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Review

Consequential lifecycle modelling of solid waste management systems – Reviewing choices and exploring their consequences

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A B S T R A C T

Application of consequential lifecycle assessment modelling has gained increased interest in the area of solid waste management. In such assessments, identification of affected technologies and choices of system boundary setting are of key importance. With the aim of investigating how previous consequential lifecycle assessments of solid waste management systems have tackled these issues, a review was performed of 36 previously published studies. The intention is to contribute to improved understanding of the challenges of performing consequential lifecycle inventory modelling of solid waste management systems, which could facilitate future studies. Results demonstrate a strong relation between the selection of affected energy production technology and overall GHG-emissions. In general, assuming that energy provision from less polluting technologies is affected by studied changes will commonly discredit waste-to-energy technologies and promote material recycling. However, made choices were also in many cases not justified. Materials substituted by waste derived goods are frequently represented by average data. The detected inconsistency in how energy provision and material provision are modelled could result in biased results, and care should be taken to minimize this risk. Four aspects are identified where current practice in system boundary setting choices is diverse and where those choices could have significant influence on overall results; counterfactual waste management, fate of materials avoided through material recycling, cascading effects and rebound effects. This paper argues that there is a need for increased transparency and coherence in identification of affected processes/technologies, as well as for a broader approach in system boundary setting, if studies have the aim of serving as relevant input for decision makers.

Contents

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1. Introduction

Life cycle assessment (LCA) has been presented as a decision-support tool, applying a holistic perspective in quantifying environmental impacts (European Commission, 2010). Its usefulness in providing valuable information for decision makers of solid waste management (SWM) systems has previously been demonstrated (EU, 2008). Nevertheless, in their review of a large set of LCAs of SWM-systems, Laurent et al. (2013) identified a certain confusion of concepts and terminology surrounding different types of lifecycle inventory (LCI) modelling frameworks. According to the authors, one of the main reasons could be a lack of adequate goal definition and insufficient reflection on the context situation in which the study is performed. Indeed, the context situation has previously been pointed out as being of key relevance for an adequate selection of LCI modelling framework, and the ILCD Handbook (European Commission, 2010) makes specific recommendations on this issue (later subjected to a critical review by Ekvall et al. (2016)). Regardless of the reasons given for selecting a particular LCI modelling approach, the choice made has undoubtedly a significant influence on the definition of system boundaries (European Commission, 2010). As discussed in Thomassen et al. (2008), ALCA and CLCA approaches are also likely to produce different results and offer different messages to end users. Attributional lifecycle assessment (ALCA) employs a system-modelling approach where the inputs and outputs are attributed to the functional unit of a product system by linking and/or partitioning the unit processes of the system according to a normative rule (Sonnemann and Vigon, 2011). Consequential LCA (CLCA), on the other hand, is change-oriented and quantifies the effects associated with changes in the life cycle of a system brought about by a decision (Weidema et al., 2009; Curran et al., 2005). In this way, the consequential approach seeks to take the environmental assessment a step further, in order to analyse how environmental burdens may vary in response to changes with market implications, where processes are linked via market mechanisms beyond the foreground system (Vazquez-Rowe et al., 2013). Such linkages can occur for example when waste is used for energy or material recovery. Recovery implies that waste-based resources are released on the market, with the effect of substituting other product-systems. In CLCA, this is modelled through system expansion (Finnveden et al., 2009). Consequential LCI modelling should, according to Weidema et al. (2009), include the unit processes that change as a consequence of a decision, and identification of these processes has been presented as the key issue in consequential LCI modelling. Finnveden et al., 2009 and Mathiesen et al. (2009) have argued that the introduction of affected processes[1] in the LCI modelling involves large uncertainties and significant time and effort are required to understand and reduce such uncertainties.

A number of reviews of SWM LCAs have been performed previously, with varying focus and objectives; some are limited to specific waste categories (Morris et al., 2013; Villanueva and Wenzel, 2007; Bernstad and la Cour Jansen, 2011; Lazarevic et al., 2010), and others to methodological conducts (Cleary, 2009; Gentil et al., 2010). Laurent et al. (2013) aimed at providing a comprehensive mapping and contextual analysis of LCA of SWM systems, but also at identifying potential misuses and misunderstandings and providing guidance to ensure robust application of the LCA methodology to SWM. The issue of consequential modelling was however not explored in details by the authors. Zamagini et al. (2012) argues that what distinguishes the two modes of LCA is the choice of the processes to be included in the system, but that the identification of those processes is often done inconsistently, using different arguments, which leads to different results. To our knowledge, no comprehensive review has until now been presented, focusing on identification and modelling of affected processes and technologies in CLCAs of SWM systems, in order to identify the augments most commonly used in this particular field, potential inconsistencies and, additionally, discuss these effects on results gained.

The objective of the present review is therefore not to add to debate on why and when attributional and consequential modelling should be used, nor the type of data most relevant to be used in different situations when applying consequential LCI modelling. These issues have been discussed thoroughly in several previous works (Suh and Yang, 2014), and is continuously an issue of debate (Ekvall et al., 2016). Rather, the objective of this paper is to describe how LCA practitioners working specifically with SWM.

a) Justify the choice of consequential LCI modelling
b) Select and justify the choice of data used in consequential LCI modelling

This was done by exploring choices made by LCA-practitioners in previously published LCAs of SWM systems performed with a consequential approach. The intention is to present a state of the art in the field and contribute to improved understanding of the challenges of performing consequential LCI modelling of SMW systems.

2. Method and material

The keywords *consequential, marginal, LCA, lifecycle assessment, life cycle assessment*, *CLCA, waste* and *residue* were applied in the databases Scopus, JSTOR and Science Direct, resulting in 177 hits. A screening of abstracts resulted in the selection of 36 scientific papers, presenting case studies where it is clearly stated that consequential modelling is applied, or where processes included in the assessment are described as marginal, i.e. affected by changes in the investigated SWM system. Only peer-reviewed original research papers published in English in scientific journals over the last ten years (2007–2017) were included. Papers studying treatment of solely residues/by-products from forest and agriculture were excluded. The main reason for this choice is the modelling of indirect land use changes (iLUC), as this is considered a topic on its own, and it would not be possible to include it in the present review

[1] The term “affected processes” will, in the present study, refer to any process (i.e. production of heat, electricity or materials) that may be affected by life cycle changes within the expanded system, while “technologies” in this context, will refer to specific technologies used for provision of these processes (i.e. wind power, production and combustion of natural gas etc.).

[2] The term “marginal” is commonly used in reviewed studies, referring to the processes and technologies affected by changes in the waste management system, with the underlying interpretation that changes are small enough to be approximated as infinitesimal, and that no changes are caused in the way the system is operated (Azapagic and Clift, 1999).

due to scope limitation. The review was performed with the intention of being as comprehensive as possible, including all studies that could be relevant for the aim of the study. However, relevant publications might have been excluded unintentionally. As the main objective of the study was to describe and discuss how researchers previously have justified the use of CLCA and the selection of data used in the modelling, conference papers were excluded as they commonly present a shortened version of a larger study, with reduced possibilities to justify made choices. The list of investigated studies and key information is presented in Table 1.

A matrix was established for the review, including i) justification of using consequential LCI modelling, ii) system boundary settings (included/excluded processes), iii) method(s) used for identification of affected processes/technologies, and iv) handling of uncertainties connected to the processes/technologies identified as affected by studied change. The outcomes of the review are presented and discussed below.

3. Results and discussion

3.1. Justification of using consequential LCI modelling

The choice between application of attributional or consequential modelling should be presented in the scope of the LCA (Laurent et al., 2013), and is of large importance since this has a strong influence on the LCA results (Ekvall et al., 2016). It could thereby be assumed that the chosen approach should have been thoroughly discussed and justified by authors of reviewed studies. However, the selected modelling approach is clearly justified only in half of the reviewed studies (SI).

In several studies, the application of CLCA is justified by a general focus on the consequences of a decision (Fruergaard and Astrup, 2011; Tonini and Astrup, 2012; Sevigné-Itoiz et al., 2014, 2015a,b). In other studies, the application of CLCA is justified more specifically by a wish to account for indirect effects and counterfactual waste management processes (Eriksson et al., 2007; Styles et al., 2016; Salemdeeb et al., 2017). Four of the reviewed studies have the outspoken aim to compare use of attributional and consequential modelling. In two of them, the two modelling approaches are applied with different objectives: the ALCA is used to attribute environmental impacts to different treatment alternatives, while the CLCA examines the potential environmental impacts from a shift between current management and a proposed alternative (Feraldi et al., 2013; Hums et al., 2016). In the other two, the comparison of results gained through ALCI and CLCI (attributional/consequential LCI) modelling is presented as an objective in itself (Boesch et al., 2014; Kua, 2015).

Justifying use of CLCI modelling by a will to focus on the consequences of a decision finds support in several references (Finnveden et al., 2009; Sonnemann and Vigon, 2011). This suggests that when LCA is used for decision-support, the LCI model should reflect the consequences of a decision in question, making consequential modelling more relevant. However, as seen in previous reviews of SWM systems, attributional modelling is by far more commonly applied on LCAs developed with the purpose of serving as decision support tool (Laurent et al., 2013). Thus, to many researchers in the field, CLCA still seems to be regarded as something new, in need of further investigation and interpretation, calling for an interest in comparing outcomes from ALCI and CLCI modelling and discussing the robustness of results gained through the use of these two approaches.

3.2. Methods used for identification of affected technologies[2]

In most reviewed studies, the substitution of conventional goods by waste derived goods constitutes the main benefit related to waste-to-energy and material recovery alternatives. The identification of what is being substituted is therefore of large importance to assess overall environmental impacts from the investigated SWM system.

Methods used for identification of affected processes in reviewed studies could be grouped into seven categories (Table 2). In several cases, one method was used for identification of one type of process and another for identification of another. Methods and choices made in identification of affected processes and technologies are discussed below, divided in provision of energy and material respectively. More details are found in the SI.

3.2.1. Identification of affected energy technologies

Electricity and heat technologies affected by a change in the SWM system can be identified either as single or mixed technologies. Amongst reviewed studies, a clear majority identifies single technologies rather than a mix of affected technologies. For identification of single affected technologies, it is clear that the work of Weidema (2003, 2009) has had a strong influence in the field (Table 2). Several studies also refer to recommendations made by Mathiesen et al. (2009), namely the inclusion of two radically different technologies as affected options for production of electricity as two alternative base-scenarios (Eriksson et al., 2007; Merrild et al., 2008; Boesch et al., 2014).

In several cases, projections or plans from national or international energy entities are used. In these cases, the affected technologies are assumed to be the mix of projected new installation capacity according to international projections (Feraldi et al., 2013, regarded as long-term marginal based on EIA (2009)), national projections (Sevigné-Itoiz et al., 2015b) or national objectives of phasing out specific fuels (Turconi et al., 2011; Martinez-Sanchez et al., 2016). Although use of such official sources could seem as a robust option, Mathiesen et al. (2009) showed that energy plans historically have rather limited relevance for projection of actual developments of national electricity markets.

Dynamic optimising models, such as energy systems analyses (ESA), can give a more complete description of the consequences of using or delivering electricity (Mathiesen et al., 2009). ESA can take into account effects on the utilisation of existing production facilities as well as effects on investments in new production facilities (Eriksson et al., 2007). Münster and Meibom (2010) argue that use of ESA is relevant for identification of affected energy production as it makes it possible to identify the combination of affected technologies in the short-as well as long-term. However, the ESAs used in reviewed LCAs were in most cases performed for other purposes than serving as LCI input. Thus, there is a risk that the ESA is outdated (Merrild et al., 2008) or that the context in which the ESA was performed not is entirely relevant for the system investigated in the LCA (Eriksson et al., 2007).

Differently from electricity production, substitution of thermal energy often depends on local conditions and production capacities connected to a specific district-heating network (Fruergaard et al., 2010). It could therefore be argued that case specific existing production capacity should be assumed affected in the short-term. The review also shows that local or national average technologies are used as representation of affected technologies for substituted heat in several studies (Boldrin et al., 2011; Turconi et al., 2011; Tonini and Astrup, 2012; Carlsson et al., 2015). In the long-term perspective, Fruergaard et al. (2010) argues that heat production from waste will contribute to phasing-out fossil fuels, a reasoning applied by Tonini and Astrup (2012), who identified different affected technologies for long- and short-term scenarios, exclusively substituting fossil fuels in the long-term horizon. However,

Table 1
Presentation of investigated studies and key data. MSW = Municipal Solid Waste, OFMSW = Organic Fraction Municipal Solid Waste, AD = Anaerobic Digestion, CHP = Combined Heat and Power.

Reference	Country	S	Treatment alternatives
Boesch et al., 2014	Switzerland	MSW	Incineration with metal recycling
Boldrin et al., 2011	General	OFMSW	Windrow composting[a]; Tunnel composting[a]; AD one stage wet[a]; Combined technology[a] (AD + Composting)
Carlsson et al., 2015	General	Food waste	Wet AD[a]
Cimpan et al., 2015	Denmark	MSW	Biowaste pre-treatment and digestion + biogas CHP[a]; WtE CHP
Eriksson et al., 2009	Sweden	Plastic waste	Incineration with CHP; Landfilling
Eriksson et al., 2007	Sweden	MSW	Incineration with CHP; Combustion of biomass with CHP; Recycling; Landfilling
Feraldi et al., 2013	US	Scrap tires	Mechanical recycling; Co-combustion at cement kiln
Fruegaard and Astrup, 2011	Denmark	MSW	Co-combustion; AD[a]; mass burn incineration
Habert 2013	EU	Industrial waste (blast-furnace slags and coal combustion fly ashes)	Co-combustion in cement kiln
Hamelin et al., 2011	Denmark	Separated animal slurry	Field spreading as fertilizer[a]; Decanter Centrifuge with Polyacrylamide[a]; Screw Press[a]; Screw Press and Pellets Production[a]
Hamelin et al., 2014	Denmark	Pig manure	AD[a]
Hums et al., 2016	US	Grease trap Waste	Conversion to biofuel; Landfilling
Johnson et al., 2013	US	Aluminium	Remelting with primary aluminium
Karlsson et al., 2015	Sweden	Faba bean	Drying + milling; Biorefinery[a]
Kimming et al., 2011	Sweden	Agricultural residues	AD[a] with CHP; Conversion to ethanol
Kua 2015	Singapore	Steel slag	Aggregate in concrete
Manfredi et al., 2011	General	MSW fractions: organic, paper, plastic, aluminium, glass	Landfilling; Recycling; Incineration or Composting[a]
Millward-Hopkin et al., 2017	UK	Fly ash from coal combustion	Use in cement production; Landfilling; Prevention
Mü; nster and Meibom 2010	Denmark	Mixed waste; organic waste; RDF	Incineration; Co-combustion; AD[a] with CHP; Gasification
Martinez-Sanchez et al., 2016	Denmark	Food waste	Incineration; Co-digestion[a]; Fodder treatment
Melamu and von Blottnitz 2011	South Africa	Sugar mill bagasse	Cellulosic ethanol biorefinery
Merrild et al., 2008	Denmark	Waste paper	Recycling; Incineration
Merrild et al., 2012	Denmark	MSW fractions: paper, cardboard, plastic, steel, aluminium, glass	Recycling; Incineration
Salemdeeb et al., 2017	UK	Food waste	Pig feed (wet/dry); AD[a]; Composting[a]
Schmidt et al., 2007	Denmark	Waste paper	Recycling; Incineration; Landfill
Sevigné-Itoiz et al., 2014	Global	Aluminum old scrap	Recycling
Sevigné-Itoiz et al., 2015a	Spain	Waste paper	Recycling
Sevigné-Itoiz et al., 2015b	Spain	Plastic waste	Mechanical recycling; Recycling to RPL; Incineration with CHP; Landfill
Sørensen and Wenzel, 2014	Denmark	Bedpans	Recycling; Incineration
Styles et al., 2016	UK	Fodder and food waste	AD[a] with CHP; Composting[a]; Incineration; Landfill
Tonini and Astrup 2012	Denmark	MSW	Enzymatic treatment in a biorefinery
Tonini et al., 2013	Denmark	OFMSW	AD[a] + Composting[a] (with CHP); Incineration with CHP; MBT + AD[a] or Composting[a]; Bioreactor landfilling; Conventional landfilling
Turconi et al., 2011	Denmark and Italy	MSW	Incineration
Turk et al., 2015	Generic	Fly ash, steel slag	Aggregate in concrete
Vázquez-Rowe et al., 2013	Luxembourg	Maize	Conversion to biomethane
Zink et al., 2014	US	Smartphone	Traditional refurbishment, Repurposing using battery power, and Repurposing using portable solar power

[a] Recycling of nutrients was assessed in the study.

the more fossil energy is substituted, the more likely is also the chance of non-fossils becoming affected by SWM systems in the future. This perspective was adopted by Fruegaard and Astrup (2011). As a result, combustion of biomass (wood and straw) was identified as the long-term affected technology.

The time horizon can be of large importance to the identification of affected energy technologies (Weidema et al., 2009). Even so, less than 40% of the reviewed studies present information of assumed time horizon for identification of affected energy technologies. Amongst these, a long-term perspective was applied in a majority of the studies. As stated by Weidema et al. (2009), even the effect of small, short-term changes are seldom isolated to the short-term perspective, since each individual short-term purchase decision will contribute to the accumulated trend in the market volume, which is the basis for decisions on capital investment, and thus long-term changes, in free market situations. Thus, a long-term perspective could be of larger relevance in this type of studies. In five studies, short-as well as long-term horizons were applied and compared.[3]

[3] It should be highlighted that different authors give different definitions to the terms short- and long-term. According to the European Commission (2010), short-term should be regarded as up to 5 years from present date. Tonini and Astrup (2012), use 15 years and Feraldi et al. (2013) use 10 years.

Table 2
Methods for identification of affected processes in reviewed studies. Sub-categories within each method are denoted with capital letters.

Method	Characteristics	Number of studies			Implication for CLCA modelling
		Power	Heat	Other[a]	
Reference to Weidema et al. (2009)	Identification of most and least competitive technology, when demand is increasing/decreasing respectively. Disregards constrained suppliers.	12	0	4	Reflecting consequences of studied decision, based on market logics.
Reference to Mathiesen et al., 2009	Fundamentally different affected energy technologies, including both fossil and renewable, should be considered (only relevant in relation to energy provision).	4	1	0	Reflecting the widest range of potential outcomes, ignoring market logics.
Statistics	A (Power and heat): Currently most common technologies (regional, national or local boundaries), based on recent statistics. B (Power and heat): Identifying the most expanding fuel in recent years. C (Power and heat): Identification of the fuel with historically highest/lowest cost. D (Power): Identification of principal fuel accommodating annual fluctuations over the last years.	4	9	9	A Ignoring consequences of studied decision on current situation. B, C and D Ignores that historic trends can be changed through political decisions, constrains etc.
Projections	A (Power): Projections of new installation capacity or national energy plans. B (Heat): Planned infrastructure projects (i.e. planned gas pipeline)	7	1	0	Projections commonly built on currently most competitive technology. Can include market logics or be based on historic trends. Plans can be less relevant in the long-term and have shown not to be representative for actual developments.
Energy system modelling	Identification of combinations of affected technologies in the short- as well as long-term, based on optimization of investments as well as production of electricity and heat, considering storage, transmission, prices and political decisions.	4	3	0	Enables modelling of short-term energy storage, price and policy dynamics over the timeframe of relevance. The context in which the ESA was performed might not being entirely relevant for the system or time horizon investigated in the LCA.
Other	- Experts opinion - Reasons of simplicity. - Creating optimized conditions for incineration vs recycling	3	2	2	Not related to the decision in question of the study.
Not stated	Choices were not motivated	10	9	1	Decreases the robustness of gained results.

[a] Recycling techniques, virgin plastics, metals, glass and paper, wood, pulp, mineral fertilizers and fuels used in the transport sector.

Contrary to previous recommendations for consequential modelling (Weidema et al., 2009; Mathiesen et al., 2009), substituted electricity was in several studies modelled as current national or regional average grid mix, based on recent statistics (Manfredi et al., 2011; Zink et al., 2014; Salemdeeb et al., 2017). Use of current grid-mix supposes that all technologies will respond in the same extent to changes in electricity demand, and includes also suppliers not affected by the change in focus of the study, why this approach should be avoided in CLCI (Hamelin, 2013).

3.2.2. Identification of affected materials

Material recycling of a product may lead to avoided production of primary material or avoided production of recycled material in another life cycle. According to Ekvall and Weidema (2004), the avoided production depends on the price elasticity of supply of collected material for recycling and on the price elasticity of demand for the recycled material in question. Assuming that the supply is completely price inelastic leads to the approximation that recycled material replaces 100% primary material. Assuming that the demand is completely price inelastic leads to the approximation that recycled material replaces 100% recycled material from other life cycles. It is obvious that this will have a large impact on results from LCAs of SWM systems including material recycling. At the same time, prices of recycled materials are commonly linked to the price of primary materials, which can fluctuate with economic cycles etc. (Frees, 2008). Thus, estimations of price elasticity may be uncertain and vary with the time horizon of a study.

The present review shows that the most common approach is assuming that the supply is completely price inelastic. This leads to the approximation that recycled material replaces 100% primary material of the same kind. According to Frees (2008), an argument for this approach is that it could be valid for materials for which there are no hinders for recycling into the same material and where there is a growing demand. However, a thorough analysis of the material markets is seldom presented in reviewed studies. Thus, this approach could be criticised for not taking into account that part of the recycled material may also replace recycled material from other systems, resulting in overestimations of environmental benefits from material recycling.

In general, it was assumed that the use of recyclables in manufacturing of new products would not influence the market situation for the product in question (Merrild et al., 2012). Production technologies for substituted products as well as recycling technologies where, with a few exceptions (Hamelin et al., 2014; Sevigné-Itoiz et al., 2014, 2015a,b), thereby commonly modelled as global or regional averages, using recent statistics. Differently from the case of district heating, materials are commonly sold on a global market, and the type of technology and electricity used in the production process by different suppliers can be of large relevance for overall environmental impacts from the material (Merrild et al., 2008; Sevigné-Itoiz et al., 2015b). Even so, different from in the case of energy, very few studies compared different choices of technologies affected by material recycling.

Only in cases where handling of one single type of waste was investigated (i.e. aluminium, paper, plastics or biowaste), did authors perform a systematic identifications of the affected virgin materials substituted through material recycling. In all these cases, authors refer to the methodology presented by Widema (2003 and 2009) for the identification of affected activities. However, also in these studies, the identified affected activities are at some point modelled as average processes. As an example, Sevigné-Itoiz et al. (2015a) identifies Brazilian bleached hardwood kraft pulp (BHKP) as the product affected by increased paper recycling in Spain. However, this process was modelled as an average process. However, according to Weidema et al. (2009) it could be assumed that the BHKP actually affected would be the one produced in the least efficient plants, reducing the demand for marginal rather than average electricity. Thus, even after an identification of affected processes, further effort is needed in order to provide relevant modelling of these processes and ensure overall consistency. In many reviewed studies this is not done, potentially due to lack of data or possibilities to adjust datasets representing average production technologies.

As stated by Hamelin (2013), the main problem with use of average data is that it includes suppliers that are not affected by the decision under study. The historical data used in construction of average data can be of higher or lower relevance in CLCA. For materials where environmental performance in extraction and production is very similar globally, the use of electricity is low and technical advances are unexpected, the use of global average data might be relevant also in consequential modelling. However, this is not the case for many materials. As an example, Merrild et al. (2008) chose to present several different virgin pulp technologies as potentially affected by increased paper recycling, showing that the choice of virgin technologies influenced the result of the LCA, mainly due to large variations in energy use. Sevigné-Itoiz et al. (2014) clearly show the large influence the choice of average and marginal data in modelling of substituted materials can have on overall results. According to their results, the avoided GHG-emissions related to aluminium recycling could be estimated to 6158 kg CO_2-eq./ton scrap collected when using average input data, while considering global markets and results marginal electricity in aluminium production result in avoidance of 18100 kg of CO_2-eq./ton of scrap collected. The difference is an effect of the identification of the actually most competitive global suppliers of virgin aluminium (including bauxite mining, alumina and smelting) and the marginal electricity mixes considered for respective supplier.

3.3. System boundary setting

The object of the consequential LCI is to include what is affected by a change in the use of a product or a service in the investigated life cycle (Ekvall and Weidema, 2004). Thus, unit processes are included in the system to the extent that they are expected to change as a consequence of a change in demand for the product/service in question (Weidema et al., 2009).

According to Ekvall and Weidema (2004), the boundaries of the system investigated should ideally be defined at the point where the consequences are so small, or the uncertainties so large, that further expansion of the boundaries will yield no information that is significant for any realistic decision. One problem with this recommendation is that it to some extent demands that the system initially must be expanded over this point, in order to identify to what extent the expansion renders significant information or not. The present review shows that previous recommendations on system boundary setting in CLCA commonly result in inclusion of processes with physical connection to the product/service in focus of the study, and energy and/or material substituted by goods generated through the investigated waste treatment (Fig. 1).

However, a different approach was chosen in some of the studies, including considerations of lost alternatives, fate of materials avoided through recycling, fate of surplus treatment capacity and lastly, rebound effects. The reasoning behind and implications of including these aspects in CLCA of SWM are discussed below.

3.3.1. Consideration of lost alternatives

The change-oriented nature of consequential modelling requires considerations of any variation in the current system resulting from investigated changes. Thus, the extent that the investigated use of waste material in our system results in reduced utilisation of other treatment alternatives is of relevance. As an example, using source-segregated household food waste for production of car fuel through anaerobic digestion in a system where this waste currently is incinerated with energy recovery, requires an assessment of the environmental loads avoided through reduced incineration. In addition, the affected processes needed for substitution of the electricity and heat no longer produced from food waste incineration could be considered. Hamelin et al. (2014) refer to goods no longer provided by the system after the investigated change as “lost alternatives”, while Salemdeeb et al. (2017) use the term “counterfactual waste management processes”, referring to the treatment alternatives from which waste is diverted. Styles et al. (2016) use the term “indirect effects”, acknowledging that, in some occasions, the alternative is not treatment, but rather use as i.e. fodder.

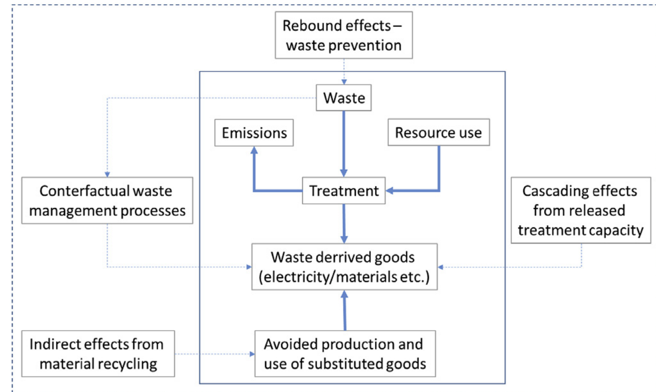

Fig. 1. Generic representation of the system boundaries most commonly applied in reviewed studies (within full line). Additional processes less commonly assessed within the studies are represented as boxes within the dotted line. Arrows represent a physical or economic connection between processes (boxes).

Counterfactual fates of waste handling are investigated by several authors (Eriksson et al., 2007; Tonini and Astrup, 2012; Feraldi et al., 2013; Hamelin et al., 2015; Styles et al., 2016; Salemdeeb et al., 2017). Styles et al. (2016) show that anaerobic digestion of brewery and bakery wastes increase GWP burdens from the system when these are diverted from current use as feed, if assumed that this increases the demand for feed produced from wheat. Tonini and Astrup (2012) argues that increased recycling of plastics and use of biomass for biogas-production in a biorefinery concept occurs at the expense of combustion of municipal waste with energy recovery. This is modelled as an increased need for electricity and heat provision from other sources, that will burden the investigated biorefinery system (Tonini and Astrup, 2012). However, the authors also consider benefits from reduced emissions from waste incineration as well as disposal of ashes and pollution control materials related to current handling (Tonini and Astrup, 2012). Counterfactual waste handling could also be relevant for secondary waste streams. Millward-Hopkins et al. (2017) investigated the effects from material recycling of low quality ashes from combustion of solid residual waste, compared to use of fly ash from coal incineration plants. According to the authors, an increased incineration of waste should be burdened by increased emissions of GHG, as it genders a need for import of high quality fly ashes from elsewhere, while low quality ashes are disposed of in the UK.

3.3.2. Indirect effects from material recycling

Increased recycling of waste materials can decrease the demand of virgin materials such as metals, plastics, paper and mineral fertilizers on the market. Recycled materials can also replace i) recycled material from other systems, ii) completely different types of material or iii) no material at all (Ekvall and Weidema, 2004). Amongst reviewed studies, it is commonly assumed that recycled material substitutes virgin material to 100%. Although this approach might be attractive, it ignores other possible applications as well as any other consequences from material recycling. The issues lifted here could be applied to all studies in Table 1 where recycling of materials were assessed.

Substitution of virgin materials with recyclables will reduce also the supply of any by-products generated in the production of virgin materials. However, the demand of these by-products might not decrease, and it is relevant to discuss to what extent alternative provision of these materials should be included in the consequential modelling. As an example, it is well known that several metals are mined only as by-products. A recent investigation of 62 different metals and metalloids show that more than 60% have companionality (i.e. the production is dependent on the mining of host metals) greater than 50% (Nassar et al., 2015). The supply of gallium, indium, selenium and tellurium, rely largely on mining ores for the production of metals such as aluminium, copper and zinc. Based on this, it could be argued that any changes in the supply of by-products from for example copper production should be assessed in CLCA of systems where the demand of virgin copper is changed. Such aspects were however not included in any of the reviewed studies.

Paper recycling is a special case in waste CLCA, where the consequence of avoided use of wood is much debated. Reviewed studies assessing paper/cardboard waste assume substitution of virgin paper production through material recycling of waste paper, releasing wood and/or land for other uses. Whether this should be accounted for or not depends on the scarcity of forest area/wood and may thus depend on the time perspective applied in the study (Villanueva and Wenzel, 2007). In some cases, biomass saved through paper and cardboard recycling is assumed to remain in the forest (Eriksson et al., 2007; Sevigné-Itoiz et al., 2015a,b). Others include the use of saved wood for energy purposes (Merrild et al., 2012; Schmidt et al., 2007). The latter is based on the assumption that wood in the future becomes a priority fuel/raw material of limited availability, due to energy policy decisions or fossil fuel scarcity (Villanueva and Wenzel, 2007). The alternative use of wood is a key issue when comparing recycling of paper and cardboard with incineration (Villanueva and Wenzel, 2007). The magnitude of the importance depends on the technology for production of electricity and heat assumedly replaced by wood, when this material no longer is used for paper production due to increased recycling. Merrild et al. (2012) assumes that coal is substituted by wood combustion, which results in vast benefits from paper recycling, while benefits according to Schmidt et al. (2007) are smaller, as natural gas assumedly was substituted by wood in their study. Fruergaard and Astrup (2011) describe biomass fuel (wood and straw) substituted by waste incineration as a non-constrained residual resource. If biomass substituted by waste-to-energy technologies is considered unconstrained, it implies that this material will not be used elsewhere to offset fossil fuels. The effect of biomass being constrained is the equivalent to assuming fossils as affected fuels. However, also residual biomass can provide several benefits to the systems they are recovered from, such as nutrient recovery and carbon storage in soils (Cherubini and Ulgiati, 2010), previously addressed in CLCA studies of bioenergy systems (Kimming et al., 2011; Ahlgren et al., 2013). Such aspects were not considered in any of the studies reviewed. If included, this could increase benefits from waste-to-energy treatment in scenarios where biomass was identified as affected technology.

3.3.3. Fate of surplus treatment capacity – cascading effects

Increased use of one treatment alternative can, in the short term, release capacity in other treatment alternative plants. As an example, increased material recycling of waste previously combusted may imply that waste incineration plants instead take in other combustible waste. This introduces the question of what type of waste management is most likely to react to this increased demand for combustible waste. According to Cimpan et al. (2015), refused derived fuel (RDF) currently landfilled in the UK is the affected waste treatment in the EU. The reason is a current lack of incineration capacity and high domestic gate fees (Eunomia, 2013). Waste management systems releasing incineration capacity could thereby be credited emissions circumvented through avoided landfilling. According to Cimpan et al. (2015), inclusion of avoided landfilling of RDF in the UK increases climate benefits from investigated Danish source segregation systems by between 15 and 30%. Villanueva and Wenzel (2007) state that alternative use of treatment capacity should be included in LCA of material recycling. However, the consequence in the long term is not avoided treatment through capacity release, but avoided construction of new capacity, as the market over time will adjust for a trend towards an increased material recovery.

3.3.4. Rebound effects

Previous reviews have shown that although prevention according to the EU waste framework directive (EU, 2008) should be the preferred alternative for waste management, this alternative is seldom included in LCAs of SWM-systems (Laurent et al., 2013). Prevention of food waste was however investigated by Martinez-Sanchez et al. (2016). Authors also included potential rebound effects associated with the affected (marginal) consumption when there is a difference in costs to consumers between alternative scenarios providing the same service. In their study, this was exemplified by increased consumption of other goods through less spending on food, when food waste is prevented on end-consumer level. According to their results, food waste prevention generates high welfare gains as more services/goods could be consumed with the same income, but this consumption could result in negative environmental impacts if additional money is spent on goods that are more environmentally damaging compared to the (prevented) food. However, saved money could also be used in products that are less polluting, but costlier compared to the pre-prevention situation. This makes assumptions of the consumption affected by the change of main importance.

In comparative LCAs, it is of key importance to maintain the same functional output from all compared systems. Thus, it is necessary to compensate any savings in monetary spending by including the consequences of a corresponding increase in marginal spending, and vice versa for an increase in cost (Brandão and Weidema, 2014). In this context, it is relevant to highlight that different treatment alternatives for SWM will result in different costs for society. However, such rebound effects are commonly not lifted in previous research in this field.

3.4. Handling of uncertainties

As stated by Björklund et al. (2003), the reliability of LCA is affected by dependence on a variety of data from different sources and more or less subjective methodological choices. When compared to attributional modelling, the uncertainties involved in the identification of affected processes have been identified as yet another source to uncertainties in CLCAs (Finnveden et al., 2009). Other authors argue that the error is greater in attributional compared to consequential LCA (Weidema et al., 2009). This statement implies that the use of historical average data in ALCA is a worse estimation of the actual environmental impacts related to a specific product or service, than the approach taken when using consequential modelling. As stated by Ekvall and Weidema (2004), describing the consequences of decisions means facing the general challenge of futures studies, as the future is inherently uncertain, and the actual future consequences of decisions are highly uncertain. Thus, such uncertainties can commonly not be reduced much further, but at least illustrated.

Several authors have suggested the use of different scenarios based on various assumptions, which together can illustrate how consequences might change under different market situations (Ekvall and Weidema, 2004; Zamagni et al., 2012). The present review shows that the approaches taken by researchers in order to investigate the influence from uncertainties commonly is rather simplified.

The most common approach amongst reviewed studies is to investigate the sensitivity of gained results in relation to the identified affected technologies for energy production. More than half (56%) of the reviewed studies performed such sensitivity analyses, and in several of these, more than one alternative technology were investigated. Mathiesen et al. (2009) recommended use of sensitivity analyses including fundamentally different technology alternatives for energy production, and this was the approach was used by most authors. Coal in power plants and coal or oil for heat production were commonly used as a worst-case scenario, compared to natural gas, wind power or use of biomass (Bosch et al., 2014; Fruergaard and Astrup, 2011; Carlsson et al., 2015; Tonini et al., 2013). In one case, the technologies identified as affected by increased need for electricity production through changes in the waste management system, were compared to current average data (Sevigné-Itoiz et al., 2015b). When waste-derived goods are used in the transportation sector, production and use of either diesel or gasoline where identified as affected processes in all cases. Only in one study was this assumption investigated through a sensitivity analysis, assuming substitution of a renewable fuel (biodiesel) (Fruergaard and Atrup, 2011).

Outcomes from these sensitivity analyses usually show a high sensitivity to the selection of affected technology for energy provision on overall contribution to GHG-emissions from different waste treatment alternatives. In general, selection of less polluting technologies discredits waste-to-energy technologies and promotes recycling (Manfredi et al., 2011; Tonini et al., 2013; Carlsson et al., 2015).

4. Conclusions

Consequential lifecycle assessments (CLCA) of solid waste management (SWM) systems is a growing area of research. Previous studies commonly do not present relevant justifications of the choices made in identification of processes and technologies affected by studied changes. At the same time, the present review clearly demonstrates a strong relation between the selection of affected energy production technology and overall GHG-emissions. Thus, increased transparency is needed in the identification of the technologies for energy production affected by changes studied.

Materials substituted by waste derived goods are in reviewed studies commonly represented by global or regional/national average data. This can cause biased results in studies where material recycling is compared to waste-to-energy. Thus, increased attention is needed to guarantee coherent consequential modelling of all parts of the investigated system in order to avoid biased comparisons between different waste management alternatives. The review also shows that increased attention should be directed to the investigation of uncertainties related to identification of material flows affected by studied changes.

In terms of system boundary setting choices, four main areas were identified where current practice is diverse and where choices could have significant influence on overall results; counterfactual waste management, fate of materials avoided through material recycling, cascading effects and rebound effects. These aspects can all be of relevance to address in CLCA of SWM systems, but were included very scarcely in reviewed studies. This indicates that CLCA currently is reduced to choices made on process modelling level, while there is a need to pay more attention to choices made on system modelling level.

The large influence from choices made in selection of substituted products/energy carriers calls for increased collaboration between the LCA-community and economists, with the aim of increasing the quality of economic modelling of substitution processes. With this said, it should be remembered that markets for different products rapidly can be altered by political measures, such as trade tariffs etc. creating inherent uncertainties that cannot be reduced. Increased transparency and justification of made choices, as well as of the effect of the same on overall results, is however a first step towards an improved interpretation and usefulness of CLCAs as decision support tool.

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Appendix A. Supplementary data

Supplementary data related to this article can be found at https://doi.org/10.1016/j.jclepro.2018.08.038.

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