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Life cycle costing of waste management systems: Overview, calculation principles and case studies

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ABSTRACT

This paper provides a detailed and comprehensive cost model for the economic assessment of solid waste management systems. The model was based on the principles of Life Cycle Costing (LCC) and followed a bottom-up calculation approach providing detailed cost items for all key technologies within modern waste systems. All technologies were defined per tonne of waste input, and each cost item within a technology was characterised by both a technical and an economic parameter (for example amount and cost of fuel related to waste collection), to ensure transparency, applicability and reproducibility. Cost items were classified as: (1) budget costs, (2) transfers (for example taxes, subsidies and fees) and (3) externality costs (for example damage or abatement costs related to emissions and disamenities). Technology costs were obtained as the sum of all cost items (of the same type) within a specific technology, while scenario costs were the sum of all technologies involved in a scenario. The cost model allows for the completion of three types of LCC: a Conventional LCC, for the assessment of financial costs, an Environmental LCC, for the assessment of financial costs whose results are complemented by a Life Cycle Assessment (LCA) for the same system, and a Societal LCC, for socio-economic assessments. Conventional and Environmental LCCs includes budget costs and transfers, while Societal LCCs includes budget and externality costs. Critical aspects were found in the existing literature regarding the cost assessment of waste management, namely system boundary equivalency, accounting for temporally distributed emissions and impacts, inclusions of transfers, the internalisation of environmental impacts and the coverage of shadow prices, and there was also significant confusion regarding terminology. The presented cost model was implemented in two case study scenarios assessing the costs involved in the source segregation of organic waste from 100,000 Danish households and the subsequent co-digestion of organic waste with animal manure. Overall, source segregation resulted in higher financial costs than the alternative of incinerating the organic waste with the residual waste: 1.6 M€/year, of which 0.9 M€/year was costs for extra bins and bags used by the households, 1.0 M€/year for extra collections and −0.3 M€/year saved on incineration.

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1. Introduction

Over the past decade, increasingly rigorous and systematic documentation of societal consequences related to solid waste management has been required by authorities, technology developers and other stakeholders. This has placed increasing emphasis on the holistic assessment of waste management, in particular on environmental impacts. Meanwhile, the Life Cycle Assessment (LCA) of waste management systems has matured significantly (Laurent et al., 2014a,b; Finnveden et al., 2009), and it is now regularly accepted as a useful source of support for overall decision-making in many countries (Carlsson Reich, 2005). While waste LCA provides a systematic framework for accounting for environmental impacts associated with waste management, most decisions related to the real-life implementation of waste technologies in modern societies are affected by economic constraints. For decision-makers, the lack of a balanced economic assessment alongside traditional LCA results therefore limits the value of the LCA itself, as economic priorities are then de-coupled from environmental aspects.

The economic characteristics of waste management have been addressed in the literature, related either to specific waste management technologies (for example Vinyes et al. 2012; Teerioja et al. 2012; De Feo and Malvano, 2012; Coelho and De Brito,

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2013) or to overall waste systems (for example Ricci, 2003; Larsen et al. 2010).

Regarding collection costs, Teerioja et al. (2012) applied a social life-cycle cost analysis, determining that the pneumatic collection system in their study was six times more expensive than a traditional door-to-door waste collection system for a specific area (0.2 km^2 with 20,000 citizens/km^2 and 2000 tonnes of MSW per year). In addition, Groot et al. (2013) developed a comprehensive cost model (including financial and carbon costs) to demonstrate that: (1) the source separation of plastic packaging waste (PPW) was over two times more expensive than post-separation and (2) for source separation options, curbside collection was 2.5 times more expensive than drop-off. Eriksson et al. (2005) assessed the welfare economics of different waste systems applied to easily degradable waste (EDW), plastic and paper. They found that incineration was better than composting and anaerobic digestion for EDW, and it was comparable to recycling for plastic and paper. While these studies naturally reach a variety of conclusions based on differences in framework conditions, very few of them include (1) details of cost calculation principles for the involved waste technologies, as in Groot et al. (2013), (2) details on assessment focus, definitions of system boundaries and assumptions, as in Carlsson Reich (2001) and Eriksson et al. (2005), or (3) clear, transparent terminology for describing assessment principles (for example Vigsø, 2004; Carlsson Reich, 2001; Eriksson et al., 2005). This clearly not only limits the transparency of these studies and the subsequent applicability of the results, but it also illustrates that the economic assessment of waste management systems is a relatively under-developed field.

The economic assessment of waste management systems and technologies involves three context-specific challenges: (1) which type of costs should be assessed (for example private or social costs), (2) for whom should these costs be assessed (for example facility operators, households, public entities or entire systems) and (3) which cost calculation principles should be applied for the individual waste technologies included in a system? Traditionally, private costs (expenses in real money flows incurred by any stakeholder, also called internal costs) are addressed in financial assessments, while social costs (i.e. the sum of private and externality costs) are included in socio-economic assessments (Nordic Council of Ministers, 2007). Waste management systems involve stakeholders with significantly different interests: (1) waste generators (for example households), (2) waste facility operators and (3) waste authorities. The financial costs of a waste management service are often paid by the waste generators (either by waste fees or through taxes), and waste operators are typically involved only in selected parts of the management chain and may consider only costs associated with relevant facilities. On the other hand, authorities, such as local governments, may be interested mainly in the socio-economic aspects of the waste management system. Existing cost assessments of waste systems in the literature offer a wide range of stakeholder's foci and associated cost calculation principles but provide limited guidance on how to assess systematically economic aspects of complex multi-stakeholder waste systems and at the same time relate these findings to LCA results.

Very few examples of combined economic and environmental assessments exist in the literature. Typically, economic assessments are carried out separately from the LCA, most often employing different system boundaries and assumptions (Hunkeler et al., 2008; Swarr et al., 2011; Norris, 2001; Carlsson Reich, 2005), while integrating economic and environmental aspects of waste management within a single assessment has been discussed only in a few cases (for example Carlsson Reich, 2005; Dahlbo et al., 2007). While a variety of approaches to cost assessment have been proposed in the literature (for example Economic Assessment, Financial Assessment, Total Cost Assessment and Cost Benefit Analysis), Life Cycle Costing (LCC) has been suggested as a consistent framework for combining LCAs and economic assessments, involving three types of LCC assessments (Hunkeler et al., 2008): Conventional, Environmental and Societal. A Conventional LCC represents traditional financial assessments (i.e. accounting for marketed goods and services) carried out typically by individual companies focusing on their ''own'' costs. The Environmental LCC[1] expands the Conventional LCC, in order to be consistent with the system boundaries of the LCA. This is also a financial assessment, albeit costs incurred by all the affected stakeholders are included. The Societal LCC further includes externality costs (i.e. it ''internalises'' environmental and social impacts by assigning monetary values to the respective effects), by using accounting prices. Societal LCCs may also be characterised as ''socio-economic'' or ''welfare-economic'' assessments. The three types of LCC thereby offer an overall framework for systematic economic assessments either in combination with LCAs or as stand-alone indicator.

Based on LCC principles and terminology, this paper aims at providing a consistent and comprehensive framework for the economic assessment of waste management systems. This is achieved by (1) developing systematic cost models for all main activities related to waste management (for example source segregation, collection, treatment and final disposal) based on transparent technical parameters associated with the involved technologies, (2) implementing the cost model framework on two selected case study examples illustrating the management of household waste and (3) on this basis, evaluating applicability and identifying critical methodological aspects related to LCC on waste management systems.

2. Methodology

2.1. Terminology

The naming principles introduced by Hunkeler et al. (2008) and Swarr et al. (2011) were applied in this study. Overall, costs can be distinguished between ''internal'' and ''external,'' whereby internal costs are monetary costs occurring both inside and outside the waste management system, while external costs (also termed ''externality'' costs) occur outside the economic system (also called ''non-marketed goods/services'' because they have no direct monetary value in the market). Internal costs can be measured either in market prices or in factor prices, the latter are market prices excluding transfers (taxes, subsidies, fees and duties used to distribute income between different agents in society, but which do not represent any resource reallocation) (Nordic Council of Ministers, 2007). The sum of internal costs and external costs represents social costs, here defined as society's costs for managing waste (Porter, 2002). The cost model differentiates between three types of costs: (1) budget costs, (2) transfers and (3) externality costs. Budget costs and transfers are characterised as internal costs, while externality costs, as the name suggests, are external. Budget costs are included in all three LCC types, transfers only in Conventional and Environmental LCCs and externality costs only in Societal LCCs. Table 1 provides an overview of cost types related to solid waste management.

Budget costs are incurred by waste agents, for example households, as waste generators or technologies/facilities operating within the waste system. Budget costs can be either ''one-off'' occurring once in the lifetime of a technology (for example capital investment or back-end costs), or recurring (for example

[1] The name ''Environmental LCC'' is used to emphasize that this type of assessment is intended to be consistent with an environmental assessment, i.e. LCA.

Table 1
Overview of costs incurred by waste agents (e.g. waste generators and waste management operators) and all members of society (waste generators, waste management operators and others) with regards to waste systems. Costs are classified into: (1) internal and external costs (and Social costs as sum of internal and external costs), and (2) budget costs, externality costs and transfers.

	Internal costs	External costs	Social costs
Incurred by	Waste agents (e.g. waste generator and operators)	All the members of society	Society
Budget cost	– Bags – Bins – Capital goods – Materials and energy consumption – Labour costs – Material and energy sales		Sum of internal costs (excluding transfers) and external costs for society (i.e. waste generator, waste operator and other agents)
Externalities cost		– Time consumption to source separate – Health issues – Disamenities – Working environment issues	
Transfers	– Fees – Taxes – Pecuniary externalities*		Not applicable

* Explained in the text (Section 2.1) and defined in the glossary.

operational and maintenance costs). In Conventional and Environmental LCCs, budget costs are accounted for in factor prices (market prices excluding transfers), whereas budget costs in Societal LCCs are accounted for in accounting prices (also called “shadow prices” or “opportunity” costs and representing the willingness to pay for a good or service). To translate factor prices into accounting prices different methods can be applied such as the Net Tax Factor (NTF) proposed by the Danish Ministry of Finances. This factor can be used in the same way as applied by Vigsø (2004) and Møller and Martinsen (2014).

Transfers are monetary flows that only represent income redistribution between stakeholders while not leading to re-allocation of resources such as land and labour or welfare change in society (Møller and Martinsen, 2014), for example environmental taxes and subsidies, or general taxes such as Value Added Tax (VAT).

Other types of transfers, sometimes referred to as “pecuniary externalities” may be related to energy and material recovery within the waste system. These transfers represent financial losses occurring when existing facilities or industries outside the system boundary of the assessment have to operate below their design capacity as a result of the additional supply of energy and/or material resources offered by the waste system. Although local changes in resource recovery from waste may generally be assumed to have a marginal influence on global primary production (for example of plastic), and pecuniary externalities can therefore be neglected, effects on more localised markets (for example heat in a local district heating network) may require attention. For example, if heat production from waste incineration increases, other marginal heat producers in the same network must reduce their production correspondingly (heat production from waste incineration has priority over other producers in Denmark) (Fruergaard et al., 2010). Reducing heat production would result in lower variable costs as well as lower revenues at the off-set facility, but all fixed costs would remain the same. Overall, this could potentially result in higher costs for heat consumers. These costs are considered transfers, since they are related neither to resource re-allocation nor to welfare changes, provided that consumers do not change their heat demands significantly (which is likely to be the case, since heat demand is fairly in-elastic in Northern Europe).

Externality costs represent effects on the welfare of individuals caused by activities which are not otherwise compensated. Externalities can be environmental, i.e. relate to the emissions in the LCA, or non-environmental in the form of noise or time spent by the households on waste sorting. Eshet et al. (2006) and Rabl et al. (2008) provided an overview of externality costs in waste management, including valuation techniques. While these values are applied here, a full description of valuation techniques is beyond the scope of this study. Any externality priced by an authority and covered by a stakeholder (also termed an “agent”) within the waste system becomes a transfer, i.e. an internal cost in the waste system, such as environmental taxes in the form of air emission taxes. Typical externalities relevant to waste systems are emissions into air, water and soil which affect human health, disturb natural environments and cause climate problems as well as disamenity impacts (for example nuisance, noise and congestion) caused by waste facilities and transportation. Other externalities, such as time spent and space used by households to sort their waste, are often excluded from the assessments due to the uncertainty in quantifying their value (Vigsø, 2004).

2.2. Assessment goals

LCCs may be applied from either a “planning” or an “analysis” perspective. Planning LCCs evaluate the economic performance of a system in response to a change in the system, while Analysis LCCs evaluate the economic performance of a system in its current state. In both cases, overall costs with respect to the delivery of a specific functional unit are evaluated. Each of the three LCC types supports different goals. A Conventional LCC is commonly used when environmental aspects are not in focus, in order to (1) assess the economic feasibility/viability of treatment solutions (for example Coelho and De Brito, 2013; Franchetti, 2009), (2) identify the economically best-performing solution (for example Karagiannidis et al., 2013; Groot et al., 2013) and (3) evaluate the economic consequences of implementing a specific waste solution (for example Gomes et al., 2008).

An Environmental LCC is typically intended to supplement an LCA with an economic performance assessment (for example Consonni et al. 2005). When all stakeholders affected by the assessed waste scenario are included in the cost assessment, either with a Conventional or an Environmental LCC, the results not only show net cost/savings but also the distribution of costs between stakeholders, i.e. which stakeholder incurs higher or lower costs. Such a result may be used potentially to evaluate needs for financial compensation between stakeholders. A Societal LCC is often used to examine the economic efficiency of specific scenarios on a societal level (Nordic Council of Ministers, 2007), in order to estimate welfare losses and gains related to re-allocating resources (Møller et al., 2014).

The system boundaries of the LCC naturally depend on the study in question, but their definition should correspond closely with those of the LCA. It should, however, be noted that Conventional LCCs may often exclude specific parts of the lifecycle, thereby reflecting the specific goal of the Conventional LCC. For example, determining a waste fee based on a Conventional LCC should exclude source separation costs incurred by households (because the goal is to determine costs downstream of the households). However, Environmental and Societal LCCs must include all of the phases of the system and thereby have system boundaries identical to the LCA.

2.3. Cost model: Structure

The proposed cost model applied a Unit Cost Method (UCM) approach, following the principles of Parthan et al. (2012). First, the waste system was divided into activities or waste stages such as source separation, collection, transportation, treatment and disposal. Then each activity was disaggregated into relevant cost items such as machinery, salaries, fuel and maintenance costs. Each cost item was classified as budget cost, transfer or externality cost.

For each cost item related to each activity (for example fuel consumption involved in the activity of collection), two characteristic parameters were defined: a physical and an economic parameter. The physical parameters described the quantity of a cost item needed to collect/treat/dispose of one tonne of waste (for example 1 l of diesel to collect 1 tonne of waste), while the economic parameter represented the unit cost of the specific cost item (0.1 €/l diesel). The unit cost of the individual cost item for the specific technology was then found by multiplying the two parameters (for example 0.1 € of diesel consumed to collect one tonne of waste). Within each activity, there were three unit activity costs (i.e. costs per tonne of waste input): (1) unit budget costs, (2) unit transfers and (3) unit externality costs. Each of these unit activity costs was then calculated by summing the unit costs for all individual cost items of the same cost type (i.e. unit budget costs, unit transfers and unit externality costs). Once all activities/technologies were defined per tonne of waste input into the specific technology, scenarios were built by linking the individual technologies with the appropriate mass and energy balances.

The LCCs of the waste system were obtained as the sum of the costs associated with all activities included in a scenario. The Conventional LCC included the sum of the budget costs and transfers for n activities involved in the scenario, as shown in Eq. (1). The budget cost of each activity i results from multiplying the unit budget cost of activity i (UBC_i) by the amount of waste input into the same activity (W_i). The transfer of activity i resulted from multiplying the unit transfer of activity i (UT_i) and the waste input amount into each activity i (W_i).

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

The Environmental LCC extended the Conventional LCC by adding transfers anticipated to be established in the near future, i.e. externalities expected to be internalized in monetary terms in a time perspective relevant for the decision being assessed. The anticipated transfer of each activity resulted 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 Eq. (2). The economic results of the Environmental LCC are complemented by an LCA for the same system. Special attention should be given to avoid double-counting of emissions, i.e. once they are internalized in the economic part, they should not be accounted in the environmental part (LCA).

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

The Societal LCC included budget costs and externality costs, both accounted for in shadow prices. The unit budget costs of activity i in factor prices (UBC_i) were multiplied by the Net Tax Factor (NTF). In Denmark, the NTF for converting factor prices to shadow prices of marketed goods is 1.17 (Miljøministeriet, 2010). The externality costs of activity i resulted from multiplying the unit externality cost of activity i (UEC_i) by the waste input amount into each activity i (W_i). The shadow prices of the marketed goods were added to the externalities cost (already in shadow prices), as in Eq. (3).

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

The Environmental and Societal LCCs were related directly to the inventory of the LCA by applying the same physical parameters as in the LCA. Fig. 1 describes the structure of the cost model, and Table 2 summarises the characteristics of each type of LCC and examples found in the literature.

2.4. Cost Model: Cost calculations

The two types of parameters associated with each cost item, i.e. physical and economic parameters, are described in the following sections.

2.4.1. Budget cost

2.4.1.1. Capital and back-end costs. "One-off" costs were allocated equally between all tonnes of waste collected/treated/disposed by a specific technology during the economic lifetime of the technology (Gluch and Baumann, 2004; Woodward, 1997). This was achieved by converting lump-sum amounts into annuities (A), in Eq. (4) when lump-sums were in the present value (P) or in Eq. (5) when lump-sums were in a future value (F), and then by dividing annuities by the annual usage rate of the technology. Indices (n) and (ir) represented the economic lifetime of the technology/piece of equipment and interest rates, respectively.

$$A = \frac{P}{\left[\frac{(1+ir)^n - 1}{ir(1+ir)^n}\right]} \tag{4}$$

$$A = \frac{F}{\left[\frac{(1+ir)^n - 1}{ir}\right]} \tag{5}$$

The annual usage rate of a technology may either be equal to the annual capacity of said technology or a fraction of the annual capacity if a facility operates below its design capacity as a consequence of the waste scenario. Determining the annual usage rate depends on the specific technology in use (see details provided in the Supplementary materials); for example, the thermal capacity of an incinerator limits the amount of waste to be treated. Thus, the annual usage rate of an incinerator (AUR_{WtE}) [tonne/year] in a specific waste scenario is inversely proportional to the calorific value of the waste, i.e. with increasing Lower Heating Values (LHV) [MJ/tonne], less waste can be incinerated. The Annual Mass Capacity (AMC) [tonne/year] of a plant was multiplied by the ratio Design Heating Value (DHV) [MJ/tonne] over the Lower Heating Value of the waste ($LHVw$) [MJ/tonne], in order to adjust the annual tonnage, as shown in Eq. (6).

$$AUR_{WtE} = AMC * \frac{DHV}{LHVw} \tag{6}$$

2.4.1.2. Operational and maintenance costs. Operational and maintenance costs can be either fixed, for example labour, maintenance and insurance, or variable, for example electricity consumption.

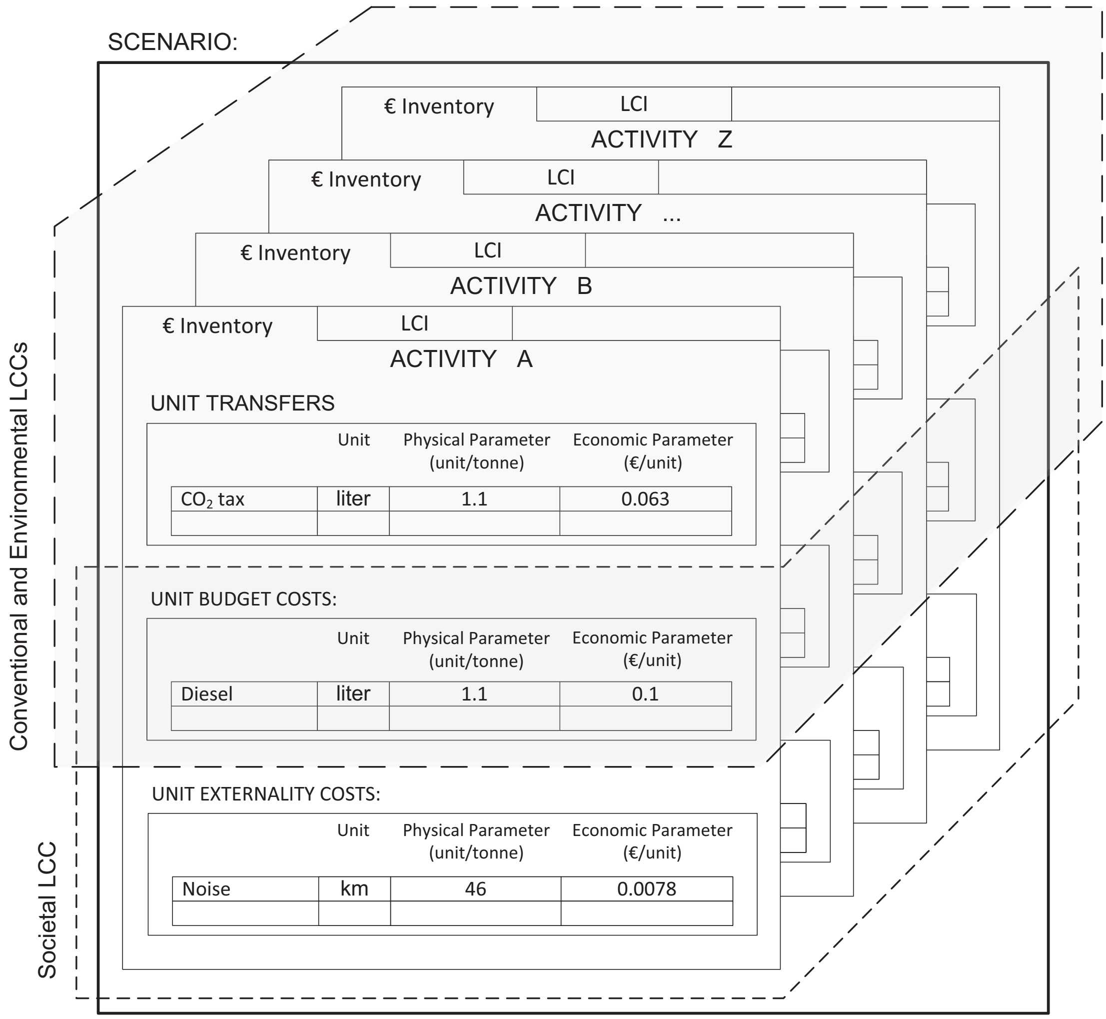

Fig. 1. Overview of the cost model structure, illustrating a range of activities (A through Z) and the cost coverage of Conventional, Environmental and Societal LCCs.

For fixed costs, annual costs (*AC*) [€/year] were divided by the annual usage rate of the plant (*AURp*) [tonne/year], in order to obtain fixed costs per item involved in treating one tonne of waste (*CIf*) [€/tonne], whereas for variable costs the physical amounts were the amounts of items needed to treat/dispose one tonne of waste (*IxT*) [kg (or MJ)/tonne] and they were multiplied by the unit price of the item (UPI) [€/kg (or MJ)], to obtain variable costs per item involved in treating one tonne of waste (*CIv*) [€/tonne], as in Eqs. (7) and (8). Physical amounts (for example 1 l diesel/tonne of waste collected) can be obtained from various sources, such as environmental reports.

$$CIf = \frac{AC}{AURp} \tag{7}$$

$$CIv = IxT * UPI \tag{8}$$

Most Operational and Maintenance (O&M) costs occur during the same year as the waste is collected/treated/disposed, but for some technologies costs may occur over time, for example in the case of leaching from landfilled waste leading to future costs for leachate treatment. Such future costs should be converted into net present value (*NPV*), Eq. (9). The same applies to revenues obtained in the future, for example from utilising biogas generated from landfilled waste.

$$NPV = \sum_{n=0}^{N} \frac{F}{(1+ir)^n} \tag{9}$$

2.4.2. Transfers

Transfers vary significantly between countries and individual waste technologies, but they are often based on measurable items, for example per waste quantities such as landfill taxes (63 €/tonne waste in DK in 2013, Fischer et al. (2012)), per direct emission such as CO_2 taxes (24 €/tonne CO_2 in DK in 2013, Energistyrelsen (2011)) or per output such as electricity subsidies (47 €/MWh electricity generated at biogas plants in DK in 2013, Energistyrelsen (2011)). The measurable item per tonne of waste (*IxT*) represents the physical parameter of the transfer, while the transfer per item (*TI*) is the economic parameter. Multiplying these two parameters results in a transfer related to the specific item per tonne of waste, i.e. unit transfer per item (*UTI*), as in Eq. (10).

$$UTI = IxT * TI \tag{10}$$

Transfers not borne by waste stakeholders, but rather borne by consumers of the co-products from the waste system (e.g. heat), are included in the assessment only if the product (e.g. heat) is not taxed when produced outside the waste system. For example, all heat produced in Denmark is taxed and the consumers pay the same tax regardless of the source. These types of transfers should not be included in the system, since they do not cause any change in the taxes paid by any waste stakeholder or heat consumer.

2.4.3. Externality costs

Externality costs are described through two parameters: an economic parameter representing the accounting price per unit of

Table 2
Characteristics of the three types of LCC.

	Conventional LCC	Environmental LCC	Societal LCC
Alternative naming	Full cost accounting Total cost accounting		Cost Benefit Analysis (CBA) Socio-economic assessment
Relation to LCA	None	Parallel LCA	Internalized LCA
System boundaries	Only internal costs	Internal costs External costs expected to be internalized	Internal costs + externaly costs
Economic cost categories	Market price (financial economics)	Market price (financial economics)	Shadow price (welfare economics)
Perspectives	Mainly 1 actor	All actors involved in the product LC	Society
Discounting	Recommended (market loan rate)	Recommended (market loan rate)	It should be stated clearly, even when discounting rate is assumed null (utility or time preference)
Guidelines for waste management	Full cost accounting (US, 2006)		Nordic Cost Benefit Analysis (Nordic Council of Ministers, 2007)
Transfers	Included	Included either in LCC or LCA	Excluded
Resource scarcity	Included (as price)	Included either in LCC (as price) or in LCA	Included in social price, i.e. private price + damage cost
Literature reviewed	Karagiannidis et al. (2013) Franchetti (2009) Coelho and De Brito (2013) Gomes et al. (2008) Groot et al. (2013) Kim et al. (2011)	Foolmaun and Ramjeeawon (2012) Assamoi and Lawryshyn (2012) Damgaard et al. (2011) Vinyes et al. (2012) Consonni et al. (2005) De Feo and Malvano (2012) Larsen et al. (2010) Sonesson et al. (2000) Zhang (2013) Carlsson Reich (2005) Eriksson et al. (2005)	Nahman (2011) Teerioja et al. (2012) Bozorgirad et al. (2013) Dijkgraaf and Vollebergh (2004) Panepinto and Genon (2011) Van Passel et al. (2013) Jamasb and Nepal (2010) Massarutto (2011) Vigsø (2004) Broitman et al. (2012) Dahlbo et al. (2007)

environmental emission, for example 40 €/tonne CO_2, and a physical parameter representing the unit environmental emission, i.e. the amount of emissions per tonne of waste, for example 10 kg CO_2/tonne waste. While accounting prices of environmental emissions should ideally correspond to society's willingness to pay in order to avoid emissions or associated impacts, they may also represent the marginal welfare abatement costs for reducing emissions in the first place (Møller and Martinsen, 2014). Please refer to Eshet et al. (2006) for further details on valuation techniques applied in relation to waste management.

Environmental externalities can be obtained from the parallel LCA, and these emissions and resource consumptions include: (1) direct emissions per tonne of waste input into a technology, (2) upstream emissions related to the consumption of commodities and capital goods within the technology and (3) circumvented emissions as a result of the downstream displacement of primary production (i.e. substitution based on material and energy recovery). Please refer to more detailed information about state-of-the-art waste LCAs elsewhere (for example Laurent et al., 2014a, 2014b; Astrup et al., 2014).

With respect to externalities, two temporal aspects have to be addressed: (1) discounting future damage relating to current emissions and (2) accounting for and discounting emissions distributed over time but nevertheless related to current waste management. Emissions occurring now (or at a specific time), for example CO_2 emissions, have damage effects distributed over time, so any associated externality costs could be discounted to a present value (or the value at the time of the emission). Future emissions, for example CH_4 emitted over time from a tonne of landfilled waste, should be accounted for within the LCA (Manfredi and Christensen, 2009), but the annual damage cost (representing damage costs at the moment of emission) should be discounted to present value (or the value at the time of treating/disposing of the waste).

Similar discounting principles apply to capital goods, since emissions related to the production of capital goods occur before the operation phase, though they have to be allocated equally to all the tonnes collected/treated/disposed by the capital good. Thus, inventories of capital goods should be annualised with a social discount rate and divided by annual usage rates. While quantification of social discount rates was beyond the scope of this study, the use of values suggested by local authorities for performance of socio-economic assessments is recommended (e.g. Miljøministeriet, 2010). Null social discount assigns equal importance to all emissions/damage regardless of the time of occurrence.

2.5. Case study scenarios

To evaluate and illustrate the applicability of the cost model, a case study was performed. The three types of LCC were performed, to assess costs related to the source separation of organic waste from 100,000 Danish households living in multi-family buildings. Scenario 1 represents the current treatment method, i.e. incineration, applied to mixed waste (after the source segregation of paper and glass), while Scenario 2 includes the source segregation of organic waste and its subsequent co-digestion with animal manure (with incineration of the residual waste). Both scenarios are illustrated in Fig. 2. Following common waste LCA principles, a zero-burden approach (Cleary, 2010) was applied, i.e. waste generation and upstream activities are excluded, indicating that products and materials are not produced with the purpose of being waste. Supplementary material (Annex 2) provides details on inventories and cost calculations for the case study.

While many objects of focus can be applied to the Conventional LCC, as all waste units/facilities/agents may be assessed individually and/or in groups, the following foci were selected for this particular case study: (1) costs for the entire system, (2) costs for an individual household represented by the waste fee, (3) costs incurred by the incinerator operator and (4) costs incurred by the collection operator. The Environmental and Societal LCCs included all costs incurred by all stakeholders within the waste system. The first Conventional LCC (for the entire system) thereby

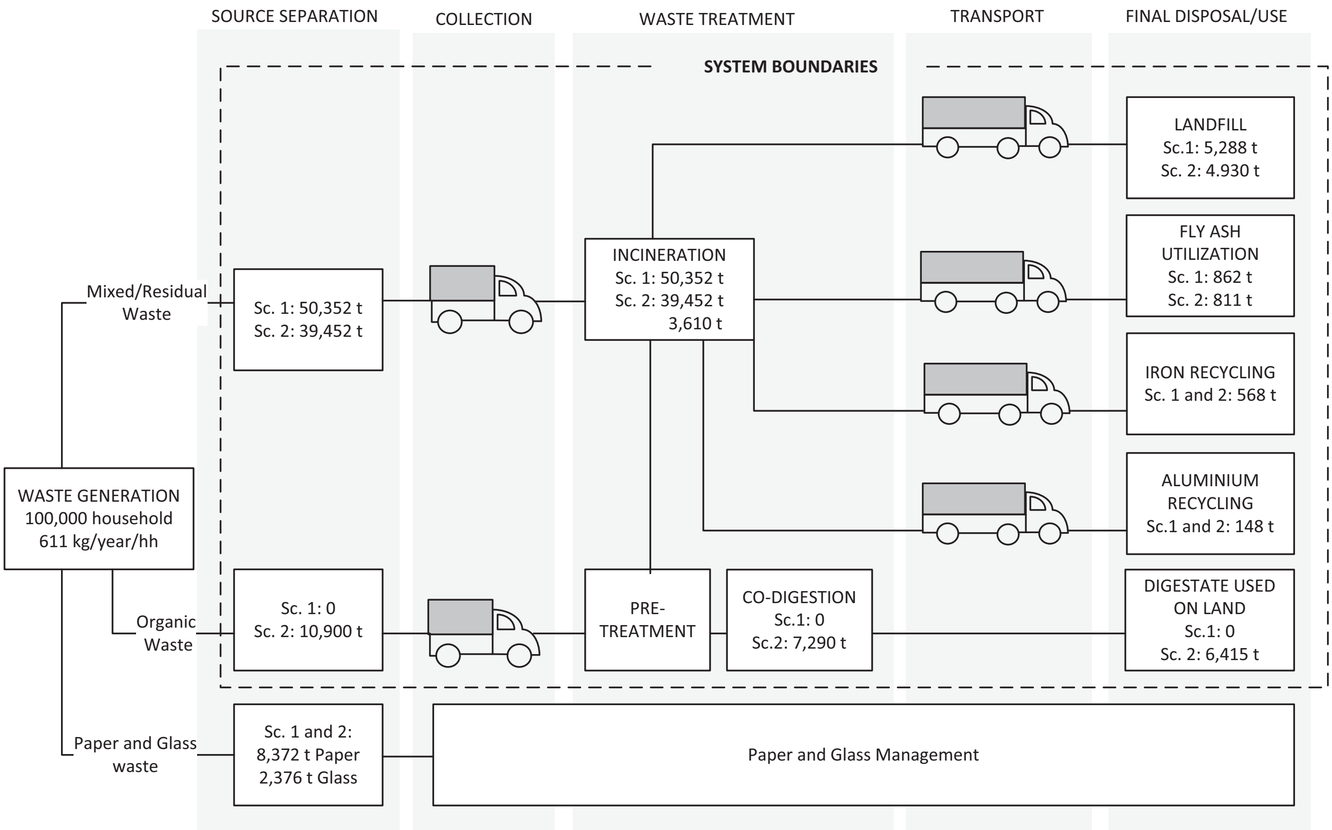

Fig. 2. Diagram of the two case study scenarios, including mass flows per functional unit and an indication of technologies associated with individual phases of the waste management system.

had the same system boundaries as the Environmental and Societal LCCs.

The LCA included in the Environmental and Societal LCCs accounted for the entire lifecycle of the waste system in the two scenarios, and consequently also for the production and disposal phases of the capital goods included in the system. Inventories related to the use phase were obtained from a recent Danish LCA study (Miljøministeriet, 2013), and those related to capital goods were obtained from Brogaard and Christensen (2012), Brogaard (2013), Brogaard et al. (2013a,b).

In the Environmental LCC, a lifecycle impact assessment (LCIA) was performed, with midpoint indicators based on the ILCD recommended methods (Hauschild et al., 2012), including the following impact categories: global warming potential, stratospheric ozone depletion, photochemical oxidant formation, terrestrial acidification, eutrophication potential, freshwater eutrophication, fossil resource abiotic depletion, mineral resource abiotic depletion, human toxicity carcinogenic effects, human toxicity non-carcinogenic effects and freshwater ecotoxicity. Results are shown in person equivalent (PE) per functional unit (see Table A3.1 in Supplementary material for normalisation factors).

The Societal LCC integrated emissions from the LCA inventory with Danish accounting prices for air emissions (Miljøministeriet, 2013) for the following compounds: carbon dioxide, methane, nitrous oxide, particulate matter, nitrogen oxides, sulphur dioxide, carbon monoxide, hydrocarbons, mercury, lead and dioxins (see Table A3.2 in Supplementary material for shadow prices).

While a detailed sensitivity analysis was beyond the scope of the study, a simple break-even analysis was performed, to evaluate the robustness of the results. This was done by: (1) identifying parameters that could potentially change ranking between scenarios for each type of LCC, and (2) finding the turning point, i.e. the value of the parameter in which both scenarios had the same costs.

3. Results and discussion

3.1. Critical aspects in the existing literature

As a basis for implementing the cost model and discussing the results obtained from the case study scenarios, key studies in the existing literature were evaluated with respect to critical assumptions and assessment principles. A range of important issues were identified.

It was found that system boundaries in regard to the economic, environmental and social assessments did not always correspond. For example, LCAs (parallel to the Environmental LCC) often omit emissions related to the production and disposal of the capital goods included in the economic assessments (e.g. Carlsson Reich, 2005; Eriksson et al., 2005; Dahlbo et al., 2007; Larsen et al., 2010), while in traditional welfare cost assessments, national geographical scopes are often applied (Nordic Council of Ministers, 2007; Møller et al., 2014) and global boundaries are typically applied in LCAs (Nordic Council of Ministers, 2007; ISO, 2006). Reapplying cost assessment results from one study as input data in other studies may induce inaccuracies and biased results which are not easily identifiable. For example, Massarutto (2011) used the cost functions of Tsilemou (2006) as the basis for financial costs in a Societal LCC, but transfers were not excluded from the cost functions, while Jamasb and Nepal (2010) included gate fees as revenues in a welfare economic assessment (called “Societal Cost

Benefit Analysis'' by the authors). This is also important when studies use European cost data, such as Damgaard et al. (2011) and Karagiannidis et al. (2013), since transfers may differ between countries.

Two aspects were rated as critical regarding the internalisation of environmental damages. Firstly, some Conventional and Environmental LCCs included anticipated transfers but used different approaches. For example, Groot et al. (2013) accounted for an imaginary CO_2 tax (without reporting the value), Kim et al. (2011) converted the GHG reduction into a monetary value based on principles set out in the Clean Development Mechanism (CDM) whereas Vinyes et al. (2012) and Zhang (2013), on the other hand, used costs for mitigating CO_2 emissions according to the international and national CO_2 markets, respectively. These are all valid approaches if transparently reported, but the results were affected by the assumption and valuation principles applied. Secondly, while resource market prices to some extent also represent resource scarcity, it may be unclear how large a fraction of the price is related to scarcity itself, i.e. the so-called resource rent (Carlsson Reich, 2005). It is considered likely that current market prices depend more on short-term resource availability than on long-term abiotic resource depletion (scarcity), and as a result market prices might not account fully for associated future damages caused by current resource consumption/savings. In an Environmental LCC, short- and long-term resource aspects are included either in the economic part (by market price) or in the environmental part (by resource depletion impact categories included in the LCA). In a Societal LCC, future damages remain unassessed until empirical studies estimate externality costs involved in resource depletion. The same principle applies to many environmental emissions whose accounting prices have not been estimated yet.

Discounting future financial costs was also found to be critical when results are meant to be disclosed in parallel with an LCA. While Hunkeler et al. (2008) consider it inconsistent with the Environmental LCC (since it is meant to accompany the LCA), most of the investigations, such as Carlsson Reich (2005) and Assamoi and Lawryshyn (2012), discounted financial costs in order to properly allocate the costs.

Finally, and as mentioned previously, Societal LCCs are welfare economic assessments, and market prices should be converted to accounting prices in order to add up two terms, namely marketed and non-marketed goods, with the same units, namely social prices, but this was only done in a few investigations such as Vigsø (2004) and Møller et al. (2014).

3.2. Case study scenarios: Conventional LCC

Fig. 3A shows the four selected Conventional LCCs for the two case study scenarios (with and without the source segregation of organic waste). The main difference in total costs between the scenarios was associated with the source separation of organics, i.e. extra bins and bags paid for by households, and the collection of organics. The differences between the waste fees in both scenarios are due mainly to collection and, to a minor extent, incineration. See Table A4.1 in the Supplementary material for detailed costs related to the Conventional LCCs.

Collection costs increased by 43% in total due to the separate collection of organic waste, and organic collection incurred higher costs per tonne than residual and mixed waste. This was caused by the fact that the number of collection points (households) required to fill a waste truck is inversely proportional to the amount of waste per collection point at a given time. This significantly affected the capital costs of trucks and labour costs because both costs were dependent on the tonnes of waste collected.

Overall collection costs per tonne were around 50 €/tonne (45.8 €/tonne for the mixed waste of Scenario 1 and 56.1 €/tonne for the residual waste of Scenario 2). The corresponding cost of emptying one container (typically paid to the collection operator by the municipality) was determined to around 3.70 €/time (assuming seven households per container, 0.5 tonne/household/year with a collection frequency of one time per week). This is similar to the amount reported by Miljøministeriet (2013), where emptying costs for a 660-l container were 3.10 €/time. The collection cost per ton of organic waste was 96.3 €/tonne equivalent to an emptying cost of 4.60 €/time (assuming 25 households per container, 0.5 tonne/household/year with a collection frequency of one time per week). The collection of organic waste was more expensive than collection of residual and mixed waste because costs are inversely proportional to the amount of waste collected in each collection point. Other options for collection could have been assessed; however, the main purpose with the case study was to show the applicability of the cost model, rather than provide an exhaustive analysis.

For incineration, the costs per tonne of residual waste (without organics) were approximately the same as for the mixed residual waste including organics, although residual waste generated more energy per tonne, due to a higher calorific value. This was because the thermal capacity of the incinerator was met with fewer tonnes in the case of residual waste (a lower annual usage rate), thereby leading to higher capital and maintenance costs per tonne of input waste. A similar effect could be observed for the incineration of solid residues from the anaerobic digestion of organics – although containing only little energy, incineration costs per tonne of digestion residue were lower than for both residual and mixed waste, because of the fixed costs being distributed between more tonnage (a higher usage rate). Overall, residual waste incineration was less costly than mixed waste incineration because of the difference in input waste amounts per functional unit, i.e. 50,352 tonnes of mixed waste vs. 39,452 tonnes of residual waste. The capital and operational costs of the scenarios (83 €/tonne in Scenario 1 and 94.73 €/tonne in Scenario 2, without revenues from energy recovery) were somewhat higher than the values found by Tsilemou (2006) (32.1 €/tonne and 15.9 €/tonne, respectively), while the incineration and downstream costs per tonne of waste (64 €/tonne in Scenario 1 and 73 €/tonne for Scenario 2, 76 €/tonne and 87 €/tonne, respectively, including transfers) corresponded well with examples of Danish incineration gate fees of 65 €/tonne (e.g. Amager Ressource Center, 2014).

3.3. Case study scenarios: Environmental LCC

The economic part of the Environmental LCC is identical to the Conventional LCC for the entire system (i.e. Fig. 3A, Total*) since there were no anticipated transfers in any of the activities involved in both scenarios, i.e. we did not expect any externalities to be internalised into monetary costs in the near future. In order to evaluate the Environmental LCC (Fig. 3B), the results from the parallel LCA thereby supplement the above discussions. Overall, both scenarios performed similarly in all impact categories with net environmental benefits for all energy-related impact categories (global warming, photochemical oxidant formation, terrestrial acidification, resource depletion (fossil) and freshwater eutrophication) and net environmental loads for resource depletion (metal), carcinogenic human toxicity and freshwater ecotoxicity. The environmental savings in the energy-related impact categories stem from the displacement of fossil-fuel based energy by the energy recovered in the incineration and co-digestion plants.

Scenario 2 provided fewer impacts than Scenario 1 for freshwater eutrophication and non-carcinogenic human toxicity, due to the displacement of mineral fertiliser when applying digestate on

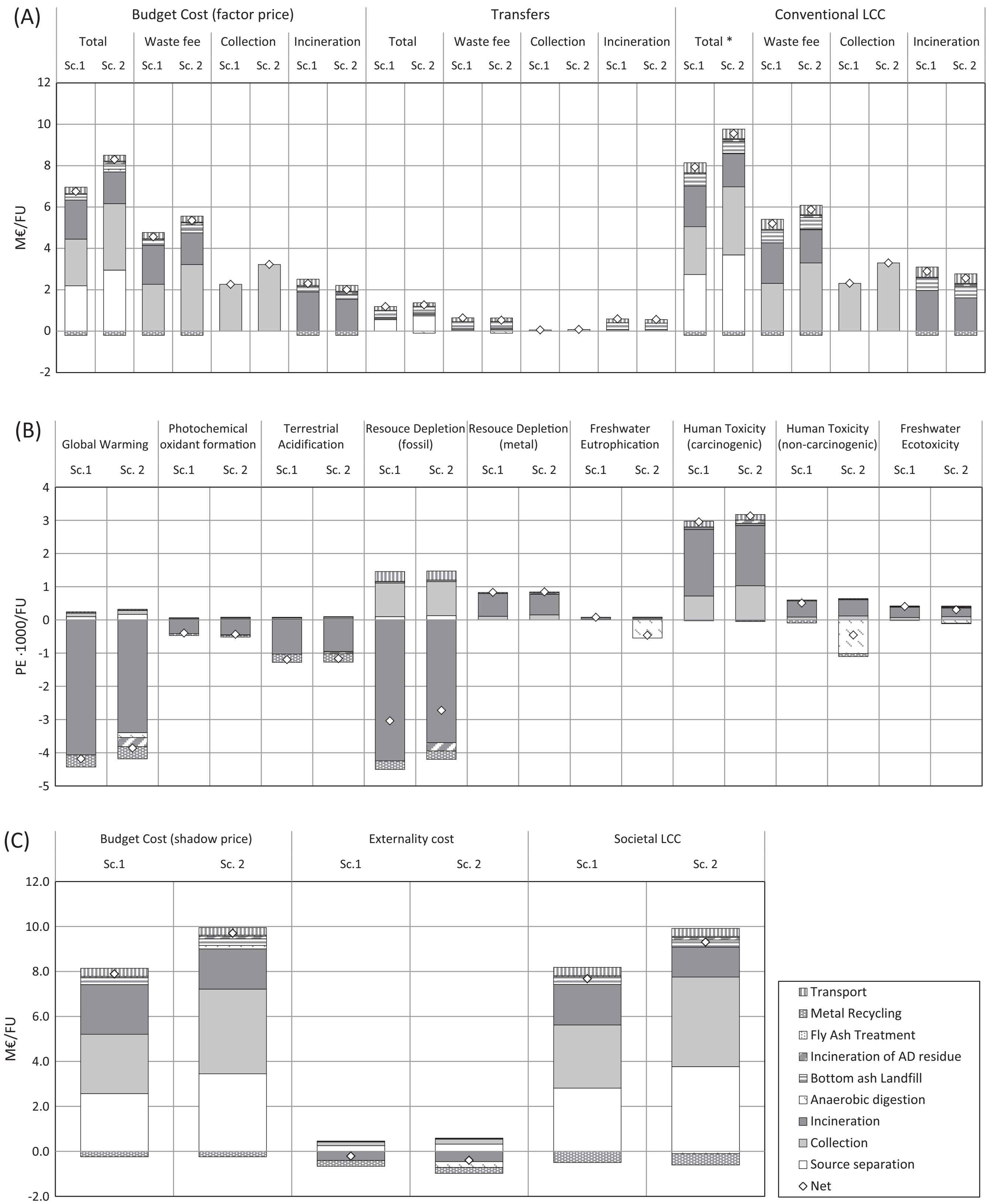

Fig. 3. (A) Conventional LCCs of the case study scenarios in millions of euros per functional unit (M€/FU). Both scenarios are compared relative to: (1) an entire waste system (Total), (2) waste fee, (3) collection and (4) incineration. (1)–(4) are the four separate foci selected for the Conventional LCC case study and are explained in Section 2.5. The legend refers to the activities included in the four assessments, i.e. (1)–(4). The "Total" in this figure is identical to the economic part of the Environmental LCC. FU stands for functional unit. (B) Environmental part (LCA) of the Environmental LCC in person equivalent per functional unit (PE·1000/FU). The Environmental LCC of the case study is identical to the "Total" in the Conventional LCC (see Fig. 3A). The legend refers to the activities included in the assessment. FU stands for functional unit and PE for person-equivalent. (C) Societal LCC of the case study scenarios in millions of euros per functional unit (M€/FU). The legend refers to the activities included in the assessment. FU stands for functional unit.

land. These environmental benefits were related mainly to circumvented phosphorus emissions into water for freshwater eutrophication and emissions of cadmium and zinc into soil for non-carcinogenic human toxicity. The production and disposal of capital goods caused most of the impacts related to carcinogenic human toxicity (chromium and mercury emissions from steel production) and the depletion of mineral resources (consumption of iron, nickel and zinc). For further details and disaggregated LCA results, see Fig. A5.1. (Annex 5) in the Supplementary material.

Overall, based on the combined economic and environmental assessments offered by the Environmental LCC, the environmental savings provided by the extra financial costs associated with organic source segregation (1.6 M€) were found to be: 907 PE for non-carcinogenic human toxicity, 527 PE for freshwater eutrophication, 55 PE for freshwater ecotoxicity and 29 PE for photochemical oxidant formation. Conversely, the same costs induced added impacts of: 697 PE for carcinogenic human toxicity, 342 PE for global warming, 339 PE for the depletion of fossil resources, 93 PE for the depletion of mineral resources and 45 PE for terrestrial acidification.

3.4. Case study scenarios: Societal LCC

Fig. 3C shows the Societal LCC for both scenarios. Budget costs were the same as in the Conventional (Total*) and Environmental LCCs, albeit now in accounting prices, i.e. 17% higher than factor prices (Miljøministeriet, 2010; Møller and Martinsen, 2014). Scenario 2 showed extra social costs of 1.6 M€/FU compared to Scenario 1.

The costs of marketed goods, i.e. represented under budget costs, were an order of magnitude higher than externality costs. Miljøministeriet (2013) reported similar differences between marketed and non-marketed goods/services, i.e. externalities, when assessing similar waste systems to the one discussed here.

Net externality costs per tonne of waste input into the specific activity were: 4.9 and 6.3€/tonne for source separation for Scenarios 1 and 2, respectively, 3.3 and 3.4 €/tonne for the collection of mixed/residual waste for Scenarios 1 and 2, respectively, 7.7 €/tonne for the collection of organic waste (Scenario 2), −8.1 and −11.7 €/tonne for the incineration of mixed/residual waste for Scenarios 1 and 2, respectively, 5.0 €/tonne for landfilling bottom ashes in both scenarios, 0.8 €/tonne for the neutralisation of Air Pollution Control residues, −1269.6 €/tonne for aluminium recycling and −124.8 €/tonne for steel recycling.

Externalities reported by Rabl et al. (2008), Dijkgraaf and Vollebergh (2004) and European Commission (2000) are much higher than the ones obtained here, which can be explained by: (1) different accounting prices per emission (see Table A3.2 in Supplementary information) and/or (2) different system boundaries and/or assumptions in the assessments, for example the type of marginal energy displaced through energy generated by incineration.

Data on externality valuations are scarce, and this may affect the outcome of the Societal LCC. No applicable valuation estimates were found available for any of the emissions into water and soil, and for some of the emissions into the air. In addition, the exclusion of non-environmental externalities such as time spent by households, due to the lack of data, was also a drawback of this assessment.

3.5. Case study scenarios: Break-even analysis

Scenario 2 was more costly than Scenario 1, but the overall results were highly sensitive to assumptions in relation to the sorting time spend by households, as well as technical parameters such as the number of households sharing containers. Changes in some of these parameters could change the ranking between scenarios. For the Conventional LCC, the total costs for Scenario 2 were 1.6 M€ higher than Scenario 1. These extra costs could be balanced by revenues deriving from digestate sales, but the agricultural sector would have to pay a minimum of 250 € per tonne of wet digestate (23% dry matter), instead of getting the digestate for free (baseline assumption). This is far from realistic in a current Danish context where farmers get the digestate for free.

For collection, the difference of 1.0 M€ between the scenarios could be reduced by 75%, i.e. to 0.25 M€, by increasing the number of households sharing one container (from 7 to 10 households sharing one container for residual waste and from 25 to 50 households sharing one container for organic waste) and consequently decreasing the loading times in the collection points (from 10 to 8 min per collection point for residual waste and from 5 to 3 min for organic waste).

The ranking of the Societal LCC could change according to the same parameters as for the total Conventional LCC, i.e. the digestate price and utilisation rate of the containers for organic waste, though for the Societal LCC the critical parameters were the time and value associated with household source segregation. While the baseline assumption in the scenarios was 0 €/h for household sorting, different approaches exist for estimating this value. For example, Ekvall (2002) proposed using one-third of the hourly wage as a cost, but if households feel a benefit because of the sorting process, i.e. 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 experienced as a burden, then the value should be positive and treated as a cost to the households.

In order to address this aspect transparently, we find it reasonable to apply a neutral value (zero) as a base assumption in the main results and then determine a break-even value for the sorting time in which both scenarios resulted in identical social costs, i.e. same value of Societal LCC. It should be mentioned that quantification of externality costs related to source segregation is beyond the scope of this investigation, and the break-even value only represents a "turning point" between the two situations. In this case study, the analysis revealed that that the turning point corresponded to a value of −1.20 €/h (13 h/year/household, according to Bruvoll et al., 2000), i.e. to balance the extra costs of treating organic waste separately, households should be willing to spend time on sorting. And the benefit from sorting should correspond to a value larger than −1.20 €/h.

3.6. Bottom-up vs. top-down approaches

The cost model applied a bottom-up approach in which cost items within each technology were calculated first, and then followed by a calculation of the technology cost and finally the cost for the entire scenario. This approach requires detailed knowledge not only of the involved technologies but also of each specific cost item related to a technology. While this may facilitate transparency and a higher level of technical accuracy, the approach also has extensive data needs. Additionally, as state-of-the-art waste LCAs also follow a bottom-up approach, applying similar principles in the cost assessment ensures the best possible correspondence between the two parts of the LCC. Examples of other cost assessments on waste management applying bottom-up approaches include Broitman et al. (2012) and SWOLF (Levis et al., 2013).

Alternatively, a top-down approach may be applied. In this case, the starting point is overall (and typically aggregated) total costs for technologies. If needed – and assumed affected by the scenarios – these costs are then modified to reflect the scenarios accordingly.

Finally, all costs are added up for the entire scenario. This approach requires detailed knowledge of the starting cost estimates and how these may be adjusted. While this approach may be easier and more accessible, lack of transparency and the risk of applying inappropriate and aggregated cost items may significantly affect the final results. Additionally, the results obtained from a top-down cost assessment may be considered static and only reflect a specific situation. In other words, applying the results or the cost data used in a top-down assessment should be done with care, for example to ensure that framework conditions are still valid in the new context. One such example is gate fees. While these may be calculated in a simple manner based on aggregated cost values for downstream activities in the waste management chain, such calculations do not account for the fact that waste operators will adjust their gate fees as soon as a change in the waste system affects their internal costs. Examples in the literature of top-down approaches applied to waste systems can be found in Miljøministeriet (2013) and Karagiannidis et al. (2013).

3.7. Dissemination of the results

The results should reflect whether the target group is a specific decision-maker (related to the case study), the scientific community or the wider public community. When results are intended only for specific decision-makers, the level of aggregation should reflect the needs of the decision-maker; for example, a single value may be preferred for the Societal LCC without the need for details related to the financial aspects of the Conventional LCC or the environmental impacts in the Environmental LCC. However, if the results are intended to provide a more generic understanding of mechanisms, and they are offered to a wider audience, then the aggregation of results and cost data cannot be recommended. To allow for the optimal applicability and transferability of results and cost data in a case study, we suggest: (1) cost calculations should be clearly disclosed, for example the equations by Gomes et al. (2008) can be transferred easily to other regions, whereas the results of Karagiannidis et al. (2013) and Sonesson et al. (2000) can only be used within the context of the case study, and (2) cost results should be provided in disaggregated cost types, for example budget, transfers and externality costs. Only then the results can be interpreted and potentially applied in the context of other case studies.

4. Conclusions

A consistent cost model for Life Cycle Costing (LCC) was developed based on a bottom-up approach including detailed cost items for all key technologies within modern waste management systems. The model followed a Unit Cost Method, in order to allow full transparency of the cost calculations and optimal applicability of the calculation principles for cost assessment of waste systems. Cost items were classified into: (1) budget costs, (2) transfers and (3) externality costs. Each cost item was defined by two parameters: a physical and an economic parameter related to the specific waste technology in question. The cost model allows calculation of: Conventional LCC, Environmental LCC and Societal LCC. The first two LCCs include budget costs and transfers, while the Societal LCC includes budget costs in accounting prices and externality costs. The cost model was applied to two case study scenarios, to demonstrate applicability and to illustrate the type of results obtained: (1) incineration of household waste, including organics, and (2) anaerobic digestion of source-segregated organic waste and the incineration of the residual fraction. Organic waste source segregation and subsequent activities resulted in an extra financial cost of 16 €/year/household (10 €/year/household for extra collection and 9 €/year/household for extra bins and bags, combined with a reduction of 3 €/year/household for incineration). These extra costs provided environmental savings for non-carcinogenic human toxicity, freshwater eutrophication, freshwater ecotoxicity and photochemical oxidant formation, but they contributed with environmental loads to carcinogenic human toxicity, global warming, terrestrial acidification and resource depletion. The case study demonstrated that valuing the time households spend on source segregation may significantly affect the results of the Societal LCC, though assigning a cost of −1.20 €/h would eliminate the overall difference between the two scenarios, i.e. households should experience a benefit from sorting. Furthermore, it was demonstrated that while the production and disposal of capital goods were not critical for the economic assessment itself, a significant influence on environmental impacts in toxicity-related impact categories could be observed. This highlights the importance of equivalent system boundaries between the economic and environmental parts of an LCC, and it also illustrates that unbalanced decisions for system cut-off cannot be advised. Critical shortcomings were also observed in the existing literature: (1) system boundaries were not always equivalent between the economic and environmental parts of assessments, (2) transfers were sometimes included in Societal LCCs, although this should not be the case, and (3) the internalisation of environmental damages in Societal LCCs was often carried out but with poor explanations despite the fact that valuation principles may affect the results. The proposed cost model offers a coherent framework for assessing both the economic and environmental aspects of waste management systems, by providing detailed cost calculations for individual waste technologies, thereby facilitating consistency with state-of-the-art Life Cycle Assessments (LCAs).

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Appendix A. Supplementary material

Supplementary data associated with this article can be found, in the online version, at http://dx.doi.org/10.1016/j.wasman.2014.10.033.

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Glossary and abbreviations

Annual mass capacity (AMC): parameter used in incineration to represent the annual usage rate of the plant, i.e. its annual capacity, when the waste has the designed Lower Heating Value (LHV), [tonne/year]

Annual usage rate (AUR): parameter used in waste technologies that represent the real annual capacity of the plant [tonne/year]

Break-even analysis: here defined as an analysis employed to determine the value of a given parameter for with the ranking between scenarios will change, other parameters remaining the same
Budget costs (BC): here defined as internal costs paid by a waste stakeholder, e.g. a waste generator or a waste operator
Unit Budget costs of an activity (UBC): budget costs per tonne of waste input into the activity
Activity budget costs: are budget costs for waste input into the activity
Externality costs (or external costs): are non-marketed costs caused an activity paid for by a party who did not choose to incur this cost (or benefit)
Factor price: market prices associated with production input factors (labour, capital and land), exclusive of transfers
Financial assessment: analysis to determine the costs and benefits of a project, and only accounts for money flows of marketed goods and services, i.e. non-market effects are not included
Fixed costs (or overheads): in waste plants, they are expenses that are independent of the amount of waste treated (opposed to variable costs)
Internal costs: expenses in real money flows incurred by any stakeholder (opposite to externality costs). Same as Private costs
Life Cycle Costing (LCC): assessment of all the costs associated with the lifecycle of a product or service
Market price: price of a good when offered in the marketplace
Net Present Value (NPV): sum of the present values of individual future cash flows
One-off costs: expenses incurred once only during the lifetime of the good, such as a capital cost or back-end costs
Pecuniary externality: transfer related to an external effect that operates through price mechanism. Looking at the overall society, pecuniary externalities offset each other, one agent gets better while another worse off, but both are external agents, none of them caused the effect
Private costs: expenses involved in providing a good/service accounted for in monetary terms, i.e. includes the cost of any production input used by the supplier and excludes all non-market externality costs. Same as Internal cost
Shadow price (or accounting price): quantitative measure of utility changes caused by a project, it represents willingness to pay for a good (market or non-market good)
Opportunity cost: value the used resources would have received in the best alternative foregone
Social costs: sum of private costs and externality costs
Socio-economic assessment (welfare economic assessment or Economic assessment): analysis employed to determine the costs and benefits of a project to the community; it employs shadow prices and the monetisation of non-market effects
Transfer (income transfer): taxes, subsidies, fees and duties used to distribute income between different agents in society, but do not represent any resource reallocation
Variable cost: in waste plants they are expenses that depend on the amount of waste treated (opposed to fixed costs)