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Implementation of life cycle assessment models in solid waste management

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DTU Environment
Department of Environmental Engineering

PhD Thesis
June 2010

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The thesis will be available as a pdf-file for downloading from the homepage of the department: www.env.dtu.dk

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Preface

The work reported in this PhD thesis, entitled 'Implementation of life cycle assessment models in solid waste management', was conducted at the Department of Environmental Engineering at the Technical University of Denmark, with Professor Thomas Højlund Christensen as supervisor. The PhD project ran from March 2006 to April 2010 and was funded by the 3R (Residual Resources Research) research school at the Technical University of Denmark three-quarters as an employee. The last quarter of the time was spent working as a research assistant.

The content of the PhD thesis is based on six papers prepared for scientific journals. The papers represent the many sub-projects included in the PhD project and conducted in collaboration with internal and external partners. In the text, the papers are referred to by the names of the authors and their appendix number, written with Roman numerals.

I Gentil, E., Damgaard, A., Hauschild, M., Finnveden, G., Barlaz, M., Thorneloe, S., Kaplan, P.O., Eriksson, O., Matsui, Y., Ii, R., Muller, O., Christensen, T.H. (2010). Models for waste Life cycle Assessment: Technical assumptions review. Manuscript submitted to *Waste Management.*

II Damgaard, A., Riber, C., Fruergaard, T., Hulgaard, T., Christensen, T.H. (2010). Life-cycle-assessment of the historical development of air pollution control and energy recovery in waste incineration.
Waste Management, Article in Press,
http://dx.doi.org/10.1016/j.wasman.2010.03.025

III Damgaard, A., Manfredi, S., Merrild, H, Stensøe, S., Christensen, T.H. (2010). LCA and economic evaluation of landfill leachate and gas technologies. Manuscript submitted to *Waste Management.*

IV Merrild, H., Damgaard, A., Christensen, T.H. (2008). Life cycle assessment of waste paper management: The importance of technology data and system boundaries in assessing recycling and incineration. *Resources, Conservation and Recycling*, 52(12), 1391-1398.

V Zhao, Y., Damgaard, A., Wang, H., Lu, W., Christensen, T.H. (2009). Life cycle assessment of the municipal solid waste management system in Hangzhou, China (EASEWASTE). *Waste Management &Research*, 27, 399-406.

VI Damgaard, A. Larsen, A.W., Christensen, T.H. (2009). Recycling of metals: accounting of greenhouse gases and global warming contributions. *Waste Management & Research*, 27, 773-780.

In addition, the following publications were produced during the PhD study:

Christensen, T.H., Bhander, G.S., Lindvall, H.K., Larsen, A.W., Fruergaard, T., Damgaard, A., Manfredi, S., Boldrin, A., Riber, C. & Hauschild, M.Z. (2007). Experience with the use of LCA-modelling (EASEWASTE) in waste management. *Waste Management & Research*, 25, 257-262.

Merrild, H., Damgaard, A., Christensen, T.H. (2009). Recycling of paper: accounting of greenhouse gases and global warming contributions. *Waste Management and Research*, 27, 746-753.

Boldrin, A., Neidel, T.L., Damgaard, A., Bhander, G.S., Christensen, T.H. (2010) Modelling of environmental impacts from biological treatment of municipal organic waste (EASEWASTE). Manuscript for *Waste Management & Research.*

The papers are not included in this online version, but can be obtained from the library at:

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May 2010
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A special acknowledgement to Professor Morton Barlaz (NCSU, USA) who, during my master thesis, gave me a first glimpse into the world of solid waste research and offered feedback on some of my work during the PhD.

Similarly, a thank you goes to the team of modellers participating in the international model comparison workshop at DTU in November 2008.

Many thanks also go to Lisbet Brusendorff and Torben Dolin from the graphic office, who produced all illustrations in the papers and the thesis.

Last, but not least, I would like to thank my family and friends outside the university for listening to talks about waste and LCA, as well as putting my mind on things totally unrelated to work.

List of Abbreviations

APC – Air-Pollution-Control
BAT – Best Available Technology
GWP – Global warming potential
LCA – Life Cycle Assessment
LCIA – Life Cycle Impact Assessment
LFG – Landfill Gas
MSW – Municipal Solid Waste
PE – Person Equivalent
SGWR - Spoiled Groundwater Resource

Summary

Technologies for waste management are ever-improving, and the number of different ways for treating waste increasing. It is therefore necessary to find ways to assess the most optimal forms of waste treatment. One of the assessment methods that have arisen to help perform this task is life cycle assessment (LCA). A number of LCA models have been developed for the general assessment of products, but increasingly models specially targeted to waste management are being made available. This raises the question of how LCA models can be implemented in solid waste management. During the work for this PhD, three areas regarding the implementation of LCA models in solid waste management were explored:

- What is the current status of LCA model implementation in solid waste management, with a focus on what crucial parts of functionality are needed in the models to ensure for a sound study?
- In what ways can LCA models be used for the development of current and emerging treatment technologies? And what information will this help generate?
- What barriers and issues arise when implementing LCA models in solid waste management?

The aim of the PhD project was to study these areas by applying an LCA model to a number of projects covering one or more of these areas and through this work identify possibilities, barriers and issues.

A good LCA can be made only when the importance of key parameters is well understood. For a successful implementation of the LCA models in solid waste management, these barriers and issues must therefore always be considered; otherwise, there will be the chance of an inconsistent result. One of the key parameters is the setting of the boundaries for the system. There is no right or wrong answer regarding where the boundary is set, but it is crucial to define what is included and what is not. Similarly, it is crucial to define whether the study uses average, marginal or a mix of input data, as this also has an important impact on the result. However, it should also be noted that if direct data is not available, then default average data will, of course, be better than no data whatsoever.

In conclusion, it can be said that taking into account the amount of data, the variability in the data quality and uncertainties with regard to the assessment method, all these factors point to the conclusion that LCAs of waste management systems cannot be an exact science. However, the goal of implementing LCA models in solid waste management is not necessarily to obtain a final single number; rather, we seek to generate an indication of the best choices when considering uncertainties. In addition, and just as importantly, performing such a study with a systematic approach will help teach the user to understand the system or technology better, thereby allowing them to optimise their system and learn what is important and what is not.

Dansk sammenfatning

Teknologier til behandling af affald bliver løbende forbedret, og antallet af forskellige måder at behandle affaldet på er stigende. Det er derfor nødvendigt at finde en metode til at vurdere, hvad der er den mest optimale kombination af teknologier til affaldsbehandling. En af de metoder, der er blevet udviklet til at hjælpe med at udføre denne opgave, er livscyklusvurdering (LCA). Tidligere er en række LCA-modeller blevet udviklet til generel vurdering af specifikke produkter, men i dag bliver der også i stigende grad udviklet LCA-modeller specielt rettet mod affaldshåndtering. Sidstnævnte rejser spørgsmålet om, hvordan LCA-modeller bedst kan blive implementeret i behandlingen af fast affald.

Under arbejdet med denne PhD blev tre områder angående implementeringen af LCA-modellerne i fast affaldshåndtering undersøgt:

- Hvad er den nuværende status mht. implementeringen af LCA-modeller i behandling af fast affald. Fokus lå på hvilke essentielle funktionaliteter, der er nødvendige i modellerne for at sikre en velfunderet undersøgelse.
- På hvilke måder kan LCA-modeller bruges til udvikling af eksisterende og nye behandlingsteknikker indenfor affaldsområdet, og hvilken ny viden kan dette medvirke til at skabe?
- Hvilke barrierer og problemstillinger opstår, i forbindelse med implementeringen af LCA- modeller i håndteringen af fast affald?

Formålet med ph.d.-projektet var således at undersøge disse områder ved at anvende en LCA-model i en række projekter, der dækkede et eller flere af ovennævnte områder og gennem dette arbejde at identificere muligheder, barrierer og problemstillinger i implementeringen af modellerne.

Et godt LCA studie kan kun foretages, når betydningen af nøgleparametre er velforstået. For at få en vellykket implementering af LCA modellerne indenfor fast affald er det nødvendigt, at disse barrierer og problemstillinger altid tages i betragtning, ellers vil der være sandsynlighed for et inkonsekvent resultat. En af de vigtigste parametre er fastsættelsen af grænserne for systemet. Der er intet rigtigt eller forkert sted at sætte grænsen, men det er afgørende at definere, hvad der er inkluderet, og hvad der ikke er. Ligeledes er det afgørende at definere, om

undersøgelsen bruger gennemsnits-, marginale- eller en blanding af gennemsnits- og marginale-data, da dette også har en vigtig indflydelse på resultatet. Det bør dog bemærkes, at hvis direkte data ikke foreligger, vil gennemsnitlige data naturligvis være bedre end ingen data.

Når der tages hensyn til mængden af data, variationen i kvaliteten af data, og usikkerhederne med hensyn til beregningsmetoder, må det konkluderes, at alle disse faktorer peger på, at LCA af affaldshåndteringssystemer ikke er den mest nøjagtige videnskab. Målet med at implementere LCA-modeller i håndteringen af fast affald er dog heller ikke nødvendigvis at opnå et endeligt præcist svar, men i lige så høj grad at frembringe en indikation på, hvad det bedste valg er, når usikkerhederne tages i betragtning. Ydermere og lige så vigtigt er det, at udførelsen af en LCA undersøgelse med en systematisk tilgang vil bevirke, at brugeren får en væsentlig bedre forståelse for, hvad der er vigtigt for systemet eller teknologien, og hvad der ikke er, hvilket dermed giver dem mulighed for at optimere deres systemer eller teknologier yderligere.

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1 Introduction

1.1 Waste Management

The waste hierarchy has governed how waste management should take place in Europe since the mid-eighties, by stating the order of treatment options for the best way to treat waste. The hierarchy is outlined as follows: prevent waste generation if possible; then reuse or recycle the waste; what cannot be recycled should be energy recovered; and, finally, the least favoured option is disposal in landfills. A number of new waste treatment technologies have come into use in the last decade, the arrival of which has begun to contest what can be considered the best treatment option in the waste hierarchy. These new treatment technologies have produced a need for ways to determine optimal treatment systems.

In the European context, this means that policy is currently moving away from the rigid waste hierarchy to a new system where the key phrase is “life cycle thinking”. This is being undertaken to assess whether the waste hierarchy is the best option or a better alternative is available, which opens up opportunities for not only many new treatment options, but also the creation of a way to establish the best system for the treatment of solid waste. The Waste Framework Directive (European Commission, 2008) does not state which assessment method should be used if deviating from the waste hierarchy, but one of the possibilities is Life Cycle Assessment (LCA), which started as an assessment method for products but has, since the early 1990s, begun to be used on waste management as well. Whereas the composition of a product is often well known and not overly complex, waste is a highly heterogeneous material that varies greatly. Furthermore, there is high diversity in the existing waste management systems already in place and the energy systems with which they interchange. This means there is high demand for LCA models that are flexible and suited to handling the complexity of changing compositions. This has therefore led to a number of LCA models targeted directly at modelling waste LCA systems. The models vary considerably both in complexity, user friendliness and the scope for what they are intended, which makes it interesting to study how they can be used within solid waste management.

1.2 Aim of the PhD

The PhD project was initiated because a number of LCA studies were performed where waste LCA models were used as a tool or instrument to achieve the goal of carrying out an assessment. However, the focus was always on the topic of the study and not on the LCA model (tool) itself. It was therefore decided to focus more on how the implementation of the LCA models themselves could be improved. Implementation can be understood in many ways. In a dictionary (Cambridge, 2010; thefreedictionary.com, 2010) its definition is:

"*To put into practical effect; carry out; implement the new procedures*"

This indicates that it is an active process where the model is used with a purpose in mind. This could be to help take a decision (technical or political), to compare different products or technologies or to study a system or process to see if new information or knowledge can be established. During this PhD study, three areas of interest formed as a part of the "implementation" of LCA models in solid waste management. The aim of the PhD study was therefore to explore these three areas.

The first area of interest was to gain an overview of the current status of LCA models being implemented in solid waste management, and as a part of this to determine the most crucial parts of the models' functionality in order to carry out a sound study. Part of this overview found that the models were used widely for the assessment of waste management systems, but less so for treatment technologies. It was therefore decided to focus in the PhD on the assessment of specific technologies and not full systems. The second areas of interest were how the LCA models could be used for the assessment or development of current and emerging treatment technologies, and what type of information this could help generate. Finally, the study looked into what barriers or issues arise with regards to a successful implementation: data quality, substitution of materials, boundaries of studies and the general product LCA methodology fitting into LCA in the solid waste management "world".

1.3 Content of this thesis

The contents of the chapters in this thesis are as follows. Chapter Two gives a short introduction to how waste management systems are viewed in this thesis, and the background for the LCA methodology used. Chapter Three discusses specific waste LCA models, the most important parameters for the models and the main differences. Chapter Four presents an overview of the current status for implementation of LCA in solid waste management based on an article screening. Furthermore, it discusses the use of LCA models in technology development, optimisation and monitoring. Chapter Five gives examples of some of the barriers and issues that arise when using LCA models in solid waste management, and suggests how some of these issues can be solved or worked around. The overall outcome of the thesis, its findings and implications are concluded in Chapter Six, where recommendations and some suggestions for further research are also found.

The research results presented in the PhD thesis are a summary of six scientific papers enclosed in the appendix.

2 Waste management systems and life cycle assessment

2.1 Waste management systems

Waste management systems cover a number of different activities that are grouped into three phases, each of which can have a number of sub-steps. A conceptual representation of this is seen in Figure 1.

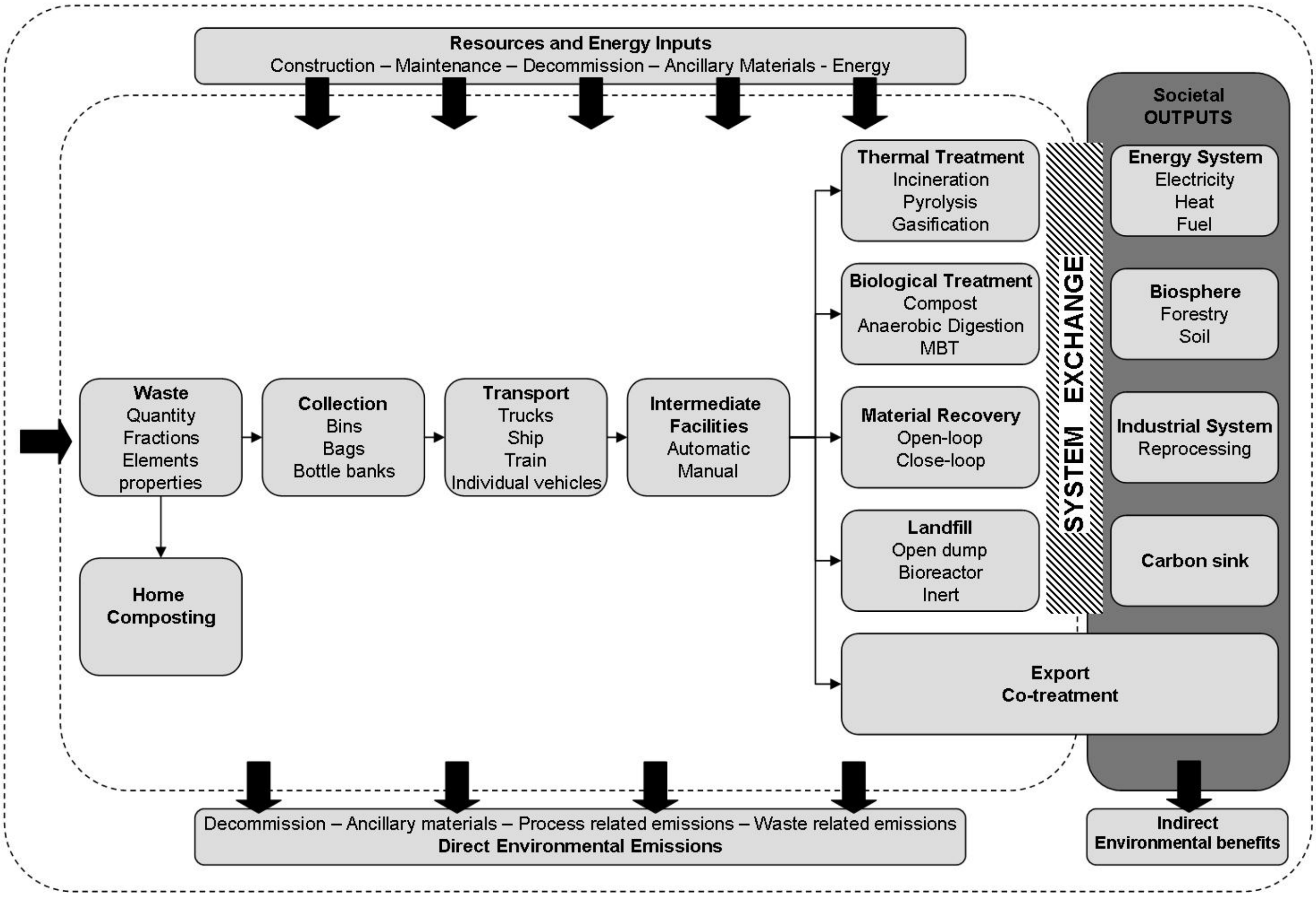

Figure 1 Generic waste management system (Gentil et al., I)

The outer dotted line represents society at large (earth system and technosphere). The inner dotted line represents waste management systems, comprising a number of waste management technologies (light shaded grey). The dark shaded grey represents the inputs and outputs of the whole waste management system. The box indicating the system exchange specifies the relationships of materials and energy flows between the waste industry and wider society through substitution.

The point of origin in a waste management system is always at the site of the waste generation. Here, the waste can either be sorted into a predefined quality by the waste producer or shipped directly for treatment. From the waste producer, collection schemes are set up to handle the collection of the waste and transport it to the treatment facilities. The collected waste will either go first to an intermediary treatment facility (for example a material recovery facility, compost facility, anaerobic digester or transfer station) from where it will go to a final treatment facility (landfill, biological matter used in agriculture, material recycling).

During each step in the system, a number of direct or indirect impacts are taking place. Emissions from the waste treatment process itself (e.g. methane released from a landfill) or from the use of auxiliary materials and energy are released into the environment. Furthermore, if the energy is produced as the product of a treatment process, it might impact the surrounding system by replacing energy that would have been needed to be produced elsewhere. This is what is called "indirect effects". Finally, materials in the waste stream might be recycled and turned into new materials, thereby replacing the need for virgin production of the same material.

2.2 Life cycle Assessment

Models for Life Cycle Assessment (LCA) are, as stated in their name, linked directly to LCA methodology. LCA is an internationally standardised methodology for environmental assessment (ISO, 2006a and 2006b), which is used to evaluate the environmental impact of a product or system. According to the ISO standard, an LCA should cover four distinct phases, as illustrated in Figure 2.

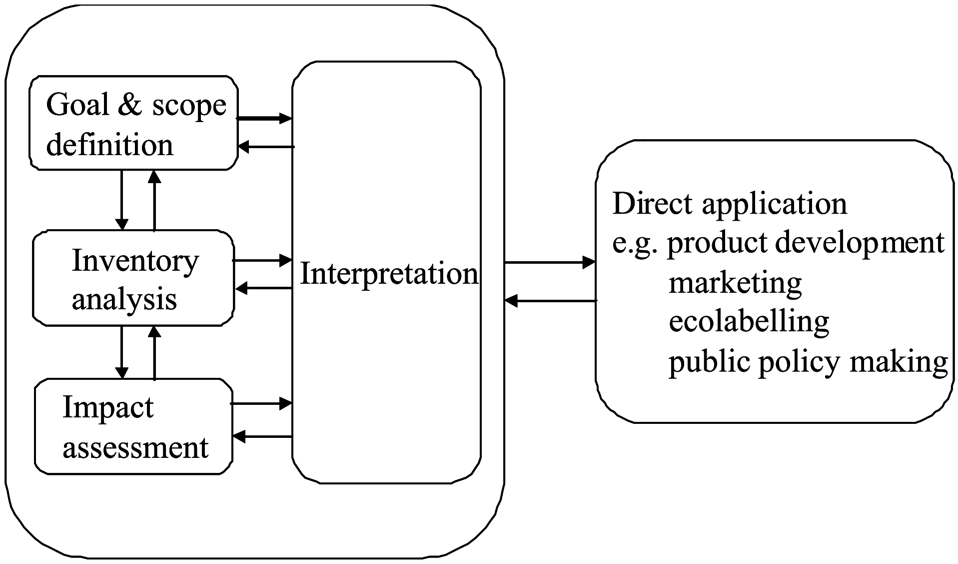

Figure 2 The four phases of an LCA (ISO, 2006a)

2.2.1 Goal and scope definition

In the goal and scope phase, the user should specify the goal and scope of a study as related to an intended application. The functional unit should be specified, which is a way to make sure that a comparison taking place is fair by always relating to the same unit. In the case of waste management, the unit could, for example, be the treatment of a tonne of waste, the treatment of all waste in a year for a studied area, etc. When specifying the functional unit, it is important to specify whether there is any spatial importance to the region or area of the study when applying the impact step. Similarly, is it important to state the time horizon of the study, how long the emissions will be studied for and what happens with the remaining substances after this period. Additionally, it needs to be stated what the boundaries are for the studies, what is included and what is excluded. Traditional product LCAs are normally referred to as "cradle-to-grave" studies, which means that all emissions are accounted for from the extraction of the materials for a product, the production of the product, the use of the product and finally the disposal of the product. In a waste LCA it is often not possible to get information about the life cycle of the product before it ends up in the waste bin, or it is not possible for the waste handler to control this part of the chain. Therefore, is waste LCAs most often bin-to-grave LCAs. This means that only the collection, transport to the treatment facility and the treatment/disposal process itself is accounted for. This signifies that a zero burden approach is applied, which means that any impacts prior to the bin are excluded from the study. This is a valid approach as long as it is stated clearly.

2.2.2 Inventory analysis

The inventory analysis is where all emissions into the environment, energy production and use and resource consumption are tallied. This is one of the main functionalities of the LCA waste models, as this is where they gather all the data required for the third step of the LCA (the impact assessment). The models used for calculating the inventory data are described in detail in Chapter Three.

2.2.3 Impact Assessment

The third phase is the impact assessment. Here, the data from the inventory analysis is applied characterisation factors, which is a way to ask “how much” this impact really is. A number of different methodologies exist which can be used for assessing the impact. In this thesis, the primary assessment method used is the EDIP97 methodology (Wenzel, 1997), which was developed at DTU. The reason for using the EDIP97 methodology was that it is the built-in impact assessment for EASEWASTE, and therefore the obvious choice. The results from the impact assessment can be given either directly in the form of the reference substance for each impact (e.g. kg CO_2 equivalents for global warming) or as a normalised unit in ‘person equivalents’ (PE), which is the amount of that impact given for all the accumulated activities for an average person in one year.

During the study, the choice of methodology and the impact of this choice were studied, which is further deliberated on in Chapter Five.

2.2.4 Interpretation

The fourth and final step is interpretation. This is where the results from the inventory analysis and impact assessment are held up against the goal and scope for the study, so that conclusions and recommendations can be established.

An LCA is an iterative process. Based on the findings of the interpretation and prior steps, it is often necessary to go back and refine parts of the goal and scope, data collection for inventory and the impact assessment. Through a number of iterations, the final recommendation can then be found.

2.2.5 Attributional and consequential LCA

The LCA methodology can be divided into two types: attributional and consequential. An attributional LCA describes a life cycle (for a system or product) with the relevant physical flows, as well as the flows to and from the life cycle and its subsystems. It applies average inventory data that attempts to represent the actual process and the subsystem it uses. A consequential LCA, on the other hand, is most used for describing a system where a decision or a change takes place (Curran et al., 2005; Weidema, 2003), and relies on marginal data for the upstream and downstream processes. “Marginal” here means the process that will always react first to a marginal change in the system. This is especially important when assessing a system with marginal energy sources, as this can have very large impacts on the overall system. In the thesis, mainly average data was used because it was very hard to acquire marginal data for all processes, the only exception being for the energy system where different marginal energy sources were evaluated. This is discussed further in Chapter Five.

From a methodological point of view, the use of this mix of marginal and average data is not completely consistent, but it is still preferable to not being able to evaluate the consequences of a system.

3 Waste LCA models

Of high importance in the implementation of LCA models in solid waste management is the quality and reliability of the available models. Winkler (2004) and Winkler and Bilitewski (2007) were the first to compare LCA models for solid waste management. Their comparison was based on a quantitative assessment, whereby they attempted to run the same scenario in six different models. Their work was important, as it highlighted that there were large discrepancies in the results when performing the LCA, and that different models for some impact categories gave directly opposing results. A closer look into their methodology, however, hinted that only some of the input parameters in the different models had been updated, and the remaining parameters were left at their default values. There was therefore a need to undertake a study of where the models differed, and consequently determine the impact of these differences. The first part led to the article attached in the appendix (Gentil et al., I). As a follow-up to this article, the model developers for six of the models met and made an effort to compare the models. This work is still unpublished, but some of the findings are mentioned in this chapter.

3.1 Differences and similarities in LCA models

LCA on waste has been performed since the early 1990s (Morrisey and Brown, 2004), and there are today more than 50 LCA models available in Europe (EPLCA, 2010). These models have all been developed with different scopes with regards to applicability, functionality, user friendliness and costs. In order to perform a study of these models and their applicability for the LCA of waste, two criteria were set up in Gentil et al. (I) to choose the models for comparison. These criteria were:

- The ability to model the environmental performance of a complete waste management system from waste collection to final disposal, including links between a potentially variable waste composition and emissions into the environment.
- The ability to model process-related emissions (dioxin formation in an incinerator) and waste-related emissions (mercury in the input waste released through the stack).

Based on these criteria, nine models were selected: EASEWASTE, EPIC/CSR, IWM2, LCA-IWM, MSW-DST, ORWARE, SSWMSS, WISARD and WRATE. A number of other models fulfilled the criteria, but were not supported or information was too scarce. As a result, they were not considered (AREA, HOLIWAST, LCA-LAND, MIMES, MSWI, WAMPS). The development time and current status of the models can be seen in Figure 3. The grey area indicates the launch time of the models. The solid line represents the active development phase and launch of subsequent versions of the same model, while the dotted line indicates the research leading to the development phase or the subsequent research not necessarily leading to an active development (use of the model as a research tool).

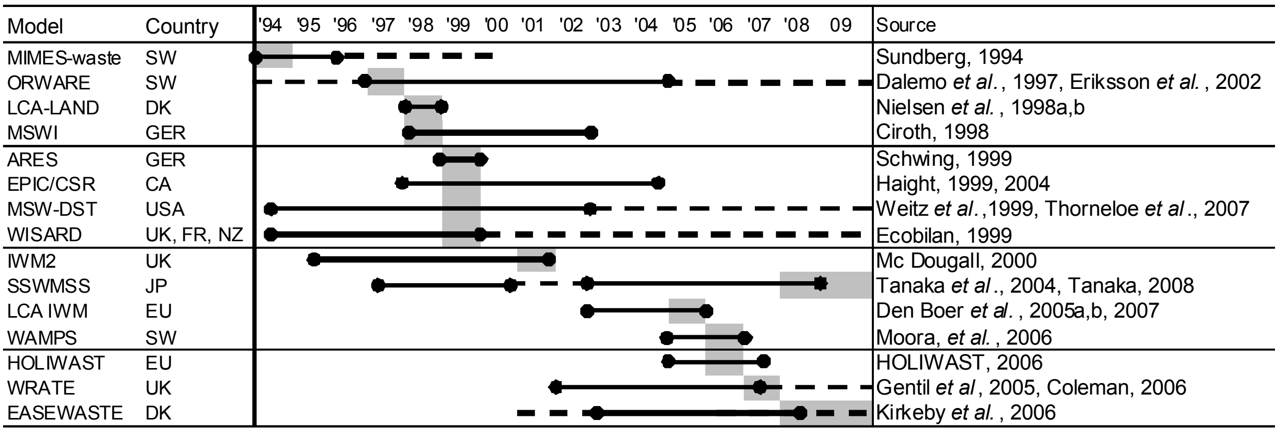

Figure 3 Timeline of selected waste LCA models (Gentil et al., I)

The figure above shows that the models have been developed over a long time, and it is obvious that this is one of the reasons why the models give different results when compared. Furthermore, when studying the model parameters, it was found that they were building on experience from the development of earlier models. An example of this is the EASEWASTE model, which in its landfill modelling was heavily inspired by the MSW-DST model, on which they then could elaborate by expanding the processes being modelled. This shows that when using the models the user begins to gain experience in what is important and what is not, and based on this can make the models more balanced and detailed in areas of importance.

Gentil et al. (I) noted the main areas of difference which would have an impact on the results of an assessment:

- Time horizon of the LCA, finite or infinite. Especially of importance for landfills and the application of biogenic waste to farmland.
- Upstream and downstream system boundaries. Upstream boundaries mean the environmental impact of the production of materials and energy used

in the modelled waste systems, and whether these are included or not. Downstream determines whether recovered materials and energy are used to substitute for virgin material and whether this is included.
- Modelling of carbon (biogenic and fossil) in the system.
- Cut of criteria; what is included and what is not.
- Detail level of the waste composition.
- Detail level of inventory.
- Energy aspect – linked to the type of LCA being performed (consequential or attributional).
- Treatment process modelling. Which processes are included in the models, and how are they technically modelled?

The different areas of importance mentioned cover quite large modelling areas, and it is consequently clear that this must have an impact on the results. These areas are, besides their description for each model in Gentil et al. (I), all discussed in detail from a more general point of view in Chapter Five, as they were found to be of overall importance for the successful implementation of LCA models in waste management. The reason for the large differences lies in the fact that the models have been developed in different countries and with different scopes in mind. There is no right or wrong method; rather, it is more a matter of knowing what type of modelling the model is suited for and – just as importantly – what it is not suited for.

3.2 Model comparison workshop

Based on the review article, it was also found that there was a need for a more detailed look at how the models actually performed when compared. Here, it was found to be extremely important to include the developers of the individual models, as they would be able to use the models as intended and know where to change the input data to compare comparables. The findings of the workshop are currently only available as an internal memo (Damgaard et al., 2010), but some are discussed in the following subsection. It was not possible to find developers for all the models covered in the review, so the models finally compared were therefore:

- EASEWASTE
- MSW-DST
- ORWARE

- SSWMSS
- WRATE
- WISARD (only participated in the initial screening and not the workshop)

In the preparation phase before the workshop, it was quickly realised that the models covered so many technologies and were so complex that not all could be covered in one sitting. It was therefore decided to focus on four areas, namely:

- Transport and collection
- Landfilling
- Material recycling
- Incineration

Each of these four areas was then defined with default input data to be used in all the models where possible, and inventory results were calculated prior to the workshop. At the workshop the initial calculated data was then compared, and based on this any differences were identified and a second calculation performed for better understanding.

3.2.1 Transport and collection

Since the initial step of a waste management system is the collection and transportation of the waste, it was an obvious starting point. The information available in the different models, as seen in Figure 4, shows that there was a large variance between the coverage of the individual models. It was therefore decided to compare the models based on fuel consumption and direct and indirect emissions.

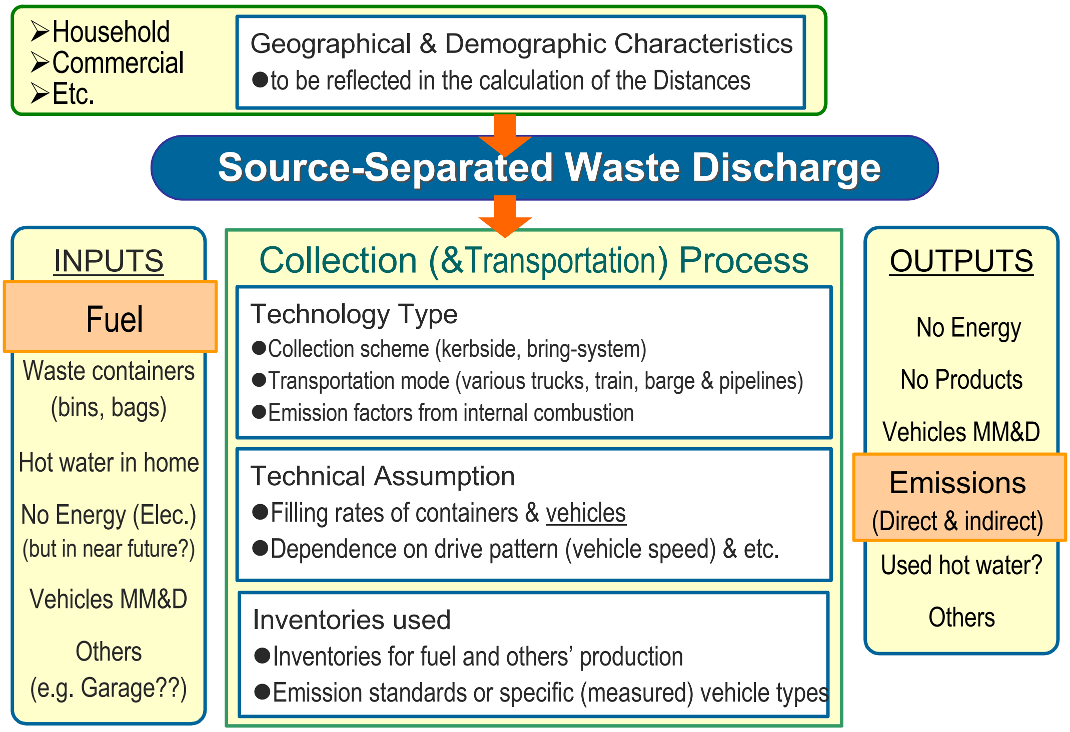

Figure 4 Aspects to be reflected in the comparison for collection and transport. Different areas included in some or all of the compared models (Damgaard et al., 2010).

The models can be split into "simple" (EASEWASTE, WRATE and WISARD) and "mechanistic" (MSW-DST, ORWARE, SSWMSS) approaches. The simple models use key aggregated values for travel distance as well as fuel consumption per distance driven, defined by the user. The mechanistic models use a more detailed method, whereby they look at distance and time between every stop to pick up waste, idling time of the engine at the stop, consumption of different driving patterns. All these parameters give fuel consumption like that found in the simple models. The initial comparison, therefore, showed large differences between the models, the reason being that the inputs from model to model were so different that they were hard to accommodate for beforehand. Nevertheless, by agreeing on a set of default inputs, fitting the mechanistic models and calculating the inputs fitting the more simple models based on this, the final result was almost identical. This shows that for something as simple as the collation and transport of waste, where the only contributing parameter is the combustion of fuel, it can still be hard to get the same results. Nonetheless, when delving into the detail, it is apparent that the underlying formula in the different models gives the same result when understanding these differences.

3.2.2 Landfilling

From the review article, landfilling was expected to provide large differences, as the scope of the models was very different, and it was also found to be true. The main reason for the large differences is that for landfill modelling a large number of parameters are combined to calculate the different emissions. Additionally, a number of the input parameters were locked in a couple of the models, so it was not possible to make the input data totally uniform. This meant that the output varied with a factor 2-3 between the different models. For models where the input parameters could be changed and made directly comparable, the difference was as low as 5 % (EASEWASTE and MSW-DST). Based on the model developers' knowledge of formulas and input data in their models, it was possible to explain the differences between the other models. This gave an understanding that even though the output was different, the underlying calculations were sound. The results of the models are credible and can therefore be used independently, but results from studies with different models should not be compared directly, as the scope of the models will not lead to the same results.

3.2.3 Incineration

Similar to landfilling, the incineration of waste was expected to offer differences in results. To keep the comparison possible, it was decided to focus on a couple of indicator emissions (NO_X and lead) as well as energy produced. The energy calculation was found to give almost identical values (variance of 7%) except for the MSW-DST model, which produced values varying by 12%. The reason for this was that MSW-DST uses the higher heating value in the calculation, and the input therefore had to be adjusted to accommodate for this factor. For NO_X and lead, the variances in calculated emissions were higher, as most of the models used default values. This meant that differences were expected. A critical difference was noted in that some models used process-specific emissions for heavy metal emissions (amount released per tonne of waste treated), whereas others used transfer coefficients (amount released per amount of heavy metal in input waste and transferred to the recipient). This shows that it is vitally important to understand how the process is modelled in order to evaluate the results.

3.2.4 Material recycling

To compare material recycling in the models we looked at the recycling of paper. It was found that all the models, except ORWARE, used the same modelling approach, and therefore obtained identical values when using the same input data, because the models all use a static multiplication of the amount of paper waste recycled, multiplied by an inventory for that process and then subtracting an identical amount of virgin paper that would otherwise have been produced. ORWARE was the only model that as an alternative used a process-based approach incorporating the composition of the waste, which meant that it was not possible to use the same default data in ORWARE, resulting in large differences. The default paper recycling processes in the models were also compared, showing that it is extremely important to use good, relevant external processes, as large differences were observed. This is a general finding for all LCA models, as the results will always reflect the input data.

3.2.5 General comments

At first glance, all of the models seem comparable. However, following a detailed study it is evident that the models differ widely. The workshop found that the models were calculating correctly, but that it is very important to have an understanding of what the model is doing, in order to gain the same result. This is because different models include a number of assumptions for each technology, which ultimately plays an essential role when calculating the life cycle inventory. For some models these assumptions can be changed, but for others it is fixed. It is therefore important to understand the nature of these parameters for a good implementation of the LCA model in a waste management study, so that the results can be interpreted correctly. Consequently, the most important issues to be aware of are covered in Chapter Five.

4 How has LCA been implemented in solid waste management?

4.1 Review of the current status of implementation

This section will describe the current status of implementation from a research point of view (articles), as well as give suggestions for new ways in which the models can be used. In order to get an overview of how the LCA models are used currently, a screening was performed, which was effected by searching the article database of the Technical Information Center of Denmark..

A search was undertaken for the keywords “LCA” and “waste” combined with one of the following: “management”, “historic”, “monitoring” and “model”. This resulted in 672 articles fitting these terms. Some articles were double hits, meaning the same article matched a number of the keyword combinations and the true amount of articles screened was lower.

A number of search criteria were then set up to narrow down the articles of interest in the following screening. The criteria were:

- Must be about municipal solid waste (not hazardous, industrial, etc.)
- The study must include an impact assessment.

Papers describing LCA models and modelling, but without an actual modelling included, were left out (some of these models are discussed in Gentil (I)). A large number of papers discussed the methodological importance of waste modelling aspects, but did not give concrete examples. These are, of course, important to acknowledge and study, but it is only when this knowledge has been used in a real study that it can be said to have been implemented. The same is true with inventory studies, a good example of which is a study by Obersteiner et al. (2007) on landfill modelling, where the authors collected the necessary data to undertake an in-depth LCA of a landfill, but had to stop at the inventory stage. It should be noted that it is acknowledged that studies in reports and other non-article formats are not included in this screening, and that although reports most likely corresponded to the vast majority of applications, the screening was only performed to gain an idea about a trend and is therefore found to be valid for purpose.

Articles fulfilling all of the above criteria were sorted into five study types:

- **Case studies**: A study describing a real case.
- **Technology comparison**: Articles that focus on comparing the impacts of different waste treatment technologies. They ensure the functional unit is the same for the input entering each treatment technology.
- **Strategic/Planning studies**: These articles focus on ways to use the modelled output for strategic decisions.
- **Technology development**: Articles that focus on what the development and improvement of a treatment technology has of improvements on the environmental impact.
- **Monitoring**: Studies that use LCA as a monitoring tool for the impact of a technology.

The tables below include the author, year and name of article, which is how they can be located in the reference list in this study. Furthermore is listed what the LCA software used were (where this was stated) and finally it is tallied how many impact categories that was covered in the article. The covered impacts only include LCA impacts, meaning costs were excluded, energy parameters when used as an impact were though included. Furthermore, SO_2 and CO_2 – indicators of acidification and global warming potential respectively – were counted as an impact in articles when used for evaluation.

Cases studies

Year	Author	Title	Impacts covered	Software used
2002	Solano et al.	Lifecycle-based Solid Waste Management. II: Illustrative Applications	5	MSW-DST
2003	Arena et al.	The environmental performance of alternative solid waste management options: a life cycle assessment study	4	Special for study
2003	Rodriguez-Iglesias et al.	Life cycle analysis of municipal solid waste management possibilities in Asturias, Spain	6	IWM
2004	Beigl and Salhofer	Comparison of ecological effects and costs of communal waste management systems	3	IWM
2004	Di Maria and Fantozzi	Life cycle assessment of waste to energy micro-pyrolysis system: Case study for an Italian town	11	SimaPro
2004	Mendes et al.	Comparison of the environmental impact of incineration and landfilling in São Paulo City as determined by LCA	3	Not stated
2005	Eriksson et al.	Municipal solid waste management from a systems perspective	6	ORWARE
2005	Lundie and Peters	Life cycle assessment of food waste management options	8	GaBi
2005	Moris	Comparative LCAs for Curbside Recycling Versus Either Landfilling or Incineration with Energy Recovery	6	MSW-DST
2006	Aye and Widjaya	Environmental and economic analyses of waste disposal options for traditional markets in Indonesia	4	SimaPro
2006	Hong et al.	Life cycle assessment of BMT-based integrated municipal solid waste management: Case study in Pudong, China	3	Not stated
2006	Kirkeby et al.	Evaluation of environmental impacts from municipal solid waste management in the municipality of Aarhus, Denmark (EASEWASTE)	11	EASEWASTE
2006	Özeler et al.	Life cycle assessment of municipal solid waste management methods: Ankara case study	5	IWM
2006	Güereca et al.	Life cycle assessment of two bio waste management systems for Barcelona, Spain	12	IWM
2007	Chaya and Gheewala	Life cycle assessment of MSW-to-energy schemes in Thailand	8	SimaPro
2007	Buttol et al.	LCA of integrated MSW management systems: Case study of the Bologna District	8	WISARD
2007	Emery et al.	Environmental and economic modelling: A case study of municipal solid waste management scenarios in Wales	5	WISARD
2008	Blengini	Applying LCA to organic waste Management in Piedmont, Italy	6	SimaPro

Cases studies continued

2008	Contreras	Application of analytical hierarchy process to analyse stakeholders preferences for municipal solid waste management plans, Boston, USA	3	Special for study
2008	Giugliano et al.	Energy recovery from municipal waste: A case study for a middle-sized Italian district	5	Not stated
2008	Liamsanguan and Gheewala	LCA: A decision support tool for environmental assessment of MSW management systems	2	Not stated
2008	Liamsanguan and Gheewala	The holistic impact of integrated solid waste management on greenhouse gas emissions in Phuket	1	Not stated
2008	Wada et al.	Evaluating waste disposal systems	3	Special for study
2008	Morselli et al.	Environmental impacts of waste incineration in a regional system (Emilia Romagna, Italy) evaluated from a lifec ycle perspective	11	SimaPro
2009	Niskanen et al.	Environmental assessment of Ämmässuo Landfill (Finland) by means of LCA-modelling (EASEWASTE)	10	EASEWASTE
2009	Zhao et al.	Life-cycle assessment of the municipal solid waste management system in Hangzhou, China (EASEWASTE)	5	EASEWASTE
2009	Banar et al.	Life cycle assessment of solid waste management options for Eskisehir, Turkey	6	SimaPro
2009	Chrubini et al.	Life cycle assessment (LCA) of waste management strategies: Landfilling, sorting plant and incineration	4	Not stated
2009	De Feo and Malvano	The use of LCA in selecting the best MSW management system	11	WISARD
2009	Tarantini et al.	Life cycle Assessment of waste management systems in Italian industrial areas: Case study of 1st Macrolotto of Prato	9	GaBi
2009	Wada et al.	Evaluating household waste treatment systems with specific examination of collection and transportation processes	1	Special for study
2009	Wittmaier et al.	Possibilities and limitations of life cycle assessment (LCA) in the development of waste utilisation systems – Applied examples for a region in Northern Germany	1	GaBi
2009	Zhao et al.	Life cycle assessment of municipal solid waste management with regard to greenhouse gas emissions: Case study of Tianjin, China	1	Not stated
2010	Khoo et al.	Food waste conversion options in Singapore: Environmental impacts based on an LCA perspective	5	Not stated

Technology Comparison

Year	Author	Title	Impacts covered	Software used
2003	Barlaz et al.	Evaluating environmental impacts of solid waste management alternatives	2	MSW-DST
2005	Finnveden et al.	Life cycle assessment of energy from solid waste—part 1: general methodology and results	4	SimaPro
2005	Moberg et al.	Life cycle assessment of energy from solid waste—part 2: landfilling compared to other treatment methods	10	SimaPro
2007	Eriksson et al.	Life cycle assessment of fuels for district heating: A comparison of waste incineration, biomass- and natural gas combustion	3	SimaPro
2008	Merrild et al.	Life cycle assessment of waste paper management: The importance of technology data and system boundaries in assessing recycling and incineration	1	EASEWASTE
2009	Kaplan et al.	Is It Better To Burn or Bury Waste for Clean Electricity Generation?	4	MSW-DST
2009	Khoo	Life cycle impact assessment of various waste conversion technologies	4	GaBi

Strategic studies

Year	Author	Title	Impacts covered	Software used
2005 a,b	Consonni et al.	Alternative strategies for energy recovery from municipal solid waste: Part A: Mass and energy balances Part B: Emission and cost estimates	6	Not stated
2007	Björklund and Finnveden	Life cycle assessment of a national policy proposal – The case of a Swedish waste incineration tax	11	SimaPro
2009	Iriarte et al.	LCA of selective waste collection systems in dense urban areas	11	SimaPro
2009	Larsen et al.	Diesel consumption in waste collection and transport and its environmental significance	4	EASEWASTE
2009	Rigamonti et al.	Influence of assumptions about selection and recycling efficiencies on the LCA of integrated waste management systems	5	SimaPro
2010	Su et al.	Su – Applying multi-criteria decision-making to improve the waste reduction policy in Taiwan	11	Special for study

Technology Development

Year	Author	Title	Impacts covered	Software used
2009	Manfredi and Christensen	Environmental assessment of Solid waste landfilling technologies by means of LCA – modeling (EASEWASTE)	11	EASEWASTE
2009	Scipioni et al.	LCA to choose among alternative design solutions: The case study of a new Italian incineration line	11	SimaPro
2010	Damgaard et al.	LCA and economic evaluation of landfill leachate and gas technologies	11	EASEWASTE
2010	Manfredi et al.	Environmental assessment of low-organic waste landfill scenarios by means of life cycle assessment modeling (EASEWASTE)	11	EASEWASTE

Monitoring

Year	Author	Title	Impacts covered	Software used
2005	Morselli et al.	Tools for evaluation of impact associated with MSW incineration: LCA and integrated environmental monitoring system.	2	TEAM
2007	Rimaityté et al.	Report Environmental assessment of Darmstadt (Germany) municipal waste incineration plant	3	LCA-IWM
2008	Riber et al.	Environmental assessment of waste incineration in a life-cycle perspective (EASEWASTE)	9	EASEWASTE
2010	Damgaard et al.	Life-cycle assessment of the historical development of air pollution control and energy recovery in waste incineration Waste Management	9	EASEWASTE

In total, the listed articles covered:

- Case studies – 34 articles
- Technology comparison – 7 articles
- Strategic/Planning studies – 6 articles
- Technology development – 4 articles
- Monitoring – 4 articles

The screening shows that the majority of the articles are in the form of case studies in which the LCA model has been used to enable this study. This was also expected, as LCA is a decision support tool, and therefore used in case studies where a decision needs to be taken between a number of different scenarios. Furthermore, it was found that there was also a good use of the models for comparing between different technologies, which is also how LCA is often used in traditional product assessments. Articles where it had been used in the analysis of the impact of strategic decisions, such as the introduction of environmental taxes on carbon emissions or in waste minimisation, was quite low, but with the introduction of “life cycle thinking” in the waste directive (European Commission, 2008) in 2008, it is expected that it will be used more for this type of analysis in years to come.

The last two types of studies, technology development and monitoring, represented the smallest segment, and it should be noted that of the eight articles in total, five were made with EASEWASTE, two of which are from this PhD. The following subsections discuss why it is relevant to begin to use LCA models for these types of studies.

Overall, there did not seem to be a trend in the amount of impacts covered in any study. In a number of the articles it was explained why the impacts were chosen, whereas other articles neglected to mention this. The technology development and monitoring studies covered most impacts, which makes sense, because these studies are not site-dependent in scope and should therefore have the overall impact in mind. A case study could, on the other hand, have a local concern in mind and therefore be focusing on a couple of specific impacts, for instance the risk of leaching from a landfill to nearby fragile wetlands, or dioxin formation in the vicinity of densely populated areas.

The models used in the articles varied widely. A number of the articles were based on specific waste LCA models such as MSW-DST, WISARD and EASEWASTE, as discussed in Chapter Three; however a large number of the studies were based on general LCA models such as GaBi and SimaPro. In this thesis, the quality of the implementation of the LCA models used in the articles has not been evaluated; rather, this study instead discusses general problems with the implementation of LCA models, which can be found in Chapter Five. Some of these problems might be related to the model used.

4.2 Use of LCA models for technology development

Most of the studies from the screening use LCA models for the specific analysis of a case area. However, another application could be for the development or optimisation of waste treatment technologies. As discussed in Chapter Three, large variations in an impact from a treatment technology can occur, depending on the model used and the technology therein. Another influence on the impact of the technology can be the age of data and spatial variations in the data.

4.2.1 Development of landfill technologies

In Damgaard et al. (III), the impacts of different landfill gas and leachate mitigation technologies were explored, which can be seen as a way to model technology development. This was done by modelling the development of conventional landfill technologies from the simple open dump to the most efficient collection and treatment of leachate and gas, where the gas was used for energy production. This produced an overview of what the different technology parameters meant for the environmental performance of each technology.

In order to facilitate the comparison, the functional unit of the study was set to the landfilling of 1 tonne of Municipal Solid Waste (MSW), which would be put in to a landfill with an average depth of 12.5 metres and at a density of 800 kg /m^3. The landfill was modelled on a 100-year period where all emissions are accounted for, while the methane generation was set to be 77 m^3 CH_4 per tonne of wet waste, corresponding approximately to 160 m^3 landfill gas (LFG) per tonne of wet waste. All of these values are, of course, variable and in the case of a real-life study determined by site-specific parameters, where the gas generated is determined by the composition of the waste. The modelling was done in EASEWASTE in the landfill module, which is described in Kirkeby et al. (2006, 2007). The LCA model will here allow simulations of what different aspects of the landfill mitigation technologies mean for the overall impact of the landfill, and can help direct focus on areas where it is most important to improve the technology. In EASEWASTE, gas and landfill parameters and emissions are defined in four time segments. The length of the segments in EASEWASTE can be defined independently for each measure. In the study these were combined to define the various conventional landfills, representing different levels of environmental protection. The key parameter values are presented in Table 1. Based on these parameters, more detailed information for the composition of leachate and gas can be added as an amount of substance per unit of gas or leachate in each time unit. This is then linked up to treatment (removal) efficiencies for the leachate treatment plants and treatment for the gas (top cover oxidation, biofilters, flaring or combustion for energy production).

Table 1 Key parameters describing the defined conventional landfill technologies in terms of measures for leachate and gas control. For each cell per period, the number of years and the amount per period or year is defined (from Damgaard et al., III).

	Time period 1	Time period 2	Time period 3	Time period 4
The dump (L1, G1)				
Gas generated (% of gas potential)	2y: 2%	3y: 8%	35y: 70%	60y: 16%
Gas collected (% of generated)	None	None	None	None
Gas oxidised by top cover (% of uncollected)	None	None	None	None
Leachate generated (mm/y)	2y: 500	8y: 500	40y: 450	50y: 450
Leachate collected (% of generated)	None	None	None	None
Leachate entering groundwater (% of generated)	2y: 100%	8y: 100%	40y: 100%	50y: 100%
The covered dump (L1, G2)				
Gas generated (% of gas potential)	2y: 2%	3y: 8%	35y: 70%	60y: 16%
Gas collected (% of generated)	None	None	None	None
Gas oxidised by top cover (% of uncollected)	2y: 0%	3y: 35%	35y: 35%	60y: 80%
Leachate generated (mm/y)	2y: 500 mm/y	8y. 250 mm/y	30y: 200 mm/y	60y:180 mm/y
Leachate collected (% of generated)	None	None	None	None
Leachate entering groundwater (% of generated)	2y: 100%	8y: 100%	40y: 100%	50y: 100%
The simple conventional landfill (L2 and, G2, G3A or G3B)				
Gas generated (% of gas potential)	2y: 2%	3y: 8%	35y: 70%	60y: 16%
Gas collected (% of generated)	2y: 0%	3y: 75%	35y: 75%	60y: 0%
Gas management	None	Flared/filter	Flare/filter	None
Gas oxidised by top cover (% of uncollected)	2y: 0%	3y: 80%	35y: 80%	60y: 70%
Leachate generated (mm/y)	2y: 500 mm/y	8y. 250 mm/y	30y: 200 mm/y	60y:180 mm/y
Leachate collected (% of generated)	20y: 95%	20y: 80%	30y: 60%	30y: 0%
Leachate entering groundwater (% of generated)	20y: 5%	20y: 20%	30y: 40%	30y: 100%
The energy recovery conventional landfill (L2, G4)				
Gas generated (% of gas potential)	2y: 2%	3y: 8%	35y: 70%	60y: 16%
Gas collected (% of generated)	2y: 0%	3y: 75%	35y: 75%	60y: 0%
Gas management	None	Flared	Elec/CHP	None
Gas oxidised by top cover (% of uncollected)	2y: 0%	3y: 80%	35y: 80%	60y: 80%
Leachate generated (mm/y)	2y: 500 mm/y	8y. 250 mm/y	30y: 200 mm/y	60y:180 mm/y
Leachate collected (% of generated)	20y: 95%	20y: 80%	30y: 60%	30y: 0%
Leachate entering groundwater (% of generated)	20y: 5%	20y: 20%	30y: 40%	30y: 100%

On the basis of this data, it was then possible to run the life cycle impact assessment for each of the defined technologies, the results of which can be seen in Figure 5.

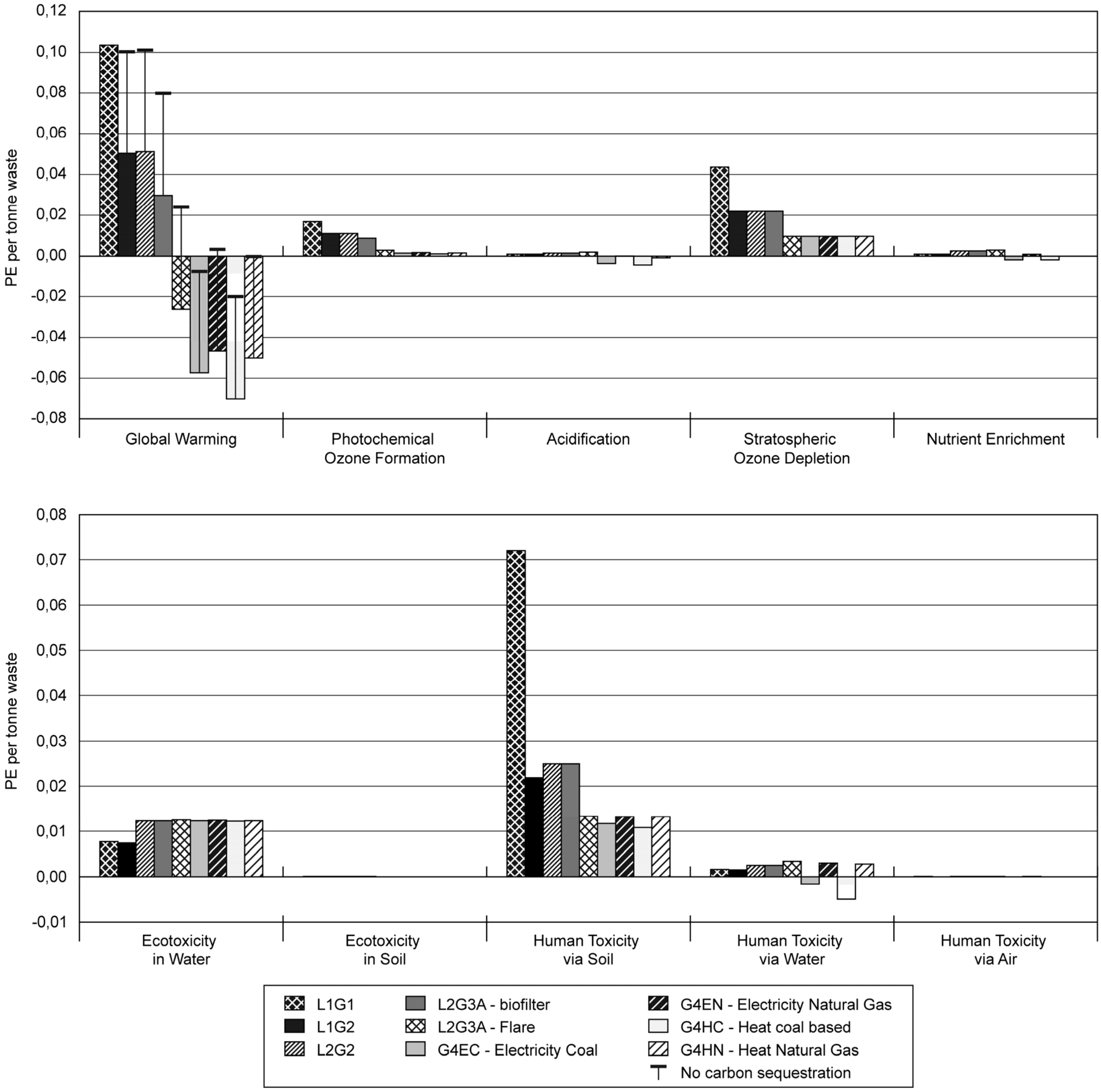

Figure 5 Impacts for the nine landfills showing standard and toxic impacts. The results are given in person equivalent (PE) per tonne of waste (from Damgaard et al., III).

Based on the detailed data behind a figure like this, it is possible to see what the optimal technology to use is based on the different impact categories. The detailed table of substances for each impact furthermore makes it is possible to locate the substances that contribute the most to each impact. In this way, it is possible for a treatment facility to optimise a technology in order to minimise a critical impact based on the LCA results. In the case of landfills, it is generally accepted that the main issue causing the global warming impact is methane, and it is therefore very clear that it is advantageous to work on lowering this emission. Nevertheless, it is not always easy to decipher the best treatment method for the other impacts. Here, a model setup like above can be used to identify the optimal treatment point.

This was recently done in a project by Møller et al. (2010), where they set up a model of a waste incinerator for a waste incineration company, Vestforbrænding I/S. The purpose of their study was to determine the benefit of increased NO_X removal, and if there was an optimal treatment point. NO_X in the incinerator was removed from the flue gas with the use of a Selective Non-Catalytic Reduction (SCNR) system, which removes NO_X by the addition of ammonia, which causes a reaction and converts the NO_X to N_2 and H_2O. The reason for removing NO_X is that it contributes to nutrient enrichment and acidification. The problem Vestforbrænding I/S wanted to study was that they knew there were occasional problems with un-reacted ammonia. This ammonia would therefore end up in either the waste water treatment system or the fly ash. From either of these points, there was a risk that the ammonia would evaporate, and as ammonia itself contributes to these two impacts there would be a trade-off between the NO_X and ammonia. The unknown factor was how large a percentage of the un-reacted ammonia would evaporate. Figure 6 shows a graph where the potential impact of nutrient enrichment is plotted as a function of the amount of ammonia used for the treatment of a functional unit of 1 GNm3 (10^9 m^3 at standard conditions) of flue gas. It is here assumed that 10% of the ammonia in the wastewater will always evaporate, but the amount of evaporation from the fly ash is unknown. It is evident that even when there is no fly ash evaporation, there is still an optimal point of ammonia usage, after which the impact from more use will actually increase.

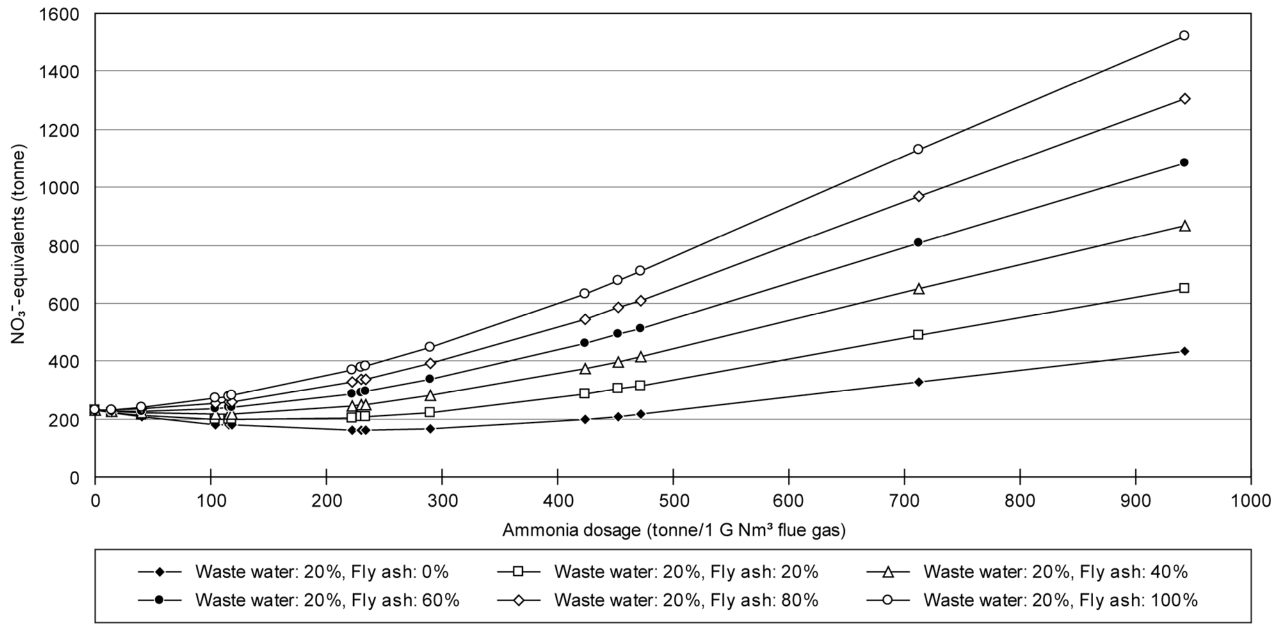

Figure 6 Potential nutrient enrichment as a function of ammonium hydroxide dosage for NOx-cleaning. Assumed realistic case with 10 % of the ammonia slip to wastewater ending up in the environment. Between 0 and 100 % of the ammonia slip to fly ash was assumed to end up in the environment as well (Møller et al., 2010).

These types of studies show that LCA models are not only used for the modelling of full systems to find an optimal system, but also they can just as importantly be used for the development or optimisation of individual treatment processes. Depending on the model used, it is, of course, easier or harder to set up the models to accommodate these types of studies.

4.3 Use of LCA models for monitoring and historical assessments

Another way to use the LCA models is to monitor a development over time. This could involve establishing how a system changes from year to year, but it could also be to monitor the performance of a technology. This can help remove myths about how good or bad a technology might be. A part of the interest was spurred by the fact that waste incineration is used widely in north European countries, but there is a lot of scepticism about waste incineration in other parts of the world. This made it interesting to assess the historical development of the impacts from waste incineration, in the form of the release of air emissions, and see if this would produce any new knowledge.

Damgaard et al. (II) examine this by monitoring the adoption over time of different air-pollution-control (APC) treatment technologies in waste incineration. The modelling looks at early burning from the start of the last century up to today's most efficient treatment technologies. Table 2 shows an overview of the emissions released through the stack after the APC is held up against EU threshold values. The functional unit is set at the combustion of 1 tonne of MSW.

Table 2 Air emissions estimated for municipal waste incineration with an increasing degree of flue gas cleaning. EU WID limit values included for comparison based on European Parliament (2000). Abridged table from Damgaard et al. (II).

Flue gas cleaning technology	EU[1] WID	APC 1	APC 2	APC 3	APC 4	APC 5	APC 6	APC 7	APC 8
Technical configuration									
Particle removal	-	No	Yes	Yes	Yes	Yes	Yes	Yes	Yes
Scrubbing [2]	-	No	No	Dry	Wet	Dry	Wet	Dry	Wet
Dioxin filter	-	No	No	No	No	Yes	Yes	Yes	Yes
Flue gas condensation	-	No	No	No	No	No	No	Yes	Yes
$deNO_x$ – technology	-	No	No	No	No	SNCR	SNCR	SCR	SCR
Air emissions									
(g per tonne of waste)									
SO_2	273	1,100	1,100	270	870	164	109	55	27
HCl	55	3,800	3,800	110	27	27	11	5	3
NOx	2730	2,200	2,200	2,200	2,200	900	900	55	55
NH_3	-	3	3	3	0	40	1	16	16
Particles	164	8,200	400	55	55	11	11	5	5
Hg	0.28	0.82	0.82	0.11	0.11	0.005	0.05	0.003	0.003
Pb	2.7[1]	82	5	1.1	1.1	0.5	0.5	0.11	0.11
Cd	0.27	5	1.1	0.16	0.16	0.11	0.05	0.01	0.01
As	2.7[1]	3	0.5	0.16	0.16	0.11	0.05	0.01	0.01
CO_2 fossil [3] (kg/tonne)	-	300	300	300	300	300	300	300	300
Dioxin [4] (μg/tonne)	0.55	16	16	3	11	0.3	0.3	0.11	0.11

[1]: The EU WID limit values have been converted based on an assumption of 5460 m3 flue gas/tonne waste. Pb and As are the value for the combined amount of Sb+As+Pb+Cr+Co+Cu+Mn+Ni+V

[2]: Acid gas absorption is for illustration assumed to be hydrated lime and limestone for semi-dry and wet systems, respectively. [3] After Astrup, 2009.

[4]: TEQ (toxicity equivalents), international, cf. Directive 2000/76/EC of the European Parliament and of the Council of 4 December 2000 on the Incineration of Waste.

Based on the emissions themselves, it is evident that there has been a very large improvement in APC technologies inasmuch that all emissions have been reduced significantly; most of the emissions are a factor 10 or more lower than the EU threshold values. Dioxin is the only emission that is not as far below, but all dioxin emissions from APCs 5-8 are still well below the threshold. The older

APC 1-4 technologies would not be constructed today, but nonetheless give a good overview of the historical development. The normalised environmental impacts seen in Figure 7 show that for all categories there has been a drastic improvement in the performance of waste incineration. The only category not following this pattern is global warming, which is growing due to the growing demand for electricity consumption in the APC equipment. In a real case this energy consumption would have been offset by generated energy, which is discussed further in Damgaard et al. (II).

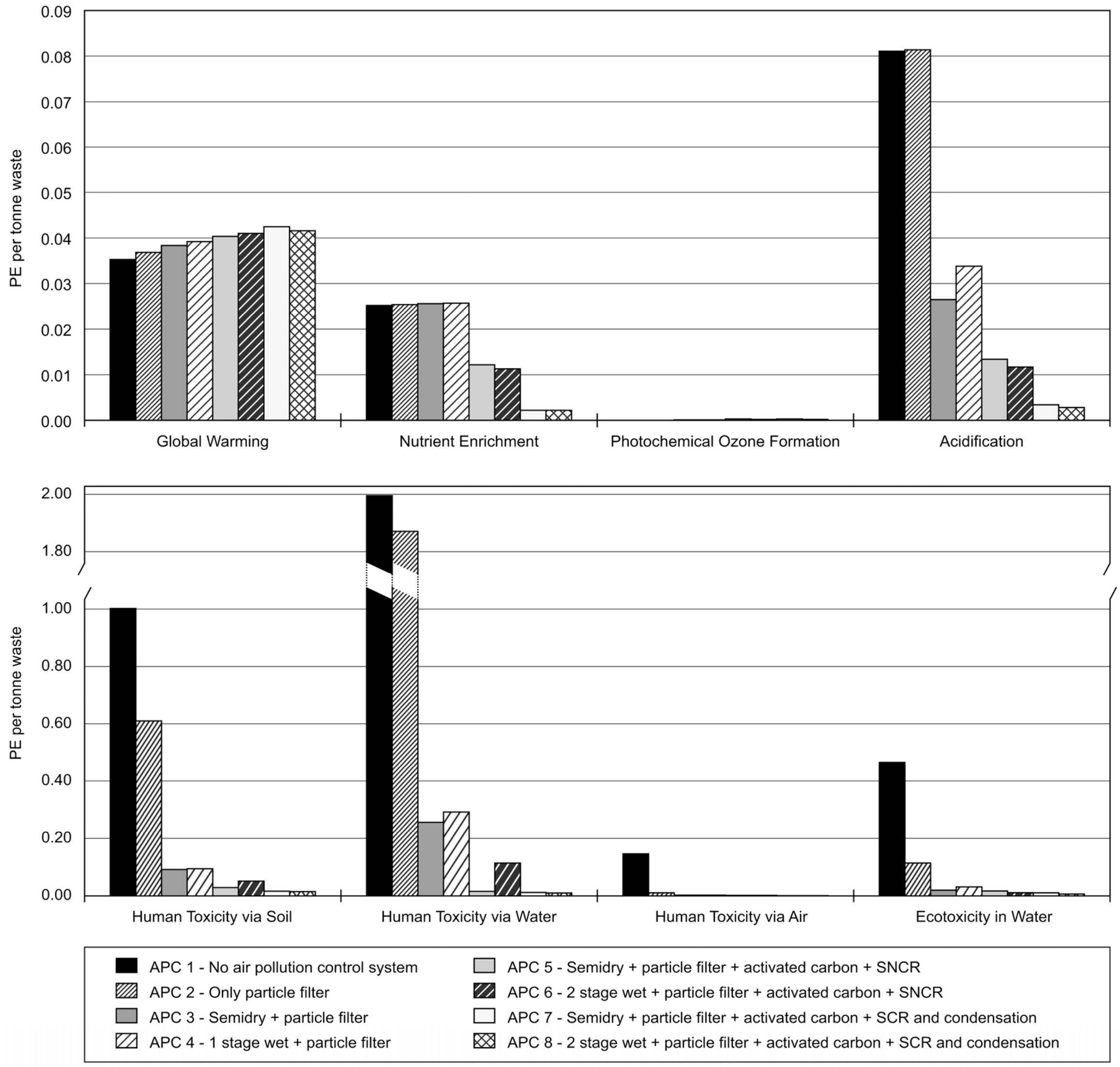

Figure 7 The normalised environmental impacts for the eight air pollution control technologies (APC1 – APC8) given in Person Equivalent (PE). From Damgaard et al. (II)

The study emphasises the importance of using the best available technologies when performing the LCAs of waste-to-energy in future waste management systems. It is not sufficient to use whichever data is available in general LCA databases, since data older than 10 years may give a very misleading picture of waste incineration. Naturally, this is also the case for other waste disposal options where there has been a large development in emission control and efficiency improvements. This way of using LCA models can help remove some of the myths that still exist about a number of treatment technologies, which are often based on old and outdated plants, and when modelling the actual plant as it looks today, they might turn out to be much better.

Similarly, this type of modelling can be used for companies that carry out annual environmental accounting of their emissions. By moving from the usual inventory reporting they could add more information by going to a mid- or end-point analysis and putting a qualitative measure on what really is important. This information would be of interest to the companies, as they would know where they need to focus if they wish to continuously improve the performance of their plant. The industry might have a very good idea on the cause of the emissions and how to abate them, but they might not have as good an idea about what the really critical emissions are. This is where an LCA model can help, as it can take monitored data and bring it to another level of understanding through qualitative characterisation.

The information could similarly be used by a company as a way to communicate to stakeholders that what they do is sustainable. The knowledge from an LCA study might be obvious for people working with these types of emissions every day, but giving it in a more comprehensive form might help convince the layman that a decision is sound. Telling a citizen that a waste incinerator only releases 0.11 μg dioxin per tonne of combusted waste will mean nothing, but explaining that this amount is equivalent to 10^{-8} of the annual impact to human toxicity from an average person will give them a measure which can help them understand that the dioxin is not a problem.

5 Barriers and critical issues in the implementation of LCA models in solid waste management

The two previous chapters have shown and discussed that there are a number of critical issues when modelling waste management systems in LCA. It is crucial that the user is aware of these issues and that they state what choices they take with regards to how they model these issues. There is not always a right or a wrong way to do it, but the choice on how it is modelled can have a determining factor on the result, so it must therefore be stated how it was modelled, what was included and, just as importantly, what was not. In addition, a number of barriers can hinder a successful implementation, which is also discussed in this chapter.

5.1 Critical issues in the implementation of LCA models in solid waste management

5.1.1 Time in waste LCA modelling

In product LCA the time factor is quite important, as for electrical products it is normally the use phase where most of the environmental impacts take place due to the use of electricity over a long time period. In LCA for waste management systems, the emissions are, most of the time, released almost instantaneously. There are, though, two exceptions to this: waste placed in landfills and waste applied to agricultural land. In these cases, the time horizon the study chooses to cover is significant. For the degradation of organic compounds an average time horizon covers around 100 years, but persistent substances such as heavy metals will remain for many millennia. There are currently two approaches to the modelling of this issue. Some models choose to model the landfill over an infinite time horizon, which means that they expect all emissions will – at some point – end up in the surrounding environment. This is done so that these processes can be compared with other processes where emissions take place instantly (waste incineration etc.). A number of models choose to model what they term “surveyable future time”, which as a default in many models is set to 100 years. After this time the emissions are no longer accounted for in the normal environmental impacts. In EASEWASTE, it has been acknowledged that there is a risk of substances still remaining after this period. A new impact category known as “stored toxicity” has therefore been introduced in EASEWASTE (adapted from Hansen et al., 2004 and Hauschild et al., 2008). The model calculates the amount of each toxic substance (heavy metals) that enters the landfill and is left at the end of the time horizon of the study, and ascribes each substance the characterisation factor for eco-toxicity to soil and water. This allows the user to choose whether he wishes to include this potential impact or not.

5.1.2 Modelling of carbon

The methodology behind the modelling of carbon in a waste LCA is critical when calculating the global warming potential (GWP). Carbon itself, when converted to CO_2, will have the same effect no matter whether it originates from a biogenic or fossil source. Nonetheless, only waste containing fossil carbon (primarily plastic products and textiles) is considered to contribute to global warming in a waste LCA. Food and paper products originate from organic material, which has been photosynthesised on atmospheric CO_2. Emissions in the form of CO_2 for these products are thus – in LCA terms – considered CO_2-neutral (Christensen et al., 2009). Similar is organic carbon when buried in the ground (landfill or agricultural land) and not released within the time horizon of the study, which was found to be a saving to the GWP.

The modelling of carbon is handled very differently in the different LCA models. IWM2 does not differ between the origins of the carbon at all, whereas EPIC/CSR does not account for the biologically originating CO_2, even though it still ought to be included in the inventory. The other models that were studied all had separate modelling for fossil and organic derived carbon. It differs whether the models acknowledge the binding of carbon in soils and landfill. However, it is imperative to understand what the model being used does assume, as this can give directly opposing results. This was seen in the Winkler (2004) study, where they did not discuss the difference in carbon approach, and therefore obtained some widely different numbers when comparing their results.

5.1.3 Setting the boundaries

An important part of a waste LCA is the correct setting of boundaries for the study, as discussed in Gentil et al. (I). These boundaries applied to a system are defined as (based on Guinée, 2002):

- The technical system and (natural) environment;
- The technical system and other technical systems (upstream and downstream boundaries such as the energy system); and
- Significant and insignificant contributions.

A fourth boundary that should be included is the time horizon, which has already been discussed above. The boundaries should be set as a part of the goal and scope of the study, and there is no right or wrong way to do this. The most important thing is to state what is included and what is not, as discussed in Damgaard et al. (VI). The boundaries for the technical system and the (natural) environment indicate what is included for waste management processes and what is not. This could be whether construction and maintenance costs are included, the demolition of buildings at the end of their life and the geographical scope of the study setting boundaries for what type of processes should and can be used.

Boundaries for the technical system and other technical systems set a boundary for how far out we extend the system (how much we include). This was studied by Merrild et al. (IV), who looked at incineration versus paper recycling. Figure 8 shows the global warming potential as a function of an increasing recycling rate, modelled with high and low efficiency for incineration and recycling technologies. The figure depicts an increasing expansion of the system boundary. The first graph only includes the incineration process and the avoided energy production versus the recycling process and avoided paper production. The second graph also includes the saved forestry from not having to harvest lumber for virgin paper production. The last graph includes the first and adds the option to use the forest biomass for energy production instead. The conclusion is clear for the high efficiency recycling process, as here recycling is always the favoured option. Then again, if the recycling process is not as efficient, it actually shows that by setting the system boundary too narrowly one can end up with a different conclusion – incineration is better than recycling. This shows that it is important to make sure all relevant processes impacted by the system and any cascading effects that might impact surrounding systems are included.

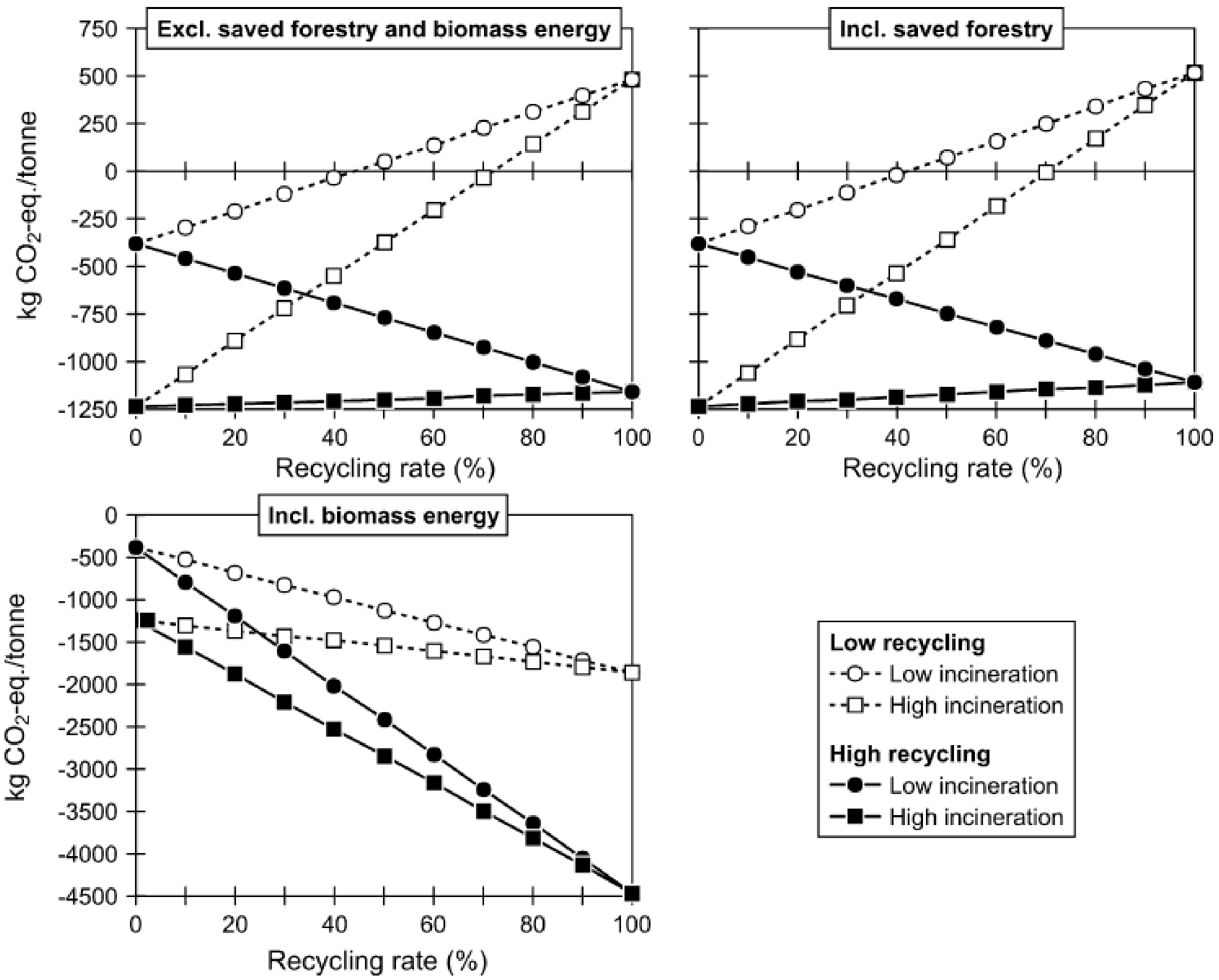

Figure 8 Global warming potentials as a function of the recycling rate for four combinations of technology levels and for three different system boundaries (from Merrild et al., IV).

The third boundary is to the point at which one separates between significant and insignificant contributions, which are normally defined by cut-off rules. The ISO 14040 (2006) standard states that it is important to understand the cut-off rules, but the standard does not state how to actually apply them. Frischknecht et al. (2007) argue that it should be done based on expert judgement, as quantitative rules would be hard to apply. For a practitioner it will often be hard to impact the choice of boundaries, as a lot of the data used will have inherent boundaries already set by the data provider. For processes where the user gathers data, one approach could be to look to other studies and see what they included and having this as a minimum goal for what should be included.

5.1.4 Waste composition

When modelling waste management systems, waste composition is one of the most important parameters. The reason for this is that all subsequent steps in waste modelling point back to waste composition. The chemical substances defined in the waste composition will end up in different places in a waste system, by the definition of the sorting of fractions at the home and in material recovery facilities (MRF). An example of this is a recent paper by Chen and Christensen (2010), where they show that differences in the moisture content of different wastes can determine what type of waste incineration technology will be the optimal choice. This indicates that it is critical to define a specific waste composition, and not just run with a default composition of waste. This is especially so in an integrated scenario where different sorting scenarios means that chemical substances will end up in different treatment processes depending on the fraction in which they originate. This is seen in Zhao et al. (V), where the waste to be handled by the waste managers is already heavily transformed, as unorganised individual collectors have scavenged and removed 23% of the waste (in the form of recyclables), thereby transforming the final waste composition to be handled. The model used for the LCA should be able to keep track of the fractions through the different treatment steps, so that emissions from treatment processes relate to the actual waste composition of the waste entering the facility, not on average emissions. This is the strength of the waste LCA models, as they are almost all predefined to handle these variations in fractional composition, and is consequently the drawback of the more general LCA models where this has to be set up manually, if at all possible.

5.1.5 Material recycling

The result of an LCA is related directly to the input data used in the model. This relates both to the direct emissions from the treatment processes and the emissions from upstream and downstream products. For treatment processes, it is clear that there will be differences depending on the treatment technology, as seen in Chapter 4 with the air-pollution-control technologies. However, indirect emissions are also important for the results of an LCA. An example of this is seen in Figure 9 from Merrild et al. (IV). This model compares reprocessing technologies with identical virgin production processes. The figure shows that for some of the recycled paper and cardboard types it can directly change the outcome of a study depending on which substitution process is chosen. This signifies that it is very important to make sure that the external processes used for avoided products represent the correct process.

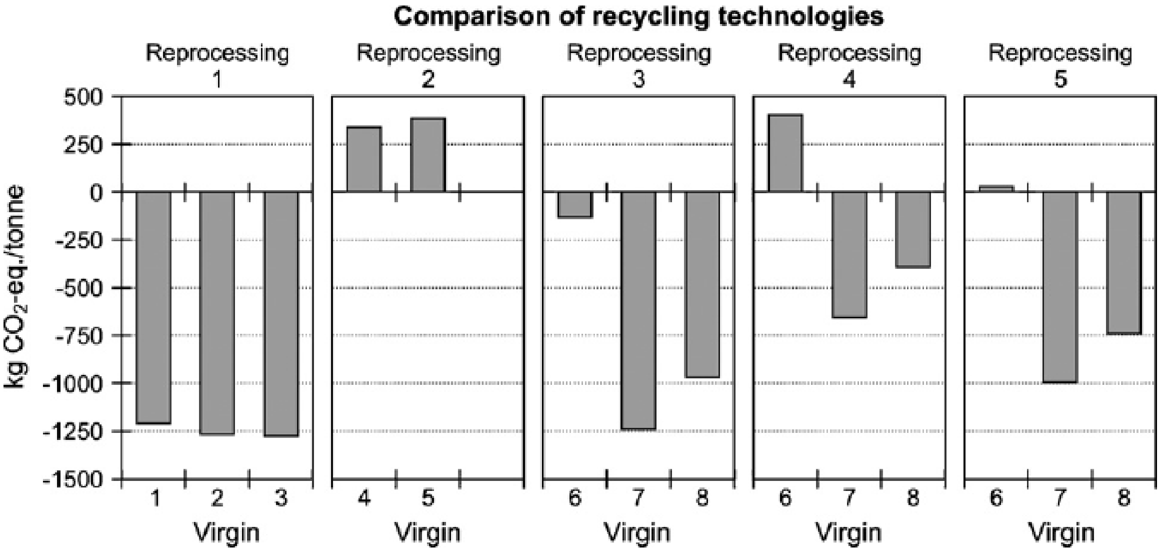

Figure 9 Global warming potentials for 14 recycling combinations modelled for five different paper and cardboard types (from Merrild et al., IV).

The problem is that recycling processes especially take place on market terms outside the direct control of the waste management system, so when the material first leaves the gate of a material recovery facility (MRF), to be sent for recycling, it is very hard to know where it will finally be recycled. Depending on the material and its inherent value, it might be recycled next door or it might be shipped to a facility very far away. Indirect emissions therefore vary considerably, as seen in Table 3, which shows the global warming factor (kg CO_2-eq. tonne^{-1} ww.) for the treatment of scrap aluminium. Here is seen a factor 4 difference on the potential savings. The main reason for this is that, similar to the paper processes in Figure 9, the avoided virgin production varies considerably. For aluminium, the reason for this variation is that virgin production is extremely energy intensive, and the type of energy used (e.g. hydro power versus coal power) varies widely from region to region, as shown in McMillan and Keoleian (2009). This indicates that it is very important to devise different scenarios that examine the importance of a parameter variation, which will then give an idea about the scale of uncertainty of the result.

Table 3 Greenhouse gas accounting and global warming factors for the treatment of 1 tonne of scrap aluminium (wet weight) in an MRF (direct and indirect emissions). After Damgaard et al., VI.

Indirect: upstream	Direct: waste management	Indirect: downstream
GWF (kg CO_2-eq. $tonne^{-1}$ ww): 6 to 45.8	GWF (kg CO_2-eq. $tonne^{-1}$ ww): 6.8	GWF (kg CO_2-eq. $tonne^{-1}$ ww): –5040 to –19 340
GWF (kg CO_2-eq. $tonne^{-1}$ ww): • Provision of diesel: 1–1.3 (GWP = 1) • Provision of electricity: 5–44.5 (GWP = 1)	GWF (kg CO_2-eq. $tonne^{-1}$ ww): • Combustion of diesel in grab cranes: 6.8 (GWP = 1)	GWF (kg CO_2-eq. $tonne^{-1}$ ww): • Reprocessing and avoided virgin production of 950 kg sorted aluminium scrap –5040 to –19 340 (GWP = 1) • Release of special emissions from virgin production (included above): CF_4: –133 to –6651 (GWP = 7390) C_2F_6: –24.4 to –1220 (GWP = 12 200)
Accounted (unit $tonne^{-1}$ ww): • Provision of diesel: 2.5 L • Provision of electricity: 50 kWh	Accounted (unit $tonne^{-1}$ ww): • Combustion of diesel in grab cranes: 2.5 L • Use of electricity (shredder and lighting): 50 kWh	Accounted (unit $tonne^{-1}$ ww): • Reprocessing of 950 kg sorted aluminium scrap • Release of special emissions from virgin production: CF_4: 0.018–0.9 kg C_2F_6: 0.002–0.1 kg
Not accounted: • Provision of waste scrap metal (collection and transport) • Construction of buildings, fixtures and other equipment	Not accounted: • Consumption of packaging material • Consumption of lubricating oil and other auxiliaries	Not accounted: • Long-haul transport of recovered scrap to smelters • Transport of residue material to treatment plant • Treatment of residue waste material

5.1.6 Energy in waste management systems

In solid waste management, the waste is often used in energy production (e.g. waste incineration or for the combustion of landfill gas). Furthermore, there is often a high use of energy in the waste treatment processes. This means that there is a large interdependency between the waste and the energy system. In waste LCAs where energy production is taking place, this often becomes the critical parameter. The reason for this is that the energy produced in an LCA is assumed to displace energy that would otherwise have been produced. This is where the choice between an attributional or consequential LCA modelling approach really has a large influence. In an attributional LCA, the displaced energy is assumed to be the average energy mix in the region where the study is taking place. Whereas in a consequential LCA it is assumed to be the energy type that will first react to changes in the market that will be displaced. This has huge implications when implemented in an LCA study, as seen in Damgaard et al. (II) where the efficiency of energy recovery in an incinerator was compared and set to substitute coal or natural gas, respectively. Figure 10 shows a graph of the great difference this will provide. If the substitutional energy had been a biogenic source, this difference would have been even larger, as demonstrated in Fruergaard et al. (2009). Since an LCA for solid waste management often will be performed to determine a future waste management plan, it will in reality always have consequences for future energy production for a region. It is therefore evident that, for most waste LCAs, the consequential approach is used. It is not always easy to verify what the future energy source will be. One way to accommodate this should be to test the choices by running different scenarios and substituting different energies to test the uncertainty of the energy parameter.

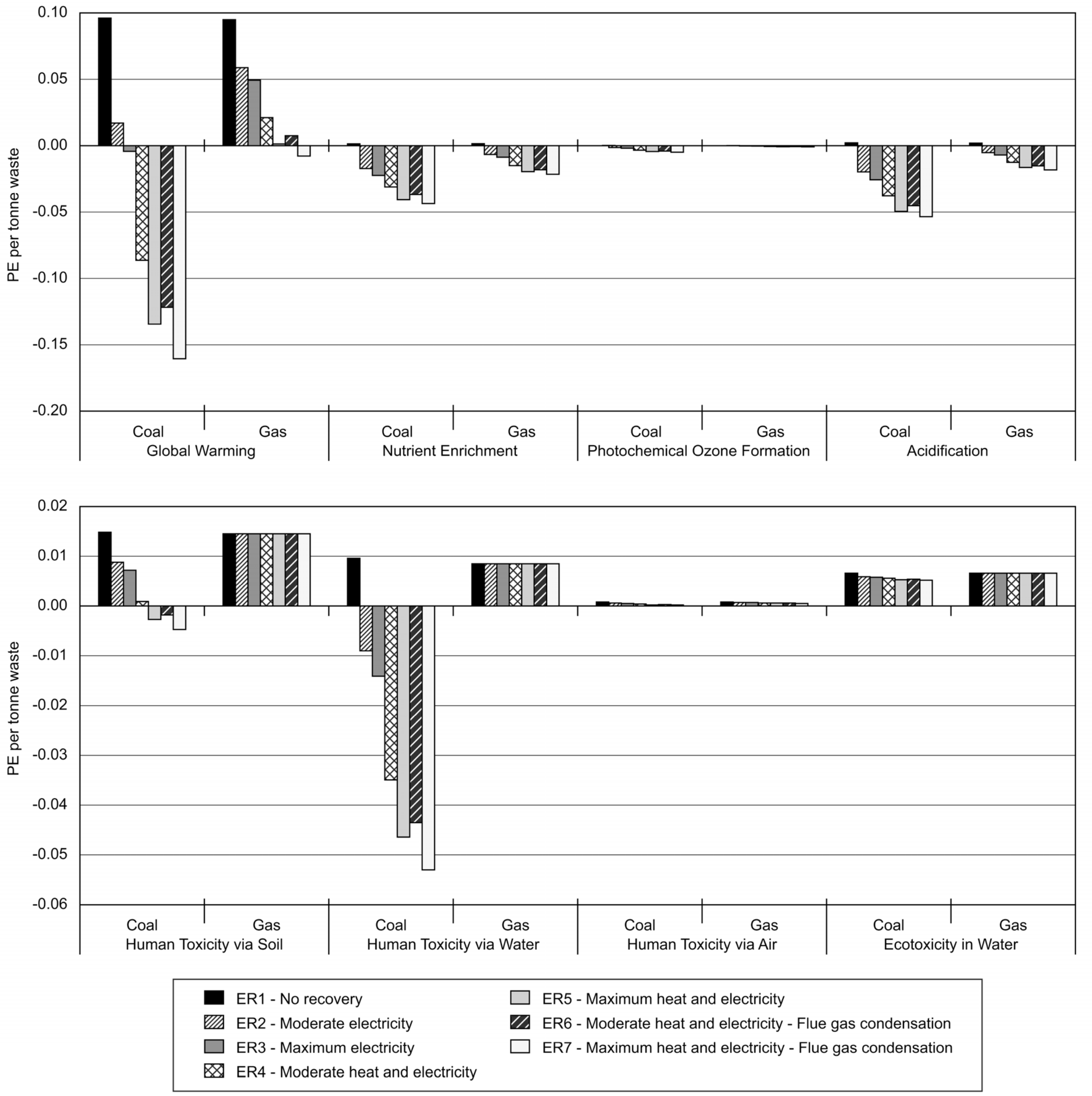

Figure 10 Standard and toxic environmental impact potentials in Person Equivalents (PE) showing the importance of the energy recovery rate and type of recovery (heat/electricity), exemplified with coal and gas substitution (from Damgaard, II).

5.1.7 LCA methodology and normalisation references

In this thesis, the main impact assessment method has been EDIP97 (Wenzel et al., 1997) and the results have been normalised with normalisation references from Sanddorf et al. (2005). It has to be kept in mind that the methodologies often have been developed with a certain regional setting in mind, and what might be a critical impact for one area might not be as critical in another. The choice of impact assessment method can thus have an impact on a study. In Dreyer et al. (2003) it was found that although results from CML2001 and EDIP97 fitted well for the non-toxic impact categories, they could give different results for toxic categories, and directly opposing results if compared with the Eco-indicator99 method. This shows that there is some uncertainty with regards to especially the toxic impact categories in the methods. In Damgaard et al. (III) it was found that some substances were contributing considerably more to the impact than expected, since the substances were expected to be degraded and therefore not to have that great an importance. By comparing the characterisation factors with those of the EDIP 2003 methodology (Hauschild and Potter, 2005) and the USEtox methodology (Rosenbaum et al., 2008), it was found that the impact from these substances was considerably lower in USEtox and EDIP2003. This shows that the choice of LCA methodology does have an impact. It is suggested to use the USEtox methodology for toxic impact categories if not having a locally developed methodology, as this method is intended for general use. This should be done as soon as a normalisation reference is developed for USEtox. For non-toxic impacts, the choice of methodology is most likely not as critical, but it will still have an impact and the choice of methodology should be stated.

In general, normalisation references should always, with the exception of GWP which is a global impact, apply to regional or local conditions. The study by Zhao et al. (V), which modelled the waste management system in the Chinese city Hangzhou, is one such example. The emissions were characterised with the EDIP97 methodology, as there was no Chinese impact methodology. Nevertheless, as the impacts took place in China it would be wrong to use a Danish normalisation reference, so a Chinese normalisation reference was consequently used. The difference between the two can be seen in Table 4. Based on the table numbers, it is evident that there are significant differences in the normalisation references. So, by applying a European normalisation reference to a Chinese study, the results from the impact assessment could be significantly changed.

Table 4 Environmental normalised potential impacts reference in China and EDIP 1997. In Zhao et al., V. (References based on Stranddorf et al., 2005 and Li et al., 2007)

Environmental impacts	Standard unit	Normalisation reference	
		China	EDIP 1997
Global warming (GW100)	kgCO2 eq year^{-1}	8700	8700
Stratospheric ozone depletion (OD)	kgCFC-11 eq year^{-1}	0.20	0.103
Acidification (AC)	kgSO2 eq year^{-1}	36	4
Nutrient enrichment (NE)	kgNO3 eq year^{-1}	62	119
Photochemical ozone formation (POF)	kgC2H4 eq year^{-1}	0.65	25

5.1.8 Impacts to groundwater

Most LCA methodologies only cover impacts to air, soil and freshwater. In solid waste management the waste can end up in landfills or biologically be applied to farmland. Here, leachate and its contained polluting substances will leach from the landfills and soil and end up in groundwater. Impacts to groundwater are usually not covered in a general LCA methodology. In the studies it has been introduced by the impact potential "Spoiled Groundwater Resource" (SGWR). The impact is calculated as the volume of groundwater that the input to the groundwater (here leachate substances) can contaminate – which goes right up to the drinking water criteria. This impact is adapted from Birgisdóttir et al. (2007), where it was used on leaching from bottom ash residues used in road construction. In EASEWASTE modelling, the emissions have been characterised with the WHO (2006) drinking water criteria instead of the Danish drinking water criteria used in Birgisdóttir et al. (2007). Similarly, as for the other impact categories, the calculation is made for each substance, and then the sum yields the potential impact. The impact is normalised with regards to the amount of contaminated groundwater per person per year in Denmark (2900 m^3/person/year (DMU & DJF, 2003)). The normalisation reference is based on the contamination by nitrate and chloride to Danish groundwater, and must be seen as a rough indicator. If used outside Denmark, a relevant normalisation reference should be found for the area where used. This impact potential is relevant only when groundwater is considered a limited resource and utilised. In Damgaard et al. (III) it was shown that by collecting leachate the toxic impact categories worsened, as the treated leachate would end up in freshwater recipients included in the standard methodologies. This shows that, if not including the groundwater impact, a waste LCA study will actually promote a no leachate collection strategy for landfilling. This is, of course, unrealistic, so it is therefore important to acknowledge the emissions for groundwater and calculate their associated impacts.

5.1.9 Zero burden?

Most LCA studies where there is substitution for virgin material taking place, or where fossil energy is substituted, will give overall savings in most categories. This is seen in Damgaard et al. (II, III and VI), Merrild et al. (IV) and Zhao et al. (V). This seems to indicate that waste management is a benefit to society, and the more waste we can treat, the better. Nevertheless, it has to be borne in mind that the reason for these savings is that we are assuming that the waste is carrying zero burden. This means that we have excluded all emissions made when producing the materials before becoming waste. If including these emissions, the waste management sector would, of course, see a net burden. It therefore has to be kept in mind that the most efficient way to lower emissions is to avoid the production of waste. However, when waste is first generated the focus should be on finding the largest overall savings for the waste management system.

5.2 Barriers in the implementation of LCA models in solid waste management

Section 5.1 discussed issues to be aware of in order to carry out a successful implementation of LCA models in solid waste management. There are, though, some barriers which were seen as a part of the thesis work.

5.2.1 Time and economy

As LCA models get better and better, they also require more and more data. This not only means that the results get closer and closer to actual reality, but it also means that more and more time and therefore money is needed to carry out the study. One way to remedy this is to expand the number of treatment processes available in the LCA models, so the user can identify the processes most similar to their needs and fit this technology to their precise case. That said, there has to be an understanding that if the time and money to set up processes precisely targeting the case are not used, they will only get average results. In a worst case scenario, this course of inaction could produce a directly opposite recommendation to what their own data would have shown. One way to remedy this could be to undertake a screening study using average data, and thereafter collect data for the most important areas identified. In this way, the amount of data needed could be cut down and hence the overall cost of the study as well.

5.2.2 User-friendly models?

Section 5.1 showed that there are a large number of issues that have to be considered when performing an LCA. Furthermore, models are now including more and more aspects of solid waste management. This requires that they manage to incorporate these aspects and at the same time be user-friendly if they are to be used extensively in the industry. A study by Eriksson et al. (2003) found that technical expert knowledge in the industry has not always been developed enough to take advantage of the possibilities in the models and get the maximum benefits available. The authors consequently found that it would be better to have an external expert to help carry out the study and bring in the required knowledge. An additional benefit of this is that it would provide the user with more credibility in the eyes of stakeholders in the final result, as the process would be undertaken and controlled by an external expert. LCA models have developed considerably since 2003 (the study was performed with ORWARE), and the newer models have a much more accessible graphical user interface, allowing easier access to view and update input data and model parameters. The need for expert knowledge has, though, not been lessened as more and more issues of importance are being raised in the models, and it can therefore still be an advantage to obtain consultancy help in setting up a first model for a waste management system or technology. Internal users can then change parameters in this model in further studies, after it has been initially set up. The client should, though, be included in carrying out the project, as it is seen that a lot of new insight can be gained in their own technologies when asked to assess them from a new angle (LCA).

5.2.3 Data

The implementation of LCA models in solid waste management requires large amounts of data – both direct and indirect emissions data. Access to these datasets, though, might be limited, as data providers often only cover data they are legally required to report. This makes it hard to assess if all the important emissions are included, or if some critical emissions are missing from the inventory. One way this could be remedied would be to look at similar processes in LCA databases and see whether they include other important emissions which should be monitored. This is, of course, only possible for direct emissions that can be measured. The issue is the same for marginal data, which is required in a consequential LCA study. In this instance, it might sometimes be necessary to compromise, as it is possible to neither get data for a marginal process nor establish what the marginal process is. In that case, it will be necessary to use average (attributional) data. The aim should, of course, always be to use marginal data where it exists, but it is believed that it is more important to include a process with marginal data than cut off that part of the system.

5.2.4 Uncertainty

Based on all the issues mentioned and the data required in the models, another barrier is uncertainty in the final result. A number of the choices when setting up the system will have large consequences for the results, and with these choices comes a certain amount of uncertainty. This was also seen in the modelling workshop, where some of the processes varied considerably. This can certainly promote scepticism in regard to whether one can actually trust the results of the study. In order to substantiate the findings and make them more credible, it is important to assess the importance of the choice made on the issues mentioned in section 5.1. This is – for most of the LCA models – easiest achieved through parameter variation and scenario analysis, as suggested by Finnveden et al. (2009). This type of uncertainty assessment entails calculating a result with a number of different data values and/or choices, e.g. using the maximum and minimum efficiency/emission, and seeing if the results are stable. A number of the product LCA models offer more advanced statistical tools such as Monte Carlo simulations. From the work within this thesis, it is believed that with the process modelling approach of waste LCA it is most beneficial to use parameter variation, as this – besides examining uncertainty – also gives the user an understanding of what is important and what is not. For an expert in LCA and waste, these findings might be obvious, but for a non-expert this can be extremely valuable new information.

6 Conclusions

The process of implementing LCA models in solid waste management is already taking place, and as the amount of models and availability of data increases, the interest in the models also grows. The aim of this PhD study was to determine how these models can be implemented in waste management, and even though the implementation already takes places there is room for improvement.

The review of the nine LCA models showed that there were differences in the data used in the models, the thinking behind the goals of the models and the methodology used in the models. All of these factors showed that the results from the models should never be compared directly. At the same time, the comparison showed that the overall recommendation when using the model for a study would be the same, as long as the same assumptions were made in the modelling (compare comparables). When using an LCA model, it has furthermore to be kept in mind where the model was developed, as conditions suitable for one place might not be fit for another. In addition, models evolve over time and newer models have learnt from past experience and built on this. This improvement is with regards to both the mathematical formulas and details in the process modelling, as well as the improved general user interface.

The screening of the published LCA studies showed that the majority of the studies were case studies of full systems. This, combined with the knowledge from the models comparison that there are large differences in the modelling of the same types of technology, confirmed the need for the second part of the PhD in making sure that the processes are modelled correctly. In order to carry out case modelling, a user will always need to decide if it is an existing or a new facility. In both cases, a default technology will often be very far from the real case technology. The study on air-pollution-control for waste incinerators showed that even as little as a few sets of year-old data gave differences in results, and a decade-old process could change the conclusion of a study. This illustrated how the implementation of LCA models in solid waste management can help remove some of the myths that still exist about some processes. It also emphasised the importance of using the best available technologies (BAT) when performing the LCAs of future waste management systems, or at least the need to be consciously aware that one is not modelling BAT if one chooses to do so. Similarly, the landfill paper showed that the leachate and gas collection efficiencies in a landfill design could be crucial for results. Since these parameters are related directly to the composition of the waste (methane potential) and location of the landfill (precipitation), getting the input data right is even more important. Using a default landfill not tailored to the study area, and not taking this into account, would therefore produce large discrepancies.

It was shown that a good LCA can be completed only when the importance of key parameters is well understood. For a successful implementation of the LCA models in solid waste management, these barriers and issues must consequently always be considered; otherwise, there will be the chance of an inconsistent result. One of the key parameters is the setting of the boundaries for the system. There is no right or wrong decision in where the boundary is set, but it is crucial to define what is included and what is not. Similarly, it is crucial to define whether the study uses an average, marginal or a mix of input data, as this also has an important impact on the result. However, it should also be noted that if direct data is not available, then default average data will, of course, be better than no data.

In conclusion, it can be said that taking into account the amount of data, the variability in the data quality and uncertainties with regard to the assessment method, all these factors point to the conclusion that conducting LCAs for waste management systems cannot be considered an exact science. Nonetheless, the goal of implementing LCA models in solid waste management is not necessarily to obtain a final single number, but rather to generate an indication of the best choices when considering uncertainties. In addition, and just as importantly, performing such a study with a systematic approach will help teach the user to understand the system or technology better, thereby allowing them to optimise their system and learn what is important and what is not.

6.1 Recommendations

The work with the different waste LCA models and the background databases shows that good documentation is extremely important in order to assess whether a process is suitable for what is to be modelled. With this in mind, and the fact that the articles demonstrate how important it is to model the technologies of today and not a decade ago, it must be said that the databases available in the dedicated waste LCA models, as well as the general LCA databases, do not always reflect the newest situation. Often, actually old data is used for wont of better options to perform a fully integrated LCA study. This is still better than not performing a study at all, but with the uncertainty this can bring, it is suggested that work is done in updating and expanding the databases with processes that reflect today's conditions.

6.2 Further research

This thesis has looked at the technical implementation of LCA models in solid waste management, the models that can be used and the technical parameters that play an important role. However, there is another side to implementation, which is how to get the LCA models intended for use into the waste management industry. How can the industry use the models for internal awareness rising, external communication of how well they perform, benchmarking and monitoring? LCA can also not stand alone, so how can the industry make it work together with an assessment of the economic and social impacts not covered by the models? All of this should be investigated further.

In the work, a general assumption was that waste is a "zero burden". As a consequence, there is a vital need for more information on what this burden really is, and where waste prevention will have the largest effect. What are the drivers that can help push towards more prevention: taxes, regulations, information campaigns?

The thesis has focused purely on municipal solid waste, and it is most likely that the need for integrated analysis is highest due to the heterogeneity of this waste type. The implementation of specific waste models for waste electronic equipment, industrial waste and hazardous waste is equally as important. The design of waste LCA models can be used for the modelling, but data for the special treatment processes needs to be created to allow the modelling in the first instance.

The work as part of the thesis also highlighted that there is still a need for more work to be done with LCA methodologies applied to the calculated inventories. As it becomes possible to track an increasing number of emissions to different recipients, it is important that the impact methods can accommodate this and help convert these emissions into a quantitative measure. Emissions to groundwater particularly are not currently assessed properly, so work needs to be done in assessing the impact of these discharges. Moreover, an assessment of what the normalisation reference should be for different regions in Europe, as well as the rest of the world, should be carried out.

The thesis focused on environmental impact categories, but none of the articles within this research examined resource depletion. Once again, the tracking of this issue is not possible in all models, although the newer models do accommodate this requirement. There is therefore a need to bring this into the modelling of waste processes to a higher degree, as there are huge resource savings to be found in the different treatment processes. The possibility of recovering phosphorous in biological waste treatment is believed to play a big role in future studies and waste management.

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8 Appendices

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