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PhD Thesis
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The synopsis part of this thesis is available as a pdf-file for download from the DTU research database ORBIT: http://www.orbit.dtu.dk

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Preface

The work presented in this PhD thesis was conducted from January 2012 to November 2015 at the Department of Environmental Engineering of the Technical University of Denmark under the supervision of Associate Professor Thomas Fruergaard Astrup.

The PhD thesis is organised in two parts: the first part puts the findings of the PhD into context; the second part consists of the four scientific journal papers listed below. The papers are referred to by their roman numerals throughout the thesis, e.g. Paper (**I**).

I Martinez-Sanchez, V., Kromann, M. A., Astrup T.F. Life cycle costing of waste management systems: Overview, calculation principles and case studies. Waste Management 2015, 36, 343-355.

II Martinez-Sanchez, V., Tonini, D., Møller, F., Astrup, T.F. Life Cycle Costing of Food Waste Management in Denmark: Importance of indirect effects. Submitted to Environmental Science and Technology.

III Martinez-Sanchez, V., Hulgaard, T., Hindsgaul, C., Riber, C., Kamuk, B., Astrup, T.F. Estimation of Marginal Costs at Existing Waste treatment Facilities. Submitted to Waste Management.

IV Martinez-Sanchez, V., Levis, J.W., Ranjithan, R., DeCarlis, J.F., Barlaz M.A., Damgaard, A., Astrup T.F. Evaluation of Externality Costs in Life Cycle Optimization of Solid Waste Management. To be submitted to Environmental Science and Technology.

In this online version of the thesis, papers **I-IV** are not included but can be obtained from electronic article databases e.g. via www.orbit.dtu.dk or on request from DTU Environment, Technical University of Denmark, Miljoevej, Building 113, 2800 Kgs. Lyngby, Denmark, info@env.dtu.dk.

In addition, the following publications, not included in this thesis, were also concluded during this PhD study:

Rigamonti, L., Grosso, M., Møller, J., Martinez Sanchez, V., Magnani, S., Christensen, T.H., 2014. Environmental evaluation of plastic waste management scenarios. Resources, Conservation and Recycling 85, 42–53

Tonini, D., Martinez-Sanchez, V., Astrup, T.F., 2013. Material resources, energy, and nutrient recovery from waste: are waste refineries the solution for the future? Environmental Science and Technology 47, 8962–9

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Thomas Fruergaard Astrup was not only a co-author but also a good supervisor. I thank him for offering me the possibility of working on this topic, for giving me the freedom to frame this PhD and at the same time guiding me while framing it, as well as for his constant support.

I would also like to thank *Charlotte Scheutz* and *Anne Harsting* for their continuous help during the PhD, *Thomas Højlund Christensen* for introducing me to the waste field, *Lisbet Brusendorff* and *Torben Dolin* for their design and graphics help, and *Hugo Maxwell Connery* for his programming support.

I would also thank my colleagues from DTU Environment for creating such a nice atmosphere at work and I'm especially grateful to the "waste group", not only for their stimulating discussions but also for all the enjoyable breaks. Special thanks go to my office mate *Kos*, my Danish friend *Line*, and my Italian office neighbours *Elisa* and *Stefania* for their moral support and friendship.

I am very grateful to all my friends in Denmark who have also helped me to pursue this PhD by making this cold country warmer and for trying not to talk about work.

Last, but not least, a big THANKS to my *family,* who have always believed in me and supported me in all my decisions, and to *Luca* for always putting life into perspective and taking good care of me.

Summary

The Solid Waste Management (SWM) sector has evolved from a simple control of emissions towards a resource recovery sector while still being constrained by strict emission regulations. For that waste authorities are paying increased attention to the waste hierarchy as a set of priorities for solid waste treatment options to boost this shift towards higher resource recovery. In this hierarchy, waste prevention has the highest priority, followed by re-use and recycling options, and what cannot be recycled should be energy recovered; and, finally, the least favoured option is disposal in landfills. However, the waste hierarchy does not consider the local needs/conditions of each geographical area, and it cannot be used to identify sustainable SWM options by itself. Environmental impact assessment can help with this task as holistic decision-support tool. Nevertheless, waste authorities need economic assessment of SWM systems alongside environmental impacts assessment to take budget constrains into account.

In light of the need for combined environmental and economic assessment of SWM, this PhD thesis developed a consistent and comprehensive method for integrated environmental and economic assessment of SWM technologies and systems. The method resulted from developing further the generic Life Cycle Costing (LCC) framework suggested by Hunkeler et al. (2008) and Swarr et al. (2011) to apply it on the field of SWM. The method developed includes: two modelling approaches (Accounting and Optimization), three cost approaches (Conventional, Environmental and Societal LCCs) and two goal perspectives (Planning and Analysis).

- The modelling approach describes how the scenarios are defined. The "Accounting approach" defines the technological pathway of each scenario before the study is performed, while in the "Optimization approach" the scenarios are the results of an optimization process.
- The cost approach describes cost principles and level of LCA integration. Conventional and Environmental LCCs are financial assessments, i.e. include marketed goods/services, but while Environmental LCCs include environmental impacts in a parallel LCA, Conventional LCCs do not. Societal LCC is a welfare economic assessment, i.e. includes marketed goods/services and effects outside the economic system (externality costs).

- The goal perspective differentiates between "Analysis" and "Planning" perspective. Analysis LCCs evaluate current status of a SWM system, while Planning LCCs focusses on the consequences a change in a system with respect to the status quo.

The applicability of the LCC framework was tested through four case studies from which the following conclusions can be drawn:

- Organic source-segregation incurs financial and social costs mainly related to the cost of bags and bins used by households, as well as extra collection costs related to the additional collection scheme for organics collection as well as extra cost of residual waste collection (compared with mixed waste).
- The environmental benefits related to food waste prevention (due to avoided food production) could be overtaken by the environmental loads associated with the alternative consumptions purchased with the savings generated from the prevented (unpurchased) food. This could be avoided if prevention campaigns were accompanied by other policies aiming at reducing the impact of alternative consumption patterns. The inclusion of these income effects is especially critical when the alternatives being compared in an LCC have significant differences on the use of scarce resources such as income, land and time. In these cases, LCC studies should be supplemented by specific analysis of potential behavioural changes in consumption patterns (defining alternative consumptions) associated with the SWM systems being assessed.
- Recycling and prevention strategies can have significant economic consequences in existing waste facilities whose operation will have to be adjusted based on the waste changes. Marginal costs of diversion strategies in existing WtE facilities depend completely on the response in such facility. However, regardless of the response type, it was demonstrated that marginal costs of diversion are several times different than average costs. Hence, when performing Planning LCC the dynamics of the SWM system (including effects in existing waste facilities) should be taken into account to avoid misleading conclusions.
- Optimization of SWM using Societal LCC demonstrated that the social optimal solution results from balancing economic and externality costs. Contrary, optimizing using either economic costs or externality costs lead to socially suboptimal solutions.

Dansk sammenfatning

Affaldssektoren har gennem det seneste årti udviklet sig fra en sektor med primær fokus på minimering af miljøpåvirkningen fra affaldshåndteringen, henimod en sektor med fokus på genvinding og udnyttelse af ressourcerne i affaldet. Affaldshierarkiet udgør en overordnet rettesnor for lovgivning på affaldsområdet og prioriterer forebyggelse af affald højest, efterfulgt af genbrug og genanvendelse, dernæst nyttiggørelse med henblik på energiproduktion, og endelig deponering af affald som den mindst prioriterede behandlingsform. Affaldshierarkiet tager dog ikke hensyn til lokale behov og betingelser, hvorved affaldshierarkiet ikke alene kan anvendes til at identificere de optimale affaldsløsninger. Miljøvurderinger, f.eks. livscyklusvurderinger (LCA), kan bidrage til fastlæggelse af de miljømæssigt optimale affaldsløsninger, men for at opnå et helhedsorienteret beslutningsgrundlag er en sammenlignelig økonomisk analyse af samme system nødvendig. Disse to typer af analyser er oftest adskilte og vanskelige at sammenligne for beslutningstagere.

For at forbedre grundlaget for beslutningsstøtte på affaldsområdet er der i denne PhD-afhandling udviklet en metode til systematisk og integreret analyse af både de miljømæssige og økonomiske aspekter af affaldsteknologier og -systemer. Metoden er en videreudvikling af en generisk metode ”Life Cycle Costing” (LCC) foreslået af Hunkeler et al. (2008) og Swarr et al. (2011). Den udviklede metode er specifikt tilrettet anvendelse på affaldsområdet og omfatter syv forskellige tilgange til LCC: i) to forskellige typer af modellering (*Accounting* og *Optimization*), ii) tre forskellige niveauer for beregning af omkostninger (*Conventional*, *Environmental* og *Societal*) og iii) to forskellige typer af målsætninger (*Analysis* og *Planning*).

- Modelleringstypen beskriver definitionen af scenarier indeholdt i analysen. *Accounting*-typen repræsenterer scenarier, som defineres på forhånd, mens *Optimization*-typen repræsenterer en analyse, hvor scenarierne er udkommet af en optimeringsproces på basis af definerede målsætninger og forudsætninger i modellen.
- Niveauer for beregning af omkostninger angiver beregningsprincipper og niveau for integrering af resultater fra en sammenhængende LCA. *Conventional* og *Environmental* LCC'er er finansielle analyser, dvs. analyserne inkluderer markedsførte produkter/services, men hvor en

Environmental LCC inkluderer miljøpåvirkninger i en parallel LCA, er dette ikke tilfældet i en *Conventional* LCC. En *Societal* LCC er en velfærdsøkonomisk analyse, dvs. analysen inkluderer markedsførte produkter/services samt effekter udenfor det økonomiske system (eksternaliteter).

- Målsætningen for en analyse kan differentieres mellem *Analysis* og *Planning*, hvor en *Analysis* LCC evaluerer status for et eksisterende affaldssystem, mens en *Planning* LCC estimerer de økonomiske og miljømæssige effekter relateret til en potentiel ændring i et affaldssystem i forhold til det eksisterende system.

De syv forskellige aspekter af LCC er blevet evalueret via fire illustrative "case studies". Følgende overordnede konklusioner kan fremhæves:

- Organisk kildesortering medfører finansielle og sociale omkostninger primært relateret til prisen på skraldeposer og containere i husstandene samt omkostninger forbundet med øget indsamling af affaldet.

- Ved forebyggelse af madaffald er der en risiko for, at miljøgevinsterne ved mindre fødevareproduktionen modsvares af en øget miljøbelastning forbundet med et øget alternativt forbrug af andre varegrupper (idet der spares penge på ikke-indkøbte fødevarer). Kampagner for forebyggelse af affald bør derfor ledsages af strategier, der enten mindsker negative miljømæssige effekter af alternativt forbrug og/eller understøtter ændrede forbrugsmønstre hos borgerne. LCC af forebyggelsesstrategier bør derfor suppleres af specifikke analyser af de adfærdsmæssige konsekvenser (f.eks. øget alternative forbrug) forbundet med forebyggelsesstrategierne.

- Genanvendelses- og forebyggelsesstrategier kan have betydelige økonomiske konsekvenser for eksisterende affaldsbehandlingsanlæg, som f.eks. bliver nødt til at justere driften som følge af ændringer i det modtagne affald. De marginale omkostninger ved sådanne ændringer afhænger af, hvordan de enkelte anlæg reagerer på ændringer i affaldsstrømmene, f.eks. import af affald til forbrænding ved faldende lokale affaldsmængder. Uanset reaktionen blev det for forbrændingsanlæg vist, at de marginale omkostninger ved ændrede affaldsstrømme afveg betydeligt fra de gennemsnitlige omkostninger. For at undgå fejlbehæftede konklusioner bør de ændrede betingelser og marginale omkostninger for eksisterende anlæg inddrages ved *Planning* LCC'er.

- Ved optimering af de samfundsøkonomiske (*Societal* LCC) konsekvenser ved specifikke affaldsløsninger blev det vist, at den samfundsøkonomisk set optimale løsning fremkommer ved en afbalancering af de finansielle omkostninger og omkostningerne forbundet med eksternaliteter. Optimering udelukkende ved brug af den ene type af omkostninger fører til samfundsøkonomisk set sub-optimale løsninger.

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Abbreviations

AFW	Animal Food Waste
BC	Budget Cost
CHP	Combined Heat and Power
EC	Externality Cost
FU	Functional Unit
GHG	Greenhouseh Gas
iLUC	Indirect Land Use Changes
LCA	Life Cycle Assessment
LCC	Life Cycle Costing
LHV	Lower Heating Value
MFH	Multi-Family Housing
MSW	Municipal Solid Waste
SFH	Single-Family Housing
SWM	Solid Waste Management
VFW	Vegetable Food Waste
WMS	Waste Management System
WtE	Waste-to-Energy

1 Introduction

Over recent decades, due to increasing resource scarcity, more focus has been given to the circular economy. For this, the solid waste management (SWM) sector has evolved from the simple control of emissions towards a resource recovery sector while still being constrained by strict emissions regulations. This change can be observed in the increase in waste-to-energy (WtE) capacities (Astrup et al., 2014), improved recycling rates (Bruvoll et al., 2002; Eurostat, 2015) and increased gas recovery from landfills (EEA, 2009).

The waste hierarchy was suggested by environmental organizations and the European Union as a set of priorities for solid waste treatment options (Berglund, 2006) to boost this shift towards greater resource recovery. In this hierarchy, waste prevention has the highest priority, followed by re-use and recycling options, and what cannot be recycled should be recovered as energy; finally, the least-favoured option is disposal in landfills. Waste authorities are paying increased attention to the waste hierarchy to meet specific environmental targets, such as the recycling targets of the European Commission for 2030 (EC, 2015). However, the waste hierarchy does not consider the local needs/conditions of each geographical area, and it cannot be used to identify sustainable SWM options by itself. For example, there is a specific break-even point in which further waste improvement (moving up in the hierarchy) is no longer justified from a resource, environmental or resource perspective. For this, an environmental impact assessment can help to find these optimal solutions.

Nevertheless, waste authorities do not base their decisions only on environmental performance of the different options because there are often budget constraints. The lack of a balanced economic assessment alongside environmental impacts assessment limits the value of both types of study, as economic priorities are then de-coupled from environmental aspects. Consequently, sustainable solid waste management development should not only take into account the environmental aspects but also the economic aspects of SWM.

While for the environmental assessment of SWM systems Life Cycle Assessment (LCA) is broadly accepted, there is no common decision-supporting tool for the economic assessment of SWM systems. This absence has incentivized each practitioner to develop her/his own economic decision-supporting tool based on the specific needs of each project. However, none of the developed tools works towards a consistent framework for the integrated environmental and economic assessment of SWM systems.

Since this is a need not only in the waste sector but also in other areas, Hunkeler et al. (2008) and Swarr et al. (2011) have suggested Life Cycle Costing (LCC) as a consistent framework for combining LCA and economic assessments of general products. They distinguished between three types of LCC: Conventional LCC, Environmental LCC and Societal LCC, each differing in the economic method applied and the integration level with the LCA. Conventional LCC is a financial assessment applied as a stand-alone economic approach. Environmental LCC is also a financial assessment, but it also includes a parallel LCA. Societal LCC is a welfare economic assessment that includes marketed goods as well as externality costs; the latter includes emissions from parallel LCA internalized in monetary terms.

Nonetheless, SWM differs from common products in that: i) waste composition is highly heterogenic, ii) waste facilities have little control over their input and do not aim at maximizing profits but rather minimizing emissions, and iii) waste systems are closely related to production sectors such as the energy sector. These particular characteristics of SWM compared with general products recommend a specific method for integrated environmental and economic assessment to meet the specific needs of SWM.

1.1 Aim of the PhD thesis

The main aim of this PhD thesis has been to provide a systematic framework for performing economic assessment of waste management technologies and systems with different levels of integration with LCA. This was done by developing further the generic LCC framework suggested by Hunkeler et al. (2008) and Swarr et al. (2011) in order to apply it in the field of SWM. This involved the following detailed objectives:

- Develop a consistent and comprehensive framework for performing LCC of SWM technologies and systems based on the learning obtained from the literature.
- Demonstrate the applicability of the developed LCC framework through several case studies.
- Assess current SWM systems (represented by the case studies) based on cost data from full-scale waste facilities with the LCC framework.

- Identify challenges in the application of the LCC as well as the critical assumptions and needs in order to perform economic assessment of SWM technologies and systems.

1.2 Content of the PhD thesis

The structure of this PhD thesis is as follows:

- Chapter 2 describes the LCC method umbrella developed and applied in this PhD thesis. This chapter elaborates upon the method developed in Papers (**I**, **III,** and **IV**).
- Chapter 3 presents the critical factors for LCC of SWM systems and builds on the lessons learned from the case studies of Papers (**I**, **II**, **III** and **IV**).
- Chapter 4 highlights the outcomes of the four case studies from Papers (**I**, **II**, **III** and **IV**).
- Chapter 5 presents a discussion of the most problematic issues identified in the case studies and provides some recommendations on how to apply LCC of SWM.
- Chapter 6 concludes the outcomes of this PhD thesis.
- Chapter 7 provides perspectives and future work for integrated environmental and economic assessments of waste management systems.

2 Method

Due to the absence of a standard method to perform economic assessments of SWM, each practitioner has developed her/his own method based on the needs of each project. Consequently, several methods exist in the literature but none of them is sufficiently comprehensive to include: i) several perspectives, e.g., of the waste operator and of the waste authority, and ii) different costs principals, e.g., financial and welfare economics.

Here, we develop a method able to meet the needs of the SWM field, and for this LCC is used as a method umbrella that covers different types of assessments, each of them having specific applications as well as limitations. The LCC method umbrella includes:

- Two modelling approaches: Accounting and Optimization.
- Three cost approaches: Conventional LCC, Environmental LCC and Societal LCC.
- Two goal perspectives: Planning and Analysis.

These approaches are described within this section, and Figure 1 illustrates the LCC method umbrella.

Figure 1: Illustration of the method umbrella used in this PhD thesis. An LCC can have either an Accounting or an Optimization modelling approach, one of three cost approaches (Conventional, Environmental or Societal LCC) and one of two goal perspectives (Planning or Analysis).

2.1 The Modelling approach

The modelling approach describes how the scenarios are defined, and it applies not only to LCC but also to LCA. The most common modelling approach is the "Accounting approach", which uses "what-if" scenarios in which the technological pathway is defined before the study is performed. On the other hand, there is the "Optimization approach" in which the scenarios are the result of the study and they are defined to meet specific objective and constraints. Juul et al. (2013) assigned the label "system assessment tools" to those models using the "Accounting approach", and "system engineering models" to those models using the "Optimization approach".

Regardless of the modelling approach used, the functional unit of the study has to be defined from the outset. This definition often includes the management of the waste generated by a specific waste generator over a specific amount of time, and its reference flow is often defined by an amount of waste with a specific composition.

2.1.1 The Accounting approach

The Accounting approach is relevant when the decision-maker: i) has an idea of the feasible SWM strategies, ii) does not have full control of the whole system being assessed due to external constraints, and iii) has the capacity to establish a trade-off between the different criteria being evaluated, e.g., giving more importance to reducing greenhouse gas (GHG) emissions than costs.

The Accounting approach has been used: i) to identify the most environmental option within the selected technological pathways though LCAs by Astrup et al. (2014), Damgaard et al. (2011) and Tonini et al. (2013, 2012), ii) to identify the most economical option within the selected technology scenarios though economic assessments by Carlsson Reich (2005), Gomes et al. (2008), Groot et al. (2013) and Teerioja et al. (2012), and iii) to compare the welfare effects of predetermined choices by DCFEE (2013), Eshet et al. (2006) and Vigsø (2004).

While these studies provided useful conclusions among the options compared, they say little about other potential technological pathways. On the other hand, they are simpler to perform and interpret than the Optimization approach, as explained below.

2.1.2 The Optimization approach

The Optimization approach can provide useful outcomes when the decision-maker has: i) autonomy to build the SWM system or control the important parts of the system (Juul et al., 2013), and ii) a single criterion (or a few criteria) as an objective.

This approach is not so common in the field of waste management, and only a few studies, such as Levis and Barlaz (2011), Levis et al. (2014, 2013) and Vadenbo et al. (2014a, 2014b), have used it to identify the waste strategies minimizing GHG emissions, landfilling and economic costs.

The limitation of this approach relates to the complexity of the computational work as well as the interpretation of the results because many simulations need to be run to understand overall trends.

2.2 The cost approach

The cost approach is used to describe cost principles as well as LCA integration. Three types of cost approaches were developed for SWM in Paper (**I**): Conventional LCC, Environmental LCC and Societal LCC.

Before going into each type of cost approach it is important to distinguish between three types of costs:

- ***Budget costs***: these represent marketed goods/services incurred by a waste generator/operator and can be either "one-off", occurring once during the lifetime of a technology (e.g., capital investment or back-end costs) or recurring (e.g., operational and maintenance costs).
- ***Transfers***: these are monetary flows representing the income distribution between stakeholders while not leading to the reallocation of resources such as land and labour or welfare changes in society (Danish Center for Environment and Energy, 2013), e.g., environmental taxes and subsidies or general taxes such as value added tax (VAT).
- ***Externality costs***: these represent the effects on the welfare of individuals of activities which are not compensated. Externalities can be environmental (e.g., relating to emissions) or non-environmental (e.g., in the form of odour from waste facilities or time spent by households on waste sorting).

Conventional and Environmental LCCs are both financial assessments (i.e., they include marketed goods/services) while Environmental LCC includes environmental impacts in a parallel LCA, Conventional LCCs do not. Societal LCC is a welfare economic assessment (i.e., it includes marketed goods/services as well as effects outside the economic system, namely externality costs). The three cost approaches are described below.

2.2.1 Conventional LCC

The Conventional LCC results from the sum of the budget costs and transfers for *n* activities involved in the SWM system, as shown in Equation 1.

The budget cost of each activity i results from multiplying the unit budget cost of activity i (UBC_i) accounted for in factor prices (i.e., market prices excluding transfers) by the amount of waste input into the same activity (W_i). The transfer of activity i results from multiplying the unit transfer of activity i (UT_i) and the waste input amount into each activity i (W_i):

$$Conventional\ LCC = \sum_{i=1}^{n}[W_i * (UBC_i + UT_i)] \quad (1)$$

Conventional LCCs have been used in the literature: i) to assess the economic feasibility of treatment solutions (Coelho and De Brito, 2013; Franchetti, 2009), ii) to identify the most economical SWM option (Groot et al., 2013; Karagiannidis et al., 2013) and iii) to evaluate the economic consequences of implementing a specific waste solution (Gomes et al., 2008).

2.2.2 Environmental LCC

The Environmental LCC results from the sum of the budget costs and transfers again but adds transfers anticipated to be established in the near future, i.e., externalities expected to be internalized in the economic system from a time perspective relevant for the decision being assessed. The anticipated transfer of each activity results from multiplying the unit anticipated transfer of activity i (UAT_i) by the waste input amount into each activity i (W_i), as shown in Equation 2.

The economic results of the Environmental LCC are complemented by an LCA for the same system without double-counting emission effects, i.e., once they are internalized in the economic part with its accounting price representing their damage, they should not be accounted in the environmental part (the LCA).

$$Environmental\ LCC = \sum_{i=1}^{n}[W_i * (UBC_i + UT_i + UAT_i)] \quad (2)$$

Environmental LCCs are typically intended to supplement LCAs with an economic performance assessment, as in Consonni et al. (2005). Very few studies addressed the combined environmental and economic assessment of

SWM, e.g., economic assessments are often performed separately from the LCA and tend to use different system boundaries and assumptions (Carlsson Reich, 2005; Hunkeler et al., 2008; Norris, 2001; Swarr et al., 2011).

2.2.3 Societal LCC

The Societal LCC results from the sum of the budget costs and externality costs, both accounted for in accounting prices, i.e., the price that reflects the opportunity cost (Nordic Council of Ministers, 2007), as shown in Equation 3. The unit budget costs of activity i in the factor prices (UBC_i) are multiplied by the Net Tax Factor (NTF) to represent the shadow prices of marketed goods/services. In Denmark, the NTF for converting the current factor prices to the shadow prices of marketed goods is estimated to be 1.17 (Danish ministry for Environment, 2010). The externality costs of activity i result from multiplying the unit externality cost of activity i (UEC_i) by the waste input amount of each activity i (Wi).

$$Societal\ LCC = \sum_{i=1}^{n} [W_i * (UBC_i * NTF + UEC_i)] \quad (3)$$

Societal LCCs are often used to examine the economic efficiency of specific scenarios at a societal level in order to estimate welfare losses and gains related to re-allocating resources, as in DCFEE (2013), Miranda (1997) and Vigsø (2004).

2.3 Goal perspectives

LCCs can be applied either from an “Analysis” or a “Planning” perspective. While Analysis LCCs aim to evaluate or describe the current status of an SWM system, Planning LCCs focus on the effects of a change in a system with respect to the status quo. They are also referred as “ex-ante” and “ex-post” studies by the Nordic Council of Ministers (2007). Ex-ante studies examine the efficiency of various options for implementing a waste policy, while ex-post studies evaluate whether the actual implemented policy has been efficient or not. This distinction is also applied in LCA, whereby Planning LCA is also called “consequential LCA” and Analysis LCA is also named “attributional LCA”.

Analysis LCCs can either be applied to identify hotspots for further research or to calculate the gate fees of waste technologies. In that case, the focus is only on the average costs of waste systems/technologies as representing the unit cost of the waste treatment/disposal.

Planning LCCs can have two time-perspectives - short- and long-term - depending on whether the study focuses on deciding the best alternative for society at present (i.e., short-term, taking into account the investments already made) or the most beneficial strategy for society in the long-term, when it is not tied to any current investment (Nordic Council of Ministers, 2007). When using the short-term perspective (i.e., the best option here and now), a change in the SWM system may affect: i) existing facilities whose operations have to be adjusted to the new situation, ii) existing facilities that may have to be decommissioned, and iii) new facilities that may have to be designed and constructed. Here, the costs of existing facilities are called "post-design costs", and costs involving the design, construction and operation of new facilities are called "design costs". In long-term perspective studies, the effects on existing studies are excluded and only "design costs" are assessed.

While design costs have been used in the literature: i) to identify economics of scale and scope (Callan, 2001; Criner, 1995; ENEA, 2007), ii) to assess the economic viability of new facilities (Coelho and De Brito, 2013; Franchetti, 2009; Kang and Schoenung, 2006), and iii) to compare the costs of different SWM options (Bel and Fageda, 2010; Consonni et al., 2005; De Feo and Malvano, 2012; De Jaeger et al., 2011; Zhang, 2013), post-design costs have not been addressed in the SWM literature so far.

SWM systems are networks of waste facilities interrelated by waste flows, whereby existing facilities are affected by changes occurring either upstream or downstream. For this reason, when performing short-term Planning LCC, the dynamics occurring within this network should be taken into account. For this, it is crucial to evaluate the responses in existing facilities, i.e., how the waste operators of existing facilities adjust their operations to changes in the system. However, these effects are often assumed to be negligible and the average costs in affected facilities are thus assumed to be constant. Nevertheless, these costs are only constant when no change occurs.

The results of Planning LCCs - both short- and long-term - can be shown as the marginal costs of the change, i.e., the total costs arising from the change or by comparing the average costs with and without change. In theory, the calculation of the marginal costs could be done with the derivative of the average costs function - the problem is that most of the time such a function does not exist or if the function exists, the change assessed can affect multiple variables that are not included in the function.

For this reason, Paper (**III**) developed a numerical method to estimate the marginal costs for existing facilities resulting from a small change in waste management based on their average costs functions to be applied in short-term Planning LCC. The idea behind the method is illustrated in Figure 2 and consists of three stages: 1) an initial stage, in which expert knowledge is used to define a cost model describing the average costs of an existing facility as a function of key plant and waste parameters, 2) a second stage, in which the results of the model are evaluated by comparing them with the literature data (real cost data), and 3) a third stage, in which marginal costs are estimated by performing a change in the waste input, and comparing the average costs with and without waste change using the same cost model defined in "1)". In the third stage, it is crucial to consider what type of response will generate the waste change in the facility. For example, if plastic waste is diverted away from the Waste-to-Energy (WtE) plant, its operators can either react by importing refused derived fuel (RDF) or biomass to compensate for thermal losses, or else not react and just incinerate less waste, albeit with a loss in utilized thermal capacity. Such a reaction should be taken into account when calculating the average costs of the new situation.

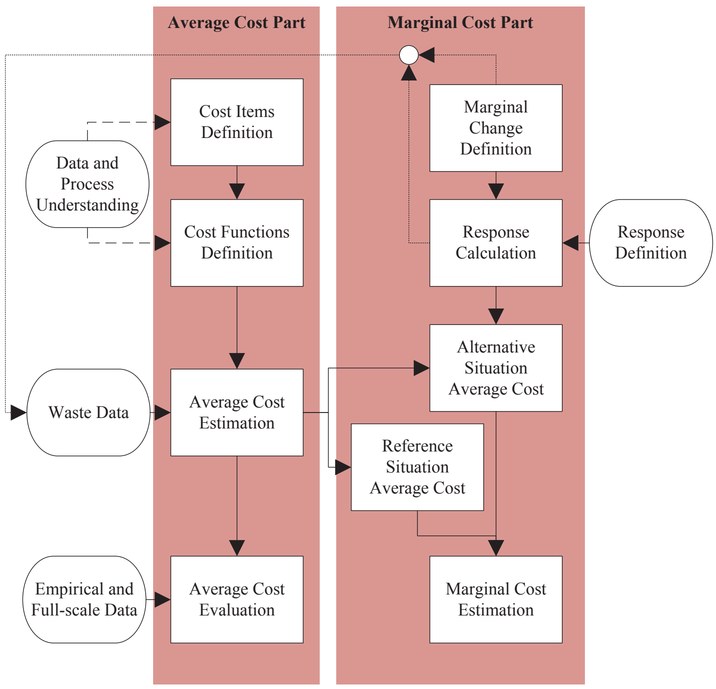

Figure 2: Illustration of the method developed to estimate the marginal costs of diversions from Paper (**III**)

3 Key factors in LCC of SWM systems

The evaluation of the key studies in the existing literature was used to identify important factors to perform LCC of SWM systems, and they are described in what follows.

3.1 Importance of the goal and scope definition

The definition of the goal and scope is the first and most important step in performing an LCC. At this stage, the LCC practitioner defines the aim of the project and which perspective to use to satisfy the needs of the project.

SWM systems involve stakeholders with different interests, namely waste generators, waste facility operators and waste authorities. While waste authorities often need a holistic perspective, i.e., covering the whole SWM system, waste operators are often only interested in the costs of single parts of the waste chain. The system boundaries of the study are established based on the stakeholder's perspective.

For example, some studies have used a technological focus to assess collection costs (De Feo and Malvano, 2012; Gomes et al., 2008; Larsen et al., 2010) or the costs of specific treatment or disposal options (Bozorgirad et al., 2013; Damgaard et al., 2011), while Dahlbo et al. (2007), Foolmaun and Ramjeeawon (2012) and Levis et al. (2014) used a system perspective to assess the whole waste system for specific waste fractions and which can be used by decision-makers who have control over the whole waste chain.

In addition, at this stage, the practitioner together with the decision-maker decides which LCC approach satisfies the needs of the project, i.e., a modelling approach, a cost approach as well as goal perspective. The definition of the cost approach is especially critical, since some aspects can only be captured by one of the cost approaches. For example, if a municipality is considering changing the waste management of its geographical area towards a system that requires more effort by households (Berglund, 2006; Hage, 2008), the value of the households' efforts should be accounted for in the assessment, which can only be done with Societal LCC.

3.2 Mechanistic vs. empirical approaches

LCC models can be mechanistic, empirical or a combination of both. While mechanistic models are based on the understanding of the system represented by sets of equations, empirical models are based on direct observations, measurement and data records. The input data availability as well as the needs of the project determine the degree of mechanistic modelling and the empirical data used.

Empirical data in LCC of SWM are often used for costs: i) which are not controlled by the decision-maker, e.g., the gate fees of downstream activities, ii) whose understanding is beyond the scope of the LCC practitioner, iii) with little influence on the overall results. This type of datum can be accurate for a case study in the same context as that in which the measurement/reporting was done, but the challenge appears when the specific data representing a case study context do not exist and data from other contexts are used. For example, the price of the collection service per tonne of residual waste in the context of organic source separation is different to the price per tonne of mixed waste without organic source separation, because the distance, time and consumption involved are different per tonne of waste for each collection scheme. Using the empirical data from one context in another can mislead, generating biased results.

Mechanistic approaches are often used for individual parts of the waste chain on which the interests of decision-makers are focused, e.g., Gomes et al. (2008) have used them to define collection costs by developing sets of equations describing costs as functions of several parameters, both physical and economic. The model itself can be transferred to other case studies by updating the values of the parameters to represent the context of the case study in question. The challenge for this approach stems from the effort needed to understand the individual parts of the SWM as well as the relation between them and the fact that the costs resulting from these models may not represent reality due to some unknown/unmeasurable aspects, such as profits.

3.3 Data limitations

Data gathering is the most demanding step in performing an LCC since cost data are not often available unless decision-makers or project partners have such data (and are willing to disclose it). This has effects on the results of an LCC, since the quality of the LCC outputs is completely dependent on the quality of the LCC inputs. In addition, to determine the accuracy of the LCC outputs it is important to validate the model with real data. However, model validation is often overlooked when performing an LCC.

3.3.1 Input data

Empirical data in LCC of SWM is highly variable according to the time, region, type and size of the facility. Finding representative data for the specific case is challenged by the lack of transparency in reporting costs by waste facilities and authorities.

Data limitations are easier to handle under mechanistic approaches, since the values of the parameters representing the case study being assessed are easier to find than the empirical data. For example, if the collection costs are defined as a function of the fuel price, such costs can be calculated easily by determining the fuel prices in the case study context.

Data gathering is easier for budget costs and transfers than externality costs. This can be explained by the fact that: i) budget costs and transfers are generally reported and/or known by waste facilities and authorities, and to some extent they are also disclosed, and ii) budget costs and transfers are less dependent on the location than externality costs. The region-specific definition of externality costs entails the local estimation of the local damage caused by an effect (e.g., emissions) and the local conversion of such damage into monetary terms. This may be the reason for the severe lack of data regarding externality costs.

Knowledge transfer between case studies is significantly challenged by the aggregation level of the results given within the studies. Generally, the higher the aggregation level, the less knowledge that can be transferred to other cases. Plenty of case studies assessing the costs of waste technologies and systems exist in the literature (e.g., Karagiannidis et al. (2013) and Sonesson et al. (2000)). Although they draw some useful conclusions for the specific case study, the knowledge transfer is limited due to the absence of: i) detailed cost calculation principles, ii) a clear definition of the assessment focus, system boundaries and assumptions, and iii) transparent terminology for describing assessment principles.

3.3.2 Model validation

Model validation serves to verify that the model represents reality, and it is necessary to juxtapose LCC outputs with real data from full-scale waste facilities.

The validation of Analysis LCC and financial assessments (i.e., Conventional LCC and the economic part of the Environmental LCC) is feasible because the results of the LCC can be verified with the current costs, such as the waste fees paid by waste generators or the gate fees of the different facilities. However, this step is almost impossible for the environmental part of Environmental LCC and the Societal LCC, since most of the environmental and non-environmental externalities are not reported or quantified.

In contrast, the validation of Planning LCC is challenged by the fact that the new facilities do not exist when the study is being performed and existing facilities whose operation will be adjusted have not yet responded to the waste-change being assessed, so real data do not exist to validate the outcome of the study.

3.4 The direct and indirect effects of SWM

Most LCC studies of SWM systems include direct costs associated with waste treatment and disposal. Such costs are related to the consumption of resources under the SWM system, including budget costs, transfers and externality costs (e.g., the consumer price for fuel, the tax associated with fuel and the emissions emitted when using this fuel). The material and energy generation in SWM systems can also have direct effects on other sectors' production, e.g., if some energy is generated in the SWM system then other energy generating sources may have to reduce their energy generation.

In contrast, most of the LCC studies exclude/neglect those indirect effects of SWM systems related to behavioural changes and/or re-allocation of scarce resources associated with a variation in the SWM system. Three examples of indirect effects are given here:

- ***Income effects*:** these relate to the fact that incomes of households are constant and/or not affected by the changes being assessed in the LCC of SWM systems, and households will have to adjust other consumption patterns to distribute their income to fit the new situation. The exclusion of this effect underestimates the relation between all the expenses of a waste generator related to the fact that an "income" is a scarce resource. If waste generators have to spend more or less in the management of their waste (e.g., through their waste fees), they will have to reduce other expenses or else use their savings in one way or another. These effects have been assessed in industrial ecology and LCA (so-called "rebound effects") in Binswanger (2001), Hertwich (2008) and Thiesen et al. (2006). Paper (**II**) represents the first attempt to include income effects in LCC of SWM systems, and further details of how this was done are explained in Section 4.2.1.

- ***Time usage effects*****:** these relate to the fact that a household's time is limited, and alternative SWM systems have different associated times attributed to households. Households will have to adjust the usage of their time in relation to other activities based on the time used for SWM activities, e.g., such as the source separation of recyclables. The usage of this marginal time can have associated budget costs, e.g.., the purchase of goods/services, externality costs, emissions associated with the production and consumption of such goods/services, as well as transfers. Some studies have quantified the time spent by households to source-separate waste, e.g., (Bruvoll et al., 2002) estimated that 185 hours are used per tonne of waste by Norwegian households and are willing to pay of US$20 per year for a company to take over the waste sorting activity. However, the inclusion of time usage effects needs to identify the marginal use of such time, e.g., the activities that will be done with the extra time (if no sorting is done by households) or the activities that will cease if more time is spent on SWM activities.
- ***Indirect Land Use Changes (iLUC)*****:** these are associated with the fact that arable land is currently fully utilized and that the additional production of agricultural products, such as food, needs to displace ecosystems (Finkbeiner, 2013; Searchinger, 2008, 2010; Tonini et al., 2015, 2012). These effects have been widely used in product and waste LCAs over the last decade, mainly estimating the GHG emissions associated. However, they have not been included in the economic assessment of SWM. In an attempt to fill this gap in the literature, Paper (**II**) developed a method to estimate economic impacts of iLUC associated with food production upstream the SWM system, and further details of how this was done are explained in Section 4.2.1.

4 LCC of SWM: Case studies

The applicability of the LCC method umbrella described in Chapter 2 was demonstrated through four case studies in Papers (**I**, **II**, **III** and **IV**). This section highlights key modelling aspects, and the main outcomes and limitations of the four cases studies. Table 1 lists the approaches used in the individual case studies (individual papers).

Table 1: Modelling, goal and cost approaches used in the case studies of this thesis.

Paper	Study subject matter	Modelling approach	Cost approach	Goal perspective
I	Danish household waste management	Accounting	Conventional LCC Environmental LCC Societal LCC	Planning
II	Danish food waste management	Accounting	Environmental LCC Societal LCC	Planning
III	Danish household waste incineration	Accounting	Conventional LCC	Planning
IV	U.S. household waste management	Optimization	Societal LCC	Planning

4.1 Danish household waste management

4.1.1 Modelling aspects

In Paper (**I**) two scenarios to manage the annual household waste generated by 100,000 Danish households living in multi-family buildings were compared. The first scenario (Sc. 1) included the incineration of mixed municipal solid waste (MSW), representing the current treatment in Denmark, and the second scenario (Sc. 2) included the source segregation of organic waste with subsequent co-digestion with manure and the incineration of the remaining residual waste (see Figure 3).

Conventional LCCs were used to assess: i) the costs of the whole system, ii) the costs incurred by a household as waste fees, iii) the costs incurred by the collection operator, and iv) the costs incurred by the incinerator operator. An Environmental LCC and a Societal LCC were used to assess the overall costs of the system, i.e., the costs incurred by all the agents involved in the waste management system, namely the same system boundaries as the whole system in the Conventional LCC “i)”.

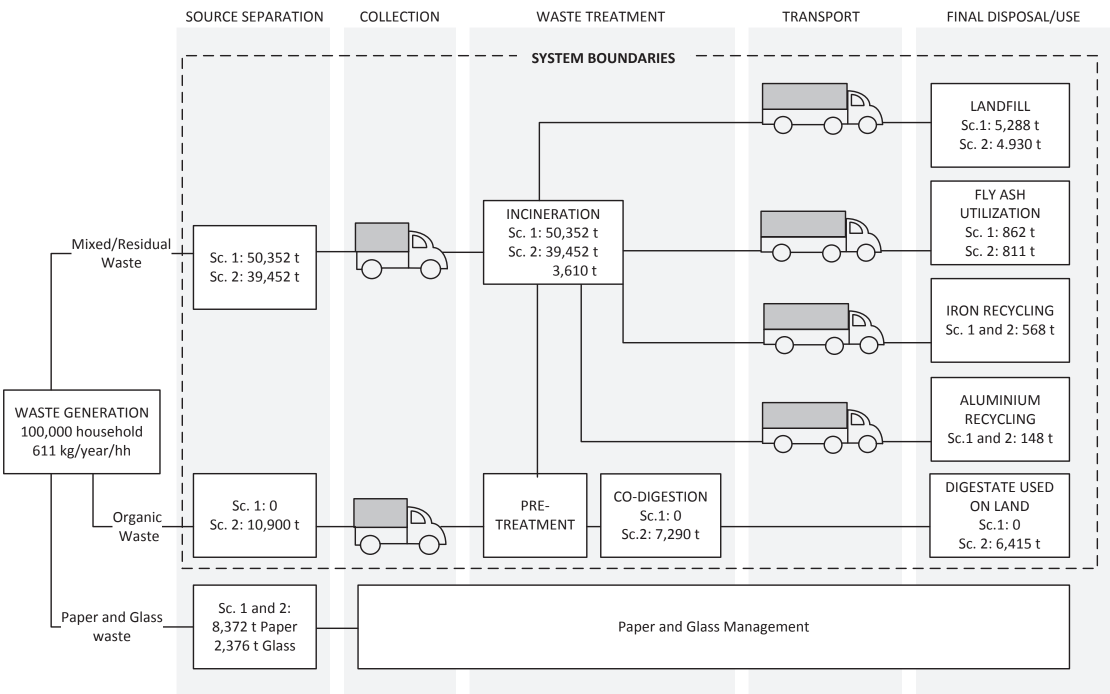

Figure 3: Illustration of the two case study scenarios assessed in Paper (**I**). The diagram includes mass flows per functional unit and the technologies used in the individual phases of the waste management system.

The LCC modelling was as mechanistic as possible, although some empirical data were also used. For example, the costs of collection, transportation and source separation were modelled as functions of several parameters (e.g., distances and the usage percentage of the bag) that were known/assumed for the case study. In contrast, this was not done for the recycling facilities downstream the incineration for the consumption of key waste facilities, such as anaerobic digestion and incineration, whose main consumption comprised empirical data assumed constant per Mg of waste (e.g., 1.5 kg of ammonia water$\cdot$Mg^{-1} waste incinerated).

The externality costs included in the Societal LCC were limited to the emissions associated with available Danish accounting prices, i.e., carbon dioxide, methane, nitrous oxide, particulate matter, nitrogen oxides, sulphur dioxide, carbon monoxide, hydrocarbons, mercury, lead and dioxins. The accounting prices of such emissions stem from Miljoeministeriet (2013). Other emissions/effects related to SWM systems were excluded due to the lack of available accounting prices.

In this case study, the externality cost related to the time used by households to source-separate is particularly critical, because the scenarios being compared do not require the same time used by households to source-separate waste. Several studies assessed the value of this time, e.g., Berglund (2006), Bruvoll et al. (2002), Ekvall (2002) and Sterner and Bartelings (1999); however, no common agreement has been reached (Nordic Council of Ministers, 2007). The overall idea is that if households feel a benefit because of the sorting activity (e.g., if waste sorting and recycling is considered to provide citizens with an added benefit that contributes positively to society), this value should be negative and counted as a benefit (Nordic Council of Ministers, 2007). Conversely, if sorting is considered to be done at the expense of other activities and is experienced as a burden, then the value should be positive and treated as a cost. In order to address this aspect transparently, a neutral value (zero) as a base assumption was used. In addition, a break-even value for the sorting time in which both scenarios resulted in identical social costs (i.e., the same value of Societal LCC) was estimated.

The assessment only included the direct costs of SWM and neglected indirect effects such as income and time-usage effects. The results of the assessment could be significantly affected by the inclusion of these two factors.

4.1.2 Key results

The Conventional LCC assessing the whole system (Figure 4A - Total) revealed that organic source segregation incurred higher financial costs than no source segregation. The extra financial costs corresponded to 16 € year^{-1} household^{-1} and were mainly related to the cost of source separation (e.g., bags and bins) and collection, and slightly compensated with some savings related to the WtE plant (since less waste was incinerated). In this case, it was assumed that the freed capacity in the WtE plant was utilized by other waste.

These extra financial costs could be balanced out if the sales from the digestate correspond to 250 € Mg^{-1} of digestate (instead of giving it away for free). However, this price is far from realistic in the current Danish context where the marketing of digestate as a good source of soil enrichment is challenged (ISWA, 2015) and the prices of mineral fertilizers are economical: 1.2 €·kg^{-1} N-fertilizer, 1.97 €·kg^{-1} P-fertilizer and 0.88 €·kg^{-1} K-fertilizer (Provstgaard and Eskildsen, 2012).

The Conventional LCC focusing, on collection costs (Figure 4A – Collection) showed that the collection costs increased with organic source separation due not only to the extra collection scheme needed for organic waste but also to the fact that the collection cost of the residual waste was 43% higher than the collection cost of mixed waste per tonne (because more collection points were needed to fill up the truck). The results showed that collection costs could be significantly reduced if the number of households sharing one container increased.

The economic part of the Environmental LCC was the same as that of the Conventional LCC when assessing the whole system (Figure 4A - Total), since in this case study there were no anticipated externalities to be internalized in the near future. The Environmental LCC showed that the extra financial costs of the source separation of organics (1.6 M€ FU^{-1}) generated environmental savings in relation to non-carcinogenic human toxicity, freshwater eutrophication, freshwater ecotoxicity and photochemical oxidant formation, but at the same time they induced loads on carcinogenic human toxicity, global warming, terrestrial acidification and resource depletion (Figures 4A and 4B). The inclusion of the capital costs appeared to be critical for carcinogenic human toxicity and the depletion of mineral resources.

The Societal LCC (Figure 4C) demonstrated that the source segregation of organic waste and subsequent treatment led to extra social costs of 1.6 M€, compared with no organic source separation, mainly due to source separation and collection costs. The externality costs of both scenarios appeared to be an order of magnitude lower than the budget costs. However, extra social costs (budget costs + externality costs) could become null if households experience the source separation of organics as a benefit, corresponding to a value of -1.2 €·$hour^{-1}$.

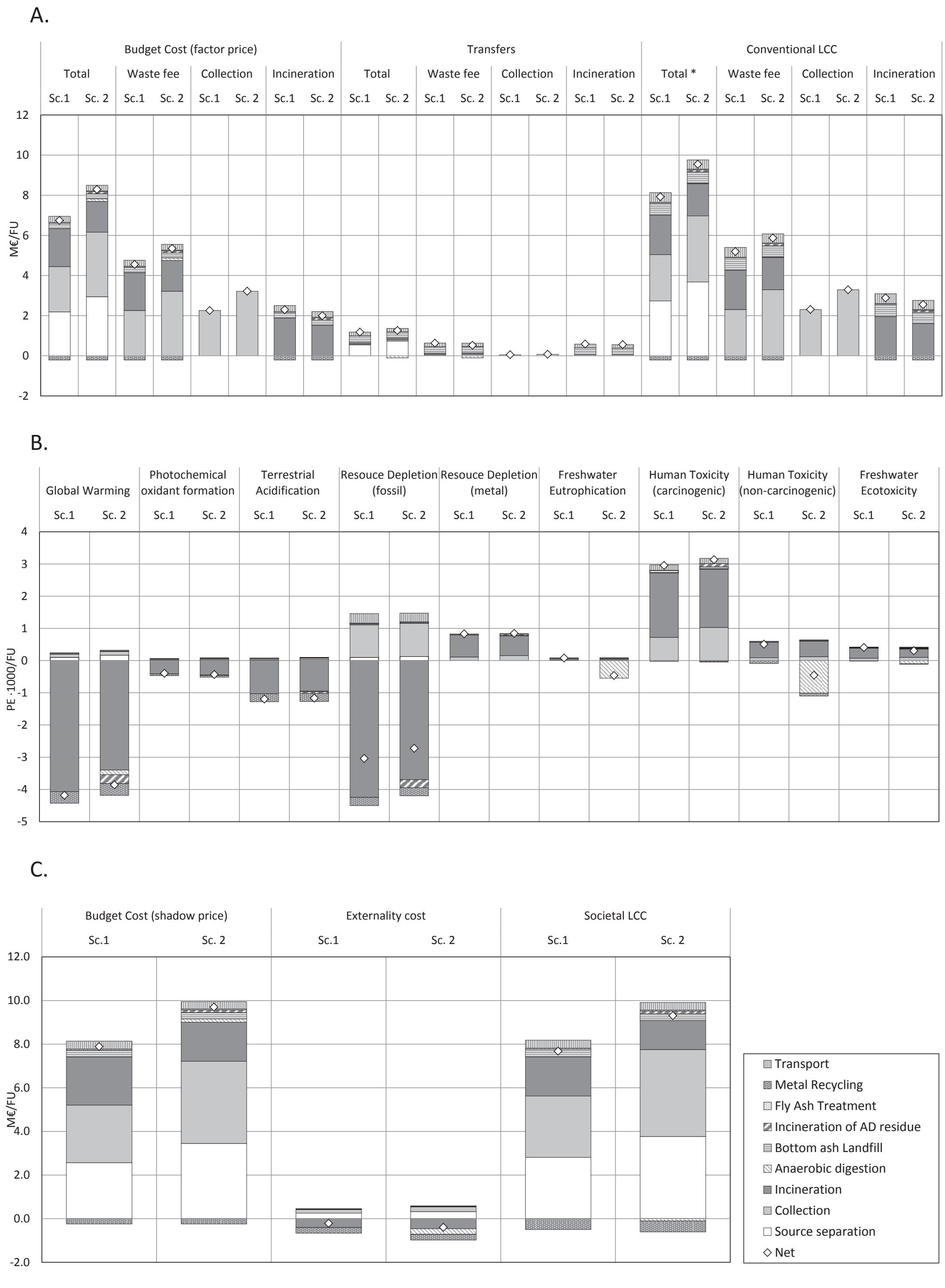

Figure 4: Results of the case studies from Paper (**I**). A) Conventional LCCs in M€ per functional unit (FU), B) the environmental part (LCA) of the Environmental LCC in person equivalent per functional unit (PE·1000/FU) - the economic part of the Environmental LCC is identical to the “Total” in the Conventional LCC. C) Societal LCC in M€ per functional unit (M€/FU).

4.2 Danish Food Waste Management

4.2.1 Modelling aspects

In Paper (**II**), the food waste hierarchy was assessed using Accounting Environmental and Societal LCCs. The functional unit of the study was the management of the annual food waste generated in Denmark, including 1,500,000 single-family houses (SFHs) and 1,000,000 multi-family houses (MFHs). The annual food waste generated per household was 210 kg in SFHs and 143 kg in MFHs, in which three-quarters was vegetable food waste (VFW) and one-quarter was animal food waste (AFW). Edible food waste represented one-half of the VFW and three-quarters of the AFW in both types of housing (Edjabou et al., 2015; Miljoestyrelsen, 2014, 2012).

Figure 5 shows the four scenarios compared, namely:

- Scenario S-IN, in which mixed waste was incinerated in a CHP plant.
- Scenario S-CD, in which source-separated food waste was co-digested with manure and the non-segregated food waste was incinerated among the residual MSW.
- Scenario S-AF, in which VFW was source-separated and treated to become animal fodder and the AFW and non-segregated VFW were incinerated together with the residual MSW.
- Scenario S-PR, in which 100% of the edible food waste was prevented and the inedible food waste was incinerated together with the MSW.

Because scenarios differ as to the upstream impacts (namely, the production and purchase of those food commodities related to edible food waste), the "zero burden assumption" could not be applied in this case study. The zero-burden assumption is often applied in waste-LCAs when the alternatives being compared have equal upstream impacts (i.e., the production of the commodities leading to waste), and to simplify the study the production of such goods is excluded from the assessment (Cleary, 2010; Gentil, 2011; Gentil et al., 2011).

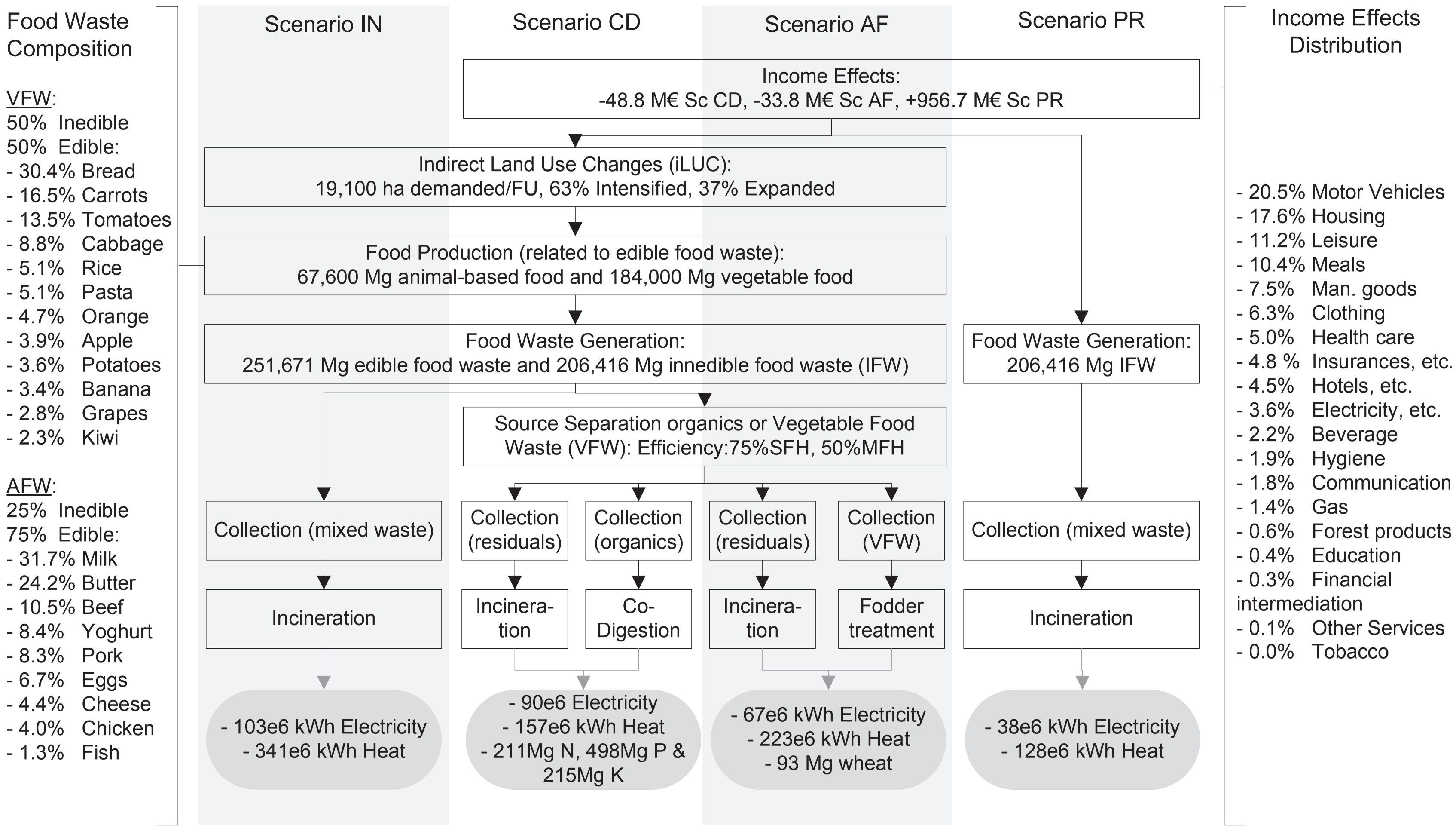

Figure 5: Illustration of the four scenarios assessed in the case study from Paper (**II**). The sketch shows the main mass and energy flows. The food waste composition includes the ratio of inedible and edible food waste for VFW and AFW. The edible food waste is disaggregated into the food commodities assumed.

To model the upstream activities related to food production it was necessary to identify the commodity composition of the edible food waste. This was not an easy task, since on the one hand waste characterization studies are not precise enough to link edible food waste to a food commodity, and on the other hand food consumption statistics do not provide knowledge of the wastage of each commodity. Thus, some estimations and assumptions were done using both data sources: i) statistics data on household food consumption (Statistics Denmark, 2015a, 2015b), and ii) food waste characterization studies (Edjabou et al., 2015; Miljoestyrelsen, 2014, 2012).

Special attention was given to two indirect effects: i) income effects related to the cost difference between the scenarios, and ii) indirect land use changes (iLUC) associated with the additional production of food related to edible food waste (in S-IN, S-CD and S-AF) and avoided fodder production (in S-AF).

To calculate income effects, the expenses of the households in S-IN were taken as a baseline, while any net change in the total expenses of households in the remaining scenarios was assumed to affect other forms of consumption (income effects). Income effects were modelled using statistics data on Danish households' consumption (Statistics Denmark, 2015a) to identify the marginal consumption by Danish households, i.e., how an extra unit of disposable income is used by an average Dane (explained below), and the input-output data from Ecoinvent Version 3 on the emissions associated with each item of such marginal consumption, as done by Thiesen et al. (2006).

The inclusion of income effects:

- Equalized the financial costs of all the scenarios to the value of S-IN (the economic part of the Environmental LCC).
- Caused environmental impacts associated with the production of the goods/services included in the marginal consumption (the environmental part of the Environmental LCC).
- Caused welfare gains on consumption in the Societal LCC, since resources were used more efficiently (budget costs) and welfare losses related to the emissions during production of the goods/services present in the marginal consumption (externality costs).

Figure 6 illustrates the calculation approach used to estimate the iLUC related to edible food waste (as well as the avoided fodder production associated with S-AF). The overall idea was that marginal food demand in Denmark is supplied by the global food market through a combination of intensification of existing production (75%) and expansion of arable land (25%) (Tonini et al., 2015), generating environmental emissions, ecosystem losses and changes in the use of labour as well as real capital changes (Fargione et al., 2008; Searchinger, 2008, 2010). The inclusion of iLUC:

- Did not cause any financial net consequence, since it was assumed that the food price remained constant in all the scenarios and the income effects equalized the total expenses of all scenarios in line with S-IN expenses (the economic part of the Environmental LCC).
- Caused environmental impacts associated with intensification because of the use of N-fertilizer, P-fertilizer and K-fertilizer, and with expansion due to carbon and nitrogen losses related to deforestation (the environmental part of the Environmental LCC).
- Welfare losses related to the environmental consequences related to carbon loss from the expansion of arable land and emissions from intensification, using the emissions from the LCA and the accounting prices of emissions.
- Welfare losses related to the socio-economic value of ecosystem losses related to expansion. For this, we estimated the combined ecosystem services value (CEVua) of the ecosystem displaced with expansion using literature data from the Economics of Ecosystems and Biodiversity (TEEB) database (Foundation for Sustainable Development, 2010).
- Did not cause any net welfare effect related to labour and real capital changes since it was assumed that the employment of real capital and labour resources move in the same direction as land use, e.g., if the production of "product A" causes the conversion of rainforest to cropland, labour and real capital will move from the tourism sector (in the rainforest) to the agricultural sector.

The LCC modelling of the SWM part used the same model as Paper (**I**) and it was as mechanistic as possible, but empirical data were also used, e.g., in the recycling facilities' downstream incineration. In contrast, the mechanistic approach to modelling upstream activities was unfeasible due to the lack of data; thus mostly empirical data were used.

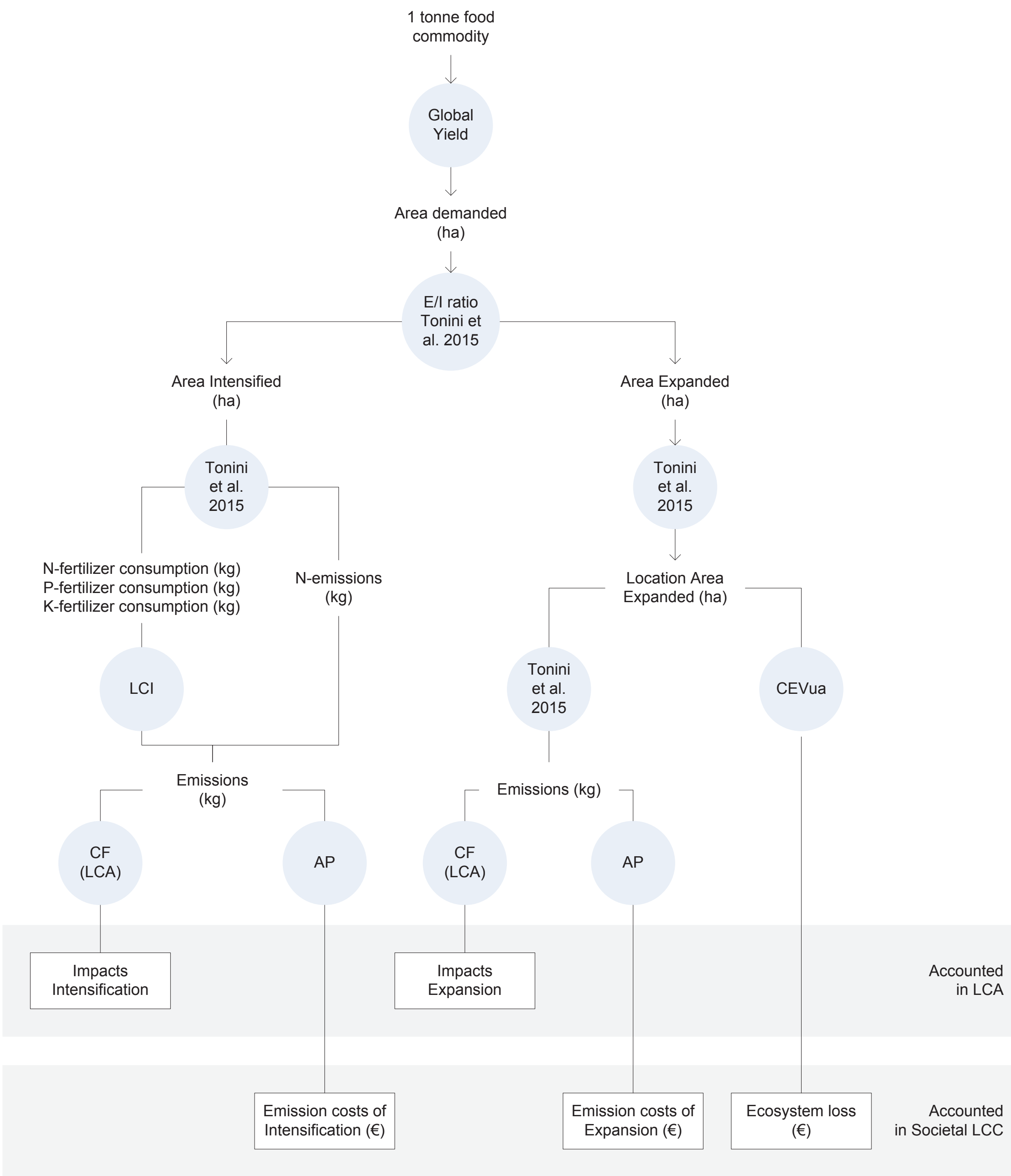

Figure 6: The iLUC calculation approach used in the investigation. E/I ratio = expansion/intensification ratio, LCI = Life Cycle Inventory, CEVua = combined ecosystem value per unit area, CF (LCA) = characterization factors of the emissions used in the LCA, AP = accounting prices of emissions.

4.2.2 Key results

The Environmental LCC revealed that although financial costs are equal between scenarios when including income effects, the economic winners and losers differ (Figure 7A). Most of the financial costs and environmental impacts were related to the food industry in the first three scenarios, while the industry related to income effects had the largest contribution for the fourth scenario (Figure 7A and 7B). However, the environmental impacts related to the income effects were highly sensitive to its composition, i.e., how savings from the prevented edible food waste were used.

The Societal LCC showed that only the prevention scenario showed welfare gains whereas the rest of the scenarios incurred welfare losses (the main contributions were again the food industry and relating to edible food waste - see Figure 7C). In all the scenarios, the externality costs were again one order of magnitude lower than the budget costs. This small magnitude may be due to the fact that only a few emissions have available accounting prices and that the accounting prices of the ecosystem are low, so iLUC was a minor element in the Societal LCC.

The results indicated that the inclusion of indirect effects - mainly income effects - can have a tremendous impact on the LCC outcomes when the alternatives being compared have different net costs associated. Based on these results, an important conclusion can be drawn: waste prevention is not by default environmentally preferable to other options, its environmental benefits are completely dependent on the use of the "money" saved by households. If alternative consumption patterns are more environmentally intensive than food production, prevention can be detrimental. To avoid this side effect, prevention campaigns should be accompanied by other economic policies aimed at reducing the impact of alternative consumption patterns.

In Societal LCC, the importance of income effects is minor due to the different magnitudes of welfare effects related to consumption (budget costs) and welfare effects related to environmental impacts (externality costs). However, this can change if the literature data on accounting prices are expanded and more emissions/effects can be included.

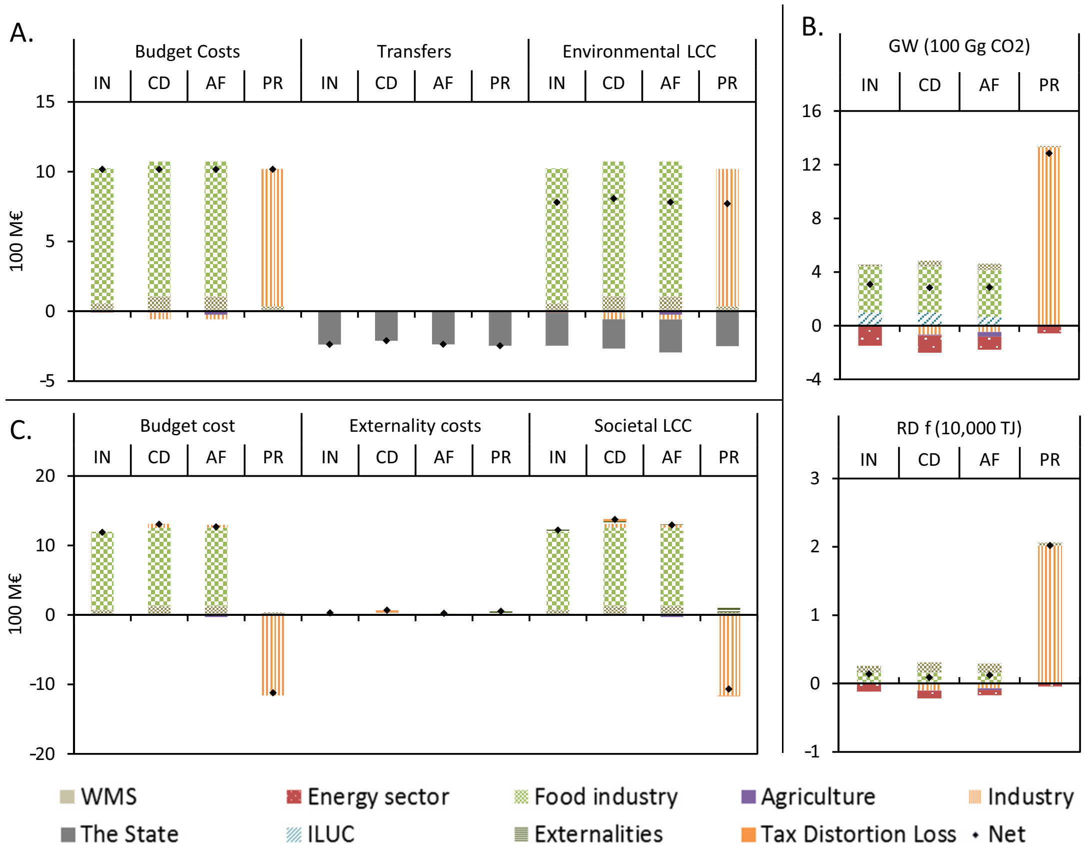

Figure 7: Results of the four scenarios assessed from Paper (**II**). A) The economic part of the Environmental LCC in 100 M€ per functional unit. B) The environmental part of the Environmental LCC in characterized impacts per functional unit. C) The Societal LCC of the four scenarios assessed in 100 M€ per functional unit. GW = global warming, RDf = fossil resource depletion.

4.3 Danish household waste incineration

4.3.1 Modelling aspects

In Paper (**III**), an Accounting Conventional LCC with a Planning approach was used to quantify the marginal and average post-design costs of WtE related to recycling or prevention strategies leading to waste diversion from WtE facilities.

The costs of WtE plants depend on the type and size of the facility as well as the waste input composition. The study included two technological options: one WtE facility that co-generates heat and power (CHP) with flue gas condensation and another that only generates power (Power). In both cases, it was assumed that there was an annual capacity of 300,000 Mg, a nominal lower calorific value of 11 GJ·Mg^{-1} and a typical Danish household waste composition of LHV of 11.2 GJ·Mg^{-1} (Edjabou et al., 2015).

The study included six potential changes in waste input in the WtE facility caused by recycling or prevention strategies that caused the diversion of fibre (i.e., paper and cardboard), glass, metals, plastics and organic waste, and residual waste (representing prevention affecting all the waste fractions in the same manner). Here, the diverted fractions are named "target fractions".

Post-design costs were estimated using the method illustrated in Figure 2, consisting of three stages: i) an initial stage, in which an expert's knowledge was used to define a costs model that describes the average costs of WtE as a function of key plant and waste parameters, ii) a second stage, in which the results of the model were evaluated by comparing them to literature data, and iii) a third stage in which marginal costs were estimated by performing several changes in the waste input composition (simulating the diversion of specific waste fractions) and using the costs model defined by "1)".

In the third stage, it was assumed that a WtE plant would exhibit three different responses to a change in waste composition: i) where biomass is added to keep constant the utilization of the thermal capacity (we assumed that the added biomass was woodchip with a water content of 41% with an LHV of 9.8 GJ Mg^{-1} and at a price of 68 € Mg^{-1}), ii) where RDF is imported, to keep the thermal capacity constant (we assumed that the imported RDF had the same composition as the RDF fraction of Evangelisti et al. (2015) with LHV 13.5 GJ Mg^{-1}, and the gate fee (revenue for the WtE facility) for this fraction was 44 € Mg^{-1}, and iii) no reaction occurs, the waste throughput drops and the capacity of the facility is not fully utilized.

4.3.2 Key results

The average costs of WtE appeared to be 26 € Mg^{-1} in the CHP case and 61 € Mg^{-1} in the Power case, and in both technological options the main contributions were due to the fixed costs (i.e., the amortization of the capital expenses, maintenance, labour and insurance costs) which were partly balanced with energy revenues. The rest of the cost items - flue gas cleaning, ash disposal and power consumption - appeared minor compared to the fixed costs and energy revenues.

The marginal costs of diversion were completely dependent on the response in the WtE plant:

- When biomass was added to compensate for the thermal losses in the waste (Figure 8A), the marginal costs of diversion were between 304 and 39 € Mg^{-1} for CHP and between 303 and 34 € Mg^{-1} for Power. Such costs were mainly due to the costs of biomass and the extra fixed costs per tonne of waste.
- When RDF was used as reaction fraction (Figure 8B), the marginal costs of diversion range from -2 to 300 € Mg^{-1} for CHP and from -2 and 294 € Mg^{-1} for Power. In this case, the extra fixed costs were partly/fully compensated by the revenues from the RDF (assumed 44 € Mg^{-1} RDF).
- When no reaction occurs in the WtE (Figure 8C), the marginal costs range from 40 to 303 € Mg^{-1} for CHP and from 35 to 296 € Mg^{-1} for Power. In this case, most of the costs relate to the lost energy generation, adding to the extra fixed cost per tonne of waste.

Although the average costs of incineration are highly affected by the energy prices, the marginal costs of diversion are not, unless there is no response to adjust operations according to the diversion. In contrast, both the marginal and average post-design costs appeared to be sensitive to the composition of the input waste, and the higher that the LHV of the waste was, the higher both costs were.

The results demonstrate that the marginal costs of "spare capacities" in WtE facilities (i.e., situations when the thermal capacity of an existing WtE facility is larger than the thermal input supplied by the waste) are much larger than the average costs of WtE. For example, the marginal cost of the diversion of one tonne of waste (with an average composition such as in the baseline case) causing "spare capacities" can be up to five times larger than the average costs of WtE when biomass is added as a reaction fraction in a CHP facility.

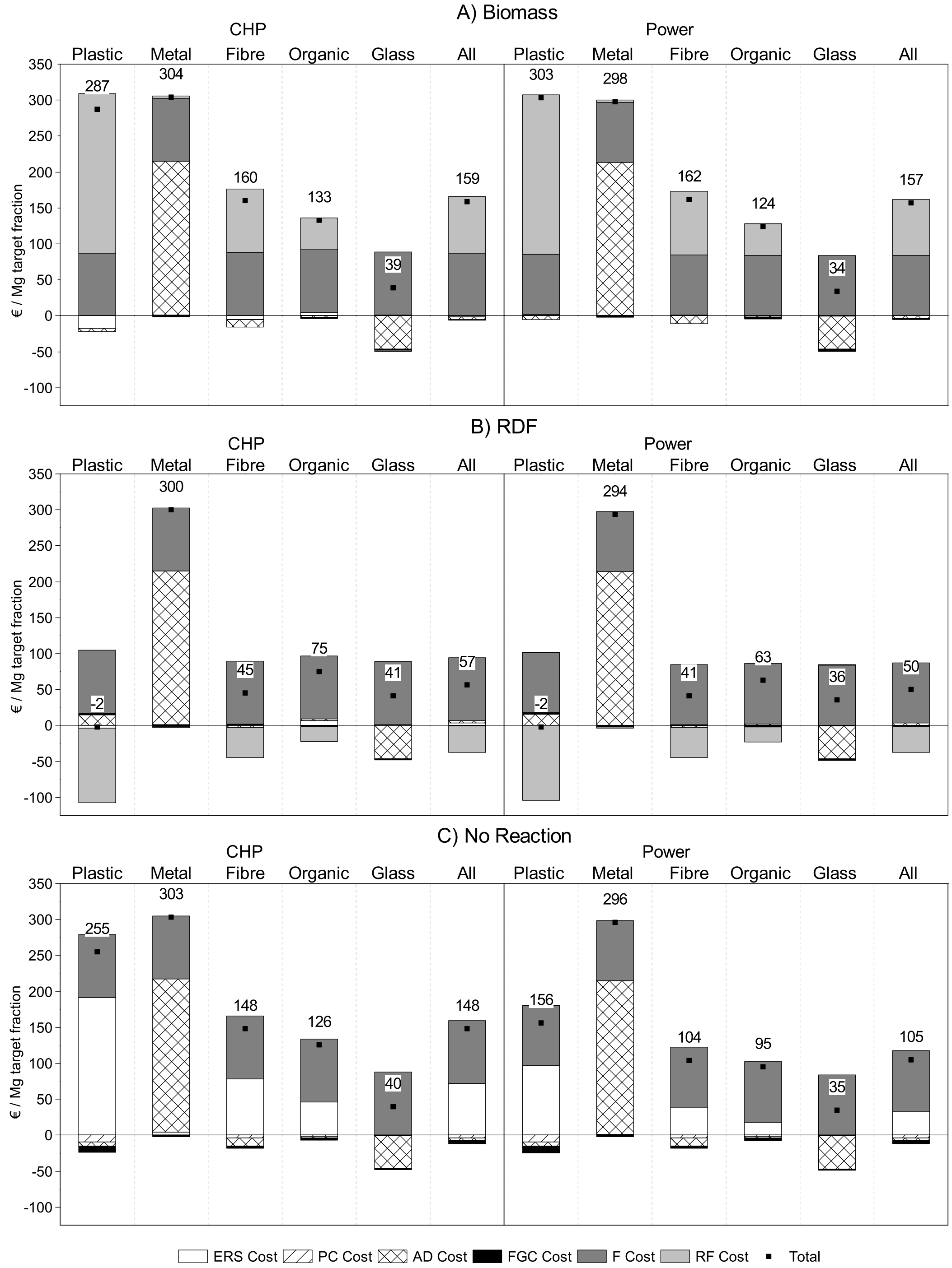

Figure 8: Marginal costs of diverting plastic, metals, fibres (paper and cardboard), organics, glass, and all waste fractions away from WtE in a CHP facility and in a Power facility in € per Mg of arget fraction (i.e. waste fraction being diverted). A) with biomass as reaction fraction, B) RDF as reaction fraction and C) no reaction from Paper (**III**).

4.4 U.S. household waste management

4.4.1 Modelling aspects

Paper (**IV**) aimed at evaluating the importance of including Societal LCC in optimization of SWM strategies. The study used a hypothetical U.S. county with an annual household waste generation of 320,000 Mg as a case study with the waste composition of 2015 from Levis et al. (2014).

SWOLF was endowed with the S-LCC functionality by adding the accounting prices of selected pollutants (described below) to the already existing calculation principles for budget costs (presented in Levis et al.[4] referred to as "engineering costs") and emissions inventory. The externality costs included the damage costs of relevant pollutants for SWM systems: GHG emissions, general air pollutants ($PM_{2.5}$, NO_X, SO_2, VOC, CO, NH_3, CO and PM_{10}) and toxic air emissions (Hg, Pb, Cd, Cr (VI), Ni, As and dioxins). While the physical emissions released from the SWM system were estimated by SWOLF, the accounting prices of the emissions stem from the literature data (Table 2).

The study used a long-term Planning approach in which existing investment/capacities in the hypothetical U.S. county were excluded, as the case study aimed at identifying the optimal solution in the long-term when society is not tied to any current investments.

The optimization followed a standard "constraint method" in which externality costs was the variable within the optimization function while budget cost was the constraint variable. The results of such optimization were: the waste flows to each waste facility, the sizes of the facilities, as well as budget costs, externality costs and GHG emissions of the strategy. The model included mass flow constraints related to minimum plant capacities and minimum plant throughputs, based on U.S. state-of-the-art waste plants. However, the model did not include potential economies of scale, i.e. a constant cost per waste fraction was assumed for each treatment/disposal facility.

Table 2: Accounting prices of the emissions used in the study in $\$\cdot kg^{-1}$

Emission	Accounting Price ($\$\cdot kg^{-1}$)	Reference
CO_2 (fossil)	0.04	(US EPA, 2015)
CH_4	0.82	
N_2O	11,62	
CO	0.87	(Matthews and Lave, 2000; Muller et al., 2011; Roth, 2004)
PM_{10}	2.95	
NH_3	25.5	(Fann et al., 2012; Muller et al., 2011)
VOC	1.5	
$PM_{2.5}$	341.8	
SO_2	64.47	
NO_x	40	(Fann et al., 2012; Mauzeral et al., 2004; Muller et al., 2011)
Hg	87067	(Rice and Hammitt, 2005)
Pb	458	(Grosse, 2002; Pizzol, 2010; Rabl et al., 2008)
Cd	115	(Rabl et al., 2008; Rice and Hammitt, 2005)
Cr	590	
Ni	11	
As	236	
Dioxins	5.50E+08	

4.4.2 Key results

Figure 9 shows the optimization results using the baseline assumptions. The extremes of the curves shown in Figure 9A represent the cases (solutions of the optimization) when minimizing externality costs without any budget constraint “min EC” and minimizing budget costs “min BC”. The cases in between these extremes were obtained using minimizing externality costs and imposing different budget constraints. The names given to these cases comprised a letter from A to K, representing the magnitude of the budget constraint, and the objective function in parenthesis – (EC) when minimizing externality.

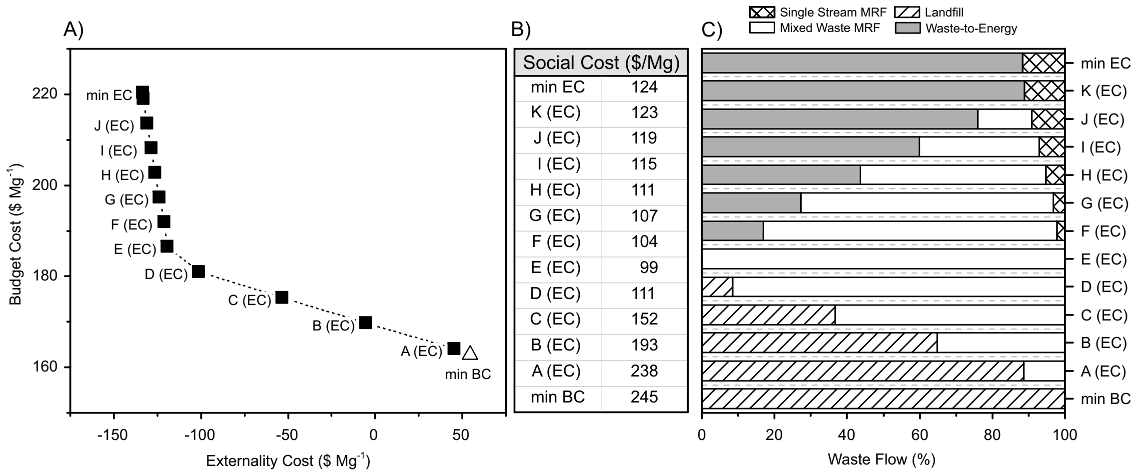

Figure 9: Optimization results represented by: A) a point (externality cost per Mg on X-axis and budget cost per Mg on Y-axis), B) social cost ($ per Mg) and C) a bar representing the waste flows of each solution after collection.

Moving from the minimal budget costs case "min BC" to the minimal externality costs case "min EC", there was a first set of cases in which mixed waste was routed either to landfill or mixed waste MRF (MW-MRF) (Figure 9C). Higher shares of MW-MRF over landfilling led to lower externality costs and larger budget costs. From case "F (EC)" to "J (EC)", there was a second set of cases in which waste was distributed between WtE, single-stream MRF (SS-MRF) and an MW-MRF. The higher share of WtE and SS-MRF over MW-MRF led to lower externality costs and larger budget costs. In the case involving minimal externality costs, "min EC", 12% of the waste was sent to SS-MRF and the rest to WtE. The streams sent to an SS-MRF included metals (ferrous and non-ferrous), PET, folding paper containers, paper bags and third-class mail paper as well as newsprint, all of which were recovered at their maximum participation rates. Other fractions, such as glass, plastic (except PET) and fibres (such as magazines, office paper and corrugated cardboard), remained with the residual waste. This can be explained by the fact that the recycling of such materials has net externality costs associated and they appeared larger than the externality costs of their incineration with energy recovery.

The technological shift from landfilling "min BC" towards the solution "min EC" was associated with increasing recycling and energy generation, which avoided externality costs related to SO_2, fossil CO_2 and other emissions from energy production, lower CH_4, NO_x and $PM_{2.5}$, direct emissions, partly balanced by higher emissions of dioxins from WtE, NH_3 and N_2O from anaerobic digestion and the utilization of digestate on land, as well as lower savings from C storage (i.e., C stored when using the digestate on land were lower than the C storage incurred when all the waste was landfilled).

The societal optimal solution of the case study, i.e. the case with the minimal social cost, appeared to be case "E (EC)" (Figure 9B). The case "min EC" had the largest externality benefits, but they came with the highest budget costs and ended up with a social cost of \$124 Mg^{-1}, slightly larger than the societal optimal solution. Conversely, the solution "min BC" had the lowest budget costs but very high externality costs that led to the overall highest social costs, namely \$245 Mg^{-1}. Figure 10 shows the optimization results as social costs versus GHG emissions per Mg of waste input. Generally, it can be concluded that reducing externality costs will typically decrease GHG emissions. However, this is not always the case as, e.g., externality costs can be reduced by switching from plastic recycling to combustion, but this may increase GHG emissions (shift from case "F (EC)" to "G (EC)"). Thus, when optimizing using only one criterion (i.e., minimizing budget costs, externality costs, GHG emissions or landfilling), there is a risk of ignoring strategies with potentially lower social costs.

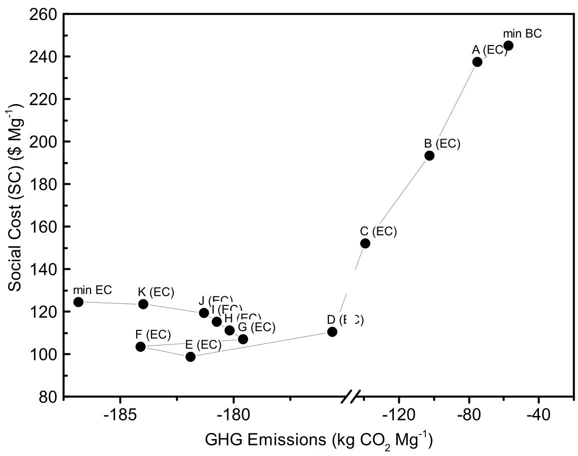

Figure 10: Optimization results shown as social costs (\$ Mg^{-1}) versus GHG emissions (kg CO_{2eq} Mg^{-1}).

Most of the externality costs were caused by SO_2, NO_x, $PM_{2.5}$, CH_4, fossil CO_2 and NH_3 emissions, while the remaining pollutants with available accounting prices had a minor impact. While considerable uncertainties are associated with the magnitude of externality costs, the results should be understood as indications of trends rather than as accurate estimates because: (1) a wide range of pollutants could not be included in the externality costs due to the absence of their accounting prices in the literature, (2) most accounting prices may represent minimum values as only certain costs were included in their estimations (e.g., human health costs), and (3) some types of externalities could not be included due to a lack of relevant data (e.g., a willingness-to-pay in the U.S. for time and space used for source-segregation in households), although the associated externality costs may be potentially important (Bruvoll et al., 2002).

The results were sensitive to the fuel mix used for electricity generation and waste composition. Future electricity and waste compositions may affect the optimization solutions. On one hand, future electricity offsets are likely to be less fossil-intensive than the current mix, thereby making externality savings less likely to compensate budget costs. This may cause a shift of the societal optimal solution towards solutions minimizing budget costs. On the other hand, future changes in waste composition may involve lower shares of recyclables (e.g., generation of paper waste is decreasing as food and yard waste are increasing), which may decrease the potential for environmental benefits (externality savings) from recycling and at the same time lead to more expensive recovery (higher budget costs).

5 Discussion and recommendations

This section discusses the main challenges when performing an LCC of SWM found in this PhD thesis and gives some recommendations on how to overcome them.

5.1 Dynamics of SWM

SWM management is commonly a service provided by local government and, over recent decades, this management is increasingly done by private and public companies. Regardless of which type of management reduces overall costs, it is clear that the nature of these organizations makes waste management a dynamic system in which waste operators will adjust their operation to respond to any changes in the system, such as any variations in waste generation, recycling activities or regulations.

For consistent and realistic Planning LCC, the dynamics of the system should be taken into account - this can only be done with mechanistic modelling since empirical costs data for future waste situations do not exist. To do this, there is an urgent need to understand better the mechanisms of the individual parts of the SWM system, e.g., waste facilities as well as the links between such parts. The development of this knowledge could help in the development of economically efficient SWM systems.

5.2 The integration of LCA into LCC

The integration of economic and environmental assessment of SWM can be achieved by applying either an Environmental LCC or a Societal LCC. While the Environmental LCCs use two indicators (i.e., a financial assessment for the economic aspects and an LCA for the environmental part), Societal LCCs use one indicator in which economic and environmental are merged. However, both LCCs have some limitations and some effects remain unassessed in the integration. While the Environmental LCC only includes the environmental effects accounted for in the LCA, leaving unassessed other externalities such as ecosystem losses or a household's time and space to source-separate waste, the Societal LCC only includes those externalities with available accounting prices. Overcoming such limitations, either by including other externalities in the Environmental LCC or by estimating more accounting prices of emissions/effects for the Societal LCC, both LCCs would have the same capabilities and could be used indistinctly.

Until these limitations are overcome, efforts should be made in sharing the knowledge of both areas, i.e., an environmental and an economic assessment. Examples of knowledge sharing might be that the development of accounting prices for Societal LCC are based on the same methods as the development of the characterization factors for the LCA. However, an additional step should be taken to convert any characterized impacts into monetary terms.

5.3 The importance of income effects

Income effects account for the fact that households' income is a scarce resource, and if the SWM alternatives being assessed do not change the net income of the waste generator, the distribution of this income will change. For example, if the waste fee of a household decreases by 10 € per year, these savings will be used for other purposes, such as the purchase of other goods and services. This extra consumption (due to the savings on the waste fees) has associated environmental impacts as well as welfare effects on consumption and environmental loads. By the same token, an increase in waste fees will lead to a decrease in other forms of consumption (due to the extra expenses spent on the waste fees) and the consequent avoided impact of their production. Thus, the estimation of income effects is crucial for Environmental and Societal LCC.

The estimation of the impacts associated with income effects is strongly challenged, mainly by the identification of marginal consumption but also by the data availability concerning the environmental impacts of such consumption. Several methods could be applied for identifying goods/services affected by a variation in the disposable income of households (Thiesen et al., 2006); however, most of them are significantly uncertain. Paper (**II**) assumed that the savings associated with preventing edible food waste would change the consumption pattern of the household towards the consumption pattern of the adjacent income group, i.e., money saved from the prevention of food is seen as extra income for the household, as done by (Thiesen et al. (2006). This approach assumes that the consumption patterns of each income group remain the same, but it could be that food prevention campaigns motivate households to change their behaviours, e.g., by consuming organic food instead of conventional food or by changing their diet. Hence, when including income effects into an SWM study, there is a need to assess the potential behavioural changes leading from the assessed change in the study.

In addition, once the marginal consumption is identified (by assuming a behavioural change), there is a need for a consistent environmental inventory to be associated, i.e., the inventories of the different products/services included in the marginal consumption should be based on the same principles (e.g., system boundaries) as the rest of the study.

5.4 The results aggregation level

The intended use of the LCC studies determines the representation of the results. Usually, the greater the focus there is on the case study, the greater the level of the aggregation that is applied. In contrast, when the LCC study aims at knowledge transfer, the results are shown (or should be shown) as disaggregated to facilitate the transferability to other case studies. Overall, a balance should be established between the level of detail needed to understand the results and the level of aggregation required to satisfy the needs of the project.

Furthermore, LCC studies include different aspects measured with the same unit ("money") and there is a tendency towards results aggregation. However, it should take into account that: 1) they represent different elements, 2) they have different uncertainties associated, and 3) the decision-maker may want to establish different priorities within them. Thus, LCC outcomes should be disaggregated into budget costs, externality costs and transfers as well as distinguishing between direct and indirect effects.

6 Conclusion

This thesis developed a methodology for the consistent and comprehensive economic assessment of SWM with different levels of LCA integration. The method distinguishes between two modelling approaches (Accounting and optimization), three cost approaches (Conventional, Environmental and Societal LCCs), and two goal perspectives (Planning and Analysis).

By applying the developed method to four case studies, several conclusions can be drawn:

- Paper (**I**): Organic source-segregation incurs financial and social costs mainly related to the cost of the bags and bins used by households, as well as extra collection costs related to the additional collection scheme for organics as well as the extra cost of residual waste (compared to mixed waste) due to the extra collection points needed to fill up the truck (since less waste is collected with each stop).
- Paper (**II**): Food waste prevention is not by default the optimal solution for SWM due to the income effects stemming from the savings generated. The benefits/loads related to prevention completely depend on the use of the savings generated. If alternative consumption patterns are more environmentally intensive than food production, prevention can be detrimental. By the same token, if alternative consumption patterns are less environmentally intensive than food production, prevention would be beneficial. Hence, LCC studies including prevention strategies need to evaluate the behavioural changes associated with prevention, since they determine the marginal consumption and the environmental aspects of the income effects.
- Paper (**III**): The marginal costs of waste diversion strategies in existing WtE facilities depend completely on the response at such facility. However, regardless of the response type, it was demonstrated that the marginal costs of diversion are several times those of the average costs. Hence, the study does not underpin the common assumption that existing facilities are not significantly affected by marginal changes occurring in the SWM system.

- Paper (**IV**): The optimization of SWM using Societal LCC demonstrated that the socially optimal solution results from balancing budget and externality costs. In contrast, optimization using either budget costs or externality costs lead to socially suboptimal solutions. SWM systems that incur minimal budget costs (e.g., landfilling) have high externality costs associated with them, while SWM systems with minimal externality costs (e.g., WtE and recycling) come with high budget costs.

The key aspects when performing the LCC of SWM systems are: 1) a clear goal definition and cost principles, 2) the inclusion of direct and indirect effects, and 3) accounting for the dynamics and relations between the SWM facilities.

- The goal of each individual study should determine which approach to use. The definition of such a goal should take into account: i) the agent perspective (i.e., whose costs are of interest), ii) the audience of the study and the intended use of the results, and iii) the goal perspective of the study (i.e., Planning or Analysis perspective).
- For an LCC to be complete, both the direct and indirect effects of SWM should be included. This is especially critical when the alternatives being compared have significant cost differences, since this would affect other consumption patterns outside the SWM, implying environmental consequences. The same applies for complete LCA to account for the fact that income is scarce, and the main change within SWM alternatives would be how waste generators distribute their income.
- For an LCC to be realistic and able to represent reality, effort should be allocated to understanding the operations of waste facilities. This is crucial in Planning LCCs in order to account for the dynamics of the gate fees.

The greatest challenge when performing LCC of SWM strategies is the lack of available data confronting the LCC practitioner. This can only be solved with:

- Close cooperation with the operators of the waste facilities as well as the waste authorities.
- Transparent disclosure of studies to enhance knowledge transfer between them.

7 Further work

According to the findings of this thesis, further research should focus on:

- The estimation of **willingness-to-pay** values for important externalities related to SWM systems, such as the **time and space used by households** to source-separate waste. As demonstrated by Paper (**I**), such values can have a significant effect when comparing waste strategies requiring different efforts on the part of households using Societal LCC.
- The development of **accounting prices for relevant pollutants of SWM** to be able to include their externality costs in Societal LCC. As demonstrated by Paper (**IV**), nitrogen-based emissions into water, zinc emissions into the air and water, and barium emissions into water, have important effects for LCA that can be included in Societal LCC due to the absence of their accounting prices.
- The further development of **mechanistic costs models** for different waste facilities/technologies to enhance the understanding of the responses in existing facilities to adjust to different changes in the SWM system as done in Paper (**I**) with collection costs and Paper (**III**) for WtE facilities.
- Establishing an **economic database** with robust data based on full-scale waste facilities with the crucial involvement of all the waste stakeholders - from waste companies to waste authorities - to be used for LCC practitioners and to help decision-makers.
- The estimation of **economies of scale** in waste facilities and their inclusion in holistic assessments of SWM systems. The costs of SWM systems depend on the size of their facilities – generally, the larger the facility the lower the cost per tonne of waste. However, larger facility capacities lead to higher collection costs and a risk of overcapacity. For sustainable SWM systems, economies of scale should be taken into account, but not as a sole criterion.

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9 Papers

I Martinez-Sanchez, V., Kromann, M. A., Astrup T.F. Life cycle costing of waste management systems: Overview, calculation principles and case studies. Waste Management 2015, 36, 343-355.

II Martinez-Sanchez, V., Tonini, D., Møller, F., Astrup, T.F. Life Cycle Costing of Food Waste Management in Denmark: Importance of indirect effects. Submitted to Environmental Science and Technology.

III Martinez-Sanchez, V., Hulgaard, T., Hindsgaul, C., Riber, C., Kamuk, B., Astrup, T.F. Estimation of Marginal Costs at Existing Waste treatment Facilities. Submitted to Waste Management.

IV Martinez-Sanchez, V., Levis, J.W., Ranjithan, R., DeCarlis, J.F., Barlaz M.A., Damgaard, A., Astrup T.F. Evaluation of Externality Costs in Life Cycle Optimization of Solid Waste Management. To be submitted to Environmental Science and Technology.

In this online version of the thesis, **paper I-IV** are not included but can be obtained from electronic article databases e.g. via www.orbit.dtu.dk or on request from.

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The Department of Environmental Engineering (DTU Environment) conducts science-based engineering research within four sections:
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The department dates back to 1865, when Ludvig August Colding, the founder of the department, gave the first lecture on sanitary engineering as response to the cholera epidemics in Copenhagen in the late 1800s.

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