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Heimersson, S., Svanström, M., Ekvall, T. (2019). Opportunities of consequential and attributional modelling in life cycle assessment of wastewater and sludge management. *Journal of Cleaner Production*, 222: 242-251.
<http://dx.doi.org/10.1016/j.jclepro.2019.02.248>

N.B. When citing this work, cite the original published paper.

Opportunities of consequential and attributional modelling in life cycle assessment of wastewater and sludge management

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ABSTRACT

Despite general agreement on the importance of adjusting each life cycle assessment (LCA) to its goal, the methodological choices in previously published LCAs on wastewater and sludge management systems are surprisingly similar, even when the information sought in the studies most likely differ. We argue that the potential of LCA may not currently be fully utilised, partly due to particular methodological challenges arising in both attributional and consequential LCAs for this type of systems. By developing the theory for handling of allocation problems in attributional LCAs, and by elaborating on the different possible foreseeable consequences in consequential LCA, we aim to facilitate both attributional and consequential LCAs, and to show the importance of this choice for a specific wastewater and sludge management system.

We introduce and apply a distinction between physically and legally joint processes as basis for the allocation of resource use and emissions in attributional LCA, and suggest that, when the joint process is not driven by commercial interests, allocation factors could be identified and quantified through stakeholder priorities. In consequential LCAs, the substitution depends on the subjective view on what consequences are foreseeable, for example based on short- or long-term considerations. All of these modelling aspects can, as our case study illustrates, affect the LCA results.

KEYWORDS

Accounting LCA, allocation problem, change-oriented LCA, multifunctional process, sewage treatment

HIGHLIGHTS

- An LCA should ideally be consistently attributional or consequential
- Such methodological consistency is possible in LCAs of sludge management systems
- The functions of a process can be legally joint
- Stakeholder priorities can be used as a basis for allocation
- Foreseeable consequences can be short-term or long-term

1. INTRODUCTION

Life cycle assessment (LCA) has been used since the 1990s for assessing the potential environmental impacts of wastewater and sewage sludge management (Tillman et al., 1998).

This has, to date, resulted in more than 100 papers published in the scientific literature (see, e.g., Corominas et al. (2013) for a review). These systems often perform multiple functions, such as wastewater treatment (WWT) and resource recovery from the sludge, e.g. in the form of biogas production and/or utilisation of nutrients by means of spreading on arable land. An LCA that aims to estimate the environmental impacts of one of the functions, for example WWT, inevitably encounters an allocation problem: what part of the environmental impacts of the system should be assigned to the function investigated?

An allocation problem can be managed in various ways, and the appropriate approaches to allocation have been debated for decades. The international standard on LCA states that allocation problems should be reduced or avoided when possible, by dividing the multifunctional process into two or more subprocesses, or by expanding the system investigated to include also the functions of the co-products (ISO, 2006). When allocation cannot be avoided, the environmental impacts of the multifunctional process should be partitioned between its functions, preferably in a way that reflects how the impacts are affected by changes in the quantities generated of the different functions.

The international standard can be interpreted in different ways. In line with, e.g., Heijungs (2014) and Sandin et al. (2015) we find it useful to distinguish between system expansion and substitution, where the latter can also be called system expansion by substitution. Pure system expansion means that the study is expanded to include all functions of the multifunctional process. This is clearly consistent with how system expansion is described by ISO (2006) and makes the study an LCA of two or more functions. Substitution, in contrast, can generate an LCA of a single function by subtracting the environmental impacts of the avoided, alternative production of the other function(s). This is also often considered consistent with the international standard, because it was how the concept of system expansion was explained in an informative annex to the original LCA standard (ISO, 1998).

The allocation procedure in the international standard does not distinguish between different purposes of the LCA; however, an early consensus document stated that the choice of allocation method should depend on the scope and purpose of the study (Consoli et al., 1993). Consensus soon grew in the LCA community that there are two types of LCA: one where it is appropriate to avoid partitioning through substitution and one where allocation problems should be solved through partitioning (Tillman, 2000). The two types of LCA were presented under many different names (Ekvall, 1999), but eventually became known as consequential LCA (CLCA) and attributional LCA (ALCA) (Curran et al., 2005).

The methodological debate still continues (Schrijvers, 2017), but now with an emphasis on how ALCA and CLCA should be interpreted (e.g. Ekvall et al. (2016)) and when they should be applied. Finnveden et al. (2009) state that “attributional LCA is defined by its focus on describing the environmentally relevant physical flows to and from a life cycle and its subsystems” and that “consequential LCA is defined by its aim to describe how environmentally relevant flows will change in response to possible decisions” (Finnveden et al., 2009).

The choice between ALCA and CLCA affects not only the approach to allocation problems, but also the choice of input data (Tillman, 2000). An ALCA aims to describe the life cycle and its subsystems. For this purpose, the production of a good in the background system is modelled using average input data. Average data represent the full mix of technologies used for producing the good. A CLCA, in contrast, aims to model that which is affected by the production and use

of the product, or by a change in the product life cycle. For this reason, many parts of the model should be based on marginal data. These are data that represent the technology actually affected by a marginal change in the total production volume of the good.

An ALCA will yield different numerical results compared to a CLCA, because the results are answers to different questions. Simply put, an ALCA estimates what share of the global environmental impact belongs to the product investigated, while a CLCA estimates how the global environmental impact is affected by the production and use of the product (Weidema, 2003). Hence, the results of an ALCA and a CLCA carry different information. If an LCA mixes elements of ALCA and CLCA approaches, the results may not be meaningful because they are not an answer to a well-defined question (cf., e.g., Brander and Wylie (2011)).

Heimersson et al. (2016) reviewed 62 LCAs of wastewater and sludge management published between 2004 and 2016. They found that most of these studies are not explicitly presented as either attributional or consequential. Most of the studies also apply a mix of attributional and consequential methodology: partitioning is typically avoided through substitution, but the substituted production is in most cases modelled using average data. Average data is also often used to model the background system. This mix of methods means that the LCA results will not carry clear information. They will not be an accurate estimate of what share of the global environmental impact belongs to, e.g., the WWT. Nor will they be an accurate estimate of how the WWT affects the global environmental impact.

The purpose of this paper is to contribute to a more consistent use of attributional and consequential methodology in LCA of the treatment of wastewater and the resulting sludge. This would result in LCAs that better answer the questions posed by the commissioners, and eventually enable wastewater and sludge treatment systems to be managed in a way that to a higher extent promotes cleaner production and sustainability. Attributional and consequential methodology in general has been described above and by, for example, Ekvall and Weidema (2004) who elaborated on the system boundaries and allocation issues in CLCA. We add to this literature by focussing on the allocation problems that are specific to environmental assessments of wastewater and sludge management. Heimersson et al. (2017) concluded that the allocation problem in such a system can be particularly challenging. In an ALCA, a challenge is to find a good basis for defining the partitioning factors; in a CLCA, the challenges include identifying what type and quantity of products are substituted and what production processes are affected. Our contribution includes an elaboration of how key concepts such as “joint processes” and “foreseeable consequences” can be interpreted and used in these studies. We also discuss alternative ways of defining factors for partitioning in an ALCA.

We apply the methods in a case study where we perform both ALCA and CLCA on wastewater and sludge management in Gothenburg, Sweden. This allows us to illustrate the effect of the choice of approach for a real-world case. The case study system includes a common type of wastewater treatment plant (WWTP) with nitrogen and phosphorus removal, treating mainly municipal wastewater. The sewage sludge is anaerobically digested and the digestion residues are, after dewatering and storage (in order to ensure sufficient hygienisation), used for agricultural purposes, in order to recover nitrogen and phosphorus. The system is thus clearly multifunctional: it performs treatment of the wastewater before release to the sea, production and use of digester biogas, and production and use of sludge as fertiliser on arable land.

2. METHODOLOGICAL ASPECTS

This section contains our methodological rationale. We summarize the scientific discussion on the applicability of ALCA and CLCA and present our position in this matter. We discuss the different approaches to the allocation problems that occur in an LCA of WWT. We also discuss concepts that need to be clearly understood to apply these approaches in practice.

2.1 Applicability of attributional and consequential LCA

As stated above, an ALCA and a CLCA generate different information. The relevance of this information has been debated among LCA researchers for almost as long as the allocation problems. Wenzel (1998) argued that LCA is only worthwhile if it directly or indirectly guides decisions and that CLCA generates the most relevant information for guiding decisions. Tillman (2000) and Lundie et al. (2007) argued that CLCA should be used for decision-making but ALCA when no specific decision is at hand. Lundie et al. added that the use of CLCA is appropriate only when the insights gained from it outweigh the uncertainties in the consequential modelling. Plevin et al. (2014) acknowledged the inherent uncertainty in the actual environmental consequences of decisions, but argued that scenario-based CLCA is still the most appropriate methodology for policy-makers because it facilitates robust decision-making – a claim that was challenged by Suh and Yang (2014) who pointed at the large difference between ideal and real-life CLCA.

Ekvall et al. (2005) stated, based on Tillman (2000) and Curran et al. (2005) that ALCA and CLCA give complementary information: ALCA about the environmental burdens of the life cycle, and CLCA about how possible decisions affect the environment. They argued that both types of information can be used for general learning purposes and for decision-making, but that both methodologies have technical as well as ethical limitations. Brander and Ascui (2015) stated that consequential methodology is appropriate for informing decision-makers that have the genuine objective of mitigating environmental impacts, but that attributional methods might be relevant to companies that are mainly concerned with regulatory liabilities based on attributional accounting. Weidema et al. (2018) argued that a socially responsible decision-maker must take foreseeable consequences into account, which implies the use of CLCA, but that information from an ALCA can be used as a complement.

Tillman (2000) argued that ALCA should be used for environmental product declarations (EPDs) and other marketing claims because it is more additive, less uncertain and easier to standardise and apply. Jelse et al. (2015) added that CLCA, because of its use of marginal data, is inconsistent with the idea behind EPDs: that it matters which of two competing products that is chosen. JRC-IEA (2010) recommended ALCA for all micro-level decision, i.e., for comparisons and decisions on specific products. Benetto (2018)¹ argued that scenario-based ALCA can be relevant also for policy-making, when policy-makers want knowledge about how the performance of the system they are responsible for depends on specific technology choices.

It is not the focus of this paper to analyse the many arguments for and against ALCA and CLCA. However, among the variety of views, we take a position close to Ekvall et al. (2005), Brander & Ascui (2015), and Benetto (2018), arguing that ALCA and CLCA can both be useful for learning as well as for decision-making on micro and macro levels. An ALCA generates information about the environmental burdens of the life cycle or system in focus without interference from the great uncertainties of marginal impacts in the background system, etc.

¹ Benetto, E., 2018. Personal communication. Environmental Research and Innovation Department, Luxembourg Institute of Science and Technology on May 23rd, 2018.

Such information is relevant to managers and policy-makers that can potentially be held responsible for the performance of the supply chain or system in focus by consumers, voters, employees, and/or their own conscience. The objective of an ALCA of WWT is to estimate what share of the global environmental impact belongs to the WWT. An ALCA of current or future municipal wastewater and sludge management can thus be relevant to policy-makers and civil servants responsible for the function of WWT.

A CLCA of WWT aims to estimate how the WWT and any changes in the process affects the environment. This information is relevant to policy-makers and civil servants responsible for the WWT, if they have the genuine objective to mitigate environmental impacts wherever they occur (cf. Brander and Ascui (2015)).

Modelling of WWT can also be part of an LCA of food products, both when sewage sludge is used in agricultural practices and as the end-of-life of the food that has passed our bodies. Both ALCA and CLCA can be relevant for decision-makers along the supply chain of the food product, depending on whether their objective is to reduce the environmental impact that belongs to the food product or to make changes in the supply chain that mitigate environmental impacts wherever they occur.

2.2 Approaches to the allocation problem

The allocation problem that occurs when an LCA includes a WWTP can in principle be dealt with through partitioning, system expansion, or substitution. Partitioning is the method typically used in ALCA (Tillman 2000). In an ALCA of WWT, this would mean that parts of the environmental impacts of the wastewater and sludge management system is assigned to the WWT. This is associated with three methodological challenges. First, the LCA practitioner must define the boundary of the system to be partitioned, i.e., identify what parts of the system are required for one of the functions only and what parts of the system that are required for two or more of the functions. The latter are in this paper called “joint processes”. Next, the physical flows to and from the joint processes must be divided into two categories: flows representing functions of the joint processes (waste treated and products generated by the processes), and flows representing impacts of the processes (energy and material used; waste and emissions generated). The latter are to be partitioned among the former. Third, it must be decided how to partition the environmental impacts, i.e., on what to base the partitioning. These three challenges are discussed in Subsections 2.3-2.5.

System expansion can be used in a comparative LCA of different options for WWT, if the compared options result in different quantities of e.g. biogas and/or nutrients. The system expansion then means that alternative production is added in the LCA model of the option with less functional output to make the two cases functionally equivalent. The result will not be an LCA of WWT, but an LCA of a basket of functions: WWT, production of biogas, and recovery of nutrients. The comparative LCA of WWT plus biogas production plus fertiliser production will be attributional if average data are used to model the alternative production, but it will be consequential if marginal data are used in the model. This paper, however, focuses on the LCA of a single function (WWT) and we will not discuss this type of system expansion further.

In a CLCA of WWT, substitution is the appropriate method (Tillman 2000). This implies that the CLCA of the function of WWT would include all of the multifunctional system. In addition, it should include the avoided production of e.g. energy carriers and fertilisers to the extent that they are replaced by the biogas and nutrients recovered in the system. This means that the allocation problem is avoided by substitution. In consequential studies, the identification of

“foreseeable” consequences is key to deciding on assumptions regarding products or services that are substituted. The challenge of deciding on the foreseeable consequences is discussed in Subsection 2.6.

2.3 ALCA: Defining joint processes

A process is physically joint with regard to its functions when the generation of multiple functions cannot be physically separated. Anaerobic digestion is an example of a physically joint process. It stabilises and hygienises sludge (i.e., the waste) from the WWTP. Simultaneously, it produces raw biogas. This process is in our system required for both these functions: treatment of waste from the WWT and biogas production. In addition, it prepares the sludge for nutrient recovery in agriculture.

Anaerobic digestion is physically separate from subsequent dewatering and the further management of the digestion residues. However, there are still strong ties between the processes. It is, in most countries, illegal to run a digestion process without dealing properly with the residues afterwards. In this sense, post-treatment is legally required for generating the functions of the digestion, i.e., waste treatment of sludge from WWT and production of biogas. We here introduce the concept of legally joint processes for processes in the system that would not be legal to exclude from the system without an adequate replacement. We thus have two different interpretations of the concept “joint process” to apply in our case study: physically and legally joint processes.

2.4 ALCA: Distinguishing functions from impacts

In an ALCA, the impacts of the joint processes should be divided between the functions of these processes. In the case of wastewater and sludge management, it can be difficult to determine whether a flow represents an impact or a function. The residue from anaerobic digestion is a sludge that contains useful nutrients but also potentially some hazardous substances. A definition is required to decide whether this sludge is a product (function) that should carry part of the environmental impacts of the digestion process, or a waste flow (impact) that should be partitioned between the waste treatment and the production of biogas. If the sludge from the digestion is considered a waste flow, the post-treatment may later turn it into a product flow. The definition should help us identify the point in the treatment where it becomes a product.

We here distinguish functions from impacts by the economic value of the flow. A flow with negative economic value is considered a waste even if it includes useful substances. If the economic value is positive, we consider it a product, even if it includes environmentally problematic substances. This distinction is in theory clear-cut. In practice, however, it can be difficult to apply because the economic value of the sludge is often close to zero and can shift over time between positive and negative, sometimes based on only a change in perception of key actors.

2.5 ALCA: Partitioning

The choice between different methods for partitioning has sometimes been described as arbitrary or at least subjective; however, an early consensus process concluded that partitioning should be based on causal relationships (Clift, 1994). Partitioning in proportion to the economic value of the products has often been advocated for ALCA, because the expected revenues from the products are often what drives the joint process (Guinée et al., 2004; Weidema, 2003). This means the (expected) economic value of the products are considered to reflect their share of the reason that someone invested in the process and continues to run it, and hence their share of the cause of the environmental impacts of the joint process.

Wastewater and sludge treatment, however, are not commercial processes in Sweden. They are not driven by expected revenues from the WWT service, from the digester biogas, etc. Instead, they are driven by societal and environmental needs for, e.g., reducing the impact on water recipients, minimising the net energy need of the plant or increasing the supply of renewable energy, and safeguarding phosphorus resources (see further description in Heimersson et al. (2017)). Economic valuation of these functions is thus not only challenging; we argue that economic value is less relevant as a basis for partitioning for these systems. We therefore need another way to estimate to what extent each driver contributes to the fact that the joint processes in the wastewater and sludge treatment system exist and are carried out.

The importance of the different drivers of the joint processes in the described system depends on a complex mix of facts and subjective perceptions and, hence, can vary between different actors. In such cases, one option could be to perform a Delphi process – a structured approach to find consensus among experts on some needed input, in this case on the relative importance of the different drivers of a specific process or system. Relevant stakeholders would then be seen as experts on their perspective. The process would, using the example of the anaerobic digestion as a joint process, aim at a relative quantification of the societal drivers behind sewage sludge treatment compared to biogas production. Delphi processes have previously been employed for other purposes in LCA: for example, for classifying the environmental impacts of wood products (Lipušček et al., 2010), or to develop environmental performance criteria for evaluation of desalination plants (Balfaqih et al., 2017).

2.6 CLCA: Foreseeable consequences

A CLCA should strive to account for the foreseeable consequences of a decision, for example the decision to treat wastewater. This decision will result, in our example, in sludge that can be used for producing biogas and for recovery of nutrients in agriculture. The biogas and nutrients are likely to replace other energy carriers and mineral fertilisers, respectively (Heimersson et al., 2017). A CLCA should account for these consequences, which means that partitioning is avoided through substitution.

The concept of foreseeable consequences, however, has subjective elements, just like the choice between partitioning methods. Different LCA practitioners and LCA commissioners might disagree on whether a consequence needs to be certain or likely in order to be included in the study, or if it is sufficient that the consequence is plausible. They can, of course, also have diverging views on whether a potential consequence is certain, likely, plausible or unlikely. They might also disagree on what the most likely consequences are – for example, if the biogas is likely to be directly combusted on site or upgraded and used as a vehicle fuel - if that is not dictated by the context.

What consequences are seen as likely or plausible can also depend on the time horizon of the study. If the biogas is upgraded for use as a vehicle fuel, it can in the short term only replace other sources of gas, such as natural gas. In the mid- to long-term, however, an increase in the availability of vehicle gas can contribute to an increase in the number of gas vehicles at the expense of vehicles using liquid fuel. With this longer time perspective, the biogas might instead replace diesel or gasoline. An earlier LCA has demonstrated the importance of the choice between replacing on-site energy production or natural gas in vehicles (Heimersson et al., 2017).

3. CASE STUDY

Before going into more detail on the modelling approaches that we apply in this study, we need to establish the details of the system that will be modelled as each system will potentially have different issues that need to be resolved.

3.1 Case study system

In order for this study to be of general interest, we have chosen a fairly common wastewater and sludge management set-up but as performed in a particular city in Sweden in order to provide a local context. The case study is based on data from the WWTP Ryaverket in Gothenburg, Sweden, operated by the municipal company Gryaab. The WWTP treats mainly municipal wastewater with some addition of industrial wastewater, for 980.000 person equivalents² (p.e.; in 2016), and is located close to the sea, where the effluent water is released. The treatment process includes a primary treatment and chemical phosphorus precipitation using ferric sulphate, followed by secondary activated sludge WWT. The mixed sludge from primary and secondary treatment is anaerobically digested, generating biogas that is subsequently upgraded to increase the share of methane, allowing it to be used as a vehicle fuel³ in trucks. The digester sludge is dewatered before end use. Several alternatives are possible for sludge end use, but in this case we assume the sludge is used on agricultural land, as such end use gives rise to challenging allocation problems, as described in the section on methodological aspects. Figure 1 shows the important aspects of the wastewater and sludge management system explored in this study. Inventory data on the foreground system is partly described in Heimersson et al. (2017). Additional inventory data on the WWT was gathered from the WWTP's environmental report from 2014 (Gryaab, 2014). Data on the background system was taken from Gabi Professional database 2016 or EcoInvent 2.0, except for the production of mineral fertilisers (Brentrup, 2015) and district heat, modelled as district heat in Gothenburg, based on Göteborg Energi (2014) and Gabi Professional database 2016.

The case study system has three main functions: it provides treatment of the wastewater (including management of the waste from the treatment), it produces energy (in the form of biogas) that is used as a vehicle fuel (and also deals with the waste from the digestion) and it provides nutrient fertiliser (and organic material) to arable land. The functional unit is in our case chosen to be the treatment of wastewater during one day; the WWT is thus here seen as the main function of the system, which is not the only possible option.

In the following, the considered system will be explored from attributional and consequential perspectives, ending up in four different ways to model the same system (varying how the joint processes are identified in ALCA and what the foreseeable consequences are considered to be in CLCA). For the comparability of results, we are modelling the system to include the same core elements in all modelling approaches, thereby illustrating the opportunities of different ways of modelling a system with more or less the same system boundaries. In a real situation, the different questions that an ALCA and a CLCA, respectively, are meant to answer may end up in very different system boundaries.

² 1 p.e. corresponds to a BOD7 of 70g per person and day

³ This is the actual local choice of biogas use but not the only way to use the biogas. In fact, it is common that most or all of the biogas is used on site for local heating needs and sometimes also electricity generation via combined heat and power. Surplus energy may then be sold, generating other allocation problems.

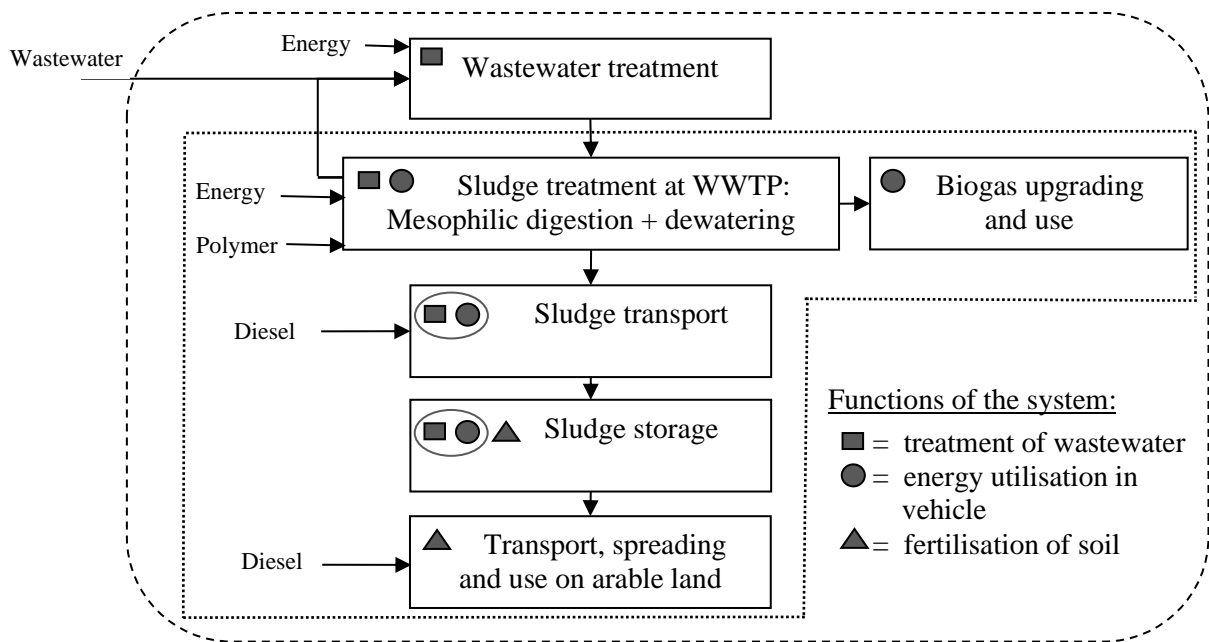


Figure 1. Case study system within dashed lines. Arrows represent physical flows. The grey symbols illustrate what functions are considered to drive the individual processes in the system when physical relationships are considered (applied in case study option ALCA 1). When two such symbols are circled, it indicates that the relative partitioning between those functions should be the same as in the preceding process. The dotted line illustrates the processes considered to be legally joint (applied in case study option ALCA 2).

3.2 Attributional modelling of the case study system

An ALCA assesses the average environmental consequences of the activities in a system, and assumes that surrounding life cycles are static, i.e., changes in the studied life cycle are not assumed to affect other life cycles.

Modelling of subsystems. When modelling subsystems connected to a life cycle, i.e., processes in the background system, in ALCA, Finnveden et al. (2009) advocate the use of average production data, i.e., data considered to reflect actual physical flows. For the purpose of the ALCA case study options, production of input materials such as energy and consumables were therefore modelled as Swedish average production, or where such data was irrelevant or lacking, EU27 average production. More specifically, electricity production was modelled as the average Swedish consumption mix (GaBi Professional database 2016).

Handling of multifunctionality. In the Theory section, two possible relationships were identified as useful bases for identifying joint processes for a system such as the one in the case study: physical and legal relationships. In both cases, the treatment of the wastewater (i.e., the processes where sludge is generated) can be considered to be done for the sole purpose of WWT. This function is the reason the WWTP was built in the first place and the WWT process does not generate any other directly useful products (functions). The need to allocate the impact from the remaining parts of the system between its three different functions, however, remains.

A situation in which subdivision of the remaining system is based on physical relationships is displayed using grey symbols in the process boxes in Figure 1. As can be seen in the figure,

three process boxes, the WWT, the biogas use and the spreading and use of sludge on arable land, have been entirely attributed to either of the system's three functions. The sludge treatment and subsequent sludge transportation are, however, considered to be joint processes between the WWT function and the biogas production and use function, as they are considered to be waste treatment of a residue from production of these two functions; the third function of the system, related to the nutrient content, has not yet appeared as no marketable product yet exists. During the sludge storage, however, the sludge is further hygienised and potentially given value as a marketable fertiliser and the process is therefore considered to be a joint process between all three functions. This is of course an assumption that is specific for this case study as there may be other situations when spreading of sludge in agriculture might not even be allowed (for example if the sludge is too polluted or if a national ban exists on spreading of wastewater sludge on arable land). For each joint process, the related impact has to be partitioned between its functions, which is discussed later. The use of a physical relationship to identify joint processes is hereafter referred to as case study option ALCA 1.

The second suggestion, to apply a legal⁴ relationship as the basis for subdivision, is motivated by three facts specific to the case study system: 1) sewage sludge can in Sweden not be landfilled as it contains too much organic material, and the WWTP therefore needs to dispose of the sludge in other ways, and one possible way is spreading on arable land; 2) sludge cannot be land applied without being properly hygienised, in this case by a combination of mesophilic anaerobic digestion and storage (which may not even be sufficient if future requirements are stricter; this is expected in Sweden); and 3) to produce biogas from sludge and utilise it as a vehicle fuel has in Sweden been driven by a political will to increase the share of biofuel usage in the transportation sector rather than a decision driven by market forces (Nordahl, 2013). The remaining system (i.e., everything except the WWT process) is therefore considered inseparable because of legal relationships and is thus handled as a black box joint process (see Figure 1) for which related impacts need to be partitioned amongst its different functions. The use of a legal relationship to identify joint processes is hereafter referred to as ALCA 2. The arguments above for, e.g., including the upgrading of the biogas in the studied system, can be compared to the theory behind the concept of 'allocation at the point of substitution' (see, e.g., Weidema (2018)). Both options for subdivision presented above leave joint processes for which related impacts need to be partitioned. As suggested in the Theory section, the partitioning could in cases like this be based on factors developed in a Delphi process. For the purpose of the current study, hypothetical partitioning factors were generated based on what was judged by the authors to be one possible result of such an exercise; this is further discussed in section 5.2.

Figure 2 presents the partitioning problems and the selected partitioning factors for the joint processes in case study option ALCA 1. Figure 2 also provides a possible graphical tool in an exercise with stakeholders. Note that the relative partitioning between WWT and biogas production should be the same in all processes that have the function to manage waste from the joint wastewater sludge treatment and biogas production (corresponding to the relative importance of the biogas and the WWT as drivers). This partitioning therefore only has to be decided for partitioning problem a) in Figure 2. This also means that partitioning problem c) in Figure 2 is reduced to finding the partitioning factor between this combination of functions and the fertiliser production.

⁴ We call it legal here although the joint process in our example should perhaps rather be said to be defined by both legal and policy issues

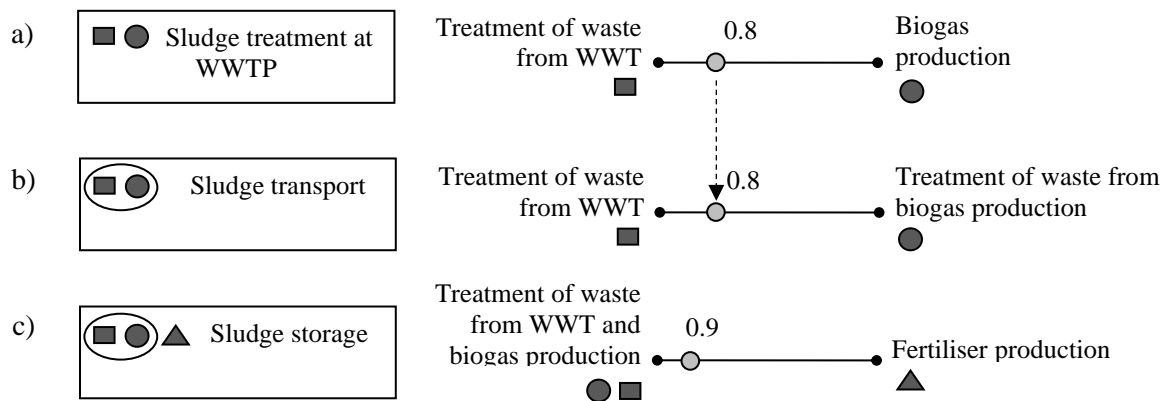


Figure 2. The partitioning problems and selected partitioning factors in case study option ALCA 1. For explanation of symbols, see Figure 1.

Figure 3 presents the partitioning problem and the selected partitioning factors for the black box joint process that was obtained assuming a legal relationship (case study option ALCA 2). Figure 3 also provides a possible graphical tool in an actual exercise with stakeholders. The selected hypothetical partitioning factors in numbers are 0.7 for the wastewater sludge treatment function, 0.2 for the energy recovery function and 0.1 for the nutrient recovery function.

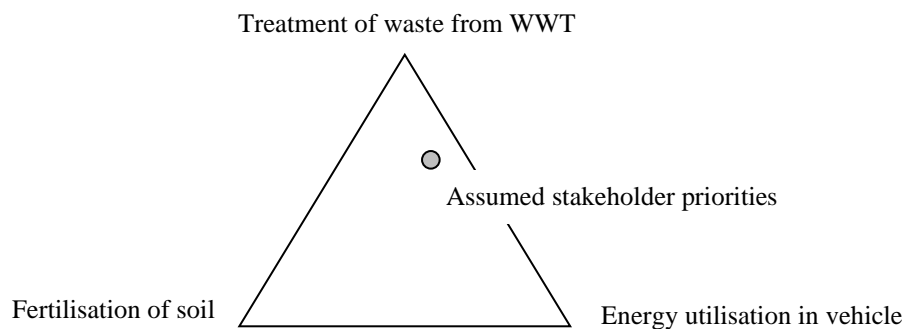


Figure 3. The partitioning problem and selected partitioning factors in case study option ALCA 2.

3.3 Consequential modelling of the case study system

In contrast to attributional modelling, consequential modelling assesses the environmental consequences of the activities in a life cycle considering interactions with dynamic surrounding life cycles.

Modelling of subsystems. Consequential modelling of a life cycle implies that marginal data is used to model subsystems of a studied life cycle when the processes or products investigated only have a marginal impact on the production volume in these subsystems. This is because the data should represent the effects of a change in the studied system, rather than the average environmental impact of producing the unit of the good (Finnveden et al., 2009). Due to lack of data on marginal production of many inputs, marginal data was used only for electricity production in this case study. Average data was, however, considered in these cases to be an adequate proxy for marginal data for other inputs.

The marginal technology for electricity use in Sweden differs depending on the time boundary chosen for the study. In a short-term perspective, marginal electricity is mainly produced in

existing coal-power plants (Sandvall et al., 2017). Data reflecting this were used in case study option CLCA 1. In case study option CLCA 2, we applied a more long-term perspective, where a change in electricity use can affect the production capacity in the electricity system. What investments are actually affected is highly uncertain. We assumed the long-term marginal electricity to be a mix of 70% wind power and 30% electricity from natural gas, which is one of the scenarios used by Sandvall et al. (2017). This mix is carbon-lean, which makes it a good contrast to option CLCA 1 and thereby serves our purpose of illustrating the influence that different modelling options can have on the results.

Handling of multifunctionality. Subdivision can be applied also in consequential studies, however if this does not fully solve the multifunctionality issue, system expansion or substitution is commonly used. Substitution has been the dominant way to handle multifunctionality in previous LCAs on wastewater and sludge management (Heimersson et al., 2016), and then generally using average production data for replaced products or services, but without any comment on the appropriateness of such a choice. In the consequential modelling of the case study system in this paper, we also use average data for the production and use of replaced products, but this was done because it was considered a relevant proxy for marginal data. In the case of biogas, the foreseeable consequence of the use of biogas is, in a short-term perspective, that marginally produced natural gas will be replaced (applied in case study option CLCA 1). Given a longer time perspective, enabling a shift in infrastructure and vehicle technology, diesel is instead assumed to be replaced (CLCA 2). In the case of replaced mineral fertiliser, average production data for calcium ammonium nitrate and triple-super phosphate (Brentrup, 2015) is used as a proxy for lacking marginal data on fertiliser production in both options. Two CLCA approaches were thus applied in the case study, one with a short-term perspective in the choice of data for the background system and foreseeable consequences, and one with a long-term perspective.

The four A- or CLCA options applied in the case study are summarised in Table 1, along with a plausible rationale for selecting each of the options.

Table 1. Summary of the options explored in the case study, and examples of questions and judgement calls that would make each option appropriate.

Option	Description	LCA questions	Judgement call
ALCA 1	A physical relationship is employed to identify joint processes. Average data is used for background processes.	What environmental impact belongs to the WWT? How can this impact be reduced?	The WWT system includes only processes that cannot be physically distinguished from the WWT or the treatment of the WWT waste.
ALCA 2	A legal relationship is employed to identify joint processes. Average data is used for background processes.		The WWT system includes also processes that must be there for legal reasons.
CLCA 1	A short time perspective on foreseeable consequences: biogas use in heavy vehicle is assumed to replace natural gas. Marginal data is used for background processes, assuming coal-fired electricity generation.	How does the WWT affect the overall environmental impact of society?	The impacts of WWT on investments in electricity production are too uncertain to account for.
CLCA 2	A long term perspective on foreseeable consequences: biogas use in heavy vehicle is assumed to replace diesel. Marginal data is used for background processes, assuming 70% wind power and 30% natural gas for electricity generation.	How does changes in the WWT affect the overall environmental impact of society?	The impacts of WWT on investments in electricity production are too important to exclude from the LCA and can be accounted for through a reasonable assumption.

4. RESULTS

Figure 4 displays the life cycle impact assessment (LCIA) results for two different environmental impact categories: climate impact, given as global warming potential (GWP), excluding biogenic carbon (as is common practice), and acidification impact, given as acidification potential (AP). The GWP and AP characterisation methods recommended by the ILCD Handbook (EC-JRC, 2011) were used for the assessment. Only an excerpt of potentially interesting environmental impact categories are thus presented here as this is not a full case study of this system; the case study is merely used to illustrate the impact of various modelling options. We chose two impact categories that are fairly common, for which results do not normally co-vary, and for which the methodology is generally accepted. Figure 4 shows that the results are dependent on the choice to do an ALCA or a CLCA and on how the modelling is done in each case.

For ALCA, the issue is how large a share of the total impact of the system that should be allocated to the WWT function and the result will therefore of course be highly dependent on the partitioning factors. These should be based on the drivers behind the processes and in our example, we have deemed the WWT function to be an important reason for the existence and operation of the system. A maximum of 100% of the impact could have been attributed to the

WWT function, compared to the 70% attributed to this function in our example (ALCA 2, see Figure 4). The type of relationship used to identify joint processes in the ALCA results in a small difference for the total GWP results. However, for AP the difference is larger due to the greater importance of the emissions after spreading of sludge on arable land (ammonia emissions to air). In fact, in the ALCA 1 model, this part is not considered to belong to the system, and with this method the importance of focusing also on what happens to the sludge after storage can not be seen. For the CLCA, a larger difference relating to modelling choices can be seen, especially for the GWP results which show to be highly dependent on in particular the modelling of the marginal electricity production (wind power and natural gas or coal power; compare especially the impact from the WWT in CLCA 1 and CLCA 2), but also on what the foreseeable consequences are considered to be, in this case the choice of replaced vehicle fuel (natural gas or diesel; compare the avoided impact from the replaced vehicle fuel in CLCA 1 and CLCA 2). The CLCA 1 model will point very strongly towards the decreased use of electricity as the main measure for reducing climate impact. We can conclude that not only the choice between attributional and consequential modelling but also the subjective interpretation of key concepts within each approach are important for the results.

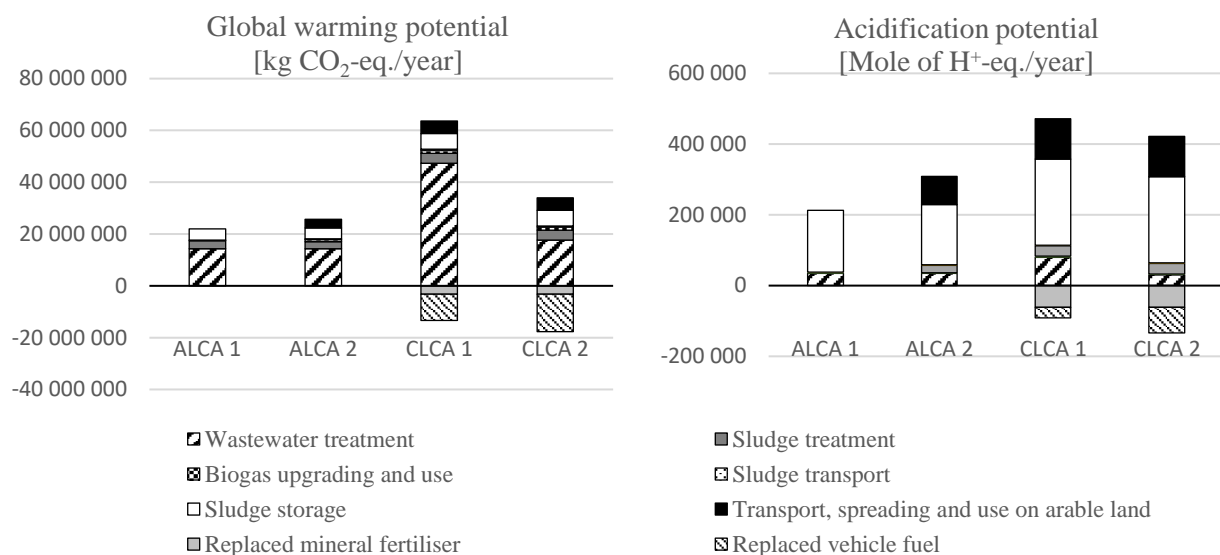


Figure 4. Climate and acidification impact for the four modelling options for the case study system.

5. DISCUSSION

Several suggestions for improved methodological practice for attributional and consequential LCAs on wastewater and sludge management systems have been demonstrated. Below, benefits and drawbacks of the suggested methodological practices are discussed.

5.1 Consequences of ways of identifying joint processes in ALCA

As could be seen in the results section, the interpretation of the concept of joint processes did not have a large impact on the LCIA results for the studied system (compare ALCA 1 and ALCA 2) for the displayed categories and the employed partitioning factors, although the ALCA 1 model misses some potentially important information on sludge handling. However, the implications of the different views embodied in the modelling approaches could be important. Subdividing the system based on a physical relationship between processes results in that the impact from sludge use on arable land is attributed entirely to the fertiliser function. As a consequence, the WWT (which is here selected to be the main function of the system) is

not assigned any responsibility for subsequent impacts related to the spreading of sludge, e.g., heavy metals and organic micro-pollutants emitted to soil or emissions to air or water related to the nutrients in the sludge. In fact, if the toxicity impact had been assessed in our case study, the different relationships used to identify joint processes could potentially have generated large differences in results. Toxicity impact was not assessed in this study because the impact assessment methodology is still considered to be quite immature (Nordborg et al., 2017) and we do not want to blur the message with a discussion on impact assessment methodology and missing data on concentrations and characterisation factors. However, we can still conclude that the use of physical relationships in identifying joint processes gives the WWTP operator less of an incentive to reduce the level of unwanted substances in the sludge. As a contrast, the WWT function has historically sometimes been attributed the full burden of the spreading and use of sludge on arable land (Doca, 2009). In the ALCA 2 approach that instead considers what processes are legally required in identifying joint processes, a large part of the system is considered joint, and all three studied functions share the responsibility of the impacts from sludge use on arable land, with the partitioning based on how strongly each function serves as a driver for that part of the system.

The concept of legally joint processes works in the ALCA of WWT, but could be problematic in a broader context. The manufacturing of most products is in the long run legal only if someone buys or otherwise deals with the products. Producing a product or buying a product might also entail a responsibility to ensure that the product reaches proper waste management after use, and when waste managers receive waste, they are typically required to deal with it properly. A whole life cycle or more could be considered legally joint, if this concept is broadly interpreted. A more careful definition and/or thorough analysis of the concept is required before its use beyond WWT can be established.

5.2 Partitioning based on stakeholder priorities

The partitioning factors employed in this paper were selected by the authors and are used here merely to illustrate the effect of method choices. The priorities are likely to differ depending on the types of stakeholders that are involved in an actual process of developing partitioning factors. A carefully designed participatory process that involves a balanced set of stakeholders can be arranged, e.g. including a workshop and a two-stage Delphi process for investigating the possibilities for reaching consensus among the stakeholders. This was done in a study that has been reported elsewhere after this study was performed (Svanström et al., 2017). In that study, no consensus was reached and average partitioning factors of about 0.7, 0.1 and 0.2, respectively, were obtained for the three same functions as are discussed in this paper: wastewater treatment and sludge management, production and use of biogas in vehicles and fertilization of agricultural soil. It can be concluded that the partitioning factors selected by the authors of this paper (0.7, 0.2 and 0.1) are similar to the average factors from the actual Delphi study done later on. For a discussion on the ranges of obtained factors in the Delphi study and on potential reasons for the differences, see Svanström et al. (2017). If we imagine that for some reason, the WWT and sludge management function was considered equally important to the other functions in our case study, achieving a partitioning factor of only 0.33 instead of 0.7, all results for model ALCA 1 would be only about half of what can be seen in Figure 4. A similar Delphi approach has been applied to discuss the primary energy content of, e.g., residual heat from industry (Ekvall et al., 2012). When consensus cannot be obtained, the method could instead generate different sets of partitioning factors that represent different groups of actors. These sets could then be used in parallel in the LCA to demonstrate the significance of the different stakeholder perspectives (for an example, see Svanström et al. (2017)). The timing of such a workshop would possibly also result in different priorities as perceptions and priorities

change over time. For example, nutrient recycling to arable land is currently not required in Sweden. However, in the future this might change (Swedish EPA, 2013).

Using a two-stage Delphi process to generate partitioning factors is, however, very time consuming for the LCA practitioner, and requires active participation of several stakeholder groups. Its usefulness is therefore likely limited to partitioning in the foreground system.

5.3 Perspectives brought by attributional and consequential modelling

The distinction between attributional and consequential modelling has arisen as an opportunity to diversify LCA studies in order to make them more aligned to the decision context. If, for example, the goal of the LCA is to compare the environmental performance of a current and a future sludge treatment alternative, and the commissioner of the study is interested in how their local actions, e.g. in Gothenburg, may contribute to a more environmentally sustainable world, then a consequential approach would be useful. A consequential approach would enable inclusion of the consequences on other life cycles from the change in the studied system (e.g. a changed energy requirement in the studied system could change the demand for electricity in the studied region, affecting the production mix). However, an attributional approach can be useful if the LCA is commissioned by, for example, the management team of a WWTP and they want to be informed about the environmental impact of the processes that they can directly influence. An attributional approach would then give the information they seek. A local policy maker or municipal civil servant, with responsibility for WWT, could also find an ALCA useful for investigating how different technological options would affect what environmental impacts that belong to the WWT.

It can be argued that the choice of an attributional approach in an LCA aimed at supporting the decision of an actor to a higher extent highlights the impacts directly related to the actor's sphere of influence and avoids blurring the information with aspects that relate to other parts of the system that the actor cannot decide on and, hence, does not feel responsibility for. A consequential approach, on the other hand, better maps actual future effects, as it takes society-wide consequences into account, and thereby implicitly assumes that responsibility is taken for a larger system. As could be seen in the case study reported in this paper, such a distinction between ALCA and CLCA can be important for LCIA results. The importance of distinguishing between these two perspectives becomes more important the larger the difference between the two perspectives. For example, when there is a large use of electricity in a system and also a large difference in environmental impact between average and marginal electricity, the choice between attributional and consequential modelling, as defined in this paper, will be important; compare the climate impact between the CLCA 1 model and any of the attributional models in Figure 4.

Earlier LCAs on wastewater and sludge management systems have generally used substitution to account for additional functions, and average data to model both the substitution and the background system. This would, using the definitions of ALCA and CLCA applied in this study (Finnveden et al., 2009), result in a hybrid LCA approach with partly attributional and partly consequential elements. The results from such studies could be challenging to interpret as they are not designed to reflect a certain perspective. This study shows that there are ways to align LCAs in this field better to either attributional or consequential thinking. It is in this context important to note that existing guidelines on LCA methodology often do not differentiate between ALCAs and CLCAs. This is, for example, the case for ISO14004:2006 and the

European Commission's new Guide for Product Environmental Footprints (PEF, 2013/179/EU)⁵.

5.4 More allocation problems will arise

This study has focused on common allocation problems in LCAs of wastewater and sludge management. It can be expected, however, that future wastewater and sludge management facilities that have developed into biorefineries with diverse and multiple outputs in line with principles of a circular economy will present more and different allocation problems to LCA analysts. For example, the generation of biopolymers, integrated with the WWT process, would create a different situation than was presented in this paper (Morgan-Sagastume et al., 2010), with a need to consider also the WWT process as a multifunctional process. It is likely that future WWTPs will be seen less as waste treatment facilities and more as recycling or production facilities.

It should also be noted that the choice of another functional unit would have generated very different results in this study. We assumed the WWT to be the main function, but in some cases, other functions could be in focus instead, for example if a commissioner would be interested in understanding if using sludge from WWT as a fertiliser would generate lower environmental impacts than other fertilisers (see, e.g., Linderholm et al. (2012)), or in making an environmental comparison between biogas from sludge digestion and other fuels. A change in functional unit would completely change the results of the case study, and pose new methodological challenges, e.g., related to accounting for the WWT function in a CLCA where it is not the main function.

6. CONCLUSION

This study builds on and develops further the ongoing scientific discussion on attributional and consequential modelling in LCA and provides methodological recommendations and new approaches for the field of LCA of wastewater and sludge management.

Climate and acidification impact results generated for a case study system show that is possible and relevant to differ between attributional and consequential methodology when assessing wastewater and sludge management systems. CLCAs can, for example, be useful in situations where a change is introduced in the WWTP, e.g. a new route for sludge handling, and the management team is interested in the environmental consequences of the change. An ALCA could be relevant for the same management team for example if they want to know how large a share of the global environmental impact that their activities are responsible for.

The study shows that LCA results for a wastewater and sludge management system depend not only on the choice between ALCA and CLCA, but also on methodological choices made when applying each of these approaches. The sensitivity of the results of the CLCA case study to the choice of foreseeable consequences and the modelling of the background system highlights a strong dependence on the chosen time perspective. For ALCAs, the concept of legally joint processes was introduced but showed a smaller influence on the case study results compared to if physically joint processes were used as basis for the allocation. To base the partitioning factors on stakeholder priorities is suggested, in order to avoid partitioning based on, e.g., the very weak economic drivers for the studied functions in this system and for environmental technologies in general.

⁵ No PEF category rules for wastewater or sludge have yet been developed (January 2018).

7. NOMENCLATURE

ALCA	attributional life cycle assessment
CLCA	consequential life cycle assessment
LCA	life cycle assessment
LCIA	life cycle impact assessment
WWT	wastewater treatment
WWTP	wastewater treatment plant

8. ACKNOWLEDGEMENTS

This research has been funded by the Swedish Research Council for Environment, Agricultural Sciences and Spatial Planning (FORMAS) [grant agreement No. 2015-1116] and the Energy Area of Advance at Chalmers University of Technology [strategic funds for collaborative links to research institutes 2016-2017].

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