Life cycle assessment of sludge management with phosphorus utilisation and improved hygienisation in Sweden

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ABSTRACT

To provide input to sewage sludge management strategies that address expected new regulations in terms of hygienisation and phosphorus recovery in Sweden, an environmental life cycle assessment (LCA) was made. The LCA identified environmental hot spots for methods that may permit sludge or phosphorus from sludge to be applied on agricultural land. In particular, thermophilic digestion, pasteurisation, thermal hydrolysis, urea treatment and mono-incineration with phosphorus recovery were compared. In addition, a sludge management system involving drying of sludge before use in forestry was investigated. The results showed that some major impacts are related to large uncertainties, such as those related to emissions from sludge storage. It also showed that large gains can be achieved when products from the systems replace other products, in particular when biogas is used to replace natural gas in vehicles, but also when sludge is used in agriculture and forestry. In general, there are small differences between the sludge management methods. Retaining the sludge matrix to allow for its utilisation in agriculture may conflict with keeping emissions to air and water from the sludge matrix low. It is recommended that any sludge management option minimises emissions from sludge to air and water and that resources are recovered and used, in line with the principles of a circular economy.

Key words | environmental assessment, LCA, phosphorus recovery, sewage sludge treatment

INTRODUCTION

Appropriate strategies for sewage sludge management have been debated for a long time (see e.g. Bengtsson & Tillman 2004). Risks related to heavy metals, organic micropollutants and pathogens create concern, but on the other hand, the need to recycle nutrients like phosphorus is increasingly discussed as an important part of a circular economy. In Sweden, new regulations for the recovery of phosphorus are expected to demand a higher level of sludge hygienisation before agricultural (or other) use (SEPA 2013). Currently, mesophilic anaerobic digestion is typical at Swedish wastewater treatment plants (WWTPs), and after subsequent storage for six months, the sludge is normally Salmonella-free and by that considered sufficiently hygienised to be spread as a fertiliser on agricultural land, according to the Swedish industry certification system Revaq (2016). In new, stricter legislation, this will probably not be sufficient as certain log reductions are proposed also for viruses and parasites. Further, phosphorus recovery from wastewater and sludge is also expected to be required in order to adhere better to the principles of a circular economy. Therefore, many wastewater treatment (WWT) organisations in Sweden are currently reviewing their sludge management strategies to prepare for the upcoming regulations. Environmental life cycle assessments (LCAs) have been used since the 90s to provide environmental input to decision-making on wastewater and sludge
management (see e.g. Corominas et al. (2015) and Yoshida et al. (2015) for reviews) and offer opportunities to, for example, discover hot spots in the life cycle or to compare different alternative options.

The Swedish Water & Wastewater Association (SWWA) and six different stakeholder organisations managing WWTPs in Sweden funded a project to provide input to new sludge management strategies. In 2015 and early 2016, the project compared different sludge management options that were assumed to meet the new hygienisation requirements and that involved the recovery of phosphorus. Researchers at Chalmers University of Technology performed an LCA, with active participation of representatives from all funding organisations. This paper reports on some of the results from the project.

**METHODS**

The studied sludge management options all involve the application of sludge or products from sludge in either agriculture or forestry (see Table 1) and they all also involve anaerobic digestion of sludge for the production of biogas. The reference system represents typical sludge treatment in Sweden with mesophilic anaerobic digestion, and assumes that all sludge is dewatered and used in agriculture. As this sludge management approach will likely not fulfil future hygienisation requirements, it was not intended to be included in the comparison among future sludge management options. The functional unit (the calculation basis) was selected to be suitable for decision-makers in the management of WWTPs, and answering the question: ‘Which available sludge management options have the lowest environmental impact and what are the hot spots for each alternative?’ The functional unit was: environmental impact per (metric) tonne of dry solids of undigested sludge (with a composition as for the Gryaab WWTP (Ryaverket) in Gothenburg, Sweden (see Mattsson 2015)). Since the selected approach aimed to describe the environmental impacts for the studied technologies in general, rather than studying the upgrading in the form of rebuilds or implementation of new process units in the sludge management at a particular plant, in LCA terms the study had an attributional rather than a consequential scope (see e.g. Ekvall et al. 2016). The reconstruction or the change itself was thus not in focus but rather the inherent characteristics of the selected sludge management options.

The environmental impact categories that were assessed in the study include the impacts most commonly assessed in similar studies, i.e., global warming, acidification, eutrophication (for marine, freshwater and terrestrial ecosystems) and photo oxidant formation potential, and were selected based on the relevance for the question at hand and the particular alternative sludge management routes to be studied (Corominas et al. 2015). The characterisation methods recommended in the ILCD handbook (EC-JRC 2011) were used. Collaboration with another on-going Swedish project, Formas LiCRA, made it possible to also look at human toxicity impact and pathogen risk for some studied alternatives, but this will only be briefly touched upon in this paper. Abiotic depletion potential was not assessed in detail; the most interesting aspect in this regard, mined phosphorus reserves, was expected to relate to only one of the activities in the life cycles of the sludge management systems, namely the saved mineral fertilisers, and therefore, a detailed inventory of the full life cycle was not considered useful.

The studied systems encompassed the sludge management and the end-use of the sludge (see Figure 1) but not the wastewater collection and the WWT. Construction of facilities, equipment and vehicles was not included, based on the low share of total impacts that is expected from the construction phase for systems like this (e.g. 3–7% in Schulz et al. 2012). The present study took into account the impacts on the environment both from direct emissions and from production and use of energy and chemicals consumed, as well as the benefits of resource recovery (energy from biogas and sludge incineration, and fertilising nutrients). Such additional functions of the systems were handled by means of substitution, which is the most common and also the recommended approach for studies
of this kind (see Heimersson et al. 2016a). This means that the systems were credited for alternative means of generating the same functions. The study was made for current Swedish conditions, and focused on technologies available today or in the near future and the way they would be implemented today. Sensitivity analyses were made in the interpretation of results, to allow for an understanding of the implications of varied assumptions or different contexts.

For managing the LCA data inventory and impact assessment, the GaBi software tool was used (www.gabi-software.com) and for some more generic data, the GaBi Professional or the ecoinvent (www.ecoinvent.org) databases were used. Some details on the data inventory are found in Table 2. The study is described in more detail in an SWWA report (in Swedish; Svanström et al. 2016).

RESULTS AND DISCUSSION

Some results will be presented and discussed in this paper; for further details, the reader is referred to the SWWA report (Svanström et al. 2016). In the graphs presented below, sections of the bar above zero are related to emissions from different parts of the system and represent an environmental impact; sections below zero represent avoided emissions related to the substitutions.

Climate impact

Results in terms of the climate impact are shown in Figure 2. In general, emissions and avoided emissions are of similar magnitude, and it is clear that the choice to manage multifunctionality by means of substitution affects the results (for alternatives to substitution for handling multifunctionality, see Heimersson et al. (2016a)). The relative significance of the avoided emissions is consistent with the outcomes of an LCA done in an Australian sludge management context (Peters & Rowley 2009), if the extreme delivery distances for Australian sludge are discounted (and hence the greater transportation impacts). Transportation was also important in some pioneering LCA work in the UK (Dennison et al. 1998) where, consequently, the reduction in sludge volume was the key to impact reduction.

Methane leakage related to anaerobic digestion is an important emission in all systems. Compared to the mesophilic digestion in other systems, the thermophilic digestion has higher impacts because of increased methane leakage and increased heat demand. However, combining all elements involved in sludge hygienisation in each system, this higher climate impact for the thermophilic digestion is not important (add, for example, pasteurisation and thermal hydrolysis to the digestion for those respective systems). To explore the effect on the results of tripling the methane leakage (not an uncommon level found in the literature (Dong et al. 2014; Mills et al. 2014)), a sensitivity analysis was made for the reference system, see Figure 3. This can clearly have an important effect.

A surprisingly large impact can also be seen from the sludge storage, for all methods where a sludge matrix still remains and is kept in open storage awaiting spreading on agricultural land; this is the case for the reference system, thermophilic digestion, thermal hydrolysis and...
pasteurisation. This relates to emissions of both nitrous oxide and methane. Impacts from sludge storage are generally not included in LCAs on sludge management (Heimersson et al. 2016b) and unfortunately, data are still scarce and cannot be modelled in detail to reflect the specific sludge treatment methods studied here (although important recent efforts have been made by Willén (2016)). Therefore, this study highlights the importance of further studying actual storage emission levels from sludge treated in different ways. A large difference can be seen for the urea treatment alternative, as the sludge storage is considered to be covered for this alternative because of the need to keep the formed ammonia in the sludge heap so that it can fulfill its function of changing the pH value for the hygienisation of the sludge. For short-time storage of sludge at the WWTP before incineration, no nitrous oxide emission data were available; the contribution from storage could therefore be under-estimated for this system, since nitrous oxide is a very potent greenhouse gas. Covered sludge storage, as in the urea case, could be an option also for sludge treated in other ways, and a sensitivity analysis was therefore made to check the possible effect of covering mesophilically digested sludge, as data were available for this variation in Jönsson et al. (2015). However, this actually did not affect the results for the reference system.

**Figure 2** | The climate impact of the studied sludge management systems, as global warming potential; the functional unit is one tonne of dry solids of undigested sludge.

**Figure 3** | The climate impact of the reference system in the first bar, with methane leakage related to anaerobic digestion tripled in the second bar, and with sludge storage varied from open to closed in the third bar, as global warming potential; the functional unit is one tonne dry solids of undigested sludge.
notably, see Figure 3, because the decrease in nitrous oxide emissions was partly compensated for by an increase in methane emissions. Also, after spreading on agricultural soil, important climate impacts arise from emission to air for systems where the sludge matrix still remains.

The largest gains in terms of climate impact stem, in our case, from replaced natural gas when the biogas is used in heavy vehicles, see further discussion later in this paper. Biogas generation was considered to be the same as for the reference system except for an increase in generation by 35% for thermal hydrolysis (Sandbacka 2015) and 3% for thermophilic digestion (Hellstedt et al. 2009). Replaced mineral nitrogen fertiliser becomes, for the climate impact, important primarily for the system where a sludge product is spread in forests, because of the high considered nitrogen replacement rate (0.85, given as nitrogen in sludge over nitrogen in replaced mineral fertiliser (Sahlén et al. 2015), compared to 0.3 for sludge on arable land (Heimersson et al. 2016a)).

### Acidification and terrestrial eutrophication

For both acidification and terrestrial eutrophication, emissions of ammonia to air turned out to determine the results; important contributions are from sludge storage and from agricultural land, after spreading. Only acidification results are shown, Figure 4, as terrestrial eutrophication results exhibit the same trends. The quantification of emissions to air from storage is, as earlier mentioned, to be regarded as highly uncertain. They are based on studies of storage of different types of manure and sewage sludge as reported in Jönsson et al. (2015) and based on Karlsson & Rodhe (2002). Note that for the short-term storage before incineration, no ammonia emissions were accounted for (due to lack of data). Emissions during storage have generally not been accounted for in earlier LCAs. Exceptions are Jönsson et al. (2015), using the same data source as the current study, and Tidåker et al. (2006), who assumed much lower ammonia emissions, as a result of a covered sludge storage.

Results are very similar for thermophilic digestion, thermal hydrolysis and pasteurisation, but emissions are lower for the urea treatment case, although they are shifted from sludge storage to agriculture because of the covered storage in that case (which is necessary to keep the ammonia in the sludge for the hygienisation). It is clear that when biological processes are allowed to continue to act in the sludge they may give rise to important emissions, and that sludge treatment methods that allow for such emissions may receive a disadvantage. Assumptions on the magnitude of ammonia emissions from open sludge storage and from agriculture are clearly important. The comparatively low impact from the scenario in which sludge is dried and used in forestry is likely at least partially a result of the lack of data on ammonia emissions to air during drying. To explore the effect on the results of varying assumptions on the type of sludge storage and on the level of ammonia emissions during drying, sensitivity analyses were made, see Figure 5 for results for the acidification impact (results are, again, similar for terrestrial eutrophication). When varying ammonia emissions from sludge storage for the reference system to represent covered storage (using data from Jönsson et al. (2015), calculated based on Karlsson & Rodhe (2002)) instead of open storage (compare first to the second bar in Figure 5),
this reduces the total impact dramatically. A large increase, on the other hand, can be seen for the drying system when assuming that half of the nitrogen lost during drying leaves as ammonia instead of the zero emissions assumed in the base case (compare the third to the fourth bar in Figure 5).

For both these impact categories (acidification and terrestrial eutrophication), emissions are for many systems much greater than the gains that can be achieved from replacing other products, but this conclusion is strongly dependent on the assumption of emission levels from open sludge storage and from agriculture. Differences between different types of technologies are larger than for the climate impact, again depending on this assumption. For acidification and terrestrial eutrophication, minimisation of ammonia emissions in sludge management appears to be critical.

**Eutrophication in freshwater and marine ecosystems**

For other eutrophication categories, emissions to water instead turn out to be dominant: phosphate for freshwater ecosystems (not shown) and nitrate for marine ecosystems (Figure 6). For freshwater eutrophication, it is the phosphate emissions from mineral fertiliser production that dominates, and therefore, large gains can be achieved when phosphorus in sludge replaces mineral phosphorus fertiliser. The relative outcomes of the sludge management alternatives therefore follow the assumed phosphorus utilisation rates for the
freshwater eutrophication category, with a clear advantage to alternatives that are considered to replace more phosphorus fertiliser. The actual phosphorus replacement rates for sludge on arable land are highly uncertain (Linderholm 2011) and the phosphorus availability in the AshDec product has also been questioned (Bäfver et al. 2013). In both cases, a replacement rate of 70% has been assumed in this study (based on Heimersson et al. 2016a).

For marine eutrophication (Figure 6), nitrate emissions to water after spreading of sludge in agriculture or forestry are instead dominant. This gives a disadvantage, again, to systems where the sludge matrix is maintained for use of the nitrogen as a fertiliser in agriculture or forestry. As can be seen in Figure 6, when mineral nitrogen fertilisers have been spread, this also gives rise to similar emissions, but generally of smaller magnitude. These saved emissions become important in particular for the drying alternative, as more nitrogen fertiliser is considered to be replaced in this case. Clearly, the incineration system gets an advantage for this environmental impact category, even if no nitrogen can be recovered, as the release of nitrogen to the environment is controlled in that case. This emphasises the importance of minimising nitrate leakage after spreading by utilising sludge in the right way, at the right place and at the right time.

**Smog formation**

Results for smog formation are dominated by NOx formed during combustion of the biogas in heavy vehicles and during the replaced combustion of natural gas, which is of similar magnitude, and the net results are therefore close to zero for all compared systems; results are not shown here.

**Modelling of electricity, heat and biogas use**

Use of both heat and electricity appears in many different activities in the studied systems, and it may influence results more than can be understood from Figures 2–6. The energy use for hygienisation was obtained from references quantifying the increased energy need if Ryaverket would implement pasteurisation, thermophilic digestion (Paulsrud & Balmér 2008) or thermal hydrolysis (Sandbacka 2015). The data for thermal hydrolysis did not include post-treatment energy recovery, although this would be possible to implement.

A potentially important difference between Sweden and other countries is how heat and electricity are supplied. To check what the effect would be of considering instead a more common energy setting in Europe, a sensitivity analysis was made for the reference system with average EU27 electricity (for 2011, the main contributors were 27.7% nuclear, 21.3% natural gas, 15.0% hard coal, 10.7% lignite, 10.3% hydro and 5.5% wind) and with a natural gas heater for heat provision (GaBi Professional database); for details on the base case see Table 2. Comparing the Reference to Scenario 1 in the left part of Figure 7 reveals that this shift between energy systems from typical Swedish to typical European clearly increases the climate impact.

For many impact categories, the use of the biogas to replace natural gas in heavy vehicles provides an important benefit. This was the choice in the base case, as most of the
biogas produced at Swedish WWTPs is used as a vehicle fuel (57% in 2014, www.biogasportalen.se) and this is the way that biogas is actually used in Gothenburg; typically, a large share is used in buses and garbage trucks. Another common situation in Europe is that the generated biogas is used to generate heat and electricity (using combined heat and power, CHP). The effect of using biogas to generate energy was explored by removing the biogas upgrading, and diverting the biogas instead to direct combustion with CHP and with electricity and heat considered to be replaced, either in the Swedish energy setting (Scenario 2) or in a European energy setting (Scenario 3). An interesting effect is that the gains from replacing heat and electricity are lower in Sweden and thereby, the net climate impact for the system is higher in a Swedish energy setting. Gains in a European setting, however, are much higher than the slight increase in impact from increased use of heat and electricity, and gains are also even larger than when replacing natural gas use in vehicles. For smog formation, shifting the biogas use from vehicle fuel to CHP in a Swedish setting strongly affects the results (Scenario 2 in the right part of Figure 7), as fewer NOx-forming activities are then replaced. Finally, had a more consequential approach been applied in the study, indirect market effects could have been considered for the use of the biogas. It could, for example, be assumed that if biogas is sold at a certain scale and for some time, it results in more gas-fuelled vehicles being driven on our roads and fewer diesel-fuelled ones, thereby actually replacing diesel use in vehicles instead; this is shown as Scenario 4 in Figure 7. This lowers both the net climate impact and the net smog formation from the system. For the biogas, it is thus important to know to what use the gas will be put, and what other products/services it will likely replace in the actual technical system in which it is embedded. However, the variations discussed in this section will not strongly affect the relative performance of the compared options.

Modelling of benefits of land application of sludge and sludge products

Also, when sludge or sludge products are used in agriculture or forestry, thereby replacing the use of mineral fertilisers, benefits can be achieved. However, these benefits are smaller than has been seen in many other earlier LCA studies, primarily because new inventory data for European mineral fertilisers, reflecting current production, were used instead of the data from the 1990s that have been used in earlier LCAs (see Heimersson et al. (2016a) for examples and further details). The assumed replacement rates for nitrogen and phosphorus are also lower than in some earlier studies, as the intention was to model what is actually replaced by the farmer or the forester and not a potential that is not likely to be realised. Varying replacement rates, however, will not change results notably as the contribution to the total impact is generally already small (see also Heimersson et al. 2016a). Replacing mineral nitrogen fertiliser, however, appears to be more important than replacing phosphorus mineral fertiliser for the studied categories, with the exception of freshwater eutrophication.

It has been suggested that there are other benefits that can be achieved when sludge is land applied than the utilisation of nitrogen and phosphorus in the sludge as a fertiliser, e.g., related to other nutrients and the soil carbon content. Peters & Rowley (2009) suggested that the water retention capacity of soil improves, and they made an effort to quantify this effect in LCA. To explore the potential effect of some benefits of land application of sludge, an innovative approach was developed and applied in the present study that was based on observed increases in harvest yields, that may be resulting from, e.g., the increase in soil carbon content, and indirect effects thereof (e.g., the increase in uptake of nitrogen), as well as the addition of other important nutrients such as potassium. The results revealed that the gains related to such benefits could potentially be more important than the gains from replacing mineral fertilisers (Heimersson et al. 2016a). However, data are scarce and the results are highly uncertain. More data are needed, and methods can likely be further refined.

Further, it has been suggested that an increase in soil carbon content can have an important effect on the climate impact as it may lead to the sequestration of carbon in soil. However, it was shown by Svanström et al. (2015) that the effect on the climate impact of letting 10% of the carbon in sludge be permanently stored in soil, which is a strong exaggeration of the potential effect, was not important.

Resource utilisation and abiotic resource depletion

One of the main ideas in the selection of sludge management routes for this study is that phosphorus in wastewater should be utilised to avoid depletion of mineral phosphorus reserves, in line with the principles of a circular economy. The discussion on ‘peak’ phosphorus (see e.g. Neset & Cordell 2012) has triggered much research and development in this field (e.g. the EU FP7 project P-REX, http://p-rex.eu/). In the suggestion for new phosphorus recycling targets in Sweden (SEPA 2013), 40% of the phosphorus
in sewage should be utilised on arable land. Strictly speaking, recycling to forest land does not qualify. When applying sludge to agricultural soil, practically all phosphorus in sewage will end up on arable land. However, for sludge incineration with phosphorus recovery, there will likely be some losses of phosphorus. The Ashdec technology studied here, however, is not an extraction process but instead treats the incineration ashes to vaporise heavy metals – the phosphorus remains in the ashes, which are then incorporated into fertiliser products. All studied systems apart from the forestry system will therefore fulfil the expected demand of at least 40% phosphorus utilisation. However, whether the phosphorus in sludge will actually replace the use of mined phosphorus will ultimately be determined by whether farmers will actually use less mineral fertilisers. The potential to recover and utilise phosphorus might actually be higher than assumed in this study, but the practical results may in fact be even lower as farmers will not risk inadequate nutrient supply. For forestry, there is a lack of data on the impact of phosphorus in sludge on the growth of trees. No phosphorus fertiliser is added in forestry today; only nitrogen fertiliser and only in certain regions (in Sweden, only in the northern parts); therefore, no mineral phosphorus fertiliser was considered to be replaced in forestry in this study.

It has also been suggested for the new regulations (SEPA 2013) that 10% of the nitrogen in sewage should be utilised on arable land. On its own, an incineration process would not fulfil this demand as all nitrogen is lost in incineration. Furthermore, application of nitrogen-rich sludge on forest land would not qualify, as use on agricultural land is targeted in the suggested regulations. All other studied systems would easily fulfil the demand. Nitrogen in mineral fertiliser comes from the technical and highly energy-demanding fixation of nitrogen from air. Nitrogen in air is not seen as a scarce resource and the rationale behind the suggested utilisation rate for nitrogen is therefore different than for phosphorus (‘peak phosphorus’ or maybe better the scarcity of P with low content of cadmium and other contaminants). It can be argued that the carbon in sludge also should have a high utilisation rate in a circular economy and be turned into e.g. biogas or biopolymers, or be utilised on arable land, or for energy recovery.

As mentioned earlier, abiotic resource depletion was not included as an impact category in this study and therefore, the inventory did not put any focus on finding complete and relevant data for these aspects. However, as much data used in this study were extracted from databases, much data on abiotic resource use were automatically included. Running the abiotic resource depletion impact assessment (using the method recommended by the ILCD Handbook (EC-JRC 2011)) revealed that it is in fact, as expected, the saved mineral phosphorus fertiliser that has a dominant contribution for our systems. However, data have not been checked in detail for completeness and consistency. Had the construction of equipment, facilities and vehicles been included in the systems, this would have resulted in abiotic resource depletion in the form of different materials used, but this is not expected to be more important than the replaced mineral phosphorus. This would be consistent with a previous Australian study that showed operational resource consumption to be much greater than resource consumption for construction (Peters & Lundie 2001).

Hygienisation requirements and pathogen risk

The studied systems were set up to follow the Swedish Environmental Protection Agency’s suggestion for new requirements (SEPA 2013), which serves as input to the new legislation. All studied alternatives are expected to fulfill the new hygienisation requirements, except for the reference system. Pathogen risk is not an existing impact category in the LCA. In the Formas LiCRA project, an effort was made to quantify pathogen risk in an LCA framework (Harder et al. 2014). Approaches were borrowed from quantitative microbial risk assessment to model exposure pathways and effects on human health (in disability adjusted life years, DALYs) of 10 pathogens for both the WWTP effluent and the land application of sludge. The results were scaled to the size of a WWTP, and put in an LCA framework alongside other impact categories in an end-point characterisation for human health (by means of certain weighting factors, all impact categories were translated into their impact on human health in the unit of DALYs, see further details in Heimersson et al. 2014). Due to lack of data and inherent uncertainties of the novel method, results are highly uncertain but, nevertheless, showed that pathogen risk can make up a significant part of the total impact on human health. This is therefore a field that should be further explored. The purpose of strengthening the sludge hygienisation requirements, however, is to reduce risks related to pathogens, and pathogen risks can therefore be expected to be lower in the future.

Metals and organic pollutants and their toxicity impact

Other concerns that are reflected in the current sludge debate are risks related to the content of metals and organic
pollutants in land applied sludge. Impact assessment methods in LCA that address human toxicity and ecotoxicity are still quite immature, and there are also large data gaps in terms of both concentrations and the actual effect of exposure to individual contaminants. Also, by necessity, LCA impact assessment models are quite generic and may be less adequate in some contexts. Further, mixture toxicity is completely disregarded in impact assessment in LCA, as characterisation is based on individual contaminants. An attempt was made within the Formas LiCRA project to model the human health impact of metals and organic contaminants, both using the existing consensus method for impact assessment (USEtox) and developing an approach more specific to land application of sludge, borrowing approaches from quantitative chemical risk assessment. The results were introduced into an LCA framework to reveal the importance of the human toxicity compared to the total impact on human health, similar to what was done for pathogen risk, as described above. Human toxicity showed to potentially have an even larger impact than pathogen risk (see Harder et al. (2017) for more details). Metals generated a larger total impact than organic pollutants. One particularly interesting result that warrants further studies is that zinc in the sludge that is land applied showed up as a dominant contributor to health impacts (Harder et al. 2017), which is not expected and probably because of an anomaly in the model as zinc is an essential element that many people even suffer from a lack of. It should be noted that also metals in the flue gases from sludge incineration turned out to have an important relative impact. Again, uncertainties are very large and the main conclusions from the study was that this is an area that needs further work, both in terms of gathering data and in terms of developing assessment methodology. Current and future efforts to reduce pollutant levels in wastewater and sludge by source reduction upstream of the plant, however, will likely lower the toxicity impact when sludge is land applied. The Swedish Revaq initiative that was started in 2002 is a good example of a source reduction effort; the initiative also offers a certification system for sludge (Persson et al. 2015).

**Further remarks**

The choice of system boundaries and functional unit were deemed reasonable for the specific study but presented different challenges. For example, the choice of starting from the undigested sludge and ignoring earlier parts of the WWTP disregarded the effect of recycled flows to these parts.

It should be noted that WWT is not included in the assessed system in this study and therefore the release of the main WWTP effluent to recipients is, for example, not considered. To check whether the sludge management is in fact an important part to focus on in the whole WWT system, other parts of the WWT were added in a special assessment. This comparison revealed that for all studied impact categories, except for marine eutrophication, sludge management contributes the majority of the total results. It is therefore relevant to focus on improving this part of the wastewater and sludge management.

To check the potential contribution of sludge management to Sweden’s total burden on the environment, a comparison was done between emissions for the reference system (without substitutions), recalculated as to represent climate impact, ammonia emissions and NOX emissions per person, and the corresponding national data per Swede as reported by the Swedish EPA for 2014 (SEPA 2016). Sludge management contributed about 0.4, 2.4 and 0.4% respectively, to total national emissions, which shows the non-negligible contribution from sludge management to total Swedish emissions, in particular for ammonia (most other national ammonia emissions are related to agricultural activities).

Note that this paper reports on an environmental LCA and that other aspects of the studied technologies, such as economy, technical difficulties in implementation, problems that can arise related to legal compliance or public acceptance or similar were not evaluated. Any implementation effort should be preceded by a thorough study of such aspects and also a new LCA that explores the particularities of the actual case in detail. Also, the study was not designed to answer the overall (and possibly more important) question on how water flows and the recycling of nutrients could be managed in society.

**CONCLUSIONS**

The study aimed to compare different sludge management options. Lack of specific data and variable data quality made it challenging to fulfil this aim. The total environmental impact turned out to be surprisingly similar for the studied technologies. The different types of technologies included in the assessment all allow large direct emissions to air from sludge, be it from storage and agriculture or from incineration, or from drying and forestry. In all cases, efforts can be made to minimise emissions, but no technologies are inherently free of such emissions. In some cases,
certain emissions to air were slightly reduced by harsher treatment of the sludge, but the lower environmental impacts from direct emissions to air were then offset by lower gains from resource recovery. The trade-off between maintaining the sludge matrix to make the sludge useful as a fertiliser and soil conditioner on the one hand and removing the risk of emissions to air during storage and after spreading on the other is clearly important to consider in future sludge strategies.

Resource recovery turned out to be very important, in particular in terms of the considered use of biogas. For nutrients in sludge, it was in particular the nitrogen that provided benefits when mineral fertilisers are considered to be replaced. A new approach that was developed and applied to account for other benefits resulting from land application of sludge, primarily the carbon content, revealed large potential gains that have hitherto not been considered in LCA.

To make it possible to evaluate the environmental impact of different sludge management options in a relevant way, and in particular for making comparisons of different strategies and upgrading options, efforts should be made to increase the knowledge of emissions to air from different sludge storage options. It is recommended that any future sludge strategy, or future LCA on sludge management, considers potential emissions from all activities in which constituents of sludge can give rise to emissions to air. Further, as resource recovery will provide important benefits, these benefits need to be quantified in a relevant way.

ACKNOWLEDGEMENTS

The Swedish Water & Wastewater Association, Gryaab AB, The Käppala Association, Stockholm Vatten AB, Sydvästra Stockholmsregionens va-verksatiebolag (Syvab), Uppsala Vatten och Avfall AB and VASYD are greatly acknowledged for financial support. The study has also received funding from the Swedish Research Council for Environment, Agricultural Sciences and Spatial Planning (FORMAS) under grant agreement No. 2012-1122, and Göteborg Stad Kretslopp och Vatten.

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First received 1 July 2016; accepted in revised form 17 November 2016. Available online 7 February 2017