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Improved life cycle modelling of benefits from sewage sludge anaerobic digestion and land application

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Abstract

Nitrogen, phosphorus and organic matter are valuable resources in sewage sludge. Life cycle assessment (LCA) can be useful for comparing the potential environmental risks of sludge management strategies to their potential environmental benefits. With growing interest in resource recovery from sludge, there is an increasing need to properly account for the benefits that can be achieved, and to handle the multi-functionality issues that then arise in LCAs. So far, both of these aspects have often been handled in a generic and seemingly arbitrary way.

The study identified and explored several alternative approaches to handle the multi-functionality in the LCA of a sludge handling system that generates both biogas and a sludge that is used on arable land; either through avoiding allocation by substituting for avoided products or services (e.g. fertilisers and natural gas), or by allocating the impact from the studied system between its functions based on economic terms. The choice of approach strongly influenced the overall LCA-result for the studied system, in particular for some of the studied impact categories. Although an attempt was made to apply economic allocation in this article, it can be concluded that no coherent basis for applying allocation was identified. Substitution was more easily applied, however, the results were highly dependent on the product assumed to be replaced by biogas and the modelling of avoided mineral fertiliser use. The previously neglected benefits related to organic matter provided by the sludge to arable land were potentially as important as the benefits of the nitrogen and phosphorus, although

the quantification of such effects will need further refinement, and are only relevant for certain soil conditions.

Keywords

Biosolids; life cycle assessment; resources; nitrogen; phosphorus; organic matter

Highlights

- A substitution approach showed to be sensitive to the modelling of replaced functions.
- Modelling of products replaced by digester biogas is important for results.
- Accounting for benefits of organic matter to arable land can be important.
- No really useful allocation basis was identified, although economic allocation was tested.

Abbreviations

C	carbon
K	potassium
LCA	life cycle assessment
N	nitrogen
P	phosphorus
WWTP	wastewater treatment plant

1 Introduction

In the EU-27 states alone, about 10 million tonnes (metric tons) of dry solids of sewage sludge (in this paper “sludge”) is generated yearly (Milieu Ltd et al., 2010). As sludge in general has a heating value of 21 MJ/kg dry matter (for activated sludge) and contains 1.5-5% nitrogen (N) and 0.8-1.1% phosphorus (P) (Metcalf & Eddy Inc et al., 2004) and large amounts of carbon (C), previous disposal methods such as landfilling of sludge are increasingly seen as a loss of potential resources, and focus is gradually shifting towards utilisation of these resources, for example through recovery of energy, materials or nutrients (in the future, even other types of resources, such as trace amounts of metals, could become worth recovering (Westerhoff et al., 2015)).

Sludge treatment through anaerobic digestion followed by spreading on arable land is today a common way to deal with sludge and it enables use of resources in sludge in two ways, through the digester biogas and through the valorisation of the nutrients and organic matter in the digester sludge. Biogas can e.g. be combusted to generate electricity and/or heat that can be utilised internally in the wastewater treatment plant (WWTP) or be sold, or be used as a vehicle fuel. Utilisation of sludge as an organic fertiliser on arable land has been shown to increase agricultural productivity in numerous studies (see Singh and Agrawal (2008) for a review). Use of sludge on arable land can provide both N, P and other nutrients and thereby recycle nutrients in society, which is consistent with the aim of developing a circular economy, and can also contribute to maintaining the concentration of soil organic carbon (SOC) (Brady et al., 2012). However, the use of sludge in food production is also associated with risks due to the heavy metals, organic micropollutants and pathogenic microorganisms in sludge (VKM, 2009), and whether these risks are acceptable is debated. Generally, to fulfil national legislative requirements, the sludge needs to be hygienised before it can be applied to arable land, and there are also threshold limit values describing how much heavy metals and nutrients that can be applied to land (see e.g. EU Directive 86/278/EEC).

1.1 Modelling of agricultural sludge use in LCA

Life Cycle Assessment (LCA) is an assessment tool that can be used to quantitatively assess and compare the environmental performance of different products or services. It has been frequently applied in the context of wastewater treatment (Corominas et al., 2013). LCA can be used to compare environmental impacts of different types of sludge handling and can, together with other types of information such as from quantitative risk assessments, support decisions on the environmentally preferable way of handling sludge. But to provide valuable decision support, LCA methodology needs to be able to both assess negative impacts of the sludge management and account for potential resource recovery. The environmental impacts of sludge use in agriculture have recently attracted attention in several modelling studies, e.g. for toxicity assessments (Harder et al., 2016) and pathogen risk (Harder et al., 2014). Heimersson et al. (2016) showed that the modelling of N, P and major C flows originating from the wastewater is very important in LCA, since the type of wastewater treatment process determines whether they end up as an emission, contributing to environmental impacts, or as a resource flow and thereby possibly lowering or counteracting environmental impacts from the assessed system.

1.2 Accounting for secondary functions during sludge treatment

The increased focus on resource recovery from wastewater and sludge during recent years results in the fact that a wastewater and sludge management system is increasingly multifunctional. A system in which sludge treatment and end-use is the main function, and with secondary functions such as production of biogas and a soil fertiliser and conditioner, is an example of such a multi-functional system. When such systems are assessed, estimates need to be made on how much environmental burden that is to be attributed to the main studied function; this depends on how the multifunctionality is handled and how large the benefits from the resource recovery are considered to be. According to ISO 14044:2006 the preferred analytical approach is to inventory the studied system in enough detail that each flow can be connected to a particular function. However, if any of the processes in a studied system delivers several functions, such subdivision is not possible. For the multifunctional system described above, the digestion can be considered to occur in order to 1) reduce the mass of sludge, 2) generate biogas for use as an energy carrier and 3) stabilise the sludge before agricultural use. Further, the application of sludge on arable land occurs in order to 1) dispose of the sludge, 2) provide nutrients and organic matter to the soil and 3) handle by-products from the biogas production. It is thus impossible to subdivide the system in order to solve the multi-functionality problem. In such cases, ISO14044:2006 recommends to expand the studied system to also include the functions provided by the co-products. This may be interpreted as expanding the functional unit to account for all functions (sometimes referred to as system expansion) (Heijungs, 2014) or to give the system a credit for the secondary functions by awarding the system negative emissions or avoided resource use corresponding to the avoided product or service that the secondary functions replace (e.g. the avoided production and use of mineral fertilisers replaced by sludge) (Koffler, 2014). The latter is referred to as substitution, and is a very common way to handle multifunctionality in LCAs on wastewater and sludge management. The least preferred option, according to ISO 14044:2006, is to allocate the burden between the different products or services. One of the products can then be considered to be responsible for the entire burden, or the burden can be allocated between the functions based on some relevant and comparable physical attribute, for example weight or energy content, or if that is not possible, on economic value. Allocation (partitioning) has not been tested earlier for a system like the one discussed above, as far as available literature reveals.

Other examples of guiding documents on multifunctionality issues are the ILCD Handbook (EC-JRC, 2010) and PAS 2050 (BSI, 2011). For a review on guidelines on how to treat multifunctionality related to co-production, recycling and energy-recovery see Schrijvers et al. (2016).

1.2.1 The benefits of digester biogas generation

If all the biogas is assumed to be consumed internally in the studied system, no multi-functionality issue will arise (e.g. if the biogas is used to heat the digesters). However, from a life cycle perspective, this may not be the environmentally preferable option, and depends on e.g. what other heating sources may be employed and the alternative use of the biogas. It is expected that biogas will increasingly be seen as a resource for which optimal use has to be decided on a case-by-case basis, and when biogas (or products from its combustion - heat and possibly also electricity) is used outside of the studied system, the multi-functionality that arises will have to be managed.

Heimersson et al. (2016) showed in a literature review that digester biogas is commonly accounted for assuming on-site combustion of the biogas that generates heat and possibly also electricity that is used internally at the WWTP as a first choice, but potential excess energy is often considered to be sold and to replace other means of heat and electricity production. Some exceptions are that Cao and Pawłowski (2013) assumed that biogas replaced diesel as a vehicle fuel while a few others (e.g. Mills et al. (2014)) assumed that it replaced natural gas production.

1.2.2 The benefits of sludge use on arable land

Heimersson et al. (2016) showed that a majority of reviewed LCA studies on wastewater and sludge management with agricultural sludge use accounted for the benefits of N and P on arable land, by crediting the studied system for the avoided production, and sometimes also the use, of mineral fertilisers. The remaining studies used EcoInvent datasets, in which the sludge management function is considered as waste treatment function that is allocated the full burden of the sludge treatment and land application processes, and no additional functions are accounted for. Heimersson et al. (2016) showed that the modelling of the amount of fertiliser that could be replaced by the sludge in a substitution approach is generally based on generic sludge-mineral fertiliser replacement ratios (the rate at which nutrients in the sludge is considered to replace mineral fertiliser nutrients, this can be on the basis of e.g. assumptions on plant availabilities) that are not specific for the conditions in the studied system.

In addition to accounting for the benefits that N and P can provide, a few studies also accounted for the value of potassium (K) or for the carbon sequestration in soil (carbon capture in soil is then assumed to contribute to reduced climate impact). One Australian study suggested a method to account for the increased water retention capacity that could result from sludge spreading, when assessing the impact category of water use (Peters and Rowley, 2009).

Increased SOC can increase crop yields, e.g. as it potentially increases soils´ N mineralisation capacity (Hedlund, 2012) and it improves the soil structure and increases the cation exchange capacity which is important for the soils ability to hold nutrients and water. As SOC in arable land vary considerably, so do the potential benefits of adding extra C to the fields. In Scania in southern Sweden, an area with low SOC, farmers show an interest in the potential to increase the SOC by using sludge (KSLA, 2012) (in addition to their main interest, the P). Schaubroeck et al. (2015) accounted for the potential benefits resulting from the organic material in sludge in Austria by assuming that the provision of humus C, in addition to N and P, through the application of sludge to arable land substituted for the production and use of peat and straw (based on C fertilizer data from Hermann et al. (2011)), in addition to replaced mineral fertiliser. However, in the regions where organic amendments are not normally added (as in Scania), e.g. because of low availability as the region´s livestock density is low, using the approach suggested by Schaubroeck et al. (2015) described above is not useful. In such cases, the potential positive effects from the C in the sludge put on soil is thus an added benefit that does not replace any other product applied on the fields (as is the case with the N and the P otherwise added as mineral fertiliser). One possibility to account for this could be to add an indicator of soil quality to the impact assessment categories (see Garrigues et al. (2012) for a review). However, this does not necessarily provide very useful information since this indicator is relevant only for a small part of the studied system; the life cycle perspective would therefore not provide additional information.

1.3 Purpose of the paper

The current paper describes a study that aims to provide input to more relevant modelling of resource recovery resulting from sludge management in LCA, by investigating different possibilities for accounting for secondary functions relating to biogas and sludge used in agriculture, either through substitution or allocation, in a system whose main function is treatment and handling of sludge, in a region where sludge is land applied but not replacing any other organic amendment. Novel approaches for applying both substitution and economic allocation are identified and their practical applicability is tested on a specific Swedish sewage sludge management system.

2 Method

The current study explores different approaches for handling the multi-functionality in an LCA of a system that performs several functions: treatment and disposal of sludge, production and utilisation of biogas and provision of nutrients and organic matter to arable land, see Figure 1. The treatment and use of 1 dry tonne of sludge was chosen as the functional unit, reflecting the choice of sludge treatment as the main studied function. This means that the biogas production and use and the potential benefits of sludge when applied to arable land are additional (secondary) functions of the studied system. In the future the view on sludge may change, but in this study sludge is considered a waste and consequently no environmental impact from its production is considered (this can be questioned, but it is the approach used in all earlier LCAs on wastewater and sludge treatment (Pradel et al., 2016)). By investigating two different approaches for handling multi-functionality for such a system (substitution and allocation) the study aims to investigate (1) the importance of this methodological choice in LCA, as well as (2) the practice of applying such substitution and allocation approaches. After identifying several possible ways of handling the multi-functionality issue for the specific situation, they were tested for a model system.

The decision context in focus in this study is the use of LCA to generate a part of the decision basis underlying strategic decisions on sludge treatment and end-use. The sludge treatment is therefore the main studied function, reflected in its functional unit, and the aim of the LCA is to assess one system in full in order to accurately capture both environmental benefits and drawbacks of that system. In another decision context, focus could instead have been on nutrient provision to soil, where the main studied function could be “provision of P to arable land” (Linderholm et al., 2012). This would make the sludge treatment a secondary function and give rise to other multifunctionality issues (but similar to the ones described in the current study).

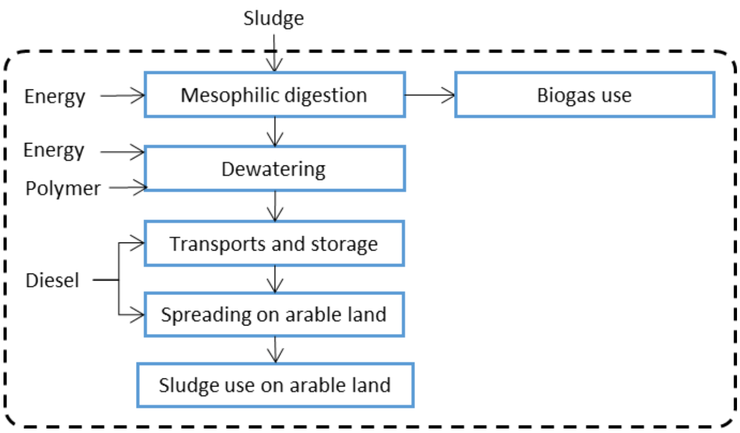

Figure 1. The model system which provides sludge treatment (main function) and also products (secondary functions) in the form of biogas for use as energy carrier and sludge for use on arable land.

2.1 Model system

In the model system, mixed primary and secondary sludge is considered to undergo mesophilic anaerobic digestion in order to produce biogas, whereafter the digested sludge is used for agricultural purposes, see Figure 1. The life cycle inventory (LCI) was compiled for mixed sludge produced at Gryaab Ryaverket (treating municipal and industrial wastewater from approximately 800 000 person equivalents in Gothenburg, Sweden) that was digested on site, transported 100 km to a storage facility, stored for a minimum of 6 months (in order to fulfil current hygienisationrequirements before agricultural sludge use set up by a Swedish industry certification system for sludge that is land applied (REVAQ, 2016)) and finally transported a further 50 km to arable land for spreading. No infrastructure, such as buildings and machinery, were included in the inventory. The biogas was assumed to generate electricity and heat through combined heat and power (CHP) combustion, as this is a common approach in Europe. The LCI is further described in a Supplementary Material. The studied system was assessed using the Gabi 6 Professional software (Thinkstep, 2015). Average data on production of energy and consumables were chosen instead of marginal data. Six impact categories were assessed, using the following mid-point indicators: global warming potential (GWP), acidification potential (AP), freshwater, marine and terrestrial eutrophication potential (EP) and photo-oxidant formation potential (POFP), using the characterisation methods recommended by the ILCD Handbook (EC-JRC, 2011). These impact categories were selected because of their relevance for assessing N, P and C benefits from resource utilisation from sludge, in accordance with Svanström et al. (2015). If the system were assessed for other purposes, e.g. to serve as a case study of the environmental impact from this type of systems, other impacts which are now excluded due to the limitations of this study, such as human toxicity and resource use, would have been relevant to include. The results presented in this study should thus not be used as guidance on benefits and drawbacks of the described system.

2.2 Alternatives for handling multifunctionality in the studied model system

Seven different modelling alternatives were selected; one baseline representing the current practice of applying substitution, four different variations in practice when using substitution, and two types of allocation. The different substitution and allocation alternatives are summarised in Table 1 in Section 3.

In the current study, the same modelling approach (substitution or allocation) is consistently used to account for both secondary functions (related to biogas and the sludge use on arable land) for each investigated alternative. Consistent use of the approach for handling multi-functionality has also been advocated elsewhere (Sandin et al., 2015). However, no efforts have been made to vary the handling of multifunctionality in background system inventory datasets.

2.2.1 Baseline

The baseline reflects common practice in LCAs on sludge management, as identified by Heimersson et al. (2016). The energy generated from biogas was assumed to replace Swedish average grid electricity production and district heat production in Gothenburg. This approach to account for gross energy values was chosen in order to clearly illustrate the importance of the utilisation of biogas; overall net results are not affected by this choice. Sludge on arable land was assumed to replace N and P mineral fertilisers (calcium ammonium nitrate and triple super phosphate) at a replacement ratio of 0.5 and 0.7 (based on earlier common practice, see section 2.2.3), respectively, with data on mineral fertiliser production from Fertilizers Europe (Brentrup, 2015). This new data set gives a smaller environmental impact to calcium ammonium nitrate and triple super phosphate production for all of the studied impact categories compared to e.g. the datasets of Davis and Haglund (1999), which has been used, directly or indirectly, in many earlier studies (e.g. Johansson et al. (2008), Linderholm et al. (2012), and many studies based on the ecoinvent database (Nemecek and Kägi, 2007)). This means that the contribution of the replaced mineral fertiliser production to the total impact is generally smaller in this study compared to previous work, and in effect, resource recovery generates smaller benefits today as alternative activities have decreased their environmental impacts.

2.2.2 Substitution 1: Alternative assumption on the use of biogas

Substitution aternative 1 investigates the effect of assuming an alternative use of biogas where the biogas is upgraded and used to fuel vehicles (which is the actual case in Gothenburg), thereby replacing natural gas. The replacement ratio was based on heat content (SGC, 2012). Data on upgrading of biogas was taken from the work of Palm and Ek (2010), on natural gas production from the Gabi professional database and on emissions from biogas use in heavy vehicles (per MJ of biogas) from Börjesson et al. (2010). The same emissions per MJ were assumed for natural gas use, and with added fossil CO_2 emissions, based on a stoichiometric calculation assuming that 1 mole of CH_4 gives rise to 1 mole of CO_2, and that Swedish natural gas contains 90% CH_4 (SGC, 2012).

2.2.3 Substitution 2: Alternative replacement ratios for N and P

Selecting replacement ratios is challenging. The plant availability of nutrients (both those originating from sludge and from fertilisers) are dependent on e.g. the local conditions of the soil, the climate, the cropping and fertiliser application practice and, in the case of sludge, e.g., the precipitation chemical used to precipitate P in the WWTP (Bengtsson et al., 1997). The true replacement ratio will thus depend on local conditions and a generic replacement ratio cannot be established. Further, the actual replacement ratio will also depend e.g. recommendations within the agricultural sector on appropriate levels and on the cost of mineral fertilizers.

Examining the references on replacement ratios in the studies reviewed by Heimersson et al. (2016), one can conclude that many of them refer (directly or indirectly) to Bengtsson et al. (1997) and/or Dalemo et al. (1998). Bengtsson et al. (1997) define “substitutability” as the plant availability of a nutrient in sludge (or urine) divided by the plant availability of the nutrient in the other fertiliser. They suggest replacement ratios of 0.5 and 0.7, for N and P, respectively, for dewatered sludge that has been stored at the WWTP, based on one growing season. Dalemo et al. (1998) suggest using a mass balance model to estimate the decreased need for mineral N fertiliser due to the use of an organic fertiliser (and discussed it for manure). Dalemo and colleagues assumed that all NO_3^-, the NH_4^+ that has not been lost as emissions and 30% of the organic N in organic waste replaced mineral fertiliser use during the first year. They assumed that during the following years, the remaining organic N in the soil is mineralised and becomes plant available in a fraction equal to the one under the first year, and is then also replacing N fertiliser. The review by Heimersson et al. (2016) showed that N replacement ratios of between 0.4 and 1 have previously been used, and for P between 0.5 and 1.

A hypothetical maximum N replacement ratio for a specific system could be calculated using a mass balance, based on the assumption that all N in the dewatered sludge or mineral fertiliser that is not emitted to air or water becomes plant available in a long-term perspective, similar to what was done by Dalemo et al. (1998). For the current model system, a maximum potential replacement ratio for N would correspond to roughly 0.8 for dewatered sludge, calculated as a quotient of the N available in the sludge after accounting for losses during storage and after spreading on arable land, and the N in the mineral fertiliser after accounting for losses after spreading on arable land (note that possible emissions of nitrogen gas are not accounted for). The described approach quantifies a theoretical potential for N utilisation, but an alternative that would be to rather base the replacement on what is actually replaced, i.e. quantifying the reduced use of fertiliser by farmers in the region of study. The Swedish board of Agriculture (Jordbruksverket, 2014) recommends farmers to judge what they call the "nitrogen effect" of organic fertilisers (other than manure for which more specific guidance is given) as the readily plant available content of the organic fertiliser. In this study, only NH_4^+-N is, as a conservative estimate, considered readily available (since the NO_3^- level in sludge is very low, and since it is difficult to estimate how much of the organic nitrogen will be mineralised in the soil in the short term, farmers do not always account for this N). In the current model system, this would correspond to approximately 30% of the total N in the dewatered sludge (depending on the timing of the spreading of the sludge, this figure could actually become even lower, a conservative assumption could be 50% of the NH_4^+-N). Remaining N in the sludge is organically bound and is less available to plants. Some of it would mineralise over time, but the timing (both the time of the spreading and when the particular crop takes up N) of this is central for whether crops can utilise it or if it leaks from the field. It is unlikely that the farmers would take the risk of reducing their mineral fertiliser dose to the extent that that crops may not be provided sufficient amounts of N (within reasonable limits, there is also a potential risk of overdosing N). Therefore, it cannot be assumed that the organically bound N would in practice substantially lower the use of mineral fertiliser. The effect of choosing a replacement ratio corresponding to the NH_4^+-N content in the sludge and thereby what can actually be assumed to be done by the farmer in Sweden was tested in Substitution 2. This assumption can be argued to reflect a static technosphere (which is common in attributional modelling).

P is present in sludge as plant available mineral P or organically bound P (that eventually can be mineralised). The leakage to air and water is likely to be small (Stutter, 2015) and it is therefore common to apply enough P fertiliser for several years of crop requirement during a single spreading. Linderholm (2011) published a review of the literature on P availability in sludge under Swedish conditions and concluded that it was not possible to quantify plant availability for P in sludge. In the later published LCA by Linderholm et al. (2012), a replacement ratio of 1 was assumed, implying that the P from the sludge gradually becomes available to an extent similar to mineral fertiliser P. However, Jönsson et al. (2015) argued that P applied to land through sludge becomes less available to plants over time and that some P is strongly bound in complexes with precipitation chemicals, and assumed a P replacement ratio of 0.6 in their LCA. In Substitution 2, a replacement ratio of 1 is tested, in order to reveal the effect of a best case scenario with maximum resource recovery. For P it could be reasonable to account for the share of the nutrients potentially becoming plant-available over time, as this is sometimes indirectly partly accounted for by farmers, if they base their P mineral fertiliser dosing on soil mapping.

2.2.4 Substitution scenario 3: Substituting also for organic matter and micronutrient provision through sludge use on arable land

In Sweden, no other organic amendments than farmyard manure are normally added to the fields in order to increase SOC (other than occasionally sewage sludge). The benefits of an increase in organic matter in the soil is difficult to quantify, but increased crop yields could be used as a proxy. It is however hard to find studies that allow for quantifying the increased SOC's impact on crop yields separately from the effect of N and P addition. However, in Substitutions 3A and 3B, two alternative ways of quantifying the replaced crop yield resulting from benefits other than N and P from sludge on arable land are tested, for a region with low SOC, and the extra crop yields are then assumed to replace crop production elsewhere.

Swedish soils are most often mineral soils and soil in southern Sweden often has a low SOC, about 1.8% C (data for Scania, Berglund et al. (2009)). The project Soilservice (part of the European Union's 7th framework program 2009-2012) reported that, in Scania, a 1% relative increase in the C stock resulted in an increase in winter wheat yield of 38 kg/ha/yr (Hedlund, 2012). The same project reported that agricultural use of sludge increases the C stock in the soil by about 0.9% per year (Brady et al., 2012). Figures of the same magnitude are reported by Börjesson et al. (2012), for sludge application in Ultuna, Sweden, between 1956 and 2009 (sludge corresponding to 4 tonnes of C per hectare every second year). Assuming that the yield increased linearly with the C up to that point, around 35 kg of winter wheat per hectare could be considered to be replaced by sludge use on agricultural soil. This figure has been divided by the amount of sludge used on annual average in Ultuna to get the average per dry matter tonne (the functional unit in this study). In this case it has been assumed that the effect during four years (the crop rotation period) can be attributed to the specific tonne of sludge. This way of quantifying the effect of the sludge on the SOC was used in Substitution 3A.

No long-term studies on sludge use in agriculture under Swedish conditions have investigated the same type of land fertilised to the same extent with either sludge or mineral fertiliser. Instead, field studies have generally tested the effect of adding sludge to arable land that was also given full mineral fertiliser rate, half rate or no fertilisers. For Substitution 3B, benefits from sludge use on arable land other than N and P were quantified using results from a long-term field experiment performed in Scania, Sweden, since 1981 (Andersson, 2012). The study reported that a supply of 1 dry tonne of sludge per hectare and year (given as a 4-year provision), combined with full N, P and K mineral fertiliser provision, increases the yield by, on average, 0.09, 0.35 and 0.02 tonnes per hectare and year for winter wheat, winter rapeseed and spring barley, respectively (based on data for 17 harvests). Assuming that this effect is due to other substances than N and P (as these nutrients are already provided in sufficient amounts by mineral fertilisers), these results could be used for our LCA to indicate the additional yield resulting from the provision of C and nutrients other than N and P and K through the sludge use on arable land. In Substitution 3B, 4-year provisions of sludge were assumed to be applied to fields with a 4-year crop rotation; spring barley, winter rape seed, winter wheat and spring barley. The selected crops represent dominating crops in Southern Sweden (Statistics Sweden, 2014). The sludge that has been applied to the fields in the Scania study is digested and contains 2.7-5.7% P and 0.37-2.1% NH_4-N (on a dry matter basis), which is similar to sludge generated in Gothenburg. As the dosing of sludge in the field trials by Andersson et al. was slightly higher per hectare than what would be the case for the model system sludge, based on current Swedish legislative allowances, results have been scaled per dry tonne of sludge. Data inventories on crop production for the different grains in the studied region were obtained from ecoinvent (due to limitations in data availability, data for German conditions were used). The selected approach is assumed to be an overestimation of the actual effect, as part of the effect might in fact be a result of the extra N and P applied through the sludge.

2.2.5 Allocation 1: All impact is burdening the sludge treatment function

In contrast to a substitution approach, it is also possible to divide the impact from a system between its usable co-functions. The simplest allocation alternative would be to allocate all the impact to one of the functions, e.g. as is done in the datasets on wastewater treatment in the EcoInvent database (Doca, 2009) where all impact from sludge land application is attributed to the wastewater treatment function, motivated by the fact that the main function of the sludge system is a waste treatment, and safe disposal of sludge is required by law, whether or not any by-products are extracted from the process. Such an approach is applied in Allocation 1 (in this case both for biogas and sludge on arable land).

2.2.6 Allocation 2: Economic allocation

Allocation of shares of the environmental impacts of the studied system to its different functions on the basis of traditional physical relationships like mass, volume or energy were rejected in the current study. This is due to the fact that the three co-functions (one waste treatment service (the sludge handling), one energy carrier (the biogas) and one fertilising agent (the agricultural sludge use)) cannot reasonably be quantified using the same physical denominator. For the same reason, allocation based on exergy, which Sandin et al. (2015) found useful for their studied biorefinery when they were faced with a similar lack of common physical denominator, was also rejected.

Economic allocation is an option, but wastewater and sludge treatment systems offer several challenges when quantifying the economic value of the different functions. Economic allocation is often based on the market value of the different products or services. In Sweden, the WWTPs are publicly owned and operated, paid for by households and other users by a combined fee for water and sanitation that differs for different municipalities. The pricing of the biogas generated in the public Swedish water sector (e.g. discussed by Nordahl (2013)) is regulated through the Public Water Supply and Wastewater Systems Act (SFS 2006:412). For sludge, WWTPs in Sweden generally have to pay a contractor who collects, stores and distributes sludge to farmers, when this is the employed route. Estimating the economic value of the sludge for the Swedish farmer is challenging as there are very large variations and uncertainties.

Despite the difficulties, economic allocation was attempted as Allocation 2 in the current study. Calculated per functional unit, Gryaab had, in 2014, an income of about 1100 SEK for the biogas (including incomes from accepting small amounts of organic waste to be co-digested with the sludge) and total operation costs for the wastewater and sludge treatment at the WWTP of 8200 SEK (calculated based on information in Gryaab (2014)). The cost of sludge treatment is often of the same order of magnitude as the cost of the wastewater treatment (UNEP, 2002) and therefore, half of Gryaab's costs were attributed to the wastewater treatment. Andersson (2012) reported a value of sludge used in agriculture of between 500-1000 SEK per tonne dry sludge in terms of increased yield as a low estimate. Using these numbers for Allocation 2 results in that the sludge handling function should be burdened with about 70% of the impact of the studied system[1].

A principally different alternative could have been be to apply allocation in the life cycle impact assessment phase (on midpoint or endpoint indicator level) instead of in the inventory phase, as discussed for monetary valuation by Pizzol et al. (2015). Similar thoughts were brought forward by Cherubini et al. (2011), presenting a partitioning method based on avoided environmental impact. But as pointed out by Sandin et al. (2015), a major problem with such an approach is that it allocates most of the environmental burden to the product that prevents most environmental impact in another system, which does not promote such utilisation of the product/service. It is also central that the value of all the compared functions (in this case sludge treatment and handling, the biogas use and sludge utilisation on arable land) should be quantified in the same phase in the LCA (i.e., the LCI phase).

3 Results and discussion

[1] Calculated as the value of the sludge treatment divided by the value of the sludge treatment, the biogas and the sludge in agriculture ((8200/2)/((8200/2)+1100+((500+1000)/2))=0.7).

The seven evaluated substitution and allocation alternatives are summarized in Table 1. Figure 2 displays the results of the different approaches for accounting for benefits and handling of multi-functionality. In general, AP and terrestrial EP results are not as strongly affected by the choice of modelling approach as the remaining studied impact categories are.

Table 1. Description of the selected modelling alternatives for handling multi-functionality in a system where sludge is treated (main studied function), biogas is generated and used, and sludge is used for agricultural purposes.

Alternative	Description
Baseline	Biogas assumed to be combusted in CHP unit, generating electricity and heat, that is replacing other means of energy production. For sludge used at arable land, mineral fertiliser is replaced at replacement ratios of 0.5 for N and 0.7 for P (based on common LCA practice, Bengtsson et al. (1997)).
Substitution 1	Biogas assumed to be upgraded and used as vehicle fuel, replacing natural gas. Sludge used on arable land - as in baseline scenario.
Substitution 2	Biogas - as in baseline scenario. For replacement of mineral fertiliser replacement ratios of 0.3 for N (calculated from the readily available N in sludge and mineral fertiliser) and 1 for P (assuming that P leakage to air and water is considered to be negligible for both sludge and mineral fertiliser) are applied.
Substitution 3A	As baseline, but also accounting for the potential crop yield-increasing effect of the organic matter from sludge use in agriculture, using data on replaced crop yield calculated from the project Soilservice (Brady et al., 2012; Hedlund, 2012) and Börjesson et al. (2012).
Substitution 3B	As baseline, but also accounting for the potential crop yield-increasing effect of the organic matter and nutrients other than N and P from sludge use in agriculture, using data from Andersson et al. (2012).
Allocation 1	All impacts allocated to the sludge management.
Allocation 2	Economic allocation. Using data on prices for sludge treatment and biogas in Gothenburg (Gryaab, 2014) and the value of the yield increase due to sludge used in agriculture (Andersson et al. 2012), thereby allocating 70% of the impact to the sludge treatment.

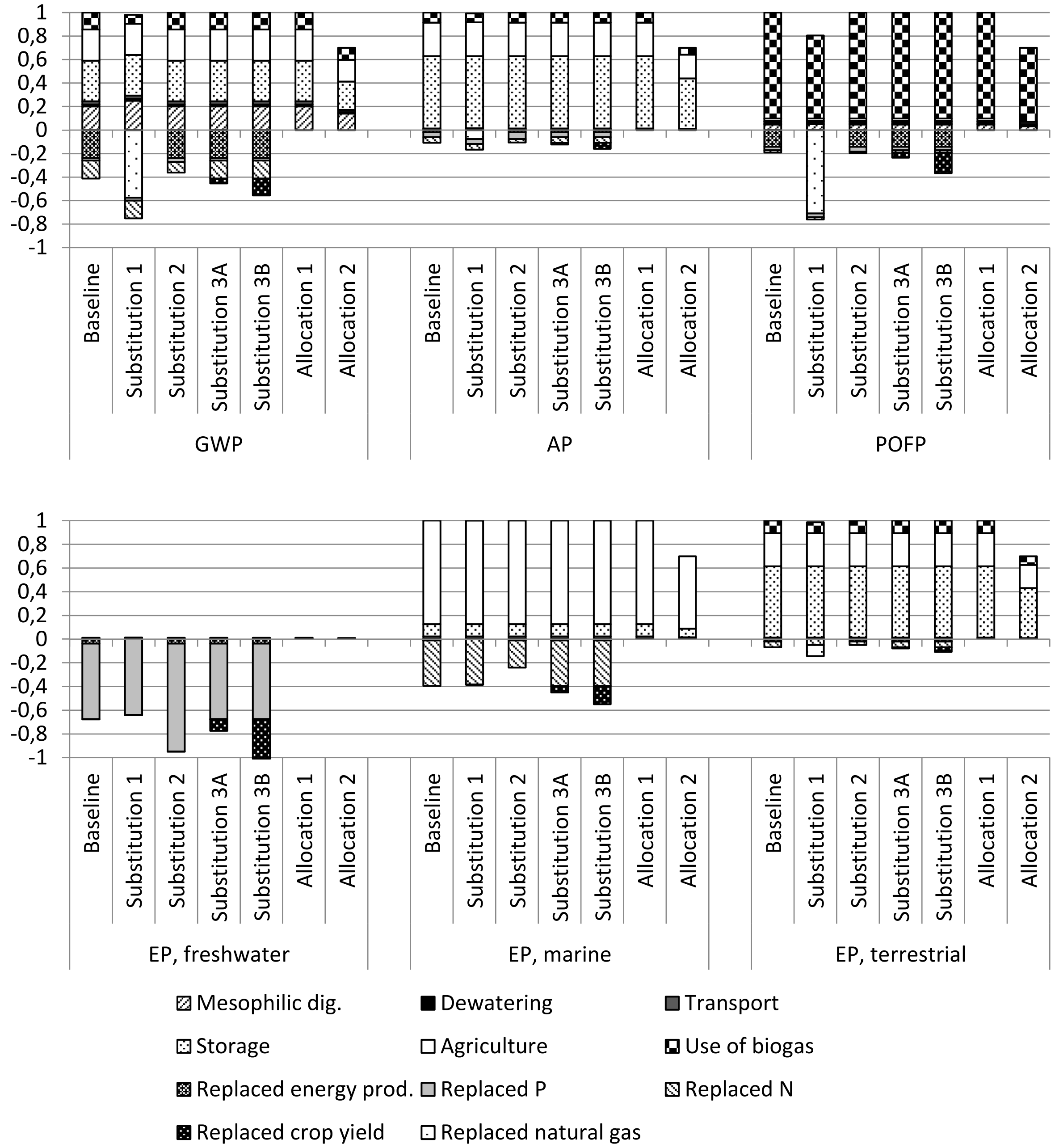

Figure 2. Gross life cycle impact assessment results (normalised against the largest absolute net result for each impact category, in order to make it possible to show results for several impacts in the same graph) for the studied model system, for the impact categories of global warming potential (GWP), acidification potential (AP), photochemical oxidant formation potential (POFP), and freshwater, marine and terrestrial eutrophication potential (EP), displayed per tonne dry matter of sludge entering the digestion. The different alternatives represent different ways of handling the multi-functionality in the system, see Table 1 for more details.

When substitution is applied, GWP and POFP are heavily affected by the modelling of the products assumed to be replaced, especially for the digester biogas, but also for the sludge on arable land. Freshwater and marine EP depends on the modelling of products replaced by sludge use on arable land. The results also indicate that accounting for benefits other than N and P in the sludge (Substitution 3A-B) can be important as it influences the overall results, although these first attempts of quantifying these benefits will need refinement. The time perspective considered for the substitution of benefits of the organic matter in sludge used in agriculture is very important as soil build-up takes time, why long-term studies are needed.

Despite the fact that P is often put forward as a main argument for sludge use on arable land results showed to be more important for replaced N than for P. Other impact categories, such as abiotic resource depletion, would perhaps better reflect the benefits of P recovery from sludge, and need future evaluation. Although the change in N replacement ratio between the baseline scenario and Substitution 2 yielded a moderate impact on the results, it indicates that the choice of markedly higher replacement ratios in some earlier studies (Heimersson et al., 2016) likely affects results substantially.

Attributing all impact to the sludge treatment function (Allocation 1) or using economic allocation (Allocation 2) gave the system equal or higher impact compared to if a substitution approach was applied. Due to the large uncertainty in the data on which the economic allocation relies, the exact approach used in Allocation 2 is not recommended for future studies. Rather, it can serve as inspiration in identifying more suitable ways to apply economic allocation.

It was noteworthy that for GWP and marine freshwater EP, the avoided emissions after mineral fertiliser use on arable land were more important for the results than the avoided production of the mineral fertiliser. Although the avoided emissions of the mineral fertiliser use phase have been included in LCAs on sludge management systems for a long time, (see e.g. Johansson et al. (2008) and Brown et al. (2010)), it is still not common practice (Heimersson et al., 2016), which it ought to be based on the results of the current study. Similar results were also shown by Schaubroeck et al. (2015).

The results shown in Figure 2 would probably be different if calculated for other regions than Sweden. The Swedish electricity mix relies largely on hydro- and nuclear power, and the district heating in Gothenburg relies largely on waste incineration and waste heat from industries, all with relatively low environmental impacts. Changing the geographical boundary to EU-27 would, for example, probably give a larger impact from the mesophilic digestion (due to its energy use) but also larger replaced environmental impacts from substituted CHP.

The model system in the current study was inventoried and assessed with an attributional inventory data approach in mind (using average data), which has been common also in earlier LCAs on wastewater and sludge management systems (Heimersson et al., 2016). The current study does not aim to provide guidance on which approach is preferable for handling multifunctionality issues in attributional and consequential studies respectively; for such guidance the reader is referred to the guideline documents (e.g. BSI (2011), ISO (2006) and EC-JRC (2010)) of choice for their study. As described in Section 1, a substitution approach can be relevant in both attributional and consequential studies, and differences could e.g. depend on the choice of data for production of the replaced product, as a consequential studies often rely to some extent on marginal data. Instead of assuming that the digester biogas would replace natural gas as vehicle fuel, a replacement of a more conventional fuel such as gasoline or diesel could be relevant although this would require a different kind of engine, and thus reflects a more long-term technology shift. Another example of how the data inventory could be adjusted to a consequential setting is when, as in Substitutions 3A-B, increased yields due to the organic matter provision to arable land through sludge is assumed to replace average crop production elsewhere in the region. If marginal crop production would instead be replaced, a decreased need for productive land (less non-cultivated land would globally have to be transformed to cropland to accommodate for the currently globally increasing needs for productive land to feed a growing population), or a decreased market prices of crops giving rise to different possible rebound effects, could be reasonable to include in the inventory. These two examples indicate the importance for the results of the choice of an average or marginal approach to data collection.

4 Conclusions

The results of the current study can be summarised in the following main conclusions:

- Substitution has historically been the predominant way of accounting for the benefits of digester biogas and sludge use on arable land. This approach was more easily applied than economic allocation to solve multi-functionality issues in this study.
- The LCA results for the model system are sensitive to both the choice of biogas use and to the modelling of replaced functions due to sludge use on arable land.
- The ratio at which the N in sludge replaces mineral N fertiliser was shown to be important for the overall LCA results. To base the substitution on the amount of mineral fertiliser that is actually, rather than theoretically, replaced is preferable in many situations.
- The potential effect of applying organic matter to arable land with no access to other organic amendment (e.g. farmyard manure) cannot be overlooked. On the contrary, the results of this study indicate that the yield increase and thereby replaced crop production elsewhere could be in the same order of magnitude as accounting for the avoided use of mineral fertilisers resulting from the N and P in the sludge. More research is however needed.

- Although a novel economic allocation was tested, the lack of relevant bases for allocation remains.

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6 References

Andersson, P.-G., 2012. Slamspridning på åkermark. Fältförsök med kommunalt avloppsslam från Malmö och Lund under åren 1981-2011. [In Swedish, Sludge spreading on arable land. Field trials with municipal sewage sludge from Malmö and Lund during 1981-2011]. Hushållningssällskapens rapportserie 16. Skåne, Sweden.

Bengtsson, M., Lundin, M., Molander, S., 1997. Life Cycle Assessment of Wastewater Systems. Technical Environmental Planning, Chalmers University of Technology, Göteborg, Sweden.

Berglund, M., Cederberg, C., Clason, C., Henriksson, M., Törner, L., 2009. Jordbrukets klimatpåverkan - underlag för att beräkna växthusgasutsläpp på gårdsnivå och nulägesanalyser av exempelgårdar. [In Swedish, Climate impact of agriculture - the basis for calculating greenhouse gas emissions at farm level and current situation analysis of example farms]. Hushållningssällskapet Halland, Sweden.

Brady, M., Hedlund, K., RongGang, C., Hemerik, L., Hotes, S., 2012. Farmer's costs of supplying soil services in diverse EU regions. Soilservice Deliverable D18.

Brentrup, F., 2015. Personal communication. Research Centre Hanninghof, Yara International ASA.

Brown, S., Beecher, N., Carpenter, A., 2010. Calculator tool for determining greenhouse gas emissions for biosolids processing and end use. Environ. Sci. Technol. 44(24), 9509-9515.

BSI, 2011. PAS 2050:2011 - specification for the assessment of the life cycle greenhouse gas emissions of goods and services. The British Standards Institute, London.

Börjesson, G., Menichetti, L., Kirchmann, H., Kätterer, T., 2012. Soil microbial community structure affected by 53 years of nitrogen fertilisation and different organic amendments. Biol. Fert. Soils 48(3), 245-257.

Börjesson, P., Tufvesson, L., Lantz, M., 2010. Livscykelanalys av svenska biodrivmedel. [In Swedish, Life Cycle Assessment of Swedish biofuels]. Lunds Tekniska Högskola, Institutionen för teknik och samhälle, Avdelningen för miljö- och energisystem, Sweden.

Cao, Y., Pawłowski, A., 2013. Life cycle assessment of two emerging sewage sludge-to-energy systems: Evaluating energy and greenhouse gas emissions implications. Bioresour. Technol. 127, 81-91.

Cherubini, F., Strømman, A.H., Ulgiati, S., 2011. Influence of allocation methods on the environmental performance of biorefinery products—A case study. Resour. Concerv. Recy. 55(11), 1070-1077.

Corominas, L., Foley, J., Guest, J.S., Hospido, A., Larsen, H.F., Morera, S., Shaw, A., 2013. Life cycle assessment applied to wastewater treatment: State of the art. Water Res. 47(15), 5480-5492.

Dalemo, M., Sonesson, U., Jönsson, H., Björklund, A., 1998. Effects of including nitrogen emissions from soil in environmental systems analysis of waste management strategies. Resour. Concerv. Recy. 24(3-4), 363-381.

Davis, J., Haglund, C., 1999. Life cycle inventory (LCI) on fertiliser production, Chemical Environmental Science. Chalmers University of Technology.

Doca, G., 2009. Life Cycle Inventories of Waste Treatment Services. Part IV "Wastewater treatment". in: EcoInvent (Ed.).

EC-JRC, 2010. ILCD Handbook - International Reference Life Cycle Data System, First ed. European Union.

EC-JRC, 2011. ILCD Handbook - International Reference Life Cycle Data System - Recommmendation for Life Cycle Impact Assessment in the Euroean context, First ed. European Union.

Garrigues, E., Corson, M.S., Angers, D.A., Van Der Werf, H.M.G., Walter, C., 2012. Soil quality in Life Cycle Assessment: Towards development of an indicator. Ecol. Indic. 18, 434-442.

Gryaab, 2014. Gryaabs verksamhet 2014. http://www.gryaab.se/arsredovisning2014/ (assessed 2015-10-18).

Harder, R., Heimersson, S., Svanström, M., Peters, G.M., 2014. Including pathogen risk in life cycle assessment of wastewater management. 1. Estimating the burden of disease associated with pathogens. Environ. Sci. Technol. 48(16), 9438-9445.

Harder, R., Peters, G.M., Svanström, M., Khan, S.J., Molander, S., 2016. Estimating human toxicity potential of land application of sewage sludge: the effect of modelling choices. Int. J. Life Cycle Ass., 1-13.

Hedlund, K., 2012. SOILSERVICE, Conflicting demands of land use, soil biodiversity and the sustainable delivery of ecosystem goods and services in Europe.

Heijungs, R., 2014. Ten easy lessons for good communication of LCA. International Journal of Life Cycle Assessment 19(3), 473-476.

Heimersson, S., Svanström, M., Laera, G., Peters, G., 2016. Life cycle inventory practices for major nitrogen, phosphorus and carbon flows in wastewater and sludge management systems. Int. J. Life Cycle Ass. 21(8), 1197-1212.

Hermann, B.G., Debeer, L., De Wilde, B., Blok, K., Patel, M.K., 2011. To compost or not to compost: Carbon and energy footprints of biodegradable materials' waste treatment. Polym. Degrad. Stab. 96(6), 1159-1171.

ISO, 2006. ISO 14044: Environmental Management - Life Cycle Assessment - Requirements and guidelines.

Johansson, K., Perzon, M., Fröling, M., Mossakowska, A., Svanström, M., 2008. Sewage sludge handling with phosphorus utilization - life cycle assessment of four alternatives. J. Clean. Prod. 16(1), 135-151.

Jordbruksverket, 2014. Riktlinjer för gödsling och kalkning 2015. [In Swedish, Guidelines for fertilization and liming 2015] http://www2.jordbruksverket.se/download/18.2b94dc5814a2e549b27e9cb7/1418215478895/jo14_12.pdf (assessed 2015-09-28).

Jönsson, H., Junestedt, C., Willén, A., Yang, J., Tjus, K., Baresel, C., Rodhe, L., Trela, J., Pell, M., Andersson, S., 2015. Minska utsläpp av växthusgaser från rening av avlopp och hantering av avloppsslam [In Swedish: Reduce greenhouse gas emissions from the treatment of wastewater and the handling of sewage sludge]. Svenskt Vatten Utveckling, Bromma, Sweden.

Koffler, C., 2014. Reply to "Ten easy lessons for good communication of LCA" by Reinout Heijungs (Int J Life Cycle Assess 19(3):473-476. DOI: 10.1007/s11367-013-0662-5). International Journal of Life Cycle Assessment 19(5), 1170-1171.

KSLA, 2012. Agronomiska perspektiv på slam som gödselmedel. Slam som produktionsresurs i svenskt jord- och skogsbruk – I. [In Swedish, Agronomistic perspective of sludge as fertilizer. Sludge production resource in Swedish agriculture and forestry]. Workshop 18 april 2012. The Royal Swedish Academy of Agriculture and Forestry, http://www.ksla.se/wp-content/uploads/2012/10/KSLA-slam-Workshop-1-Agronomiska-perspektiv.pdf (assessed 2015-07-07).

Linderholm, K., 2011. Fosfor och dess växttillgänglighet i slam - en litteraturstudie. [In Swedish, Phosphorus and its plant availability in sludge - a literature review] Svenskt Vatten Utveckling, report no. 2011-16. Sweden.

Linderholm, K., Tillman, A.M., Mattsson, J.E., 2012. Life cycle assessment of phosphorus alternatives for Swedish agriculture. Resour. Concerv. Recy. 66, 27-39.

Metcalf & Eddy Inc, Tchobanoglous, G., Burton, F.L., Stensel, H.D., 2004. Wastewater engineering, treatment and resure, fourth edition ed. McGraw-Hill, New York.

Milieu Ltd, WRc, RPA, 2010. Environmental, economic and social impacts of the use of sewage sludge on land. Final report, Part III: Project Interim Reports. European Commission, DG Environment.

Mills, N., Pearce, P., Farrow, J., Thorpe, R.B., Kirkby, N.F., 2014. Environmental & economic life cycle assessment of current & future sewage sludge to energy technologies. Waste Manage. (Oxford) 34(1), 185-195.

Nemecek, T., Kägi, T., 2007. Life Cycle Inventories of Agricultural Poduction Systems, Data v2.0. ecoinvent report No. 15. ecoinvent centre, Swiss Centre for Life Cycle Inventories, Zûrich, Schwitzerland and Dûbendorf, Germany.

Nordahl, O., 2013. Biogas från VA. Prissättning och hantering av affärsrisker. [In Swedish, Biogas from water services. Pricing and management of business risks]. Svenskt Vatten Utveckling report 2013-19, Stockholm, Sweden.

Palm, D., Ek, M., 2010. Livscykelanalys av biogas från avloppsreningsverksslam [In Swedish, Life Cycle Assessment of biogas from sewage sludge]. Svenskt gastekniskt centrum, report SGC 219, Sweden.

Peters, G.M., Rowley, H.V., 2009. Environmental comparison of biosolids management systems using life cycle assessment. Environ. Sci. Technol. 43(8), 2674-2679.

Pizzol, M., Weidema, B., Brandão, M., Osset, P., 2015. Monetary valuation in Life Cycle Assessment: A review. J. Clean. Prod. 86, 170-179.

Pradel, M., Aissani, L., Villot, J., Baudez, J.C., Laforest, V., 2016. From waste to added value product: Towards a paradigm shift in life cycle assessment applied to wastewater sludge - A review. J. Clean. Prod. 131, 60-75.

REVAQ, 2016. REVAQ, Renare vatten - bättre kretslopp, Regler för certifieringssystemet. Utgåva 3.3 [In Swedish, Cleaner water - more recycling, Rules for the certification system. Issue 3.3]. Stockholm, Sverige.

Sandin, G., Røyne, F., Berlin, J., Peters, G.M., Svanström, M., 2015. Allocation in LCAs of biorefinery products: Implications for results and decision-making. J. Clean. Prod. 93, 213-221.

Schaubroeck, T., De Clippeleir, H., Weissenbacher, N., Dewulf, J., Boeckx, P., Vlaeminck, S.E., Wett, B., 2015. Environmental sustainability of an energy self-sufficient sewage treatment plant: Improvements through DEMON and co-digestion. Water Res. 74, 166-179.

Schrijvers, D.L., Loubet, P., Sonnemann, G., 2016. Critical review of guidelines against a systematic framework with regard to consistency on allocation procedures for recycling in LCA. International Journal of Life Cycle Assessment 21(7), 994-1008.

SGC, 2012. Basic Data on Biogas. Svenskt Gastekiniskt Centrum, Sweden.

Singh, R.P., Agrawal, M., 2008. Potential benefits and risks of land application of sewage sludge. Waste Manage. (Oxford) 28(2), 347-358.

Statistics Sweden, 2014. Production of cereals, dried pulses and oilseeds in 2014. Preliminary statistics for counties and the whole country.

Stutter, M.I., 2015. The composition, leaching, and sorption behavior of some alternative sources of phosphorus for soils. Ambio 44(2), 207-216.

Svanström, M., Laera, G., Heimersson, S., 2015. Problem or Resource - Why It Is Important For the Environment to Keep Track of Nitrogen, Phosphorus and Carbon in Wastewater and Sludge Management. J Civil Environ Eng 5:200.doi:10.4172/2165-784X.1000200.

UNEP, 2002. Environmentally sound technologies in wastewater treatment for the implementation of the UNEP global programme of action (GPA) "Guidance on municipal wastewater". http://www.unep.or.jp/ietc/publications/freshwater/sb_summary/10.asp (assessed: 2015-10-22).

Westerhoff, P., Lee, S., Yang, Y., Gordon, G.W., Hristovski, K., Halden, R.U., Herckes, P., 2015. Characterization, Recovery Opportunities, and Valuation of Metals in Municipal Sludges from U.S. Wastewater Treatment Plants Nationwide. Environ. Sci. Technol. 49(16), 9479-9488.
VKM, 2009. Risk assessment of contaminants in sewage sludge applied on Norwegian soils. Norwegian Scientific Committee for Food Safety (VKM), Oslo, Norway.