THESIS ON CIVIL AND ENVIRONMENTAL ENGINEERING

Life Cycle Assessment as a decision support tool for system optimisation – the case of waste management in Estonia

HARRI MOORA

2009

TALLINN UNIVERSITY OF TECHNOLOGY
Faculty of Civil Engineering
Department of Environmental Engineering

Dissertation was accepted for the defence of the degree of Doctor of Philosophy in Civil and Environmental Engineering on May 12, 2009.

Supervisors: Professor Enn Loigu, Department of Environmental Engineering, Tallinn University of Technology

Assistant Professor Andrius Plepys, International Institute for Industrial Environmental Economics, Lund University

Opponents: Associate Professor Arvo Iital, Department of Environmental Engineering, Tallinn University of Technology

Assistant Professor Carl Dalhammar, Lund University

Defence of the thesis: June 22, 2009

Declaration:

Hereby I declare that this doctoral thesis, my original investigation and achievement, submitted for the doctoral degree at Tallinn University of Technology has not been submitted for any academic degree.

Harri Moora

ISSN 1406-4766

ISBN 978-9985-59-915-0

EHITUS JA KESKKONNATEHNIKA

Olelusringi hindamine kui süsteemi optimeerimise otsuse toetusvahend - Eesti jäätmekäitluse näitel

HARRI MOORA

2009

TABLE OF CONTENTS

ABBREVIATIONS

BMW	Biodegradable municipal waste
BSES	Baltic sustainable energy scenario
C_2H_4	Ethane
CBA	Current trend scenario
CO_2	Carbon dioxide
CH_4	Methane
CHP	Combined heat and power
COD	Chemical oxygen demand
CTS	Current Trends Scenario
EEA	European Environment Agency
EIA	Environmental impact assessment
EU	European Union
GDP	Gross domestic product
GHG	Greenhouse gases
HCl	Hydrogen chloride
IOA	Input-output analysis
IO LCA	Input-output life cycle assessment
LCA	Life cycle assessment
LCI	Life cycle inventory
LCIA	Life cycle impact assessment
LCC	Life cycle costing
MSW	Municipal solid waste
MW	Megawatt
N	Nitrogen
NH_3	Ammonium hydroxide
NMVOC	Non-methane volatile organic compounds
N_2O	Nitrous oxide
NO_x	Nitrogen oxide
NPP	Nuclear Power Plant
P	Phosphorus
PP	Power plant
PPS	Purchasing Power Standard
SO_2	Sulphur dioxide
VOC	Volatile organic compounds
WAMPS	Waste management planning system (LCA model)
WTE	Waste-to-energy

List of publications included in this thesis

Paper I: Moora, H., Stenmarck, Å., Sundqvist, J-O., 2006. Use of Life Cycle Assessment as decision-support tool in waste management planning – optimal waste management scenarios for the Baltic States. Environmental Engineering and Management Journal, September, Vol. 5, No. 3, 2006, p. 445-456

Paper II: Moora, H., Sundqvist, J-O., Stenmarck, Å., 2007. LCA-based decision-support tool for waste management planning – optimal waste management scenarios for the Baltic States. Conference Proceedings - 3rd International Conference on Life Cycle Management (LCM2007), Zurich, 26.-29.08.2007, University of Zurich at Irchel.

Paper III: Moora, H., Voronova, V., Reihan, A., 2009. The Impact of Municipal Solid Waste Management on Greenhouse Gas Emissions in Estonia. Leal Filho, W. & Mannike, F. (eds.) Interdisciplinary Aspects of Climate Change. Series: Environmental Education, Communication and Sustainability. Vol. 29. p. 311-325. Peter Lang Publishers.

Paper IV: Moora, H., Lahtvee, V., 2009. Electricity Scenarios for the Baltic States and Marginal Energy Technology in Life Cycle Assessments – a Case Study of Energy Production from Municipal Waste Incineration. Oil Shale. (accepted)

1 INTRODUCTION

1.1 Municipal solid waste as a problem

The generation of municipal solid waste (MSW) in the European Union (EU) has increased steadily in the past decades as Europeans have become richer and consume more. This has been the main reason for a parallel rise in waste amounts and the related environmental impacts and waste management problems. The amount of MSW per year is expected to grow by 25% within the EU from 2005 to 2020, with striking differences between Member States (ETC/RWM, 2007).

The new EU Member States including Estonia have recently experienced a rapid economic development, resulting in a significant increase of waste quantities, while their waste management systems still require much effort to be adjusted to the European state-of-the-art. The municipal waste management system in Estonia must comply with the principles and targets of the European waste policy and directives. Therefore, waste management has become one of the key issues in governments and policy-makers as well as the general public.

Municipal waste disposal has the potential to cause a number of impacts on health and the environment, including emissions to air, surface water and groundwater, depending on how it is managed. The contribution of waste management sector (especially landfilling) to climate change has been recently more and more discussed. Waste also represents a loss of natural resources (such as the metals or other recyclable materials it contains, or its potential as an energy source).

Obviously solid waste management is a complex issue, combining management, technical, economic, political, social and behavioural aspects. Waste management usually involves dealing with complex systems, managing a large workforce and working together closely with the public and business companies. Problems related to setting up a system and financial aspects are common, as well as mistakes in the selection and maintenance of technologies. Political decisions and companies' struggle for market share have an impact on the waste management infrastructure and technologies chosen to deal with the collection and utilisation of waste. Social norms affect which waste management choices become a daily routine and which are just short-term experiments. Psychology plays an important role in determining how waste management systems should be best designed to ensure public support and participation in waste management initiatives and practices. Therefore, the preparation and management of optimal solid waste management systems needs inputs from a range of disciplines, and careful consideration of local conditions. There are many factors that vary from place to place and they must be considered in the design of a system.

Waste itself is also an important factor. Typical MSW in developed countries contains a variety of waste materials. The increasing amount of waste products that contain toxic substances (e.g. small electronic equipment) can cause significant problems in waste treatment facilities. This, combined with new emerging waste management technology options, makes the planning of a waste management system a challenging task for waste management planners.

1.2 The waste hierarchy

The current EU waste policy as well as the related legal requirements and targets are based on a concept known as the waste hierarchy. This means that, ideally, waste generation should be prevented or reduced, and that which is generated should be recovered by means of re-use, recycling and other recovery operations, thus reducing disposal/landfilling operations (Figure 1).

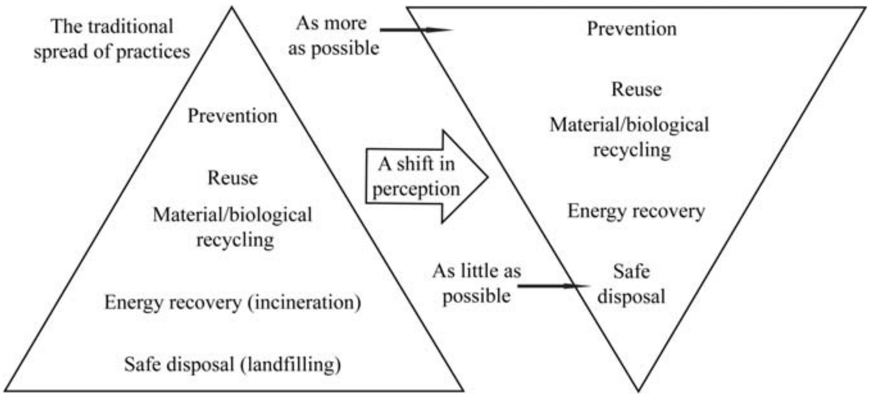


Figure 1. The waste hierarchy: change of priorities in waste management strategies

Waste management has been governed by the waste hierarchy for decades. However, this concept and the ranking of waste management options have caused a lot of discussion. For example the position of waste incineration with energy recovery in the hierarchy was a subject of intense debate under the process for the review of the EU Waste Framework Directive 2008/98/EC (EC, 2008). Thus, the following question can be asked: is the waste hierarchy based on scientific evidence or on presumptions and affective values?

The experience of the old Member States shows that implementation of the EU waste policy can be achieved in many ways. Some of the solutions can be more eco-efficient and sustainable than others (CEC, 2005). The waste hierarchy is a good guideline to assess waste management options, in particular when waste management plans are being developed or revised at the national or regional level. However, the hierarchy may not necessarily be regarded as being a rigid prescription, because the environmental impact of a waste management system depends on a number of local factors, such as the environment, economy, society and technology related factors. Taking into account specific regional features and aiming of at cost-effectiveness the implementation of the waste management hierarchy may require re-prioritisation of waste management objectives and subsequently demand an input for designing optimal waste management strategies.

As highlighted in the European Commission's Strategy on the prevention and recycling of waste (EC, 2005a), in the Thematic strategy on sustainable use of resources (EC, 2005b) and the revised European Waste Framework Directive, life cycle thinking is essential for the planning of sustainable waste management. This means that the EU waste policy has to be implemented in a sustainable manner, considering the full life cycle of the resources and quantifying the environmental and economic benefits/trade-offs associated with alternative waste management options for achieving the targets. Derogations from the waste hierarchy could be granted based against scientific proof given by environmental assessment tools, such as life cycle assessment (LCA).

1.3 Application of LCA in municipal waste management planning

A large number of analytical tools for assessing environmental impacts are available (Wrisberg et al., 2002; Finnveden and Moberg, 2005; Kates et al., 2005; Ness et al., 2007). Life cycle assessment as one of the comprehensive decision-support tools for analysing complex socio-economic systems has gained high recognition in many applications, including waste management planning (Morrisey and Brown, 2004; Olofsson, 2004; Moberg, 2004; Ekvall et al., 2007). Several LCA models for MSW planning have been developed during the last decade (EASEWASTE by Kirkeby et al. 2006; LCA-IWM by den Boer et al. 2005; WISARD and WRATE by the UK Environmental Agency; ORWARE). These models are very different with respect to level of detail, user-friendliness, flexibility, transparency, databases provided and level of impact assessment. The inherent complexity and high requirements for the background data often hinder the use of LCA models in a decision making process.

The main challenges relate to the scope of covering the interactions of a given waste management system with its local conditions. Different studies though based on the same (LCA) logic show different results (Winkler, 2005; Winkler and Bilitewski, 2007), which may hinder the applicability of LCA for waste management planning. The applicability of LCA for waste management planning is restricted by certain limitations (Finnveden et al., 2007; Ekvall et al., 2007). Some of them are characteristic to LCA methodology. Others are relevant for specific local waste management conditions such as financial information, waste composition, the design of waste collection scheme and mode of transport or raw material and energy sources used for electricity and heat supply.

A comprehensive analysis of waste management systems requires the collection of specific data, which often is rather difficult and time-consuming. This leads to reliance on assumptions or use of surrogate data from other regions (e.g. average waste composition in EU). Depending on the LCA's objective and scope, the error introduced by such assumptions and data can be substantial.

The costs and benefits of various waste management alternatives (especially the environmental impacts and costs) have not been properly taken into account in the waste management planning process in Estonia, where the use of LCA for these purposes is still in its infancy. The major limitation is the lack of relevant data and methodological expertise. It is also difficult to compare specific waste management data with the data used by LCA models developed for other countries.

In order to make the results of LCA studies more reliable and to save time and resources it is important that these limitations, including the sensitive input characteristics and critical assumptions that have larger impact on the final results of LCA, are known and understood by both the LCA practitioners and users (decision-makers) of such studies.

1.4 Aim of the thesis

This thesis investigates the validity of high-level policy frameworks by the example of a municipal solid waste management system. The thesis questions and explores the validity of the waste hierarchy as a guide for strategic decision-making at the regional level. The research is based on simulations of the consequences of different waste management treatment options and technologies in Estonia by using LCA methodology.

The secondary goal of this research is to better understand the applicability of lifecycle thinking and LCA-modelling for analysing complex technical, socio-economic and environmental systems such as waste management. The applicability of LCA is restricted by certain limitations. In this thesis the specific local characteristics and critical assumptions that could have a larger impact on the final results of LCA are discussed. Finally, the research provides background insights for the development of a common reference system (predefined regional data sets) for the LCA in waste management.

The discussion focuses mainly on the environmental impacts characteristics of different waste management systems. A discussion of economic issues is limited.

The following research questions are put forth:

1) How can regional contexts influence the European waste hierarchy as a general policy framework for municipal waste management strategies?
2) What are the specific system characteristics that influence decision-making process the most?

2 METHODS

2.1 Research design

The purpose of this section is to present the research logic by linking the research results (published in various articles) and explaining how they contribute to the overall research goal.

The research is built on analysing the practical applications of LCA in actual case studies in Estonia providing insights from the user point of view. This research has evolved from a number of studies conducted by the author in 2000-2008. The thesis summarises and discusses research findings published in four academic articles selected for the thesis (see the list below). Other referred papers and studies of the author contribute to the content in the form of data and analytical results. The discussion and analysis of findings is conducted along the framework described in the methodology section.

Papers included in the thesis

Paper I: Moora, H., Stenmarck, Å., Sundqvist, J-O., 2006. Use of Life Cycle Assessment as decision-support tool in waste management planning – optimal waste management scenarios for the Baltic States. Environmental Engineering and Management Journal, September, Vol. 5, No. 3, 2006, p. 445-456

Paper II: Moora, H., Sundqvist, J-O., Stenmarck, Å., 2007. LCA-based decision-support tool for waste management planning – optimal waste management scenarios for the Baltic States. Conference Proceedings - 3rd International Conference on Life Cycle Management (LCM2007), Zurich, 26.-29.08.2007, University of Zurich at Irchel.

Paper III: Moora, H., Voronova, V., Reihan, A., 2009. The Impact of Municipal Solid Waste Management on Greenhouse Gas Emissions in Estonia. Leal Filho, W. & Mannike, F. (eds.) Interdisciplinary Aspects of Climate Change. Series: Environmental Education, Communication and Sustainability. Vol. 29. p. 311-325. Peter Lang Publishers.

Paper IV: Moora, H., Lahtvee, V., 2009. Electricity Scenarios for the Baltic States and Marginal Energy Technology in Life Cycle Assessments – a Case Study of Energy Production from Municipal Waste Incineration. Oil Shale. (accepted)

Paper I and II

The results from early stages of this research are presented in Papers I and II. Here the main aim was to test the validity of the waste management hierarchy (in the regional context) and evaluate the environmental and economic performance of different technological options for MSW management in the Baltic States. The secondary aim of these papers was to better understand the suitability of LCA methodology for modelling waste management systems. For that reason a simplified screening level LCA model (WAMPS) was developed and tested with the specific regional particularities. The results of the research were based on the case studies carried out in Estonia.

In Paper II the updated version of the WAMPS model was used where some important prerequisites were changed. In addition, Paper II investigated how municipal waste performs in comparison with other fuels (biomass) when used in combined heat and power plants. Paper II also analyses the capabilities of this kind of models and discusses the feasibility of their application in the policy making process (at local level).

Paper III and IV

In the later stages of the research the interactions of the analysed waste management systems with the climate change impact were studied. Paper III examines the climate change impact in Estonia in terms of GHG emissions from MSW management between years 2000 and 2020. Two most feasible waste management options for Estonia, material recycling and composting, and material recycling with intensive incineration, were compared in terms of their possible contribution to climate change. As such the paper also contributes to the discussion related to the implementation of the waste hierarchy in Estonia.

Based on the results and experiences of the LCA case studies the sensitive data points and critical assumptions relevant for specific local circumstances were analysed. Waste data is one of the crucial input parameters of LCA in waste management. Lack of information on actual waste composition is one of the main barriers to waste management planning (planning of recycling and energy recovery potential) in the Baltic States. Paper III provides the most updated information on the composition and generation of MSW in Estonia. The composition of mixed municipal waste in Estonia was explored in an empiric study by carrying out a country-wide waste sorting analysis.

The case studies show that the environmental impacts of electricity production could account for a major portion of the total environmental burden and economic benefits of the studied systems. For example, waste incineration in combined heat and power (CHP) plants reduces the need for other energy resources causing a marginal effect on the electricity system. Due to the specific nature of the Baltic electricity system, it is important that the waste management decision makers in these countries are aware that among other LCA related sensitive data issues the possible background electricity source could significantly influence the results of LCA. In Paper IV the possible short- and long-term future electricity scenarios for the Baltic States were analysed to identify the possible marginal electricity sources, which should be used in consequential LCA studies in these countries. To illustrate how the choice of electricity input data could influence the results of the LCA a case study on municipal waste incineration with energy production was carried out.

2.2 Methodology

2.2.1 Overview of decision support tools

Several tools for environmental systems analysis have been developed to support different types of decisions (Wrisberg et al., 2002; Finnveden and Moberg, 2005; Kates et al., 2005; Ness et al., 2007). They may be divided into procedural and analytical tools (Wrisberg et al., 2002). Procedural tools focus on providing the procedures leading to decision-making, while analytical tools provide information that could be used for system optimisation, comparison of alternatives, communication, etc.

Typical procedural tools are for example Strategic Environmental Assessment (SEA) and Environmental Impact Assessment (EIA). The most common analytical tools are: Life Cycle Assessment, Material Intensity per Unit Service (MIPS), Environmental Risk Assessment (ERA), Material Flow Accounting (MFA), including Substance Flow Analysis (SFA), Cumulative Energy Requirement Analysis (CERA), Environmental Input-Output Analysis (IOA), analytical tools for eco-design, Life Cycle Costing (LCC), Total Cost Accounting (TCA), and Cost Benefit Analysis (CBA). Regarding the scope of the tools the first seven focus on an analysis of physical flows, the last three mainly on monetary flows. However, integration of environmental, economic as well as social aspects into one tool is becoming more popular (e.g. life cycle costing in LCA or environmental CBA, etc.) (Reich, 2005).

Which tool to use in a specific decision-making situation depends on the decision context. When linking tools and decision context, some aspects can influence the choice of tool, whereas others influence how the tools are used. The choice of an appropriate tool in different contexts is largely decided by two aspects: the object under study (e.g. products, services, policies, plans, regions, organisations, etc.) and the impacts of interests (environmental, economic, social, etc.) (Finnveden and Moberg, 2005). At the same time there are many aspects which could influence how the selected tool is used. Some of them could also influence the choice of the tool. Examples are complexity, degree of aggregation and level of detail, scope (site-specific or general), scale of the decision and preferences, credibility, cultural context (Finnveden and Moberg, 2005).

The scale of the decision will influence the amount of resources put into the analysis. This could influence how the tool is used (screening level or detailed study). In a decision-making process, both local or site specific and site-independent information may be of interest. Some of the above mentioned tools are applicable more on site-specific objects while others may be used for different objects. For example, if the decision maker is concerned about local effects, e.g. when deciding the location for a waste incinerator, site specific assessment should be made using specific tools such as ERA or EIA. If, on the other hand, there is a need to compare the environmental impacts of different waste management scenarios (e.g. recycling versus incineration), life-cycle based assessment tools such as LCA would be appropriate.

Different tools also can complement each other by adding different types of results. For example, the results of an EIA study could be combined with LCA results which give additional information in the life cycle perspective.

An important aspect recently much discussed is whether the system analysis tool can be used for change-oriented studies (consequential) or retrospective studies (attributional) (Tillman, 1999; Weidema, 2003; Finnveden and Moberg, 2005). Change-oriented studies analyse the consequences of choices or decisions (e.g. assessment of waste management or energy plans, improvement of a production process, etc). Ideally, the data and system boundaries used should reflect the *ex ante* or the predicted changes in the future, which may depend on the scale of the change and its time span. Retrospective studies usually describe a system as it was (*ex post* or past activities) or provide information on the consequences of individual actions.

If the aim is to assess future consequences of a decision on the system level, consequential approach should be used (Weidema, 2003) (see Chapter 2.2.3 below). The issue related to the need for different sets of data and methodology depending on whether the tool is used for attributional or consequential studies, is more discussed in the context of using LCA and not so much covered in other tools.

Compared to many other decision-support tools life cycle thinking and especially LCA have gained wide acceptance in providing policy relevant and consistent results (Björklund and Finnveden, 2005). It can be used both as a descriptive tool as well as a change-oriented tool with different choices of data and methodology. Recently also economic aspects have been integrated into LCA (input-output LCA, life cycle costing in LCA, etc.), which makes the tool even more suitable in the decision making process. Its strength compared to many other environmental system analysis tools is also that the framework, terminology and methodological choices of LCA are standardized by the International Standardisation Organisation (ISO) (ISO, 2006). This gives LCA greater credibility.

The broad system perspective makes LCA a very useful tool also for an environmental and economic comparison of waste management options (e.g. testing the validity of the waste hierarchy). Waste management is a complex system that is difficult to study. The system complexity is even bigger if links to other sectors such as manufacture of products, energy production and agriculture are taken into account. This means that impacts of a waste management system could be both local and global. The benefit of using LCA in analysing different waste management systems is that it provides a comprehensive view of the processes and impacts involved. This is important, since the indirect impacts caused by background systems (e.g. energy and material production), often are bigger than the direct impacts of the waste management system itself

(Ekvall et al., 2007). LCA helps to analyse local environmental pressures and waste management costs, while considering these in the broader context of the benefits (e.g. recycling of materials or substitution of fossil fuels in the energy production) and trade-offs felt elsewhere across the society (e.g. contribution of waste management in the climate change impact).

However, LCAs like other system analysis tools in general are a simplification of the complex reality. Therefore it is clear that LCA studies will always be open for criticism. There are several limitations in LCA such as limitation in predicting future, understanding the processes, choosing appropriate time frame, limitations in being site-specific and using a wide system approach, lacking knowledge in specific impact categories (e.g. toxicity), etc. Data and methodological uncertainties are still large and clear conclusions are difficult to draw (Ekvall et al., 2007). Most of these limitations, including the ones discussed in the current thesis, are however common for other systems analysis tools as well. This means that LCA and other system analysis tools should be considered as decision support tools which provide relevant information, support the decision making process and do not substitute the crucial role of a decision maker.

2.2.2 Introduction to life cycle assessment

Life cycle assessment is a tool for evaluating impacts and consumption of resources and was initially developed for evaluating the whole life cycle of products including extraction of resources, production, distribution, use and disposal (i.e. from cradle to grave). The term ‘product’ can include not only product systems but also service systems such as waste management. An ISO standard has been developed for LCA providing a framework, terminology and some methodological choices (ISO 1998, 2000a, 2000b, 2006). Specific requirements are necessary in LCA studies considering that different LCAs would follow the same routines and would thus be more comparable with each other.

According to the ISO standard (ISO, 2006) complete LCA must follow a systematic approach with four iterative steps (Figure 2):

1. **Definition of goal and scope**, attempts to define the extent of the inquiry as well as specify the methods used to conduct it in later steps. One selects a product system, functional units, boundaries, allocation methods, and impact categories during this defining phase.
2. **Life cycle inventory (LCI)**, where involved processes are identified, all relevant data (input and output) are collected and allocation is conducted.
3. **Life cycle impact assessment (LCIA)** is aimed at understanding and evaluating the magnitude and significance of the potential environmental impacts. This includes:

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

3.2 Classification, assignment of the inventory data to the impact categories.

3.3 Characterisation, quantification of contributions to the chosen impact categories.

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

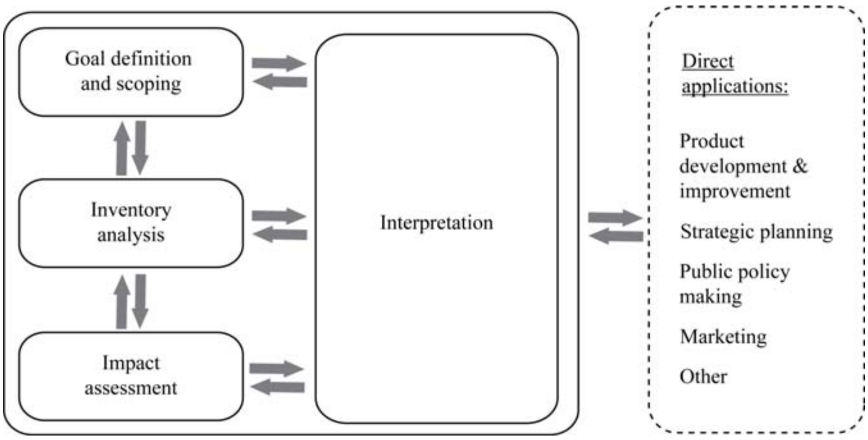


Figure 2. Framework for LCA according to ISO 14040

In addition to the mandatory elements there are optional elements and information which can be included in a Life Cycle Impact Assessment (LCIA). Weighting may be included to convert and possibly aggregate indicator results across impact categories, resulting in a single result. The most controversial issue (and thus a limitation) is the inherent subjectivity in valuation steps, where different environmental impacts could be sometimes weighed against each other. The prevailing approaches for valuation – panel methods, distance-to-target and economic valuation methods – are all subjective since they lack solid scientific backing.

Normalisation is another optional element whereby the magnitudes of the impacts are related to reference values, e.g. total contribution to an impact category by nation.

In principle, LCA attempts to model all important types of environmental impacts of the product system. In reality LCA will often be limited to the environmental impacts, which can be quantified using existing methodologies. For example, due to incomplete data and lack of consensus on assessment methodology, the impacts of toxic chemical emissions and land use are poorly represented in many LCA models (Reap et al., 2008). Typically most LCAs include global warming, acidification and eutrophication.

LCA can be made on different levels depending on the need in the given decision support. The difference between the levels is related to the effort invested in data collection and calculations and thereby the detailing, the thoroughness and the precision obtained.

Three main levels of LCA applications can be as follows (Wenzel, 1998):

1. **Life cycle thinking**, conceptual qualitative assessment of inputs, emissions, etc.
2. **Simplified or screening LCA**, including quantitative information based on readily available data in databases or screenings with limited data collection.
3. **Detailed or full LCA**, including quantitative information and new data inventory.

To save time and resources it is recommended that assessment should be simple in the beginning (screening LCA), and if needed, a more detailed assessment can be conducted. This is relevant also for waste management planning in Estonia, where the resources for detailed LCA are limited.

LCA was initially developed as a tool for assessment of environmental aspects, but nothing prevents assessment of economic and social aspects. Integration of economic aspects in LCA has recently gained momentum (Rebitzer and Hunkeler, 2004). This is reflected by the development of tools such as Life Cycle Costing (LCC) and Input-Output LCA (IO LCA).

In the course of the development of LCA in the last decade, the economic discipline of IOA (Leontief and Ford, 1970; Leontief, 1986; Leontief et al., 1983) has been re-discovered as a useful source of knowledge in strengthening LCA. It is very difficult to model the whole product system involving the entire supply chain in a detailed way in the conventional process-based LCA. IO LCA is considered to be a practical solution for the long-recognized problem of incomplete systems specification in process-LCA due to boundary cut-offs (Norris, 2002). It uses aggregate sector-level data to define how much environmental impact can be attributed to each sector of the economy and how much each sector purchases from other sectors. Such analysis can account for long chains (for example, building an automobile requires energy, but producing energy requires vehicles, and building those vehicles requires energy, etc.).

Hybrid LCA (a hybrid approach combining both methods) describes approaches to blending data from IO and process-based models (Suh, 2003; Williams, 2004;

Heijungs et al., 2005; Suh and Huppes, 2005). For example, one might use process LCA to capture all the aspects that can be measured within the scope of the study and use IO-LCA to capture the supply chain outside of the system boundary.

Both of these methodologies provide new possibilities for more complete LCA applications to support decision-making. However, the main limitation for the usage of IO-LCA method in Estonia is the poor level of existing national and regional databases on economical and environmental statistics (input-output databases).

2.2.3 Overview of important methodological issues in LCA

Although the LCA has reached a certain level of harmonisation and standardization, there are important methodological choices which could have a significant impact on the results (Reap et al., 2008).

Below some of these methodological issues relevant for the research presented in the thesis, are shortly discussed.

Functional unit

One of the first critical stages of LCA is the definition of an appropriate and specific functional unit. The functional unit is a key element of LCA. It is a measure of the function of the studied system and it provides a reference to which the inputs and outputs can be related. This enables comparison of two essential systems. In comparative LCA studies, the choice of the right functional unit is crucial and can lead to intensive discussions.

The classical difficulty in setting a functional unit is when a system provides more than one function (multi-functionality of products and services in general) (Plepys, 2004), specifically in the case of waste management systems (Björklund et al., 1998; Finnveden, 1999).

System boundaries and allocation

System boundaries determine which unit processes should be included in a LCA study. LCA is based on the material and energy flows over system boundaries. It is of absolute necessity to have well-defined system boundaries, in order to obtain unambiguous results. Usually the system boundaries can be:

- Geographical boundaries. Denote a geographical area where the results of LCA are valid.
- Time boundaries. Denote the time for which LCA results are valid.
- Life cycle boundaries.
- Techno-sphere/bio-sphere boundaries.

A traditional problem in LCA is how to deal with processes or groups with more than one input and/or output, e.g. processes or productions with co-products of economic value (multi-output processes), or waste management where several waste components are treated in the same process with common consumption of raw material and common formation of emissions (multi-input processes). For example, MSW is a mixture of different materials. Different emissions are produced when waste is treated in an incinerator or a landfill. The difficulties lie in how the emissions shall be shared between different input parameters.

There are various ways to implement the allocation problem. Two major approaches are recommended (ISO, 1998):

1. Wherever possible, allocation should be avoided by:
   a. Dividing the unit process to be allocated into two or more sub-processes and collecting the input and output data related to these sub-processes (subdivision),
   b. Expanding the product system to include the additional functions related to the co-products. After expanding the system, subtracting equivalent product from main product inventory (system expansion).
2. Where allocation cannot be avoided, the inputs and outputs of the system should be partitioned between its different products or functions in a way that reflects the underlying physical relationships between them (physical allocation); i.e. they shall reflect the way in which the inputs and outputs are changed by quantitative changes in the products or functions delivered by the system.

The former approach implies the need for more data to be collected, which is always a trade off with time and resources for making the analysis. The latter approach is by nature subjective as often there is no scientific background on what basis the allocation or division should be done.

As allocation is a source of controversy, it is difficult to give short and exact guidelines on the best allocation principle (Finnveden, 2000; Rebitzer and Hunkeler, 2004; Heijungs and Guinee, 2007). However, the use of expanded system boundaries is generally recommended in the waste management related change-oriented LCA studies (Ekvall and Finnveden, 2001; Weidema, 2003). The advantage of avoiding allocation through system expansion is that it makes it possible to model the indirect effects of actions (Ekvall and Finnveden, 2000). A drawback of using system expansion is that the models get larger and more complicated. Several critical assumptions concerning e.g. materials and energy sources being replaced should be answered.

System delimitation

Important developments in the LCA methodology during the last decade have focused on improving the understanding of how market information can provide a transparent procedure for unambiguous delimitation of the described systems - the product life cycles - i.e. what processes to include and what processes to exclude from the systems.

When dealing with system delimitation in LCA modelling, two approaches are referred to (Tillman, 1999; Guinee et al., 2002; Weidema, 2003):

1. Attributional or retrospective approach
2. Change-oriented or consequential approach

Attributional approach usually applies average or supplier-specific data and allocates between co-products by applying allocation factors (Ekvall et al., 2005). On the other hand, consequential approach strives towards modelling the actually affected processes, which implies that marginal suppliers or technologies are affected and co-product allocation is avoided by system expansion (Weidema et al., 1999; Weidema, 2003; Ekvall and Weidema, 2004).

Traditionally in LCA data describing current production processes have been used. Occasionally the average of several processes is used, e.g. electricity data is often calculated as a weighted average of various electricity sources (hydropower, nuclear power, coal-based power, etc.). This is also true for most of the earlier LCA studies carried out in Estonia.

When using LCAs for assessing future plans and strategies (e.g. waste management plans), the actual affected processes, suppliers and technologies (those that reflect the real technological consequences and thereby also the real environmental consequences of the decisions made) should be used in the model rather than weighted-out averages of all technologies (Ekvall and Weidema, 2004). The fundamental issue is that when an LCA identifies e.g. a certain material as being advantageous, a demand for this material is created. In a free market the response to this demand will typically be an increased production by the manufacturer who is the most competitive and is not constrained with respect to the size of the production. Therefore the data to be used in the LCA should be data from this manufacturer and not necessarily data from the present suppliers of the material. Ekvall and Weidema (2004) provide guidelines for identifying such marginal technologies in consequential LCA.

In this thesis the change-oriented or consequential LCA approach is used for analysing waste management system in Estonia.

Choice of appropriate time-frame

The choice of the time frame is important for waste management related LCAs. For example emissions from landfills may prevail for a very long time, often thousands of years or even longer (Sundqvist, 1999). Therefore the potential emissions from landfilling have to be integrated over a certain time period. The choice of the time period can have significant influence on the results for materials that are persistent (e.g. plastics) and for substances which are slowly leaching out, e.g. metals from municipal solid waste and ashes (Finnveden, 2000).

The choice of the time frame is discussed in many surveys (Finnveden et al., 1995; Sundqvist, 1999; Udo de Haes et al., 1999). The principles for determining the length of the time period can be roughly arranged under two time horizons:

1. A short-term period (surveyable time period), usually up to 100 years. It is defined as the time period until the landfill reaches some kind of pseudo-steady-state. For a conventional municipal solid waste landfill this corresponds to the end of methane production phase.
2. A longer period (called hypothetical, infinite time period), which is the period until the landfilled material is completely released to the environment

Data quality

Although the problem of data quality is not intrinsic to the LCA methodology only, it could have a serious impact on the results of a LCA study. Different data sources can give even larger differences in results than different allocation approaches. Reliability of the results of LCA studies strongly depends on the extent to which data quality requirements are met. The following parameters should be taken into account:

- Time-related coverage (old vs. new data)
- Geographical coverage (location-specific)
- Usage of average or marginal data (data resolution, i.e. level of aggregation)

A comprehensive analysis requires the collection of an enormous amount of data. In many cases it is not possible to obtain these data, resulting in simplifications. Knowledge gaps are often filled in by qualified assumptions or by import of data from databases and from other regions. The possible data drawbacks are usually addressed through the process of sensitivity analysis, where the data causing the greatest impact on final results are identified and examined more carefully.

2.2.4 The WAMPS model

As part of the research a screening level LCA model was developed and tested in case studies. This LCA software tool called WAMPS is intended to be applied during the waste management planning process to find optimal solutions and alternative waste treatment technologies for waste management systems. WAMPS presents the environmental and economic consequences of different waste management scenarios in a life cycle perspective. The focus is on environmental consequences, while assessment of economic consequences is more simplified.

This model allows scenario analysis of different waste management systems. It enables decision makers without an in-depth knowledge of LCA methodology to learn how changes in the system affect its environmental and economic impacts. It also allows us to choose the most optimal system solution based on these two aspects.

The model was developed by the Swedish Environmental Research Institute (IVL) and is based on a more detailed LCA model ORWARE (Björklund, 2000; Eriksson et al., 2000; Sundqvist et al., 2002). The author of this thesis contributed to the model calibration and refinement, testing and regional database generation.

Functional unit

The basic functional unit in WAMPS is the amount of waste treated within a specific region. This gives two possibilities to formulate a LCA study:

1. **Different waste management options are compared**, e.g. what are the optimal waste treatment options and technologies for a future waste management system. It is advisable that the studied scenarios handle the same waste amount in the same region.

2. **Study of the development of waste management over time**. In this case development over time is studied. During the studied period the waste amount is expected to change. In this case the waste amount, as well as source separation and choice of treatment/recycling methods may be different in different scenarios.

Sub-models

WAMPS consists of a number of sub-models which may be combined to describe a waste management system for a region or municipality. Each sub-model describes a process in a real waste management system, e.g. landfill, incineration, waste collection and transport, material recycling, composting and anaerobic digestion. Materials turnover is characterised by (1) the supply of waste materials and process chemicals, (2) the output of products and secondary wastes, and (3) emissions to air, water and land. Energy turnover is the use of various energy carriers such as electricity, coal, oil, heat, and recovery of e.g. heat, electricity or biogas from waste treatment processes. The concept set up by the WAMPS model is shown in Figure 3. The solid line defines the boundaries of the waste management core system where wastes are treated and different products are formed.

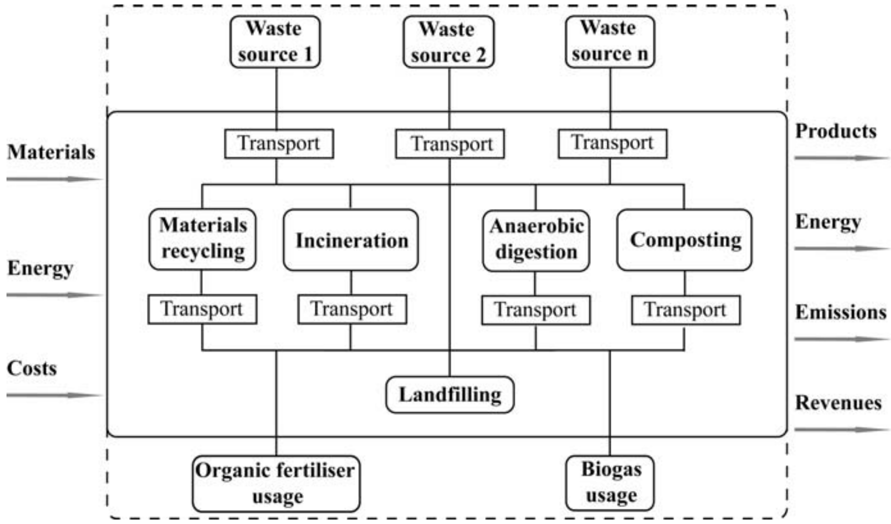


Figure 3. A conceptual model of a complete waste management system comprising different processes described by different sub-models (Sundqvist et al., 2002)

Some of the sub-models of WAMPS are described as follows.

Landfill

A conventional municipal waste landfill that meets the demands of the EU Landfill Directive 1999/31/EC (EC, 1999) is modelled in WAMPS. The landfill model includes emissions to air and water, turnover of energy and gas production as well as costs of the landfill process. Leachate water is treated in a local treatment plant (biological treatment), reducing COD, N and P emissions to water. Landfill is equipped with a landfill gas collection system. There is a possibility to adjust the efficiency of landfill gas recovery as well as the amount of gas used for energy (electricity and/or heat) production in the model. There is one important difference between landfilling and most of the other treatment processes. Waste landfilled today will cause emissions during a very long period of time. A surveyable time period of about 100 years, i.e. the time until the most active processes in the landfill have ended and the landfill has reached a pseudo steady-state, is used in modelling the emissions from landfill. This technosphere boundary was chosen because of the focus of the studied environmental impact categories. The landfill model is described in detail in Fliedner (1999), Sundqvist (1999).

Incineration

An incineration plant (mass-burn technology with combined heat and electricity production) is a Best Available Technique (BAT) compliant incineration plant, meeting the EU Waste Incineration Directive 2000/76/EC (EC, 2000b) requirements with a good margin. The plant has an advanced flue gas cleaning system, including a flue gas condensation step where energy is recovered by heating pumps. The incineration model includes turnover of energy, emissions to air and water and costs related to the incineration process. Effects of landfilling of ashes and slag generated in the process are also included. The emission data is based on emissions from a real incineration plant in Sweden. The original incineration model is described in Sundqvist (1999, 2002).

Composting

The composting model has four sub-models: home composting, open windrow composting, close windrow composting and reactor composting. These are the most likely approaches feasible in the Estonian conditions. The degradation process is the same for all the composting types except for the degradation speed. The different compost sub-models generate the same composition of the compost product when processing the same type of waste. The models include energy consumption, emissions to air and water, transports linked to the process, saved amount of Nitrogen (N)- and Phosphorous (P)- fertiliser and costs of the process. Input data in WAMPS to the composting model represent the share of degradable waste in waste treated with biological methods. The composting models are described in Sonesson (1996, 1998).

Anaerobic digestion

The model for anaerobic digestion includes the production and use of biogas. The model also includes energy consumption, emissions to air and water, transports linked to the process, effects of the spreading of the digestate, saved amount of N- and P- fertiliser, and costs of the process. There are two options for the use of biogas: either for production of energy or use as vehicle fuel (for buses). Input data to the model is the amount of waste treated with anaerobic digestion of the total amount of waste treated with biological methods. The type of energy source replaced is also filled in (same as the replaced energy source from incineration and/or landfill gas but with the possibility to have different replaced sources for heat and electricity). The original anaerobic digestion model has been described in Dalemo et al. (1997).

Collection and transport

There are various types of sub-models describing vehicles for different modes of transport. For collection of waste there are back-packer and front-loader models for waste collection vehicles. For the transport of primary and secondary waste there are three sub-models for ordinary truck as well as truck and trailer. They have the same structure while the parameters differ. Data on average load, average speed, etc. are used as input in all transport sub-models. Collection includes the direct collection of waste at the collection points (at houses, or at various collection stations) plus the transport to a near waste site for storage or treatment or for transfer to bigger recovery facilities. The model for collection includes energy consumption, emissions to air and water and costs of the collection system. The collection model is valid for vehicles running on diesel. The transport sub-model is described in Dalemo et al. (1997).

Recycling

There are several recycling sub-models, each one modelling the recycling of different materials. The recycling sub-models include energy consumption, emissions to air and water and costs of the process. The model also includes saved emissions and saved energy consumption from the corresponding processes in the background system. Input data is the share of each fraction sorted out from the ground composition.

The studied system

The total system analysed in WAMPS consists of the following parts (Figure 4):

- Waste management system with different sub-models, i.e. the core system of the waste management system
- Compensatory or background systems, i.e. systems to which the waste management model has a relation to (e.g. energy generation)
- Key flows of material and energy connected to up-stream and down-stream systems

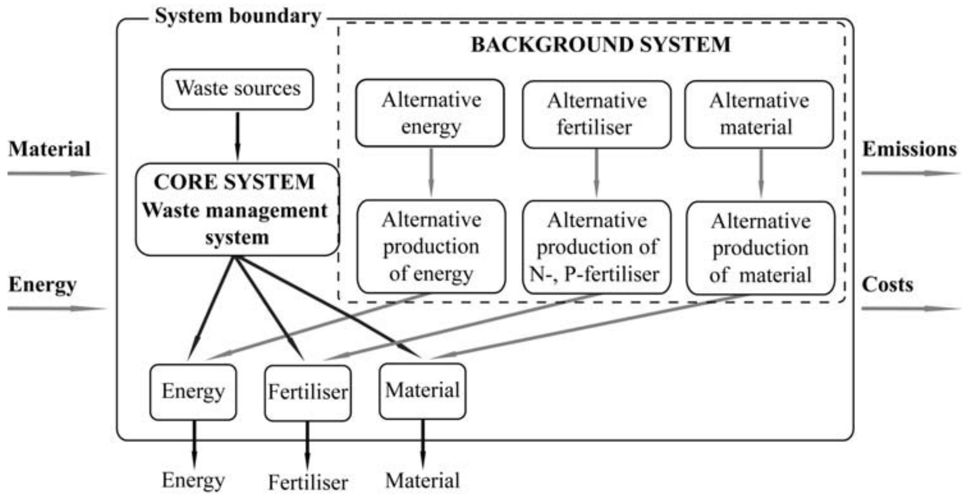


Figure 4. The studied system: the core waste management system and background systems (Sundqvist et al., 2002)

The core system describes the physical waste management system studied, including collection, treatment, and final disposal of waste generated within a defined geographical area and time-space. Up- and downstream processes are defined as the processes that impact the core system when using materials and energy. Upstream processes include for example waste sources and electricity and fuel generation, and downstream processes characterise the use of organic fertiliser and energy utilisation.

The allocation problem in WAMPS is addressed by expanding the system boundaries to include the so-called compensatory or background processes. This method was chosen as the least controversial and more adequate for a more objective analysis. The background systems outside the core waste management system enable a quantitative comparison of environmental, economic and energy parameters between the use of waste as a raw material and the use of virgin raw material. Background systems also have up-stream and down-stream processes.

WAMPS compares a core waste management system with a background system. A waste management system can produce different products depending on the choice of waste treatment and recovery options: heat, steam, electricity, vehicle fuel (biogas), compost, paper, plastics, metals, etc. In a background system similar products are produced from virgin origin. When a product is produced from waste, it substitutes a product in the background system. Each waste product has an alternative in the background system with a virgin raw material source and a production process that is included in the model. In WAMPS a number of recovery options are compared with the background system and the potentially 'saved emissions' are assessed. The background system consists of heat production (alternative to waste incineration and combustion of biogas and landfill gas), electricity production (alternative to waste incineration and combustion of biogas and landfill gas), vehicle fuel production (alternative to biogas), fertilizer production (alternative to compost and digestate) and production of materials (plastic, newsprint, paper packages, glass, steel, aluminium, etc.). The parameters and data of these systems were chosen based on how much they are characteristic to Estonia, which is important to keep in mind in order to have more adequate results. All these products from the background system can also be produced by various waste recovery methods.

The net emissions from the studied system are calculated according to:

$$E_{net} = E_{waste} - E_{Background}$$

E_{net}: Net emission (tonnes/year or kg/year)

E_{waste}: Emission from a waste process that produces a certain amount of product/energy (tonnes/year or kg/year).

$E_{Background}$: Emission from the same amount of alternative virgin production in the background system (tonnes/year or kg/year).

This calculation can give negative net emissions. This means for example that waste incineration could give lower emissions than the corresponding energy production in the background system.

Environmental impact categories

The substance flow analysis carried out in WAMPS generates data on emissions from the system. WAMPS primarily calculates emissions of specific compounds or elements, such as CO_2, SO_2, NO_X, etc. from each sub-model. Emissions from all sub-models are summed. Different emissions are then classified into environmental impact categories. WAMPS is designed to bring a more global dimension to waste management planning. Therefore, it focuses on four main global/regional environmental impact categories that are the most relevant for waste management: global warming, eutrophication, acidification and photo-oxidant formation. The spectrum of the chosen impact categories was deemed sufficient for the purpose of this research. Uncertainties for these impact categories are also assumed to be lower than in other possible impact categories (e.g. toxicological impact categories) (Finnveden and Lindfors, 1998; Reap et al., 2008).

Within each impact category contributions of each emission are weighed together with characterisation factors (Sundqvist et al., 2002):

- **Global warming:** all emissions are expressed as CO_2-equivalents: 1 kg of methane (CH_4) is equal to 21 kg of fossil carbon dioxide (CO_2), and 1 kg of nitrous oxide (N_2O) is equal to 310 kg of fossil CO_2.
- **Eutrophication:** all emissions are expressed as oxygen demand (COD): 1 kg of NO_X is equal to 6 kg of COD, 1 kg of NH_3 is equal to 16 kg COD, 1 kg of NO_3^- is equal to 4.4 kg COD and 1 kg of phosphorus (P) is equal to 140 kg COD.
- **Acidification:** all emissions are expressed as SO_2-equivalents: 1 kg NO_X is equal to 0.7 kg SO_2, 1 kg of NH_3 is equal to 1.88 kg SO_2, and 1 kg of HCl is equal to 0.88 kg of $SO_{2.}$
- **Photooxidant formation:** Photooxidant formers have been divided into volatile organic compounds (VOC) and NO_X. CH_4 is included in the VOC but with a relatively low factor: 1 kg of CH_4 is equal to 0.006 kg ethane (C_2H_4), CO is equal to 0.03 kg C_2H_4 and NMVOC (non-methane volatile organic compounds; used as a summary parameter) is equal to 0.416 kg of $C_2H_{4.}$

Environmental cost

The model enables us to calculate the environmental or external costs (broader costs for environmental damage) of the studied waste management systems by aggregating the results of environmental impact assessment using monetary weightings for the emissions. The ORWARE weighting method (based on marginal damage function of emissions and willingness-to-pay estimations from ECON, 1995, except for eutrophication, which is based on Gren, 1993) is used for the calculation of environmental or external costs of environmental impacts in WAMPS (Sundqvist et al., 2002). In addition to the above mentioned environmental impact categories, the emissions of heavy metals (lead, cadmium and mercury) are taken into account when calculating the environmental costs.

This method is not suitable for calculating the total environmental costs of the studied waste management system. However, it can be used when comparing different waste management options.

Economic costs

The economic cost information of the alternative waste management options is usually the most important issue for local decision makers. The total economic impact of the waste management system in the WAMPS model is calculated on a more general level. It covers the following items:

- Investment costs (excluding transport and recycling) periodised by an annuity factor depending on depreciation time (varies for different kinds of equipment and structures) and interest rate
- Operational costs
- Revenues
- Taxes and charges

The sum of these costs gives the total annual cost from which a cost per ton of waste is calculated.

3 VALIDITY OF THE WASTE HIERARCHY FOR ESTONIAN CONTEXT

In the new EU Member States, including Estonia, the municipal waste management is under rapid change. Decision-makers on a local level face a complex task when planning a future waste management system. A wide variety of technological options, increasingly diverse waste fractions, environmental restrictions and EU-wide recovery targets mean that a lot of considerations have to be made. The solutions to municipal waste management should not only be environmentally adequate but also cost efficient and socially accepted. Although authorities in Estonia have had some access to waste management related economic cost information, so far they have lacked comparable environmental information to assess the environmental aspects of alternative waste management options.

The waste hierarchy is suggested as a guiding principle by the EU. The fact that landfilling is the worst option for MSW treatment is generally accepted. However, choice of the most optimal waste management solution has been under heavy discussion among local decision-makers in Estonia. Especially preference between incineration and recycling is often discussed. Another open question is where to place biological treatment such as composting in the waste hierarchy.

This chapter is based on Papers I, II and III and aims to test the validity of the waste hierarchy by evaluating the environmental and economic performance of a number of MSW management scenarios and treatment options in a life-cycle perspective (WAMPS model was used).

The research is outlined as the results of illustrative case studies carried out in Estonia. In the first case study alternative future waste management scenarios for a specific region (large urban region - Tallinn and surrounding municipalities) were compared. In addition, several sensitivity scenarios were developed to study the influence of the choice of different energy sources in the background system (electricity and district heat). The second case study complements the first one by analysing the two most feasible waste recovery options (incineration with energy recovery and composting) in terms of their possible contribution to climate change. Although in both cases the focus is on Estonian conditions, it is expected that the results are applicable also for other similar regions in the Baltic States.

The results of the case studies have been published in several research reports (Moora, 2007a, 2007b). They were also used in the development of a waste plan for Tallinn municipality 2006-2011 and for elaboration of the recently adopted National Waste Plan for 2008-2013.

3.1 Waste definition and fractions studied

According to the European Waste Catalogue (Commission Decision 2000/532/EC) (EC, 2000a), municipal waste is defined as household waste and similar commercial, industrial and institutional wastes including separately collected fractions. In this research the same basic definition for MSW is used.

MSW composition as well as the classifications used to collect data on waste fractions varies widely among regions and countries. Although the official data on MSW in Estonia have improved during the recent years, they are still unreliable and need to be analysed before they can be used for waste management planning purposes. The data on municipal waste generation and fractional composition in this research are based on the results of earlier studies (Tallinn, 2005) and corrected statistical data which were verified by a country-wide waste sorting survey of MSW in 2007/2008 (Moora and Jürmann, 2008a, 2008b) (see the summary of results in Appendix 1).

The MSW sorting survey that was carried out by the author in the framework of this research, was based on the methodologies described in international standards and guidelines (Nordtest, 1995; ASTM, 2003). The study was conducted during four seasons (October 2007 - August 2008). The composition of mixed municipal waste was studied in 8 areas covering a large city (Tallinn), small towns and rural areas. Loads of ordinary waste transport vehicles were used for sampling and sorting. Sorting was done manually into 33 sub-categories (11 primary fractions). In addition, the efficiency of a packaging collection system was studied by analysing the selected packaging collection containers and separately collected packaging waste fractions in sorting facilities. One of the objectives of the study was to develop a standardised procedure for waste composition analysis in Estonia.

The fractional division of MSW in sorting studies differs widely (Dahlen and Lagerkvist, 2008). Based on the recommendations of international guidelines (Nordtest, 1995; ASTM, 2003) and the experience gained from the sorting survey the following 11 primary municipal waste fractions were analysed:

1. Plastic (subdivided into mixed plastic and plastic packaging - soft and hard fraction)
2. Glass (mixed glass and glass packaging)
3. Metals (steel and aluminium, separately steel and aluminium packaging)
4. Paper and cardboard (mixed paper, newsprint, paper and cardboard packaging)
5. Organic waste (mixed organic waste, garden waste and kitchen/food waste)
6. Wood (wooden furniture, packaging, etc.)
7. Hazardous household waste (small chemical waste, medical waste, etc.)

8. Waste electrical and electronic equipment (batteries, small WEEE except large items such as refrigerators, washing machines, etc.)
9. Textile (textiles and leather, carpeting, etc.)
10. Other combustible waste (unclassified combustibles, rubber, miscellaneous combustibles)
11. Non-combustible waste (unclassified non-combustibles, ceramics, ashes, soil, mineral waste, etc.)

The studied MSW in this thesis includes most of the packaging waste (except transport packaging such as metal barrels, wooden cargo pallets, etc.).

Biodegradable municipal waste (BMW) is a mixture of waste materials that decompose in aerobic or anaerobic conditions. In this study BMW contains the following waste fractions: paper and cardboard, wood, biodegradable part of textile waste, organic waste (garden and food waste).

Specific issues related to the development of MSW generation rates and composition are described in more detail in Chapter 4.1.

3.2 Waste management options considered

Various options are available for the treatment of either the whole MSW or materials separated from it for recovery/recycling or pre-treatment prior to disposal. The waste management options considered in this research are outlined as follows (Figure 5):

- Landfilling
- Incineration in terms of mass-burn combustion of mixed MSW with energy recovery
- Material recycling
- Biological recycling in terms of composting

These waste management options were chosen on the basis that they reflect the existing practices (landfilling and material recycling) as well as the current developments in municipal waste management (incineration and composting). The current research does not study anaerobic digestion with energy recovery. Anaerobic digestion is an emerging option for the treatment of MBW in Europe. However, this is still a relatively expensive solution which is still used mainly for the treatment of large organic waste streams from agriculture, waste water treatment, etc.

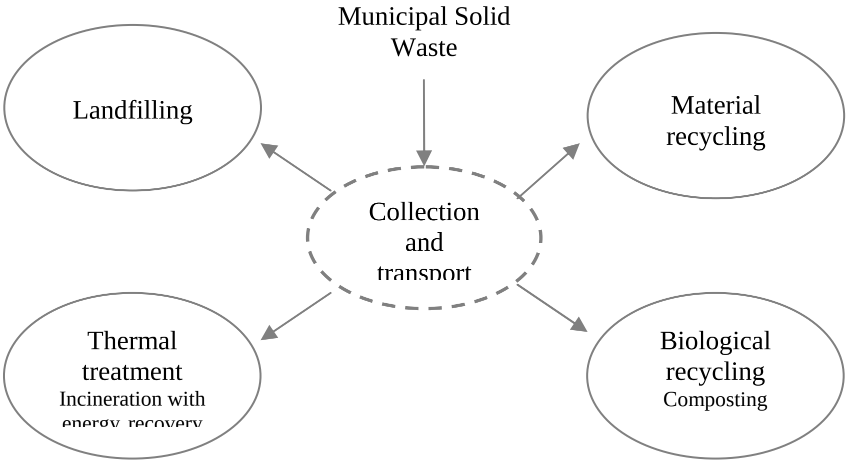


Figure 5. Waste management options considered

There are a number of aspects that could affect the development of a waste management system. A short overview of the analysed waste management options/technologies and the related aspects (including specific legal requirements) is given below.

Landfilling

Landfilling has been the predominant treatment method for MSW in Estonia. Approximately 76% of MSW was landfilled in 2007 (see Figure 6). However, in recent years Estonia has achieved a relatively high share of MSW recovery (mainly due to the high recycling rate of packaging waste). From strategic and environmental perspectives, landfilling is the least desirable option for the management of MSW. Discussion about phasing out a significant share of landfill-deposited waste in Europe is driven mainly by the climate change dialogue. Landfills release significant amounts of greenhouse gases, such as methane (CH_4) whose global warming potential is about 21 times higher than that of CO_2. Another major emission of landfill is leachate water, polluted by both organic compounds and metals. In Estonia like in many other European countries, public opposition to the establishment and operation of landfills has become an increasingly important issue that municipalities and waste management companies constantly need to address.

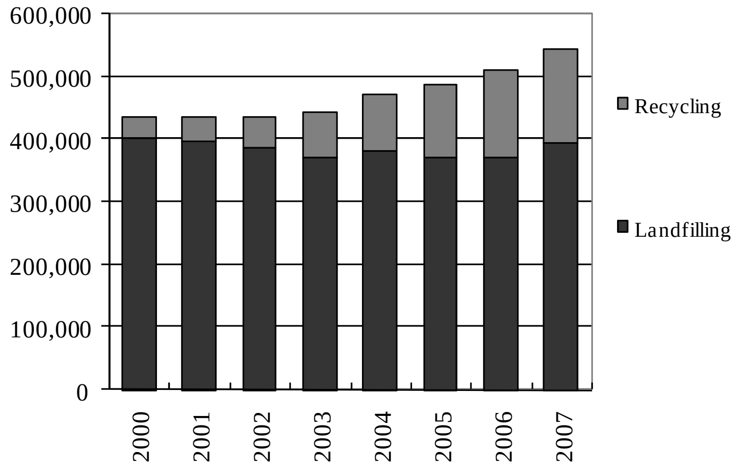


Figure 6. Municipal waste generation and the share of landfilling in Estonia (tonnes)

All old landfills for depositing MSW in Estonia will be closed by July 16, 2009. After that only new landfills that comply with the technical requirements of the EU Landfill Directive 99/31/EC (EC, 1999) will remain operational. The Landfill Directive, which is transposed into Estonian legislation through the Waste Act (2004) and through the Regulation of the Minister of Environment of April 29, 2004, no. 38, sets provisions covering location of landfills, technical and engineering requirements for aspects such as water control and leachate management, protection of soil and water and landfill gas emissions control. With the Council decision 2003/33/EC the Landfill Guideline (EC, 2003) is implemented by establishing criteria and procedures for the acceptance of waste in landfills.

The experience with new landfills in Estonia indicates that there are problems in assuring compliance with technical requirements (e.g. water control, leachate treatment and gas collection systems). Especially leachate treatment has necessitated additional efforts in terms of technology and related financial resources, since the initially designed treatment capacity has been insufficient for the load of leachate.

The Landfill Directive defines also progressive targets for the diversion of biodegradable fraction of MSW away from landfill. The main implication of this approach is that there is an absolute limit placed on the quantity of biodegradable municipal waste (in tonnes), allowed to be landfilled by specific dates. According to the Directive landfilling of biodegradable municipal waste has to be reduced compared to the baseline of 1995 in the following ways - down to 75% by 2006, down to 50% by 2009, and down to 35% by 2016. Estonia has transposed the target dates to take place four years later than those prescribed in the Landfill Directive.

However, the targets are stricter than those in the Landfill Directive. According to the Estonian Waste Act the proportion of biodegradable waste out of the total amount by weight of municipal waste deposited in landfills must be reduced to:

- 45% by July 16, 2010
- 30% by July 16, 2013
- 20% by July 16, 2020

Based on the results of the sorting study carried out in the framework of this research, it is likely that the first target (in 2010) for diverting biodegradable waste from landfill will be met. However, the next targets in 2013 and especially in 2020 will be very challenging for Estonia (Figure 7).

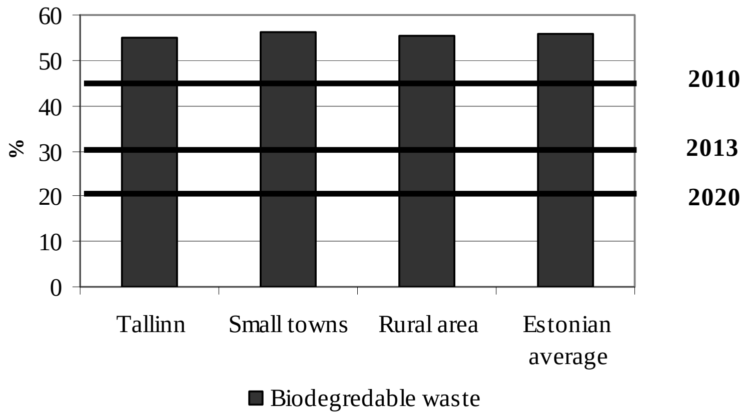


Figure 7. The share of MBW in landfilled MSW in Estonia (Moora and Jürmann, 2008a)

Incineration

A modern waste incinerator is a complex industrial process plant involving several process steps to optimise energy production and to minimise unwanted emissions. There are three main incineration technologies: (1) mass-burn, (2) fluidised bed, and (3) refuse-derived fuel incineration. The main differences between these technologies are the quality of input waste and the way it is incinerated.

In this research mass-burn technology is chosen as probably the most suitable large-scale technology for Estonian conditions. The key characteristic of mass-burn technology is the incineration of mixed waste with little or no pre-treatment (size reduction or removal of non-combustibles). Unsorted waste is transferred through the furnace on a series of moving grids with air flow supplied from underneath and over the waste. The grids shake the waste to ensure that there is sufficient air-circulation for efficient combustion. This technology has proven to tolerate significant fluctuation in waste composition and sizes, which generally has a relatively low average gross calorific value, in the range of 9-10 GJ/tonne – about one third of that of coal or plastic (Smith et al., 2001).

The main pollutants of concern are dioxins, acid gases, nitrogen oxides, heavy metals and particulates. These are present in bottom ash, fly ash and combustion gases, although flue gas cleaning reduces pollutant emissions to the air to a large extent. The main residue from incineration is bottom ash (volume-reduced inorganic ash), which could be disposed of in an ordinary landfill. Fly ash can contain enough dioxins and metals to necessitate treatment as hazardous waste.

There is no experience of mass-burning of MSW in Estonia. However, the discussion of where and when to build a waste incinerator has recently increased. There are two competing project plans to build a waste incinerator in Tallinn, the capital of Estonia. The interest in waste incineration with energy recovery is based on the consideration that it provides additional revenue from electricity and heat generation, reducing at the same time the total costs of waste treatment. With the increasing fossil fuel prices, the interest in energy recovery from waste is increasing. Many bigger cities in Estonia have a well developed district heating system which gives a relatively good position for MSW incineration in combined heat and power plants (CHP).

The results of the sorting study indicate that landfilled MSW in Estonia has a relatively high calorific value due to a high share of combustible materials such as plastic and paper. However, the limiting factor for MSW incineration is related to the higher recycling targets in the future. The share of MSW which can be sent to incineration by 2020 cannot exceed 50% of the total MSW generated (EC, 2008). Future estimations of waste generation (see Chapter 4.1.2) show that waste incineration capacity in Estonia is limited. The maximum amount of MSW which could be sent to incineration is approximately 300,000 tonnes per year. To avoid excess investment in incineration capacity and the consequential technological lock-in, it would be reasonable to build only one large-scale incineration plant in Estonia.

All waste incinerators shall comply with the requirements of the EU Waste Incineration Directive 2000/76/EC (EC, 2000b). The Waste Incineration Directive has been transposed into Estonian legislation through the Waste Act (2004) and through the Regulation of the Minister of Environment of June 4, 2004, no. 66.

Material recycling

Typical recyclable waste fractions in MSW (for material recycling) are plastic, glass, metals, paper and cardboard. Various options are available for the collection and treatment of recyclable materials. Two main types of waste management systems are used, depending on whether a mixed MSW or source separation of various waste fractions is undertaken. In this research source separation followed by collection (kerb-side and/or bring systems) and treatment/recycling facility is used in case studies. This approach is essential to ensure high quality of the collected recyclable materials (especially for paper and plastics) and therefore it should be preferred to other options based on the treatment of mixed MSW.

The possibilities to recycle waste materials in Estonia are limited. All metal waste and most of the collected paper and cardboard as well as plastic waste are sent for recycling outside Estonia. Glass (packaging) is recycled mainly in Estonia.

The main driver for the development of a separate waste collection and recycling system is the EU Directive on Packaging and Packaging Waste 94/62/EC (EC, 1994), which has been transposed into Estonian legislation through the Packaging Act (2004) and the Packaging Excise Duty Act (1996). The Packaging Act sets targets for the recovery and recycling of packaging waste in accordance with the Packaging Directive. Accordingly, at least 55% of the total mass of packaging waste a year shall be recycled. Separate recycling targets per material are as follows:

- 70% of glass
- 60% of paper and cardboard
- 60% of metal
- 45% of plastic

The packaging collection and recovery system, organised by producer responsibility organisations, is functioning relatively well, especially in bigger cities and towns. In recent years the minimum recovery target (50%) of the packaging waste has been met in Estonia. The recycling rate of different waste fractions can be influenced by the fluctuation of market prices of recyclable materials. The high share of impurities in the source separated waste is another limiting factor of an efficient recycling system (see also Chapter 4.3.1).

When planning a future municipal waste management system it is important to take into account the revised Waste Framework Directive 2008/98/EC (EC, 2008) which sets new recycling targets to be achieved by the EU Member States by 2020, including recycling rates of 50% for MSW.

Composting

Composting and the related process of anaerobic digestion (not studied in this thesis) are used mainly for treating organic fraction such as food waste and garden waste. Composting can be undertaken with minimal equipment at home in most households with suitable garden space or in a centralised way by collecting and treating organic waste on an industrial scale. In this study the following composting technologies are considered: (1) home composting, (2) open windrow composting, (3) closed windrow composting with forced aeration, and (4) reactor composting. Open windrow composting is usually suited to garden waste composting. Organic waste streams that contain food waste are usually due to hygienic and quality requirements composted in closed systems. The advantages of closed composting methods include the ability to maintain proper moisture and oxygen levels for the microbial populations to operate at peak efficiency to reduce pathogens, while preventing excess heat or cool, which can crash the system. Closed and aerated systems also facilitate the use of biofilters to treat process air to remove particulates and mitigate odours that is the main local nuisance.

In the course of this research also the share of home composting in Estonia was calculated. Based on earlier studies (Oras, 2002a; Oras, 2002b; Moora, 2007b) it could be assumed that approximately 3% of the total MSW generated in 2005 was composted at home.

There are several regions in Estonia that have started to develop centralised collection of organic waste (e.g. Tallinn City and Central Estonian waste management region). Most of the organic waste is composted in open windrows. Tallinn landfill has gained first experiences in using closed windrow composting technology.

The quality and marketing of the end product (compost) is the most crucial composting issue. First experiences with composting in Estonia show that the main problem is related to the low quality of compost and lacking a market for that product (see also Chapter 4.3.1).

Tackling BMW is a relevant topic in EU waste strategies, due to its implication in the policies of soil conservation and the EU Landfill Directive. In particular, with the aim of meeting the targets provided by the Landfill Directive, all Member States are obliged to set up national strategies for reducing the amount of biodegradable municipal waste to be landfilled.

Since the share of organic waste fraction is relatively high in biodegradable municipal waste, there are basically two options to meet the targets: introduction of an intensive organic waste collection and treatment (e.g. composting) system or mass-burn incineration.

3.3 Case 1: Optimal waste management scenario for Tallinn region

3.3.1 Introduction

Tallinn, the capital of Estonia, is the largest city in Estonia. Most of its population live in the city and “suburban” area. Tallinn region (Tallinn city and surrounding municipalities, Harju County) was chosen for analysis because it represents a characteristic large city area where the majority of municipal waste (50%) in Estonia is generated. In 2005, the total amount of MSW generated in the region amounted to approximately 232,000 tonnes per year (Table 1). The composition of MSW generated in the region is presented in Figure 8.

Table 1. *Waste management situation in 2005*

	Tallinn region
Population	521,000
MSW generated (t)	232,000
MSW generated (kg/cap)	445
Share of biodegradable waste of total MSW	66%
Share of packaging waste of total MSW	26%
Number of households	215,000
Number of detached houses in city area	35,000
Number of detached houses in rural areas	13,000

Approximately 80% of MSW was landfilled in 2005. Tallinn landfill is a modern landfill that meets all the EU requirements and has been operational since 2003. Separate waste collection and recycling are more developed in the Tallinn region than in other areas in Estonia. Packaging and waste paper collection network is relatively well developed. Tallinn has also started to introduce a separate collection system for organic waste. The city of Tallinn has large dwelling areas with district heating. A plan to build an incineration plant has recently come under discussion in Tallinn.

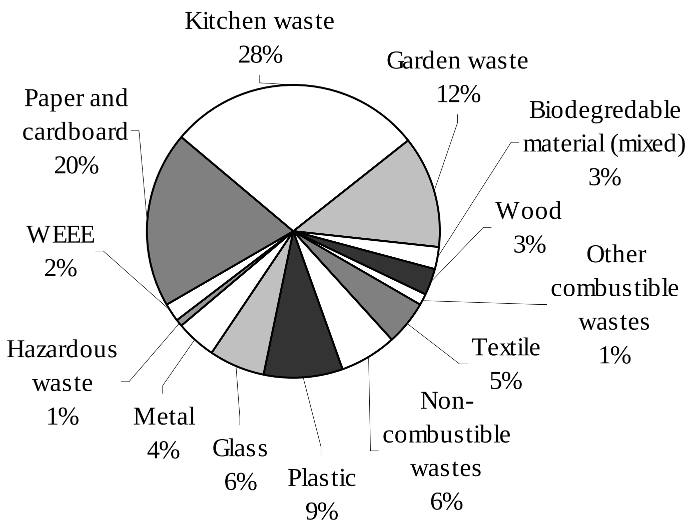


Figure 8. Waste composition in Tallinn region (2005)

3.3.2 Scenarios and major assumptions

A base scenario and five alternative waste management scenarios were developed and analysed. The base scenario describes the waste management infrastructure in 2005. Other scenarios analyse alternative waste treatment systems and their environmental impacts and economic costs. Two of the alternative waste management scenarios (scenario 1 and 2) helped to quantify the potential environmental and economic benefits and trade-offs related to subsequent compliance with the legal requirements. These scenarios project the waste management situation in 2013. Both scenarios are in compliance with the requirements and recycling targets of the following legal acts:

1) Packaging Act (Packaging Directive 2004/12/EC) – minimum packaging recycling target 55% for packaging waste.
2) Waste Act (Landfill Directive 1999/31/EC) - target amount of biodegradable municipal waste going to landfills must be reduced to 30% by 2013.

The additional three scenarios (scenario 3, 4 and 5) consider more intensive recycling, incineration and composting (respectively). The aim of these scenarios is to compare the environmental and economic performance of these waste recovery technologies in the light of the waste hierarchy. The high waste recovery and incineration levels reflect the situation already achieved in some leading EU countries. In addition, two sensitivity scenarios (scenario 6 and 7) were developed to study the influence of the choice of different energy sources in the background system. Table 2 outlines the main considerations of each scenario.

Although the waste amount and compositions are likely to change over time, the same basic waste amount and composition managed within the same system area is used for all the analysed scenarios. Therefore, the results of this case study should not be used as predictions of future emissions or emissions savings.

Table 2. Main characteristics of studied waste management scenarios

Scenario	Material recycling	Composting	Incineration	Landfilling	Compliance with directives
0 Base scenario	15%	5%	-	80%	None
1 Recycling + Incineration	23%	9%	50%	18%	Both Directives
2 Recycling + Composting	23%	34%*	-	43%	Both Directives
3 Intensive Recycling	30%	5%	-	65%	Packaging Directive
4 Intensive Incineration	15%	5%	75%	5%	Landfill Directive
5 Intensive Composting	15%	45%**	-	40%	Landfill Directive
Sensitivity scenarios					
6 Max Incineration (fossil fuel)	-	-	95%	5%	-
7 Max Incineration (bio fuel)	-	-	95%	5%	-

*Compliance with the target for 2013
**Compliance with the target for 2020

When developing the scenarios, the following general assumptions were made:

- In all scenarios it is assumed that 40% of landfill gas is recovered and combusted in a gas engine with 90% efficiency (60% of heat and 40% of electricity). This is calculated based on estimations from the Tallinn landfill and experiences from Swedish landfills (Sundqvist, 1999; Sundqvist et al. 2002).
- It is assumed that maximally 40% of the compost produced is used for agricultural purposes (substituting mineral fertilisers). This is based on the experience from EU countries with more advanced composting systems (Barth, 2006). The rest of the compost is used for landscaping, as a filling

material or topsoil cover in landfills. No energy is generated from the studied composting processes. It is assumed that the amount of organic waste (appr. 10,000 t/y) composted at homes remains the same in all scenarios.

- In all scenarios (except sensitivity scenarios) it is assumed that the energy produced from landfill gas and waste incineration replaces fossil fuel based energy in a background system: the produced electricity replaces oil shale based electricity and the produced heat replaces natural gas based heat used for district heating.
- The incinerator complies with all the EU requirements and it is assumed that the gross efficiency of energy recovery from the incineration process is relatively high - 80%. 80 MW of useful energy is produced, out of which 20% is produced as electricity and 80% as district heat. This assumption is based on the technical description of the planned waste incinerator in Tallinn (Moora, 2007a).
- The financial data used for economic cost calculation (energy and fuel prices, wages, market prices for recyclables, etc.) are based on the data from 2005. Cost data are based on own survey and data from relevant literature (Hogg, 2001, 2002 ;Tsilemou and Panagiotakopoulos, 2006).

Base scenario (scenario 0)

Base scenario characterises the waste management situation in 2005. Most of the MSW was at that time landfilled (80%). However, a reasonable amount of waste was recycled (15% of MSW). Tallinn landfill has a gas recovery system but it was not yet operational in 2005. However, in this study it is assumed that 40% of the landfill gas is recovered. Approximately 11,500 tonnes of organic waste (5% of total MSW) was composted. 80% of the organic waste (mainly garden waste) was composted at home and the rest was composted in a centralised way using open windrow composting.

Recycling scenario with incineration (scenario 1)

In this scenario increased amounts of recyclable materials (mainly packaging, paper, cardboard and metals) are separately collected and recycled to fulfil the recycling targets of the EU Packaging Directive. The recycling level is expected to be 23%. 50% of MSW is incinerated with energy recovery. As incineration is already contributing to the reduction of biodegradable waste, the share of biological recycling is not expected to exceed 9% of the total MSW. Centrally collected kitchen waste is composted using closed windrow composting with forced aeration. Collected garden waste is composted in open windrows. Material recycling and incineration leads to a relatively small amount of rest waste, which is landfilled (18% of the total MSW).

Recycling scenario with composting (scenario 2)

In this scenario the requirement to divert biodegradable waste away from landfilling, is fulfilled by increasing composting to 34% of the total MSW. Approximately 32,000 tonnes of organic waste (mainly kitchen waste) is composted using closed composting method with forced aeration. The rest of the organic waste is composted in open windrows (including home composting). Approximately 43% of the remaining waste is landfilled.

Intensive recycling scenario (scenario 3)

In this scenario maximum recycling rates (85% for cardboard, newspaper and glass, 80% for other recyclables) have been achieved (kerb-side collection of source separated waste fractions). Composting is at the base scenario level.

Intensive incineration scenario (scenario 4)

An increased amount of combustible materials (75%) is incinerated. Recycling and composting are at the base scenario level. A small amount of rest waste (5%) is landfilled.

Intensive composting scenario (scenario 5)

An increased amount of organic waste (45%) is collected (kerb-side collection) and mainly centrally composted using closed windrow with forced aeration and reactor-composting method (without gas collection and energy recovery). Recycling is at the base scenario level.

Sensitivity scenarios (scenario 6 and 7)

Two theoretical/unrealistic waste management scenarios where most of the combustible waste materials (95%) are incinerated, were developed. The aim was to test how a choice between fossil fuel and non-fossil fuel based energy in the background system would influence the environmental performance of waste incineration. In scenario 6 it is assumed that the produced energy is substituting fossil fuels in energy production (oil shale for electricity generation and natural gas for heating). In scenario 7 non-fossil fuel is substituted in the background system (biomass for both electricity and heat).

3.3.3 Results

Environmental impacts

Results of the environmental impact assessment of the analysed scenarios are shown in Figures 9-12. The diagrams show net emissions from the waste management system minus saved emissions in the background system. When the emissions from the waste management system are less than the saved emissions in the background system then net result is negative. Negative results are thus avoided impacts.

Global warming

Landfill is a major source for greenhouse gas emissions (mainly CH_4), despite of the fact that landfill gas is recovered at a high rate. Therefore, the base scenario has the highest contribution to climate change impact.

Material recycling and waste incineration give a negative net impact because fossil fuels are saved when materials are recycled and the produced heat and electricity substitute the heat and electricity produced from natural gas and oil shale in the background system. The more materials are sent to recycling or incineration, the less waste is landfilled, which in turn results in bigger savings of GHG emissions. This double effect, less waste to landfill and substitution of virgin material and fossil fuel in heat and electricity production, makes the recycling and incineration scenarios (scenario 1 and 4) the best options regarding the global warming impact (Figure 9).

In the other scenarios (scenario 2, 3 and 5) the amount of waste landfilled is still rather high and therefore these scenarios have higher contribution to climate change.

Acidification

Acidifying substances are mainly gases such as SO_2 (from e.g. fossil fuel combustion), HCl (from waste incineration), NO_X (from all combustion processes: waste incineration, energy production, engines, etc.) and NH_3 from combusting process and spreading of compost.

Landfilling gives high acidifying emissions due to emissions from landfill gas combustion. Composting gives also high emissions due to NH_3 releases from the composting process (Figure 10). Both recycling and incineration have a positive impact in terms of substituting virgin material and energy in the background system. This makes the two recycling and incineration scenarios (scenarios 1 and 4) the most favourable regarding acidification impact.

Eutrophication

Eutrophication is dominated by various nitrogen and phosphorous emissions to water/soil (N- and P- compounds and COD) and air (NO_X in combustion gases and NH_3 releases from spreading compost).

Landfilling gives the highest eutrophication impact depending on N- and P-compounds in leachate water. Composting causes emissions from spreading the compost. Recycling of materials is better than waste incineration regarding the eutrophication impact (Figure 11).

Photo-oxidant formation

Photooxidants (counted as ethene-equivalents) are dominated by NMVOC (non-methane volatile organic compounds) and CH_4 emissions from landfilling. Landfill also gives the highest NO_X emissions from landfill gas combustion.

Photooxidant formation shows the same tendencies as global warming and acidification, with the corresponding discussion (Figure 12). Also here incineration is the most favorable option.

Sensitivity scenarios

The comparison of the studied environmental impacts and environmental cost of the two sensitivity scenarios (scenarios 6 and 7) shows that if the heat and electricity produced in waste incineration substitute non-fossil fuel (biomass) in the background system, then waste incineration is not the most preferable waste management option any more. This indicates clearly that the environmental ranking of the studied waste-to-energy (WTE) options is very sensitive regarding the assumption of a possible background energy source.

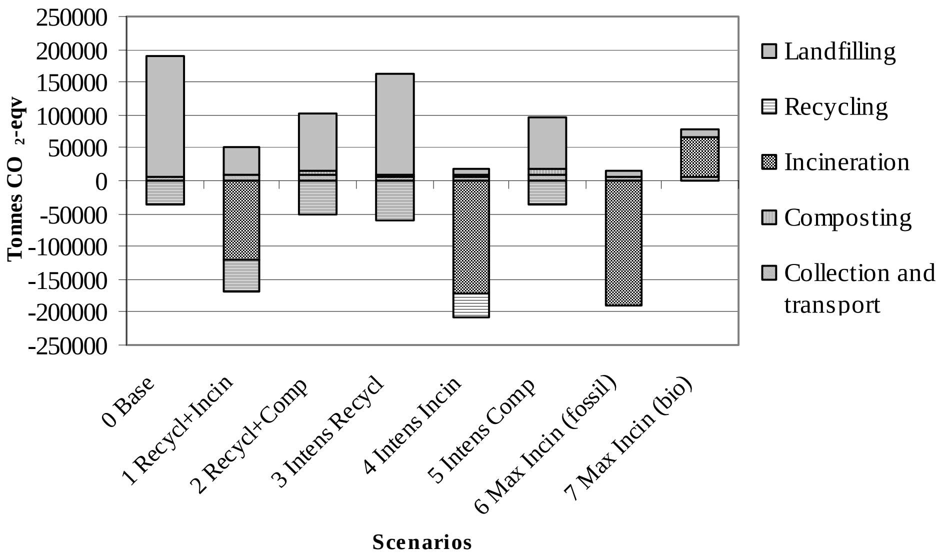


Figure 9. Emissions of greenhouse gases

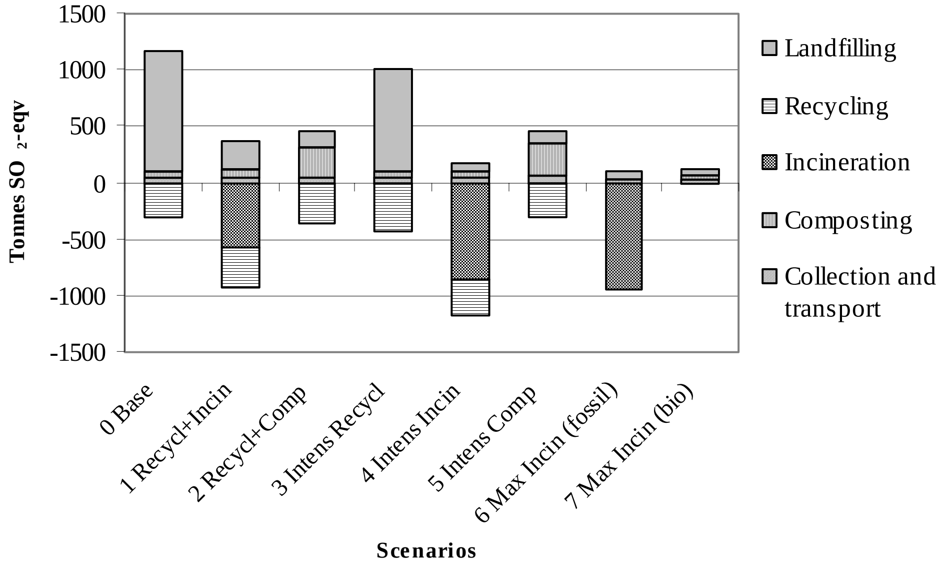


Figure 10. Emissions of acidifying substances

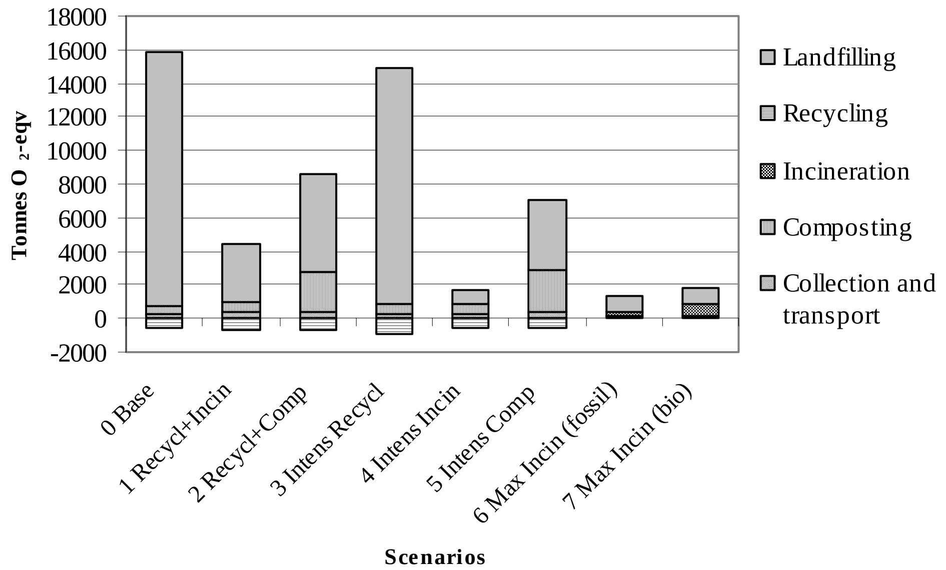


Figure 11. Emissions of eutrophicating substances

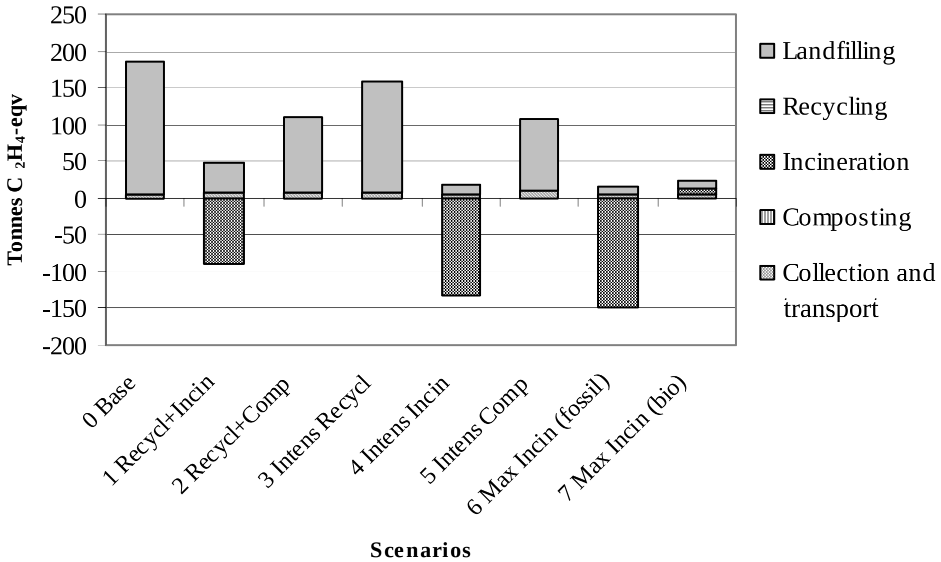


Figure 12. Emissions of photooxidants

Environmental costs

Environmental costs are an aggregation and weighting of the different environmental impacts calculated above. The environmental cost assessment shows the following ranking order of the studied scenarios (Figure 13):

1. All recycling and incineration scenarios (scenarios 1 and 4) create the lowest environmental costs.
2. Intensive composting (scenario 5) and recycling scenario with composting (scenario 2) becomes the next option from environmental cost perspective.
3. The two scenarios of intensive recycling (scenario 3) and base scenario have higher environmental costs than the three scenarios above.
4. When comparing the two sensitivity scenarios, then similar to the results of environmental impact assessment, the incineration scenario with substitution of biomass in the background system (scenario 7) has totally opposite results with relatively high environmental costs.

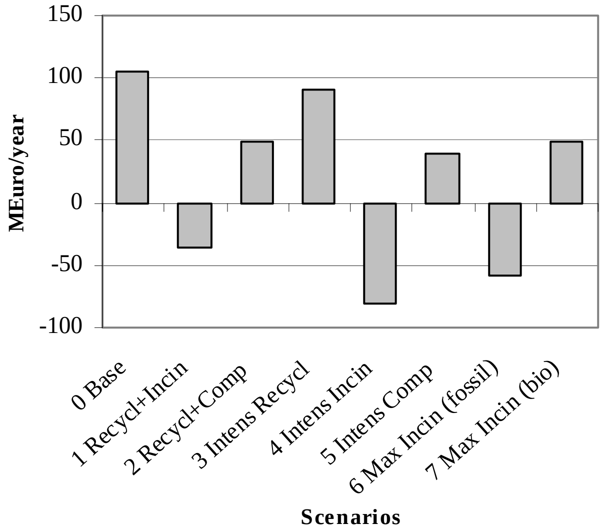


Figure 13. Comparison of environmental costs

Economic cost

Financial costs are more difficult to estimate because the calculations are based on very general cost data with a lot of uncertainties involved. For example, the value of recycled materials and fuel prices is highly volatile due to the fluctuations that are inherent to markets with constrained supplies. The economic cost calculation is based on financial data from 2005. Therefore, the results can be used for a comparative analysis of different waste management scenarios and not for preparing investment plans.

However the following conclusions can be drawn (Figure 14):

1. Investment costs are the highest for incineration, but at the same time incineration gives the highest revenues, which results in the lowest total annual costs (or costs per ton).
2. Recycling requires lower investments but there are large uncertainties in the calculation of related operational costs (collection cost). Total costs of recycling are higher than incineration costs.
3. Investment costs of composting depend on the used technology. At the same time composting gives very low revenues, and therefore the total economic costs for composting scenarios are the highest.

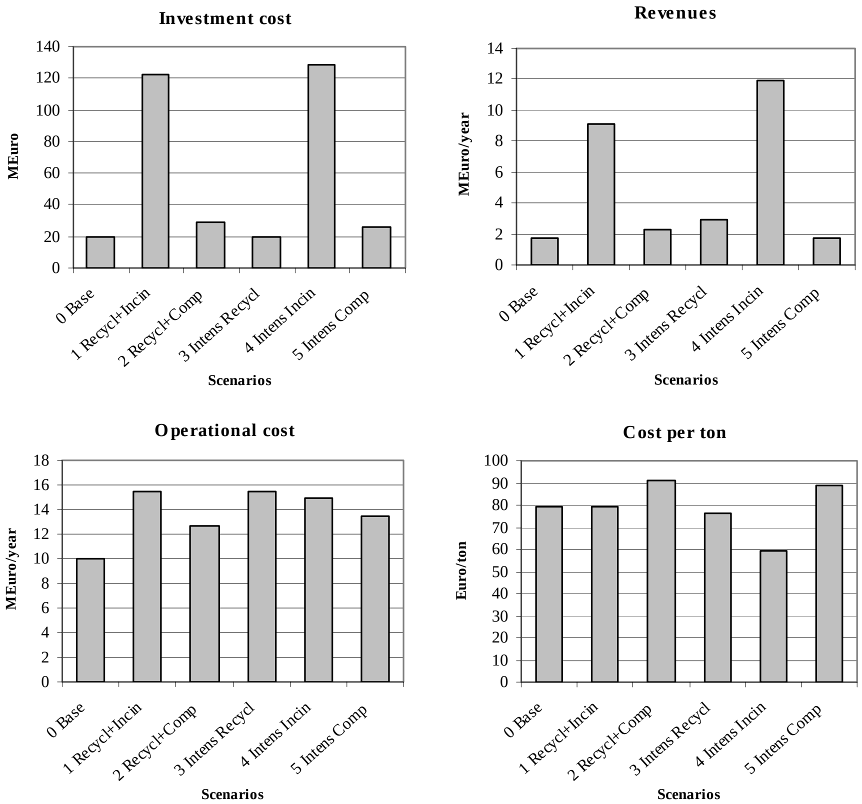


Figure 14. Economic costs of different scenarios

3.4 Case 2: Global warming impact of MSW management in Estonia

3.4.1 Introduction

The scope of this case study is the whole Estonia and the study focuses on the evaluation of the possible reduction of climate change potential by alternative waste management options. Global climate change impact is one of the most significant environmental impacts of waste management. The topic has been recently heavily discussed because waste management has become highly integrated with energy systems.

The contribution made by the waste management sector to climate change is primarily determined by the volume and composition of municipal solid waste as well as the waste management options chosen. The quantity of MSW in Estonia has rapidly risen in line with economic growth and growing consumption. According to specified statistical data, approximately 435,000 tonnes of municipal waste (302 kg per person) were generated in 2000. In recent years (2004-2007) the amount of municipal waste has been growing on the average by 5% per year and reached 540,000 tonnes in 2007 (400 kg per person). Since the number of population is expected to remain roughly the same, economic growth or specifically private final consumption will be the key driving force behind the growing waste volumes in Estonia. The forecast of the municipal waste generation in this case study is based on future estimations of final consumption by households expressed in Purchasing Power Standard (PPS). The growth rate of municipal waste generation is expected to decrease in the coming years due to the slowdown of economic growth. However, in the period 2000-2020 the generation of municipal waste is projected to increase by approximately 58% (Figure 15). In 2020, the generation of municipal waste per person is estimated to be 509 kg (690,000 tonnes). In general, this is in line with the projections made by the EEA - in the new Member States, the generation of MSW is projected to increase by approximately 50% from 2005 to 2020 (ETC/RWM, 2007).

The aim of this case study is to evaluate the climate change impact of the possible future waste management options in Estonia and not to predict the exact GHG emissions generated in the waste sector.

3.4.2 Scenarios

For the GHG emission calculation the waste management situation in 2000 was taken as a starting point or a base scenario. Two waste management scenarios were developed to analyse possible future alternative waste treatment options and their climate protection potential by 2020. Since the pros and cons of waste incineration as a possible MSW management option have been under discussion in Estonia, the incineration-based scenario was compared with the scenario where legal targets are achieved by intensive biological recycling (composting).

The predictions about the future MSW generation presented in the earlier chapter were considered when developing the alternative future scenarios. It is assumed that waste composition remains the same during the studied period.

The basic assumptions are in general the same as in the first case (see Chapter 3.3.2). For both future scenarios it is assumed that all landfills will be equipped with a landfill gas collection system at the latest by 2010 and the landfill gas recovery rate will increase up to 50% by 2020. Before 2010 the collected gas is flared and after 2010 it will be used for electricity and heat production, which is substituting oil shale based electricity and natural gas based heat used for district heating. The energy produced in waste incineration is also substituting the electricity produced from oil shale and heat from natural gas.

Both alternative future scenarios are in compliance with the recycling targets of packaging waste and the landfilling limits for biodegradable waste (Table 3).

Table 3. Main characteristics of studied waste management scenarios

Scenario	Material recycling	Biological recycling (composting)	Incineration	Rest waste (landfilling)
2000 Base scenario	4%	4%	-	92%
2020 Scenario 1 Recycl + Incin	27%	15%	45%	13%
2020 Scenario 2 Recycl + Comp	27%	37%*	-	36%

*Compliance with the target for 2020

Base scenario (scenario 0)

In 2000, waste management in Estonia primarily involved landfilling of MSW (92% of the total MSW). There was no landfill gas collection in landfills at that time. Only a small amount of packaging waste (mainly PET-bottles and cardboard) was collected separately and sent to recycling. There was no centralised collection system for biodegradable waste. Approximately 17,000 tonnes of biodegradable waste (mainly garden waste) were composted in the households (4% of the total MSW). It is assumed that the amount of home composting will remain the same till 2020.

Material recycling with intensive incineration (scenario 1)

Scenario 1 is a projection for 2020, where the dominant option of MSW management in Estonia is incineration. 45% of MSW is incinerated in a mass-burn incineration plant. This assumption is based on the plans to build an incinerator close to the capital of Estonia, Tallinn. The incineration plant is expected to start its operation in 2012. In this scenario an increased amount of packaging materials is separately collected (through bring collection system) and recycled to fulfil the recycling targets of the EU Packaging Directive. As incineration is already contributing to the reduction of biodegradable waste, the share of biological recycling is not expected to exceed 15% of the total MSW. A relatively small amount of rest waste is landfilled (13% of the total MSW).

Material recycling with biological recycling in terms of composting (scenario 2)

Scenario 2 is a projection for 2020, where the legal targets are archived by material and biological recycling. Also in this scenario material recycling is expected to amount to up to 27% of the total MSW. The Landfill Directive requirement to divert biodegradable waste away from landfilling is met by increasing composting to 37% of the total MSW. An increased amount of wet biodegradable waste is composted using additionally centralised reactor-composting method (without gas collection and energy recovery). It is assumed that the remaining waste will be deposited in a landfill.

3.4.3 Results

The results of the scenario analysis in terms of net GHG emissions are shown in Figures 15 and 16. The results of the case study indicate that net GHG emissions from the management of municipal waste in Estonia are projected to decline significantly by 2020 from a peak of around 1.1 million tonnes CO_2-equivalents per year in 2000, largely because of an increased recovery of MSW and the diversion of waste away from landfills.

When comparing the two studied scenarios we can see that the incineration scenario (scenario 1) has a higher climate protection potential than the alternative composting scenario (scenario 2). In scenario 1 where high rates of recycling and incineration with energy recovery are attained, net emissions of CO_2-equivalents are even negative. The reason for the negative net GHG emissions is a relatively low amount of waste sent to landfills as well as a high share of material recycling (avoided primary production of materials) and recovered energy in incineration plants (avoided emissions as a result of substituting heat and electricity produced from natural gas and oil shale in the background system). Incineration gives approx. 75% and recycling almost 25% of the total avoided emissions.

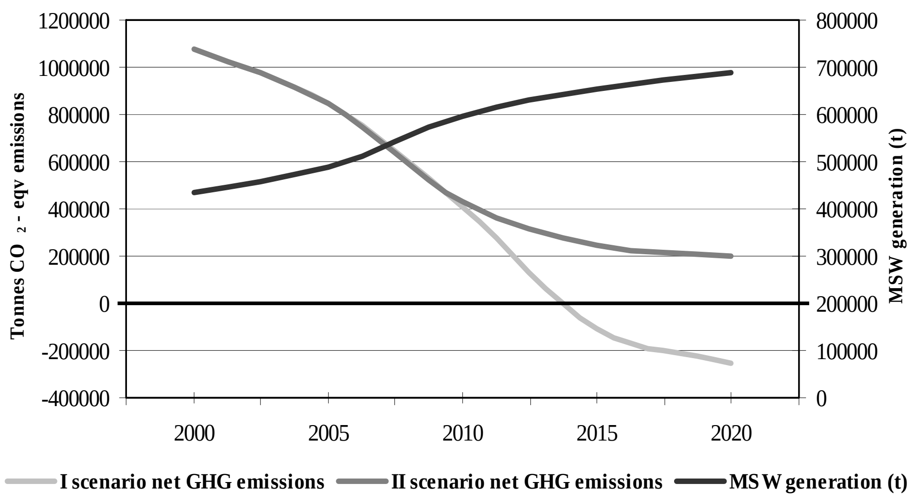


Figure 15. Emissions of net GHG from studied waste management scenarios, 2000-2020

In scenario 2 sources of GHG savings are mainly material recycling and the avoided emissions from landfilling. As in this scenario composting without energy recovery is applied, the net GHG emissions are higher than in scenario 1.

Direct emissions from landfills continue to be a major source of GHG emissions till 2020 despite of the fact that the landfilling rate will decrease significantly and a relatively high share of landfill gas is recovered in both studied scenarios. The GHG emissions from waste collection and transport will increase by 2020 due to increased recycling. In scenario 2 a higher collection rate of biodegradable waste causes slightly more emissions of CO_2-equivalents. In spite of that, the collection and transport of waste accounts for a relatively small amount of the estimated net GHG emissions in both future scenarios.

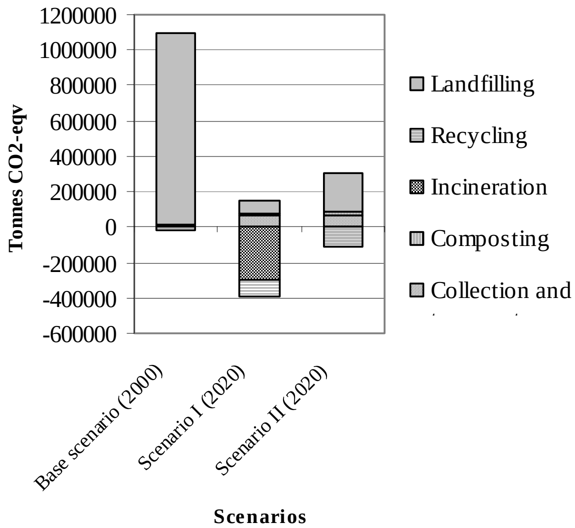


Figure 16. Emissions of GHG from studied waste management practices and scenarios

3.5 Analysis and interpretation

3.5.1 Ranking of waste management options

The LCA modelling results of the case studies presented in this research indicate that in general terms the waste hierarchy is valid as a guiding principle. There are, however, certain local valuations that can lead to exceptions to the traditional order of preferred waste management options.

It is clear that landfilling of MSW is the least preferred waste management option regarding environmental impact. This is valid even if landfill gas is recovered at a high rate. As indicated by the results of case 1, landfilling has also relatively high environmental and economic costs. Therefore, landfilling of MSW should be avoided as far as possible, both because of the environmental impact and because of the low recovery of resources.

Composting of degradable waste (without energy recovery) has hardly any advantages with respect to the environment and energy turnover when being compared to other waste recovery options (such as recycling and incineration). The economic costs of this waste management method are also relatively high. One of the specific drawbacks of composting in Estonia is the low quality of compost and lack of market for that product (see Chapter 4.3.1). However, composting has a potential if landfilling is avoided and incineration or anaerobic digestion are not feasible.

Materials recycling and waste incineration show the best results both in terms of the studied environmental impacts and economic costs. It is important to stress that if high rates of recycling and incineration with energy recovery are attained, net emissions may even become negative, which means that these waste management options can partly offset the emissions that occurred when the products were manufactured from virgin materials and energy was produced from fossil fuel (oil shale and natural gas). This is an important aspect especially in the context of climate change, since in Estonia the electricity produced with current oil shale combustion technology (including old boilers) has the highest climate change impact in terms of CO_2 emissions among other fossil fuels (e.g. heavy oil or coal).

In a systems perspective there are small differences between recycling and incineration. Waste incineration is the best option when the produced energy can be utilised to a high extent and if it is assumed that the produced heat and electricity substitute fossil fuel in the background system. However, when the substituted energy is a non-fossil fuel (e.g. biomass), then waste incineration is not the most preferable waste management option any more.

Material recycling and incineration should not be seen as competing options but as completing options. Intensive recycling and especially incineration of MSW lead to a lower landfilling rate of MSW compared to other possible waste management scenarios. It must be noted that material recycling alone, when meeting the recycling targets of the EU Packaging Directive (EC, 1994), does not reduce significantly the waste amount directed to landfill.

As the case studies show, the type of waste collection system and transport distances have low influence on the total environmental impacts compared to other waste treatment options. Increased collection and transport did not affect the environmental ranking of the studied scenarios. However, the design of collection system (kerb-side collection or bring system, number of collection points and other logistical aspects) could significantly influence the economic cost of the collection system (see Chapter 4.3.2). At the same time, the results of the case studies indicate that long-distance transport of already collected waste to treatment/recovery facilities (if the distance is less that 500 km and the transport is done in an efficient way by truck and trailer) does not significantly increase the economic costs of the waste management system. In Estonia the distances to the existing and possible future waste treatment facilities (e.g. landfills or a planned incineration plant) are relatively short.

In general the environmental and economic hierarchy go hand in hand with the exception that intensive composting system seems to be the most expensive waste management option (see Table 4). From a purely economic cost perspective, incineration provides more income than recycling of waste fractions with a high heating value, depending on the costs of separate collection. Therefore, it could be expected that the incineration of MSW will significantly influence the development of waste management systems, including investments in other waste recovery technologies. If waste incineration starts to affect the fulfilment of waste recovery targets, government intervention (e.g. introducing waste incineration tax) may be necessary.

Table 4. Environmental and economic hierarchy of MSW management based on Estonian conditions

Environmental hierarchy	**Economic hierarchy**
1.-2. Recycling and incineration	1. Incineration
	2. Recycling
3. Composting	3. Landfilling
4. Landfilling	4. Composting

A conclusion may be drawn, taking into account the studied environmental impacts and economic costs, that the scenario where a high share of recyclable waste fractions are sent to material recycling and maximum amount of rest waste is incinerated with energy recovery, is the optimal scenario for Estonia, possibly forming a basis for the development of a future waste management system. It should be noted that in this scenario waste treatment technologies are combined in such a way that the ultimate legal requirements (recovery targets and restrictions for biodegradable waste landfilling) are met, while minimising the overall economic costs.

In general, the results of the case studies concur with those of previous LCA studies in waste management systems (Kirkeby et al., 2006; Koneczny et al., 2007a; Koneczny and Pennington, 2007b) giving preference to the potentials offered by WTE options (waste incineration). This is mainly because of the specific regional assumptions (e.g. oil shale as a electricity source) used.

3.5.2 Limitations

The scope of this research is limited to the chosen waste management options/technologies. Although the studied waste management technologies represent the most likely options for MSW treatment in Estonia, there are also other existing and future technologies (e.g. mechanical biological treatment, anaerobic digestion, etc.) that may have different performances to those described in this study.

There are several sensitive assumptions and data gaps that could influence the ranking of the studied waste management scenarios and treatment options.

The scenarios applied in the case studies, as well as the associated emissions and results, are not actual predictions of future situations, as these can be influenced by changes in waste generation and composition. The data on waste composition reflected the past situation. In reality, it may be expected that waste generation and composition will change over time. How rapid and severe the change will be is an uncertainty. Therefore it is important to study the possible future change in the amount as well as composition of waste and its possible impact on the results.

Another major sensitive regional factor that could significantly influence the ranking of studied scenarios, is the future marginal energy source (both for heat and electricity production).

These above mentioned specific local data caps and assumptions that could influence the results of LCA the most, are discussed in more detail in the following chapter.

It is important to note that not all relevant environmental impacts are included in this study. The focus is on four major environmental impact categories, because uncertainties associated with these impact calculations are somewhat lower than in other categories (Finnveden and Lindfors, 1998; Finnveden, 2000; Reap et al., 2008). There could also be significant uncertainties associated with the financial data used for economic cost calculation. Modern waste management presents a high level of complexity and thus, also other local and regional aspects (e.g. land use, toxicological impacts and social aspects) have to be considered selecting a most optimal waste management scenario for a specific region.

Despite the abovementioned limitations it can be concluded that the results of the research are fairly robust and in general similar to many of the more in-depth LCAs in waste management. However, as discussed above in this thesis, LCA is not the only tool for environmental system analysis of waste management. This method should be used in combination with other tools (such as EIA) to give more comprehensive decision support.

4 DATA GAPS AND CRITICAL ASSUMPTIONS

The applicability of LCA is restricted by certain limitations. As the case studies show there are several data gaps and critical assumptions specific for regional context that could have a larger impact on the final results of the LCA. This indicates for decision-makers that the interpretation of the results of LCA studies should be drawn with caution. Most of those assumptions relate to future situations and developments (e.g. waste generation, future marginal energy source, prices of secondary raw materials and energy). Investments in the waste management area can be long-term. For example, to be economically feasible, a waste incineration plant must have a life span of at least 15 to 20 years. At the same time changes in waste generation and composition as well as energy price could significantly influence both environmental impacts and the economics of waste incineration. It is therefore important to be able to calculate in which scenarios would incineration be economically feasible and what would be the environmental consequences.

In Estonia, there is a need for this type of regional data relevant for system studies but, unfortunately, it is mostly lacking. The collection of specific information is often rather time-consuming and it is difficult to predict future developments. This leads to reliance on assumptions or use of surrogate data from other regions (e.g. average waste composition in the EU). However, it is important that LCA practitioners and the end users of such studies (decision-makers) have sufficient knowledge and understanding of those specific regional characteristics and possible restrictions related to the choices that could influence the results.

This chapter focuses in more detail on the two most crucial local/regional characteristics that could have a significant impact on the results of change-oriented or consequential LCAs in waste management - waste generation and composition and replaced marginal energy in the background system. Other input characteristics, such as the quality of recycled material and financial data are discussed on a general level. The discussion on a possible marginal electricity source is based on findings presented in Paper IV.

The discussion is mainly limited to the Estonian context. However, the described sensitive input characteristics related to waste management are also relevant to other LCA studies in the Baltic States. The discussion on the marginal electricity source is relevant for any LCAs in the Baltic States.

4.1 Municipal waste generation and composition

Data on waste generation and composition is the common basis for environmental and economic assessments of waste management systems. These data are essential for the development of waste management systems and for the planning of treatment and disposal facilities.

LCAs for waste management planning often use the amount of waste generation as a given input parameter (White et al., 1999; Koneczny et al., 2007a). Thus the impacts of economic dynamics and other factors are not always taken into consideration for an accurate assessment of future waste generation and composition (Figure 17).

This could lead to elevated environmental burdens and higher costs due to e.g. overcapacity of waste processing facilities. Therefore, it is important to base the waste management planning decisions on more carefully designed long-term forecasts of waste generation and composition.

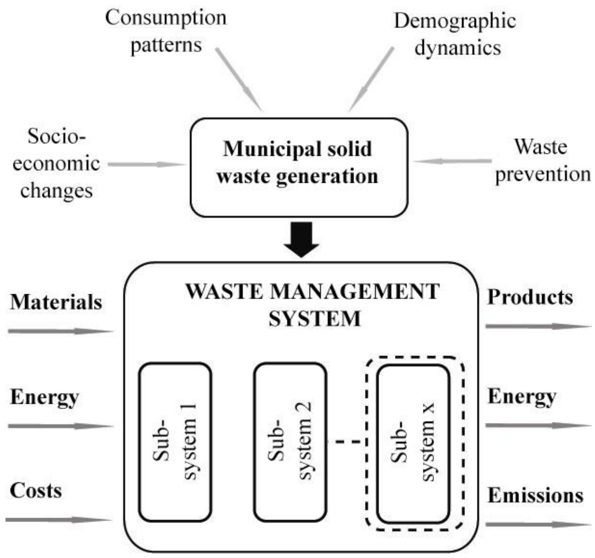


Figure 17. Factors that influence waste generation and composition in waste management systems

It may be assumed that the tendencies in municipal waste generation described below can be followed also in other countries with similar socio-economic structure and development as Estonia (e.g. the other Baltic States).

4.1.1 Problems with data quality and availability

Data on waste generation and composition in many European countries are quite poor in terms of availability, comparability, consistency and quality (ETC/RWM, 2007). This is especially relevant for the new EU Member States including Estonia. Where data do exist, they are often expressed in an inconsistent manner, data sources cover different time periods and geographical locations.

As the various surveys and studies indicate, the amount of MSW generated and its composition vary significantly among Member States (AWAST, 2003; Weidema et al., 2007; Koneczny et al., 2007a). It makes comparisons of waste data between countries difficult. This is especially relevant when comparing countries or regions with high income inequality and differences in economic growth (e.g. old and new EU Member States).

In Estonia the main source of waste data is the database of the waste register at the Environment Information Centre of the Ministry of Environment, which is based on systematised waste reporting. Although the waste data in this official database have improved in recent years, the database is still not complete and therefore it is necessary to specify the data before they can be used for waste management planning purposes.

There are many forecasting models to assess future waste generation rates (Beigl et al., 2008). These models use a variety of factors that could influence the regional municipal waste generation rate (e.g. social, economic, demographic dynamics, consumption patterns, etc.). The high heterogeneity of MSW streams and their diversity in economy make forecasting of waste generation highly complex. For example, the global economic crisis that started in 2008 has significantly influenced the waste generation rate in Estonia and made earlier waste generation forecasts questionable. How this affects the forecasting of waste trends (and therefore the relevant decisions) will be discussed below.

4.1.2 Characteristics of municipal waste in Estonia

To specify and validate the municipal waste generation and composition data in Estonia, a country-wide waste sorting analysis of mixed municipal waste was carried out by the author of this thesis (see also Chapter 3.1). The most updated information on MSW generation and composition was compiled based on data about separately collected waste fractions (waste register), composition of residual waste (derived from sorting analysis) (Moora and Jürmann, 2008a) and data obtained from earlier studies (Tallinn, 2005; Moora, 2007a, 2007b) (see summary results of the survey in Appendix 1).

In order to allow a better understanding of the waste generation trends and specific characteristics that could influence future municipal waste streams in Estonia, the results of the waste survey were analysed and compared with earlier waste data. As part of the research, the question of how the rapid economic growth has influenced waste generation and fractional composition of MSW in Estonia, was studied. A waste sorting analysis was carried out during the time period (2007/2008) when Estonia had the highest economic growth rate with an average 10% a year (in terms of GDP). In addition, the possible waste composition variability in different areas (large city, small towns and rural areas) was studied. Also the influence of the recent economic decline on the earlier municipal waste generation forecasts was examined.

Municipal waste composition trends

The fractional composition of MSW has changed significantly during the relatively short period from 2005 to 2008 (Figure 18). Due to the rapid economic growth and increased consumption levels the share of packaging waste in total MSW has increased from 26% in 2005 to 37% in 2008. This is also the reason why the total percentage of packaging related materials (plastic, glass and paper/cardboard) appears to have increased significantly, while the share of organic waste (food and garden waste) has decreased.

Such a significant change in the municipal waste composition influences the environmental and economic performance of a waste management system (e.g. the need to ensure higher material recycling levels). Therefore, the trend of packaging waste increase (especially the disproportionate increase in the generation of plastic and paper/cardboard) and the related decrease in the share of organic waste fractions in total MSW have to be taken into account when modelling future waste management scenarios.

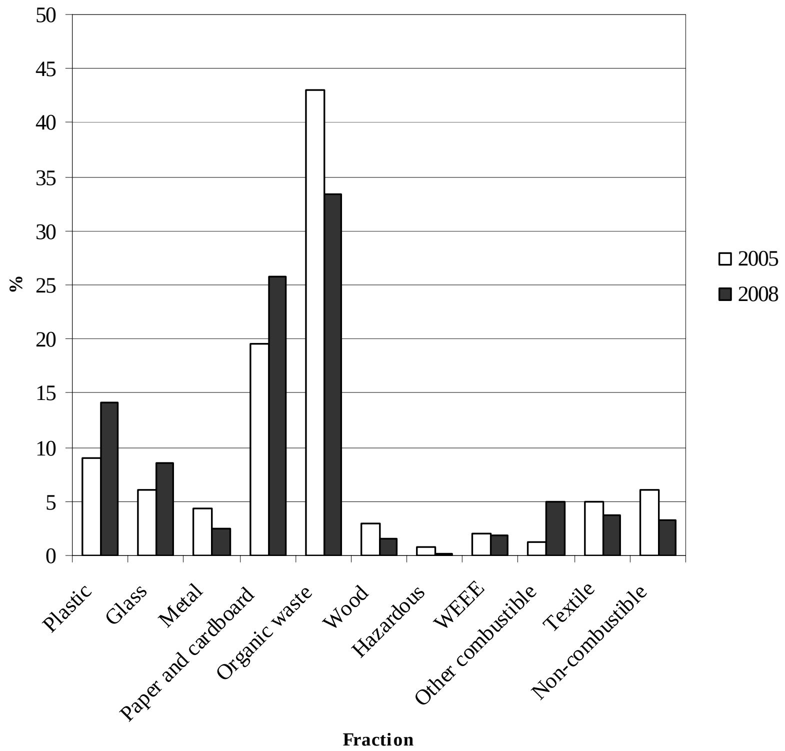


Figure 18. Changes in MSW composition 2005-2008

It may be expected that due to socio-economic differences and the level of source separation in different regions (urban and rural areas) the collected mixed municipal waste would show variations in composition. However, the results of the waste sorting analysis indicate that the composition of mixed municipal waste does not significantly vary in the studied areas (Figure 19). It may be the result of the so-called home treatment (burning and composting) of certain waste in rural areas.

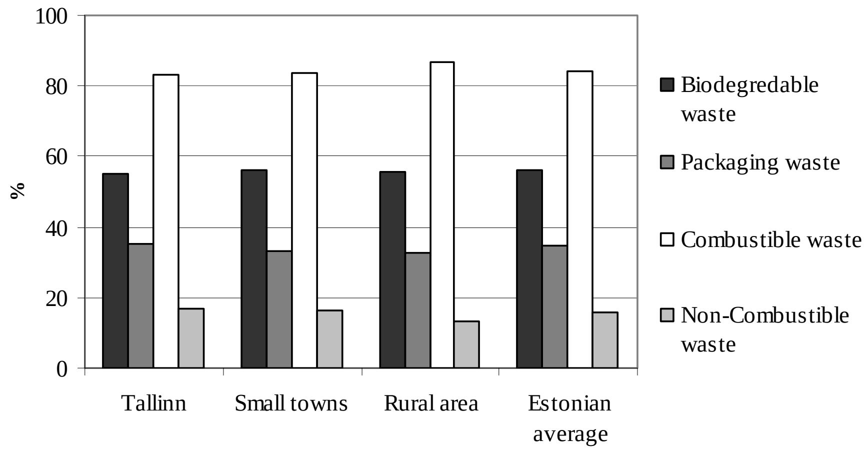
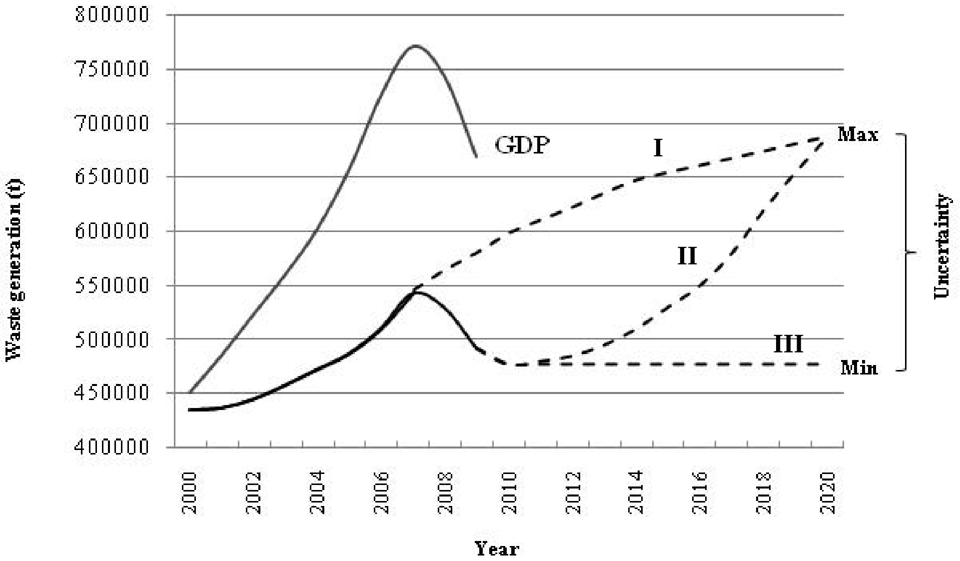


Figure 19. Composition of mixed municipal waste (landfilled) in characteristic regions in Estonia

Uncertainty in forecasting future waste generation

The quantity of municipal waste strongly depends on the socio-economic conditions of the region (Beigl et al., 2004). The generation of MSW relates mainly to the nature and intensity of economic activities and the size of the population. In general there is a strong link between Gross Domestic Product (GDP) and waste generation. The quantity of municipal waste in Estonia has rapidly risen in line with economic growth and growing consumption (see Figure 20). Since the number of population is expected to remain roughly the same in Estonia, the possible economic development will be the key driving force behind the changes in waste volumes in the next decade.

The earlier forecast of the municipal waste generation in Estonia (see also case study 2 in Chapter 3.4) was based on the assumed continuous GDP growth and final consumption by households expressed in Purchasing Power Standard (PPS). However, fluctuations in the economic situation could lead to changes in waste generation. This is well illustrated by the impact of the unexpectedly serious global economic decline that has already influenced the municipal waste generation in Estonia (see Figure 20). Recent indicators show that MSW generation has dropped in correlation with GDP decrease.



I – Earlier forecast (case 2) **II** – Max. MSW generation **III** – Min. MSW generation

Figure 20. Municipal waste generation forecast at different levels of economic growth (GDP measure)

It can be anticipated that municipal waste generation in Estonia will start to grow along with the recovery of economy. In general it is expected that in a long-term perspective municipal waste amounts will continue to grow. However, it is very difficult to predict the growth rate and time line (see Figure 20).

Based on the observed economic impact on the total amounts of MSW, it can be concluded that the underlying assumptions and expectations concerning forecasts are not always well defined and are, therefore, not transparent. The exact forecasts cannot be made. The forecasts are limited by:

- a failure to consider social and especially economic trends related to the region
- lack of reliable data and forecasts with regard to these trends.

4.2 Marginal energy source

Energy is a major consideration in LCA, both since energy is part of any LCA study and since different ways to model energy systems in LCA have brought about a debate on LCA methodology (see also Chapter 2.2.3). Many LCA studies use average data (e.g. average electricity mix of a certain region or country) to model the background systems, the systems that indirectly are affected by the actual system under the study. The use of average data to model these systems may be relevant if the aim is to analyse the impacts of past activities (attributional LCA study). However, if the aim is to model the future consequences of a decision (consequential LCA study), the use of average data may be misleading, since these data are historical data and therefore cannot capture future consequences resulting from changes in the system (e.g. changes in electric power production system). It should be noted that when applying average power production data, the results can be seriously affected by the delimitation of the market on which the action is taken. Consequential or change-oriented LCA modelling is mainly characterised by including affected (marginal) technologies and processes instead of average technologies (Weidema et al., 1999; Weidema, 2003).

Modern waste management systems are closely connected to energy systems. WTE facilities, such us waste incineration in combined heat and power plants, reduce the need for other energy sources and can therefore be expected to have marginal effects on the production of energy carriers such us heat and electricity (Figure 21).

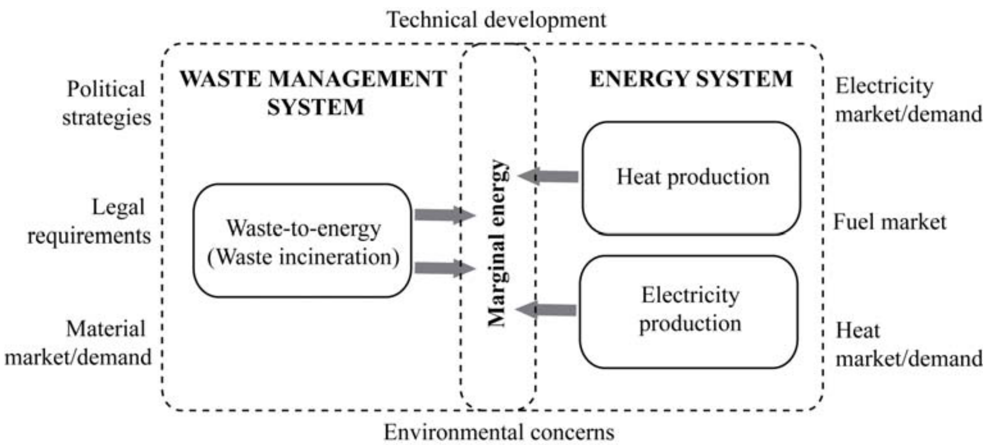


Figure 21. Relation between waste management and energy systems and factors influencing both systems

Since the choice of energy is decisive for the results in many LCA studies, it is important that the LCA practitioners have an understanding about the development of energy systems on the local and regional level.

In Estonia as in other European countries, extensive changes are taking place in the energy systems. Therefore it is difficult to determine marginal energy sources which should be used in consequential LCAs. This is especially relevant for electricity, because electricity is usually drawn from the public grid continuously supplied by a variety of power plants located in Estonia and the neighbouring countries. The sources of electricity vary during the day and over the year and the exact source of a given kWh of electricity cannot be identified. The heat market is usually limited with a local district heating system, making the marginal energy source easier to define.

To ensure that the used energy data are consistent with the rest of the system analysis, it might be necessary to carry out a separate energy system study. However, this could add significantly to the cost of the assessment. Therefore it would be meaningful for consequential LCA studies where the energy system is likely to have a significant influence on the results, to test the sensitivity of the results by using two energy sources: one with high CO_2 emissions (fossil fuels) and one with low or no CO_2 emissions (renewable sources).

4.2.1 Heat production

Waste incineration in a large-scale CHP is particularly suitable in urban locations where there is likely to be a sufficient amount of waste as well as a heat demand within a reasonable distance of the incineration plant. In Estonia there are only a few bigger cities that are suitable locations for a large-scale waste incinerator (e.g. Tallinn and Tartu). Heat produced in waste incineration is usually transferred to the district heating system where an incineration plant usually functions as a base-load heat supplier.

Development of district heating systems in Estonia

As in many other Eastern European countries, district heating has a relatively large share in the Estonian heat market. Approximately 50% of the households are connected to district heating systems. The share of district heating in heat consumption is approximately 70%. Most of the district heating systems are, however, small and heat is supplied from relatively small-scale boilers (80% of the boilers are less than 1 MW). Bigger district heating networks are located in larger cities (Tallinn, Tartu, Pärnu, Kohtla-Järve, Narva) where the share of district heating is close to 90%.

The poor condition of district heating networks and the increasing prices of heat have reduced the prospects of centralized heating and many consumers have preferred local heating (Hlebnikov and Siirde, 2008). Therefore, the share of district heating has dropped significantly during the last decade. In bigger cities the share of district heating has stabilized in recent years and it is expected that the current level will be maintained in the coming years.

The main fuels used for the heating sector in Estonia are natural gas, biomass and oil shale (less shale oil and oil shale gas) (Figure 22). Oil shale based heat is mainly used in cities close to the large oil shale fuelled power plants in the north-western part of Estonia.

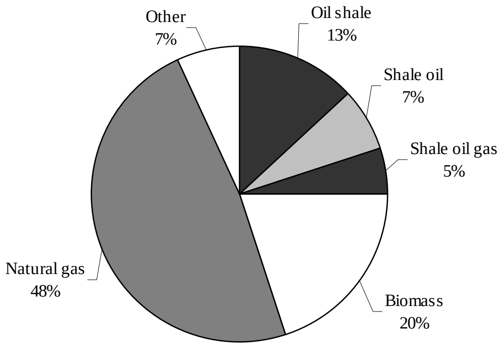


Figure 22. Fuels used for heat production in Estonia (2006)

The predominant fuels vary in district heating systems. For example, in Tallinn most of the heat is produced from natural gas (ca 98% in 2007), whereas in Tartu ca 60% of the heat is based on biomass. Due to the increasing price of fossil fuels and stricter environmental requirements more biomass is used as the fuel of choice, resulting in less use of fossil fuel. At the beginning of 2009 large-scale bio-fuelled CHP plants started to operate both in Tallinn and Tartu.

Possible marginal heat production technology

From an energy system perspective, waste incineration in CHP has an impact on the local district heating system where it affects alternative marginal fuels and technologies. The impact of waste incineration depends on local conditions and possible development trends of the district heating system where the waste incineration plant will operate. These local aspects have to be studied when defining the marginal heat production technology.

Short-term impacts include reductions in the operating hours of the existing power plants as well as possible avoidance or postponement of investments in new plants for district heat production. In general, marginal heat production technology is the technology that has the highest variable cost among alternative heat sources.

Long-term marginal heat production technology is determined by whether the local market for district heat shows an upward or downward trend. If the local market for district heat increases in the future, the new capacity added will be generally the most preferred technology, usually the technology that satisfies the given load shape at the lowest price. If the demand for district heat decreases, the long-term marginal heat production technology will be the least preferred technology.

The present and predicted fuel prices as well as the current heat production capacity and possible development trends of the district heating systems indicate that in most of the larger cities in Estonia (e.g. in Tallinn) the marginal heat technology is based on fossil fuels (mainly natural gas). In short- and long-term, natural gas will most probably be a more expensive fuel than biomass or municipal waste. On the other hand, natural gas allows for simultaneous production of district heat and a large amount of electricity. Natural gas fuelled CHP could economically compete with biomass and waste over the base load district heat if electricity prices are sufficiently high.

In certain situations when the aim is to invest in a new district heat plant (for the purpose of replacing older facilities or for system expansion), waste incineration usually competes with alternative fuels such us biomass. Biomass could be a marginal heat source also in district heating systems if it has a dominant share of the market (e.g. in Tartu).

It may be argued that, in certain situations, waste itself could be labelled as marginal fuel, i.e. if a planned incinerator competes with another waste incinerator in the same district heating system. However, the amount of municipal waste that could be incinerated in large scale CHPs, is limited. Therefore waste is most probably not labelled as a marginal heat source in Estonia.

4.2.2 Electricity source

Due to historical reasons, the electricity systems of the three Baltic States have been a common system that has links to Russia and Belarus and operates in parallel with their power systems. Until very recently, this system had no links to other European countries. In the years to come the Baltic electricity sector is expected to undergo major changes. Since the electricity market will be more liberal and open, with links to the Central European and Nordic electricity systems, it will be less relevant to refer to separate national systems in the future.

Therefore, when defining possible future marginal electricity sources that should be used in consequential LCAs in Estonia and the other Baltic States, it is important to focus on the possible developments of the common Baltic electricity system.

The Baltic electricity system

Electricity production in the three Baltic States differs considerably. The Estonian electricity production is dominated by fossil fuel sources based on a small number of large fossil-fuel power plants. The primary fuel for electricity production is oil shale, although also natural gas, oil shale gas, shale oil, diesel oil, wood and peat are used as fuels; in addition, small hydropower plants and a growing number of wind turbines are in operation. The Latvian electricity system is largely based on hydropower and co-generation of fossil fuels (mainly natural gas, and to some extent, coal). The Lithuanian electricity system is dominated by nuclear power production. The share of the installed electricity production capacities in the three Baltic States (IEA, 2006) is presented in Figure 23.

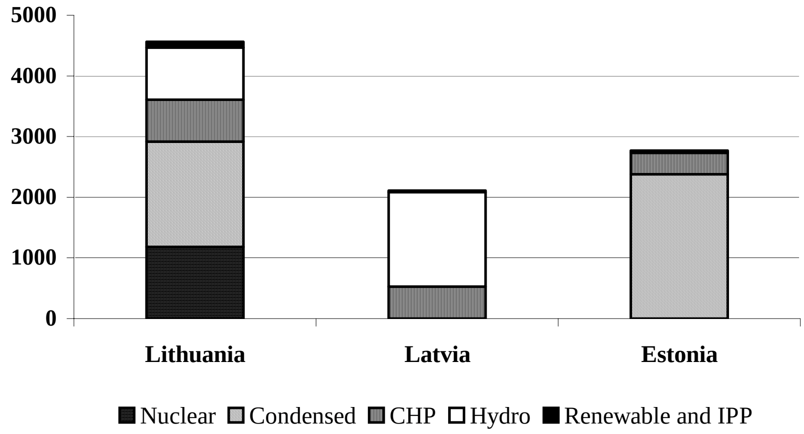


Figure 23. Installed electricity production capacities in the Baltic States 2005 (in MW)

Estonia and Lithuania are net electricity exporters. Both the Ignalina Nuclear Power Plant (NPP) in Lithuania and large oil shale fired thermal power plants in Estonia were built to supply electricity to the north-western regions of the former Soviet Union. Latvia is a net importer of electricity, buying from the other Baltic States as well as from Russia.

In the years to come, the Baltic electricity market is expected to undergo major changes. Up till recently, the electricity sector was characterised by vertically integrated monopolies, but at present the sector is undergoing reform processes to meet the requirements of the EU Directives regarding liberalisation of the electricity sectors. Decommissioning of the second unit of Lithuanian Ignalina NPP in 2009, closing down the worn-out oil shale power production capacities in Estonia by the end of 2015 and opening up the electricity market poses new challenges and forces to seek alternative electricity sources to cover the growing electricity demand in the Baltic States. While the other Baltic States have opened their electricity markets, Estonia has been granted the right to keep its market partly closed until 2013.

Possible future electricity scenarios

It is difficult to predict the mix of possible future electricity production technologies because it depends on assumptions regarding the growth of electricity demand, future fuel prices, electricity production costs, limitations due to energy security, environmental taxes, national policy incentives to support renewables, etc.

As the earlier discussion on waste generation shows (see Chapter 4.1.2), the forecasting of future can be highly uncertain.

Therefore, as recommended, two *extreme* sets of assumptions for future electricity production scenarios in the three Baltic States were studied in order to identify possible marginal electricity sources that can be used in consequential LCAs.

1. Current Trends or Business as Usual Scenario (CTS) assumes that future electricity production is based mainly on conventional fuels and technologies.
2. Baltic Sustainable Energy Scenario (BSES) assumes a more sustainable, renewable energy oriented electricity production.

Both scenarios target the year 2020. Scenarios are described and compared via energy balance (production, import-export and consumption of electricity). For baseline projections data from the National Energy Reports (2005) were used. For projecting energy sector developments, various studies available from public sources, addressing the availability of resources, were used (MoEAC, 2004; Tammoja et al., 2004; Siirde, 2004; MoEoL, 2007; Rasburskis et al., 2006; Jaskelevicius, 2007; INFORSE, 2008).

The energy sector development goals and measures for Estonia as well as assumptions related to possible power production investment projects by major market players are mainly based on the National Electricity Sector Development Plan 2005-2015 and the new drafted National Electricity Sector Development Plan 2008-2018. For Latvia, the projections within the Current Trend Scenario are based on the recently adopted Guidelines for Energy Sector Development 2007-2016. For the development of Lithuanian energy sector, the projections are based on the National Energy Strategy, adopted by the Lithuanian Parliament in January 2007.

Current Trend Scenario (CTS)

The CTS foresees that current development trends in the three Baltic States will continue. This means that concentrated power production will largely continue to prevail and no significant changes in the power supply mix will take place in these countries besides those already agreed within the EU accession process (closure of Ignalina NPP and phase-out of Narva PP old oil shale power units), by which phased out power capacities will be mainly replaced by conventional technologies (nuclear and fossil fuel).

In Estonia the additional oil shale based power production capacities (at least 2x300 MWel) will be renovated to meet the necessary demand of electricity consumption. The total capacity of the installed oil shale boilers will be about 1,100-1,200 MWel. However, oil shale use will be limited by the high cost of CO_2 quota at the carbon emission market. Also power production from natural gas will increase in order to cover the growing electricity demand and balance the electricity system. Wind power development will be modest due to the continuous uncertainty in state incentive policy. Biomass based CHP will take near maximum from the supply market which is restricted by the small heat capacity of district heating systems.

A shortage of power supply from domestic suppliers is expected to appear due to the phase-out of old capacities and lack of new capacities. The shortage will be covered by import from Nordpool and later by the new Ignalina NPP. Also a possibility to build a small nuclear power plant in Estonia should be considered. If Estonia were to build a nuclear power plant it would most probably not be operational before 2025.

In Latvia, investments in the new natural gas and coal based production capacities will have been made by 2020. Together with the development of the carbon emissions market, the interest to utilize biomass potential in the country will grow significantly. No big changes will occur as regards a wider use of hydropower. Due to Latvia's participation in the new Ignalina NPP project, a part of domestic demand will be covered by import from Lithuania and, to a smaller extent, by import from Russia and Nordpool.

In Lithuania, after the closure of Ignalina NPP, the modernized Lithuanian Power Plant will become the major source of electricity production, along with the CHPs located in bigger cities. Natural gas will be the dominating fuel at these power plants. A new nuclear capacity of 3,200 MW will be built by 2020 and after that domestic demand will be largely covered by nuclear power. Natural gas will mainly be used to run reserve plants due to NPP breaks and overhauls. Hydro-, wind and biomass energy share will remain small as all government resources will be used to cover the construction costs of the new NPP, thus no state funds will be allocated to support renewable development.

Baltic Sustainable Energy Scenario (BSES)

Another way of handling the discussion on which type of electricity will be marginal in the future is to assume that the aim is towards a more sustainable energy production. For BSES the electricity demand level for 2020 is calculated first by reducing it by the assumed energy saving. Energy saving potential assumptions are based on the official national strategies of the three Baltic States. In order to identify the calculated demand, a more sustainable energy production fuel-mix is predicted, taking into account available technologies. The main assumption is that the renewable potential of all three Baltic States can be fully used by available technologies via implementation of proper incentives and lifting of market restrictions (existing 2005-2008) by the governments.

In Estonia a large part of the oil shale based power production capacities will be phased out and only renovated blocks (about 400 MWel) will stay operational after 2015. Oil shale use will be limited also by the high cost of CO_2 quota at the carbon emission market. Wind energy development will be active; about 1,200 MW of wind turbines, many of them offshore, will be installed. Biomass based small size CHP will supply power to district heating, utilising this demand to the maximum, and new large consumer self-supply CHPs will be constructed. Estonia will become a net exporter of renewable electricity due to large-scale wind energy development and new connections to Nordpool. In order to compensate for wind deviations, new connections to larger markets (Nordpool) and cooperation with Sweden and other countries in respect of hydro reserves will play an important role. Due to the large share of wind capacity gas turbines will be built and the share of natural gas will remain relatively high (Palu et al., 2008). In the future some gas could be extracted from biomass and oil shale.

In Latvia, investments in new natural gas and possibly also clean-coal based production capacities will have been made by 2020. Together with the development of the carbon emissions market, the interest to utilize the high biomass potential of the country will be significant compared to 2005. No big changes will occur in a wider use of hydropower. Energy saving will be seriously promoted (supported) by the government, thus efficiency measures will be applied by consumers and demand increase will therefore be under control.

In Lithuania, by 2020 no new nuclear capacities will be built and domestic demand will largely be covered by natural gas based power of the existing reserve capacities which will be renovated to meet environmental standards. Wind share will increase as all government resources will be used to support carbon-free technologies deployment. The biomass sector will surge upward and new small-scale producers will operate everywhere in rural areas utilizing agro-waste and energy culture in electricity production.

Possible marginal electricity sources

Changes in electricity demand and supply can be viewed against changes they initiate in the electricity generation system. Usually, the electricity generated from nuclear or hydropower with relatively low variable cost provides the base load for electricity generation. Moreover, a nuclear power plant should work in a most stabile manner without too high fluctuations in production – largely due to safety considerations. A power plant that is turned off and on depending on the dynamics in the system (when electricity supply or demand changes), is labelled as a marginal producer. Usually all power grids do have marginal energy capacities to be utilised in cases when the electricity demand spikes. Such capacities are those where the per-unit production costs for electricity are the highest (in some countries such “expensive” plants are also those whose energy carriers such as coal, oil or gas are loaded with environmental taxes).

In general, the system response to changes in output demand (e.g. increased or decreased demand for energy) will vary in short- and long-term.

In heat production the short-term output responses to electricity demand changes typically occur at power plants that have the highest variable cost at the time of the demand change.

In the long term, the response will be changes in the timing, and perhaps the nature of investments in new production capacities. The long-term marginal electricity technology is determined by whether the electricity demand increases or decreases. If the electricity demand increases in the future, the type of new capacity added will be generally the one which is the most preferred technology, usually the one which satisfies the given load shape at the lowest price.

If the demand decreases, the long-term marginal technology will be the least preferred technology (Weidema et al., 1999). It is important to note that marginal technologies could be technologies that are able to respond to the demand instantly. Therefore, the long-term base-load electricity production capacity (e.g. nuclear power) could be counted as marginal only when electricity intensive process industry or electricity producing activities are modelled.

Different sources of electricity can be argued to be marginal in the Baltic States. In the short- to medium-term future, the present and predicted future cost structure as well as the existing power production capacity in the three Baltic States indicate that natural gas (in Latvia also coal) is the main marginal electricity source for the region. Taking into account the possible future scenarios of the Baltic energy systems, either reflecting the current trend scenario or a more sustainable future, these fuels will most probably remain marginal also in long-term. As a shortage in power supply from local suppliers is foreseen in a short- and medium term perspective, the marginal electricity sources of possible import markets have to be taken into account. According to a number of studies (Weidema et al., 1999; Sjödin and Grönkvist, 2004), in Central Europe and Nordic countries coal-condensing power as the most expensive electricity production technology available in the market is the short-term marginal electricity source. In the Nordic region natural gas is expected to be the long-term marginal source due to efforts to lower emission levels. However, recent studies indicate that the future marginal electricity source maybe also CO_2 free (Finnveden, 2008). The question of a possible future electricity import from Russia is still open, but it may be expected that coal will be the origin of the electricity imported from Russia in both short- and long-term.

It could be argued that in certain situations also other technologies/fuels could be labelled as marginal sources in the Baltic region. Taking into account that the Estonian electricity market is not yet fully open and there is currently insufficient flexibility, it is possible that in Estonia where the majority of electricity is produced from oil shale this fuel may be counted as the marginal electricity source for a shorter period. However, the position is that both oil shale as well as nuclear power are used as a base load technology, which is not adjusted to follow changes in electricity demand. Therefore, normally such technologies are not labelled as marginal electricity sources. However, with growing electricity demand and high cost of CO_2 quota at the carbon emission market, it could be said that the planned nuclear power plants could politically or environmentally (CO_2 free electricity source) be regarded as the preferred technology and therefore defined in certain conditions as a long-term marginal electricity source.

Table 5. Possible marginal electricity sources in the Baltic States

Short-term marginal electricity source	Power plants that have the highest variable cost among those at the time of the demand change - natural gas and coal fired power plants. Oil shale for a shorter period in Estonia.
Long-term marginal electricity source	CTS: natural gas and coal fired power plants BSES: renewable sources such us biomass and wind power

In the case of a sustainable electricity scenario, renewable sources meet the demand that would eventually have been met with fossil fuels. Therefore biomass fired CHP plants could be marginal technologies in long-term (for example for heat and electricity production in district heating systems). If wind power obtains a significant share, it could be one of the long-term marginal electricity sources in the future. However, this will take place only after the elimination of the current constraint related to the technical problems of power system steering (Palu et al., 2008). Hydropower with relatively low variable costs and limited and inflexible power capacity is not labelled as a marginal electricity source in the Baltic States.

In the future, under a more sustainable and liberal electricity market, there may be more demand for electricity with a lower environmental impact. As a consequence, if electricity is purchased directly from a specifically contracted production plant (i.e. renewable sources, including wind or hydropower), electricity data from these plants should be used in environmental assessments instead of data from marginal sources.

4.3 Other issues

4.3.1 Quality and market of recovered waste materials

Another important issue related to sensitive data for LCA is the size of the fraction of recovered waste. There are several factors that could limit the final recovery of the waste fraction and therefore affect the studied waste management system.

When performing a LCA study, it is important to consider the following factors:

- The quality of collected recyclable waste fractions
- Possible market for recyclable waste fractions

In Estonia the quality of source separated waste materials (especially packaging waste) could be relatively low because of contaminants and impurities. Therefore the percentage of losses and sorting out during the process could be relatively high.

To analyse the content of impurities in the source-sorted waste streams the collected waste fractions were sampled and characterised several times during the year (2007/2008) in selected waste collection containers and sorting plants (Moora and Jürmann, 2008b). As the results of the analysis show, the average content of impurities is the highest in the mixed packaging collection system (15% of collected waste). In addition to impurities some of the collected packaging waste of low quality or low economic value is sorted out. For this reason the total loss could be up to 40% of the total collected packaging waste. This part of the low-quality and dirty material is treated today as rest waste (landfilled). The content of impurities in the separately collected waste paper and glass packaging stream is however very low (average 3%).

The same aspects influence the recycling of collected organic waste. It is often assumed that the quality of compost is good and therefore a relatively high share of produced compost replaces mineral fertilisers. This is not the case in Estonia where the experiences of composting of organic waste from households show that the quality of compost is low. The quality aspects are related mainly to source separated food waste collection system, whereas the centrally collected green waste has sufficient quality.

Another limitation for an environmentally beneficial use of compost is the very low market demand for such product in Estonia. Today most of the MBW based compost is used as a filling material and for landscaping the landfills.

The development of a quality assurance system and market for composting will take years. The future of composting depends largely on the decisions concerning the national requirements related to the EU regulation on animal by-products and the implementation of the EU directive on incineration and co-incineration. Therefore, it may be assumed that the maximum amount of produced compost which will replace industrial fertilisers in Estonia will stay far below the average of the current practice in more advanced European countries (average utilisation of produced compost for agricultural purposes is 40%) (Barth, 2006).

4.3.2 Data on economic costs

Cost estimation is a basic requirement for planning MSW management systems since cost effects are an important factor for decision-making. The economic analysis of a possible waste management system is a complex task, made even more difficult due to the scarcity of real cost data. This is especially relevant for Estonia since there is no practical experience with modern waste treatment systems and technologies. Therefore, only limited information is available for making cost estimates for future waste management options. Cost data published in different international sources (Hogg, 2000, 2001; Hannequart and Radermaker, 2002) is scarce and often not applicable for the specific local conditions and operating practices in Estonia.

Waste management planners in Estonia have had the best knowledge/access concerning mixed MSW collection and landfilling related financial cost information. The biggest uncertainty is related to the additional cost of collection and treatment of organic waste. In the existing composting facilities in Estonia, the costs of organic waste composting exhibit considerable variation (15-30 EUR/t). These costs are very much affected by specific local conditions such us the chosen technology, amount and composition of the inflowing waste as well as the possible end use of the compost.

With regard to large-scale WTE facilities (e.g. waste incinerators), the cost data is more reliable than for other types of waste recovery technologies. This is because the technologies are more established and therefore the estimates of investments and operational costs can be made on the existing reference facilities in other parts of Europe. At the same time, possible future revenues from recyclables and especially energy recovery are highly uncertain. The future energy price could significantly affect the overall economic performance of a WTE facility and the whole waste management system.

5 CONCLUSIONS

5.1 Addressing research questions

The main aim of the thesis was to test the validity of the waste hierarchy in the regional context and evaluate the environmental and economic performance of alternative technological options for municipal waste management in Estonia.

The results of the research indicate that in general terms the **waste hierarchy** is valid as a guiding principle for waste management. There are, however, certain local or regional valuations that lead to exceptions to the traditional order of preferred waste management options. Thus, a conclusion may be drawn that waste hierarchy may require regional re-prioritisation taking into account specific local conditions.

Based on the results of LCA modelling (by using the WAMPS model) the following conclusions concerning the comparison of the studied waste management options have been reached.

1. Landfilling of MSW is the least preferred waste management option regarding environmental impact. This is valid even if landfill gas is recovered at a high rate. Therefore, landfilling of MSW should be avoided as far as possible, both because of the environmental impact and because of the low recovery of resources.
2. Composting has hardly any advantages with respect to the environment and energy turnover when being compared to other waste recovery options (such as recycling and incineration). However, composting has a potential if landfilling is avoided and incineration or anaerobic digestion are not feasible.
3. Materials recycling and waste incineration show the best results both in terms of the studied environmental impacts and economic costs. It is important to stress that if high rates of recycling and incineration with energy recovery are attained, net emissions may even become negative. It means that these waste management options can partly offset the emissions that occurred when the products were manufactured from virgin materials and energy was produced from fossil fuels. This is especially important concerning the climate change impact.
4. In a systems perspective there are small differences between recycling and incineration. Waste incineration is the best option when the produced energy can be utilised to a high extent and if it is assumed that the produced heat and electricity substitute fossil fuel in the background system. However, when the substituted energy is a non-fossil fuel (e.g. biomass), then waste incineration is not the most preferable waste management option any more. From a purely economic cost perspective, incineration provides more income than recycling of waste fractions with a high heating value.

5. The type of waste collection system and transport distances have low influence on the total environmental impacts compared to other waste treatment options. However, the design of a collection system may significantly influence its economic cost.
6. In general the environmental hierarchy corresponds to economic hierarchy with the exception that intensive composting system seems to be the most expensive waste management option.

A conclusion may be drawn that the scenario where a high share of recyclable waste fractions are sent to material recycling and maximum amount of rest waste is incinerated with energy recovery, is the optimal scenario for Estonia, possibly forming a basis for the development of a future waste management system.

An LCA of waste management system includes a number of uncertainties with respect to both the system (e.g. definition and boundaries) and input data and assumptions. As the case studies show there are several **data gaps and critical assumptions specific for regional context** that could have a significant impact on the final results of the LCA. It is important that LCA practitioners and decision-makers have sufficient knowledge and understanding of those characteristics and the possible restrictions related to the choices that could have the strongest influence on the results. Most of those assumptions relate to future situations and developments.

The results of the research allow us to conclude that there are two main regional characteristics (waste data and marginal energy source) needing more attention when assessment of the life-cycle based environmental and economic performance of a waste management system is undertaken. The most sensitive input data and assumptions are as follows:

1. Data on waste generation and composition are the common basis for environmental and economic assessments of waste management systems. Therefore, it is important to make all waste management planning decisions on the basis of more carefully designed long-term forecasts of waste generation and composition.
2. As the recent developments in waste generation in Estonia indicate, fluctuations in the economic situation may lead to significant changes in waste generation. Earlier assumptions and expectations concerning forecasts were not always well defined. The forecasts are limited by a failure to consider social and especially economic trends related to the region as well as lack of reliable data with regard to these trends.
3. The choice of a replaced marginal energy source in the background system could be decisive for the results of waste management related LCA studies. For example, the results for incineration will change drastically if it is assumed that

energy produced from the incineration of wastes replaces electricity from fossil fuels or non-fossil fuels.

4. Energy systems are undergoing great changes in Estonia. Therefore it is difficult to determine marginal energy sources which should be used in consequential LCAs. This is especially relevant for electricity, because electricity is supplied by a variety of power plants located in Estonia and the neighbouring countries. The heat market is usually limited to a local district heating system, making the marginal heat producer easier to define.
5. The present and predicted fuel prices as well as the current heat production capacity and possible development trends of the district heating systems indicate that in most of the larger cities in Estonia (e.g. in Tallinn) the marginal heat technology is based on fossil fuels (mainly natural gas). In long-term perspective and in certain situations also biomass could be labelled as marginal heat source.
6. The described electricity scenarios demonstrate that, no matter what the future energy supply in the Baltic States is based on – conventional or more sustainable technologies and fuels, it is clear that fuel and technology mixes will be more complex than today. Thus, the share of today's dominant sources like oil shale in Estonia or nuclear power in Lithuania will diminish and the role of the marginal electricity sources will significantly grow. Based on the present cost structure, existing power production capacity and import markets, mainly natural gas and coal fired power plants are the short-term marginal sources of electricity in all three Baltic States. It can be assumed that in the long term, despite of the changes in the electricity market and establishment of new connections, the same fuels will most probably remain the marginal electricity sources. In the case of a more sustainable electricity future trend with an additional renewable electricity production capacity or electricity conservation measures undertaken, also renewable sources such as biomass and wind power could be labelled as marginal electricity sources.
7. To reduce the uncertainty related to energy data, it might be necessary to carry out a separate energy system study. However, this could significantly add to the cost of the LCA. Therefore it would be meaningful to test the sensitivity of the results of LCA by using two energy sources: one with high CO_2 emissions (fossil fuels) and one with low or no CO_2 emissions (renewable sources).
8. Other uncertain factors to which special attention needs to be paid, are the quality and market for the recovered waste materials as well as the future costs of waste management options (especially treatment of organic waste).

The discussion in this thesis focuses mainly on the Estonian context. However, most of the results and discussions are probably also relevant for regions and countries with a similar socio-economic structure and waste management development as Estonia (e.g. the other Baltic States).

Finally, it may be concluded that LCA can provide useful information to test the validity of the waste hierarchy and evaluate the different waste management options and technologies. However, LCA and other system analysis tools should be considered as decision support tools that provide relevant information but do not substitute the crucial role of a decision maker.

5.2 Future research

Several questions in connection to this thesis remain to be addressed in further research.

The scope of this research is limited to the chosen waste management options/technologies. Although the studied waste management technologies represent the most likely options for MSW treatment in Estonia, there are also other existing and future technologies. Thus it is necessary to widen the research scope to explore the pros and cons of these new waste management technologies in the future life-cycle based environmental and economic assessments. For this, the technical and environmental parameters of these technologies should be studied while taking into account the local context.

It is also interesting to follow the developments of the connections between energy (especially district heating) and waste management in the context of future policy instruments and technology developments. Also other possible future development trends (e.g. waste generation and composition, economic costs) need more in-depth research. In addition, major characteristics with possible influence on future developments in the waste and energy sector need to be specified more precisely.

Modern waste management presents a high level of complexity, thus selection of a better waste management scenario requires the consideration of many aspects. Social aspects play an important role in the planning of sustainable waste management systems. In order to be able to better explain the dynamics of the future socio-technical waste system, social aspects need to be integrated in system analysis tools such as LCA.

REFERENCES

Azapagic, A., Clift, R., 1999. Allocation of environmental burdens in multiple function systems. Journal of Cleaner Production, vol. 7, no. 2, pp. 101-119.

ASTM, 2003. Standard Test Method for Determination of the Composition of Unprocessed Municipal Solid Waste. In: ASTM D5231-92. American Society for Testing and Materials, US.

Barth, J., 2006. Status of organic waste recycling in the EU. 1st Baltic Biowaste Conference - Development of Sustainable Biowaste Management in the Baltic States, its Framework and Instruments for a Successful Introduction. 23rd and 24th of May 2006, Tallinn, Estonia.

Beigl, P., Wassermann, G., Schneider, F., Salhofer, S., 2004. Forecasting municipal solid waste generation in major European cities. In: Pahl-Wostl, C., Schmidt, S., Jakeman, T.(Eds). iEMSs Int. Congress - Complexity and Integrated Resources Management, Osnabrück, Germany.

Beigl P., Lebersorger S., Salhofer S.P., 2008. Modelling municipal solid waste generation - A review. Waste Management, vol. 28, no. 1, pp. 200-214.

Björklund, A., Bjuggren, C., Dalemo, M., Sonesson, U., 1998. System boundaries in waste management modelling - a comparison of different approaches. In Sundberg, J. Nybrant, T. and Sivertun, Å. (Eds), proceedings of Systems Engineering Models for Waste Management, 25-26 February 1998, Göteborg, AFR-report no. 229, Swedish Environmental Protection Agency, Stockholm, Sweden.

Björklund, A., 2000. Environmental systems analysis of waste management - experiences from applications of the ORWARE model, PhD thesis. Royal Institute of Technology, Stockholm, Sweden.

Björklund, A., Finnveden, G., 2005. Recycling revisited - life cycle comparisons of waste management strategies. Resources, Conservation and Recycling, vol. 44, pp. 309-317.

CEC, 2005. Report from the Commission to the Council and the European Parliament on the National Strategies for the Reduction of Biodegradable Waste Going to Landfills Pursuant to Article 5(1) of Directive 1999/31/EC on the Landfill of Waste. Commission of the European Communities. Brussels, 30.03.2005.

Dahlen, L., Lagerkvist, A., 2008. Methods for household waste composition studies. Waste Management, vol. 28, pp. 1100-1112.

Dalemo, M., Frostell, B.M., Jönsson, H., Mingarini, K., Nybrant, T., Sonesson, U., Sundqvist, J.-O., Thyselius, L., 1997. OWARE - A simulation model for organic waste handling system. Resources Conservation & Recycling, vol. 21, pp. 17-37.

den Boer, E., den Boer J., Jager, J.(Eds), 2005. Waste management planning and optimisation. Handbook for municipal waste prognosis and sustainability assessment of waste management systems (LCA-IWM). Stuttgart.

ECON, 1995. Miljokonstnader knyttet til ulike typer avfall. ECON-report no. 338/95. ECON Energi, Oslo, Norway.

Ekvall, T., Finnveden, G., 2000. The application of life cycle assessment to integrated solid waste management. Part 2 - Perspectives on Energy and Material Recovery from Paper. Institution of Chemical Engineers. Trans IChemE, vol. 78.

Ekvall, T., Finnveden, G., 2001. Allocation in ISO 14041 - A critical review. Journal of Cleaner Production, vol. 9, pp. 197-208.

Ekvall, T., Weidema, B.P., 2004. System boundaries and input data in consequential life cycle inventory analysis. International Journal of Life Cycle Assessment, vol. 0, pp. 161-171.

Ekvall, T., Tillman, A.-M., Molander, S., 2005. Normative ethics and methodology for life cycle assessment. Journal of Cleaner Production, vol. 13, pp. 1225-1234.

Ekvall, T., Assefa, G., Björklund, A., Eriksson, O., Finnveden, G., 2007. What life cycle assessment does and does not do in assessments of waste management. Waste Management, vol. 27, pp. 989-996.

Eriksson, O., Frostell, B., Björklund, A., Assefa, G., Sundqvist, J.-O., Granath, J., Carlsson, M., Baky, A., Thyselius, L., 2000. OWARE - A simulation tool for waste management. Resources, Conservation and Recycling, vol. 36, no. 4, pp., 287-307.

ETC/RWM, 2007. Environmental outlooks: municipal wastes. Working paper 2007/1. European Topic Centre on Resource and Waste Management. Copenhagen. Available http://waste.eionet.europa.eu/publications.

European Commission, 1994. European Parliament and Council Directive 94/62/EC of 20 December 1994 on packaging and packaging waste. Official Journal, L 365 , 31.12.1994.

European Commission, 1999. Council Directive 1999/31/EC of 26 April 1999 on the landfill of waste. Official Journal, L 182 , 16.07.1999.

European Commission, 2000a. 2000/532/EC: Commission Decision of 3 May replacing Decision 94/3/EC establishing a list of wastes pursuant to Article 1(a) of Council Directive 75/442/EEC on waste and Council Decision 94/904/EC establishing a list of hazardous waste pursuant to Article 1(4) of Council Directive 91/689/EEC on hazardous waste. Official Journal, L 226, 6.9.2000.

European Commission, 2000b. Directive 2000/76/EC of the European Parliament and of the Council of 4 December 2000 on the incineration of waste. Official Journal, L 332/91, 28.12.2000.

European Commission, 2003. Council Decision of 19 December 2002 establishing criteria and procedures for the acceptance of waste at landfills pursuant to Article 16 of and Annex II to Directive 1999/31/EC (2003/33/EC). Official Journal, L 011, 16.01.2003.European Commission, 2005a. Taking sustainable use of resources forward: A Thematic Strategy on the prevention and recycling of waste. COM(2005) 666 final.

European Commission, 2005b. Thematic Strategy on the sustainable use of natural resources. COM(2005) 670 final.

European Commission, 2008. Directive 2008/98/EC of the European Parliament and of the Council of 19 November 2008 on waste and repealing certain Directives. Official Journal, L 312/3, 22.11.2008.

Finnveden, G., Albertsson, A.-C., Berendson, J., Eriksson, E., Höglund, L.O., Karlsson, S., Sundqvist, J.-O., 1995. Solid waste treatment within the framework of life cycle assessment. Journal of Cleaner Production, vol. 3, pp. 189-199.

Finnveden, G., Lindfors, L.-G. 1998. Data quality of life cycle inventory data - rules of thumb. International Journal of Life Cycle Assessment, vol. 21, pp.191-237.

Finnveden, G., 1999. Methodological aspects of life cycle assessment of integrated solid waste management systems. Resources, Conservation and Recycling. vol. 26, pp. 173-187.

Finnveden, G., 2000. On the limitations of life cycle assessment and environmental systems analysis tools in general. International Journal of Life Cycle Assessment, vol. 5, pp. 229-238.

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

Finnveden, G., Björklund, A., Moberg, Å., Ekvall, T., 2007. Environmental and economic assessment methods for waste management decision-support: possibilities and limitations. Waste Management & Research, vol. 25, pp., 263-269.

Finnveden, G., 2008. A world with CO2 caps - Electricity production in consequential assessments. International Journal of Life Cycle Assessment, vol. 13, pp. 365-367.

Fliedner, A., 1999. Organic Waste Treatment in Biocells - A Computer-based Modelling Approach In the Context of Environmental Systems Analysis. Master of Science Degree Thesis, TRITA-KET-IM 1999:5, Dept. of Chemical Engineering and Technology, Royal Institute of Technology, Stockholm, Sweden.

Guinee, J.B., Goree, M., Heijungs, R., Huppes, G., Kleijn, R., de Koning, A., van Oers, L., Wegener Sleeswijk, A., Suh, S., Udo de Haes, Hr.A., de Bruijn, H., van Duin, R., Huijbregts, M.A.J., 2002. Handbook on Life Cycle Assessment.

Operational Guide to the ISO Standards. I: LCA in Perspective. IIa: Guide. IIb: Operational Annex. III: Scientific Background. Kluwer Academic Publishers, Dordrecht.

Gren, I.M., 1993. Alternative nitrogen reduction policies in Mälar region, Sweden. Ecological Economics, vol. 7, pp. 159-172.

Heijungs, R., Suh, S., Kleijn, R., 2005. Numerical Approaches to Life Cycle Interpretation - The case of the Ecoinvent´96 database. International Journal of Life Cycle Assessment, vol. 10, no. 2, pp. 103-113.

Heijungs, R., Guinee, J.B., 2007. Allocation and "what if" scenarios in life cycle assessment of waste management systems. Waste management, vol. 27, pp. 997-1005.

Hannequart, J.-P., Radermaker F., 2002. Methods, costs and financing of waste collection in Europe. General review and comparison of various national policies. Brussels, Association of Cities and Regions for Recycling: 19.

Hlebnikov, A., Siirde, A., 2008. The major characteristic parameters of the Estonian district heating networks and their efficiency increasing potential. Energetika, vol. 54, no. 4, pp.67-74.

Hogg, D., 2001. Costs for Municipal Waste Management in the EU, Eunomia Research & Consulting. Directorate General Environment, European Commission: 79.

Hogg, D., 2002. Composting waste - Assessing the costs and benefits. Waste Management World, vol. March/April, pp. 35-41.

IEA, 2006. Key World Energy Statistics. International Energy Agency. Available http://www.iea.org

INFORECE-Europe, 2008. Action Plan of Vision 2050 for Lithuania. Available http://www.inforse.dk/europe/pdfs/Actions-for-Lithuania.pdf

ISO, 1998. ISO 14041 International Standard. Environmental management - Life cycle assessment - Goal and scope definition and inventory analysis. ISO 14041:1998. International Organisation for Standardisation.

ISO, 2000a. ISO 14042 International Standard. Environmental management - Life cycle assessment - Life cycle impact assessment. ISO 14042:2000. International Organisation for Standardisation.

ISO, 2000b. ISO 14043 International Standard. Environmental management - Life cycle assessment - Life cycle interpretation. ISO 14043:2000. International Organisation for Standardisation.

ISO, 2006. ISO 14040 International Standard. Environmental management - Life cycle assessment - Principles and framework. ISO 14040:2006. International Organisation for Standardisation.

Jaskelevicius, B., 2007. Kyoto Protocol requirement and wind energy evolution in Lithuania. Ekologija, vol. 53, no. 3, pp., 16-23.

Jungmeier, G., Werner, F., Jarnehammar, A., Hohenthal, C., Richter, K., 2002. Allocation in LCA of Wood-based Products Experiences of Cost Action E9. Part I - Methodology. International Journal of Life Cycle Assessment, vol. 7, no. 5, pp. 290-294.

Kates, R.W., Parris, T., Leiserowitz, A., 2005. What is sustainable development? Goals, indicators, values and practice. Environment vol. 47, pp. 8-21.

Kirkeby, J. T., Birgisdottir, H., Hansen, T. L., Christensen T. H., Bhander, G. S., Hauschild, M., 2006. Environmental assessment of solid waste systems and technologies: EASEWASTE. Waste management & Research, vol. 24, no. 1, pp. 3-15.

Koneczny, K., Dragusnau, V., Bersani, R., Pennington, D. W., 2007a. Environmental Assessment of Municipal Waste Management Scenarios: Part I - Data collection and preliminary assessments for life cycle thinking pilot studies. European Commission.

Koneczny, K., Pennington, T. (Eds), 2007b. Environmental Assessment of Municipal Waste Management Scenarios: Part II - Detailed Life Cycle Assessments. European Commission.

Leontief, W. and Ford, D., 1970. Environmental repercussions and the economic structure: An input-output approach. Review of Economics and Statistics, vol. 52, no. 3, pp. 262-271.

Leontief, W., 1986. Input-Output Economics. New York, Oxford University Press.

Leontief, W., Koo, J.C.M., Nasar, S., Sohn, I., 1983. The future impact of non-fuel minerals in the U.S. and world economy. Lexington, Lexington Books.

MoEAC, 2004. Long-Term National Development Plan for the Fuel and Energy Sector until 2015. Ministry of Economic Affairs and Communication. Tallinn, Estonia.

MoEoL, 2007. Guidelines for Energy Sector Development 2007-2016. Project. Ministry of Economics of Latvia. Riga, Latvia.

Moora, H., 2007a. Installation of waste incineration unit in Iru power plant - Life cycle based environmental and economic assessment. SEI-Tallinn. [in Estonian]

Moora, H., 2007b. Life cycle based environmental and economic assessment of municipal waste management alternatives in Estonia. SEI-Tallinn. [in Estonian]

Moora, H., Jürmann, P., 2008a. The analysis of the quantity and composition of mixed municipal solid waste in Estonia - Municipal solid waste composition study. SEI-Tallinn.

Moora, H., Jürmann, P., 2008b. The analysis of the quantity and composition of mixed municipal solid waste in Estonia - Packaging waste composition study. SEI-Tallinn. [in Estonian]

Ness, B., Urbel-Piirsalu, E., Anderberg, S., Olsson, L., 2007. Categorising tools for sustainability assessment. Ecological Economics, vol. 60, pp. 498-508.

Norris, G.A., 2002. An introduction to Input-Output LCA Theory and Methodology; Its Strengths and Weaknesses and a comparison between Input-output LCA and Process LCA. Presentation from the 16th Discussion Forum on LCA. Lausanne, Swtzerland. 10.04.2002.

Nordtest, 1995. Solid waste, municipal: sampling and characterisation. Nordtest method NT ENVIR 001, Finland.

Palu, I., Tammoja, H., Oidram, R., 2008. Thermal power plant cooperation with wind turbines. Estonian Journal of Engineering, vol. 14, no. 4, pp. 317-324.

Plepys, A., 2004. Environmental Implications of Product Servicising. - The Case of Outsourced Computing Utilities. International Institute for Industrial Environmental Economics. Lund, Lund University: 119 (201 incl. articles).

Rasburskis, N., Gudzinskas, J., Gylys, J., 2006. Combined heat and power production: social-economic and sustainable development aspects. Journal of Civil Engineering and Management, vol. 12, no., 1, pp. 29-36.

Reap, J., Roman, F., Duncan, S., Bras, B., 2008. A survey of unresolved problems in life cycle assessment. Part 2: impact assessment and interpretation. International Journal of Life Cycle Assessment, vol. 13, pp. 374-388.

Rebitzer, G., Hunkeler, D., 2004. Life Cycle Costing in LCM: Ambitions, Opportunities and Limitations, discussing a framework. International Journal of Life Cycle Assessment, vol. 8, pp. 253-256.

Reich, M.-C., 2005. Economic assessment of municipal waste management systems - case studies using combination of life cycle assessment (LCA) and life cycle costing (LCC). Journal of Cleaner Production, vol.13, pp. 253-263.

Siirde, A., 2007. Potential of combined heat and power production in Estonia. Tallinn University of Technology. Report, p. 27. Tallinn. [in Estonian]

Sjödin, J., Grönkvist, S., 2004. Emission accounting for use and supply of electricity in the Nordic market. Energy Policy, vol. 32, no. 12, pp. 1555-1564.

Smith, A., Brown, K., Ogilvie, S., Rushton, K., Bates, J., 2001. Waste Management options and climate change. Final report to the European Commission, DG Environment. AEA Technology, p. 204.

Sonesson, U., 1996. The ORWARE Simulation Model - Compost and Transport Sub-Models. SLU report no. 215, Department of Agricultural Engineering, Swedish University of Agricultural Sciences, Uppsala, Sweden.

Sonesson, U., 1998. Systems analysis of waste management - The ORWARE model, transport and compost sub-models. Doctoral Thesis, Agraria 130, Swedish University of Agricultural Sciences, Uppsala, Sweden.

Oras, K., 2002a. Statistical work on developing a methodology for determination of waste generated in non-covered areas by the waste collection system and the estimation of generated municipal waste quantities. Statistical Office of Estonia.Oras, K., 2002b. Statistical work on determination of relevant information available at municipal waste collectors. Statistical Office of Estonia.

Suh, S., Huppes, G., 2005. Methods for Life Cycle Inventory of a product. Journal of Cleaner Production, vol. 13, no. 7, pp. 687-698.

Suh, S., 2003. Input-output and hybrid Life Cycle Assessment. International Journal of Life Cycle Assessment, vol. 8, no. 5, p. 253.

Sundqvist, J.-O., 1999. Life cycle assessment and solid waste - Guidelines for solid waste treatment and disposal in LCA. AFR report 279, Swedish Environmental Research Institute, Stockholm, Sweden.

Sundqvist, J.-O., Granath, J., Reich, M. C., 2002. Hur skall hushållsavfallet tas om hand? Utvärdering av olika behandlingsmetoder. IVL, Stockholm, Sweden.

Tallinn, 2005. The generation and composition of household waste in Tallinn. Tallinn Environmental Board.

Tammoja, H., Raesaar, P., Valtin, J., Tiigimägi, E., 2004. Electricity consumption in Estonia 2005-2015. Tallinn University of Technology. Report. [in Estonian]

Tillman A.-M.., 1999. Significance of decision-making LCA methodology. Environmental Impact Assessment Review, vol. 20, pp. 113-123.

Tsilemou, K., Panagiotakopoulos, D., 2006. Approximate cost functions for solid waste treatment facilities. Waste Management and Research, vol. 24, pp. 310-322.

Udo de Haes, H.A., Jolliet, O., Finnveden, G., Hauschild, M., Krewitt, W., Mueller-Wenk, R., 1999. Best available practice regarding impact categories and category indicators in life cycle impact assessment. International Journal of Life Cycle Assessment, vol. 4, no. 66, p. 74.

Weidema, B.P., Frees, N., Nielsen, A.M., 1999. Marginal production technologies for life cycle inventories. International Journal of Life Cycle Assessment, vol. 4, no. 1, pp. 48-56.

Weidema B.P., 2003. Market information in life cycle assessment. Danish Environmental Protection Agency. Environmental Project no. 863 .Copenhagen.

Weidema, B.P., Villeneuve, J., Tojo, N., Neubauer, A., Fehringer, R., Favoino, E., Wesnas, M., Bräuer, I., 2007. Stakeholder perspectives for decision making in municipal waste management planning. Submitted to International Journal of Intergrated Waste Management, Science and Technology.

Wenzel, H., 1998. Application Dependency of LCA Methodology: Key Variables and Their Mode of Influencing the Method. International Journal of Life Cycle Assessment, vol. 3, no. 5, pp. 281–288.

White, P., Franke, M., Hindle, P., 1999. Integrated waste management: A lifecycle inventory, Aspen Publishers, Maryland, USA.

Williams, E., 2004. Energy Intensity of Computer Manufacturing: Hybrid Assessment Combining Process and Economic Input-Output Methods.

Winkler, J., 2005. Comparative Evaluation of Life Cycle Assessment Models for Solid Waste Management. International Journal of Life Cycle Assessment, vol. 9, no. 6, pp. 156–157.

Winkler, J., Bilitewski, B., 2007. Comparative evaluation of life cycle assessment models for solid waste management. Waste Management, vol. 27, pp. 1021-1031.

Wrisberg, N., Udo de Haes, H.A., Triebswetter, U., Eder, P., Clift, R., 2002. Analytical tools for environmental design and management in a systems perspective. Kluwer Academic Publishers, Dordrecht, p. 275.

SUMMARY

The new European Union (EU) Member States including Estonia have recently experienced a rapid economic development, resulting in a significant increase of waste quantities, while their waste management systems still require much effort to be adjusted to the European state-of-the-art. The municipal solid waste (MSW) management system in Estonia must comply with the principles and targets of the European waste policy and directives.

The current EU waste policy as well as the related legal requirements are based on a concept known as the waste hierarchy. However, this concept and the ranking of waste management options have caused a lot of discussion in many countries.

Life cycle assessment (LCA) as one of the comprehensive decision-support tools for analysing complex socio-economic systems has gained high recognition in waste management planning and policy-making. Interest in LCA is growing also in Estonia. However, the inherent complexity as well as lack of methodological expertise and specific local data hinder the use of LCA models in decision making.

The main aim of this thesis was to test the validity of the waste management hierarchy in the regional context by evaluating the environmental and economic performance of a number of MSW management scenarios and treatment options in Estonia. For that reason a screening level LCA model WAMPS was developed and tested by using specific regional particularities. Also the specific local characteristics and critical assumptions that could have a larger impact on the final results of LCA were discussed. As such the thesis provides background insights for the development of a common reference system for the waste management related LCA in Estonia.

The research is outlined as the results of illustrative case studies. In the first case study alternative future waste management scenarios for a specific region (large urban region - Tallinn) were compared in terms of their environmental impacts and economic costs. The second case study complements the first one by analysing the two most feasible waste recovery options (incineration with energy recovery and composting) in terms of their possible contribution to climate change.

The results of the case studies indicate that in general terms the waste hierarchy is valid as a guiding principle for waste management. However, there are certain local or regional valuations that lead to exceptions to the traditional order of preferred waste management options. Thus, a conclusion may be drawn that the waste hierarchy may require regional re-prioritisation taking into account specific local conditions. Materials recycling and waste incineration with energy recovery show the best results both in terms of the studied environmental impacts and economic costs when being compared to other waste recovery options (such as composting and landfilling). Therefore, it could be concluded that the scenario where a high share of recyclable waste fractions are sent to material recycling and maximum amount of rest waste is incinerated with energy recovery, is the optimal scenario

for Estonia, possibly forming a basis for the development of a future waste management system.

The case studies show that there are several data gaps and critical assumptions specific for regional context that could have a significant impact on the final results of the LCA.

As the recent developments in waste generation in Estonia indicate, fluctuations in the economic situation may lead to significant changes in waste generation and composition. Earlier assumptions and expectations concerning forecasts of waste generation and composition were not always well defined. Therefore, it is important to make all waste management planning decisions on the basis of more carefully designed long-term forecasts of waste generation and composition.

Modern waste management systems are closely connected to energy systems. Waste-to-energy facilities, such us waste incineration in combined heat and power plants, reduce the need for other energy sources and can therefore be expected to have marginal effects on the production of energy carriers such us heat and electricity. The case studies show that the results of LCA change drastically if it is assumed that energy produced from the incineration of wastes replaces electricity from fossil fuels or non-fossil fuels. It is difficult to determine future marginal energy sources which should be used in consequential LCAs. This is especially relevant for electricity, because electricity is supplied by a variety of power plants located in Estonia and the neighbouring countries. The heat market is usually limited to a local district heating system, making the marginal heat producer easier to define.

The present and predicted fuel prices as well as the current heat production capacity and possible development trends of the district heating systems indicate that in most of the larger cities in Estonia (e.g. in Tallinn) the marginal heat technology is based on fossil fuels (mainly natural gas). In a long-term perspective and in certain situations also biomass could be labelled as the marginal heat source.

Based on the present cost structure and the existing electricity production capacity and import markets, mainly natural gas and coal fired power plants are assumed to be the short-term marginal sources of electricity for a common Baltic electricity market. In the long term, despite of the changes in the electricity market and establishment of new connections, the same fuels will most probably remain the marginal electricity sources. In the case of a more sustainable electricity future trend with an additional renewable electricity production capacity or electricity conservation measures undertaken, also renewable sources such as biomass and wind power could be labelled as marginal electricity sources.

The discussion in this thesis focuses mainly on the Estonian context. However, most of the results and discussions are probably also relevant for regions and countries with a similar socio-economic structure and waste management development as Estonia (e.g. the other Baltic States).

KOKKUVÕTE

Euroopa Liidu uute liikmesriikide, sh Eesti majandus on viimastel aastatel kiiresti kasvanud, mistõttu on järsult suurenenud ka olmejäätmete kogus. Samas peab Eesti jäätmekäitlus lähiajal läbi tegema mitmeid olulisi muudatusi ja arenguid, et jõuda vastavusse Euroopa jäätmepoliitika ja õigusaktide nõuetega.

Euroopa Liidu jäätmepoliitika ja jäätmekäitlussüsteem on üles ehitatud nn jäätmehierarhia põhimõttel. Samas on jäätmehierarhia kohandamine ning jäätmekäitlusvalikute tegemine põhjustanud liikmesriikides ägedaid vaidlusi.

Olelusringi hindamine (*life cycle assessment*, LCA) kui keeruliste sotsiaal-majanduslike süsteemide analüüsimise üks põhjalikumaid meetodeid on jäätmekäitluse kavandamisel ja selle poliitika kujundamisel laialt kasutamist leidnud. Ka Eestis tuntakse selle meetodi vastu üha suuremat huvi. Samas on selle meetodi keerukus, võimalike kasutajate metodoloogiline pädevus ning kohaliku tasandi teabe puudumine siiani suuresti takistanud selle vahendi laiemat kasutamist Eestis.

Käesoleva doktoritöö peamine eesmärk on hinnata jäätmehierarhia paikapidavust, arvestades kohalikke ja regionaalseid tingimusi. Selleks uuriti alternatiivsete jäätmekäitlusstsenaariumite ja -tehnoloogiate keskkonnamõju ning majanduskulusid Eestis. Uuring viidi läbi doktoritöö käigus välja töötatud LCA mudeli WAMPS abil.

LCA tulemusi võivad oluliselt mõjutada spetsiifilised kohalikud või regionaalsed sisendandmed ja nende kvaliteet ning hindaja tehtud valikud. Käesolevas töös antakse ülevaade sellistest olulisematest sisendparameetritest ja valikutest, mida tuleks arvesse võtta jäätmekäitlussüsteemide olelusringipõhisel modelleerimisel. Tulemused panustavad jäätmekäitluse LCA Eestile eriomase andmebaasi loomisse.

Töö põhineb kahe juhtumuuringu tulemustel. Esimene juhtumuuring hindas ja võrdles Tallinna piirkonnas rakendatavate võimalike jäätmekäitlusstsenaariumite keskkonnamõju ja majanduskulusid. Teine juhtumuuring analüüsis Eestis tekkivate olmejäätmete käitlemise arengutest tulenevat olelusringipõhist mõju kliimamuutustele. Võrreldi jäätmepõletusel ja kompostimisel põhinevate võimalike jäätmekäitlusstsenaariumite panust kasvuhoonegaaside tekkesse.

Uuringu tulemused näitavad, et jäätmehierarhiaga toodud tavapärane jäätmekäitlusmooduste järjestus on üldplaanis rakendatav. Samas, võttes arvesse kohalikke tingimusi, võib taaskasutusmooduste järjestus olla piirkonniti erinev.

Jäätmete materjalina ringlussevõtt ja põletamine koos energiakasutusega näitavad parimaid tulemusi nii keskkonnamõju kui ka majanduslike näitajate poolest, võrreldes teiste uuritud jäätmekäitlusmoodustega (jäätmete ladestamine prügilasse ja kompostimine). Nii võib väita, et Eesti jaoks on kõige optimaalsem jäätmekäitlusstsenaarium, mille puhul suunatakse võimalikult suur kogus olmejäätmeid taaskasutusse materjali ringlussevõtuna ning ülejäänud jäätmed põletatakse võimalikult suures koguses.

Jäätmekäitlussüsteemide modelleerimise tulemused näitavad samas, et teatud kohaliku tasandi sisendteave ja valikud võivad oluliselt muuta LCA tulemusi.

Teave olmejäätmete tekke ja liigilise koostise kohta loob aluse jäätmekäitluslahenduste kavandamisele. Eestis on sellekohane teave olnud üsna puudulik. Majanduse kiire kasv ja sellele järgnenud järsk langus on selgesti näidanud eelnevate prognooside ja hinnangute üledimensioonitust.

Ajakohased jäätmekäitlussüsteemid on tihedalt seotud energiasüsteemidega. Näiteks asendab jäätmepõletusel toodetav energia taustsüsteemis toodetud energiat, mõjutades seega nn marginaalseid energiatootjaid/tehnoloogiaid (soojatootmist ja/või elektritootmist). Asendatavaid marginaalseid energiaallikaid võib olla keeruline tuvastada. Seda eriti elektrienergia puhul, kuna võimaliku marginaaltehnoloogia tuvastamiseks tuleb uurida Balti riikide ühtset elektrienergiasüsteemi ja selle tulevikuarenguid. Asendatavat soojatootjat/kütust on üldjuhul lihtsam välja selgitada, kuna üldjuhul on see piiratud kohaliku kaugküttesüsteemiga.

Võttes arvesse kütuse tänaseid ja prognoositud hindu ning suuremate linnade kaugküttesüsteemi arenguid, võib eeldada, et jäätmepõletamisel põhinev soojuse tootmine asendab fossiilkütustel (eelkõige maagaas) põhinevaid soojatootmis-tehnoloogiaid. Teatud tingimustel võib jäätmekütusega konkureerida ka biomass.

Käesoleva töö käigus koostatud Balti riikide võimalikest elektritoomis-stsenaariumitest lähtudes võib eeldada, et nii lühi- kui ka pikaajalises perspektiivis (kuni aastani 2020) põhinevad elektritootmise marginaaltehnoloogiad fossiilkütustel. Kui aga lähtuda sellest, et Balti riikide elektritootmise arengud on enam suunatud säästlike elektritootmistehnoloogiate kasutuselevõtule ja energiasäästule, siis võivad marginaalelektri allikad olla ka taastuvad energiavarud, näiteks tuul ja biomass.

Kuna marginaalenergiaallikat (eriti elektrienergia puhul) võib olla raske valida, siis tuleks LCA uuringute puhul esitada tulemused, mis lähtuvad nii fossiilkütuste kui ka taastuvate energiavarude asendamise eeldusest.

Käesolev doktoritöö keskendub eelkõige Eesti tingimuste uurimisele. Samas võib eeldada, et suurem osa töö tulemustest on võimalik kasutada olelusringi hindamise protsessis ka teiste sotsiaal-majanduslikult lähedastes ning jäätmekäitlusarengute poolest sarnastes regioonides ja riikides (nt Baltimaad).

Acknowledgements

I would like to convey my appreciation to a number of people for their help in writing this doctoral dissertation.

First of all, I would like to thank my supervisor Andrius Plepys (IIIEE, Lund University) for his help and valuable input over the years. Thank you Andrius for the fruitful discussions and constructive comments on the thesis drafts.

Thanks are due also to my other supervisor Prof. Enn Loigu (Tallinn University of Technology) for his great support and encouragement to finalise my work.

I would like to thank my colleagues at SEI-Tallinn for good cooperation and support. Especially people with whom I have shared an office: Viire, Karin, Evelin and Peep. We have shared many valuable hours discussing both research and non-research related topics.

Thanks also to "WAMPS team", Jan-Olov Sundqvist, Åsa Stenmarck and Jurate Miliute, for shaping up my knowledge on LCA.

Special thanks to Peeter Eek (Head of the Waste Management Department, Estonian Ministry of Environment) in sharing his knowledge about waste management and giving me a chance to be involved in many interesting projects.

Financial support from the Stockholm Environment Institute for finalising the research is much appreciated.

And last but not least, many thanks to my family for their patience.

APPENDIX 1. Results of municipal waste sorting study

The fractional composition of mixed municipal waste in 8 study areas 2007/2008 (Moora and Jürmann, 2008a)

Category	**Tallinn**			**Small towns**		**Rural area**			**Estonian average**
	Kesk-linn	Nõmme	Haabersti	Paide	Jõhvi	Ida-Viru-maa	Pärnu-maa	Rapla-maa	
1. Plastic	17,84	20,55	17,12	21,55	16,75	20,36	23,24	19,01	18,63
2. Glass	9,52	6,78	9,93	6,63	9,09	6,07	11,62	5,53	8,32
3. Metal	2,44	3,33	2,5	2,21	2,53	2,37	3,36	2,73	2,58
4. Paper and cardboard	20,57	15,22	16,18	16,52	18,81	13,86	12,25	16,27	17,53
5. Organic waste total*	32,71	35,79	38,59	33,78	40,29	40,91	32,39	36,99	36,65
5.1 Kitchen waste	*25,25*	*28,25*	*34,9*	*29,76*	*33,18*	*35,53*	*27,88*	*32,11*	*30,00*
5.2 Garden waste	*6,49*	*6,4*	*2,51*	*2,7*	*5,42*	*3,55*	*2,83*	*3,48*	*5,27*
5.3 Other org waste	*0,97*	*1,14*	*1,18*	*1,32*	*1,69*	*1,83*	*1,68*	*1,40*	*1,38*
6. Wood	0,62	0,3	0,44	0,19	0,34	0,23	0,24	0,38	0,44
7. Hazardous waste	0,18	0,46	0,22	0,14	0,1	0,24	0,27	0,28	0,22
8. WEEE	0,68	0,35	0,47	0,61	0,28	0,85	0,26	0,67	0,58
9. Other combustible material	6,06	5,96	5,8	5,82	4,19	7,28	8,09	9,82	6,34
10. Textile	5,08	5,97	4,86	6,04	3,17	4,09	4,18	4,04	4,43
11. Other non-combustible material	4,28	5,29	3,9	6,5	4,46	3,73	4,08	4,29	4,28
Total	100,0	100,0	100,0	100,0	100,0	100,0	100,0	100,0	100,0

* Organic waste – garden and kitchen waste and other organic waste (excluding paper, cardboard and wood waste)

Category	**Tallinn**			**Small towns**		**Rural area**			**Estonian average**
	Kesk-linn	Nõmme	Haabersti	Paide	Jõhvi	Ida-Virumaa	Pärnu-maa	Rapla-maa	
Biodegradable total	55,63	53,17	56,57	52,13	60,20	56,47	45,87	54,47	55,97
Packaging waste total	36,93	33,68	34,85	34,94	31,37	32,21	41,90	32,67	34,49
Combustible waste	82,90	83,79	82,98	83,91	83,55	86,73	80,39	86,50	84,02
Non-combustible waste	17,10	16,21	17,02	16,09	16,45	13,27	19,61	13,50	15,98

APPENDIX 2. Curriculum vitae

1. Personal data

Name:	Harri Moora
Date and place of birth:	13.03.1969, Tallinn

2. Contact information

Address:	Kentmanni 9-13, Tallinn
Phone:	+372 5052365
E-mail:	harri.moora@seit.ee

3. Education

Educational institution	Graduation year	Education (field of study/year)
Lund University	1998	Environmental Management and Policy/MSc
University of Tartu	1993	Geology/BSc

4. Language competence/skills (fluent; average, basic skills)

Language	Level
Estonian	Mother tongue
English	Fluent
Russian	Average
Finnish	Fluent

5. Special Courses

Period	Educational or other organisation
28-30.10.2008	Bridging environmental and economic assessments for decision support (II) - Expert Seminar and PhD Course at Aalborg University, Denmark
15-21.09.2006	Bridging environmental and economic assessments for decision support (I) - Expert Seminar and PhD Course at Aalborg University, Denmark

6. Professional Employment

Period	Organisation	Position
Since 1998	Estonian Institute for Sustainable Development, Stockholm Environment Institute	Programme director
1995	Geological Survey of Estonia, Department of Marine Geology and Geophysics	Researcher

7. Defended theses

MSc thesis: The Implications of the IPPC Directive and the BAT Concept for the Estonian Industry. IIIEE, Lund University, 1998.
BSc thesis: The development of coastal ridges in the north-estonian clint bays. Tartu University, 1993.

8. Main areas of scientific work/Current research topics/Projects and publications

Sustainable use of resources and waste management
Life cycle assessment
Sustainable consumption and production

Projects:

- 2007-2008 - The analysis of the quantity and composition of mixed municipal solid waste in Estonia - Municipal solid waste composition study.
- 2007-2008 - Evaluation of the economic and environmental costs and impacts of landfill tax/charge system in Estonia.
- 2007 - Installation of waste incineration unit in Iru power plant - Life cycle based environmental and economic assessment.
- 2004-2007 - Evaluation of life cycle based environmental impacts and economic cost of alternative scenarios for municipal waste management in Estonia.
- 2004-2007 - Project on Regional Cooperation in Waste Management (RECO) – international waste management and spatial planning project to support the development of management of municipal waste in the Eastern Europe in order to minimize negative environmental effects, lower life cycle costs for waste management and meet EU and national waste policies and legislation (INTERREG IIIB).

Articles included in this thesis:

Moora, H., Stenmarck, Å., Sundqvist, J-O., 2006. Use of Life Cycle Assessment as decision-support tool in waste management planning – optimal waste management scenarios for the Baltic States. Environmental Engineering and Management Journal, September, Vol. 5, No. 3, 2006, p. 445-456

Moora, H., Sundqvist, J-O., Stenmarck, Å., 2007. LCA-based decision-support tool for waste management planning – optimal waste management scenarios for the Baltic States. Conference Proceedings - 3rd International Conference on Life Cycle Management (LCM2007), Zurich, 26.-29.08.2007, University of Zurich at Irchel.

Moora, H., Voronova, V., Reihan, A., 2009. The Impact of Municipal Solid Waste Management on Greenhouse Gas Emissions in Estonia. Leal Filho, W. & Mannike, F. (eds.) Interdisciplinary Aspects of Climate Change. Series: Environmental Education, Communication and Sustainability. Vol. 29. p. 311-325. Peter Lang Publishers.

Moora, H., Lahtvee, V., 2009. Electricity Scenarios for the Baltic States and Marginal Energy Technology in Life Cycle Assessments – a Case Study of Energy Production from Municipal Waste Incineration. Oil Shale. (accepted)

Other publications and reports:

Moora, H., 2007. Installation of waste incineration unit in Iru power plant - life cycle based environmental and economic assessment. SEI-Tallinn. [in Estonian]

Moora, H., 2007. Life cycle based environmental and economic assessment of municipal waste management alternatives in Estonia. SEI-Tallinn.

Moora, H., Jürmann, P., 2008. The analysis of the quantity and composition of mixed municipal solid waste in Estonia - Municipal solid waste composition study. SEI-Tallinn.

Moora, H., Jürmann, P., 2008. The analysis of the quantity and composition of mixed municipal solid waste in Estonia - Packaging waste composition study. SEI-Tallinn. [in Estonian]

Belmane, I., Karaliunaite, I., Moora, H., Uselyte, R., Viss, V. 2003. Eco-design in the Baltic States' Industry. The Nordic Council of Ministers. TemaNord 2003:559

Moora, H., 2001. Packaging and Packaging Waste management in Estonia, Implementation of EPR in Estonia. Conference proceedings: The European Roundtable of Cleaner Production, 2-4 May 2001, Lund, Sweden.

Moora, H., Pallo, T**.,** 1999. Integration of EMS and Cleaner Production Principles in Estonian Industries. Conference proceedings: The European Roundtable of Cleaner Production, September 28 - October 1, 1999, Budapest, Hungary.

Elulookirjeldus

1 Isikuandmed

Ees- ja perekonnanimi:	Harri Moora
Sünniaeg ja -koht:	13.03.1969, Tallinn
Kodakondsus:	Eesti

2 Kontaktandmed

Aadress:	Kentmanni 9-13, Tallinn
Telefon:	+372 5052365
E-posti aadress:	harri.moora@seit.ee

3. Hariduskäik

Õppeasutus	Lõpetamise aeg	Haridus (eriala/kraad)
Lundi Ülikool, Rootsi	1998	Keskkonnajuhtimine ja -poliitika/MSc
Tartu Ülikool	1993	Geoloogia/BSc

4. Keelteoskus (alg-, kesk- või kõrgtase)

Keel	Tase
Eesti keel	Kõrgtase
Inglise keel	Kõrgtase
Vene keel	Kesktase
Soome keel	Kõrgtase

5. Täiendusõpe

Õppimise aeg	Täiendusõppe läbiviija nimetus
28-30.10.2008	Bridging environmental and economic assessments for decision support (II) – Ekspertide seminar ja PhD kursus, Aalborgi Ülikool, Taani
15-21.09.2006	Bridging environmental and economic assessments for decision support (I) - Ekspertide seminar ja PhD kursus, Aalborgi Ülikool, Taani

6. Teenistuskäik

Töötamise aeg	Tööandja nimetus	Ametikoht
alates 1998	Stockholmi Keskkonna Instituut, Säästva Eesti Instituut	Programmi direktor
1995	Geoloogia Keskus, Meregeoloogia ja Geofüüsika osakond	Teadur

7. Kaitstud lõputööd

Magistritöö: IPPC direktiivi ja PVT kontsepti mõju Eesti tööstusele. IIIEE, Lundi Ülikool, 1998.
Bakalaureusetöö: Põhja-Eesti klindilahtedes paiknevate põikvormide kujunemisloost. Tartu Ülikool, 1993.

8. Teadustöö põhisuunad

Ressursside säästlik kasutamine ja jäätmekäitlus
Olelusringi hindamine
Säästlik tarbimine ja tootmine

Projektid:

- 2007-2008 - Eestis tekkinud olmejäätmete (sh pakendijäätmete ja biolagunevate jäätmete) liigilise koostise analüüs Eestis.
- 2007-2008 - Jäätmete saastetasu rakendamise analüüs, uued suunad ja ettepanek uute tasumäärade rakendamiseks aastatel 2010-2015.
- 2007 - Iru elektrijaama jäätmepõletusploki olelusringipõhine keskkonnamõju ja majanduskulude hinnang.
- 2004-2007 - erinevate olmejäätmete käitlusalternatiivide olelusringipõhise keskkonnamõju ja majanduskulu hinnang. Jäätmealase olelusringi hindamise mudeli arendamine.
- 2004-2007 - rahvusvaheline jäätmealane projekt RECO. Ida-Euroopa riikide jäätmekäitlussüsteemide arendamine, olelusringipõhiste keskkonnamõjude ja majanduskulude optimeerimine (INTERREG IIIB).

Käesoleva töö aluseks olevad artiklid:

Moora, H., Stenmarck, Å., Sundqvist, J-O., 2006. Use of Life Cycle Assessment as decision-support tool in waste management planning – optimal waste management scenarios for the Baltic States. Environmental Engineering and Management Journal, September, Vol. 5, No. 3, 2006, p. 445-456

Moora, H., Sundqvist, J-O., Stenmarck, Å., 2007. LCA-based decision-support tool for waste management planning – optimal waste management scenarios for the Baltic States. Conference Proceedings - 3rd International Conference on Life Cycle Management (LCM2007), Zurich, 26.-29.08.2007, University of Zurich at Irchel.

Moora, H., Voronova, V., Reihan, A., 2009. The Impact of Municipal Solid Waste Management on Greenhouse Gas Emissions in Estonia. Leal Filho, W. & Mannike, F. (eds.) Interdisciplinary Aspects of Climate Change. Series: Environmental Education, Communication and Sustainability. Vol. 29. p. 311-325. Peter Lang Publishers.

Moora, H., Lahtvee, V., 2009. Electricity Scenarios for the Baltic States and Marginal Energy Technology in Life Cycle Assessments – a Case Study of Energy Production from Municipal Waste Incineration. Oil Shale. (accepted)

Muud publikatsioonid ja aruanded:

Moora, H., 2007. Kütusena jäätmeid kasutava soojus- ja elektrienergia koostootmisploki rajamine Iru Elektrijaama territooriumile. Aruanne, Iru Elektrijaam, SEI-Tallinn.

Moora, H., 2007. Olmejäätmete käitlusalternatiivide keskkonnamõju ja majanduskulu olelusringipõhine uuring. Uurimistöö aruanne. SEI-Tallinn.

Moora, H., Jürmann, P., 2008. Eestis tekkinud olmejäätmete koostise ja koguste analüüs. Uurimistöö aruanne. SEI-Tallinn.

Moora, H., Jürmann, P., 2008. Eestis tekkinud pakendijäätmete koostise ja koguste analüüs. Uurimistöö aruanne. SEI-Tallinn.

Belmane, I., Karaliunaite, I., Moora, H., Uselyte, R., Viss, V. 2003. Eco-design in the Baltic States' Industry. The Nordic Council of Ministers. TemaNord 2003:559

Moora, H., 2001. Packaging and Packaging Waste management in Estonia, Implementation of EPR in Estonia. Konverentsi kogumik: European Roundtable of Cleaner Production, 2-4 May 2001, Lund, Sweden.

Moora, H., Pallo, T**.,** 1999. Integration of EMS and Cleaner Production Principles in Estonian Industries. Konverentsi kogumik: European Roundtable of Cleaner Production, September 28 - October 1, 1999, Budapest, Hungary.

APPENDIX 3. Selected papers

PAPER I

Moora, H., Stenmarck, Å., Sundqvist, J-O., 2006. Use of Life Cycle Assessment as decision-support tool in waste management planning – optimal waste management scenarios for the Baltic States. Environmental Engineering and Management Journal, September, Vol. 5, No. 3, 2006, p. 445-456

PAPER II

Moora, H., Sundqvist, J-O., Stenmarck, Å., 2007. LCA-based decision-support tool for waste management planning – optimal waste management scenarios for the Baltic States. Conference Proceedings - 3rd International Conference on Life Cycle Management (LCM2007), Zurich, 26.-29.08.2007, University of Zurich at Irchel.

PAPER III

Moora, H., Voronova, V., Reihan, A., 2009. The Impact of Municipal Solid Waste Management on Greenhouse Gas Emissions in Estonia. Leal Filho, W. & Mannike, F. (eds.) Interdisciplinary Aspects of Climate Change. Series: Environmental Education, Communication and Sustainability. Vol. 29. p. 311-325. Peter Lang Publishers.

PAPER IV

Moora, H., Lahtvee, V., 2009. Electricity Scenarios for the Baltic States and Marginal Energy Technology in Life Cycle Assessments – a Case Study of Energy Production from Municipal Waste Incineration. Oil Shale. (accepted)

14. **Alvina Reihan**. Analysis of long-term river runoff trends and climate change impact on water resources in Estonia. 2008.

15. **Ain Valdmann**. On the coastal zone management of the city of Tallinn under natural and anthropogenic pressure. 2008.

16. **Ira Didenkulova**. Long wave dynamics in the coastal zone. 2008.

17. **Alvar Toode**. DHW consumption, consumption profiles and their influence on dimensioning of a district heating network. 2008.

18. **Annely Kuu**. Biological diversity of agricultural soils in Estonia. 2008.

19. **Andres Tolli**. Hiina konteinerveod läbi Eesti Venemaale ja Hiinasse tagasisaadetavate tühjade konteinerite arvu vähendamise võimalused. 2008.

20. **Heiki Onton**. Investigation of the causes of deterioration of old reinforced concrete constructions and possibilities of their restoration. 2008.