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ENVIRONMENTAL ASSESSMENT CHALLENGES OF WASTE MANAGEMENT SYSTEMS

Focus on sensitivity analysis of modelling choices

DISSERTATION

eingereicht an der

LEOPOLD-FRANZENS-UNIVERSITÄT INNSBRUCK
FAKULTÄT FÜR TECHNISCHE WISSENSCHAFTEN

zur Erlangung des akademischen Grades

DOKTORIN DER TECHNISCHEN WISSENSCHAFTEN

Innsbruck, März 2016

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Innsbruck, 18.03.2016

Maria Ortner

Acknowledgements

Thanks to my supervisor Univ.-Prof. Dr.-Ing. Anke Bockreis for her support during my PhD and for giving me the opportunity to integrate my background of environmental management in our working group.

Thanks to Ao.Univ.Prof. Dipl.-Ing. Dr.nat.techn. Stefan Saalhofer for being my second reviewer and for the interesting discussions at the yearly PhD-seminars.

I would like to thank my colleagues at the department of environmental engineering at the Universität Innsbruck who supported and accompanied me while completing my PhD thesis. Special thanks to Dr.-Ing. Wolfgang Müller for his support in the different research projects and the joint recreational activities.

Thanks to the Marietta Blau Grant from the Federal Ministry of Science and Research of Austria for enabling a seven month long research period as a guest researcher at the Department of Environmental Engineering at the Technical University of Denmark (DTU). Thanks to my colleagues at DTU environment who welcomed me warmly in their research group and gave me interesting new inputs for my professional but also personal life.

I also would like to thank the vice rectorate for research of the University of Innsbruck for the financial support for six month of my PhD by the doctoral scholarship for the promotion of young researchers.

Finally I would like to thank my family and friends who reminded me that there is a life outside university. Special thanks to my parents who always encouraged and supported me during my studies. And special thanks to my boyfriend Mario for the support and the PhD-free weekends and evenings.

Kurzfassung

Diese Dissertation zeigt den Einfluss unterschiedlicher Modellierungsentscheidungen auf die ökologische Bewertung von abfallwirtschaftlichen Systemen mittels Sensitivitätsanalysen und deren Anwendung auf verschiedene abfallwirtschaftliche Fallstudien auf.

Nach einer Einleitung zu der ökologischen Bewertung von abfallwirtschaftlichen Systemen, der Methodik der Ökobilanzierung, deren Stärken und Schwächen sowie gesetzliche Verankerung wird anhand der Fallstudie „Landfill Mining“ die geschichtliche Entwicklung dieses abfallwirtschaftlichen Prozesses und die damit verbundenen Herausforderungen an ökologische Bewertungen dargestellt. Anschließend folgt die Vorstellung der unterschiedlichen Arten von Unsicherheiten innerhalb von Ökobilanzen in der Abfallwirtschaft und der in dieser Dissertation angewandten Methodik der Sensitivitätsanalyse zur Erhebung und Reduzierung derselbigen. Im Rahmen der zweiten Fallstudie „Sammlung und Verwertung von Altspeisefetten aus Haushalten in Österreich“ wird diese Methodik angewandt und Erkenntnisse diskutiert.

Mit diesen Grundlagen wird eine umfassende Sensitivitätsanalyse anhand der dritten Fallstudie „Biologische Abfallbehandlungsverfahren für Bioabfall aus der getrennten Sammlung von Haushalten“. Hierbei wurde der Fokus auf Softwareprogramme als Unsicherheiten in der ökologischen Bewertung von abfallwirtschaftlichen Systemen gelegt. Aufbauend auf bisherigen Forschungserkenntnissen wird ein schrittweiser Analyseansatz zur Erhebung des Einflusses des gewählten Softwareprogrammes vorgestellt und für den Vergleich von zwei Softwareprogrammen angewendet. Der Vergleich umfasst eine Analyse der inkludierten Datenbanken, der implementierten Wirkungsabschätzungsmethoden mit den dazugehörigen Charakterisierungsfaktoren, die Zuordnungsschemen von Elementarflüssen, ausgewählte automatisierte Berechnungsmodelle und die damit generierten Massen- und Substanzbilanzen. Abschließend werden Empfehlungen für den Umgang mit Softwareprogrammen in der Erstellung von ökologischen Bewertungen und den damit verbundenen Modellierungsentscheidungen und Unsicherheiten gegeben.

ABSTRACT

This thesis and the included papers show the influence of modelling choices on the environmental assessment of waste management systems by applying sensitivity analysis on different case studies in the field of waste management.

First, the relevance of environmental assessment of waste management systems, the method of life cycle assessment (LCA), the strengths and weaknesses of this concept and its legal basis are introduced. In this context, the historical development of the first case study "Landfill Mining" and the related assessment challenges are presented.

Next, the different types of uncertainties within the life cycle assessment of waste management systems are stated and the applied method of sensitivity analysis to assess and reduce uncertainties is explained. In the course of the second case study, "Collection and utilization of waste cooking oil from households in Austria", the method is applied and findings are discussed.

Based on these findings, a profound sensitivity analysis was conducted on the third case study "Biological waste treatment option for biowaste". Here, the emphasis was put on software tools as an uncertainty factor within the ecological assessment of waste management systems". On the basis of research findings, a stepwise approach for the assessment of the influence of the chosen software tool in the course of an environmental assessment is presented and applied on the comparison of two different software tools. The comparison includes an assessment of the included databases, the offered life cycle impact assessment (LCIA) methods, the implemented characterization factors and inventory system of elementary exchanges, the calculation models implied and their effect on generated mass and substance balances.

Finally, recommendations are given concerning conducting an environmental assessment with supporting software tools and the related modelling decisions and uncertainties.

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List of Abbreviations

AD plant	Anaerobic Digestion plant
CF	Characterization Factors
CM	Characterization Models
CML	Institute of Environmental Sciences of Leiden University
EASETECH	Environmental Assessment System for Environmental TECHnologies
EC	European Commission
EIA	Environmental Impact Assessment
EPD	Environmental Product Declarations
EPLCA	European Platform on Life Cycle Assessment
EU	European Union
FM	Fresh Matter
GaBi	Ganzheitliches Bilanzieren
GHG	Greenhouse Gas
GWP	Global Warming Potential
IC	Impact Categories
ILCD	International Reference Life Cycle Data System
ISO	International Organization for Standardization
JRC	Joint Research Center
LCA	Life Cycle Assessment
LCI	Life Cycle Inventory
LCIA	Life Cycle Impact Assessment
LCT	Life Cycle Thinking
LFM	Landfill Mining
mPE	mili Person Equivalent
NPK	Nitrogen, Phosphorous, Potassium
POFP	Photochemical Ozone Formation Potential
RFD	Refuse Derived Fuels
SC	Sensitivity Coefficient
SFA	Substance Flow Analysis
SR	Sensitivity Ratio
VS	Volatile Solids
WCO	Waste Cooking Oil
WMO	World Meteorological Organization
WMS	Waste management systems
WtE	Waste-to-Energy
WtM	Waste-to-Material

LIST OF PAPERS

The following papers listed below constitute an integral part of this thesis. They are annexed to this thesis in part II of the printed version, but not in the online version due to copyright issues. Copies of the papers may be obtained from journal publishers.

I. Landfill Mining: Objectives and Assessment Challenges
Maria E. Ortner, Julika Knapp, Anke Bockreis (2014)
Published in Waste and Resource Management 167/2, pp. 51 - 61.
DOI: 10.1680/warm.13.00012
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This paper was awarded with the Thomas Telford Premium Prize for best paper in the journal at the ICE Publishing Awards 2015

II. Ökologische Bewertung unterschiedlicher Verwertungspfade von Altspeisefetten aus Haushalten in Österreich (in German)
Maria E. Ortner, Wolfgang Müller, Irene Schneider, Anke Bockreis (2015)
Published in Österreichische Wasser- und Abfallwirtschaft 67/9-10, pp. 359 – 368,
DOI: 10.1007/s00506-015-0259-2
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III. Environmental assessment of three different utilization paths of waste cooking oil from households (2016)
Maria E. Ortner, Wolfgang Müller, Irene Schneider, Anke Bockreis (2016)
Published in Resources, Conservation and Recycling 106, 59–67
DOI: 10.1016/j.resconrec.2015.11.007
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IV. The greenhouse gas and energy balance of different treatment concepts for bio-waste
Maria E. Ortner, Wolfgang Müller, Anke Bockreis (2013)
Published in Waste Management & Research 31/10, Supplement, pp. 46 - 55.
DOI: 10.1177/0734242X13500518
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V. Influence of the choice of software tools on LCA modelling decisions and LCA results
Maria E. Ortner, Anders Damgaard, Charlotte Scheutz, Anke Bockreis (submitted)
Submitted to the International Journal of Life Cycle Assessment

The following additional publications also resulted from the work during the PhD study. Although they are not part of this thesis, their content is also related to the topics presented here and could be of interest to the readers. Some publications are not that closely related to the topic, but demonstrate the broad range of research done during the PhD study. Copies of the papers may be obtained from the journal publishers or conference organizers.

1. Ökobilanzieller Vergleich verschiedener Sammel- und Verwertungsoptionen von Altspeisefetten aus Haushalten (in German)
   Maria E. Ortner, Wolfgang Müller, Anke Bockreis (2015)
   Published in proceedings of 5. Wissenschaftskongress Abfall- und Ressourcenwirtschaft, Innsbruck, Austria, pp. 239 - 242.
   ISBN: 978-3-902936-66-0

2. Kreislaufführung von Holzaschen - Verwertung im alpinen Wald (in German)
   Marina Fernández-Delgado Juárez, **Maria E. Ortner**, Heribert Insam, Alexander Knapp (2014)
   Published in proceedings of DepoTech 2014. Abfallwirtschaft, Abfallverwertung und Recycling, Deponietechnik und Altlasten, Leoben, Austria, pp. 415 - 418.
   ISBN: 978-3-200-03797-4, S.

3. Ökobilanzielle Bewertung von Holzascheausbringung im Alpinen Raum (in German)
   Maria E. Ortner, Wolfgang Müller, Anke Bockreis (2014)
   Published in proceedings 4. Wissenschaftskongress Abfall- und Ressourcenwirtschaft, Münster, Germany, pp. 193 - 196.
   ISBN: 978-3-9811142-4-9

4. The Challenges of Landfill Mining
   Maria E. Ortner, Anke Bockreis (2014)
   Published in proceedings of SUM 2014 – Second Symposium on Urban Mining, Padova, Italy.
   ISBN: 978-8-862-65031-1

5. Landfill Mining - Stand der Forschung in Deutschland und in Österreich (in German)
   Anke Bockreis, **Maria E. Ortner** (2013)
   Published in proceedings of 2. Darmstädter Ingenieurkongress Bau und Umwelt. Aachen, Germany, pp. 681 - 684.
   ISBN: 978-3-8440-1747-2

6. The greenhouse gas and energy balance of different treatment concepts for bio-waste
   Maria E. Ortner, Wolfgang Müller, Anke Bockreis (2013)
   Published in proceedings of ISWA World Congress Vienna, Austria.
   ISBN: 978-3-200-03229-3

7. Landfill Mining: Bisherige Entwicklung und Potentiale (in German)
   Maria E. Ortner, Anke Bockreis (2013)
   Published in proceedings of Österreichische Abfallwirtschaftstagung 2013, Innsbruck, Austria
   ISBN: 978-3-902810-76-2.

8. Verfahrensvergleich zur Verwertung biogener Abfälle (in German)
   Anke Bockreis, **Maria E. Ortner**, Wolfgang Müller (2013)
   Published in proceedings of Österreichische Abfallwirtschaftstagung 2013, Innsbruck, Austria.
   ISBN: 978-3-902810-76-2.

9. CO_2- und Energiebilanz verschiedener Verfahren der Bioabfallverwertung (in German)
   Wolfgang Müller, Anke Bockreis, **Maria E. Ortner** (2012)
   Published in proceedings of DepoTech 2012, Abfallwirtschaft, Abfalltechnik, Deponietechnik und Altlasten, Leoben, Austria, pp. 107 - 112.
   ISBN: 978-3-200-02821-0

The results of the scientific work were also presented at various conferences. These, hereby listed conferences are part of the scientific exchange of ideas and experience and form an invaluable part of the formation of this PhD thesis. The presentations were given by the author of this thesis during her PhD study.

1. Vergleich von Softwareprogrammen für die ökobilanzielle Modellierung und Analyse von abfallwirtschaftlichen Prozessen (in German)
   Ökobilanzwerkstatt 2015, Pforzheim, Germany, 15.09.2015.

2. Ökobilanzielle Bewertung in der Abfallwirtschaft mit Fokus auf Abfällen biogenen Ursprungs (in German)
   Doktorandenseminar der Abfallwirtschaft 2015, Wien, Austria, 10.09.2015.

3. Ökobilanzieller Vergleich verschiedener Sammel- und Verwertungsoptionen von Altspeisefetten aus Haushalten (in German)
   5. Wissenschaftskongress Abfall- und Ressourcenwirtschaft, Innsbruck, Austria, 19.03.2015.

4. Ökobilanzielle Bewertung der Sammlung und Verwertung von Altspeisefetten (in German)
   Ökobilanzwerkstatt 2014, Dresden, Germany, 09.09.2014.

5. Ökobilanzielle Bewertung in der Abfallwirtschaft mit Fokus auf Abfällen biogenen Ursprungs (in German)
   Doktorandenseminar der Abfallwirtschaft 2014, Weimar, Germany, 20.10.2014.

6. Ökobilanzielle Bewertung von Holzascheausbringung im Alpinen Raum (in German)
   4. Wissenschaftskongress Abfall- und Ressourcenwirtschaft, Münster, Germany, 27.03.2014.

7. Die Bewertung der Ressource Wasser im Rahmen von Ökobilanzen (in German)
   Workshop ,,Der Wasserfußabdruck als Beispiel für ökologische Bewertungsindikatoren von Städten und Gemeinden", Innsbruck, 20.05.2014.

8. Ökobilanzielle Bewertung von Holzascheausbringung im Alpinen Raum „Projekt AshTreat" (in German)
   Ökobilanzwerkstatt 2013, Graz, Austria, 23.09.2013.

9. The greenhouse gas and energy balance of different treatment concepts for bio-waste
   A Life-Cycle Perspective on Landfill Mining (poster presentation)
   ISWA World Congress 2013, Wien, Austria, 09.10.2013

10. Ökobilanzielle Bewertung in der Abfallwirtschaft (in German)
    Doktorandenseminar der Abfallwirtschaft 2013, Duisburg, 03.11.2013.

11. Landfill Mining: Bisherige Entwicklung und Potentiale (poster presentation in German)
    Österreichische Abfallwirtschaftstagung 2013, Innsbruck, Austria, 18.04.2013.

During the PhD study I worked on different scientific research projects, which the herein mentioned or included publications are derived from.

- Project Bioenergy from the kitchen – Economic and ecological assessment of the collection and utilization of waste cooking oil funded by the EU-program INTERREG – Bavaria Austria

- Project ENERALP - Optimization of alpine energy systems through spatial planning concepts and standardized methods for energy potential analysis
    - Part E: Bio-waste from restaurants and canteens as a resource for energy production
    - Part F: Organic Fraction of Municipal Solid Waste as Potential Renewable Energy- and Feedstock-Resource

  Funded by the K1-competence center „alpS – Centre for Climate Change Adaptation“ from the COMET-Programme of the Austrian Federal Ministry of Science, Research and Economy and the Austrian Ministry for Transport, Innovation and Technology.

- Project BIENE - Bioenergy systems in alpine regions:
    - Part D: AshTreaT - Wood Ash Recycling in Alpine Forests

  Funded by the K1-competence center „alpS – Centre for Climate Change Adaptation“ from the COMET-Programme of the Austrian Federal Ministry of Science, Research and Economy and the Austrian Ministry for Transport, Innovation and Technology.

Moreover I co-supervised the following master thesis:
- Paul Neururer (2014): Determination of feasibility criteria for landfill mining

A part of the research for the PhD was supported by the Marietta Blau Grant of the Federal Ministry of Science and Research of Austria. The grant enabled a seven month long research period as a guest researcher at the Department of Environmental Engineering at the Technical University of Denmark (DTU). During this period (March – September 2015) the research published in paper V was conducted.

Six month of my PhD were sponsored by the doctoral scholarship for the promotion of young researchers of the University of Innsbruck.

PART I: THESIS

1. INTRODUCTION

“Rather than seeing models as describing literal truth, we ought to see them as convenient fictions which try to provide something useful.”
David Frame

1.1. BACKGROUND AND MOTIVATION

With the extensive introduction of the sustainability concept in the years around 1980 the environmental consequences from waste management systems have become a relevant topic for governments, local authorities, plant owners and other stakeholders. Different quantitative assessment tools can serve to provide informed and science-based support, for more environmentally sustainable decision-making in waste management. Ever since the EU Waste Framework Directive (2008/98/EC) (European Parliament 2009) stated that Life Cycle Thinking (LCT) can be used to complement the waste hierarchy in order to make sure the best overall environmental option is identified (Article 4(2)), Life Cycle Assessment (LCA) is one of the leading environmental assessment methods for waste management systems.

During the last decade the number of LCA studies of waste management systems has increased significantly, leading to somewhat different conclusions and recommendations on the same subject of investigation. Reasons for this can be found in the methodology which has its limitations. Besides objective and science based modules, also subjective modules and decisions are included. The LCA practitioner has a central role when it comes to the reliability and uncertainty of an LCA study and it is important to understand that the environmental information generated is neither complete, nor absolutely objective or accurate. However, this also applies to other methods for environmental systems analysis as well.

In order to quantify the impact of modelling decisions of the LCA practitioner, profound sensitivity analyses were conducted in the course of different ecological assessments of defined waste management case studies. The approach of sensitivity analysis was chosen, because this phase of an ecological assessment is based on scientific findings and knowledge and not influenced by subjective values.

1.2. RESEARCH OBJECTIVES

The objectives of this thesis were to:

– Investigate the development of a current relevant waste management system and identify the related challenges for state of the art ecological assessment practices on the example of "Landfill Mining".
– Identify sensitivities of ecological assessment studies in the field of waste management on the example of the waste fractions "biowaste from separate collection in households" and "waste cooking oil from separate collection from households".
– Assess the role of modelling choices and the type of software tool applied as uncertainty factors in life cycle assessment studies of a defined waste management system.

1.3. STRUCTURE OF THE THESIS

The thesis is structured in five chapters:

– Chapter 2 describes the context of environmental assessment of waste management system, presents the method of Life Cycle Assessment (LCA), related legislative regulations and the challenges of waste management systems for the ecological assessment (Papers I)
– Chapter 3 presents the work performed on sensitivities and uncertainties within waste-LCA studies on the example of the waste fraction "waste cooking oil from households" (Paper II, III).

– Chapter 4 presents modelling choices and LCA software tools as sources of uncertainty within waste LCA (Paper IV, V)
– Chapter 5 gives conclusions and recommendations for further work.

1.4. CONTENTS OF THE PAPERS OF THIS THESIS

The five included papers use the data of three different case studies (elaborated in the corresponding chapters) investigating or applying environmental assessment approaches on different waste management systems reflecting assessment challenges and different types of uncertainties. The distribution of the included papers on these topics can be seen in Figure 1.

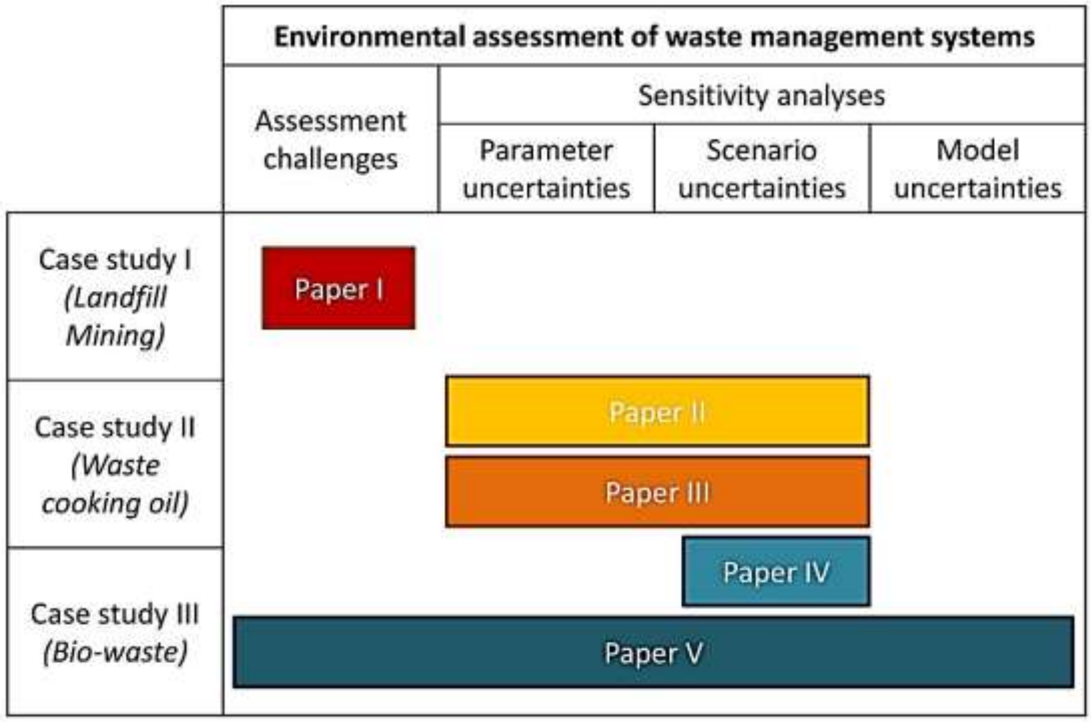

Figure 1: Topics and case studies for the included papers (Paper I: Ortner et al. 2014; Paper II: Ortner et al. 2015; Paper III: Ortner et al. 2016a; Paper IV: Ortner et al. 2013; Paper V: Ortner et al. 2016b)

Case study I: "Landfill Mining"

The term landfill mining (LFM) characterizes the process of dismantling a landfill with the aim to recover untreated wastes and to utilize them as resources, such as secondary raw material or refuse derived fuel (RDF). In literature, the first LFM project documented dates back to the mid of the last century. The sector of waste and resource management has evolved significantly through major developments in the field of technology, patterns of resource consumption, standard of living, composition of landfilled waste and regulatory frameworks ever since then. Accordingly did the environmental and economic performances of LFM activities.

(Elaborated in more detail in chapter 2.2. Case study I: Landfill Mining).

Paper I (Ortner et al. 2014) elaborates the development of LFM as a current relevant waste management system and identifies challenges for state-of-the-art ecological and economical assessment practices of the same. The study includes a literature review on LFM focusing on the objectives, trends and findings of 60 LFM projects conducted during the period 1953—2009 and their implication for ecological assessment. An emphasis is put on the effect of legislative regulation changes on LFM activities and the current situation and potentials in Austrian and German landfills. The results showed that a main objective has always been the recycling of excavated materials. However, the focus shifted from using organic and mineral fractions for compost or cover material to the extraction of more valuable fractions like metals. Further categorized key drivers and objectives were environmental pollution concerns, land recycling, recovery of landfill volume, post-closure activities and landfill rehabilitation. The frequency distribution of the classified objectives varies over time. The review of the state of the art of ecological assessment studies in the field of LFM of the last decade showed that one of the major challenges in evaluation are the integration of long-term environmental impacts (after 100 years), the quantification of socio-economic effects and the availability of reliable and suitable data sources.

Case study II: "Utilization of waste cooking oil from households"

Waste cooking oil (WCO) from households mainly results from cooking and frying food with vegetable oil and is classified as a domestic waste stream. Due to its high heating value WCO is a suitable feedstock for energy generation from a waste-to-energy perspective. However, inadequate disposal of waste cooking oil through sewage systems can cause economic and environmental problems, such as impeding sewage treatment at wastewater treatment plants (WWTP). In addition, it leads to the loss of a valuable resource with high energy content.

(Elaborated in more detail in chapter 3.3.
Case study II: Waste cooking oil from households)

In **Paper II and III** (Ortner et al. 2015; Ortner et al. 2016a) sensitive model parameters in the course of an ecological assessment (following the principles of the Life Cycle Assessment methodology) of three different utilization concepts for WCO are elaborated. The utilization options investigated were (1) the conversion of WCO to biodiesel, (2) the direct combustion in a cogeneration plant and (3) the production of biogas within an agricultural biogas plant. The comparative assessment showed that all three options resulted in a reduction of greenhouse gas (GHG) emissions. The ranking of the greenhouse gas balances between the different scenarios was the esterification of the WCO (scenario 1; savings of 3,089 kg CO_2eq t^{-1} WCO) before the utilization of WCO in a cogeneration plant (scenario 2; savings 2,967 kg CO_{2eq} t^{-1} WCO) before using WCO as a co-substrate in an agricultural biogas plant (scenario 3; saving of 1,459 kg CO_{2eq} t^{-1} WCO). However, a sensitivity analysis showed that this ranking can change when certain sensitive parameters are varied. In **Paper III** the sensitivity analysis was extended. For the assessment of parameter uncertainties, the method of perturbation analysis and for the quantification of scenario uncertainties the method of scenario analysis were applied. Focus was put on resulting energy outputs and their substitution potential concerning primary production of fossil energy carriers. The findings endorse that the definition of reference systems in the course of an environmental assessment has a major influence on the overall results.

Case study III: "Biological waste treatment options for biowaste"

In accordance with the Waste Framework Directive (European Commission, 2008) the term biowaste used in this thesis includes garden and park waste, and food and kitchen waste from households, restaurants, caterers retail premises, as well as comparable waste from food processing plants. Although regarded as the least desirable option, around 40 % of the generated biowaste is still landfilled in the EU (in some member states still 100 %). In contrast, the process of anaerobic digestion generates a renewable energy source (biogas) and a residual material (digestate). Both digestate and compost from aerobic treatment produced from separately collected biowaste can be utilized in agriculture and horticulture as fertilizer and soil conditioner and thus preserve natural resources such as peat from being extracted.

(Elaborated in more detail in chapter 4.1.2.
Case study III: biowaste from households)

Paper IV (Ortner et al. 2013) investigates the greenhouse gas and energy performance of biowaste treatment plants for three characteristic biowaste treatment concepts: composting, biological drying for the production of biomass fuel fractions, and anaerobic digestion. The ecological assessment was conducted in accordance with the LCA method defined in the ISO 14044. To enable a direct comparison, the mass balance and inventory were calculated for all process concepts assuming the same biowaste amounts and properties. All three concepts contributed to a reduction of greenhouse gas emissions and show a positive balance for cumulated energy demand (CED). However, in contrast to other studies the environmental advantages of anaerobic digestion compared to composting were smaller as a result of accounting for the soil improving properties of compost. This finding can be classified into the category of the so called scenario uncertainties, which state that the definition of system boundaries (in this example the inclusion of soil improving properties of compost) can influence the final result substantially.

The scientific findings of **Paper I-III** and the inventory and assessment results of **Paper IV** served as a basis for the case study modelled in the course of the software comparison conducted in **Paper V** (Ortner et al. 2016b). Paper V assesses how sensitive environmental assessment results are to the usage of different types of software tools and to different software users and their modelling choices during the conduction of an environmental assessment study. Firstly, the software models were directly compared when modelling case study III on the basis of the same inventory data and the same methods of Life Cycle Impact Assessment (LCIA) (list of ILCD recommendations). Secondly, two user groups were defined and compared within each software tool. For assessing the effect of different modelling choices the usage of aggregated datasets versus own modelled datasets was analyzed with a process variation analysis. Lastly, the consequences of using different background data were assessed. The results showed that modelling the same case study based on the same inventory data did not lead to the same final results in all impact categories considered. Reasons were found in differently generated mass and substance balances due to different calculation models and methods within the two software tools. Furthermore, the LCIA characterization methods listed under "ILCD recommendations" in both software tools differed in the version applied, the number of elementary changes included and accounting of long-term emissions. The modelling choice of using aggregated or own modelled data had a higher influence on the overall results than the choice of the background data.

2. CONTEXT

"The whole is more than the sum of its parts."

Aristoteles

2.1. ENVIRONMENTAL ASSESSMENT OF WASTE MANAGEMENT SYSTEMS (WMS)

Waste management systems (WMS) are complex phenomenon with a range of consequences for society and the involved stakeholders. Ever since the introduction of the concept of sustainability the environmental performance is, besides social and economic parameters, of great relevance. There exist a number of tools for the assessment of environmental impacts and for supporting decision making in WMS, such as Environmental Impact Assessment (EIA), Substance Flow Analysis (SFA) and the here in more detail discussed Life Cycle Assessment (LCA) (Finnveden and Moberg 2005; Finnveden et al. 2007). LCA allows expanding the perspective beyond the WMS itself. Environmental consequences from WMS often derive to a great extend from the impacts of surrounding systems rather than from the emissions from the WMS itself. The broad perspective of LCA makes it possible to take the significant environmental benefits that can be obtained through different waste management processes into account (Ekvall et al. 2007). These benefits include e.g. energy recovery from thermal waste utilization, which reduces the need for fossil energy sources and material recovery from recycling processes substituting production of primary raw material. Because of this, LCA has gained in acceptance as a tool for waste management planning on local level and policy-making at national and international levels (which will be discussed in more detail in 2.1.2. Legal framework of LCA in waste management).

In the scientific literature a number of publications can be found describing how to apply the method of LCA on WMS ((Barton et al. 1996; Finnveden 1999; Del Borghi et al. 2009); summarizing the status quo of LCA in WMS (Laurent et al. 2014a; Laurent et al. 2014b) and discussing the limitation of the tool in the context of WMS (Ekvall et al. 2007; Clavreul et al. 2012; Lazarevic et al. 2012). One of the main results of the latter is that it is important to understand that the environmental information generated is neither complete, nor absolutely objective or accurate (which applies also to other methods for environmental systems analysis) (closer discussion in section 2.3. Limitations of LCA of WMS).

In the following an overview of the method of LCA and the legal framework of LCA in the context of waste management will be given.

2.1.1. Life Cycle Assessment (LCA)

The method of life cycle assessment is defined as the "compilation and evaluation of the inputs, outputs and the potential environmental impacts of a product system throughout its life cycle" (ISO 2006). Within this definition the term 'product' can include also services such as waste management.

With the ISO 14044 an international standard for LCA has been developed, providing a framework, terminology and some methodological choices. The methodological framework comprises four phases, namely the goal and scope definition, the inventory analysis, the impact assessment and the life cycle interpretation (Figure 2).

In the goal and scope phase the objective of the study, the system boundaries, the functional unit (reference unit, e.g. the provision of a service within a specified context) and modelling assumptions are defined in detail. During the Life Cycle Inventory (LCI) analysis the inputs and outputs of the system are quantified. The LCI data of a studied system usually includes a simple mass flow model, relating material outputs to material inputs and an inventory of so called elementary exchanges (emissions and resources used). Furthermore interactions with other processes or systems are taken into consideration, such as consumption of heat, electricity, auxiliary goods and services.

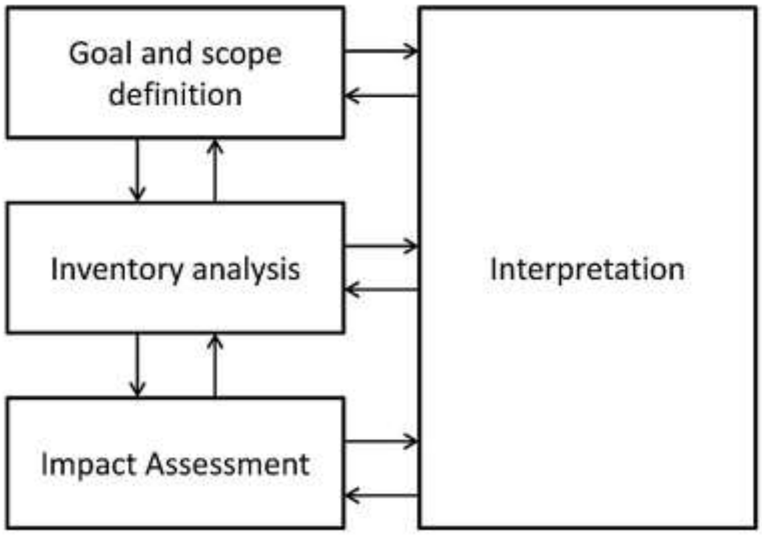

Figure 2: Phases of a LCA according to the ISO framework (ISO 2006).

The Life Cycle Impact Assessment (LCIA) converts the data from the LCI into environmental impacts. The LCIA phase is sub-divided into analytical steps, some of which are regarded as optional. The obligatory elements of classification and characterization are based on more or less traditional natural science. Their aim is to describe the contribution from the system assessed to a number of environmental impact categories, such as global warming potential, resource depletion, human and eco-toxicity. One of the optional elements of LCIA is called normalization which consists of dividing the characterized impacts by certain normalization references. This step allows the scaling of the impacts relatively to the impact of, e.g. a population, thus enabling for some degree a comparison across different impact categories. The findings of the LCI and LCIA phases are evaluated in relation to the defined goal and scope in order to reach conclusions and recommendations during the interpretation phase. LCA is an iterative process, this means that e.g. the interpretation phase can be performed at all stages for several times in order to redefine the goal or the scope of the study, or to refine data collection. A number of explaining handbooks are available (e.g., Guinée et al. 2004; European Commission -- Joint Research Centre -- Institute for Environment and Sustainability 2010), as well as scientific reviews of previous developments (Pennington et al. 2004; Rebitzer et al. 2004; Finnveden et al. 2009).

2.1.2. LEGAL FRAMEWORK OF LCA IN WASTE MANAGEMENT

In the European Union (EU) a number of life-cycle-based or related policies exist (e.g. “Thematic Strategy on the prevention and recycling of waste” (European Commission 2005); Integrated Product Policy Communication (European Commission 2003)). The EU Waste Framework Directive (2008/98/EC) (European Parliament 2009) states the general principles of good management and handling of waste in the community. In article 4(1) the five-step waste hierarchy defines the legally binding priority order for waste management (waste prevention before preparing waste for re-use, followed by recycling and other recovery, and disposal (such as landfilling) as the least desirable option). Article 4(2) constitutes that Life Cycle Thinking (LCT) can be used to complement the waste hierarchy in order to make sure the best overall environmental option is identified.

Furthermore, the European Platform on Life Cycle Assessment (EPLCA) was established by the European Commission (EC). The platform aims at increasing the availability of quality-assured life-cycle data and published several waste specific LCA guide lines (JRC European commission 2011a; JRC European commission 2011b; JRC European commission 2011c). On an international level countries like Australia, Canada, Brazil, United States of America or Thailand consider life cycle approaches as politically relevant by means of national and international ecolabelling schemes, carbon-footprint labels, Environmental Product Declarations (EPD), ecodesign activities, and various resource protection and solid waste management considerations (Wolf et al. 2012).

Laurent et al. (2014a) conducted a profound literature review on the state-of-the-art of LCA studies in the field of WMS. The evolution over time of the scientific publication of waste-LCA studies showed a rapid increase between 2003 and 2009. The authors correlated this development with the above listed inclusion of legislation striving for more Life Cycle Thinking in decision making in the EU. The influence of legislative regulations on waste management practices and assessment approaches was analyzed in **Paper I** (Ortner et al. 2014) and the results are presented in the following two chapters.

2.2. CASE STUDY I: LANDFILL MINING

Paper I analyzed the influence of legal requirements on waste management practices and the associated assessment challenges. The analysis was performed on the example of the introduction of the EU Landfill Directive and its consequences for landfill mining initiatives **(case study I)**. In a literature review the development of main objectives for conducting LFM projects over time were identified in order to create a basis for the future selection of adequate assessment tools. 60 LFM projects conducted during the period 1953—2009 were analyzed. Stricter legislative regulations and framework conditions flagged the international field of waste management in the 1990ies. In the European Union an important milestone was the introduction of the EU Landfill Directive in 1999 (Directive 1999/31/EC on Landfill of Waste, Council of the European Union). The Landfill Directive established stringent technical requirements for landfills and limited the quantity of biodegradable municipal waste to be landfilled. As a consequence, the period assessed was divided into before and after the year 2000 and a focus was put on examining of whether and what kind of shift in interest for LFM activities occurred because of this new regulations.

Figure 3 Figure 3 (a) presents the defined categories of objectives for LFM projects and their relative frequency distribution in percentages. For one fifth of the assessed projects the reuse of recovered material (in literature also referred to as Waste-to-Material, WtM) and energy recovery from excavated material (Waste-to-Energy, WtE) were the main drivers. Environmental pollution concerns were primary focused on by up to 22 %. These LFM projects involved measurements concerning surface and groundwater protection, hazard control in terms of landfill stability, as well as landfill gas and leachate emissions. Pilot and demonstration studies assessing the technical and economic feasibility of LFM projects contributed to 16%. 15 % of the projects observed listed no specified motives. The recovery of landfill volume and landfill rehabilitation measurements, which include the processes of reclaiming, treatment, reassembly or relocation of the excavated material, contributed to 8 % and 11 %. The objective of cultivating land space for further utilization within urban development plans, such as the creation of parks or industrial areas is represented by 7 % of all projects. An insignificant percentage stated the reduction in post-closure activities and costs as the main target.

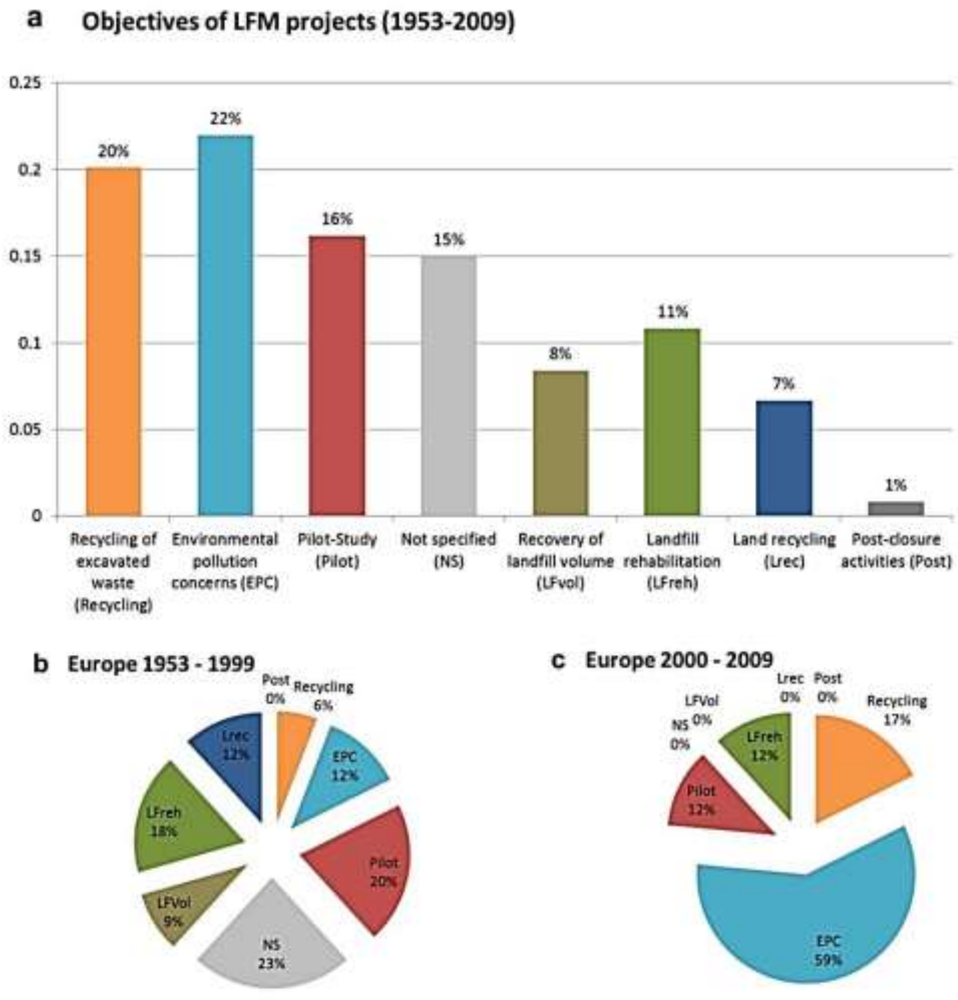

Figure 3: Categorization of the main objectives of LFM projects (a) in the period of 1953 – 2009 on an international level (b) in the time before the introduction of the EU Landfill Directive (1999/31/EC) (Europe) (c) after the year 1999 (Europe). (Relative frequency in %, including multiple motives) (from Paper I)

The evaluation of the development in European LFM activities before and after the introduction of the EU Landfill Directive 1999 (Figure 3 b and c) showed that environmental pollution concerns sharply increased from 12 % to 59 %. The objective of material recycling grew, partly also due to recent developments in the state of the art technologies for separating and recycling of waste, which have improved the level of recycling. It should be stressed that the focus has moved from using organic and mineral fractions for compost or cover material to the extraction of more valuable fractions like metals. Landfill rehabilitation activities declined and others such as land recycling or creation of more landfill volume disappeared completely.

This shift in objectives is also reflected in the different stakeholders participating in the field of LFM. Due to improved technologies for waste recycling, government regulations aiming at waste reduction and the declining long-term development for solid waste to be disposed of at landfills leads to a shrinking market size for companies owing landfills in Europe. Consequently landfill owner are looking for alternative business opportunities (Van der Zee et al. 2004). For local governments local pollution concerns or the recovery of land for urban development strategies in densely populated areas are of higher relevance (Van Passel et al. 2012). Other stakeholder groups are residents, companies processing secondary raw materials, environmental groups or owners of local power plants. All these interest groups pursue different aims and interests. However, within a decision-making process the aim of and incentives for a landfill mining project considerably influence the assessment method applied and as a consequence the output on which the decision is based (Van der Zee et al. 2004; Frändegård et al. 2012; Van Passel et al. 2012). This is one of the challenges and limitations for LCA in WMS which will be discussed in more detail in the following sub-section.

2.3. LIMITATIONS OF LCA OF WMS

Beside the strengths and advantages of the LCA approach discussed earlier, there are a number of weaknesses LCA practitioners and clients working with LCA results should be aware of.

Firstly, LCA is not completely an objective assessment tool (Miettinen and Hämäläinen 1997). There are inherent subjective parts which are influenced by the values and perspective of the LCA practitioner and LCA commissioner. The goal and scope definition includes subjective elements, such as modelling choices related to system boundaries and functional unit, that should be guided by the preferences of the real decision makers and the information requirements related to the application. **Paper I** showed that different stakeholders within the waste management system follow different objectives. Municipalities can use the tool when deciding on how to design their waste management system, which collection scheme should be implemented, how and where the collected waste should be treated or disposed. For companies the assessment of the efficiency and effectiveness of their waste technologies, e.g. energy consumption or degree of purity of their secondary raw material, is of greater interest. Legislative units can use the tool in order to define limits of emissions or harmful substances. Residents and Non-governmental organization usually are interested in the regional consequences of e.g. a thermal treatment facility.

An objective classified step within LCA is the compilation of system inputs and outputs and keeping mass and energy balances during the life cycle inventory phase. The LCIA phase includes objective and subjective steps. The objective parts are classification and characterization, which should be based on scientific knowledge of different impact pathways and environmental models. Normalization, weighting and also the phase of interpretation are considered as less scientific and objective parts, where the preferences and values of the decision makers usually are the guiding factors.

Secondly, there exist general methodological limitations for LCA which have been profoundly discussed in literature (Finnveden 2000; Reap et al. 2008b; Reap et al. 2008a). Table 1 gives an overview of general LCA limitations and states literature references for detailed information. However, in this work only the waste specific limitations and problems will be discussed in more detail in the following chapters.

Table 1: General limitations of LCA studies (based on Reap et al. 2008a)

LCA phase	Problematic decisions	Literature
Goal and scope definition	– Definition of functional unit – Boundary selection (spatial, temporal, technological) – Alternative scenario consideration (modelling of a business-as-usual scenarios as reference) – Choice of attributional or consequential modelling – Lack of social and economic impacts	(USEPA 2006; Reap et al. 2008a; Zamagni et al. 2012)
Life Cycle Inventory	– Allocation – Cut-off criterias – Local technical uniqueness	(Ekvall and Finnveden 2001; Heijungs and Guinée 2007; Reap et al. 2008a)
Life Cycle Assessment	– Impact category and methodology selection – Spatial variation – Local environmental uniqueness – Dynamics of environment – Time horizons	(Reap et al. 2008b)
Interpretation	– Weighting and valuation – Uncertainty in the decision process	(Reap et al. 2008b)
All	– Data availability and quality – Data source selection – Limits due to time and budget restrictions	(Weidema and Wesnaes 1997; Guo and Murphy 2012)

Uncertainty and sensitivity analysis are commonly used to point out the potential shortcomings and limits of a LCA study to decision makers. This topic will be discussed in more detail in the next chapter.

3. Sensitivity Analysis within LCA of WMS

"In this world nothing can be said to be certain,
except death and taxes."
Benjamin Franklin

Models are simplifications or approximations of the reality. LCA practitioners have to decide on system boundaries and assumptions in order to depict complex natural and anthropogenic systems. These assumptions should be as simple as possible and as complex as needed. As a consequence a LCA study is always subject of uncertainties.

Different strategies and methods exist for assessing uncertainties, which can be divided in two main objectives: uncertainty quantification and sensitivity analysis. The aim of **uncertainty quantification** is to describe the range of possible outcomes with a given set of inputs, wherein each input has some uncertainty. The robustness of the overall result is in the center of assessment e.g. in order to give decision makers a profound bases. **Sensitivity analysis** on the other hand aims to describe how sensitive the outcome variables are to variation of individual input parameters. Given multiple input parameters, sensitivity analysis can help to determine which ones drive the majority of the variation in the outcome. The purpose lies in the reduction of uncertainty.

The focus of the thesis lies in the sensitivity analysis of WMS. In this chapter well established methods of sensitivity analysis, different types of uncertainties and the results of assessing them in the course of **case study II** are presented.

3.1. METHODS OF SENSITIVITY ANALYSIS

The ISO (International Organization for Standardization) standard defines sensitivity analysis as the determination of the influence of variations in assumptions, methods and data on the results. Alerted assumptions, methods or data are applied on the results obtained and compared to the original results. Typically the influence on the results of varying the assumptions and data by a range of e.g. +/- 25 % is checked. The sensitivity can be expressed as the absolute deviation of the results or as the percentage of change. Significant changes in the results, e.g. larger than 10 %, are identified.

In this work the proposed methodology for sensitivity analysis in LCA of WMS of Clavreul (2013) is applied as this method was development on the basis of a profound literature study and analysis of past and present methods and tools for uncertainty quantification and sensitivity analysis within waste LCA. The author recommends local one-at-a-time approaches, because these calculations are simple to implement and results easy to communicate. These are the reasons why these techniques have been the most used among the scientific community for years (Saltelli et al. 2008). It needs to be stated that there exist limitations in particular related to non-linearity in waste-LCA models.

The method from Clavreul (2013) recommends to begin with a contribution analysis followed by a sensitivity analysis itself consists of two steps, namely the perturbation analysis and the scenario analysis. In the following these methods are explained in more detail.

3.1.1. Contribution analysis

The contribution analysis, also sometimes called dominance analysis or analysis of key issues, is a common method presented by Heijungs and Kleijn (2001). By decomposing the LCA result (characterized, normalized or weighted impact) into its individual process contributions a quick overview of the important contributors is derived. All processes with positive and negative environmental impacts are subdivided into their sub-components. This allows a detailed analysis of a process or even substance level. From an application-oriented point of view, opportunities for the redesign of products or processes can be derived from the knowledge of the share of a certain process or life cycle stage in an impact category or an emission. For analysis-oriented uses the determination of the elements that contribute most to a certain impact category or to an emission and the data corresponding to those elements is of higher interest.

3.1.2. Perturbation analysis

A perturbation analysis aims on the identification of the effect of a single parameter change on the overall result (Heijungs and Kleijn 2001). Each single parameter value is marginally varied and the variation of the result is calculated. For analysis-oriented uses this analysis method provides a list of input parameters of which small imprecisions have significant consequences on the final results. Thus, the LCA practitioner knows where to prioritize in a more detailed analysis and where not. For application-oriented uses the results of this sensitivity analysis step suggests ideas for product or process improvement.

Two ratios are of special interest: the sensitivity coefficient (SC) and the sensitivity ratio (SR). The sensitivity coefficient expresses the ratio between the two absolute changes:

$$SC = \frac{\Delta result}{\Delta parameter} \quad (1)$$

The sensitivity ratio expresses the ratio between the two relative changes:

$$SR = \frac{\frac{\Delta result}{initial\_result}}{\frac{\Delta parameter}{initial\_parameter}} \quad (2)$$

A SR of 2 expresses that an increase of a parameter value by 10% leads to an increase of the overall result by 20%. **Paper II** provides a detailed analysis of parameter sensitivities on **case study II**.

3.1.3. SCENARIO ANALYSIS

The scenario analysis consists in testing different options within a system individually and observing the effect of these changes on the final result. The results received after variation can be compared with the baseline results in order to identify the uncertainties that change some scenario results significantly or the ranking between alternatives.

3.2. TYPES OF UNCERTAINTIES

Several authors have suggested typologies to describe the different types of uncertainties in LCA. The structure of this work follows a well-established definition introduced by Huijbregts (1998) which divides uncertainties into three groups: (1) parameter uncertainties refer to the uncertainty in values due to e.g. inherent variability, measurement imprecision or paucity of data; (2) scenario uncertainties are due to the necessary choices made to build scenarios; and (3) model uncertainties are due to the mathematical models underlying LCA calculations. These three types are presented in more detail for the context of waste management in the next sub-sections.

3.2.1. PARAMETER UNCERTAINTIES WITHIN LCA OF WMS

Parameter uncertainties refer to the uncertainty in values due to e.g. measurement and sampling errors or imprecision, inherent variability, or insufficiency of data. Within waste management systems typical parameter uncertainties are chemical compositions of fractions (such as water content, heating value), transport distance, consumption of materials and energy, emission factors, sorting efficiencies, degradation rates of organic matter, the applied characterization factors in the LCIA, etc. (Clavreul et al. 2012).

In **Paper III** a contribution and a perturbation analysis were conducted to quantify parameter sensitivities and their application on the example of collecting and utilizing waste cooking oil from households (**case study II**).

3.2.2. SCENARIO UNCERTAINTIES WITHIN LCA OF WMS

Scenario uncertainties arise due to the necessary choices made to build scenarios. Typical scenario uncertainties within LCA of WMS are the definition of system boundaries, the selected databases (e.g. EcoInvent), the defined time-horizon of inventories, allocation decisions, chosen characterization and normalization methods, choice of specific technologies for waste treatment and the avoided material or energy production, etc. (Clavreul et al. 2012).

Scenario uncertainties were quantified in **Paper II and III** with the help of scenario analysis.

3.2.3. MODEL UNCERTAINTIES WITHIN LCA OF WMS

Model uncertainties are due to the mathematical models underlying LCA calculations. This can relate to the assumed linearity of emissions, the modelling of waste- and process-specific emissions, the defined models for substances fate and the related method of calculating characterization factors, assumption of linearity of response, applied model for biodegradation or plant uptakes and fertilizer substitution (Clavreul et al. 2012).

Model uncertainties were analyzed in detail in **Paper V** and the results are presented in **chapter 4**.

3.3. CASE STUDY II: WASTE COOKING OIL FROM HOUSEHOLDS

Paper II and III investigate the greenhouse gas (GHG) balances of three different utilization concepts for waste cooking oil (WCO) originating from a separate collection system for private households. The utilization pathways analyzed were (1) the esterification of WCO to biodiesel, (2) direct combustion in a cogeneration plant and (3) the fermentation and conversion to biogas within an agricultural biogas plant. Figure 4 shows the system boundaries on the example of the biodiesel scenario. The starting point of the assessment is the delivery of the WCO by the citizens at public collection points followed by collection, processing of the WCO, transport of the WCO to further utilization facilities (esterification plant, cogeneration plant, and agricultural biogas plant), the energetic utilization of the generated products and the associated substitution of fossil energy carriers. The functional unit was defined as the management of 1 t [fresh matter (FM)] waste cooking oil in terms of collection, treatment and utilization. Details on the chosen reference systems and applied modelling principles are presented in **paper II and III**.

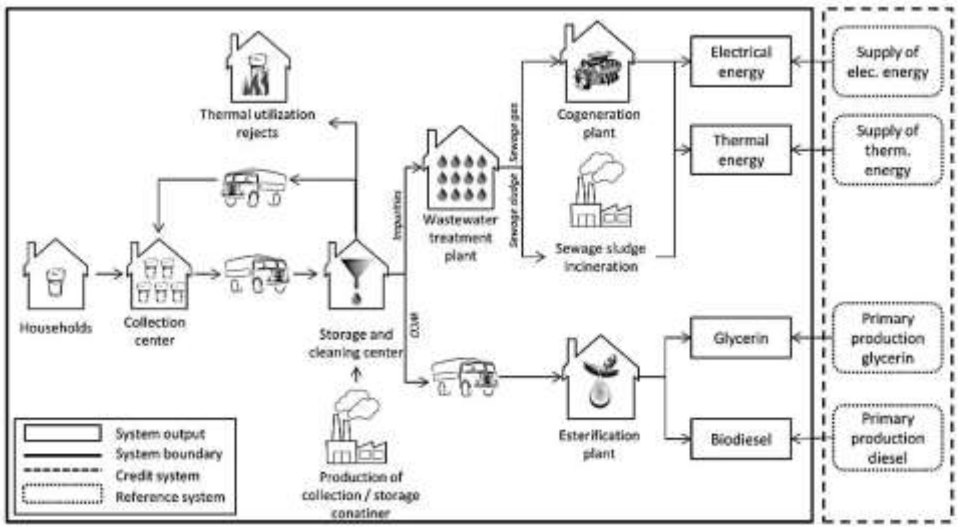

Figure 4: Process flowchart of the WCO collection system with following esterification (WCO: waste cooking oil) (from paper III)

3.3.1. ASSESSED PARAMETER UNCERTAINTIES IN CASE STUDY II

In **paper III** a special focus was put on parameter uncertainties which were assessed by contribution and perturbation analysis of twelve parameters. Focus was also put on the substitution potential of the resulting energy outputs concerning primary production of fossil energy carriers.

The results of the assessment showed that all three options contribute to a reduction of greenhouse gas emissions with esterification of the WCO (scenario 1) being the most favorable environmental option closely before scenario 2 (utilization of WCO in a cogeneration plant) and with twice as much savings as scenario 3 (WCO as a co-substrate in an agricultural biogas plant). The contribution analysis in Figure 5 shows that the main contribution to the overall environmental performances derived from the substitution processes. The environmental burden resulting from the collection and storage system, transport and processing and cleaning plant were in a similar range for all concepts. This is due to the fact that the same amount of collected WCO per year was assumed. Transport expenditures had a small impact on the overall result.

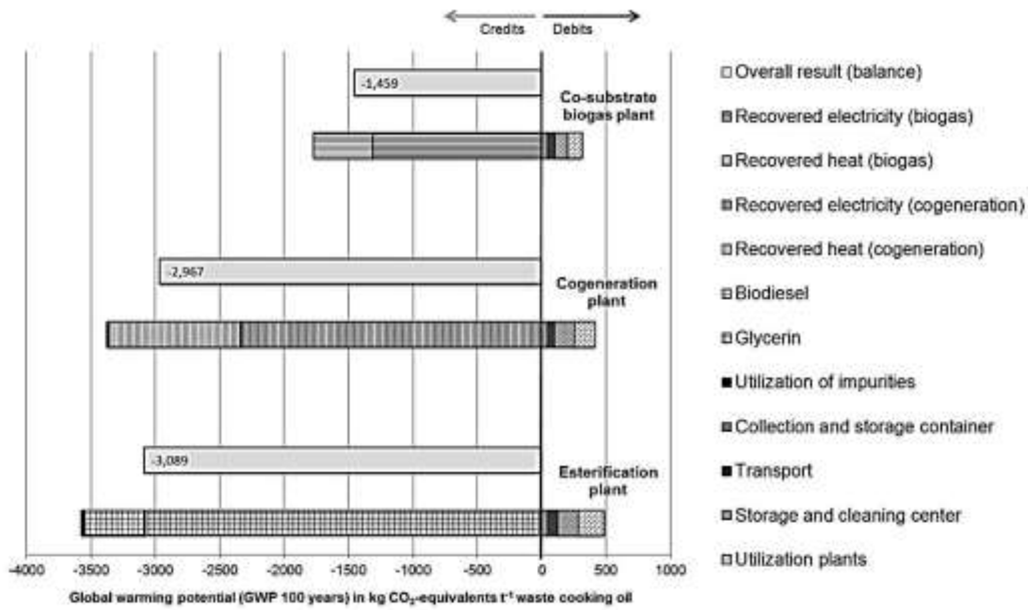

Figure 5: Contribution analysis of the comparison of greenhouse gas balances of the three utilization scenarios for WCO. Positive value range shows the environmental burden (debits). Negative value range shows the saved environmental impacts (credits) (from paper III).

In a next step twelve parameters were defined and varied individually by +10% and their sensitivity ratios were calculated. The subject of transportation and its contribution to the environmental performance of waste management systems is often discussed amongst practitioners. As a consequence, the parameter of transport distance was included in the perturbation analysis though, the contribution analysis showed that it was of minor influence on the overall result.

Figure 6 shows the SR as absolute values and presents the impact of the parameters selected on the GHG emissions of each scenario. Negative values express that, an increase of these parameters result in a growth of environmental burdens and thus, deteriorate the result. For scenario 1 the greenhouse gas emission profile of the reference system “fossil diesel” for the generated biodiesel showed the highest SR of 1. This means that, by increasing this parameter by 10 %, the benefits of the biodiesel scenario also increased by 10%. The second highest SR was found to be the electrical efficiency of the biogas powered cogeneration plant in scenario 3 with 0.9. And the variation of the electrical efficiency of the plant oil powered cogeneration plant in scenario 3 led to a SR of 0.79. The SR of the GHG emission profile of the substituted glycerin and of the thermal efficiency of the two cogeneration plants (scenario 2 and 3) were between 0.15 and 0.34. For all scenarios the SR of the parameter of the utilization plants lied between -0.05 and -0.07 and for the transport distances between -0.01 and -0.02.

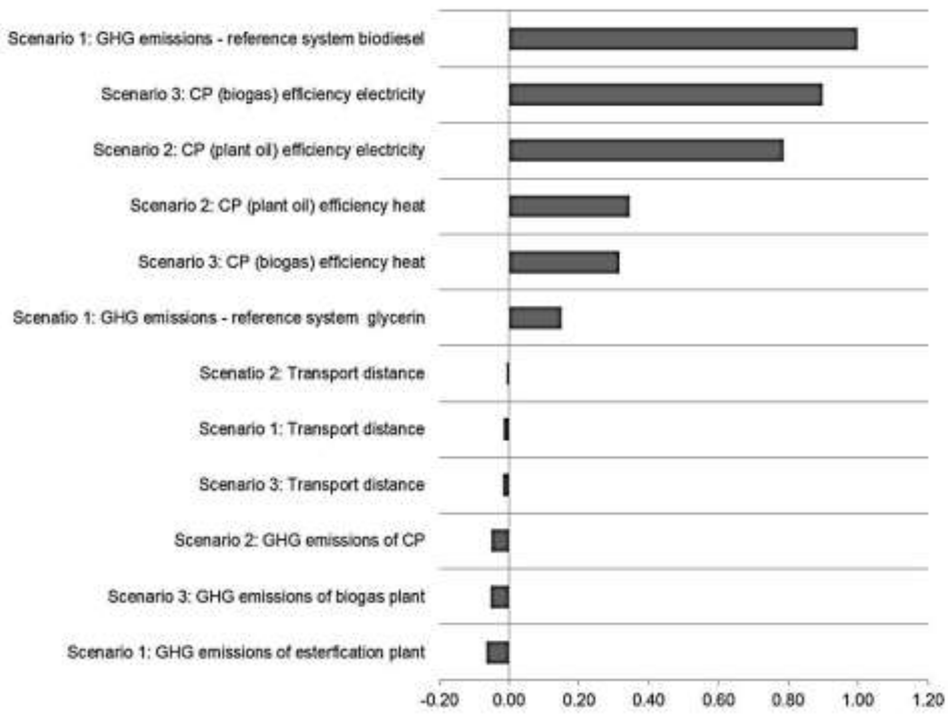

Figure 6: Parameter sensitivity ratios with respect to the GHG emissions of the three scenarios analyzed. Sensitivity ratios are presented in absolute values (GHG: greenhouse gas; CP: cogeneration plant). (from paper III)

All in all the result of the perturbation analysis showed that the results of case study II are solid as all parameter tested remained beneath a sensitivity ratio of 1 or 10%. Furthermore, the findings confirm the issue of the LCA community that the environmental impacts of the surrogated primary productions have a greater influence on the overall environmental performance of a waste management system than the treatment plants or transportation processes.

3.3.2. ASSESSED SCENARIO UNCERTAINTIES IN CASE STUDY II

Based on the findings of the perturbation analysis, that the final results are sensitive to the defined reference systems, a scenario analysis was conducted.

The GHG balances (Figure 5) of the three scenarios assessed showed that scenario 1 (biodiesel) outperformed the other two scenarios. However, the difference of 122 kg CO_{2eq} t^{-1} WCO between scenario 1 (biodiesel) and 2 (cogeneration plant) is not very significant. Therefore, the uncertainty of the ranking between scenario 1 and 2 was tested by varying the reference system chosen in scenario 2 (cogeneration plant). In Figure 7 the outcomes of the variation of the environmental performance from scenario 2 (cogeneration plant) are presented, when applying different energy carriers as a reference system for the electric energy produced (Figure 7a) and for the thermal energy output (Figure 7b).

Considerable deviations can be found in both figures. In the cases where hard coal and heavy fuel oil were used as reference systems for the electrical energy output, the overall result (balance) of the cogeneration scenario outperformed the biodiesel scenario. The variation of energy carriers for electrical energy led to a standard deviation of 1,037 CO_{2eq} t^{-1} WCO (44%) in the overall results. The variation of energy carriers for the thermal energy output resulted in a smaller standard deviation of 376 kg CO_{2eq} t^{-1} WCO (14%). In this scenario analysis the reference scenario (biodiesel) was always performing best.

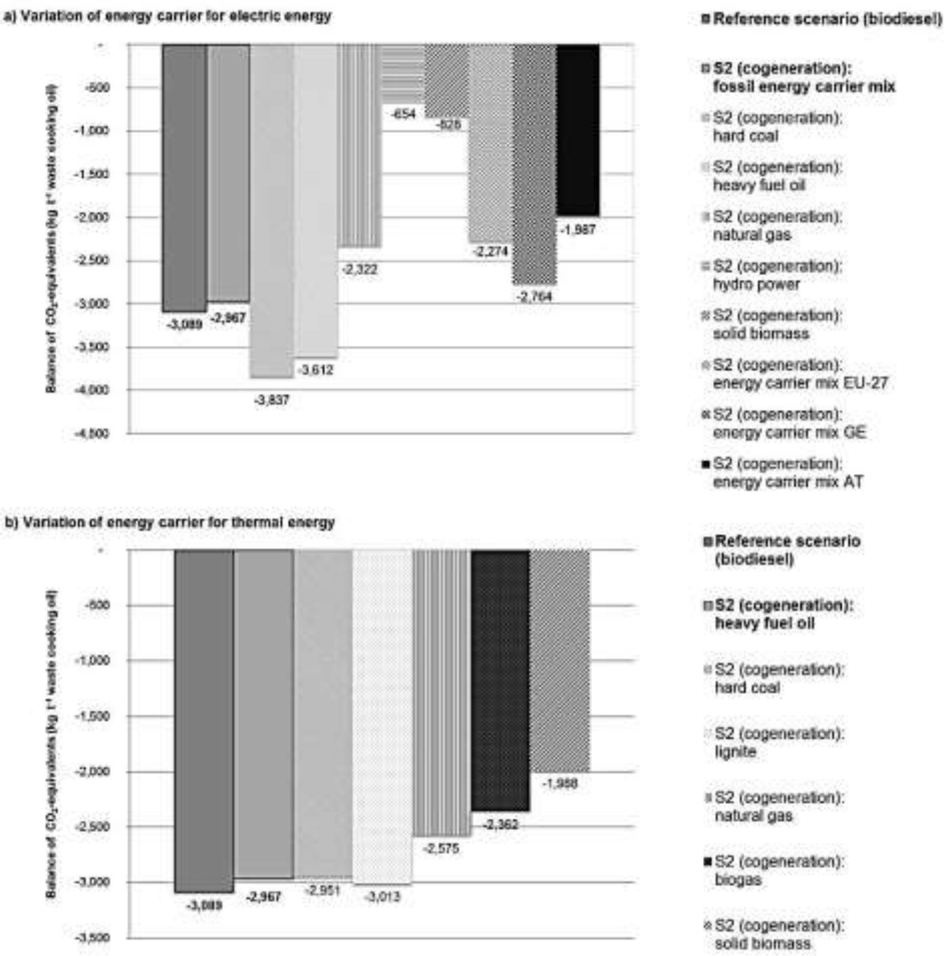

Figure 7: Scenario analysis of the reference systems for the produced energy (expressed in CO_2eq ton^{-1} WCO). a) Variation of the electrical energy carriers. b) Variation of the thermal energy carriers. (S2: scenario 2) (from Paper III).

4. SOFTWARE TOOLS AS UNCERTAINTY FACTORS WITHIN LCA

"Essentially, all models are wrong, but some are useful."
George E. P. Box

In **paper V** all three types of uncertainties explained in chapter 3 were elaborated on the example of **case study III**. However, a focus was put on model uncertainties in terms of the software tool applied.

Working with the support of software programs is state of the art in the field of LCA. However, modelling choices of LCA practitioners depend on the infrastructure of the software tool used, comprising the included datasets, calculation models, LCIA methods, etc. **Paper V** provides an overview of several studies in the field of comparing LCA software tools for waste management (Winkler and Bilitewski 2007; Turconi et al. 2011; Herrmann and Moltesen 2015; Kulczycka et al. 2015; Speck et al. 2015). It can be summarized, that the general consensus was that the choice of software tools can influence the final results of a LCA study to a not negligible degree, a fact that is made worse as a review on waste LCAs found that in more than 30% of the studies the software tool applied is not stated (Laurent et al. 2014b).

The objective of **paper V** was therefore to quantify the impact of the choice of software tool by comparing the structure of two specific LCA software tools from a user-perspective when modelling a waste management related LCA. Special focus was put on the differences in the default modelling setting of the software tools concerning the included databases, the LCIA methods for selection and the possibility to define specific process parameters and flow characteristics. Two LCA software tools were compared. The software GaBi from thinkstep AG (GaBi 2015) was chosen as a representor for one of the major players of the commercially available LCA software on the market. As a representative of the University based softwares on the market the software EASETECH (Environmental Assessment System for Environmental TECHnologies") from the DTU (Clavreul et al. 2014) was taken.

4.1. ASSESSMENT APPROACH

The approach presented in this section is the result of the work in **Paper V**. The stepwise approach for the comparison of EASETECH and GaBi is structured as follows:

- Definition of user groups and scenarios.
- Screening of databases for datasets on „biological waste treatment“ (case study III).
- Selection of suitable datasets.
- Comparison of LCIA methods included.
- Modelling of scenarios.
- Calculation of deviations between overall results of the scenarios.
- Detailed analysis on process and substance level.

Each assessment step will be explained in detail in the following sections.

4.1.1. DEFINITION OF USER GROUPS AND SCENARIOS

To assess the impact of different modelling choices on the LCA result, two user groups were defined: 1) default users and 2) expert users. Both user groups had the task to model **case study III** (Biological waste treatment options for biowaste) based on the same life cycle inventory.

Default user group: Aggregated EcoInvent datasets were applied for foreground processes and adjusted as far as possible to the requirements of case study III. Detailed information on the adjustments of the datasets can be found in Online resource 1a of paper V. EcoInvent data was also used for background processes.

Expert user group: Foreground processes were modelled based on expert (case specific) knowledge. For the background processes either Ecoinvent data or EASETECH/GaBi data (depending on the software tool) was used.

In a next step three cases were defined to be calculated in the two software tools (Figure 8): 1) Expert A using EcoInvent data for background processes in both softwares, 2) Expert B using EASETECH/GaBi data for background processes, 3) Default user applying EcoInvent data for foreground and background processes. Consequently, three comparisons were made and discussed from these three cases:

I. **Direct comparison between the two LCA software tools with optimal data** (Scenario 1 in Figure 8). „Expert A $_{\text{EASETECH}}$" is compared with „Expert A $_{\text{GaBi}}$". Both experts make use of the individual software infrastructure with data being recalculated to fit the structure modelled. The foreground processes are based on the same inventory from case study III. Comparability of the final results is assured as both user groups are using EcoInvent data for the background processes. The main differences between the two users can be reduced to the calculation models and the LCIA characterization methods within the individual software.

II. **Assessment of the effect of different modelling choices within the software tools.** In GaBi, "Expert A $_{\text{GaBi}}$" was compared with "Default user $_{\text{GaBi}}$" and in EASETECH, "Expert A $_{\text{EASETECH}}$" was compared with "Default user $_{\text{EASETECH}}$" (Scenario 2 + 3 in Figure 8).

The main difference in modelling is the usage of aggregated data sets versus own modelled processes for foreground processes.

III. **Assessment of the influence of upstream and downstream data**. "Expert A $_{\text{EASETECH}}$" was compared with "Expert B $_{\text{Ecoinvent}}$" and "Expert A $_{\text{GaBi}}$" was compared with "Expert B $_{\text{Ecoinvent}}$" (Scenario 4 + 5 in Figure 8). Both expert users modelled the foreground processes identically. The modelling difference can be found in the data used for the background processes (e.g.: Expert A $_{\text{EASETECH}}$ uses EASETECH data, Expert B $_{\text{Ecoinvent}}$ uses EcoInvent data).

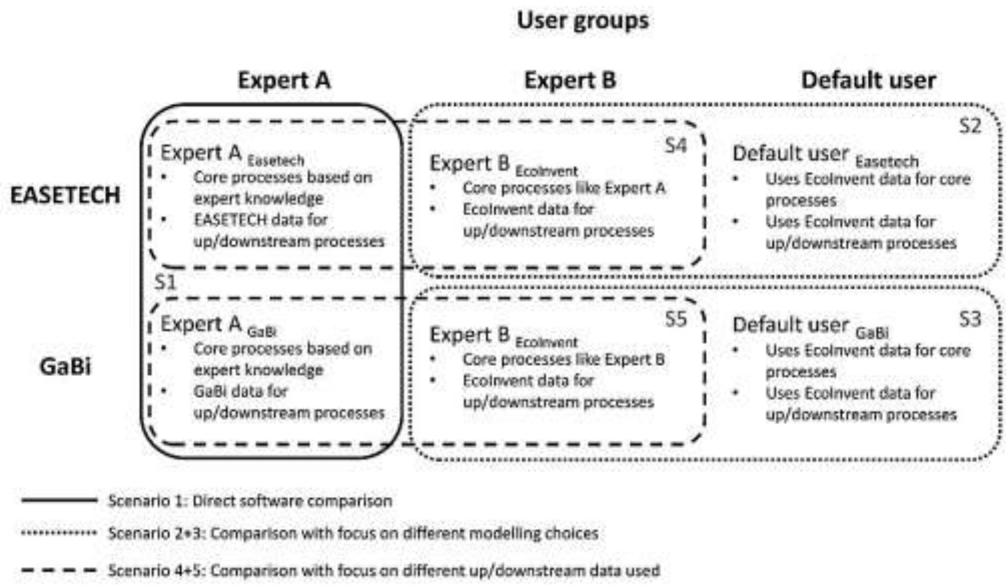

Figure 8: Overview of the defined user groups and scenarios for both LCA software tools assessed (from paper V).

The deviation between the different scenario results was calculated as percentage difference according to formula 3 (presented for Scenario 1, based on ISO 14044).

$$\%\Delta = \frac{Result\ Expert\ A\ EASETECH - Result\ Expert\ A\ GaBi}{Result\ Expert\ A\ GaBi}\ x\ 100 \quad (3)$$

4.1.2. CASE STUDY III: BIOWASTE FROM HOUSEHOLDS

The case study chosen for the comparison was an anaerobic digestion plant based on the assessment conducted in **paper IV**.

Paper IV investigates the greenhouse gas (GHG) and energy performance of three characteristic biowaste treatment concepts: (1) composting; (2) biological drying for the production of biomass fuel fractions; and (3) anaerobic digestion. The LCI of the concept of anaerobic digestion with maturation of the solid digestate served as a basis for the case study modelled in the course of the LCA software tool comparison in **paper V**.

Figure 9a presents the system boundaries of **case study III**. The starting point of the included processes is the delivery of the biowaste by the waste management organization followed by a mechanical pretreatment process, a continuous dry anaerobic digestion process for 2.5 – 3 weeks, utilization of the resulting biogas, post-composting of the solid digestate, a sieving process, the energetic and material utilization of the generated products and the associated substitution of the reference systems (fossil energy carriers, mineral fertilizers). The functional unit was defined as the treatment of 1 tonne [fresh matter (FM)] biowaste from households with separate collection in Austria.

Figure 9b illustrates the differences in the process chain that appeared when applying the aggregated EcoInvent dataset in Scenario 2 and 3 (default user groups compared to expert user groups). In the system of the aggregated dataset, the mechanical pretreatment does not remove metals from the input material, no RDF production takes place and the other process steps (except for the use on land) are aggregated into one dataset.

Details on the chosen reference systems and applied modelling principles are presented in **paper IV and V**.

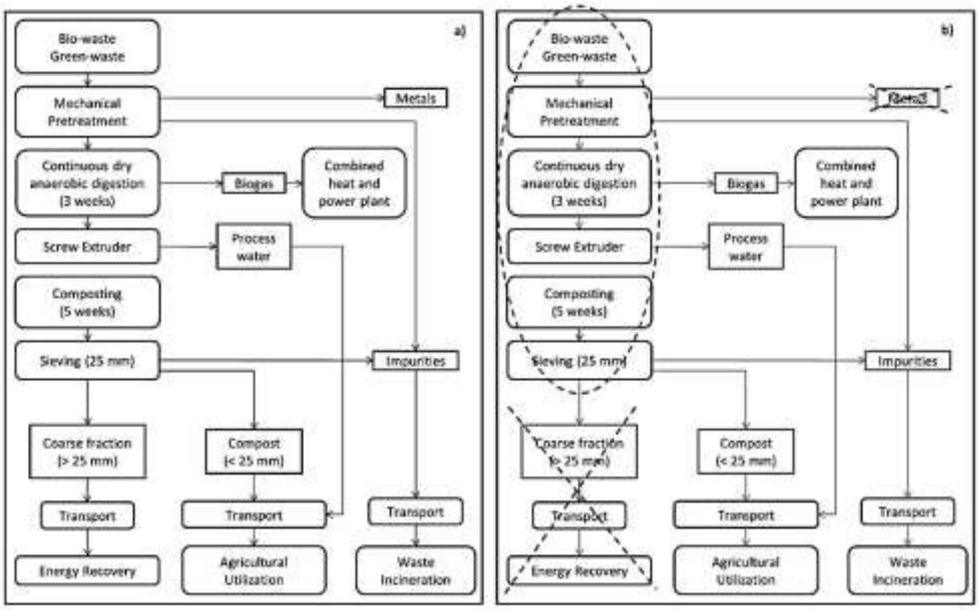

Figure 9: The flow chart of the biological treatment where a) shows the system both user groups had to model and b) shows the change in the treatment process chain when using aggregated EcoInvent datasets (default user group) (from paper V).

4.2. ASSESSMENT OF DATABASES AND SELECTED DATASETS

The software versions of EASETECH and GaBi used in the comparison were the newest versions available at the time the study was conducted. The GaBi version used was 7.0.0.19 (compilation), with Database (DB) version 6.110 including EcoInvent database version 2.2 (integrated version). The EASETECH version used was 2013, 1.5.07, with DB from June 2015. The EcoInvent datasets imported into EASETECH were from the same database version 2.2 as in GaBi.

The first step of the assessment of the databases was a screening using defined keywords (biological treatment, anaerobic digestion, fermentation, biogas plant, biowaste, composting, compost, biogas, fertilizer, nitrogen, phosphorous, potassium, NPK, phosphor, spreading) in the database of GaBi, EASETECH and EcoInvent. In a second step, the multitude of matches found in the first search was screened against a lower number of keywords (biowaste, biological treatment, fertilizer). In a final step quality indicators according to Weidema and Wesnaes (1997) were applied in order to assess the compliance of the found datasets. Table 2 lists the main technological characteristics of the foreground processes used by the expert user group. The foreground processes modelled based on own data by the expert users were: anaerobic digestion plant, cogeneration plant, use on land of compost and presswater. A detailed list of all datasets used in both software tools can be found in Online Resource 1a and 1b of **paper V**.

Table 2: Main technical parameters of the foreground technologies used by the expert users (from paper V).

Technology	Description	Source
Anaerobic digestion	Thermophilic reactor with post-composting CH_4 yield: 65% degradation of volatile solids (VS) Biogas produced: 108 Nm^3/tonne wet weight Pretreatment of biowaste. Separated contaminants are inventoried as household waste for incineration. Metals are recycled.	(Ortner et al. 2013)
Cogeneration	Electricity recovery: 35% Heat recovery: 45%	Gemis (2010)
Land application compost and presswater	Digestate loading and spreading by hydraulic loader and spreader	(Ortner et al. 2013)
Incineration of contaminants	Municipal solid waste incineration plant Efficiency electricity recovery: 10% Efficiency heat recovery: 30%	(Ortner et al. 2013)
Incineration of biomass	Biomass incineration plant Efficiency electricity recovery: 27% Efficiency heat recovery: 35%	(Ortner et al. 2013)
Metal recycling	90% recycling efficiency	(Ortner et al. 2013)
Transportation distances	Compost and presswater: 15 km Impurities, RDF and metals: 120 km	(Ortner et al. 2013)

4.3. ASSESSMENT OF LIFE CYCLE IMPACT ASSESSMENT METHODS

The characterization models for the Life Cycle Impact Assessment have been listed as one of the major potential differences between LCA software tools in previous studies (Herrmann and Moltesen 2015; Speck et al. 2015). Both software tools offer ILCD (International Reference Life Cycle Data System, European Comission 2010) recommended list of impact categories and characterization models recommended as suggested default assessment methods. A detailed analysis of the characterization models listed in the ILCD recommendations and the number of compartments and sub-compartments within both software tools was conducted in order to verify compliance. Therefore, the implemented characterization models, the version of the model, the version of characterization factors applied were investigated for every impact category.

Table 3 presents the results of the comparison of 13 impact categories. In six impact categories (global warming, terrestrial eutrophication, acidification, freshwater eutrophication, ionizing radiation and photochemical ozone formation), the LCIA characterization models and the version of the models were consistent. For five impact categories (ecotoxicity, marine eutrophication, human toxicity cancerous, human toxicity non-cancerous, resource depletion) the two software tools offered the same LCIA characterization models, however different versions thereof. For ozone depletion, the LCIA characterization models EDIP and ReCiPe are marked as consistent in Table 3, as both are based on the recommended model of the World Meteorological Organization (WMO). The assessment showed that concerning particulate matter two different ILCD recommended characterization models were integrated in the softwares (EASETECH: Humbold (2009); GaBi: Riskpoll).

Table 3: Results of the comparison of the implemented characterization models (CM) and characterization factors (CF). (✓: consistent; —: dissenting) (from paper V).

		Criteria of compliance				
Impact category	Characterisation models (CM) applied	CM	Version of CM	Version of CF	Nr. of comp.	Long-term emissions
Global warming	Easetech: IPCC 2007 GWP 100a GaBi: IPCC 2007 GWP 100a	✓	✓	✓	—	included in GaBi
Eutrophication, terrestrial	Easetech: Accumulated exceedance GaBi: Accumulated exceedance	✓	✓	✓	—	included in GaBi
Acidification	Easetech: Accumulated exceedance GaBi: Accumulated exceedance	✓	✓	✓	—	included in GaBi
Photochemical Ozone Formation Potential (POFP)	Easetech: ReCiPe Midpoint (H) w/o LT, GaBi: ReCiPe, LOTOS-EUROS model	✓	✓	✓	—	included in GaBi
Eutrophication, freshwater	Easetech: ReCiPe Midpoint (H), w/o LT GaBi: EUTREND model, ReCiPe	✓	✓	✓	—	included in GaBi
Ionising radiation, human health	Easetech: ReCiPe Midpoint (H) w/o LT, GaBi: ReCiPe	✓	✓	✓	—	included in GaBi
Resources, depletion	Easetech: CML 2013 GaBi: CML2002	✓	—	—	—	
Eutrophication, marine	Easetech: ReCiPe Midpoint (H) w/o LT, GaBi: ReCiPe, EUTREND model	✓	—	—	—	included in GaBi
Ecotoxicity	Easetech: USEtox, W/O Longterm, DTU updated GaBi: USEtox	✓	—	—	—	included in GaBi
Human toxicity cancer effects	Easetech: USEtox, W/O Longterm, DTU updated GaBi: USEtox	✓	—	—	—	included in GaBi
Human toxicity non.canc. Effects	Easetech: USEtox, W/O Longterm, DTU updated GaBi: USEtox	✓	—	—	—	included in GaBi
Ozone depletion	Easetech: EDIP w/o LT, ODP 100a w/o LT GaBi: WMO model, ReCiPe	✓	—	—	—	included in GaBi
Particulate matter	Easetech: updated from Humbert 2009 (after ILCD 2011) GaBi: RiskPoll	—	—	—	—	included in GaBi

An interesting finding was, that GaBi included short and long-term emissions (listed separately) as default setting for all impact categories (except for resource depletion), whereas EASETECH only covered short-term emissions in the default settings. For the direct software tool comparison in Scenario 1 long-term emissions were excluded from GaBi results by subtracting them manually in an external calculation model. No function for excluding them automatically within GaBi was found.

During the LCIA elementary exchanges get assigned to a number of compartments (air, water, soil, etc.) and associated sub-compartments. EASETECH maps emissions to the compartments air, water, soil and resources, which are further divided into sub-compartments. This means that e.g. for the compartment air, it is further specified in which sub-compartment an emission is released which can be: indoor, low population density, long-term lower stratosphere and upper troposphere, non-urban air or from high stacks, urban air close to ground and unspecified. **Paper V** showed that GaBi uses different numbers and terms for compartments and sub-compartments and different characterization factors than EASETECH. The effects of this different inventorying of elementary exchanges on the final results are analyzed in detail in section 4.4.2. Analysis of deviations on substance level.

4.4. ASSESSMENT OF THE CHOICE OF DIFFERENT SOFTWARE TOOLS

The objective of Scenario 1 was to assess potential differences and deviations in the final results of the two software tools analyzed. Therefore, user “Expert A $_{EASETECH}$” was compared with “Expert A $_{GaBi}$” (Scenario 1). Both expert users had the task to model the foreground processes of case study III based on the same inventory but making use of the individual software infrastructure. For the background processes EcoInvent data was used in both software tools.

In Table 4 the overall results are presented in person equivalents/1000 (mPE). The normalization was conducted separately in a calculation program, as the two software tools did not offer the same normalization factors. The normalization references from the Prosuite project were applied, which were developed specifically for the recommended ILCD method (Online Resource 3, paper V). It needs to be emphasized that the deviations are strongly influenced by the absolute numbers of the results. The closer the absolute result is to zero, the higher the deviation will be. This is a consequence of the mathematical effect of dividing a low number with another low number. Therefore, only deviations within one impact category should be compared. As in absolute numbers the results between different IP can differ by a factor 10 or more, this is also reflected by the deviations.**Fehler! Verweisquelle konnte nicht gefunden werden.**The analysis revealed that in six impact categories (terrestrial eutrophication, acidification, ionising radiation, freshwater eutrophication, climate change, photochemical ozone formation potential) the deviations were between 0.01 % and 1.07 %, which is considered acceptable (the ISO 14044 considers deviations > 10% as significant). This result was expected as both software tools used the same version of LCIA characterization models presented in Table 3. It was found that the main reason for the small deviations in these impact categories derived from the different calculation models, that caused differences in the mass and substance balances and the subsequent life cycle inventory. This will be explained in more detail with the help of a contribution analysis in the next section.

For three impact categories (ozone depletion, ecotoxicity and marine eutrophication) the deviation was between 15% and 58%. And four impact categories (human toxicity cancer effects, resource depletion, human toxicity non.can. effects and particulate matter) showed a deviation higher than 60%. The main reason for the deviation in these impact categories was related to the different version of the implemented LCIA characterization models (Online Resource 2, paper V). The impact category "resource depletion" holds a special position. Here, the high deviation resulted from comparing the aggregated impact from fossil and elemental resources in GaBi with the independent impact of elemental resources in EASETECH (a separate impact category for fossil resources is available). As these two different CML (Institute of Environmental Sciences of Leiden University) impact categories represent the default settings in the two software tools, this was an important difference, which needed to be pointed out for LCA practitioners.

Table 4: Deviation of overall results in different impact categories in Scenario 1 (mPE: mili person equivalent; CM: characterization model; CF: characterization factor). Sorted after increasing deviations in results. Grey boxes mark the main reasons for deviation for each impact category; potentially: if long-terms emissions were not excluded in GaBi they would contribute substantially to deviations (from Paper V).

				Reasons for deviations					
				Calculation models	Life Cycle Impact Assessment				
	EASETECH (mPE)	GaBi (mPE)	%Δ	Substance/ mass balance	CM	Compartm., sub-comp.	CF	Long-term emissions
Eutrophication, terrestrial	74	74	0					potentially
Acidification	28	28	0					potentially
Ionising radiation	-1	-1	0					potentially
Eutrophication, freshwater	-2	-2	0					potentially
Climate change	-12	-12	1					potentially
POFP	-3	-3	1					potentially
Ozone depletion	-1	-1	15					potentially
Ecotoxicity	1476	2502	41					potentially
Eutrophication, marine	3	7	58					potentially
Human toxicity cancerous	76	683	89					potentially
Resources depletion	-11	-5	113					
Human toxicity	1759	494	256					potentially
Particulate matter	-30	-5	473					potentially

It is important to mention that the results in the individual impact categories of both software tools showed the same tendency and did not contradict each other. Still, in the impact categories with a deviation > 15% the LCA practitioner must pay outmost attention. In the optional step of weighting during the interpretation phase the EASETECH results could lead to a different ranking between individual impact categories than the results from GaBi.

4.4.1. Analysis of deviations on process level

The deviations and reasons for deviations between the two software tools were assessed in more detail with the help of contribution analysis. The aim was to analyze how the deviations are influenced by differences in the mass balance due to different calculation models within the two software tools. A contribution analysis on a process level was conducted for the impact category "global warming potential (GWP)" of "Expert $_{\text{GaBi}}$" with "Expert $_{\text{EASETECH}}$". Therefore, the LCIA results were disaggregated to visualize the individual contributions of the individual processes.

It was shown that the deviation on a process level can be significantly higher or lower than the deviation between the overall results (stated as "total" in the Table 5). In the impact category GWP the overall deviation amounts 0.71% whereas the deviation of the process "use on land of presswater" is 13%. This was explained by differences in the calculation models included in the different software tools. In EASETECH the properties of the input material are defined at the beginning by the user and the substance and mass balance can be kept throughout the system with the help of e.g. transfer coefficients. This modelling approach is called "input specific modelling". "Expert $_{\text{EASETECH}}$" applied the "use-on-land" calculation model which allowed detailed definition of the distribution of biogenic carbon, nitrogen and phosphorous. In the software GaBi, mass and substance balances can also be modelled with the help of process and plan parameters. "Expert $_{\text{GaBi}}$" modelled process emissions with emission factors (e.g. 100 g NH_3 t^{-1} fresh matter (FM) input material). Further explanation of the differences concerning the implemented calculation models and their effects on the final result can be found in Online Resource 4 of **paper V**.

Table 5: Contribution analysis of 13 impact categories of scenario 1 „Expert $_{GaBi}$ "compared to „Expert $_{EASETECH}$" (AD plant: anaerobic digestion plant; USO: use on land; CM: contaminants; BM: biomass, mPE: mili person equivalent; %Δ: deviation stated in % with GaBi as reference result). Classification of deviations: white: minor; grey: +/- 10 % (absolute) difference from overall deviation (from paper V).

	Eutrophication terr.			Acidification			Ionising radiation			Eutrophication freshw.		
Processes	Easetech (mPE)	Gabi (mPE)	%Δ	Easetech (mPE)	Gabi (mPE)	%Δ	Easetech (mPE)	Gabi (mPE)	%Δ	Easetech (mPE)	Gabi (mPE)	%Δ
AD plant	24	24	0	21	21	0	0	0	0	1	1	2
Cogeneration	-7	-7	0	-19	-19	0	-1	-1	0	-1	-1	2
USO compost	1	1	1	0	0	0	0	0	0	-1	-1	1
USO presswater	57	57	0	30	30	0	0	0	2	-1	-1	1
Incineration CM	0	0	0	-1	-1	0	0	0	0	0	0	13
Incineration BM	-1	-1	2	-3	-3	2	0	0	2	0	0	22
Recycling metals	-1	-1	0	-1	-1	0	-1	-1	0	-1	-1	0
Transport	1	1	1	0	0	1	0	0	1	0	0	1
Total	74	74	0	28	28	0	-1	-1	0	-2	-2	0

	Global warming			POFP			Ozone depletion			Ecotoxicity		
Processes	Easetech (mPE)	Gabi (mPE)	%Δ	Easetech (mPE)	Gabi (mPE)	%Δ	Easetech (mPE)	Gabi (mPE)	%Δ	Easetech (mPE)	Gabi (mPE)	%Δ
AD plant	28	28	0	5	5	0	1	1	13	128	23	454
Cogeneration	-34	-34	0	-7	-7	0	-1	-1	15	-223	-42	436
USO compost	-3	-3	4	0	0	1	0	0	13	750	987	24
USO presswater	-1	-1	13	0	0	2	0	0	17	847	1142	26
Incineration CM	0	0	0	0	0	0	0	0	13	-7	95	107
Incineration BM	-1	-1	1	-1	-1	2	0	0	16	-12	311	104
Recycling metals	-1	-1	1	0	0	0	0	0	103	-12	-19	40
Transport	0	0	1	0	0	1	0	0	6	4	5	23
Total	-12	-12	1	-3	-3	1	-1	-1	15	1476	2502	41

	Eutrophication mar.			Hum. toxcitiy canc.			Resource depletion			Hum. toxcitiy non canc.			Particulate matter		
Processes	Easetech (mPE)	Gabi (mPE)	%Δ	Easetech (mPE)	Gabi (mPE)	%Δ	Easetech (mPE)	Gabi (mPE)	%Δ	Easetech (mPE)	Gabi (mPE)	%Δ	Easetech (mPE)	Gabi (mPE)	%Δ
AD plant	8	2	413	5	6	21	14	9	48	3	3	2	41	21	96
Cogeneration	-8	0	11052	-9	-12	19	-26	-18	46	-5	-5	2	-63	-28	123
USO compost	-1	0	912	25	303	92	-3	-1	309	827	230	260	-2	-1	203
USO presswater	4	5	16	33	352	91	-1	-3	45	933	266	251	10	11	13
Incineration CM	0	0	765	8	11	23	-1	-1	66	0	0	89	-4	-2	120
Incineration BM	-1	0	1033	27	36	25	-4	-2	60	0	1	81	-11	-5	110
Recycling metals	-1	0	7364	-14	-14	3	6	6	1	1	-1	173	-2	-2	1
Transport	1	0	24262	1	1	14	4	3	12	0	1	100	0	0	27
Total	3	7	58	76	683	89	-11	-5	113	1759	494	256	-30	-5	473

4.4.2. ANALYSIS OF DEVIATIONS ON SUBSTANCE LEVEL

In a next step a contribution analysis on a substance level was conducted for the LCIA results of the impact category “particulate matter” of “Expert $_{GaBi}$” and “Expert $_{EASETECH}$”. The objective was to analyze whether the overall deviation was caused by the substance balance generated by the calculation models or by the implemented LCIA method. The substance balances in Table 6 show no significant deviations between the two software tools. However, the comparison of the LCIA results showed that there exist significant deviations for certain substances. Different reasons were found in this context. The first is that different LCIA methods for this IC were implemented in the two software tools (GaBi: RiskPoll; EASETECH: updated version from Humbert (2009)). Consequently, different characterization factors were found. For example GaBi did not assign impacts to “nitrogen oxides”, but implemented characterization factors (CF) for “nitrogen dioxide” (0.0072 kg PM2,5-equ.) and “nitrogen monoxide” (0.0111 kg PM2,5-equ.). EASETECH uses ecoSpold substance naming, and thus all emissions of nitrogen oxide emissions are accounted for as “nitrogen dioxide” with a CF for “nitrogen oxides” (“air - unspecified” 0.0072 kg PM2,5-equ.; “air - urban air close to ground”: 0.0078 kg PM2,5-equ.).

In GaBi, only an allocation of the elementary exchange “carbon monoxide emissions to air” to “fossil” but not to “non-fossil” was possible for GaBi-specific or self-defined flows. "Carbon monoxide, non-fossil" emissions were only available for EcoInvent flows. EASETECH allowed the grouping between “fossil” and “non-fossil”. However, as the same characterization factors are used for the “fossil” and the “non-fossil” emission group the overall result is not affected, when assigned differently. The emission group of “Particulates” was inventoried to a higher number of sub-compartments in EASETECH than in GaBi and also different characterization factors were assigned. Further information on the differences concerning the implemented LCIA models can be found in Online Resource 2 of **paper V**.

Table 6: Contribution analysis on substance level for the impact category „particulate matter" of scenario 1 „Expert $_{GaBi}$ "compared to „Expert $_{EASETECH}$" (%Δ: deviation stated in % with GaBi as reference result). Classification of deviations: white: minor; grey: +/- 10 % (absolute) difference from overall deviation (from Paper V).

	Substance balance (kg)			Characterised results (kg PM2,5-Equiv.)		
	Easetech	GaBi	%Δ	Easetech	GaBi	%Δ
Nitrogen oxides	-9.2E-02	-9.2E-02	1	-7.7E-04	-	
Ammonia	6.6E-01	6.6E-01	0	4.4E-02	4.4E-02	0
Carbon monoxide, fossil	-2.1E-02	-2.1E-02	0	-7.8E-06	-7.3E-06	7
Carbon monoxide, non-fossil	1.3E-02	1.4E-02	1	6.9E-06	4.8E-06	43
Sulfur dioxide	-4.3E-01	-4.3E-01	1	-2.8E-02	-2.6E-02	5
Sulphur trioxide	-1.6E-09	-1.6E-09	1	-8.6E-11	-7.9E-11	8
Particulates, > 2.5 µm, and < 10µm	-1.3E-02	-1.3E-02	0	-2.7E-03	-	
Particulates, < 2.5 µm	-3.2E-02	-3.2E-02	1	-9.6E-02	-3.2E-02	198
Particulates, > 10 µm	-2.5E-02	-2.3E-02	10			
Total	6.2E-02	6.2E-02	1	-8.3E-02	-1.5E-02	473

4.5. ASSESSMENT OF DIFFERENT MODELLING CHOICES WITHIN ONE SOFTWARE TOOL

4.5.1. MODELLING CHOICE I: AGGREGATED VERSUS SELF-MODELLED PROCESSES FOR FOREGROUND PROCESSES

The objectives of Scenario 2 and 3 were to assess the effects of different modelling choices concerning foreground processes within the individual software tools. Therefore, "Expert B $_{EcoInvent}$" was compared with the "Default user $_{EASETECH}$" in Scenario 2 and "Expert B $_{EcoInvent}$" was compared with the "Default user $_{GaBi}$" in Scenario 3. The differences in the modelling choices were that the default users applied aggregated EcoInvent datasets for foreground processes and adjusted them as far as possible to the requirements of case study III. These adjustments in the datasets are presented in more detail in Online Resource 1a and 1b of paper V. The changes in the process chain when using an aggregated dataset are shown in Figure 9 in "Chapter 4.1.2. Case study III". The expert users modeled foreground processes themselves based on expert knowledge. For background processes both user groups used the same EcoInvent datasets.

A consequence of the default user´s modelling with aggregated dataset was a change in the characteristics in the input material. It was not possible to adjust the physical characteristics or the content shares of heavy metals and nutrients in the input material of the EcoInvent dataset to the inventory of case study III. Still, the biogas yield of the aggregated EcoInvent dataset of the AD plant was adjusted by the default users in both software tools.

Table 7 presents the results of Scenario 2 and 3 for 13 impact categories normalized to mili person equivalents (mPE). Nine impact categories (IC) in EASETECH and six IC in GaBi showed a deviation higher than 80%. In GaBi the deviation of the remaining ICs was between 10 and 80 %. In EASETECH, only the IC "ionizing radiation" stayed below the 10% deviation limit.

Partly high differences between the results of the two default users in GaBi and EASETECH were found. This is traced back to the different inventorying of the elementary exchanges into compartments and sub-compartments. In addition, the modelling principles of GaBi (e.g. modelling with partly aggregated processes, definition of parameter for emission factors) allowed the "Default user $_{\text{GaBi}}$" more adjustment of the aggregated EcoInvent datasets to the case study than EASETECH where emissions are partly directly linked to the characteristics of the input material.

Table 7: Results of the user comparison in Scenario 2 and 3: percentage deviation in the overall results due to different modelling choices of using aggregated data. Within EASETECH "Expert B $_{\text{EcoInvent}}$" was compared with the "Default user" (Scenario 2). In GaBi "Expert B $_{\text{EcoInvent}}$" was compared with "Default user" (Scenario 3). Classification of deviations: white: minor (less than 10%); grey: substantial (between 10 and 80%); black: major (more than 80%) (from Paper V).

	EASETECH			GaBi		
	Scenario 2			Scenario 3		
	Default User (mPE)	Expert B$_{\text{EcoInvent}}$ (mPE)	%Δ	Default User (mPE)	Expert B$_{\text{EcoInvent}}$ (mPE)	%Δ
Eutrophication, terrestrial	32	74	56	36	74	52
Acidification	1	28	97	4	28	85
Ionising radiation	-1	-1	10	0	-1	63
Eutrophication, freshwater	2937	-2	128266	2946	-2	129131
Climate change	-4	-12	65	-3	-12	73
POFP	-7	-3	112	-5	-3	42
Ozone depletion	-1	-1	29	-1	-1	25
Ecotoxicity	4668	1476	216	5065	2502	102
Resources depletion	-30	-11	163	-19	-5	260
Eutrophication, marine	-9	3	431	4	7	45
Human toxicity non-cancerous	117	1759	93	1105	494	124
Particulate matter	-62	-30	105	-19	-5	263
Human toxicity cancerous	4854	76	6315	764	683	12

4.5.2. MODELLING CHOICE II: DIFFERENT DATABASES FOR BACKGROUND PROCESSES

The objectives of scenario 4 and 5 were to assess the effects of different dataset choices within the individual software tools. Therefore, “Expert A $_{\text{EASETECH}}$” was compared with the “Expert B $_{\text{EcoInvent}}$” in Scenario 4 and “Expert A $_{\text{GaBi}}$” was compared with the “Expert B $_{\text{EcoInvent}}$” in Scenario 5. The differences in the dataset choices were that “Expert A $_{\text{EASETECH}}$” applied datasets from the EASETECH database for background processes, “Expert A $_{\text{GaBi}}$” used GaBi datasets and “Expert B $_{\text{EcoInvent}}$” used EcoInvent datasets accordingly. All expert users modelled the foreground processes identically.

In Table 8 the results of Scenario 4 and 5 are presented for 13 impact categories normalized to mili person equivalents (mPE). In EASETECH, seven impact categories (IC) showed a deviation of more than 80%, five IC deviated between 10 and 80% and only one IC (terrestrial eutrophication) was below 10%. In GaBi four IC reached a deviation of higher 80%, five IC deviated between of 10 and 80% and four deviated below 10%. Eight IC showed a lower deviation between the two expert users in GaBi than within EASETECH. This difference is due to the fact that the temporal, geographical and technical quality indicators of the background data from GaBi and EcoInvent are closer than those from EASETECH and EcoInvent. Further information can be found in the Online Resource 1a, 1b of paper V.

Table 8: Results of the user comparison in scenario 4 and 5: effect of different data used for the background processes. In EASETECH “Expert A $_{\text{EASETECH}}$” using EASETECH data was compared with “Expert B $_{\text{EcoInvent}}$” using EcoInvent data (Scenario 4). In GaBi “Expert A $_{\text{GaBi}}$” using GaBi data was compared with “Expert B $_{\text{EcoInvent}}$” using EcoInvent data (Scenario 5). Classification of deviations: white: minor (less than 10%); grey: substantial (between 10 and 80%); black: major (more than 80%) (from paper V).

	EASETECH			GaBi		
	Scenario 4			Scenario 5		
	Expert A$_{\text{EASETECH}}$ (mPE)	Expert B$_{\text{EcoInvent}}$ (mPE)	%Δ	Expert A$_{\text{GaBi}}$ (mPE)	Expert B$_{\text{EcoInvent}}$ (mPE)	%Δ
Eutrophication, terrestrial	78	74	5	78	74	5
Acidification	38	28	38	38	28	39
Ionising radiation	-2	-1	40	-1	-1	28
Eutrophication, freshwater	-35	-2	1431	-1	-2	44
Climate change	-7	-12	42	-9	-12	22
POFP	0	-3	110	0	-3	100
Ozone depletion	0	-1	89	0	-1	98
Ecotoxicity	1643	1476	11	2105	2502	16
Resources depletion	-8	-11	30	-34	-5	530
Eutrophication, marine	8	3	196	7	7	6
Human toxicity non-cancerous	6877	1759	291	494	494	0
Particulate matter	14	-30	146	12	-5	335
Human toxicity cancerous	870	76	1050	643	683	6

4.5.3. SENSITIVE PROCESSES WITHIN CASE STUDY III

A process variation was performed for Scenario 2 and 3. The objective of the process variation was to show which process has the highest influence on the overall result and therefore represents an important modelling decision for the LCA practitioner when modelling case study III. For this reason processes of the system of the default user were varied with the equivalent processes from Expert B $_{\text{EcoInvent}}$ in GaBi. The results of the same process variation for the EASETECH user groups can be found in Online Resource 5 of paper V.

In a first step, the process of anaerobic digestion plant was varied by replacing the AD plant (aggregated dataset) of the default user with the AD plant (self modelled) from Expert B. As a consequence, the rest of the process chain changed, because of different set up of the EcoInvent dataset (explained above). Changes in the overall result and on process levels were calculated and expressed in percentage deviation, with the system of the default user serving as the baseline scenario. In order to relate the deviations with the changed process, all the processes had to be changed one after the othjer. The process variation was performed for two impact categories, which were selected according to their general relevance for waste management systems (GWP) and absolute mPE emissions within the case study (eutrophication, terrestrial).

Table 9 shows that for GWP the emissions of the “use on land of compost and presswater” and “anaerobic digestion plant” processes had a significant effect on the overall results. Whereas for the IC “terrestrial eutrophication” the overall result was highly sensitive to the variation of the anaerobic digestion plant.

Table 9: Process variation in Scenario 3 (Expert $B_{EcoInvent}$ compared with Default $user_{GaBi}$). Results expressed in mili person equivalents (mPE) for a) global warming potential (GWP) and b) terrestrial eutrophication potential (EP). Scenario 3: base line scenario; variation "AD plant": effects on process level and overall results due to variation of the process "anaerobic digestion plant"; Variation "cogeneration": effects on process level and overall results due to variation of the process "cogeneration plant"; Variation "USO": effects on process level and overall results due to variation of the use on land process. (AD plant: anaerobic digestion plant; USO: use on land; CM: contaminants; BM: biomass, mPE: mili person equivalent; %Δ: deviation stated in %) (from paper V)

a)

	Global warming potential (mPE)											
	Scenario 3			Variation "AD plant"			Variation "Cogeneration"			Variation "USO"		
Processes	Default User $_{GaBi}$	Expert B $_{EcoInvent}$	%Δ	Default User $_{GaBi}$	Expert B $_{EcoInvent}$	%Δ	Default User $_{GaBi}$	Expert B $_{EcoInvent}$	%Δ	Default User $_{GaBi}$	Expert B $_{EcoInvent}$	%Δ
AD plant	33	28	17	33	28	17	33	33	0	33	33	0
Cogeneration	-33	-34	1	-33	-33	0	-33	-34	1	-33	-33	0
USO compost	-3	-4	25	-3	-2	58	-3	-3	0	-3	-20	87
USO presswater	0	-1	100	0	-1	100						
Incineration CM	0	0	100	0	0	100						
Incineration BM	0	-1	100	0	-1	100						
Recycling metals	0	-1	100	0	-1	100						
Transport	1	0	17	1	0	110	1	1	0	1	0	112
Total	-3	-12	73	-3	-10	67	-3	-4	12	-3	-21	85

b)

	Eutrophication terrestial (mPE)											
	Scenario 3			Variation "AD plant"			Variation "Cogeneration"			Variation "USO"		
Processes	Default User $_{GaBi}$	Expert B $_{EcoInvent}$	%Δ	Default User $_{GaBi}$	Expert B $_{EcoInvent}$	%Δ	Default User $_{GaBi}$	Expert B $_{EcoInvent}$	%Δ	Default User $_{GaBi}$	Expert B $_{EcoInvent}$	%Δ
AD plant	45	24	87	45	24	87	45	45	0	45	45	0
Cogeneration	-11	-7	52	-11	-11	0	-11	-7	52	-11	-11	0
USO compost	0	58	100	0	1	100	0	0	0	0	4	100
USO presswater	0	57	100	0	1	100						
Incineration CM	0	0	100	0	0	100						
Incineration BM	0	-1	100	0	-1	100						
Recycling metals	0	-1	100	0	-1	100						
Transport	2	1	132	2	0	317	2	2	0	2	0	315
Total	36	74	52	36	14	160	36	39	9	36	38	6

5. CONCLUSIONS, DISCUSSION AND OUTLOOK

"There are known knowns. There are known unknowns.
But there are also unknown unknowns;
things we don't know we don't know.
It is the latter category that tends to be the difficult one."
Donald Rumsfeld

5.1. CONCLUSION

This thesis and the included papers show the importance of assessing and considering the effect of modelling choices on the environmental assessment of waste management systems by applying sensitivity analysis to different case studies.

After introducing the relevance of environmental assessment studies in the field of waste management, the legal basis, challenges and limitations of the Life Cycle Assessment method were discussed. The challenge of uncertainties of assessment results, focusing on modelling decisions was highlighted using three waste management case studies. A review was carried out on the historical development of "Landfill Mining", providing a global overview of the main drivers of this waste management process and related assessment challenges.

Typical uncertainties within the Life Cycle Assessment of waste management systems were presented and the applied method of sensitivity analysis to assess and reduce uncertainties was explained. Parameter and scenario uncertainties were assessed in detail in the course of the second case study “collection and utilization of waste cooking oil from households in Austria”.

On the basis of these research findings, a stepwise approach for the assessment of software tools as an uncertainty in the course of an environmental assessment was developed, and applied, for the comparison of two different software tools. Focus was put on the choice of LCA software tools and the associated limitations during the actual modelling process. The comparison included an analysis of the included databases, the offered Life Cycle Impact Assessment methods, the implemented characterization factors and inventory system of elementary exchanges, the calculation models implied and their effect on generated mass and substance balances. Concerning the included LCIA methodologies, the comparison showed that the implemented ILCD recommended list of impact categories and methodologies differed in seven out of thirteen impact categories. This was a very critical finding, as it makes it harder for practitioners to compare LCA results, which are stated to follow the ILCD recommendations. The contribution analysis on a substance level of the impact category “particulate matter” showed how careful LCIA practitioners have to choose and work with LCIA methods. The software users need to analyze the LCIA methods in detail, in advance, closely assessing the substances included and also the compartments and sub-compartments to which they are assigned. It was found that compartments and sub-compartments for the inventorying of elementary exchanges were defined differently within both software tools, which affected the overall results. Furthermore, software users need to determine whether long-term emissions are included as a default setting.

The assessment of different modelling choices showed that the application of aggregated or self modelled datasets for foreground processes led to substantial and major deviations in all impact categories (except for one in EASETECH). The modelling choice of selecting different datasets for background processes revealed lower deviations than the choice of foreground processes. Still, substantial and major deviations were found in more than half of the impact categories assessed in both software tools.

Three case studies were used throughout the work to exemplify (1) Landfill Mining, (2) Utilization of waste cooking oil from households, and (3) Biological waste treatment options for biowaste.

5.2. DISCUSSION AND OUTLOOK

The aim of this thesis was to increase the awareness of the LCA community and the recipients of the LCA studies about the influence of defined modeler choices on the validity of the results and the conclusions drawn from LCA studies in the field of waste management. Modelling choices within environmental assessments are always interlinked with human judgement, which again differs from person to person.

The results obtained from this thesis are, in part, very specific to the defined case studies. Concerning the software comparison elaborated on in chapter 4 other substances, emissions and impact categories can deviate to a different extent when modelling other waste management technologies. However, general weaknesses and limitations of software tools were uncovered with this case study which can be valuable for the LCA community in general. Furthermore, it needs to be stated that it was not the aim of paper V to define which software tool is better or worse. The selection of the software tool strongly depends on the goal and scope of the study. A software, such as “GaBi”, allows the modelling of systems with little complexity within reasonable time and sufficient quality. The input-specific modelling principle of EASETECH enables the assessment of effects of changes on a detailed substance and process level of a waste management system. This might be of greater interest for practitioners in the field of waste management.

In general the findings of this thesis underline the need for LCA practitioners and the recipients of LCA studies to be critical with LCA results. It is of outmost importance that LCA modelers are critical with their own decisions and modelling choices, such as the definition of substituted primary production processes. LCA software users need not trust the tools blindly, nor transfer the responsibility for the validity of their results to the software tool applied, or the provided data. It was shown that sensitivity analysis helps to reflect and reduce the uncertainty inherent to any complex decision making process and increases reliability and transparency in LCA results.

Still, in the future more transparency is needed, regarding these concerns. LCA practitioners should state the chosen LCA software tool, the LCIA methodology applied plus the version of the method and the implemented characterization factors, as the stated ILCD recommended normalization methods also differed between the two software tools assessed.

Future research in the field of waste LCA and LCA software tools should focus on harmonizing the different inventorying principles of elementary exchanges, such as definition of compartments and sub-compartments. The results highlight the need for a common format (and even database) for impact methods and elementary exchanges which ensures equivalent characterization across software tools and models. There are already very good initiatives concerning harmonization within the LCA community such as the European Life Cycle Database or the open source software “OpenLCA” from Green Delta. Still, the findings of this study are once more a request for more transparency in LCA studies from all parties concerned, the software providers, the database providers and the LCA modelers.

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PART II: PUBLICATIONS