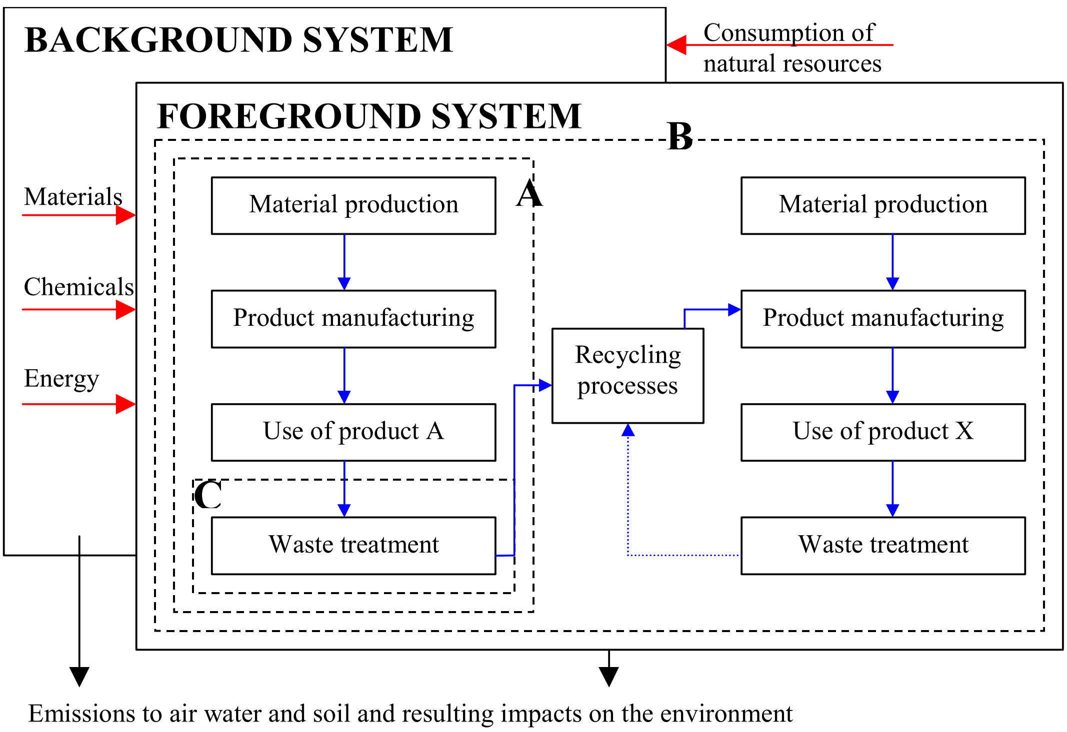

Figure 2-2 Different system boundaries for LCA

For the following description, note that:

1. System within the dotted line marked A is referred to as alternative A.
2. System within the dotted line marked B is referred to as alternative B.
3. System within the dotted line marked C is referred to as alternative C.

Alternative A shows system boundaries for an LCA of product A. The waste treatment after product use is included within the system boundaries as this life cycle stage will itself require energy and materials and cause impacts on the environment. Traditionally, several studies have located the waste treatment outside the system boundaries (A minus C) reporting only a the amount of waste leaving the system boundaries (and in some cases type of treatment). In case of recycling of waste, system boundaries can be extended as shown in alternative B. In LCAs evaluating waste treatment options, the system boundaries can be set where the waste is introduced into the system (alternative C). This is however only possible when it is presumed that all preceding processes are the same for all options, or that they do not influence on the waste composition.

The third point following Figure 2-1 and alternative C related to Figure 2-2 expresses the same system –the municipal waste treatment system- and are the focus of this document. The life cycle of waste starts when products are thrown in the trash bin and ends when the waste material is disposed and degraded and/or is brought back to the technological system through recycling and energy recovery. A coarse generic illustration of such a system is given in Figure2-3.

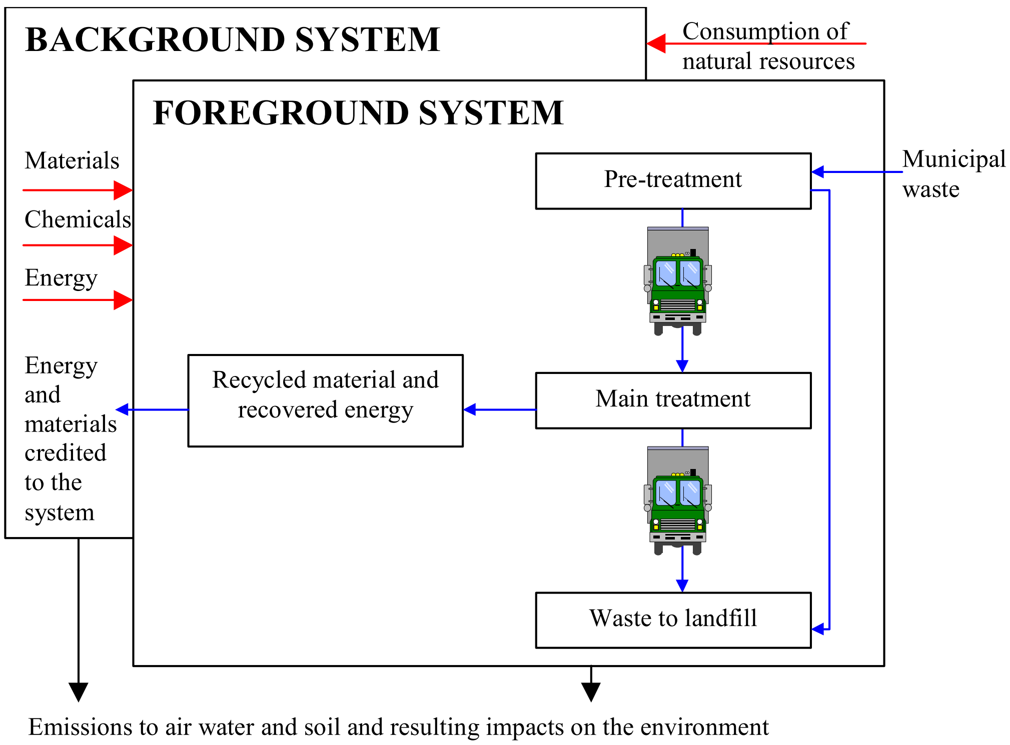

Figure2-3 System boundaries for waste treatment options

2.2 Life cycle phases and work process

The general methodological structure of LCA, which is used as basis in this guideline, follows the ISO 14040 series:

- ISO 14040:1997 – Principles and framework

- ISO 14041:1998 – Goal and scope definition and inventory analysis
- ISO 14042: 2000 – Life cycle impact assessment
- ISO 14043: 2000 – Life cycle interpretation

The names of the ISO publications more or less reflect the main phases of an LCA. The phases are given below and illustrated in Figure 2-4.

- **Goal and scope definition**, where the goal and scope of the study are defined.
- **Inventory analysis**, which involves compilation and quantification of inputs and outputs, for a given life cycle system.
- **Impact assessment**, which aims at understanding and evaluating the magnitude and significance of the potential environmental impacts of a life cycle system.
- **Interpretation**, in which the findings of either the inventory analysis or the impact assessment, or both, are combined consistent with the goal and scope in order to reach conclusions and recommendations.

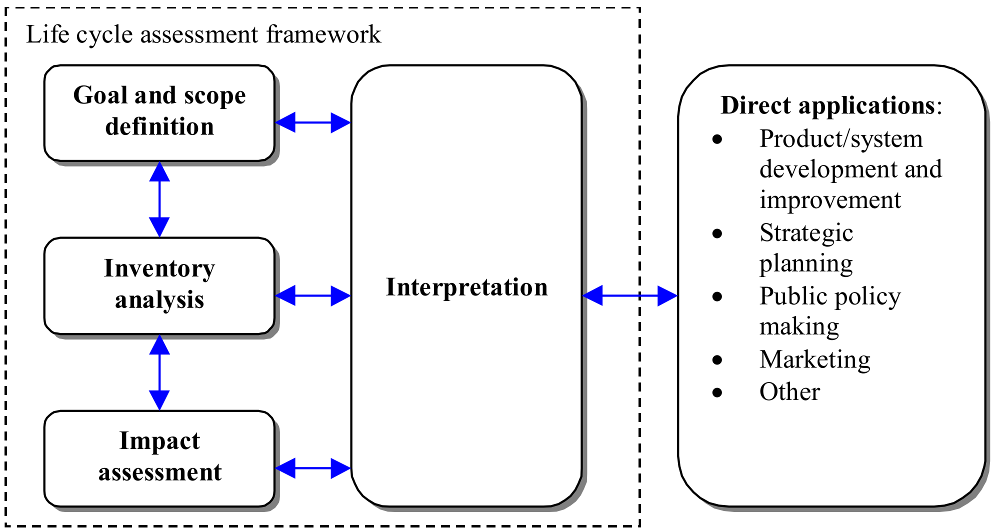

Figure 2-4 Phases of an LCA

An LCA study does not always need to use impact assessment. In many cases inventory data alone are sufficient for an evaluation. The term LCI (life cycle inventory) is used to indicate that a study has excluded the impact assessment phase.

It is beneficial to perform the LCA in at least two iterative steps. The first time one goes through the LCA phases. First, a key issue identification should be carried out. A rather broad system should be defined and rough data can be used. Dependent on the outcome of a sensitivity and uncertainty assessment, a more detailed study should be performed with revised system boundaries and focus on high quality data going through all the phases again this time with specific focus on the points identified during the first iteration.

It is referred to the ISO 14040 series documents for further general details about the contents of each life cycle phase. For details related to assessment of municipal waste is referred to chapter 3 in this document.

2.3 Applications

LCA results may be useful inputs to a variety of decision making processes. Life cycle assessment in a waste management perspective is specifically targeted towards:

1. Identification of the most environmentally significant processes during the waste treatment chain.
2. Identification of the most significant environmental burdens during a waste management scenario.
3. Identification whether improvement proposals result in local optimisation (shift of environmental burdens to other sites), or if they are environmentally for the better for the whole waste management system.
4. Assessment of the environmental performance of a waste management scenario in a life cycle perspective. Assessment of several scenarios can be used to compare the performance of alternative systems or with defined criteria.

Applications of LCA are presented in the following.

Strategic planning and decisions

LCA can be applied to compare the environmental performance of alternative systems that shall fulfil a specific service function. This can be e.g. industrial production systems, transport systems or municipal waste treatment systems. The later application example will be the focus of this guideline.

LCA can help organisations responsible for municipal waste flows, or suppliers of waste treatment systems, to understand the pros and cons of their own systems, and it can identify focus areas for system performance improvement, data quality improvement and reporting.

Product development

LCA ensures that the whole product life cycle is taken into account. This means that an overall product environmental performance improvement can be achieved. By combining LCA with product quality assessment, improved environmental performance can be achieved without compromising the overall quality of the product. A life cycle approach lies inherently in the eco-design concept. Eco-design is promoted through several large industry corporations, designer organisations and through the New Approach legislation in EU (product focus) such as the *Directive on Electrical and Electronic Equipment* (EEE) which is under preparation. Regarding waste management, eco-design principles adopt goals such as:

- Design for cleaner production, including less production waste.
- Design for durability.
- Design for longevity.

- Design for reuse and recycling (simple disassembly, reduced material complexity, use of recyclable materials, component recovery through closed loop re-manufacturing and secondary application).

The eco-design concept is well documented and guidelines exist both general and for specific product groups (see. e.g. the new publication *Sustainable Solutions. Developing Products and Services for the Future /62/*). ISO has its own Work Group TC 207/WG 3 named *Integrating environmental aspects into product development (DFE)*.

Acquisition and procurement

Traditionally, acquisition and procurement processes balance the functional performance of products against factors such as cost, quality and service. LCA can play an important and synergistic role with existing processes. LCA will identify alternatives that provide decreased environmental burdens. This is particularly relevant for large consumers such as large companies, public administration and development projects.

In a waste management perspective, LCA will identify products with low degree of waste generation through their life cycle.

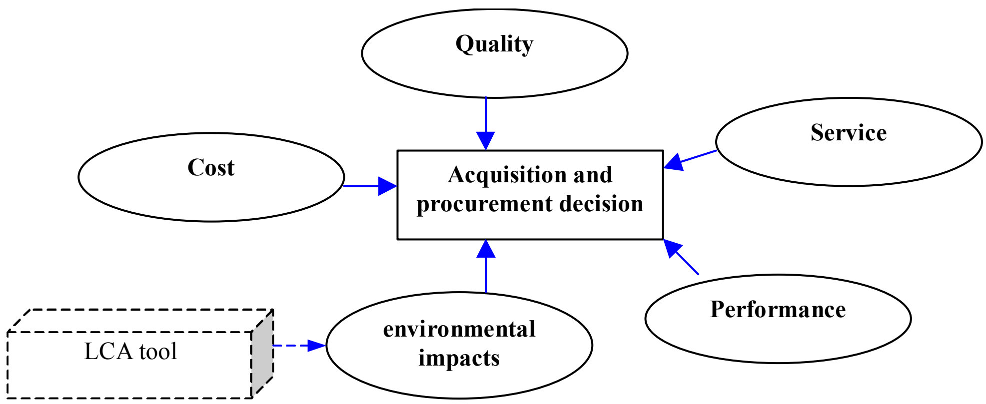

Figure 2-5 Acquisition and procurement decision strategy

Product environmental labelling and declaration

Product environmental labelling and declarations have the goal to:

- Stimulate changes in consumer behaviour that will ultimately lead to meaningful and measurable improvement in the environmental aspects of consumer products.
- Communicate accurate, verifiable, and non-deceptive environmental information to consumers to help them make product choices.
- Educate consumers about the environmental aspects of products.

LCA of products can in this context be used to:

- Give background information that enables the labelling program to set labelling criteria which ensure that labelled products can be called “environmentally preferred” in a meaningful way. Input to Type I environmental labelling (see ISO 14024).
- For specific environmental claims, ensure that the attribute to be communicated is environmentally relevant in the context of the product’s life cycle. Input to Type II environmental labelling (see ISO 14021).
- For product information programs, ensure that meaningful and environmentally relevant information about the life cycle is provided to the consumer. Input to Type III environmental labelling (see ISO 14025).

E.g. the EU eco-label award scheme and the Swedish and Norwegian environmental product declaration programs require LCA.

Policy and regulations

LCA can be used in pilot applications that primarily involve assessing technological alternatives for research development and for rulemaking.

E.g. can LCA, in a waste management perspective, be used to assess the environmental burdens of waste treatment alternatives and, combined with findings from other studies, introduce research programs, regulations and/or incentives that promote the better alternatives.

3 GUIDELINE FOR LCA APPLIED FOR RESIDUAL MUNICIPAL WASTE AND SLUDGE TREATMENT ALTERNATIVES

3.1 Prioritised topics for the guideline

In the Nordtest document on status of LCA in the waste management sector, prioritised research and development areas are identified /5/. These areas are also prioritised issues in this guideline. The prioritised issues are:

- System boundaries: Limitations of the system to be studied.
- Allocation: Substitution of energy and material by recycling and recovery.
- Inventory data: Emissions from landfill, incineration and biological treatment; emissions from compost and sludge used in agriculture; how to take into account long term emissions.
- Impact assessment: Identification of impact categories of particular interest for waste management studies; characterisation factors for toxicological impacts.

The SETAC-Europe LCA Working Group “Data Availability and Data Quality”, subgroup “Energy, Transport and Waste Models” has finalised a report with recommendations and references concerning waste (and energy/transport) /18/. The report identifies waste incineration, landfill, composting/digesting and recycling as the main waste treatment processes.

In this guideline the focus is on mixed municipal waste. Less focus is put on pure material recycling processes, e.g. for paper, plastics, glass and metals. Although, treatment of biowaste is included because the treatment alternatives for this fraction are not as established. Prioritised processes are incineration, landfill and composting/digestion.

3.2 Function and functional unit

Definition of the functional unit is a part of the *Goal and Scope phase* of the LCA methodology. The primary purpose of the *functional unit* is to provide a reference to which the input and output data are normalised (in a mathematical sense). Therefore the functional unit shall be clearly defined.

Having defined the functional unit, the amount of product necessary to fulfil the function unit must be quantified. The result of this quantification is *the reference flow*.

Comparison between systems shall be made on the basis of the same functional unit, and the functional unit should to the extent possible reflect all functions of the product. If two different waste treatment systems are being compared the functional unit should be *ton waste of a specified composition*.

The calculation could be made based on the average annual amount of waste treated. This period of time reflects all activities that cause environmental impacts, including non-continuous activities like maintenance. A longer period can be selected if activities that are important in an environmental perspective occur less frequently than once per year (e.g. accidents).

The reference flow will be the amount of waste treated over the defined period of time.

The environmental impacts caused during the defined period of time is then normalised by the reference flow. The result will then be environmental impacts per ton waste.

The functional unit should also reflect waste quality by defining the waste composition that the study is relevant for.

Summarised, the functional unit for municipal waste should then take into account:

- The period of time to which the environmental impacts and waste generation should be related. Note that this is *not* the same as the time perspective of the emissions (which can be centuries after the treatment) and the resulting impacts (which can be centuries after the emissions' occurrence).
- Amount of waste generated
- Composition of waste. One cannot compare treatment alternatives if the composition of waste that enters the system boundaries is significantly different.

An example of application of a functional unit for municipal waste can be (random numbers is used).

- 35 000 ton/year mixed municipal waste is treated.
- Composition is specified.
- 3 years are the period of time where all planned non-continuous activities are included. The resulting reference waste flow is 105 000 ton waste.
- Emissions and resource consumption are estimated for the reference flow based on quantification models.
- Environmental impacts are quantified and divided (normalised) by 105 000 ton, and all results are presented per ton waste.
- The results are valid for the specified waste composition and time horizon.

3.3 System boundaries

3.3.1 Unit processes and input and outputs of unit processes

The system boundaries define the *unit processes and input and outputs of unit processes to be included* in the system to be modelled.

This guideline is limited to LCA studies where the products generating municipal waste are fixed with respect to design, materials, mass/volume and consumption. With these limitations, life cycle stages and unit processes that should be taken into consideration are listed in Table 3-1. The table also gives comments and recommendations related to each issue.

Table 3-1 Life cycle stages and unit processes to be taken into consideration

Life cycle stage/unit process	Comments/recommendations valid for studies on the full function of a waste treatment system or process.
Household and/or industry distribution of waste on reception facilities	Waste bins where the waste has different destinations and/or treatment. Can be excluded from the system if common for all treatment alternatives under study.
Collection and transportation	Processes for transporting waste to treatment facilities, waste treatment products to final consumption should be included. As transport processes usually give small contributions to the total life cycle impacts, they can be excluded for ancillary materials, if not already included in ready-made cradle-to-gate inventory data for the ancillary materials. Transportation for collection of the waste will normally be important.
Production and use of fuel, electricity and heat	Important to include. See comments in next row.
Manufacture of ancillary materials	Flows are divided into *primary flows* and *secondary flows*. The primary flows are the materials that the product is built up from. The secondary flows are auxiliary materials and energy that enables an activity to be performed. Several tiers of auxiliary flows may extend further and further from the main sequence. The analyst should set criteria on how many tiers of auxiliary flows will be included. The criterion is typically set from 0-2 tiers of auxiliary flows. 0 tier means that a material flow is only identified by the input amount and not by the upstream life cycle. 1 tier means that the material flow used in a process unit is included by its upstream life cycle, but the materials used in the upstream life cycle flow is not. It is common to use ready-made cradle-to-gate data for secondary flows (cradle-to-gate is the part of the life cycle including everything from resource extraction to ready-made product, but not use and disposal). The selection of tiers is then not a relevant issue.
Waste treatment processes	Waste treatment systems consist of the degradation system and other processes like pumps, cutting equipment, pre-heating etc. It is important to include both environmental impacts related to the degradation process itself and supporting processes.
Recycling/recovery of materials and/or energy	Important to include. 1. Energy recovery from incineration. 2. Energy recovery of bio-gas from anaerobic digestion. 3. Energy recovery of landfill gas. 4. Recovery of soil improvement material from composting and anaerobic digestion. 5. Recovery of materials from recycling processes.
Manufacture, maintenance and decommissioning of capital equipment	Usually of little importance. Should only be included on request, or if the capital equipment itself is the product subject to an LCA.
Additional operations such as building lighting and heating	Usually of little importance. Should only be included on request.

It is an iterative process to identify the inputs and outputs from the process units that should be included in the study. The initial selection is typically made using data that can easily be made available. The most significant process units and inputs/outputs to focus upon in a more detailed study should be identified through established *criteria* and *sensitivity analysis*.

Criteria that are common to use to limit life cycle systems are given in Table 3-2 /2/, /11/.

Table 3-2 Criteria for limiting system

Type of criteria	**Criteria quantification**
Cut-off based on ***mass relevance.*** I.e. all inputs to a process unit that cumulatively contribute more than a defined percentage to the total mass input are included.	Always cut off the flow with the lowest contribution first, then the second lowest contribution. Continue this process until the defined percentage cut-off criterion is reached. Typical cut-off criteria are 1 - 5%.
Cut-off based on ***energy relevance***. I.e. all inputs to a process unit that cumulatively contribute more than a defined percentage to the total energy input are included.	Always cut off the flow with the lowest contribution first, then the second lowest contribution. Continue this process until the defined percentage cut-off criterion is reached. Typical cut-off criteria are 1 - 5%.
If independent expert judgement or quantification of ***environmental relevance*** allows for it.	Input and output that contribute more than an additional defined percentage to the estimated quantity of each individual data category are included. Typical cut-off criteria are 1 - 5%. E.g. flows that contribute to less than 5% of the total CO_2 emissions are excluded (if CO_2 is the only selected data category).*

*For the chemical-related toxicity categories you can not apply a fixed weight or volume-based cut-off criterion since some substances are so potent that even minute quantities can contribute significantly to the overall toxicity impact (e.g. the toxic metals).

As an example of input/output limitations we can use a process unit consisting of a waste treatment process, represented by incineration. Input data are as follows:

- Waste: 750 GJ/yr (96,8%)
- Oil: 20 GJ/yr (2,6%)
- El. power: 5 GJ/yr (0.7%)

With an energy based cut-off criterion of 1 % the el. power input is excluded. With a criterion of 3% only el. power is excluded as the sum of oil and el. power is above 3%. With a cut-off criterion of 5% both oil and el. power are excluded as inputs.

For better understanding of the system under study a flow diagram should be prepared. Figure 3-1 shows an example of a flow diagram for alternatives for treatment of municipal waste from a Norwegian community after source and central separation. Production and use of fuel, electricity and heat and manufacture of ancillary materials are not illustrated even though included in the case study.

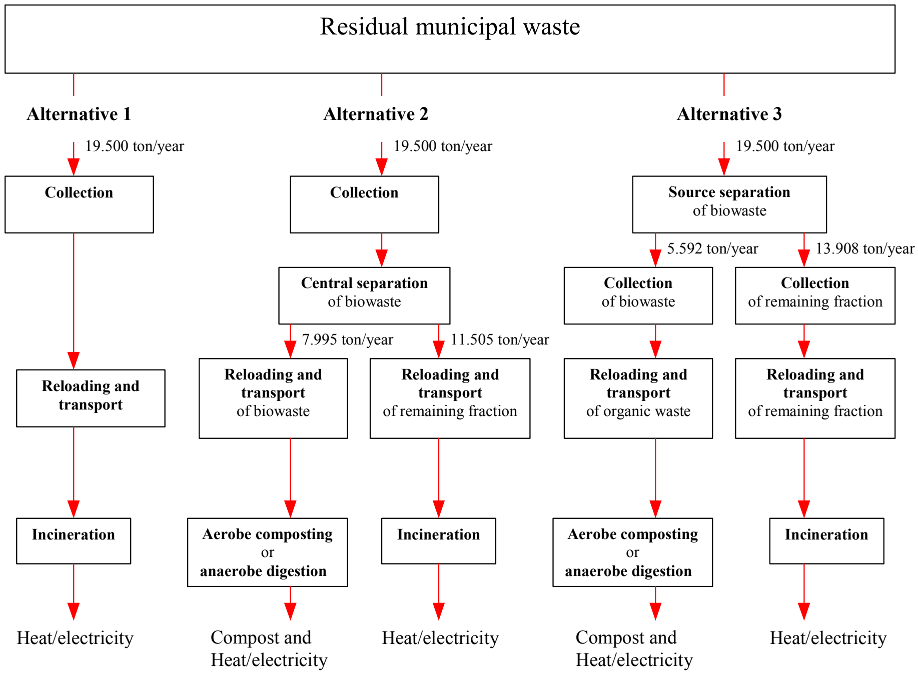

Figure 3-1 Example of a flow diagram

A description should be related to the flow diagram which explains the activities taking place within each process unit (box in the flow diagram), the type of inputs and outputs of each process unit and the locations at which the activities are taking place.

3.3.2 Upstream and downstream system boundaries

The upstream system boundaries are the boundaries that define where the technical system boundaries shall start, i.e. the cradle of each material or energy flow. The downstream system boundaries are the boundaries that define where the technical system shall end, i.e. the process that is regarded as the grave for any material flow. Processes might occur with the material after the downstream boundaries, but they are preferably insignificant or they are so long-term and unknown that uncertainty makes it difficult to include.

In the previous sub-section it was described how to set boundaries within a defined system. In this section focus will be on how to limit a system with respect to start and end of the waste life cycle.

The starting point should be the point at which the waste appears, e.g. from households. For comparison of systems it is of vital importance that the systems are defined with the same starting point and the same composition.

The systems can be further limited by excluding those parts, subsequent to the point when waste appear, which are identical in all systems. This is illustrated in Figure 3-2 where the process units within the dashed square are common for both alternative systems. These processes are therefore excluded from the systems to be studied.

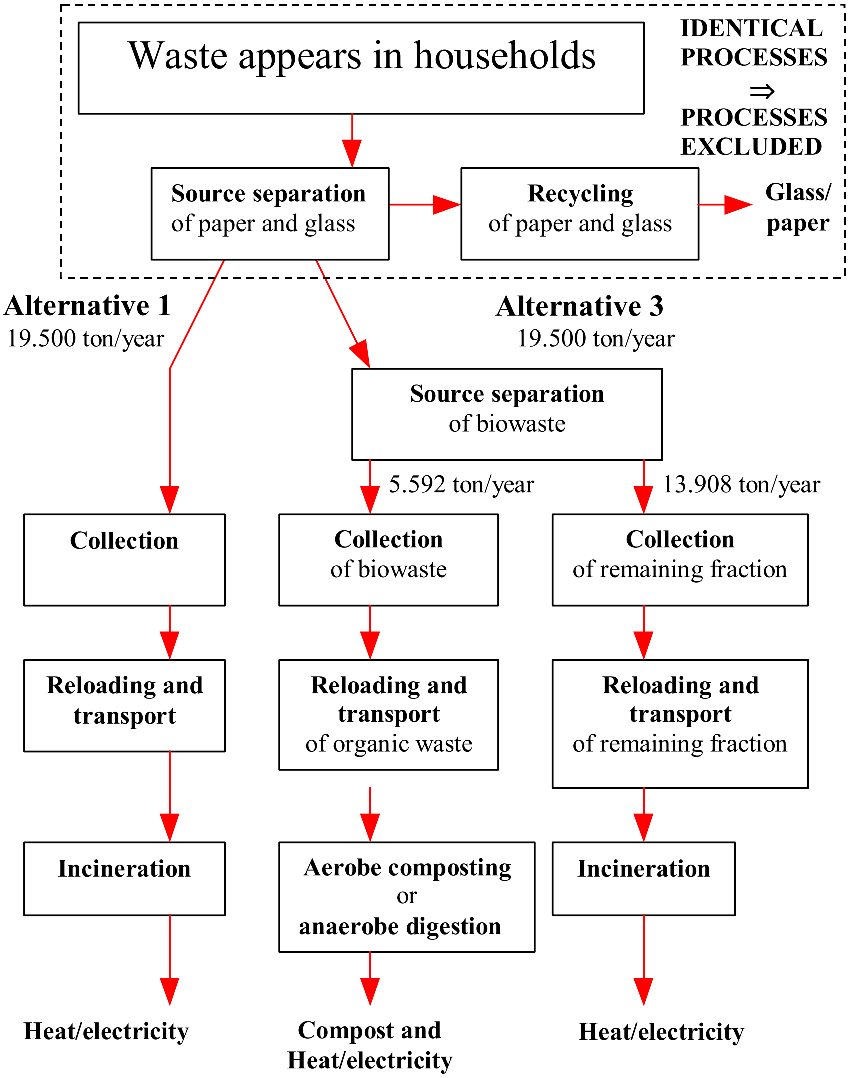

Figure 3-2 Illustration of waste life cycle upstream cut-off

The end of the waste life cycle can also seem somewhat blur. It is therefore important to provide a description that clearly defines the downstream end of the waste life cycle. Typical questions to be raised are:

- How far do we follow products from recycling (e.g. compost)?
- How far do we follow products replaced by products from recycling (e.g. fuel oil replaced by recovered heat from waste incineration)?
- For how long do we take into account emissions and resource consumption related to a landfill?

3.3.3 Products from recycling and recovery

As a minimum the products from recycling should be followed until the product is at a level where it can be regarded to replace an alternative product. If an earlier cut-off is practised, the system under study can gain more benefit from the replacement than it should have.

The product from the recycling process may also introduce large or new environmental impacts during the use stage. The recycled product use should then be included within the system together with credits for substituting the use of the replaced product.

As an example we use compost, recovered heat and recycled paper. These products can replace respectively fertiliser, oil and pulp based on wood. An illustration of the life cycle processes of the products from recycling/recovery is given in Figure 3-3.

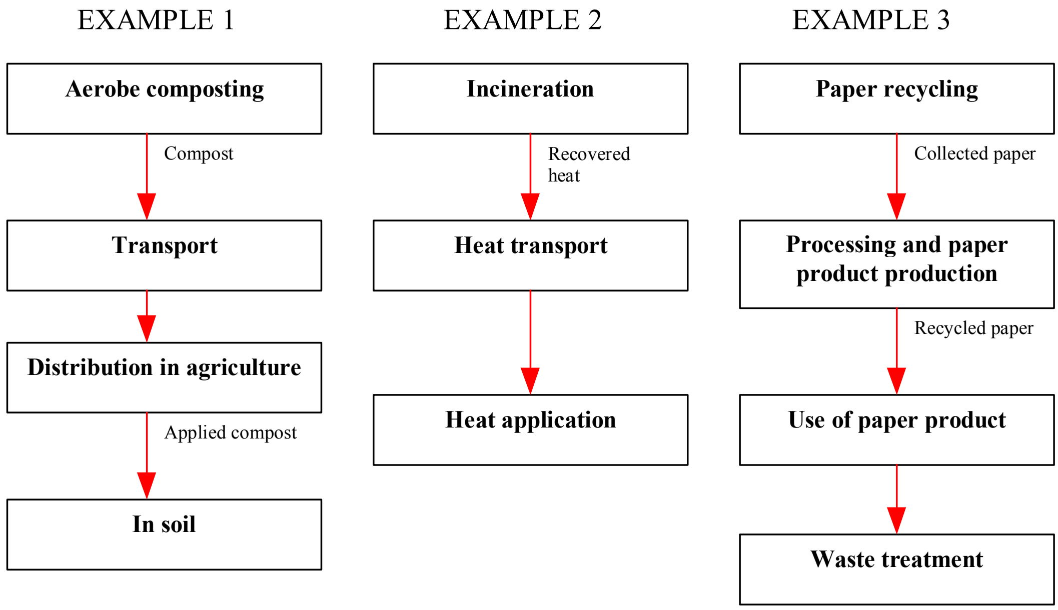

Figure 3-3 Illustration of examples of life cycle of products from recycling

Cut-off can be introduced at several levels for each of the examples in Figure 3-3. Ideally, all products from recycling/recovery should be followed until they cease to exist. However, there can be arguments for introducing a cut-off at an earlier stage. Cut-off at a stage preceding the point where the product ceases to exist can be used if:

- The environmental impacts of the remaining life cycle are of the same type and magnitude as the product to be replaced.
- The remaining life cycle gives insignificant environmental impacts compared to the total life cycle impacts.
- Data for the remaining life cycle is not available but it can be assumed that one of the previous two bullet points applies.

In all cases the reason for the chosen cut-off must be argued for.

3.3.4 Products replaced by products from recycling and recovery

It is of major importance that the product to be replaced by a recycling product and the recycled product has system boundaries that involve the same life cycle stages and are based on the same cut-off criteria. If not, the life cycle inventory analysis and impact assessment will under or over estimate the environmental burdens caused by the life cycle system. It is crucial to avoid this when making comparative studies.

E.g. if a soil improvement product derived from treatment of sludge to a certain extent replaces fertilisers, it is important that the sludge product is analysed to the same level as the artificial fertiliser. A bias would lead to a higher/lower environmental burden for the overall life cycle system than in an ideal situation where both products are described and analysed on the same level. This is illustrated in Figure 3-4. We here see that it is wrong to include all process levels for the product from recycling, while the product to be replaced includes less process levels. If the environmental impacts from the product from recycling is followed until the product is in the soil, the same should account for the product to be replaced.

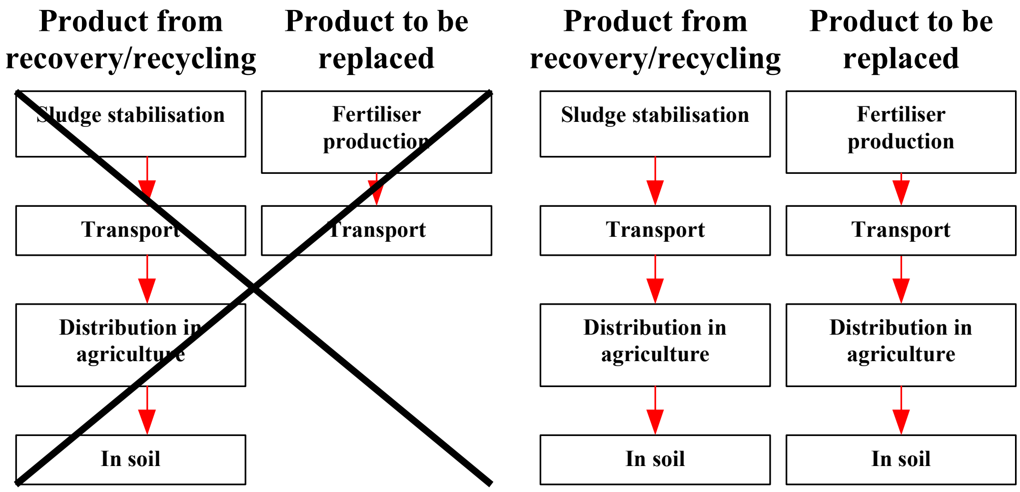

**Figure 3-4	Example of system for product to be replaced**

3.3.5 Time aspect for landfills and soil products

Waste in landfills and soil improvement products will have an impact on the environment that lasts for a long period of time. This is the case for e.g. leakage of metals and gases from degradation such as methane. The challenge to be dealt with here is to select an appropriate time interval and the time dependent emission function to be integrated over the selected time interval (see Figure 3-5).

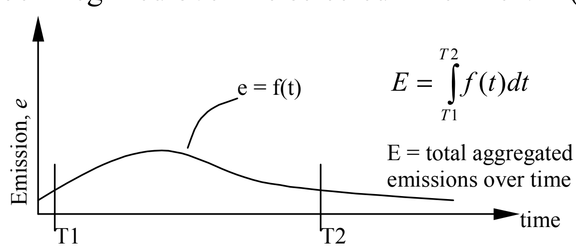

**Figure 3-5	Integration of environmental impacts from landfill over time**

The selection of a time interval is an ethical question based on the fact that by limiting the exposure time, effects on future generations will be omitted.

The ISO 14040/41 standards do not give any specific recommendations. However SETAC recommends that the emission *E* should be integrated over an infinite time period (from T1=0 to T2=∞). If this is not possible a time interval of 100 years should be applied. Third priority is any other time interval.

In Nordic LCA studies on waste, both the infinite time approach and more limited time intervals are applied. Often without any discussion of the consequences of the selected approach.

In the Danish LCA-LAND study, 100 years is used to estimate emissions from 5 different waste component categories at landfills. The model is limited to landfills in Denmark, Germany and the Netherlands. The model can be found online at http://www.ipt.dtu.dk/.

The Swedish ORWARE model describes an average Swedish landfill and divides the time for emissions into the *surveyable time period* and the *remaining time period*. The surveyable time period is until the major part of CH_4 has ceased (100 years), and the remaining time period is when all components have been released to the environment.

A clear recommendation on best practice will not be given here, although some basic decisions must be taken and made clear in an LCA report.

- Decide time intervals for different substances and processes.
- Give arguments for the selected time intervals.
- Make sure that the selected approaches are consistent.
- Discuss consequences for the results if other approaches are selected.

Normally, landfill data are not developed specifically for each study. That is usually a too comprehensive task. In stead readymade data are used, like e.g. the LCA-LAND data. It is then a satisfactory argument to use the approaches of the data source. But again it is emphasised that the approaches must be consistent throughout the study.

3.4 Inventory data

3.4.1 General

If an LCA study involves specific waste treatment processes, attempts should be made to collect and apply data that are as specific as possible for the processes in question.

In the case of more generic studies, such as e.g. a basis for political decisions, generic data should be applied. However, it is important that the generic data represent the temporal and spatial boundaries of the study.

In the following sub-sections parameters, and to a certain extent data, that are commonly applied in life cycle assessment of waste are presented. Focus will be on energy use and recovery and emissions to air water and soil. Information on other environmental aspects is given in section 3.4.10.

3.4.2 Waste composition

For all treatment alternatives for municipal waste the environmental impact from the treatment is partly a function of the composition of the waste. The collection of inventory data for waste composition is defined by the scope of the study. There are three alternative approaches for collection of inventory data:

1. To collect data on composition of the waste and the emissions caused by the treatment from the geographical area under study. This enables calculation of emission factors that are specific for the waste flow in question. Different local waste separation regimes can lead to large variations in waste composition.
2. To use average “default” waste composition. This allows for the use of already developed related average emission factors. Such an approach is recommended in less comprehensive screening studies.
3. Not use waste composition, but limit the study to “average” municipal waste. This will limit the study to be process specific and does not allow for calculations that shall reflect how changes in waste composition affect the LCA results.

Data for composition of waste with respect to waste fractions is developed on national level in many countries, for specific waste treatment plants and for specific municipal waste management companies (see e.g. table 3.3 for different waste fractions). It is important to make sure that the collected data reflect the source separation that is valid for the study. E.g. if an LCA is to be performed of the treatment chain related to a municipal waste flow with source separation of paper, it is important to collect composition data that reflect such a separation regime.

Composition is also related to the content of the basic chemical compounds such as carbon (C), nitrogen (N), sulphur (S), chloride (Cl) and metals. The contents of these compounds in the municipal waste partly decide the amount of pollutants emitted to air, water and soil. By developing factors for emissions to air, water and soil related to treatment method and technology a model that predicts emissions can be developed. However, such a model is not able to reflect changes in waste fractions. To be able to do that the content of chemical compounds must be given per waste fraction, like in /9/ or /15/.

<u>Process and product approach</u>

These two phrases are used to describe two different emission modelling approaches.

- The **process approach** uses ready-made emission and resource consumption factors for different waste treatment methods and underlying technological variations. Data might also be developed for different types of waste. I.e., if you define waste treatment method, technology and waste type, then a ready-made factors that fits with the definition are applied.
- The **product approach** uses waste treatment method specific models that calculate emission and resource consumption factors based on waste composition (waste fraction composition and contents of the waste fractions).

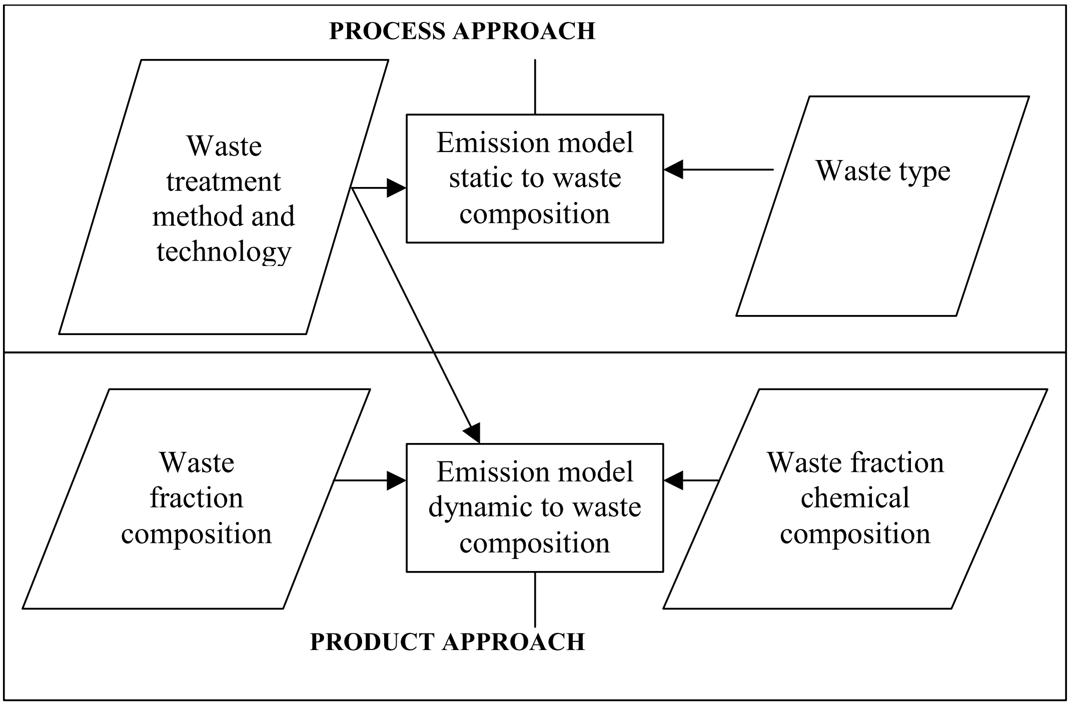

Figure 3-6 Illustration of inputs to models predicting waste treatment emission factors

3.4.3 Incineration

3.4.3.1 System description

Data of interest related to waste incineration are:

- Type of technology. This influences how pollutants are distributed on emissions to air and water and residuals. The technology can be separated based on type of flue gas cleaning systems and furnace types.
- Waste distributed on waste fractions (applies for the whole study). Data for the specific waste in question are sometimes available, but in most cases generic data must be applied.
- Waste fraction physical and chemical data (applies for the whole study).. Combined with data on how waste is distributed on waste fractions, this allows for development of product specific emission factors. Data for the specific waste in question are sometimes available, but in most cases generic data must be applied.
- Amount and type of support fuel. Some type of fossil fuel is usually incinerated to generate enough heat to during waste incineration start-up and to keep the incineration process stable.
- Type and amount of auxiliary materials. These materials is applied in the flue gas cleaning. Relevant auxiliary materials can be $Ca(OH)_2$, NaOH, coke, ammonia, limestone, urea and waste water treatment chemicals.
- Overview of the range of pollutants decided to be studied.
- Emission to air and water. Can be collected from specific plants or estimated from models based on waste composition and technology. Emission to water presumes that a wet flue gas cleaning system is installed.
- Process specific energy consumption and emissions such as operation of vehicles and machinery.
- Incineration residues and residue contents. Can be given specifically for the defined plant(s). or estimated based on models taking into account waste composition and incineration technology.
- Potentially recovered energy from a specific plant. Is estimated based on models that take into account heating values and dry matter in waste components and efficiency of the installation.
- Type of energy substituted by recovered energy.
- Recovered energy. Should be based on data valid for the defined plant(s).

The different topics listed above are described in further detail in the following chapters in this guideline. An overview of a system model for incineration is given in Figure 3-7. Note that some process units are given at a coarse level. These can be further refined (e.g. the deposition boxes, the production chain boxes and the incineration process itself). Further, emissions, resource consumption and energy use flows are not shown for the process units (except incineration). Finally, transportation is excluded.

All the flows in the system will be relative to the municipal waste flow entering the system. This municipal waste flow is the whole, or a share of the reference flow resulting from the defined functional unit.

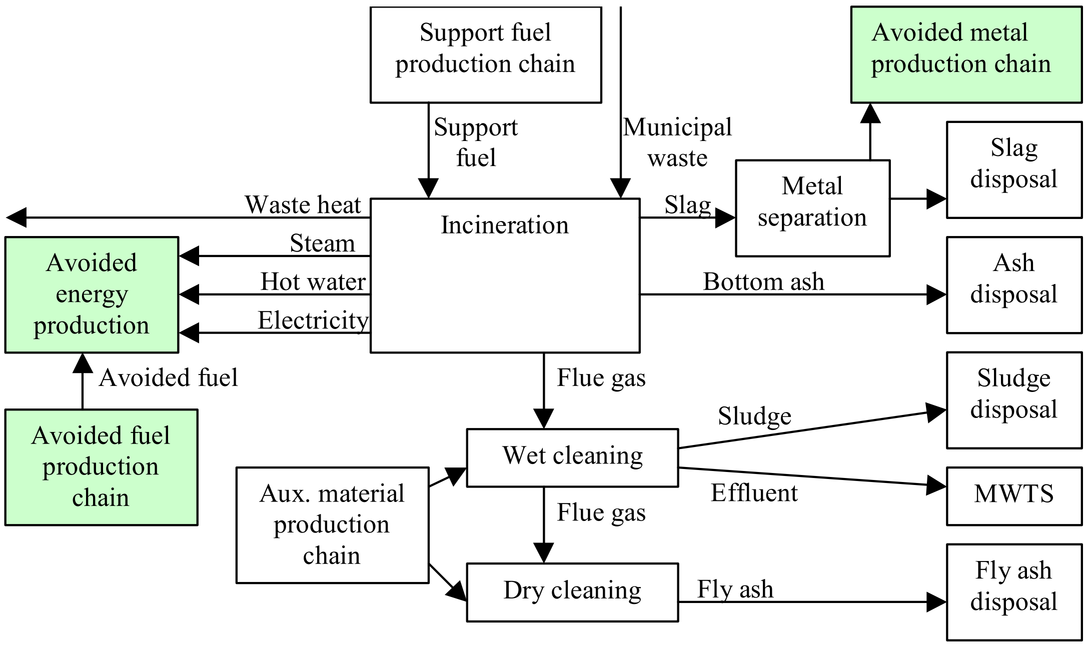

Figure 3-7 Process flow chart for the incineration system

3.4.3.2 Emission of CO_2

CO_2 emissions are estimated from the carbon content of the incinerated material. The carbon content contributes to emissions such as CO_2, CO, CH_4, non-methane volatile organic compounds (NMVOC) and carbon in soot. The relative distribution of carbon between the different components depends on the operation of the incineration plant. CO_2 is by far the component that binds most of the carbon (above 97%) /10/.

Exhaust gas cleaning or incineration technology does not influence CO2 emissions. It is therefore common to differentiate CO_2 emissions on waste composition only.

Emission of CO_2 from incineration of biological waste material does not contribute to net emissions of greenhouse gases and should therefore not be accounted for. It is therefore necessary to separate between fossil carbon and biological carbon.

Calculation of net CO_2 emissions from waste incineration is based on the fossil carbon content of the waste (kg fossil carbon/kg waste), multiplied by the amount of CO_2 generated per amount of carbon (kg CO_2/kg fossil carbon).

The fossil carbon content of different waste fractions is estimated in several Nordic studies. Table 3-3 gives data based on two selected Nordic studies.

Table 3-3 Carbon content in waste fractions (dry matter)

Reference /9/, g C /kg waste			Reference /15/, g C /kg waste		
Waste fraction	**Fossil C**	**Biological C**	**Waste fraction**	**Fossil C**	**Biological C**
Food waste	0	434	Food, garden waste	0	500
Newspaper	8	440	Wood	0	495
Corrugated cardboard	0	500	Other degradable	0	400
Mixed cardboard	170	400	Newspapers, magazines	0	422
PE	856	0	Milk cartons	125	375
PP	855	0	Mixed cardboard	0	422
PS	889	0	Other paper	0	433
PET	640	0	Napkins, coffee filters	0	458
PVC	401	0	Diapers	0	500
			Plastic foil	644	0
			Hard plastic packaging	656	0
			Other plastic	590	0
			Textiles	278	278
			Fine matter	75	75
			Vacuum cleaner bags	150	150
			Other combustible	400	93

Further, /9/ and /15/ give the following amount of CO_2 generated per amount of carbon:

- /9/: 3,67 kg CO_2/kg fossil carbon (equivalent to 100% conversion)
- /15/: 3,49 kg CO_2/kg fossil carbon (takes into account conversions to other substances)

Based on the composition of the waste (examples given in section 3.4.2), fossil carbon content of the waste fractions (as given in Table 3-3) and kg CO_2/kg fossil carbon, the amount of CO_2 per amount of mixed municipal waste is calculated (kg CO_2/kg waste).

If the composition of the waste is not known or does not fit with the given/available waste fraction carbon content data, average municipal composition figures can be used, at least in a screening study. E.g. Norwegian figures indicate average emissions from incineration plants of 0,29 kg CO_2/kg household waste (including water content in the waste) and 0,43 kg CO_2/kg household waste (dry matter) /15/.

3.4.3.3 Emissions to air (not CO_2)

Other emissions to air vary with age, incineration technology, flue gas treatment technology and composition of the waste. Data can be retrieved in several ways:

1. Process specific data from one incineration plant: This is recommended in cases when the analyst knows that the waste is going to be incinerated in the particular plant where the data is derived from, or in a similar plant. The data are static and they are not able to reflect situations where the composition of the waste is changed.
2. Average process specific data from several incineration plants: This is recommended in studies with a broader geographical scope. E.g. in case of national or regional studies. It is then important that the selected plants represent

the variation in technology in use within the boundaries in question. These data are also static and they are not able to reflect situations where the composition of the waste is changed.

3. Product specific data: Emission data are modelled based on knowledge about waste composition. Should be applied for system development purposes, where the effect of system changes shall be quantified (e.g. the effect of introducing source separation of a material).

Emissions to air from a waste incineration plant applied in an LCA of waste should be given as weight pollutant emitted per weight waste incinerated (e.g. *g NO_x/kg waste*). When deriving plant specific data, these are usually given at one of the formats listed in Table 3-4 (example with NO_x as pollutant is used). How to estimate *g NO_x/kg waste* from the given format is also given in the table.

Table 3-4 Formats for emission to air data and related estimation to derive at wanted format

Format	Estimation	Comment
g NO_x/m^3 exit gas	Multiplied with exit gas rate at same conditions (m^3 exit gas/h) and divided by waste flow (kg waste/h).	At specific conditions for ambient pressure and temperature, and exit gas O_2 content. Exit gas rate are usually measured at existing plants and can be estimated for new plants
g NOx/h	Divided by waste flow (kg waste/h).	
g NOx/kg waste		Wanted format

It should be possible to retrieve plant specific data on emissions to air for that are regulated by the authorities, as these data are publicly available. Usually the regulated parameters are those with limit values according to EU directive on the incineration of waste /12/. The directive gives the minimum requirements that *new* waste incineration plants have to comply with. Existing plants have to comply with the directive within the end of 2005. The directive gives requirements as concentrations in the exit gas at defined conditions. The requirements of the directive can be regarded as a worst case for emissions to air from a waste incineration plants that must be compliant with the directive or related national legislation.

Limitations on which emissions that shall be taken into account are made in the scope of the LCA. If there are no arguments for restricting the number of pollutants, the parameters regulated in the EU directive should at least be taken into account in addition to CO_2 in the scope when studies include waste incineration. The parameters are:

- Dust (can be further specified by dividing into particle size; PM_{10}, $PM_{2,5}$)
- TOC (can be further specified into chemical components or groups of these)
- HCl
- HF
- SO_2
- NO_x
- CO
- Cd+Tl
- Hg

- Metals (includes Sb, As, Pb, Cr, Co, Cu, Mn, Ni and V)
- Dioxins/furans

Table 3-5 gives emission measurements made in 1999/2000 at an existing waste incineration plants in Norway /13/. It is seen that there are great variations in the emission factors, mainly due to different pre-treatment, incineration and cleaning technologies. All plants incinerate mixed municipal and industry waste. Still, there might be large differences in waste compositions probably also affect the emission factors.

Table 3-5 Emission to air from waste incineration in Norway, 1999/2000 /13/

	Unit	**Frevar**	**BiR**	**Nir**	**Ålesund**	**Heimdal**	**Klemets-rud**	**Brobekk**
Start year		1984	2000	2000	1987	1986	1985	1987
Waste flow	ton/yr	70000	90000	30000	37000	90000	160000	100000
Relative to waste								
Dust	g/ton	2,1	7	2,29	42,9	91,1	14,3	79,5
Hg	g/ton	0,001	0,01	0,008	0,11	0,11	0,273	0,088
Metals*	g/ton	0,69	0,07	0,1	0,64	0,4	23,4	9,14
CO	g/ton	252	32,5	1,14	1	333	250	0,22
TOC	g/ton	1,4	2	3,43	0,56	28,9	3,19	19
HF	g/ton	0,28	0,7	1,14	4,82	0,48	1,95	2,29
HCl	g/ton	2,8	4,5	11,4	0,08	0,08	146	42,3
SO_2	g/ton	21	14,5	74,3	155	5,56	78,4	386
NO_x	kg/ton	2,54	0,67	0,41	1,53	1,47		2,26
Dioxins	mg/ton	0,029	0,06	0,046	2,23	4,11	11,7	13,9

* Includes Cd, Hg, Tl,, As, Pb, Cr,, Cu, Mn and Ni

If it is preferred to model the emissions to air, in stead of using data from existing plants like those in Table 3-5, the model developed by the Danish part of the EUREKA project on technical data for waste incineration can be applied /10/. Here emission factors are established for the relevant compounds for a range of cleaning technologies, given that the waste content of C, N, S, Cl, and metals are known.

By selecting data from specific plant as those listed in Table 3-5 (or other plants), a *process specific* data collection approach is applied. The benefit of such an approach is that it is easy to derive updated data. By using the approach from the EUREKA project, both a product specific and process specific approach is selected. This approach requires that the contents of the waste are known and the flue gas cleaning technology to be applied.

A product specific approach is also applied in the Swedish ORWARE project /9/. This model is however simpler than the EUREKA model because there are less possibilities for variation in technology.

3.4.3.4 Emissions to water

Emissions to water from waste incineration are only related to plants that have wet exhaust gas cleaning systems. The wastewater is then released to the municipal waste water system and treated at the local wastewater treatment plant (see Figure 3-8). Some studies only follow the emissions until after the effluent cleaning. This is acceptable if there are no subsequent treatment, but if treatment is present it should be included so that the actual effluent to the recipient is quantified.

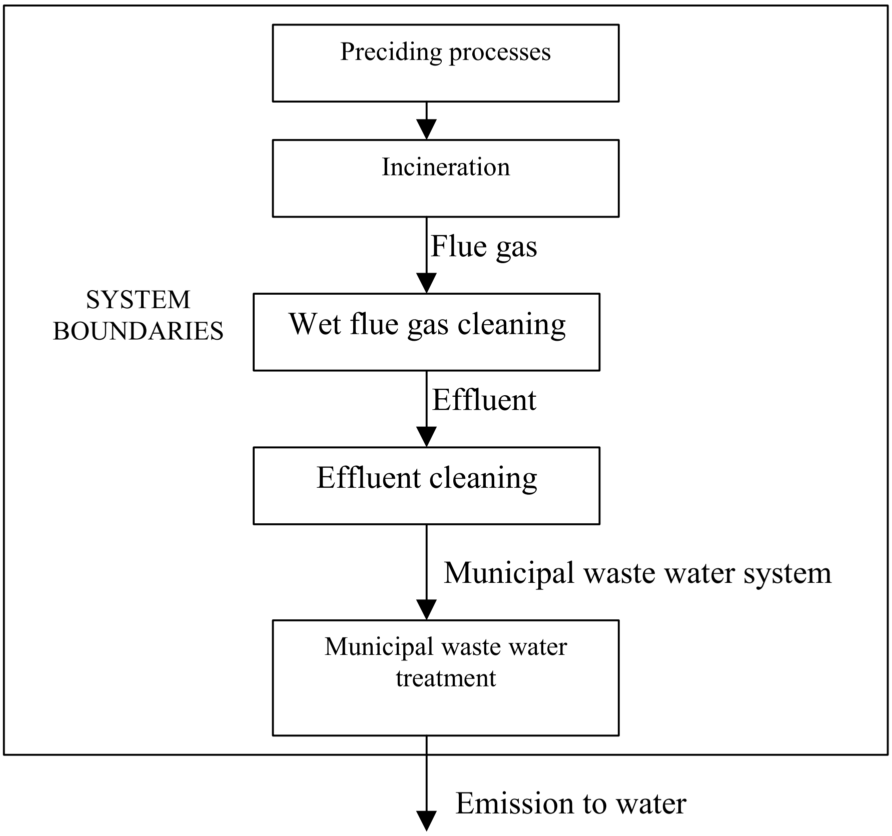

Figure 3-8 Illustration of flows, process units and system boundaries from incineration to emission to water

In general, no studies are identified during the preparation of this guideline that quantify the contribution from effluent cleaning to the total emissions from municipal wastewater treatment plants to natural water resources. However, plant specific data can be derived for emissions that leave the plant into the municipal waste water system. These data must only be used combined with careful evaluations and discussions about how the effluents affect the local wastewater treatment plant and their fate after the treatment.

In the same way as for emissions to air, the EU directive on incineration of waste /12/ can give input to the scope on pollutant to include in the study when wastewater from exhaust gas cleaning is involved as a process in the system. The requirements are given for suspended substances, Hg, Cd, Th, As, Pb, Cr, Cu, Ni, Zn and dioxins and furans as mg/l wastewater.

If data for these compounds are collected they should be applied by multiplying with the amount of wastewater emitted per ton waste incinerated (l waste water/ton waste). The compound emitted through the wastewater emissions can then be expressed as mg/ton waste.

3.4.3.5 Incineration residues

Incineration residues come in the form of slag (unburned materials), bottom ash (ash collected at bottom of kiln), sludge (from wet flue gas cleaning system) and fly ash (collected at dry flue gas cleaning systems). The type and amount of residues generated are of course very much dependent on the technology applied at the incineration plant.

The incineration residues contain metals and dioxins/furans that may leak into the environment if not handled properly. The residue fractions with the highest concentrations of toxic compounds are usually regarded as hazardous waste (e.g. fly ash) and should be treated accordingly. Process units that should be taken into account when quantifying the environmental impact of residues are illustrated in Figure 3-9.

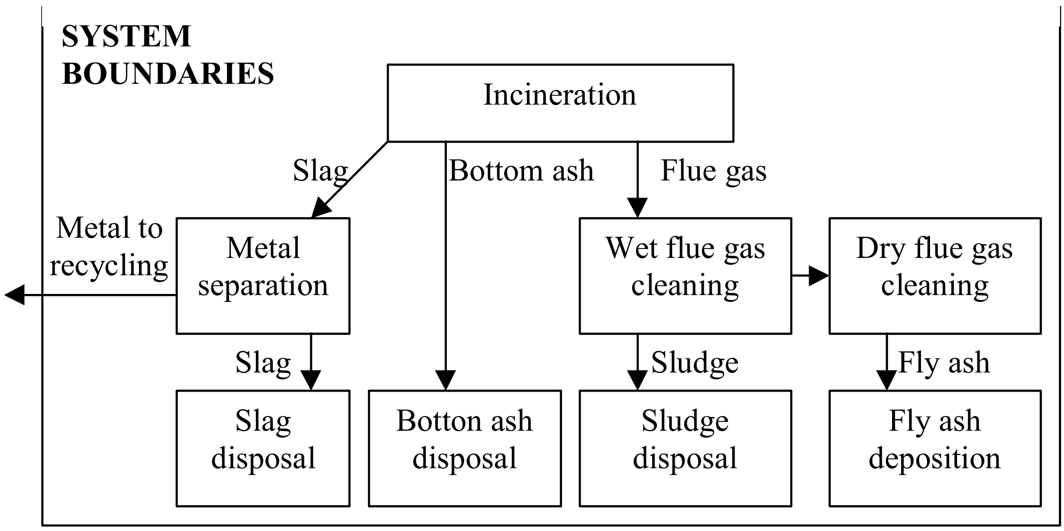

Figure 3-9 Illustration of flows, process units and system boundaries from incineration to final disposal of residues

In many LCAs the system boundaries are set in a way that incineration residues are only quantified as "waste". By using such an approach much information is lost. This might be information such as collection and recycling of metal contents of the slag, and how the remaining residues are deposited and what impacts to the environment that are caused by this deposition.

To be able to quantify the potential environmental impact of incineration residues a process approach or a product approach can be selected, as for all other flows in and out of the incineration plant.

In a process approach the amount of residues are measured (kg residue/ton waste), together with the content of chemical compounds (g compound/kg residue). Then the leakage and land occupation can be estimated based on leakage rates and land occupation factors for different deposition methods. This approach is not able to reflect changes in the waste composition.

In a product approach the amount of residue and its contents of basic chemical compounds are modelled. The modelling takes into account waste fraction distribution, content of chemical compounds in the waste fractions, and an input/output mass balance for different technologies. The product approach is able to reflect changes in the waste composition.

Parameters that should be taken into account regarding the environmental impact of deposition of residues should at least be those toxic compounds regulated by authorities. In addition land-use is an important parameter in many LCAs.

The most comprehensive Nordic work on modelling the environmental impact of incineration residues is probably the LCA-LAND project (ref. http://www.ipt.dtu.dk/ap/lceresearch.htm). The model is developed for landfills and can be applied for deposited incineration residues. It is based on a large number of assumptions and approximations concerning landfill properties, waste product properties and characteristics of various kinds of environmental protection systems (e.g. landfill gas combustion units and leakage treatment units). The model is useful for estimation of emissions from waste products disposed in landfills and it has been made operational in the computer tool LCA-LAND. In the model, waste products are subdivided into five groups of components: general organic matter (e.g. paper), specific organic compounds (e.g. organic solvents), inert components (e.g. PVC), metals (e.g. cadmium), and inorganic non-metals (e.g. chlorine,) which are considered individually. The assumptions and approximations used in the model are as far as possible scientifically based, but where scientific information has been missing, qualified estimates have been made to fulfil the aim of a complete tool for estimation of emissions. Due to several rough simplifications and missing links in the present understanding of landfills, the uncertainty associated with the model is relatively high.

3.4.3.6 Recovered energy

Recovered energy ratio is the exploited energy from the incineration plant divided by the energy produced by the plant that can potentially be exploited. Energy is exploited as steam used in industrial processes, hot water used in district heating and electricity production.

The recovered energy ratio is varying considerably from plant to plant and over the year. Annual variation is usually a result of variations in ambient temperature which influence the need for district heating. However, the energy recovery can be optimised by adjusting the amount of waste incinerated. This requires an intermediate storage of the waste during the summer season.

In Sweden the recovered energy is close to 100% due to a comprehensive use of district heating using hot water.

In Norway the energy recovery lies around 70% on average (varied from 50-84% in 1999/2000 for existing plants) /13/. Norway does not have much district heating, and therefore the potential for energy recovery is lower than e.g. in Sweden. Although, increased district heating is a national target. Steam to industrial processes is the most important form of energy recovery in Norway.

3.4.4 Landfills

3.4.4.1 System description

The landfill system is relevant to apply to both to the direct municipal waste flow, and to residual waste flows resulting from other treatment methods, such as incineration and biological treatment. Data and information normally applied in LCA landfill emission models are:

- Overview of landfill technologies applied in the temporal and spatial boundaries of the study. This is mainly related to types of leakage water prevention (e.g. membranes of different leaking potentials) collection and treatment and collection, combustion and energy exploitation of landfill gas.
- Distribution of main waste flow on waste fractions (applies for the whole study).. The main waste flow should be separated on waste fractions that act differently in the landfill and give significantly different type and magnitude of emissions and gas production.
- Waste fraction contents (applies for the whole study).. This gives the substances available for pollutant and product generation.
- Overview of the range of pollutants decided to be studied.
- Time period for estimation of emissions. Some emissions can occur over a very long period of time. The time period is a temporal cut-off.
- Product specific models for potential generation of pollutants and distribution of pollutants on environmental compartments. The models must match the time periods selected for generation of pollutants and generation of impacts.
- Share of leakage water and gas collected and treated. Duration of collection and treatment might be relevant in the future as models depending on duration might be developed.
- Leakage water and landfill gas treatment technology emission factors.
- Process specific energy consumption and emissions such as operation of vehicles and machinery.
- Recovered energy, which is estimated based on models that take into account produced gas, share of gas collected, heating value of gas and the efficiency of the installation(s).
- Type of energy substituted by recovered energy.

A process flow chart for a landfill is given in Figure 3-10. Note that some process units are given at a coarse level. These can be further refined (e.g. the avoided energy and related production chains). Further, emissions, resource consumption and energy use flows are not shown for the process units. Finally, transportation is excluded.

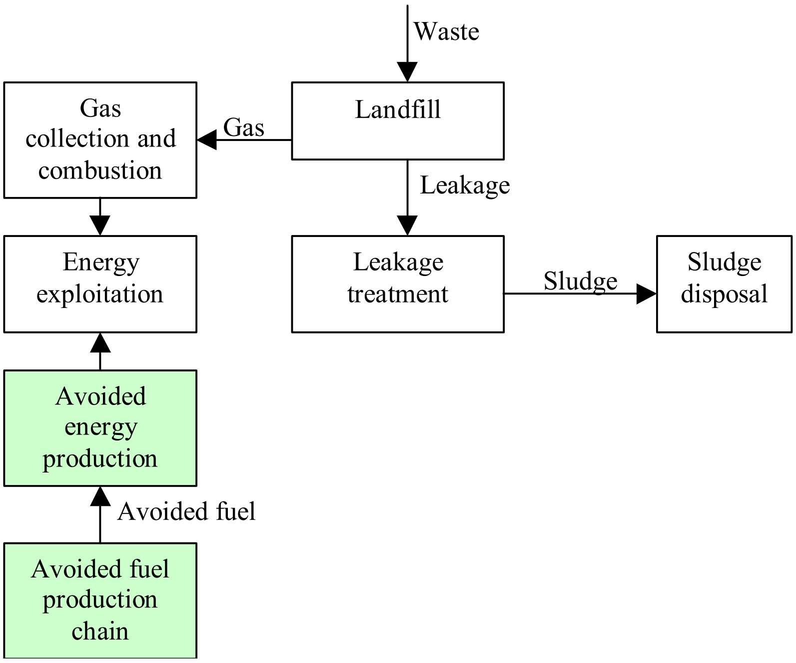

Figure 3-10 Process flow chart for the landfill system

A particular pollutant emitted to an environmental compartment after a specific emission treatment ($E_{pollutant,treatment,compartment}$) can be expressed as:

$$E_{pollutant,treatment,compartment} = (M \cdot C \cdot F \cdot G \cdot A \cdot R)_{pollutant,treatment,compartment}$$

Where: M is the waste flow input to the landfill (kg/functional unit)

C is the content of the basic substance in the waste forming the pollutant (g $_{substance}$/kg $_{waste}$)

F is the share of the basic substance going to the compartment of interest (g $_{compartment}$/g $_{substance}$)

G is the generation potential of the pollutant from the basic substance (g $_{pollution\ pot.}$/g $_{substance}$)

A is the share of the pollutant that is treated (g $_{treated}$/ g $_{pollution\ pot}$)

R is the emission reduction factor for the pollutant after treatment (g $_{removed}$/ g $_{treated}$)

Pollutants that are not treated are given by:

$$E_{pollutant,non-treatment,compartment} = (M \cdot C \cdot F \cdot G \cdot (1 - A))_{pollutant,non-treatment,compartment}$$

The sum of the two equations above expresses the total emission of a pollutant to a given compartment.

However, note that some pollutant emissions can not be modelled this way. E.g. amount of VOC and BOD to treatment should rather be related to the amount of waste, rather than a basic substance generation potential.

As an example let us define the following task: Calculate landfill emission to water given that:

- 10000 kg waste is delivered to landfill per functional unit (FU) (M=10000 kg/FU).
- Focus on cadmium (Cd). The Cd content of the waste is 0,1 mg/kg waste (C= 0,1 mg/kg).
- Cd in waste is released to water as Cd, i.e. pollutant equals the substance (G=1 mg/mg).
- 10% of the Cd ends in landfill waste water after 100 years. The remaining Cd remains in the landfill. (F=0,1).
- 80% of the landfill wastewater is treated. (A=0,8).
- The landfill wastewater treatment reduces the Cd content of wastewater with 90%. The removed Cd ends in treatment sludge (R=0,1).

The emission to water from the landfill is then given by:

$$
\begin{aligned}
E_{cd,water} &= E_{cd,water,treatment} &&+ \quad E_{cd,water,non\text{-}treatment} \\
&= (10000*0,1*0,1*1*0,8*0,1) &&+ \quad (10000*0,1*0,1*1*(1-0,8)) \\
&= 10 &&+ \quad 20 \\
&= \underline{30\ \text{mg/FU}}
\end{aligned}
$$

The Danish LCA-LAND model /31/ is a product specific landfill model, based on a large number of assumptions and approximation concerning landfill properties, waste product properties and characteristics of various kinds of environmental protection systems (landfill gas combustion and leakage treatment). This is probably the most comprehensive work in Europe related to product specific emissions from landfills. The model can be used as a basis to establish emissions per ton waste from waste composition and the waste fraction's content of pollutants.

The model calculates emissions to air, water, what remains in the landfill after 100 years and recovered energy. It takes into account all the input data given in the bullet list above, except the latter one related to substituted energy. Most of the input parameter can be varied. The only standard parameters are:

- The time period.
- Fraction of leakage treated at landfills with leakage treatment units (80%).
- Fraction of gas collected at landfills with combustion plants (50%).
- Fraction of precipitation entering landfills equipped with water stopping top covers (5%).
- Fraction of precipitation entering landfills without water stopping top covers (50%).

The model contains default values for The Netherlands, Denmark and Germany. It can be applied for other countries and regions as well as long as the necessary input data are available. LCA-LAND should be regarded as a model which processes input data given by the analyst into inventory results, not a data source.

3.4.4.2 Emission of CO_2 and CH_4

Main focus has been placed on the bulk emissions to air, which is the greenhouse gases methane (CH_4) and carbon dioxide (CO_2). It is commonly assumed that approximately the first months there are aerobic conditions in the landfill, which means that CO_2 is formed. After that there are anaerobic conditions, which means that CH_4 is formed in addition to CO_2.

As discussed in section 3.3.5, the time period taken into account partly decides the CH_4 and CO_2 generated per ton waste. As a starting point the carbon content in the waste flow available for degradation decides the potential emissions of CO_2 and CH_4. This should be specified during the waste composition data collection (ref. section 3.4.2). The total available carbon in the waste, minus carbon washed out with the leakage water, is available for CO_2 and CH_4 generation.

There might also be carbon left in the landfill after the defined emission time frame has run out. In a Swedish LCA study on solid waste /9/ the biological share of these amounts of carbon are transformed in CO_2 equivalents which are regarded to decrease the contribution to global warming (the carbon sink concept). I.e. the landfill is a carbon sink if carbon is not released to air or water, but remains in the landfill for an infinite time. If the carbon had not ended up in the landfill it would have be released and contributed to global warming.

It is important to use a product specific approach to estimate CH_4 and CO_2 generation. This is first of all because biologically based carbon is CO_2 neutral and a product specific approach is needed to keep track of the share of biological carbon.

A share of the landfill gas is often collected and combusted. The combustion transforms most of the CH_4 into CO_2, although some minor amount can remain throughout the combustion.

Figure 3-11 shows CO_2 and CH_4 emitted from a landfill and which emissions that should be regarded as decreasing, neutral or increasing CO_2 equivalent emissions. Box B and D-H contributes to global warming, while A and C are regarded as CO_2 neutral. It is common to pay less attention to box D and H due to a very small contribution compared to the other boxes. If carbon remains in the landfill after the defined time frame (surveyable time) for emission of CO_2 and CH_4, it must be decided whether to use the carbon sink approach or not /9/. If the carbon sink approach is used, box I will decrease the contribution to global warming and box J will be neutral. If the landfill is not regarded as a carbon sink, box I will be neutral and box J will increase the contribution to global warming in an infinite time perspective. That is *if* the remaining carbon is taken into account at all. If the remaining carbon is *not* taken into account, the contribution from box I and J will be zero.

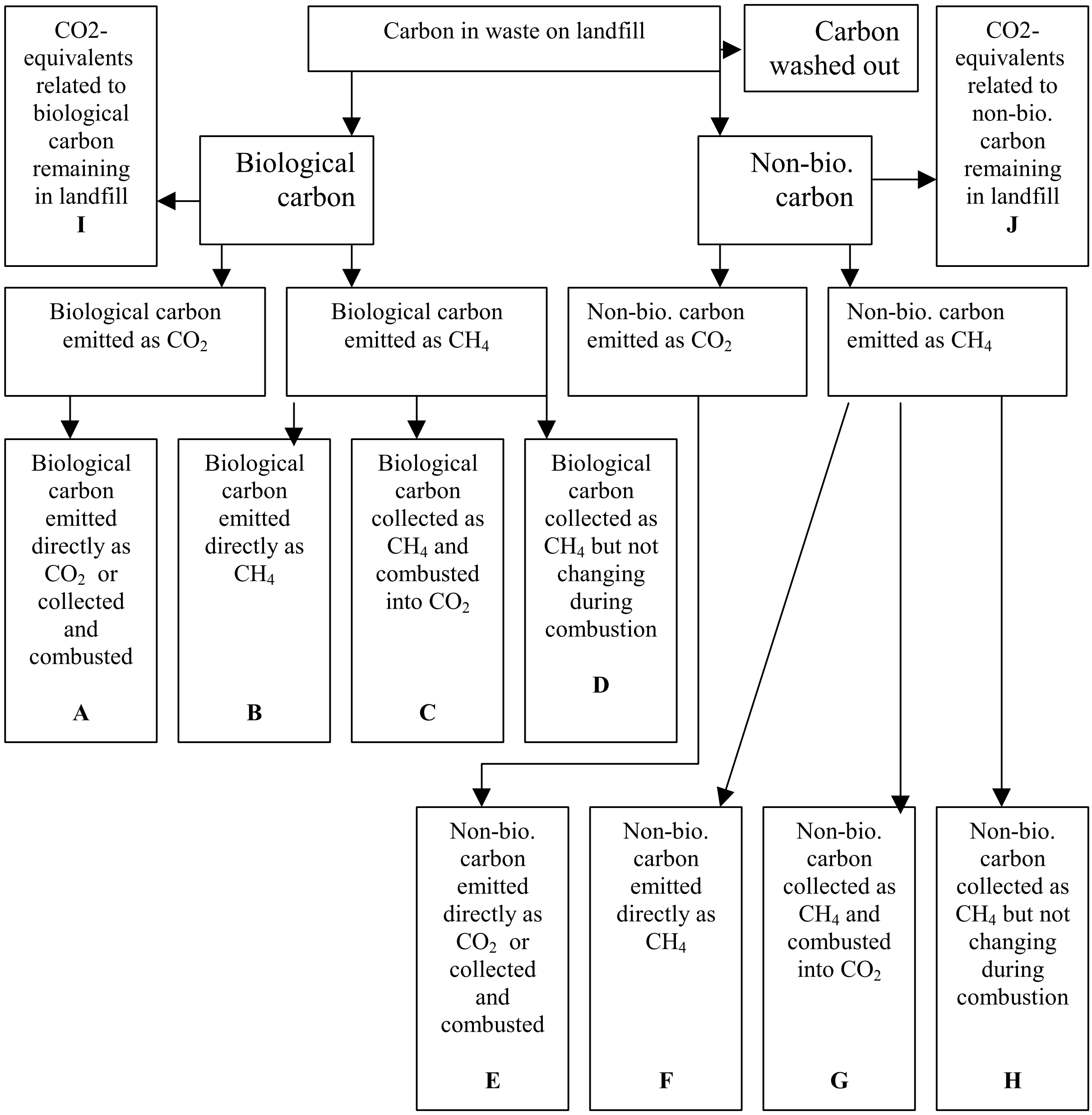

Figure 3-11 CH_4 and CO_2 emissions from landfill

The biological carbon content of different waste fractions and the related CO_2 and CH_4 emissions are given in Table 3-6 /15/. It is assumed that the biological carbon content is to equal the bio-available content. This is not always the case, e.g. lignin is biological but not bio-available. The approach applied does therefore not take into account the carbon that remains in the landfill (non-available biological carbon). Further, in the bio-available definition chemical reactions involving carbon are included in addition to the biological degradation. In sum this might overestimate the potential for gas development.

It is recommended to perform sensitivity studies on how to deal with the remaining carbon. Alternative scenarios could be:

- Not to include remaining carbon (as in Table 3-6).
- To use the carbon sink approach.

- To assume that all carbon in landfill is emitted in an infinite time perspective.

Table 3-6 Greenhouse gas emissions (kg/ton waste fraction) from landfill /15/

Waste fraction	Dry matter (%)	Bio-available carbon (% of C)	Total potential[1]		After CH_4 combustion[2]		Contribution to global warming[3]	
			CO_2 (kg/ton)	CH_4 (kg/ton)	CO_2 (kg/ton)	CH_4 (kg/ton)	CO_2 (kg/ton)	CH_4 (kg/ton)
Food, garden waste	30	100	269	99,4	400	49,7	0	49,7
Wood	80	100	739	252	1072	126	0	126
Other degradable	25	100	179	66,3	267	33,2	0	33,2
Newspapers, magazines	90	100	714	240	1031	120	0	120
Milk cartons	90	75	634	213	916	107	0	107
Mixed cardboard	90	100	714	240	1031	120	0	120
Other paper	90	100	733	246	1059	123	0	123
Napkins, coffee filters	60	100	530	169	753	84,4	0	84,4
Diapers	30	100	289	92,1	411	46,1	0	46,1
Plastic foil	90	5	53	19,0	78	9,5	0	9,5
Hard plastic packaging	90	0,1	1	0,3	1.4	0,2	0	0,2
Other plastic	100	0,1	1	0,4	1.5	0,2	0	0,2
Textiles	90	50	466	159	677	79,6	0	79,6
Fine matter	50	50	70	23,9	102	12,0	0	12,0
Vacuum cleaner bags	100	50	280	95,5	406	47,8	0	47,8
Other combustible	75	20	138	47,3	201	23,7	0	23,7
Glass	100	100	9	3,3	13.4	1,7	0	1,7
Iron	100	10	8	2,9	11.8	1,5	0	1,5
Other metals	100	0,1	0	0	0.0	0.0	0	0.0
Other non-combustible	100	0,1	0	0	0.0	0.0	0	0.0

1. 57% of carbon is emitted as CH_4 for food, garden waste and glass, while 50% for the remaining waste fractions. Only CO2 is generated the first half year. Complete degradation of bio-available carbon in the landfill. Degradation of other types of carbon like lignin and those used in plastics are not taken into account and are assumed to be remaining in the landfill.
2. 50% of the methane is collected and combusted. 0,054 kg CO2 is emitted per MJ methane combusted. Methane has here a heating value of 49 MJ/kg (2,65 kg CO_2 pr. kg CH_4).
3. All CO_2 has its origin from bio-available carbon and is therefore not accounted for.

Due to the great variation in degree of flaring and energy recovery of landfill gas, it is important that the system under study reflect the actual technology applied in the geographical and temporal scope of the study.

3.4.4.3 Emissions to air (not CO_2 and CH_4)

Pollutants are emitted to air from landfills through direct evaporation from the landfill and through landfill gas combustion off-gases. Although methane and carbon dioxide are the bulk constituents, landfill gas typically contains in the order 120-150 trace components, constituting approximately 1% of volume (according to USEPA). The wide range of trace compounds that may be present are mainly determined by the types of waste deposited. It is therefore a benefit to apply product specific models as a basis for estimating emissions to air from landfills.

There is lack of data on emissions to air directly from landfills. Limited process information exists for evaporation of metals, volatile organic compounds (NMVOC), dioxines and other toxic pollutants. For most pollutant there is not enough data to establish product specific models.

For combustion off-gases better background data exist. But it is not identified studies that attempt to relate landfill gas to waste fractions and waste fraction contents (except for CO_2 and CH_4). Hence, it is not possible to establish product specific models for emission from landfill gas combustion either.

Information that has been gathered about toxic emissions emitted directly to air from landfill is given below.

- Dioxines directly from landfill: No data are found.
- Dioxines from landfill fires: 1 mg TEQ/ton mixed waste /30/. This figure must be combined with landfill fire frequency if taken into account. Often accidents are excluded from an LCA study.
- Hg directly from landfill: Measurements at Grønmo landfill in Oslo, Norway indicate that 1% of Hg in waste to the landfill is emitted to air. Studies carried out in Sweden indicate 0,01-0,2 g/ton waste (average Swedish waste to landfill) /13/. Note that both these studies are 10-15 years old.

Pollutants that are typically emitted from landfill gas combustion are given in Table 3-7 with examples of emission factors per kg gas combusted. These should not be regarded as default values, only examples, based on measurements from a single landfill with a simple flare technology.

Based on the amount of landfill gas generated per ton waste (e.g. as given in Table 3-6), and the share going to combustion/flaring, it is possible to calculate the figures in Table 3-7 into emissions per kg waste.

Table 3-7 Emission factors for landfill gas combustion/flaring /15/

Parameter	Unit per kg gas	Mixed waste
CO	g	39,7
NO_x	mg	162
SO_2	mg	931
PM	mg	882
PAH	mg	1,23
Hg	μg	5,88
Dioxins	pg	539

If other emissions than CO_2 and CH_4 from landfill are to be included in an LCA study, efforts should be made to collect more relevant data than those given above, where the main focus should be placed on the toxic compounds.

3.4.4.4 Emissions to water

It is particularly leakage of nutrients and metals that have negative impacts on the environment.

Parameters that typically are measured in leakage water are listed below /13/. According to the Norwegian State Pollution Control Agency (SFT) the first 11 parameters should be measured 4 times per year and the remaining ones 2 times per year. It should therefore be possible to at least derive process specific data for these parameters (if included in the study scope).

- Amount of leakage water
- Chemical oxygen demand (COD)
- Tot-N
- Ammonia
- Mercury (Hg)
- Lead (Pb)
- Cadmium (Cd)
- Iron (Fe)
- Chloride (Cl)
- Sodium
- Borium
- Biological oxygen demand (BOD)
- Arsene (As)
- Phenol
- Aromates
- Tot-P
- Potassium
- Sulphate
- Aluminium
- Polyaromatic hydrocarbons (PAH)
- Chlorinated organic compounds
- Zinc
- Chromium (Cr)
- Copper (Cu)
- Nickel (Ni)

Environmental authorities in the other Nordic countries might have other focus parameters and measurement criteria. Note that short-term measured data cannot be used to model long-term emissions. However if measured data are collected over many years from many sites and the waste composition is roughly known, one could use the measured values to see if there are large differences between model outputs and the measurements.

To be able to quantify leakage from landfill using a product approach one must:

- Select the period of time for which emissions shall be quantified (ref. section 3.3.5).
- Gather data on generated leakage rates for all components for the selected time interval.
- Gather data for the type and share of pollutants removed from water by leakage treatment.
- Calculate leakage to the environment (g pollutant/kg waste) based on waste composition, leakage rates and share of leakage collected in wastewater treatment systems and reduction factors for wastewater treatment.
- Finally the equation given in section 3.4.4.1 can be applied.

The selected time period is the same as for all pollutants (usually 100 years).

In ORWARE and LCA-LAND it is assumed that 80% of the leakage from landfills are collected and treated. In Norway in 1995 about 50% of all waste at landfill had leakage water treatment. This figure rises to 70-100% in the more populated areas around the Oslofjord (ref. http://www.ssb.no/).

If a leakage treatment system is in place, not necessarily all leakage water is collected and treated. As for landfills without leakage treatment, direct emissions must be taken into account.

Due to the great variation in collection system for leakage water, it is important that the system under study reflect the actual technology applied in the geographical and temporal scope of the study.

To estimate leakage to soil and water from a landfill in an LCA, it is recommended to use a product based model to estimate the maximum emissions (no leakage water collection). Then process specific figures, representative for the geographical and temporal boundaries of the study, should be applied for the share of leakage water that is collected and the efficiency of this treatment. The efficiency of the treatment varies with substance and treatment technology.

Future landfills might be located close to the sea. In this way drinking water resources (ground water) are protected and leakage treatment considered less important. This means that we will have higher emissions from landfills in the future but that the resulting impacts will be of lower concern. E.g. risk assessment will show lower risks but a life cycle impact assessment will show higher impacts. This illustrates the importance of not looking solely on LCA results when considering environmental performance. Parallel evaluations based on different methodologies are often necessary. It also illustrates the need to continuously develop the "waste-LCA methodology", in this case to integrate risk into the assessment.

3.4.4.5 Energy recovery

Energy recovery related to landfills is relevant when the landfill gas is collected and incinerated with energy recovery. The energy can be exploited both as heat, electricity or mechanical energy, as other types of fuels.

The main component of landfill gas is CH_4 (about 50%). As an approximation it is common to assume that the energy recovery is related to CH_4 alone.

The energy recovered based on a product specific approach can be estimated from:

- The amount of CH_4 produced by the waste flow in question. This is calculated based on the waste composition as all waste fractions have their own specific CH_4 generation potential based on bio-available carbon and defined period of time where CH_4 generation takes place.
- The degree of CH_4 collected and incinerated.
- The net heat value of the CH_4.
- Energy losses from combustion to delivered energy.

New landfills are usually required to have gas collection and flaring systems, while there are several old landfills that are lacking such systems.

- In Norway in 1995 approximately 25% of all landfills had gas collection system (ref. www.ssb.no).
- In Sweden in 1998 approximately 25 % of all landfills had gas collection (ref. http://www.environ.se/)

Most LCA studies assumes that in case of a landfill gas collection system, approximately 50% of the gas is collected and combusted. Assuming that the Norwegian and Swedish figures are valid for the present situation in Nordic countries (the values are probably higher as the large sites with much gas production probably has gas collection), about 13% of all landfill gas is collected with a variation within countries ranging from 0-50%.

The net heat value of CH_4 is typically 50-55 MJ/kg (depending on the conditions under which the gas is incinerated).

As a example on how estimate recovered energy we assume the following:

- Paper is sent to landfill.
- 240 kg CH_4 per ton paper is generated.
- 80% of the landfill gas (and CH_4) is collected and incinerated.
- It is assumed that only CH_4 gives energy in the incineration process.
- CH_4 has a heat value of 50 MJ/kg.
- The efficiency of the incineration and energy recovery process is 90% (10% energy loss).

The recovered energy, which in a life cycle perspective substitutes another energy carrier, is estimated to be:

$$240 \cdot 0{,}8 \cdot 50 \cdot 0{,}9 = 8640 \text{ MJ/ton paper.}$$

3.4.5 Aerobic composting

3.4.5.1 System description

The composting system is relevant to apply to organic waste, although some small non-organic fractions can be expected due to fractions passing through source- or central separation. Data and information normally applied in a LCA composting model are:

- Overview of composting technologies applied in the temporal and spatial boundaries of the study. The technology usually includes pre-treatment (disintegration and mixing of the organic waste), some type of composting process(es) and post treatment (e.g. stabilisation , sifting and maturing). Also different types of air and water emission purification technologies are applied.
- Distribution of main waste flow on waste fractions (applies for the whole study).. The main waste flow should be separated on waste fractions that act differently in the composting process and give significantly different type and magnitude of emissions and gas production.
- Waste fraction contents (applies for the whole study). This gives the substances available for pollutant and product generation.
- Overview of the range of pollutants decided to be studied.
- Product specific models for potential generation of pollutants and distribution of pollutants on environmental compartments, residues and compost.
- Share of air and water emissions collected and treated.
- Efficiency of emission treatment.
- Process specific energy consumption and emissions such as operation of vehicles and machinery.
- Compost and residue generation.
- Amount and type of fertiliser substituted by produced compost based on compost quality.
- Amount and type of auxiliaries and related cradle-to-gate data.

Composting is modelled in several studies. Although some exceptions exist, most of the models are process specific based on one the following assumption:

- That the waste fraction composition does not change and that the waste composition for which process data is collected, are representative for the waste flow composition under study. This is e.g. the case if only one waste fraction is treated (food waste)-
- If the waste composition changes, all organic waste behaves approximately similarly (contain the same pollutants and have the same potential to generate products, product characteristics and emissions).

A process flow chart for composting is given in Figure 3-12. Note that some process units are given at a coarse level. These can be further refined (e.g. the composting plant). Further, emissions, resource consumption and energy use flows are not shown for the process units. Finally, transportation is excluded.

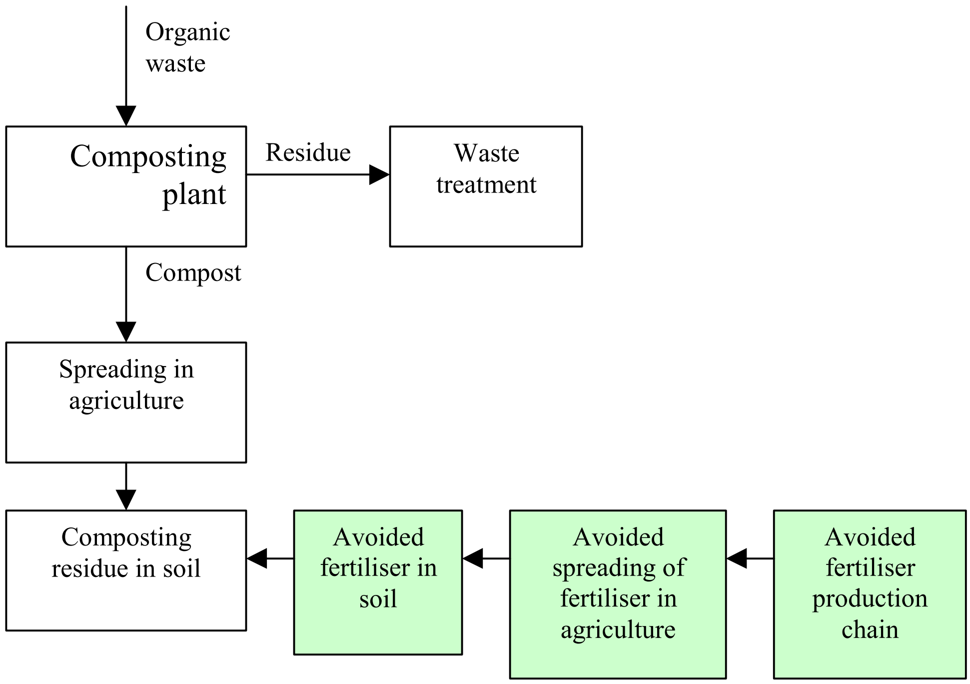

Figure 3-12 Process flow chart for the aerobic composting system

3.4.5.2 Emission of CO_2 and CH_4

As long as the waste that is degraded is organic waste and sufficient oxygen access is secured, generation of CH_4 is small. Nonetheless, specific data should be collected to document whether the CH_4 level is insignificant. Studies have shown that CH_4 can constitute over 10% of the air emissions from a closed and controlled composting plant.

It has also been documented that 1,5-2% of the biologically available carbon are emitted as CH_4 during composting of green waste /33/.

The same study says that about 80% of the carbon is emitted as CO_2. The waste contained 30% carbon (weight), which gives a CO_2 emission factor of 880 kg/ton green waste.

The emitted CO_2 is regarded to be greenhouse gas neutral.

For the remaining carbon other carbon-related emissions must be estimated during the selected period of time for emission generation. After that, it must be decided whether the carbon sink approach is used, if it will be emitted during a infinite period of time, or whether it is excluded from the further estimates and evaluations (as for carbon remaining in landfills).

3.4.5.3 Emissions to air (not CO_2 and CH_4)

The approach for establishing product related emissions to air are similar to what is valid for emissions from landfill. The potential for generation of a pollutant is based on the contents of the waste flow and distribution factors for emissions to air and water and what remains in the compost and residue.

However, this study has not identified such models except for nitrogen /9/. This means that it is only possible to establish product specific models for these substances and related emissions.

If no product specific models are identified, process models must be applied. Process emissions to air from composting are available from measurements performed at composting plants. It is important that the measurements from plant technologies that are representative for the system under study are applied as there can be great variations. It is emphasised that little data exist, mainly due to the fugitive characteristics of the emissions. Performed measurements are mainly related to substances generating odour, and they are measured as concentrations. However, some data are developed. Air emissions from the composting process are given in Table 3-8. Figures for other emissions than the ones listed in the table have not been identified by this study.

Table 3-8 Process specific emissions to air from composting

Parameter	Unit	Mixed organic waste		Green waste (garden waste)	
		Bio-reactor, gas treatment unknown /32/	Various techn. without gas treatment /34/	Wood box without gas treatment /33/	Various techn. without gas treatment /34/
NH_3	kg/ton waste	0,024	1,3	0,16	0,38
N_2O	kg/ton waste	0,08	-	0,17	-
TOC	kg/ton waste	0,9	-	-	-
CO	kg/ton waste	-	-	0,27	-
VOC	kg/ton waste	-	0,80	-	1,7

To what extent emission control is efficient depends on the composting and control technology. In-vessel composting methods can collect approximately 100% of the air emissions for purification. This is not possible for more open methods. Efficiency data can be collected from scientific studies and from control equipment suppliers. Biological filters are perhaps the most common emission purification technology. An American study estimates the efficiency of such filters to be 75% and 90% for captured NH_3 and VOC respectively /34/.

3.4.5.4 Emissions to water

Water leaches from the compost as a result of the water content in the waste. In addition it comes from watering the compost and/or from rainwater. The heat in closed composting vessels can evaporate water that is condensed in colder areas and released. The amount of water generated will depend on several factors, but a rough estimate is 250-300 kg water per ton waste /32/. The concentration and amount of pollutants washed out with the water depends on the concentration of substances in the waste and the amount of water emitted.

Water emissions to ground, groundwater and surface water can more or less be avoided by appropriately designed composting facilities.

In case of central composting, the run-off water is collected and purified in a local wastewater treatment unit, or sent to the municipal water collection and treatment system.

In case of home composting, it is assumed that insignificant amounts of pollutants emitted in run-off water as long as the waste is garden waste.

Based on the above emissions to water from composting is regarded as a minor problem.

No product specific models have been identified in this study related to emission to water. Table 3-9 shows process specific emissions factors without water treatment. These emissions will be significantly reduced if water treatment is applied. To derive more relevant figures, pollution reduction factors can be applied according to e.g. municipal sewage treatment plants.

Table 3-9 Emission factors for run-off water from composting /32/

Parameter	Water in waste (mg/litre)	Condensed water (mg/litre)	Rain water (mg/litre)
COD	20.000-100.000	500-2.000	500-2.500
BOD_5	10.000-45.000	100-1.000	100-1.200
TOC	5.000-18.000	<50-500	<50-500
P_{tot}	50-150	<1	<1-50
NH_4-N	50-800	<5-100	15-300
NH_3-N	<5-190	<1	<5-150
Cl^-	2.000-10.000	-	30-500
K^+	1.000-7.300	-	-
Zn	1-8	0,2-0,6	<1-2
Pb	0,01-0,02	<0,1	<0,1-0,2
Ni	0,07-2,6	<0,04	<0,05-1
Co	0,01-0,2	<0,05	<0,05-0,2
Cd	0,01-0,2	<0,02	<0,05-0,2
Hg	-	<0,0005	-

3.4.5.5 Compost

Data for the amount of compost generated for various presumptions are shown in Table 3-10.

Table 3-10 Amount of compost generated

Waste to composting	Technology	Compost	Additives	Ref.
Organic fraction of household waste	Reactor composting in closed room	590 kg pure compost per ton waste	340 kg wood per ton waste is added as stabilising substance.	/7/
Organic fraction of household waste	Reactor composting in boxes	450 kg pure compost per ton waste	170 kg wood per ton waste is added as stabilising substance.	/7/
Organic fraction of household waste	Open string technology	350 kg pure compost per ton waste	290 kg wood per ton waste is added as stabilising substance.	/7/
Sludge	Open string technology	600 kg pure compost per ton waste	400 kg wood per ton waste is added as stabilising substance.	/8/
Food waste	Open string technology	500 kg pure compost per ton waste	-	/9/

The amount of residues that are separated as non-compost is 50-300 kg per ton waste. This material flow should be subject to further waste treatment and should be treated as such in an LCA study /7/, /8/.

Compost quality, how to estimate the amount of substituted fertiliser, content of toxic compounds and leakage of these are treated in section 3.4.9.

3.4.6 Anaerobic digestion

3.4.6.1 System description

The anaerobic digestion system is relevant to apply to organic waste, although some small non-organic fractions can be expected due to fractions passing through source- or central separation. Data and information normally applied in an LCA anaerobic digestion model are:

- Overview of technologies applied in the temporal and spatial boundaries of the study. The technology usually includes pre-treatment (disintegration, mixing and pre-heating of the organic waste), some type of digestion process(es) and post treatment (e.g. stabilisation , sifting and maturing). Also different types of air and water emission purification technologies are applied.
- Distribution of main waste flow on waste fractions (applies for the whole study).. The main waste flow should be separated on waste fractions that act differently in the process and give significantly different type and magnitude of emissions and gas production.
- Waste fraction contents (applies for the whole study). This gives the substances available for pollutant and product generation.
- Overview of the range of pollutants decided to be studied.
- Process specific energy consumption and emissions such as operation of vehicles and machinery.
- Product specific models for potential generation of pollutants and distribution of pollutants on environmental compartments, residues and compost.
- Share of air and water emissions collected and treated.
- Efficiency of emission treatment.
- Compost, residue and energy generation (or any other generated product).
- Type and amount of fertiliser substituted by produced compost based on compost quality.
- Type and amount of energy substituted by recovered energy.
- Amount and type of auxiliaries and related cradle-to-gate data.

Relevant waste fractions for anaerobic digestion are organic waste such as food waste, paper and cardboard, garden waste, edible oil and fat and sludge.

Micro-organisms digest the waste and/or sludge in a controlled environment without any presence of air. This process produces biogas (mainly methane), which can be collected and exploited. The residues can be further treated to become compost.

An overview of a system model for anaerobic digestion is given in Figure 3-13

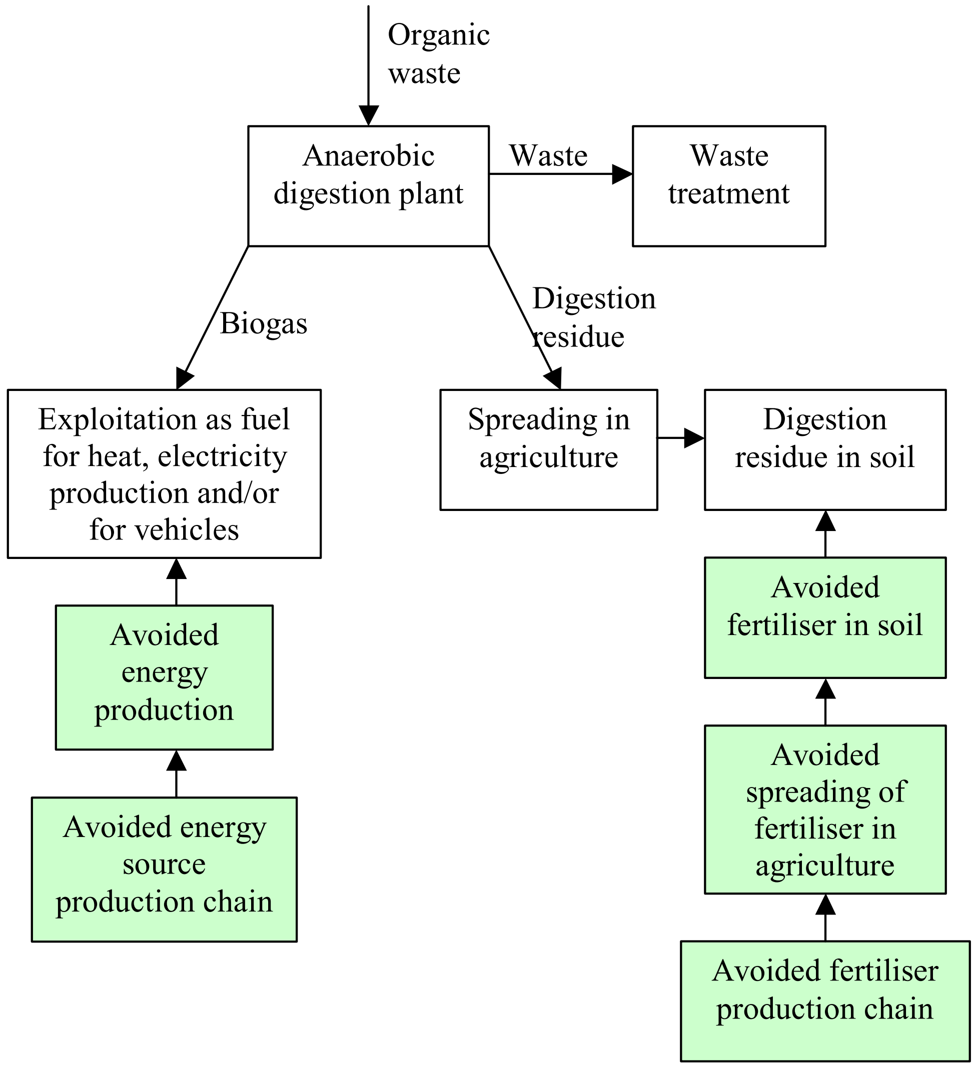

Figure 3-13 Process flow chart for the anaerobic digestion system

Related to the figure, note that some process units are given at a coarse level in the figure. These can be further refined. The following is not reflected:

- E.g. the bioreactor plant can be split into processes such as pre-treatment (homogenisation and thermal treatment), dewatering, digestion and composting. The thermal treatment process is for sterilisation purposes and to improve the digestion process.
- Transportation processes, emissions, resource consumption, auxiliary material use and energy use flows are not shown in the figure.
- A liquid phase can be separated from the waste flow in the dewatering process. This liquid can be used as a carbon source for biological wastewater treatment plants. This potential product, and the product that presumably is substituted (e.g. ethanol), is not included in the figure.

The bioreactors can be divided into wet and dry processes. The wet process mixes the waste with so much water that it can be pumped through the process (typically 15% dry matter). On the negative side this requires energy, water, larger reactor volumes and produces more effluent. The dry process has typically 30-35% dry matter.

Process temperature is used to regulate the digestion time, hence also the reactor capacity or volume. It is in this respect common to separate between mesophilic (30-40^{0}C) and thermophilic (50-60^{0}C) processes. Experience from a bioreactor plant in Finland using a wet process gives 10 days digestion time for a thermophilic process and 20 days for a mesophilic process /7/.

Table 3-11 gives some typical data collected from various bioreactor treatments of household waste. Note that this process includes separation of the organic fraction from the mixed residual household waste.

Table 3-11 Typical process data for anaerobic digestion

Environmental parameter	**Unit (per ton waste)**	**Plant 1 /7/**	**Plant 2 /7/**	**Plant 3 /9/**	**Plant 4# /29/**
Technology	-	wet mesophilic	dry mesophilic	wet mesophilic	wet thermophilic
Waste type	-	organic waste	organic waste	organic waste	Sludge
Electricity consumption	MWh	0,05	0,038	0,009	
Heat consumption	MWh		0,025	0,138	0,94
Waste water	m3	0,56			
Reject for other treatment	ton		0,22		
Produced compost	ton	0,67	0,69	0,86	0,58
Produced biogas	m3	130	145	99	370

All data are given per ton dry substance entering the plant Note that the process requires more energy than the others because it contains thermal hydrolysis. This gives a quicker process and better compost quality.

There are significant technical differences from plant to plant, and as seen from the table, this leads to significant differences in the performance of the plants. It is therefore important to use plant specific information for the waste type defined by the system description and within the geographical and spatial boundaries defined in the scope of the study.

Also, the compost product could be of varying quality. This will quantitatively not be taken into account unless the "in soil" environmental impacts are taken into account. As a minimum requirement such differences must be described qualitatively in comparative studies.

Based on experience from LCAs carried out for anaerobic digestion plants, the aspects influencing the environmental performance the most are the amount of recovered energy and the amount of composting residue substituting fertilisers. In addition comes the emissions caused by the plant and the consumption of energy and materials such as e.g. lime added to sludge to improve compost quality. In the latter case it is important also to include the production chain for the added materials.

3.4.6.2 Emission of CO_2 and CH_4

The main purpose of anaerobic digestion is to generate biogas that can be exploited as an energy source. The process takes place in a closed and controlled environment with no access to air where bacteria digest the organic waste. As the biogas is collected and combusted, it is transformed mainly into CO_2, but CH_4 will also be present in the off-gas. As the waste flow is approximately 100% organic, all CO_2 emissions are greenhouse gas neutral.

CH4 might also be emitted due to fugitive emissions during biogas storage.

Emissions of CH_4 and CO_2 are also related to the pre- and post-treatment processes, which requires fuel and/or electricity as heat and mechanical energy sources.

The heat consuming processes at the anaerobic digestion plant is often supplied with energy from the recovered biogas.

Both the type and magnitude of fuel and electricity consumption will be plant specific.

3.4.6.3 Emissions to air (not CO_2 and CH_4)

Other emissions to air are (as for CO_2 and CH_4) related to:

- Direct emissions from the degradation process.
- Combustion of fuel supplied to the plant.
- Production of electricity supplied to the plant (common for all processes within the same geographical boundaries).
- Combustion of the biogas (either at the site or it can be exported as a fuel, e.g. for buses).

Emission factors for fuel can be derived from onsite measurements or from generic data related to similar transport means or machinery.

Emission factors for biogas combustion are available from e.g. specific sites combusting biogas and from the companies responsible for biogas a bus fuel (e.g. bus companies in Uddevalla, Sweden or Fredrikstad, Norway).

3.4.6.4 Emissions to water

The water content of the waste/sludge is usually undergoing several processes and chemicals might be added. Hence, it is difficult to estimate the content of pollutants in water based on the waste/sludge composition. The excess water can be exploited as a carbon source and as source very little reject water is generated. Potential reject water is usually treated in wastewater treatment plants. The amount of reject water and the contents of pollutants are impossible to quantify without also specifying the applied technology. Therefore, no generic data is given here, and it is recommended to only apply data that is specific for the relevant technology.

3.4.6.5 Energy recovery

The energy provided by the CH_4 from the digestion system per functional unit (FU) will vary a great deal depending on technology. Hence, it is difficult to give generic figures for a geographical area unless data are collected from a representative share of the relevant plants in the area. Generally, the recovered energy is calculated by:

$$E = G \cdot H \cdot R \cdot M$$

E is the recovered energy (MWh/FU)

G is produced biogas (m^3/ton waste)

H is heating value of the biogas (MWh/m^3)

R is share of produced energy that is exploited, subtracted what is used by the process itself ((MJ $_{exploited}$ - MJ $_{internal\ use}$)/MJ $_{produced}$)

M is the amount of waste (ton waste/FU)

How the recovered energy is distributed on heat and electricity production and what type of energy that is substituted is of course specific for the selected plants that are representative for the temporal and spatial boundaries of the study.

An anaerobic digestion plant operating with high temperatures, such as e.g. plants with thermal hydrolysis as pre-treatment, may require so much energy that all the biogas is used as energy for internal heat production. If we look at the biogas production in Table 3-11 and assume a biogas heat value of 6 kWh/m^3, the energy efficiency of the plants (1-1 relationship between consumed energy and produced energy) is in the range 0,58-0,94. Note that low energy efficiency is not necessarily negative. It can indicate that there is more focus on compost quality and by-products that require high treatment temperatures.

3.4.6.6 Compost and other products

As for all the other parameters related to anaerobic digestion, also the products vary a great deal. This is both the amount of product and the type of products. Some plants are focused on biogas generation and others on soil improvement products and other by-products (e.g. carbon source). Again data should be quantified based on a process specific data for relevant technology.

Compost quality, how to estimate the amount of substituted fertiliser, content of toxic compounds and leakage of these are treated in section 3.4.9.

3.4.7 Biocells

3.4.7.1 System description

In principle biocells are anaerobic digestion (bioreactors) carried out in batches, usually under less controlled ambient conditions. It can also be seen as an improved landfill, especially with respect to biogas collection and treatment of leakage.

Compared to a landfill biocells have:

- More efficient biological turnover
- Better collection of biogas
- More efficient land use
- Low production of leakage water
- Better quality of leakage water

Note that even though bioreactors usually have better performance data than biocells, it usually also calls for larger investments. Also, biocells can be more technically feasible than bioreactors when the input is residual waste not only containing organic waste.

The biocell system is relevant to apply to mixed municipal waste, preferably with a high organic content. Data and information normally applied in an LCA biocell model are:

- Overview of technologies applied in the temporal and spatial boundaries of the study.
- Distribution of main waste flow on waste fractions (applies for the whole study).. The main waste flow should be separated on waste fractions that act differently in the process and give significantly different type and magnitude of emissions and gas production.
- Waste fraction contents (applies for the whole study). This gives the substances available for pollutant and product generation.
- Overview of the range of pollutants decided to be studied.
- Process specific energy consumption and emissions such as operation of vehicles and machinery.
- Product specific models for potential generation of pollutants and distribution of pollutants on environmental compartments, residues and compost.
- Share of air and water emissions collected and treated.
- Efficiency of emission treatment.
- Residue and energy generation (or any other generated product).
- Type and amount of energy substituted by recovered energy.
- Amount and type of auxiliaries and related cradle-to-gate data.

Little information has been derived that gives inventory data for biocells. However, some information is given below based on the only identified Nordic study on this waste treatment alternative /37/. This study includes collection of experience data and information and a LCI model (applied in ORWARE).

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Note that even though bioreactors usually have better performance data than biocells, it usually also calls for larger investments. Also, biocells can be more technically feasible than bioreactors when the input is residual waste not only containing organic waste.

In general the system model for a biocell is the same as for landfill. The differences are related to the input and output data. The main purpose of the biocell is to recover as much biogas as possible. To do so it is more common to add auxiliary materials than it is for landfill. Such materials, that should be included in an LCA, are typically:

- Water and air injection.
- Phosphorous, to optimise degradation (1-2 kg/ton household waste).
- Pre-combusted waste (ash), to establish anaerobic conditions at the bottom of the cell. A decision must be taken on whether the ash shall be regarded as a product, a resource or waste. If it is regarded as a product the production of ash (incineration process) should be allocated to the system. If it is regarded as a resource, no production impacts are allocated. If it is perceived as waste, the system should be credited the reduced amount of waste. To make such a decision, the analyst could evaluate the monetary flow related to the ash. If the biocell company buys the ash it should be regarded as a product. If the company get paid for receiving the ash, it should be regarded as waste. If the ash is free it should be regarded as a resource (not limited).

An overview of a system model for biocell is given in Figure 3-14. It can be argued that the fraction remaining after opening the biocell can be used as soil improvement products. In that case the system should include this post-treatment of the product and the production chain of the substituted product. The main reason for not including it here is that the acceptance for such a product is unlikely due to the content of pollutants and lack of data. Inclusion of the relevant processes would be more or less based on speculations.

Note that some process units are given at a coarse level. These can be further refined (e.g. the avoided energy and related production chains). Further, emissions, resource consumption and energy use flows are not shown for the process units. Finally, transportation is excluded.

All the flows in the system are relative to the waste flow entering the system. This municipal waste flow is the whole or a share of the reference flow resulting from the functional unit.

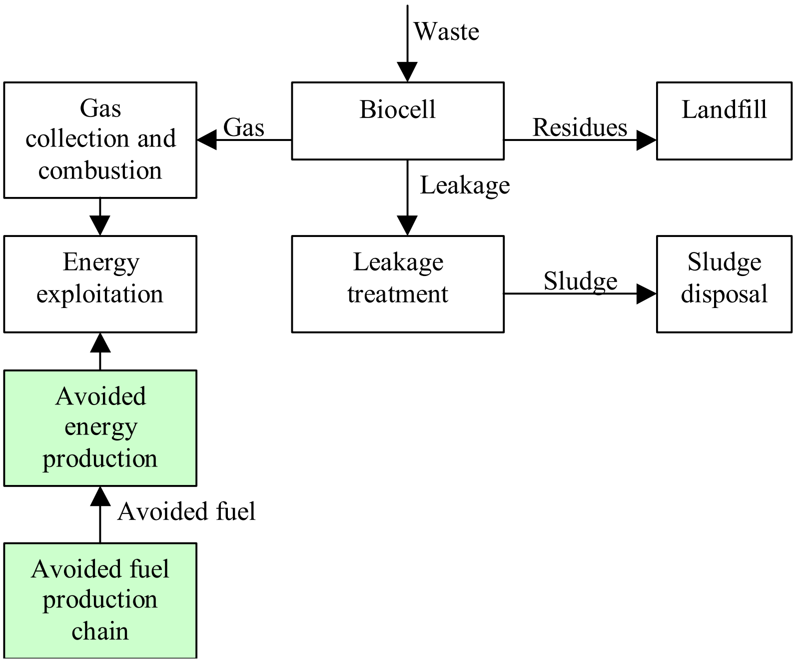

Figure 3-14 Process flow chart for the biocell system

As for all other treatment alternatives the data applied to the model can be based on a product or process approach or a combination of these. The process approach uses plant specific data, but does not have the ability to reflect changes in the waste composition in the calculated results. The product based approach gives this opportunity, but should be calibrated against process data representative for the spatial and temporal scope of the study to minimise errors.

As for landfill, which takes place over a long period of time, the emissions can be divided into surveyable time emissions and remaining time emissions (ref. section 3.3.5). However, the surveyable time could be set similar to the biocell lifetime (from closing to opening). The remaining time would then be relevant for the biocell residues if sent to landfill.

3.4.7.2 Emission of CO_2 and CH_4

Biocells are constructed in such a way that methane tends to oxidise when passing through the protection layer. It is indicated that the methane emissions are less than 10% of the formation (i.e. less than 10 m^3/ton mixed household waste, based on the figures given in section 3.4.7.6).

Experience data from operation of 12 Swedish biocells for 5 years gives the following results, which must be multiplied with the biocell lifetime (10-15 years):

- CH_4: 3-10,3 m^3/yr/ton (average 5,7 m^3/yr/ton). Less than 10% of this is emitted.
- CO_2: 2,9-7,6 m^3/yr/ton (average 4,9 m^3/yr/ton). All is emitted. The major part of the CO_2 is non-fossil based. The fossil/non-fossil CO_2 must be estimated based on the waste fractions and their degradation within the biocell lifetime.

In general, the different categories of CH_4 and CO_2 emissions as described in Figure 3-11 for landfill, also apply for biocells.

The methane gas formation will of course vary with the ambient conditions, technical conditions and the waste composition. The ORWARE biocell model enables differentiation between household waste mix, sludge and ash and slag from incineration.

Experience shows that the biogas production does not increase proportional to the content of fast biodegradable waste such as food and garden waste. In order to obtain high rates of biogas production under the whole treatment period, the amount of fast biodegradable waste must be limited.

3.4.7.3 Emissions to air (not CO_2 and CH_4)

No data or information has been identified. However, there are reasons to believe that the potential substances are the same as for landfill. Although, in smaller amounts due to the closed environment and less mass transport of gases and micro pollutants.

3.4.7.4 Emissions to water

Typically 0,3 litre leakage water is produced per kg household waste (compared to 2 litre/kg for landfill and 0,1 liter/kg for bioreactors).

Both less leakage water and lower concentration of pollutants in the water result in less leakage impacts compared to a traditional landfill. The main reason for less water generation is due to less intrusion of water into the biocell. The level of pollutants is lower because the bottom layer consists of pre-combusted material and anaerobic conditions are established here. The layer then acts as an anaerobic filter for the leakage water, degrading dissolved carbon substances. Experience data shows 30-85% reduction of COD concentration versus a traditional landfill.

3.4.7.5 Biocell residues

As previously mentioned it can be argued that the fraction remaining after opening the biocell can be used as soil a improvement product. However, the acceptance for such a product is unlikely due to the content of pollutants and lack of data. In a future perspective, with improved source separation and subsequent improved waste quality, this usage could be possible.

A more probable usage is as top/side covers at landfills or new biocells. However, this might require further stabilisation to avoid potential odour problems.

The residues might also end up as landfill waste, with related environmental impacts.

3.4.7.6 Energy recovery

Some characteristics for biocell energy recovery are:

- The period of time that methane is produced is 10-15 years.
- For mixed household waste ~200 m^3/ton biogas is generated (~250 m^3/ton including air intrusion). About 100 m^3/ton is methane gas.
- Typically 60-70% of the methane gas is collected in a biocell.
- An LCA must consider how the gas is exploited and what type and magnitude of energy sources that is substituted by the energy recovery.

3.4.8 Substituted energy

Energy sources in other systems are substituted when recovered energy from the system under study are exploited and replaces other energy sources. This is relevant for energy recovered from incineration plants, energy recovered when incinerating collected landfill gas, energy recovered through use of collected biogas from anaerobic digestion, and energy recovered when using biogas from biocells.

3.4.8.1 Substituted energy sources

The amount of energy substituted equals the amount of energy from the waste treatment that is exploited.

The distribution of substituted energy sources is illustrated in Figure 3-15.

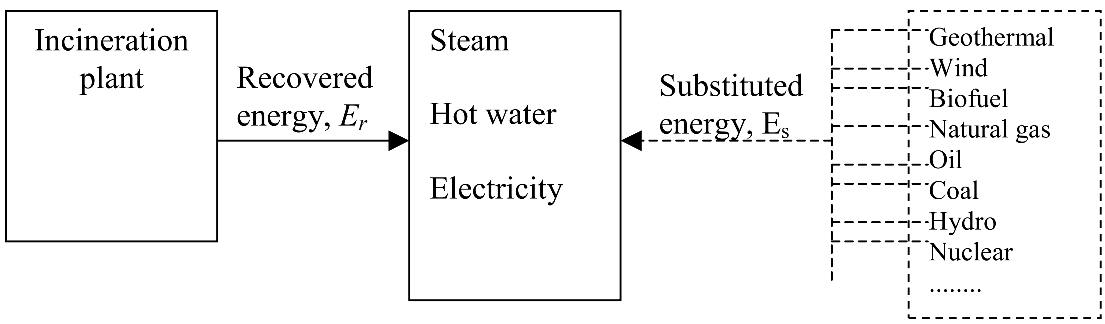

Figure 3-15 Illustration of distribution of substituted energy

First of all it must be decided whether to use a retrospective or a prospective approach (should be defined in the scope). In a retrospective study the historic and present energy supply situation is used as a basis. The following steps should be followed when identifying substituted energy sources in a historic/present perspective:

- Quantify how exploited energy is distributed on steam to industrial processes, hot water to district heating and electricity production.
- Check with the companies that use/produce steam what would be the alternative energy source(s) if the steam is not provided. If several energy sources are relevant the distribution of these in a long term must be quantified.
- Check with the electricity producers, what will be the energy source for electricity production, if additional electricity is needed.

In a prospective study future energy supply scenarios are defined. In general, the marginal energy source should be applied and the retrospective and prospective approaches can lead to different marginal energy carriers.

In a short-term perspective, at an existing plant, the marginal energy carrier is usually the lowest priced energy source that is technically and legally feasible. In most cases this involves some kind of fossil fuel (oil, coal, natural gas). However, also other energy forms might be relevant, especially if energy is recovered in the form of electricity.

In a long-term perspective the marginal energy source will be the energy source taken out if energy consumption is reduced, or the energy source installed if the energy consumption is increased. This is perhaps more a result of political goals, rather than short-term economic considerations.

Whether to use a short- or a long-term perspective will depend on the goal and scope of the study. E.g. to analyse future scenarios (prospective approach) a long-term perspective should be preferred.

In many studies where electricity is replaced, the present form of producing electricity is selected as the energy form to be replaced. This is not the marginal energy form (although it can be). The energy form can be the energy source mix used for electricity production in the nation in question. The energy mix used for electricity production in the Nordic countries in 2000 according to Nordel (http://www.nordel.org/) are given in Table 3-12.

Table 3-12 Total electricity generation by energy source, and net imports and exports 2000, TWh (ref. NORDEL statistics at http://www.nordel.org)

	Denmark	Finland	Iceland	Norway	Sweden
Net imports	0.7	11.9			4.7
Geothermal power			1.3		
Wind power	4.2	0.1		0.0	0.4
Other *	3.8			0.4	
Biofuel	1.7	13.2			3.6
Natural gas	8.0	8.0		0.2	0.5
Oil	0.1	1.5			2.8
Coal	16.4	8.4			1.9
Nuclear power		21.6			54.8
Hydropower	0.0	14.4	6.4	142.1	77.8
Net exports (negative value)				-19.0	

* In Denmark orimulsion (a fossil fuel produced from natural bitumen mixed with water) and refinery gas.

As the liberalised market for electricity in Europe comes into effect through improved distribution nets, it may be difficult to define the energy mix of a region, as energy will flow across national borders. It is then relevant to apply the energy mix used for electricity production on an average European level.

In case of introducing recycling of materials in stead in incineration it is important to realise that very often the fuel substituting a specific waste fraction can be another waste fraction. This is because the incinerator capacity is often limited which means that if one waste fraction is recycled instead, another waste fraction (which perhaps is currently landfilled) can be incinerated. The capacity restriction will however vary from region to region. In same areas there might even be over-capacity. It is therefore important to relate such a presumption to the geographical boundaries of the study.

3.4.8.2 Pollution related to substituted energy sources

The pollution and related impacts from energy sources that are substituted by recovered energy should preferably be subtracted from the life cycle inventory of the municipal waste. This is equivalent to a system expansion which is ISO's first recommendation before allocation is investigated. This means that emission data (and other environmental impact data) for all energy sources (at least the ones listed in Table 3-12) can potentially come into use.

There are several problems related to derive pollution data, both due to great variations in energy plants using the same energy source, and due to the variety and forms of environmental impacts. E.g. combustible energy sources such as fossil fuels have environmental impacts that are traditionally treated in an LCA (emissions to air and water). Hydro and nuclear power give environmental impacts such as land occupation, esthetical disturbance, hazardous waste and human/environmental risk, which are aspects that can be quantified with established methodological models.

Due to the great number of energy producing installations and the variety in technology, it is not reasonable to list plant specific pollution data for substituted energy sources within the frames of this study. However, if the available LCA study scope and resources allow for it, plant specific data should be collected and applied.

Ideally, the whole life cycle of the substituted energy sources should be taken into account. This means that e.g. for oil, coal, natural gas nuclear fuels, extraction, production and distribution should be taken into account in addition to the operational pollution. Such data are available for most energy sources in life cycle inventory databases.

Life cycle assessments have been performed for most fuels and energy sources. Hence, it should not be difficult to obtain generic data. An example of a comprehensive data source is the EU project ExternE. The results of work performed in several European countries (including Denmark, Sweden, Finland and Norway) are presented in reports available online at http://externe.jrc.es/.

3.4.9 Substituted fertilisers and fertiliser impacts

3.4.9.1 What is substituted?

Both sludge treatment, aerobic composting and anaerobic digestion of organic waste are processes that can produce products that can be used as fertilisers or additions in soil products due to high contents of nutrients.

The products can replace artificial fertilisers, although there are great uncertainties related to what extent the artificial fertilisers are replaced. I.e. how many ton artificial fertiliser is replaced by one ton compost?

The nutrient content of the compost is defined by the content of nitrogen (N) and phosphorous (P). Hence, the compost can replace both artificial N-fertiliser and P-fertiliser. Table 3-13 gives content of nutrients in sludge and compost. The sludge related data are collected from 18 Norwegian sludge treatment plants, while the figures for compost (from municipal waste) are derived from a range of European studies /28/.

Table 3-13 Content of nutrients in sludge and compost /28/

Parameter	Sludge (kg/ton dry matter)	Compost (kg/ton dry matter)
NH_4-N	0,4 - 7,8	0,18 - 0,78
Tot-N	3 - 29	7,9 - 23,3
Tot-P	4 - 22	1,9 - 5,4
Ca	3 - 190	27 - 35,3
K	0,7 - 2,7	5,3 - 14,8

It is more difficult to obtain representative figures for compost than sludge due to a larger variation in composition. The level of available N is lower and the compost will to a less extent replace fertilisers compared to sludge. The homogeneity will be influenced by the organic waste composition, which again is influenced by the composition at the source, the quality of organic waste separation and the treatment technology.

Note that the approach presented here takes into account the nutrition potential of compost as a basis for assessing what products that are substituted and estimation of the amount of avoided artificial fertiliser.

There might be other benefits of artificial fertilisers that are lost due to the substitution, e.g. lime in fertiliser that affect soil acidity. Or vice versa if compost has other benefits that the replaced product has not, e.g. increasing the soil's organic carbon content, added structure to soil and changed water balance. Such additional effects are difficult to quantitatively take into account in an LCA. However, it is important to address these issues when defining the functions of the competing products.

3.4.9.2 How much is substituted?

The amount of artificial fertiliser substituted depends on whether the soil limits the amount of compost with respect to N or P. To decide this, figures for recommended annual doses for N and P in soil can be used (kg/ha-year). The ratio between recommended N-dose and recommended P-dose can be used as a reference value. If the exploitable N/P ratio in compost is larger than the reference value the compost is N-limited and vice versa. As a simplification the total N and P content can be used in stead of the exploitable, although it is recommended to use the exploitable content.

$$\frac{N_{comp}}{P_{comp}} \approx \frac{N_{comp,\exp l}}{P_{comp,\exp l}} > \frac{N_{\lim it}}{P_{\lim it}} \Rightarrow N - \lim ited$$

$$\frac{N_{comp}}{P_{comp}} \approx \frac{N_{comp,\exp l}}{P_{comp,\exp l}} < \frac{N_{\lim it}}{P_{\lim it}} \Rightarrow P - \lim ited$$

Where: N_{comp} = The N-content (kg/ton) in the compost

P_{comp}= The P-content (kg/ton) in the compost

$N_{comp,expl}$ = The exploitable N-content (kg/ton) in the compost

$P_{comp,expl}$ = The exploitable P-content (kg/ton) in the compost

N_{limit}= Recommended annual N-dose in soil (kg/ha-year)

P_{limit}= Recommended annual P-dose in soil (kg/ha-year)

A Swedish study presents maximum N-dose = 90 kg/ha-year and P-dose = 15 kg/ha-year, which gives N_{limit}/P_{limit} = 90/15 = 6. The residue of anaerobic digestion is analysed with an N-content of 7,6 kg/ton and a P-content of 1,1 kg/ton. The N_{comp}/P_{comp} ratio is then 6,9. This is larger than 6 and the compost is therefore N-limited /9/.

The limit for exploitable N-content in the compost is calculated by:

$$\frac{N_{comp,limited}}{P_{comp,expl}} = \frac{N_{limit}}{P_{limit}} \Rightarrow N_{comp,limited} = \frac{N_{limit}}{P_{limit}} \cdot P_{comp,expl} = 6 \cdot 1{,}1 = 6{,}6 \quad kg/ton$$

Alternatives to the N- or P-limited approach are to use both the N- and P-content directly or it is possible only to focus upon N or P as a basis for substitution.

N-limited substitution

Most N in compost is bound up organically, while only a small share is mineral based nitrogen (NH_4^+, NO_2^- and NO_3^-). Studies on sludge carried out by Planteforsk in Norway shows that about 80% of mineralised N and 10% of organic N can be exploited the first year, and 10% of the remaining N each year after that. These figures are directly comparable with the nitrogen content of artificial fertilisers /7/.

Based on Table 3-13 and only taking into account the first year of nutrition, the available N is 0,6 – 9,1 kg/ton for sludge and 0,9 – 2,9 kg/ton for compost (dry matter). NO_2^- and NO_3^- are assumed negligible. Note that taking only the first year into account is an underestimation. Ideally integration of the nutrition uptake function should be performed over the period of time where the nutrition takes place.

To estimate the amount of N-fertiliser that is replaced, it is necessary to know the N-content of the fertiliser. Figures for N-content of N-fertilisers produced by the worlds largest mineral fertiliser producer (Hydro Agri) are given in Table 3-14 (ref. http://www.agri.hydro.com/).

Based on the fact that available N in sludge and compost is 0,5 – 4 kg/ton and the second column in Table 3-14, the amount of sludge or compost to replace 1 ton artificial N-fertiliser is calculated. This is given in the third and fourth column in Table 3-14.

Table 3-14 Nitrogen content of N-fertilisers supplied by Hydro Agri and the magnitude substituted per ton compost (dry weight)

Nitrogen fertilisers	Tot-N (kg/ton)	Substitution sludge (kg/ton dry matter)	Substitution compost (kg/ton dry matter)
Calcium nitrate	155	4 - 59	6 - 19
Calcium ammonium nitrate	250 - 280	2 - 36	3 - 12
Ammonium nitrate	340	2 - 27	3 - 9
Urea	460	1 - 20	2 - 6
Nitrogen solutions (mainly UAN)	280 - 320	2 - 32	3 - 10
Ammonium sulphate	210	3 - 43	4 - 14

P-limited substitution

To be able to estimate the amount of P-fertiliser that is replaced, it is necessary to know the P-content of the fertiliser. Figures for P-content of P-fertilisers produced by the worlds largest mineral fertiliser producer (Hydro Agri) are given in Table 3-15 (ref. http://www.agri.hydro.com/).

Assuming that all phosphorous in sludge and compost are available for uptake, the content given in Table 3-13, and the second column in Table 3-15, the amount of sludge or compost to replace 1 ton artificial P-fertiliser can be calculated. This is given in the column 3 and 4 in Table 3-15.

Table 3-15 Phosphorous content of P-fertilisers supplied by Hydro Agri and the magnitude substituted per ton compost (dry weight)

P-fertilisers	P (kg/ton)	Substitution sludge (kg/ton dry matter)	Substitution compost (kg/ton dry matter)
Hydro-P™ 8	80	50 - 275	24 - 68
Raw phosphate	160	25 -138	12 - 34
Hydro-PK™ 5-17	47	85 - 468	40 - 115

Note that these fertilisers also contain kalium (latter product) and sulphur that can give additional positive effects.

A study performed by The Norwegian Crop Research Institute (Planteforsk) reveals that 35% of the P in compost from biowaste is bioavailable, and 8% of the P in compost from sludge /36/. This indicates that the assumption made above, saying that all phosphorous is available for uptake, is not valid. Applying this would reduce the values in the table above significantly.

3.4.9.3 Processes related to compost exploitation

There are mainly two processes related to compost exploitation that are associated with environmental impact. These are, in addition to transport activities, the spreading of the compost and the leakage of pollutants from the compost and into the recipients.

The latter processes is often omitted in LCA applied in the waste management sector, but should ideally be included as there could be significant differences between artificial fertilisers and compost with respect to the contents of pollutants.

Spreading of compost

The spreading process itself will probably not differ between fertiliser and compost. However, the amount of material that is spread is greater for the compost, and therefore it requires more energy for the transport and spreading. If the spreading of fertiliser and compost can be regarded to be about the same, this process will not have to be taken into account.

The main environmental impact associated with this process is the consumption of fuel and related combustion exhaust gases. Data for emitted exhaust gases per unit fuel are usually easy to obtain (e.g. from spreading vehicle manufacturer) and are therefore not treated any further here.

The fuel is usually diesel. The amount of diesel consumed per ton sludge or compost (F) can be given as F = C*L/A, where:

- *A* (ton/ha) is the amount of compost per area. This parameter can be given specifically for the study or national regulations can be used as a maximum area. Remember to take into account the wet fraction if *A* is given based on dry fraction.
- *L* (m/ha) is the driving distance of the tractor/spreader per ha of spreading. This parameter can be estimated based on the working width of the spreader.
- *C* (MJ/m) is the fuel consumption per driving distance of tractor/spreader. This parameter can be derived e.g. from vehicle manufacturer.

Sludge and compost in soil

Sludge and compost always contain some pollutants that can be transferred to the soil. The content of pollutant in sludge and compost will of course depend on the pollution level in the origin flows (wastewater and organic waste) and the technology applied to treat these flows. It can be assumed that all the pollutants end up in the soil.

However, note that artificial fertilisers will also contain pollutants that must be considered if the compost pollutants are considered.

Table 3-16 and Table 3-17 give data for pollution level in sludge and compost /28/. The data are from Norway and collected during 1997-2000. The data are mainly for exemplification as these data tend to be very case specific.

The figures for organic pollutants in sludge are gathered from 7 Norwegian municipal wastewater treatment plants. One sample is mixed over a month of sludge production. Five such samples are taken from each plant.

The figures for organic pollutants in compost are gathered from 9 samples of compost from Norwegian household waste

The figures for heavy metals in sludge are gathered from mixed monthly samples from all Norwegian wastewater treatment plants with dehydration of sludge.

The figures for heavy metals in compost are gathered from mixed samples from 9 Norwegian composting plants (two reactors and seven open air plants). Source separated organic waste from households was composted.

Table 3-16 Contents of pollutants in sludge and composted sludge /28/

Parameter	Unit	Sludge			Composted sludge
		Average	Min.	Max.	
Dioxins/furanes	ng/kg TS	10,6	3,1	69,3	-
PCB	mg/kg TS	0,05	0,02	0,10	
PAH	mg/kg TS	6,0	0,7	30,3	0,084
Creosols	mg/kg TS	35,5	n.d.	470	
Nonylphenol/-ethoxilates	mg/kg TS	171	22	650	
Phtalates (DBP and DEHP)	mg/kg TS	81	n.d.	192	7,4
LAS	mg/kg TS	85	n.d.	424	116
Cd	mg/kg TS	1			
Pb	mg/kg TS	21			
Hg	mg/kg TS	0,9			
Ni	mg/kg TS	15			
Zn	mg/kg TS	317			
Cu	mg/kg TS	244			
Cr	mg/kg TS	25			

Table 3-17 Contents of pollutants in compost from organic fraction of municipal waste /28/

Parameter	Unit	Average	Min.	Max.
Dioxins/furanes	ng/kg TS	4,4	0,5	11.9
PCB	mg/kg TS	0,024	0,003	0,078
PAH	mg/kg TS	1,36	n.d.	3,77
Creosols	mg/kg TS	2	n.d.	22
Nonylphenol/-ethoxilates	mg/kg TS	n.d.	n.d.	-
Phtalates (DBP and DEHP)	mg/kg TS	8,0	n.d.	29,2
LAS	mg/kg TS	85	14	185
Cd	mg/kg TS	0.36	<0,3	0,59
Pb	mg/kg TS	20	<5	37
Hg	mg/kg TS	0,11	<0,05	0,38
Ni	mg/kg TS	10	<2	17
Zn	mg/kg TS	197	46	320
Cu	mg/kg TS	52	24	78
Cr	mg/kg TS	14	<5	20

3.4.9.4 Processes related to substituted fertiliser

Compost can substitute fertiliser. The allocation principle presented in section 3.5.2 required that the system producing the compost will get subtracted the environmental burdens associated with the substituted fertiliser (or the alternative system will have it added).

Environmental burdens associated to artificial fertilisers are related to the whole life cycle of the product. This includes:

- The cradle to gate production chain
- Distribution
- Spreading
- Fertiliser in soil

Note that it is important to set the fertilisers product system cut-off at the right stage. If the compost system does not include the environmental impacts of spreading or compost in soil, neither should spreading and fertiliser in soil be included (and vice versa).

To obtain LCI data for the life cycle of fertilisers is a study in itself. Usually, there are not enough resources available in a waste LCA to develop fertiliser LCI data specifically for the geographical and temporal boundaries of the study. Hence, ready-made data should be applied. Such data are available in various LCA databases, but note the time range and geographical boundaries they represent. There are large differences between types of fertilisers, production technology (old Eastern Europe technology versus modern Western Europe) and national power supply systems (e.g. hydropower versus coal power). One should therefore make sure that applied fertiliser data that comply the scope of the study.

It will be a too comprehensive task for this guideline project to collect and present LCI data for different artificial fertiliser products. For ready made LCI data, or the basis for developing such, it is referred to LCA databases (or the studies providing the basis for the database data), large fertiliser producers (like Hydro Agri) and the European branch organisation European Fertilizer Manufacturers Association (EFMA). The latter has developed several Best Available Technique (BAT) documents for various fertiliser products and for production and application (ref. http://www.efma.org/index.asp). Many LCI data can be established based on these documents.

3.4.10 Other environmental aspects than emissions and material consumption

Waste treatment also has some other obvious environmental disadvantages than emissions, energy and material consumption and waste generation, the most important ones being:

- Land occupation
- Odour
- Noise
- Accidents resulting in emissions (e.g. landfill fires).
- Injuries and fatalities from accidents.
- Esthetical impacts

One problem with these environmental burdens is that they are usually not given in a unit that enables aggregation of contribution from various processes. If they are given in units that can be aggregated, there is often lack of existing representative data, and much project resources must be invested to derive figures.

This study has not identified any LCAs on waste where the environmental burdens above are included. However, methodologies exist in the general LCA literature that enables them to be included (except for esthetical impacts).

- Land occupation is perhaps the environmental burden that is most commonly applied in LCAs of the burdens above. Guideline on how to measure land area occupation is described in section 3.6.1.

- Odour is usually a result of exposure from a range of gases. Odour can then be included as an impact category by including the emission of odour generating gases in the inventory analyses. The problem is that inventory data to a little extent exist.
- Noise is measured in dB(A), but it is not possible to aggregate noise measured in dB (A) from different locations. However, a certain noise level can be transferred to a potential influence area, which can be aggregated.
- Accidents with both environmental and human health consequences are difficult to predict due to great variations in accident frequency. Also, it is not a part of normal planned operation, which is often a presumption in an LCA. However, the Swedish ORWARE model includes a landfill fire model.
- Esthetical impacts are usually not described in quantitative terms and are very site specific. This environmental burden is usually not included in LCAs, but is commonly treated in environmental impact assessments (EIA).

3.5 Allocation

According to the ISO14040 series selection of allocation principle should be performed according to the following hierarchy:

1. System expansion.
2. Allocation applying a relevant technical criterion.
3. Allocation applying an economic criterion.

Allocation is partitioning the input or output flow of a unit process to the product system under study. An allocation principle is as such a principle that describes how the flows shall be partitioned.

With respect to LCA applied for municipal waste, allocation is particularly relevant with respect to:

- How to allocate environmental burdens from waste treatment to specific input waste fractions (multi input problem). E.g. if the flow of interest is municipal waste and this waste is incinerated or sent to landfill together with other types of waste (e.g. mixed industrial waste), how do we allocate the environmental burden from the incineration or landfill to the municipal waste under study? (See Figure 3-16 for illustration.)
- How to allocate the environmental benefit generated by a waste system that produces product that are applied in other systems (open loop recycling). E.g. soil improvement products or heat produced from composting or incineration, how does the waste system under study benefit from the fact that the produced soil improvement can replace fertilisers and heat can replace other energy sources?

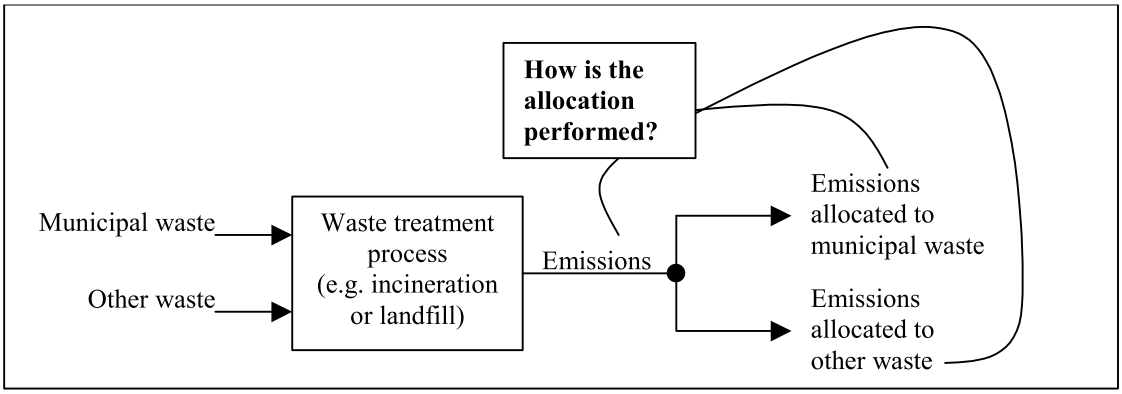

Figure 3-16 Multi input waste allocation problem

3.5.1 Multi input recycling

Multi input recycling can be relevant is the case illustrated in Figure 3-16 or in similar problems related to transportation.

If a product based approach is used in the inventory phase there will be no allocation problem. This because the emissions from the waste treatment is a direct function of the inputs. However, if a process approach is used to establish data, allocation must be applied.

Related to transportation, where the waste flow under study is transported together with other waste, volume should form the basis for allocation. This is because volume is usually the limiting factor on the capacity of transportation means.

As waste flows usually have approximately the same economic value, economic allocation will usually mean the same as allocation based on mass in situations as those illustrated in Figure 3-16. As an example, assume that the waste treatment process in Figure 3-16 gives 1000 kg NOx/year. The mix of input waste per year is 40% municipal waste and 60% other waste on a mass basis. If the system under study only included the municipal waste flow, a mass based allocation approach would allocate 400 kg NOx/year to the municipal waste flow. Here may exist a technical criterion which differs if the emission is product specific (i.e. determined by the composition of the product or waste stream and nor of the process) as NOx is partially in the above allocation example.

Allocation of produced energy to input flows should be based on the energy content of the input flows. Similarly, metal emissions should be allocated to input flows based on their content of these metals, and CO2-emissions according to their C-content.

3.5.2 Open loop recycling

The open loop recycling problem is usually solved through system expansion in most LCAs applied for waste. The system expansion approach is illustrated in Figure 3-17.

For illustration purposes, landfill treatment, that result in no form of recycled/recovered products, is compared to composting, that result in a recycled material (soil improvement product).

Note that the composting system also represents other waste treatment alternatives that give recycled/recovered products, such as incineration (heat and electricity) and material recycling (glass, paper, metals etc.).

In the first instance (first row of figure 3-17) the functions of the two waste management options are:

- Waste management through landfill of waste.
- Waste management through composting waste.

A related functional unit to the above functions would be *treatment of X ton waste*. This would however be wrong as it does not reflect the additional function of the composting (material production).

On the second row in the figure a material production function (virgin material) is added to the landfill system to make it equivalent to the recycling system (this is known as system expansion). A related functional unit would *be treatment of X ton waste and production of Y ton material*.

On the third row in the figure the virgin material is credited to the composting function to isolate the waste landfill waste management function. Of course it can be performed the other way around, where the recycled material is moved over and the composting is isolated.

This is known as the avoided product allocation and it is identical to system expansion – the preferred procedure according to the ISO standard for life cycle assessment. The related functional unit would now be *treatment of X ton waste* again. However, due to the system approach the compost system would have a system definition that makes the functional unit correct (in contrast to the first definition).

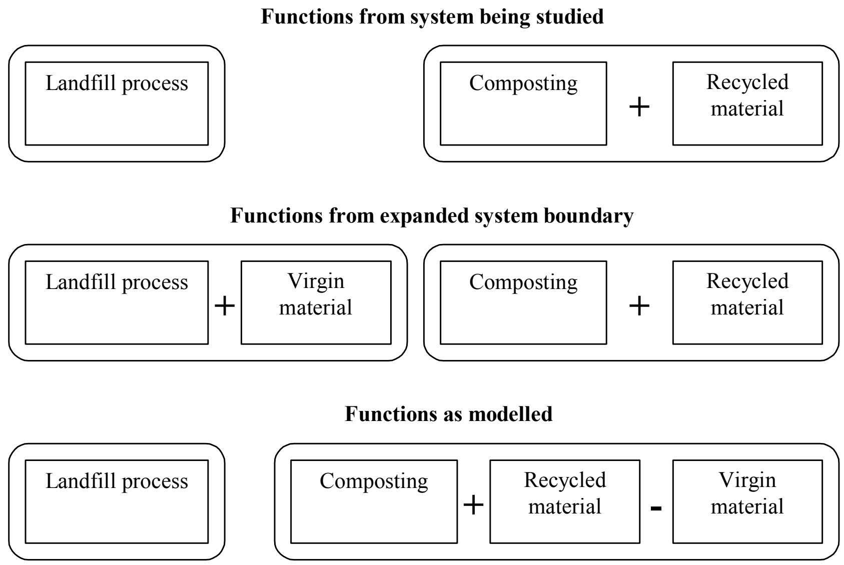

Figure 3-17 System expansion methodology

As a more specific example let us consider the comparison of recycling and incineration of waste paper. It is assumed that the recycled paper replaces virgin paper, while the recovered energy replaces oil combustion. According to Figure 3-17 and the level “functions from expanded system boundary”, the system models could become as illustrated in Figure 3-18.

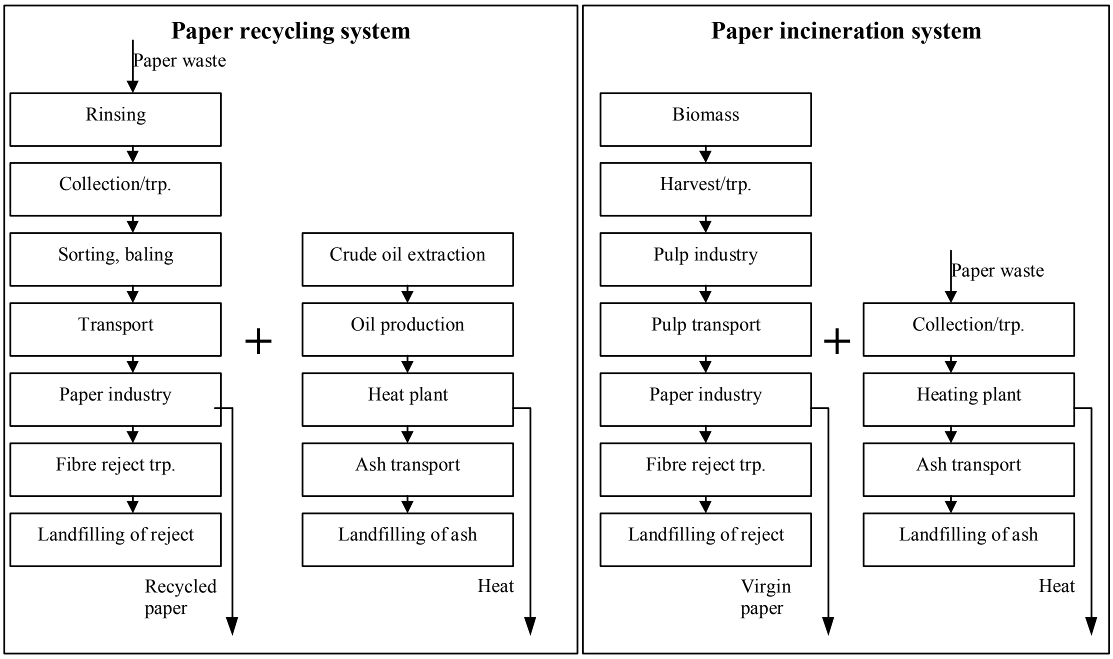

Figure 3-18 Systems for paper recycling and incineration after system expansion

3.6 Impact assessment

The general methodology on how to perform quantitative life cycle impact assessment (LCIA) is described in numerous methodology reports and in ISO 14042. Hence it will not be described in any detail in this report. If an LCIA is carried out, it consists of some mandatory elements and some optional elements /3/.

The mandatory elements (selection of impact categories and impact indicators, classification and characterisation) convert LCI results to indicator results for each defined impact category. The optional elements are normalisation, grouping or weighting and data quality analysis techniques. This is illustrated in Figure 3-19.

LIFE CYCLE IMPACT ASSESSMENT

Mandatory elements

Selection of impact categories, category indicators and characterisation models

↓

Assignment of LCI results (classification)

↓

Calculation of category indicator results (characterisation)

↓

Category indicator results (LCIA profile)

↓

Optional elements

Calculation of the magnitude of category indicator relative to reference information (normalisation)
Grouping
Weighting
Data quality analysis

Figure 3-19 Elements of an LCIA /3/

For the mandatory part of the LCIA, each impact category uses constant characterisation factors to calculate the potential contribution to the impact categories from the components identified in the LCI.

$$S_{ji} = M_i Q_{ji}$$

- S_{ji} Potential contribution to impact category *j* from component *i*
- M_i Amount of component *i* from LCI results
- Q_{ji} Characterisation factor for component *i* to impact category *j*

The sum of all S_{ji} describes the total contribution to impact category *j* from all components:

$$S_j = \Sigma S_{ji}$$

3.6.1 Impact categories, indicators and characterisation models

LCA applied for municipal waste management usually includes the same environmental impacts as LCA studies in general. Therefore, the same environmental impact categories as those established in generic LCA guidelines should be used. Based on the Danish UMIP study /19/ and the Nordic Guideline on Life-Cycle Assessment /11/ the impact categories listed in Table 3-18 can be applied. Which impact categories to include and how to perform the impact assessment must be defined in the scope of the study. The listed impact categories covers the categories used in most known LCAs applied in the waste management sector, including /6/, /7/, /8/, /9/, /16/ and /20/.

For many of the impact categories the classification and characterisation is quite straight forward, because the impact categories have been in use for a long time, the number of substances which contribute is manageable, and a certain degree of international consensus exits on which indicators to use and how contributions shall be modelled. A selection of approaches is briefly described in the table.

Ozone depletion is hardly ever an issue related to waste management systems in the Nordic countries due to the prohibition of ozone depleting substances. This is confirmed by the findings in /9/.

According to the Nordtest State of the art study /5/ it is especially assessment of toxicity impacts that needs further development with respect to application in the waste management sector. This impact category is treated separately in section 3.6.2.

The impact assessment can stop after the characterisation has been performed, or it can continue with normalisation and/or weighting (sometimes normalisation lies inherently in the weighting method).

Note that new characterisation models and characterisation factors are not developed within a specific waste management decision support projects. This is done in separate research project dedicated to that purpose. Most LCA practitioners use some kind of commercial LCA computer tool. These tools usually have several alternative characterisation models and associated characterisation factors. The LCA practitioners usually select one of the available models in the tool, or put in new models based on available research reports.

Not all components identified in an LCI can be assigned to an impact category. In such cases these substances should be listed separately. Also, there are weighting methods that weight the LCI components directly, rather that estimating the impact category indicator scores first.

Table 3-18 Commonly used impact categories, indicators and characterisation models

Impact categories	Commonly used indicator(s)	Characterisation model(s)
Global warming	Global warming potential (GWP) = CO_2-equivalents.	GWP for substances as defined by the Intergovernmental Panel on Climate Change (IPCC) and given e.g. in /19/.
Ozone depletion	Ozone depletion potential (ODP) = CFC11-equivalents.	ODP for substances as defined by the World Meteorological Organization (WMO) and given e.g. in /19/.
Photo-oxidant formation	Photochemical ozone creation potential (POCP) = C_2H_4-equivalents..	POCP for substances for relevant background concentration level of NO_x as given e.g. in /19/.
Acidification	Hydrogen ion (H^+) generation potential expressed as SO_2-equivalents.	H^+ generation potential taking into account regional/national recipient buffer capacity (removal of nitrates by plant harvesting). See /19/ for Danish adoption.
Eutrophication	Nutrient enrichment of water and soils.	Nitrogen limited recipients. Phosphorous limited recipient. Combined nitrogen and phosphorous limited. All three models with or without N to air. For all models see /11/.
Toxicity for ecosystems and humans	See section 3.6.2	See section 3.6.2.
Abiotic resource consumption	Weight (ton) Volume (m^3)	Usually the resources are split into renewable and non-renewable resources. The LCI results are transformed from weight to volume or vice versa by using material density.
Biotic resources consumption	Weight (ton) Volume (m^3)	The LCI results are transformed from weight to volume or vice versa by using material density.
Fresh water consumption	Weight (ton) Volume (m^3)	The LCI results are transformed from weight to volume or vice versa by using water density.
Land consumption	m^2 m^2*year	Land areas are usually split into area categories reflecting the present exploitation. See /11/ for examples of land area categorisation.
Materials not followed to cradle	Weight (ton)	All (selected) materials aggregated. Selected materials given separately.
Energy not followed to cradle	Energy content (MJ)	All (selected) forms of energy aggregated. Aggregation within the groups renewable and non-renewable energy.
Waste not followed to grave	Weight (ton) Volume (m^3)	All (selected) forms of waste aggregated. Aggregation within the groups non-hazardous waste and hazardous waste.
Smell	Potentially affected area (m^2)	Affected area is based on experience or dispersion modelling combined with smell threshold values. Can be split on area type as for land consume.
Noise	Potentially affected area (m^2)	Affected area is based on experience or noise modelling combined with noise acceptance criteria. Can be split on area type as for land occupation.

Table 3-19 and Figure 3-20 illustrates an LCIA profile. To be able to present the results in Table 3-19 on a common axis, for each impact category the sum of all three waste treatment system alternatives is set to 1. The alternatives are expressed as their relative contribution to 1. It should be noted that some of the impact scores may be very large and others insignificant - the figure does not tell anything about the relative size of the different impact scores.

7 of the categories in Table 3-18 are included. The profile is related to the comparison of 3 alternative treatment methods for 19500 ton household waste; 1) anaerobic digestion of biowaste and incineration of residues, 2) aerobic composting of biowaste and incineration of residues, 3) Incineration. The negative indicator scores occur due to the environmental benefit of recovered energy and recycled material. Note that the results are only valid for the boundaries, limitations and data applied in the specific study.

It is seen from the figure that 3) incineration is ranked as the best alternative for all impact categories, except waste generation.

Table 3-19 Example of LCIA profile /7/

Impact category	Unit	Anaerobic digestion	Aerobic composting	Incineration
Eutrophication	kg PO_4	-4.98e+04	-3.41e+04	-9.91e+04
Eco-toxicity	m^3 water/air	-2.66e+10	-1.91e+10	-4.97e+10
Global warming	kg CO_2	-5.97e+06	-4.96e+06	-1.36e+07
Acidification	kg SO_2	-2.42e+04	7.16e+03	-7.42e+04
Photo-oxidant formation	kg ethylene	8.60e+02	1.24e+03	6.69e+02
Human toxicity	kg body weight	-7.23e+06	-5.17e+06	-1.36e+07
Energy	MJ	-7.32e+07	-5.29e+07	-1.14e+08
Solid waste	kg	1.14e+04	4.11e+05	5.65e+05

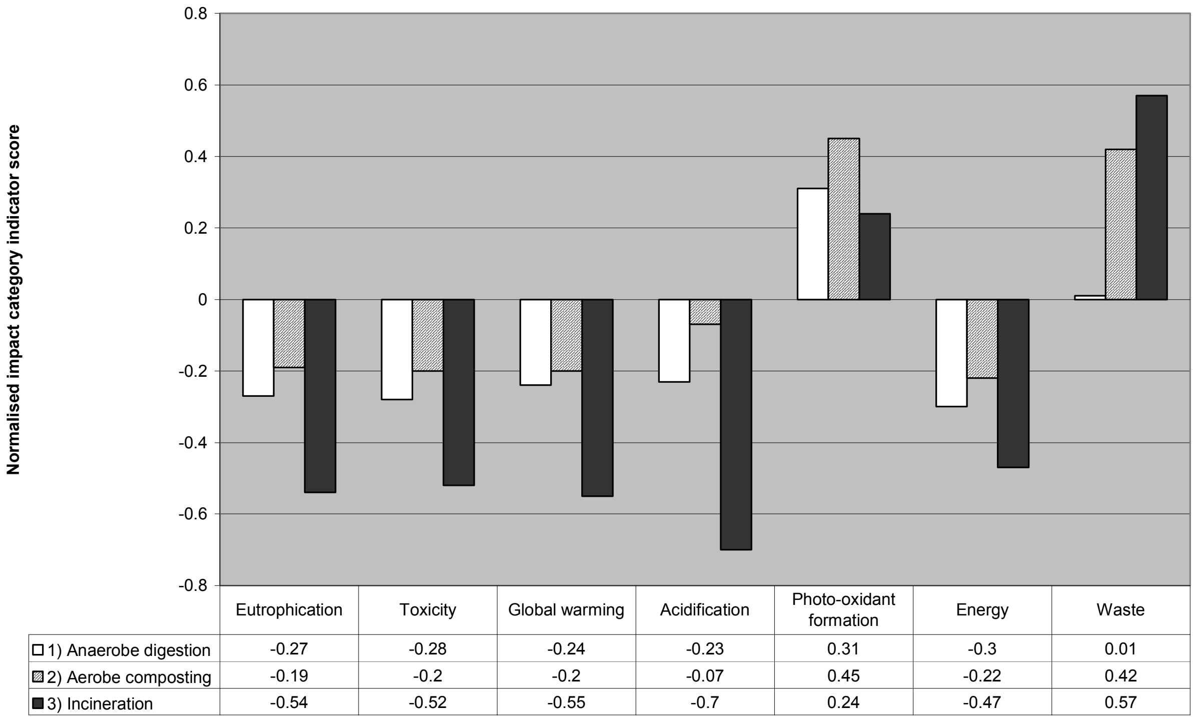

Figure 3-20 Example of LCIA profile /7/

3.6.2 Toxicity

The toxicity impact category is an important category for LCAs applied in the waste management sector. This, because emissions like dioxin, PCB, PAH and heavy metals are toxic and they are generated by a range of relevant processes related to waste management, such as:

- Toxic pollutants emitted to air from incineration and through evaporation from landfills, composting and bioreactors.
- Toxic pollutants leaching from landfill containing municipal waste and incineration residues.
- Toxic pollutant in soil improvement products recovered from composting of biowaste.
- Toxic pollutant emitted during production of consumed energy and auxiliary materials.
- Toxic pollutant emitted during production and use of substituted energy and materials.

The impact category toxicity is often divided into human toxicity and eco-toxicity. The reason for collecting them in a common category is because the toxic impact model regard the recipient as the same and does not include the different fates that the pollutants might have on humans and eco-systems.

The influence that the long release time horison for metals in landfills has on the toxicity but not on the calculated impact potentials for LCIA is a matter of discussion. The discrepance is caused by "dilution in time" which means that environmental concentrations below the landfill may be slightly increased for thousands of years. A risk assessment may tell us that this does not cause any significant risk but an LCIA looks at the mass emission which may be very large and hence cause a large impact potential. Dilution in time is an issue particularly for landfills as opposed to most other processes in the life cycle. It gives problems with the traditional LCIA approach based on mass loads and may call for alternative approaches (part of the justification for the distinction between the "short term" emissions (<100 years) and the long term emissions (>100 years) applied for landfills by many researchers.

The toxicity category is very complex. The main reasons for this are a large number of mechanisms, an enormous number of contributing substances, many affected natural resources and the inter-media transport of substances in the eco-system. In addition to dividing into human toxicity and eco-toxicity it is also common to divide eco-toxicity into aquatic, terrestrial and sediment eco-toxicity.

Several characterisation models exist for this impact category. Models applied in identified LCA studies on waste management are:

- Dutch USES-LCA model. This is a multi-media fate model that predicts the environmental concentrations after emission, and compares the concentrations with no-effect concentrations. Separates on aquatic eco-toxicity, terrestrial eco-toxicity, sediment eco-toxicity and human toxicity. See /23/ for further description (report can be downloaded from /22/). Applied in /9/.
- UMIP model for human toxicity. Separates on toxic exposure through air, water and soil. The model takes into account distribution on different environmental

compartments, exposure routes, human intake, transfer to human body and component toxicity. See /19/ for method description. Applied in /9/.

- UMIP model for eco-toxicity. Separates on acute aquatic eco-toxicity, chronic aquatic eco-toxicity and soil eco-toxicity. The model takes into account distribution on different environmental compartments, bioaccumulation and toxicity. See /19/ for method description. Applied in /9/.
- Dutch model for human toxicity and eco-toxicity proposed by the Centre for Environmental Studies (CML), University of Leiden, in 1992 (CML-92). The human toxicity effect factor for each component is equal to the inverse of the human tolerable daily intake (mg/kg body weight). Emissions to air and water are treated separately and then added. The eco-toxicity effect factor for each component is equal to the inverse of a threshold concentration (mg/m^3) in water. There is thus no consideration of the substance's environmental fate in the model. See /21/ for further description of method. Both categories are applied in /6/, /7/ and /8/. Note that the CML toxicity impact assessment methods have been extensively updated since 1992. The upgrading e.g. includes USES-LCA. See /22/ for overview of updated methods and models.

SETAC recommends that impact assessment of toxicity should take into account /24/:

- The toxicity of the component.
- Differences in human toxicity and eco-toxicity.
- Fate and exposure (not the case CML-92)
- Background concentration dependency (not the case for any of the models)
- Regional geographical differentiation (not the case for any of the models)

Based on the amount of data included in the models, the recommendations of SETAC and what is in use in the latest LCA studies in the waste management sector, it is recommended to use one of the first two models listed above. It is referred to /11/ and /24/ for overview of other models generally applied in LCA studies.

In some studies toxicity is excluded from the LCA study with the argument that no credible methods exist and due to lack of data /16/. This is OK as long it is clearly stated in the scope of that study that the toxicity impact assessment is not included, and as long as the goal of the study can be met without inclusion of chemical impacts in the impact assessment.

As for other impact categories, new characterisation models and characterisation factors for toxicity impact assessment are not developed within a specific waste management decision support projects (ref. last paragraph in section 3.6.1).

As an example, Table 3-20 lists some of the most common metals related to waste treatment processes and relates human toxicity characterisation factor for emissions to air based on the USES-LCA and UMIP methods. Only metals that have given values in both models are included in the table.

The effect factors from the two models cannot be directly compared, as the models for deriving values are different. However, the relative importance of the metals can be compared using a reference metal that equals 1. We use cadmium as a reference metal. All other metals are then given as cadmium equivalents. The results of this calculation are given in column 4 and 5 in the table. In column 6 the ration between the USES-LCA effect- factors and the UMIP effect factors are given. It shows that there are large differences in the prioritisation of metals in the two models, especially for chromium, cobalt, copper and lead.

This difference may change the ranking of alternative waste treatment solutions. Especially related to human toxicity. There are reasons to believe that similar differences also exist for other components that are toxic to humans and ecology. Due to this, it is recommended to apply both models in an LCA if the project frames allow for it.

Table 3-20 Human toxicity effect factors for emissions to air /19/, /22/

	USES-LCA Human health Metals emitted to air Effect factors	UMIP Human health Metals emitted to air Effect factors	USES-LCA given as Cd-equivalents	UMIP given as Cd-equivalents	USES-LCA/ UMIP ratio
Cadmium	1.50E+05	1.10E+08	1.00E+00	1.00E+00	1.00E+00
Chromium III	6.50E+02	1.00E+06	4.33E-03	9.09E-03	4.77E-01
Chromium IV	3.40E+06	1.00E+06	2.27E+01	9.09E-03	2.49E+03
Cobalt	1.70E+04	9.50E+03	1.13E-01	8.64E-05	1.31E+03
Copper	4.30E+03	5.70E+02	2.87E-02	5.18E-06	5.53E+03
Lead	4.70E+02	1.00E+08	3.13E-03	9.09E-01	3.45E-03
Mercury	6.00e+03	6.70E+06	4.00E-02	6.09E-02	6.57E-01
Molybdenum	5.40E+03	1.00E+05	3.60E-02	9.09E-04	3.96E+01
Nickel	3.50E+04	6.70E+04	2.33E-01	6.09E-04	3.83E+02
Selenium	4.80E+04	1.50E+06	3.20E-01	1.36E-02	2.35E+01
Thallium	4.30E+05	5.00E+05	2.87E+00	4.55E-03	6.31E+02
Vanadium	6.20e+03	1.40E+05	4.13E-02	1.27E-03	3.25E+01
Zinc	1.00E+02	8.10E+04	6.67E-04	7.36E-04	9.05E-01

3.6.3 Normalisation

To be able to present the impact assessment results on a common axis, to enable comparison and/or to form the foundation for subsequent weighting, normalisation is performed. This means that the impact category indicator results are divided with associated reference values.

$N_j = S_j/R_j$

- N_j The normalised indicator score of impact category *j*
- S_j The total indicator score of impact category *j*
- R_j Reference value of impact category *j*

Examples of references used in normalisation are:

- The status of relevant impact categories within the geographical area for which the study shall be representative (e.g. national, European or global) within a specified period of time (usually latest available year). The same geographical area and time frame is applied for all impact categories.
- The status of relevant impact categories within the geographical area that the impact category has an effect (e.g. global reference for global warming and

national reference for acidification) within a specified period of time (usually latest available year).

Note that normalised results do not state which impact categories are the most important for a waste treatment system, although some clue is given when there are magnitudes of difference between different normalised impact scores. To state importance weighting must be performed. The normalisation will only show to which problems the waste system contributes the most.

3.6.4 Weighting

Weighting is the optional step of an LCIA where the different impact categories are weighted so that they can be compared among themselves. The aim is to arrive at a further interpretation and aggregation of the data of the impact assessment. Some weighting methods apply weighting factors directly to the LCI results. The weighting factors are then usually established partly based on an inherent effect factor.

$$W_j = N_j F_j$$

- W_j The weighted category indicator score of impact category *j*
- N_j The normalised indicator score of impact category *j*
- F_j Weighting factor for impact category *j*

Aggregation of impact categories can be carried out after weighting.

$$W = \Sigma W_j$$

Note that some weighting methods use different weighting models for resources consumption, ecological effects and human health/work environment. In these cases aggregation is only possible within the same type of model if not other is specified.

Many weighting methods exist, but no methods have been identified that are particularly developed for application in LCAs in the waste management sector. Hence, the range of weighting methods available for generic LCA studies are also applicable in the waste management sector.

It will be a too comprehensive task for this guideline to go through alternative weighting methods. Based on availability of weighting methods in LCA computer tools, the newest and most commonly used weighting methods applied in the Nordic countries are:

- Environmental Design of Industrial Products (UMIP) /19/. The method uses a distance to target approach.
- Eco-indicator. Eco-indicator 99 latest version /25/.
- Environmental Priority strategies (EPS). EPS 2000 latest version /26/.

All the above weighting methods require that the impact assessment and normalisation is carried out in a specific way before weighting can be performed. The references given in the bullet list above describes the methods and gives weighting factors.

It is emphasised that weighting is a controversial issue and there is no consensus within the Nordic countries or other international fora on recommended weighting methods. The only recommendation made in several publications is that more than one weighting method should be applied to a study if weighting shall be carried out. This should especially be valid for LCA applied in waste management as comparison results often are made public and can generate a lot of basis for discussion.

3.7 Interpretation of results

The interpretation phase of an LCA is defined by ISO as /1/:

The phase of life-cycle assessment in which the findings of either the inventory analysis or the impact assessment, or both, are combined consistent with the defined goal and scope in order to reach conclusions and recommendations.

The procedure of interpretation is further elaborated as /4/:

To analyse and report results, reach conclusions, explain limitations and provide recommendations for an LCI or an LCIA study.

A request to analyse, conclude and recommend presumes that there is a question to answer or a problem to solve. Related to waste management the main questions where LCA can help answer are:

- What part of the waste treatment system should be in focus for environmental improvement?
- In case there are several solutions for improvement, how good are the solutions compared to each other in an environmental perspective?
- What are the total environmental impacts associated with different conceptual waste treatment alternatives and how do they perform compared to each other?

The main issues recommended to be included during the interpretation phase of a quantitative LCA are /27/:

- Based on knowledge about the system, identify the methodological choices that significantly affect the performance of the system.
- Define data quality indicators and evaluate data quality. Where possible, estimate uncertainty ranges.
- Completeness check. Determine if missing information, such as data gaps, data quality gaps, information gaps on technical methodological choices, are crucial to the goal and scope of study.
- Sensitivity analysis. Determine if a sensitivity analysis, that is a study of the influence of identified technical and methodological variables, is necessary. If yes, design a factorial scenario calculation plan. Carry out the calculations in a deterministic way, i.e. without considering data uncertainty.
- Uncertainty analysis. Determine whether or not an uncertainty analysis, i.e. replicate calculations of scenario with varying values of selected data elements, is necessary. If yes, make replicate calculations of at least one experiment with selected Y-parameters, representative of identified clusters. Determine if the spread of the replicates is larger than the variance between different scenarios.
- Conclude, from the uncertainty analysis, whether the data quality is sufficient or not. If yes, determine whether or not there are significant differences between the scenarios, and the cause of such differences.

Sensitivity or uncertainty analyses should be performed on the major assumptions and uncertainties. E.g. a sensitivity analysis can reflect assumptions about changes in the market. In a declining market, it is usually not invested in new technology. The old technology will then not be exchanged in a future scenario and the contribution from old technology should be included (if old technology exist within the geographical boundaries). In an increasing market new technology will be built and therefore BAT scenarios can be applied in a future scenario.

Another example is a study with the goal to compare two alternative solutions for treatment of paper waste (paper recycling and incineration) /27/. Sensitivity analysis and uncertainty analysis were performed. The variables in the sensitivity analysis were:

- Input data (generic or specific)
- Heat production from fuel (oil or biomass)
- Transport distance to paper industry (106 or 300 km)
- Paper composition (100% cardboard, 50% each of cardboard and liquid cardboard, 100% liquid cardboard)

The sensitivity analysis gives the possibility to reach conclusions within some specified presumptions. E.g. it is concluded that material recycling is better the incineration with respect to CO_2 emissions provided that biofuel is used to produce replacement heat from incineration, and that the waste paper contains at least 50% liquid cardboard.

A general important conclusion is that there is no such thing as an unambiguous environmental effect of a change of the waste paper treatment technology. There are reasons to believe that this conclusion is also valid when assessing other waste flows and other waste treatment alternatives.

4 OVERVIEW OF LCA STUDIES IN THE WASTE MANAGEMENT SECTOR

The following chapter gives an overview of the current work on LCA in the waste management sector. It lists groups working on LCA in the waste management sector, provides links to web pages and overview of projects, studies and LCA models being made. Some of the information in the following chapter may become outdated soon after the publication of this report as the projects proceed.

4.1 ORWARE – Sweden

ORWARE is an acronym for Organic Waste Research. It is a model for analysing both environmental and economic aspects of waste management strategies, based on life cycle perspective and developed in cooperation between several Swedish research institutes and institutions. Among treatment methods that can be simulated with the model are incineration, composting, anaerobic digestion, biocell, landfilling, sewage treatment and transport. The model includes production of material, energy and plant nutrients (N and P) by waste treatment, which can substitute virgin raw material in the studied system. There are several ORWARE related projects running at the moment but the main LCA-based project is funded by the Swedish National Energy Administration (STEM) is finished after running for four years. A final report from the project is under editing /38/. Further information about the ORWARE project can be found on the projects web site: http://www.ima.kth.se/forskning/orware

4.2 ESRG - Sweden

The Environmental Strategies Research Group (ESRG) has performed a study in order to evaluate different strategies for the treatment of solid waste based on a life cycle perspective. The goal of the study was to identify advantages and disadvantages of different methods for the treatment of solid waste, and to identify critical factors in the systems, including the background systems. The waste fractions considered were the compostable, combustible and recyclable fractions of municipal solid waste. The waste treatment options considered were landfilling, incineration, recycling, digestion and composting. The project was completed in August 2000 and the result is presented in the report "Life Cycle Assessment of Energy from Solid Waste" /9/. The report and further information can be downloaded from the ESRG web site: http://www.fms.ecology.su.se

4.3 The LCA-LAND model and projects in Denmark

At the Department of Manufacturing Engineering at the Technical University of Denmark a model for analysing emission from municipal solid waste landfills and waste incineration plants in Denmark, the Netherlands and Germany has been developed. The model was developed as a part of the project LCAGAPS, a EUREKA project, which focuses on developing solutions to remediate identified lacks and shortcomings of existing life cycle assessment methods. The model is product specific, which means that emission from the waste treatment is allocated to the products being landfilled or incinerated. In the model, waste is divided into five groups: Specific organic compounds (e.g. organic solvents), general organic matter (e.g. paper), inert components (e.g. PVC), metals and inorganic non-metals (e.g. chlorine). Different types of solid waste are a composite of these five groups and the model calculates emission to water and air from products during the first 100 years of the landfill. The model has been made operational in a computer tool called LCA-LAND /40, 41, 42/.

A project is now running by Cowi Consult in Denmark in co-operation with the Department of Manufacturing Engineering at the Technical University of Denmark, concerning how to model landfilling of different types of residuals from incineration and electricity production /43/.

Web site:

Department of Manufacturing Engineering at the Technical University of Denmark: http://www.ipt.dtu.dk/engelsk/index.html

4.4 WISARD - EA and Ecobilan (UK)

In the UK three LCA models for solid waste management have been developed and applied. These are WISARD (developed by Environment Agency and Ecobilan), IWM2 (Procter and Gamble) and the Wasteman model (AEA technology) /44/.

The Environment Agency (EA) of England and Wales initiated in 1994 a life cycle program for waste management. In December 1999, WISARD (Waste-Integrated Systems Assessment for Recovery and Disposal) computer software designed to help waste managers identify more sustainable integrated approaches to waste management was launched. The tool includes the data on waste management operations and processes compiled under the Agencys programme, as well as background data on raw materials, energy and other processes in life cycle from Ecobalance UKs (The Ecobilan Group) proprietary life cycle database, DEAM /45, 46/.

The EA LCA research programme has focused on the development of the WISARD LCA software and numbers of projects are running simultaneously. These projects are:

- Data development and refinement for WISARD e.g. home composting LCA data and the collection of financial data on the waste management.
- The development and enhancement of the WISARD software
- Guidance on the use of Impact Assessment in LCA to local authorities.

A number of LCA studies using the WISARD program have been performed in the UK. These are e.g. assessment of the Scottish Waste Strategy and Area Waste Management Plan by the Scottish Environmental Protection Strategy (SEPA), consultant led studies concerning the development of local authority municipal waste management strategies and applications of WISARD for test/controversial waste planning applications /44/.

Web site:

Environment Agency: http://www.environment-agency.gov.uk
Ecobilan: http://www.ecobilan.com/uk_wisard.php

4.5 IWM2 - Proctor and Gamble (UK)

Dr. P. White *et al.* /47/ published in 1995 the book "Integrated Solid Waste Management – A Lifecycle Inventory". Included with the book is a software model (IWM-1) that allows the prediction of the overall environmental burdens and economic costs of municipal waste management. This model has been used by several local authorities in the UK and other EU countries during the development of their integrated waste management /48/. In a second edition of the book, an upgraded version of the model, IWM-2, is provided on a CD /49/. The model has been developed and made more user-friendly for waste managers. IWM-2 is designed to be an "entry level" LCI model for solid waste and appropriate to users starting to apply lifecycle thinking to waste systems. Among sections of waste management that are treated in the model are waste collection, sorting, biological treatment, thermal treatment, landfilling and materials recycling. Proctor and Gamble are currently using the model in countries with developing economies such as Mexico, Brazil, Russia and China /50/.

4.6 IWM model for municipalities – Canada

In Canada, the Environmental and Plastic Industry Council (EPIC) and Corporation Supporting Recycling (CSR) commissioned the development of an environmental analysis model to evaluate the life cycle environmental and energy effects of waste management processes. The object of the project was to provide Canadian municipalities with tools that will enable them to evaluate the environmental and economic performance of the various elements of their existing or proposed waste management systems. The model uses life cycle methodology to quantify the energy consumed and the emissions released from a user specified waste management system. It uses data specific to the user municipality to ensure applicability of the results and accuracy but at the same time default values have been provided to allow the user to undertake a first level screening evaluation.

The model includes the processes: waste collection, waste transfer, sorting of recyclable materials at a material recovery facility, reprocessing of recovered materials into recycled materials, composting, energy recovery and landfilling. Recycled materials, compost and recovered energy are accounted for as avoided burdens i.e. avoided production of virgin materials, conventional soil amendments and energy produced form combustion of fossil fuels. Additional information on the boundaries, data sources, parameters and assumptions used in the development of the model is provided in a Project Report available from the EPIC and CSR.

For further description of the model, its applicability and information about availability of the model, refer to the project web site:
http://www.iwm-model.uwaterloo.ca
http://www.iwm-model.uwaterloo.ca/iswm_booklet.pdf

4.7 U.S. EPA model

Through funding by the United States Environmental Protection Agency (U.S. EPA), a municipal solid waste decision support tool (MST-DST) and life-cycle inventory

(LCI) database for North America have been developed. The MST-DST methodology incorporates both full cost accounting and life cycle inventory analysis (LCI) and is now being used in variety of case studies across the United States.

The solid waste management systems analysed may be existing systems, entirely new systems or a combination of both, based on user specific data on municipal solid waste generation, requirements to the system, etc. The processes that can be modelled include collection, transfer, separation, composting, incineration, landfilling and digestion with biogas production. Through an optimisation module the user can identify objectives as minimizing total cost or life cycle parameter such as energy consumption and greenhouse gases. Because much of the data needed for modelling are not readily available to the user, effort has been expended in developing realistic and credible default values for input parameters. To provide a wider accessibility at a lower cost, development of a web-based version of the MST-DST is now being considered /51/.

For further information about availability of the MSW-DST, LCI database and project documentation, refer to the project web site:
http://www.rti.org/page.cfm?objectid=760BD7F2-7050-4FD3-B0EB101FB48210C8

4.8 International expert group on life cycle assessment for integrated waste management

Members of the group are experts on life cycle assessment for integrated waste management from ten countries from all over the world.

The objective of the group is to promote more sustainable waste management through the appropriate use of life cycle techniques and the goal is the development and use of life cycle tools for integrated waste management. The group intents to achieve it's goal by e.g. exchanging information on research and development projects, exchanging inventory data, agreement on the way that major technical issues are dealt with, identification of data gaps, research needs and scope for collaboration /44/.

According to the Secretarial of the group the, the groups web page has recently closed but it contained information about members meetings and technical documents /44/.

5 FINDINGS OF LCA IN THE WASTE MANAGEMENT SECTOR

In chapter 3, various aspects of LCA in the waste management sector were outlined and suggestions made for the best practice. Although limited attention has been given to waste treatment in LCA compared to other stages in the products life cycle, a lot of studies have already been performed. In the following chapter the findings of these studies will be discussed. However, the conclusions from the LCA studies cannot be regarded as general in any way as the results of LCA studies are site dependent and depend on assumptions and choices made in each study separately.

5.1 LCA as a basis for decision making

When a municipality decides to carry out an LCA study the intention is usually to compare environmental burdens of future alternatives, in waste treatment, to the current situation or test the current waste management plan or strategy /6, 7, 48, 52, 53/. Many of these studies are also intended to provide the overall economic cost of the system /16, 48, 51/. Finnveden and Ekvall /54/ reviewed several LCA studies concerning recycling of paper packing products and concluded that these studies were unable to decided whether recycling or incineration is better from an environmental perspective. This was mainly because the studies did not take into account all the relevant environmental impacts. The results also depended on a number of key issues which were uncertain (i.e. aspects of the studied system) and the valuation element also includes ideological and ethical aspects, which cannot be finally decided. Besides, since the environmental impact depends on other policy decisions, the question of whether or not to recycle or incinerate waste paper is too narrow a formulation. Other policy areas, such as heat and electricity production, waste management and forestry had to be considered as well. In a paper by McDougall and White /48/ a number of lifecycle inventory (LCI) case studies were reviewed. McDougall and White conclude that LCI could be used as a tool to demonstrate the environmental and economic benefits and the necessity of a certain type of waste management. However, the tool cannot make decisions based solely on the information it provides. The decision making process required to improve waste management strategies, still must come from a dialogue between waste managers, politicians, planners and the public /48/. In LCA studies made for three municipalities in Sweden, using the ORWARE model, no conclusion could be reached regarding whether one waste treatment alternative was better than the other except that landfilling usually was the worst choice /16/. Each of the alternatives (incineration, composting, anaerobic digestion and recycling) had its pros and cons. Utilisation of energy and material from the waste gave credits to the treatment alternative both from environmental and economical perspective. Therefore, the choice of treatment method had effects outside the waste treatment system as regards the production of electricity, heat, plastic, cardboard and fertilisers.

From the results and the discussion above, it can be concluded that even though the LCA method and the LCA studies can be improved, one can usually not draw the conclusion that any product A is environmentally preferable to a given product B from the results of an LCA study. LCA will play a role in providing a better basis for decision making by identifying key issue aspects, which are of importance when making a decision.

5.2 Data gaps

In chapter 3 data quality in an LCA study was addressed. LCAs require the acquisition of significant amounts of data and the quality of that data determines the utility of the final LCA. After studying several case studies and databases Finnveden /55/ concludes that data gaps limit the inclusion of several impact categories or cause them to be less well covered and therefore limits the types of conclusions that can be drawn form these studies. These impact categories were e.g. land use and impact on biodiversity, human and ecotoxicological impact categories, eutrophication of aquatic systems and photo-oxidant formation /55/. Human and ecotoxicological impact categories have severe data gaps due to the large number of possible pollutants that end up in the waste or are produced by waste treatment and lack of knowledge of the behaviour of all these pollutants. In LCA studies of future waste management options in three municipalities in Sweden where the ORWARE model was used, ecotoxicological impacts were not quantified due to data gaps and lack of methods to weight different emissions /16/. In a study by Finnveden *et al.* /9/ treatment of various fractions of municipal solid waste with different alternatives were analysed. Due to data gaps more emphasis was put on the total energy use and emission of greenhouse gases in the study (as these impacts categories are better known) than toxicological impact categories. Finnveden *et al.* /9/ conclude in the study that emission with toxicological impacts and impacts from land use need further attention.

Data for the stages of the lifecycle where direct measurements are possible are normally more certain than data from e.g. landfill where data have to be estimated. Long timeperspective makes experiments and field studies on landfills difficult to perform and therefore the uncertainty with landfill models may be large. In the case studies done by Det Norske Veritas for municipalities in Norway, future options in treatment of municipal solid waste and sludge were analysed /6, 7/. Existing process and transport data from the municipalities or neighbour municipalities were used. Data from background processes were from LCA databases. Landfilling of waste was not an alternative in the studies. Instead the impact category “solid waste” was used and the amount of waste produced by the different waste treatment alternatives was reported as “solid waste”. This limits the conclusions that can be drawn from the study as impact of various “solid waste” fractions are not studied, but it simplifies the study. In models such as the ORWARE model, US. EPA model and the IMW2 model, landfill is included. However, these landfill models are based on number of assumptions and predictions about future behaviour of the landfill.

According to the above, data gaps are associated with specific impact categories, mainly concerning toxicological effects, and processes that can not be measured due to long duration. As mentioned in the former subchapter, these data gaps limit the usefulness of LCA as a decision supportive tool because not all impacts are considered to the same extent. Ranking of waste treatment alternatives relative to those environmental impacts as well as weight total impact may also be wrong.

5.3 Ranking type of waste treatment

The waste hierarchy of solid waste i.e. the preference of recycling over incineration over landfilling is often taken as a rule of thumb (difference between incineration, composting and anaerobic digestion is usually small). However, by using life cycle assessment the validity of the waste hierarchy has been tested and proven to be dependent on assumptions and value choices that can be made /9/. Often different choices can lead to more variations in the final result of an LCA study than the differences between the alternatives that are studied. Therefore, the type of waste treatment can not be ranked relative to the environmental impact, without making assumptions and taking choices of value into consideration and different choices are appropriate for different decisions and perspectives. The effect of different choices should be analysed by sensitivity analyses when comparing different waste treatment options.

Choices that affect the comparison of different waste treatment relative to environmental impact are e.g. time aspect of landfilling /e.g. 9/, substitution of new material by recycling /e.g. 57/, energy utilisation from waste /e.g. 7, 9/, choice of allocation principles /61/ and impact categories /55/.

5.3.1 Time perspective

As noted in chapter 3 emissions from landfills may prevail for a very long time, often thousands of years or longer. The choice of time frame in the LCA of landfilling may therefore clearly affect the results. Choosing a short time perspective i.e. shorter than 100 years, makes the landfill a carbon sink relative to other treatment options e.g. incineration, as e.g. plastic material is not degraded /9/. Likewise, metals have not leached out of the landfill during such a short time /59/. Therefore, short time perspective credits the landfill alternative in the LCI as less emission has occurred.

5.3.2 Recycling of material

Recycling of material and energy from waste can be done in several ways and recycled material can substitute virgin material in several ways. The choice of substituted virgin material or energy and the quality of the recycled materials affect the ranking of recycling compared to other waste treatment alternatives. The key factors when crediting the recycling of paper are what energy is replaced by energy from incineration of wastepaper, what material is replaced by the recycled fibres, how pulpwood savings are used when recycled fibres replace virgin fibres and external fuel and electricity demand in paper production /54, 57/. If heat from incineration replaces fossil fuel, recycling will lead to increased use of fossil fuels and associated impacts. However, in studies where wood for paper production has been “saved” due to recycling of paper and instead used as fuel, recycling benefits since the use of fossil fuel can be reduced /54/. In a study by Finnveden /9/, quality of recycled material (paper and plastic) was modelled so that one kg of waste material would not replace exactly one kg of virgin material. This was because the losses and sorting out during the process, and in the case of paper and board products the fact that the quality would not be as good and therefore a larger amount of fibre would be necessary in the recycling case. When recycling organic fertiliser products (sewage sludge, reactor compost, and anaerobic digestion sludge) two quality aspects have to be considered, the nutrient availability and the content of polluting compounds from the waste /5, 52, 53/. Metal content of biologically treated waste may limit the use of organic fertiliser produced. In a study carried out by Björklund *et al.* /52/ on waste management in Stockholm it was concluded that the spreading of organic fertiliser products had to be limited so as not to exceed the regulatory limits of metals (g metal/ha, year). The results of a case study for Uppsala municipality were similar, i.e. metal content of sewage sludge and compost limited their use in agriculture but digester sludge could be used to provide the entire dosage need of phosphorus /53/.

5.3.3 Energy recovery

Energy recovery from waste can be e.g. heat and electricity from incineration and methane gas from landfilling or anaerobic digestion. In a study carried out by Det Norske Veritas /7/ for a municipality in Norway, anaerobic digestion with methane gas production, composting and incineration were compared. The results of the study were mostly dependent on energy recovery possibilities of the treatment methods. Incineration and anaerobic digestion were ranked higher than composting because of heat and biogas production. Ranking of incineration compared to anaerobic digestion was however dependent on the energy efficiency of the incineration plant and which energy source the gas produced substituted. If the methane gas substituted heat production by oil, anaerobic digestion was credited very high. However, if it substituted electricity production, the ranking of anaerobic digestion was not as high because electricity is mainly produced by hydropower in Norway, which is relatively "clean". In a study carried out by Finnveden *et al.* /9/ substitution of various avoided heat sources was analysed for ranking of landfilling, incineration and recycling. The energy sources were forest residues, natural gas and "saved" forest from paper recycling. As more energy is recovered through incineration than landfilling, the use of non-renewable heat sources (natural gas) lowered landfilling to the least preferred option. In a Swedish study where the ORWARE model was used /60/, the results were the same i.e. composting, which produced the least useable energy from the waste (compared to anaerobic digestion and incineration), became the worst scenario when coal was used instead of biofuel for heat production.

It can be concluded that the effect of various energy source substitutions is site dependent as the energy production at different sites varies. Substitution of non-renewable energy sources credits the system more than substitution of renewable energy sources.

5.3.4 Collection and transportation

Collection and transportation of waste are unit processes, which should be taken into consideration when making a life cycle assessment of waste management. Emission from transport vehicles can represent a large part of the emission from the foreground system /56/. Fuel consumption for waste transport may increase as nutrient recycling and source separation increase. It should however be noted that transportation may also decrease as a result of increased recycling as transport of virgin material is decreased /54/. Several studies have been performed to analyse the importance of transport on LCI results /57/. The conclusion of these studies is that transportation has limited influence on LCI results concerning energy demand and emission of CO_2, SO_2 and NO_x, under the assumption that the transportation is reasonably efficient (i.e. no transport of small volumes in cars). Other types of environmental problems, such as cancer and respiratory diseases may however be influenced by transportation. Finnveden *et al.* /9/ studied the effect of different transportation distance by truck to treatment facility and the effect of transporting waste by passenger car. The study showed that different distance for transportation of waste by truck to treatment facilities did not influence the ranking of treatment options studied. However, transport of waste by passenger cars from household to collection points influenced the results significantly concerning the impact categories photochemical oxidant formation and human and ecotoxicological impacts /9/. In a study by carried out by Det Norske Veritas for Bærum municipality in Norway, different transport distance to three waste incineration plants had effect on ranking of these plants relative to photochemical oxidant formation /7/. In LCA studies performed for the municipalities Uppsala, Stockholm and Älvdalen longer regional transport was of little significance as long as the transport was carried out in an efficient manner /58/.

Choice of collection and transportation may have influence on some impact categories but in general, as long as the transportation is reasonably efficient, it will have no effect on the conclusion of an LCA study.

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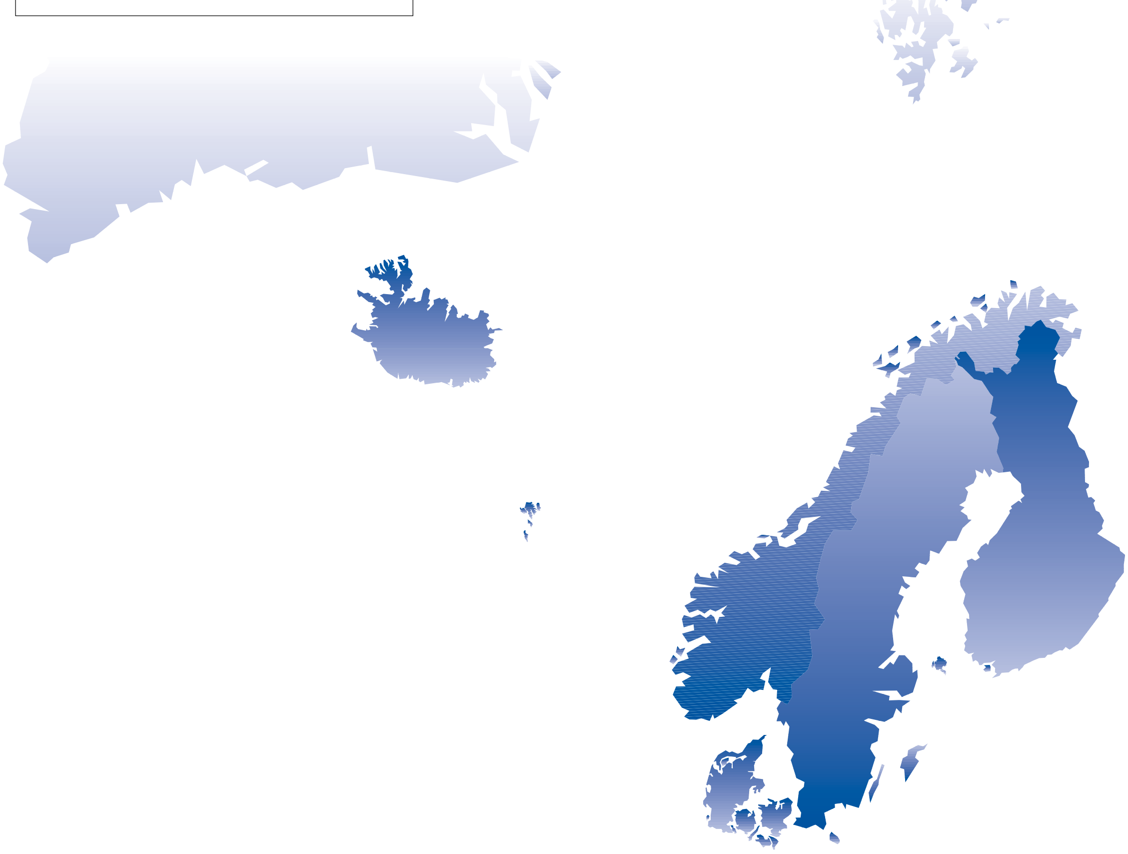

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