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Analyzing the process of struvite recovery with Life Cycle Assessment

– A case study

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ABSTRACT

In this study a comparative life cycle assessment was conducted, where a wastewater treatment method with a newly integrated struvite precipitation process was analyzed and compared with the conventional enhanced biological nutrient removal (EBNR) - based wastewater and sludge treatment. A case study was carried out based on data from a wastewater treatment plant in south Sweden, which was operating with EBNR, and started to use integrated struvite precipitation on pilot-scale in 2013. The aim of the study was to analyze and compare the environmental impacts of the two scenarios (Reference and Struvite scenario), and to identify which one was more favorable with regard to phosphorus recovery potential. The model included the avoided fertilizer production by the products (sewage sludge and struvite) but it did not include the environmental impacts from fertilizer application. Data regarding the two scenarios was acquired from the treatment plant and the operator of the struvite removal while the impact values were gathered from life cycle assessment databases.

The assessment showed that both scenarios had net negative impacts for all of the impact categories investigated due to conventional chemical fertilizers being replaced. The differences among the avoided impacts of the two scenarios were not considerable, indicating that the struvite precipitation in the plant, during the time period investigated, did not have much effect on the treatment. The Struvite scenario proved to be less favorable in the assessment, due to its increased energy consumption and the fact that the favorable effects of the struvite production did not suffice to balance the extra use of resources of the scenario. The avoided fertilizer production was found to be the most important for the result of the assessment. The chemicals used for the struvite precipitation exclusively showed little effect on the impacts, as the model was not sensitive to these factors.

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INTRODUCTION

Phosphorus (P) has been widely considered as waste, a potential polluter in streams and aquatic environment. Traditionally, wastewater treatment methods were focusing on the elimination of P from the inlet water in order to prevent eutrophication and other pollutions in aquatic ecosystems (Ashley et al., 2011). As a key component for plant growth, this macronutrient has crucial importance regarding to the sustainability of ecosystems and agriculture (Jönsson, 2004). Since phosphorus is a non-renewable (in human perspective, looking at its global cycle) and a non-replaceable nutrient which is used in an open-loop system, its recovery from waste streams is essential to cope with the need from agriculture (Dawson and Hilton, 2011). Treatment methods of wastewater should aim not only to eliminate phosphorus from the water but also to recover it, and make it available for further use.

Nutrient recovery has been getting significant attention across Europe. The EU and states like Switzerland and Norway put emphasis on phosphorus recovery specially, and encourage research and implementation of such technologies since 2007. Projects within the Seventh Framework Program and Horizon 2020 are aiming to increase the European phosphorus recycling rate from wastewater (CORDIS, www; P-REX, www). Sweden has a significant potential for P recovery, in which 90% of the wastewater treatment plants (WWTPs) operate with biological treatment and capture the majority of phosphorus, as 95% of P entering the plants is concentrated in the sewage sludge (Linderholm et al., 2012).

Returning phosphorus from waste streams to agricultural fields would reduce the need for chemical fertilizers and consequently the associated environmental impacts. Moreover it would reduce the overall dependence on the finite phosphate rock and increase phosphorus security at community level (Cordell et al., 2011). Solutions for an increasingly closed-loop system for P have been proposed and used in the past decades (Ashley et al., 2011). In WWTPs five basic ways are possible for P recovery: (1) after water treatment, the sludge can be utilized directly for agricultural purposes, (2) dissolve P from the incinerated sewage sludge, (3) reclaim P from the original sludge or (4) from the sludge liquor, and (5) through chemical precipitation (Cordell et al., 2011; Woods et al., 1999). In case 3, phosphorus can be precipitated from the sludge liquor in controlled environment through the addition of metals. Doing so, magnesium can contribute to the formation of struvite, a phosphate mineral, which can be used as a high quality agricultural fertilizer.

Struvite precipitation in WWTPs has been known for decades, and has significant P recovery potential (Maaß et al., 2014; Muster et al., 2013). Nevertheless, it is also associated with environmental burdens of energy and other resource needs. To realize, assess, and quantify these environmental impacts, life cycle assessment (LCA) as an environmental assessment tool was used in this study. LCA has already been proven to be a useful tool to investigate different wastewater treatment methods (Corominas et al., 2013). Using LCA, the aim was to assess the potential environmental impacts of struvite precipitation from wastewater sludge and to compare them with the impacts from conventional biological phosphorus removal. Both methods eliminate P from wastewater and offer possibilities for P recovery. However the usability of the sludge and the struvite can be considerably different and their productions are associated with different environmental loads.

This study was carried out on a wastewater treatment plant, Öresundsverket located in Helsingborg, in the south of Sweden. Struvite recovery, as a sludge treatment process was introduced in the plant in 2013, in pilot stage. This greatly affects the availability of data (more about the limitations later). Prior to this introduction, the plant was operating with conventional biological nutrient removal (EBNR) (detailed in ‘Materials and methods’).

Goal

The goal of the present study was to analyze and to compare the environmental impacts of two wastewater treatment processes, the EBNR integrated with struvite recovery and the conventional EBNR, at a particular treatment plant in Sweden. It aimed to identify which process is more favorable with respect of P recovery potential, and which one of them that was associated with more serious environmental impacts. This comparative life cycle assessment comparing the treatment process integrated with struvite precipitation, and the conventional biological nutrient removal in a case study was aimed to provide information to help realizing possible improvement potential of the struvite recovery process.

LITERATURE STUDY

Life cycle assessment (LCA)

All products and services produced can mean environmental burdens. Therefore LCA is a highly useful tool, since this method is able to provide a holistic perspective on these environmental burdens. It is a comprehensive method for analyzing systems regarding their environmental impact. Investigating the life cycle of a product (or process) means that its production, utilization and waste management (from cradle to grave) are all accounted for. The scope of LCA is generally on technical systems, and their connection to natural systems that is aimed to be described and quantified. The method allows a comparison of products or processes regarding their environmental impacts, and, since it can handle several environmental issues, also trade-offs between different environmental impacts can be identified (Jensen et al., 1997; Baumann and Tillman, 2004).

The methodology for LCA is described in international standards (ISO 14040 and ISO 14044). The purpose of the study, to whom and in which form the results will be communicated, are key questions and should be thoroughly answered at the start of the assessment. Further on, a model of the system being studied is built according to the scope and the aim that were set previously. The result is a flow model where all the environmentally relevant flows should be considered. These flows are quantified, in which all the data required should be collected and analyzed for the inputs and outputs of the whole system. The results of this inventory are then converted into impact categories that describe the potential effects on the environment for a number of categories. It serves as a stepwise aggregation of the information into fewer parameters. These impact categories reflect the information in a more effective way and serve as a basis for comparison (Baumann and Tillman, 2004).

LCA is traditionally a decision support tool, but nowadays it can be used for numerous purposes: for product development, within an industry, or identification of improvement possibilities, or also as a strategic planning tool (also within an industry); in market communication – eco labeling; to support governmental policy making in areas like energy production and waste management (Jensen et al., 1997). As every analyzing and assessment tool, LCA also has limitations. Uncertainties, as such, greatly affect the outcome of the assessment. Uncertainties can arise from data used for the analysis, as well as from the model that was used to describe the studied system. Uncertainties also arise from the methodology, like the choice of time perspective, assumptions made, or from the allocation of environmental burdens to different lifecycles. Other than the mentioned ones, it has to be stated that, though all the relevant environmental impacts ought to be considered in an LCA, this is not always fulfilled. The perspective of the LCA practitioner and commissioner can also have significant effect on the assessment (Ekvall et al., 2007).

LCA on wastewater treatment

LCA has been used intensely for evaluating wastewater and sludge treatment processes since the end of the 20th century (Emmerson et al., 1995; Roeleveld et al., 1997). As the analysis improved within this field, LCA was used to compare different wastewater treatment configurations. These studies showed a trade-off between eutrophication, toxicity and global warming potential caused by emissions to water, sludge treatment and disposal and the use of energy and chemicals respectively. Technological improvement gave rise to new treatment processes regarding wastewater and sludge and LCA was applied to compare them with conventional methods. (Corominas et al., 2013)

Several LCA studies were carried out regarding the topic of wastewater sludge applications and nutrient recovery. Lundin et al. (2000) compared conventional wastewater systems with source separation systems, where urine and black water were treated separately. Not only lower emissions and more efficient recycling of nutrients were experienced for the separation systems, but the study also highlights the importance of the setting of boundaries of the LCA on wastewater and sludge treatment methods. It states that next to the wastewater treatment the surrounding systems, like power generation, fertilizer production, and agriculture should be considered also (depending on the purpose of the study). The expansion of models in such way, to include the effects of these mentioned infrastructure systems, is needed to show the full impact on society of e.g. nutrient recycling. Corominas et al. (2013) points out the same, indicating that advantages of treatment processes resulted from the recovery of P and other nutrients are correctly evaluated when the model used in the analysis was expanded to include the avoided production of commercial fertilizers.

Different recycling and disposal options for sewage sludge were analyzed using LCA by Lundin et al. (2004). Besides the agricultural use of the sludge, incineration with P recovery from the ash, and two other methods with P recovery were investigated. The study found that the direct agricultural use of the sludge was the least preferable from the environmental point of view. Though this process has benefits, in terms of recycling P and nitrogen to the soil, the form of P in the sludge as ferric phosphate is considered less available for plants than the mineral form in fertilizers. Further, the direct application of sludge introduces a larger content of heavy metals to the soil than the other options. It had the largest eutrophication and acidification potential, and energy was required for the transportation and spreading, whereas energy could be recovered with the other options.

Svanström et al. (2008) compared handling options for sewage sludge, using LCA. Not only the impacts of the core systems were calculated but also the avoided use of fertilizers, the avoided environmental emissions from those fertilizers and from the sludge disposed were accounted for, using system expansion. Nevertheless, a large uncertainty can be seen regarding the estimation of these environmental emissions. Obvious from the study was that these system expansions had significant effect on the outcome of the study, in which numerous environmental benefits were realized.

Linderholm et al. (2012) compared three P recovery options (from wastewater, through sewage sludge recycling, struvite precipitation in the WWTP and recovery of P from the ash from incinerated sludge) with the utilization of virgin P. The boundaries of the system were expanded to include not only the recovery processes, but also the transportation, spreading and usage of the reclaimed P. The effect of the different forms of P in the reclaimed fertilizer products and the investments at the WWTP were not included in the study. Looking at the results the reuse of sludge was the most and the recovered P from ash was the least preferable in terms of energy use and emission of greenhouse gases. In case of cadmium (Cd) content - referring to the heavy metal content of the P-sources, sludge contained the most, the chemical fertilizer and the struvite nearly the same and less than the sludge, and the recovered P from the ash was the best option with insignificant concentration of Cd.

Goals for wastewater treatment systems should go beyond protection of human health and surface water quality, and target ecological sustainability, in which minimizing loss of resources, reducing the use of energy and water, reducing waste generation, and facilitate nutrient recycling. This change of paradigm can be properly addressed by using LCA, both at research stages of new technologies, and on pilot and full scale level of practice (Corominas et al., 2013). There is a need of more environmental analysis or LCA focusing on case studies dealing with struvite recovery in order to further improve the method, and to foster nutrient recovery. This case study aimed to assess the potential benefits of struvite recovery at a particular WWTP, and to acquire information regarding the impacts of sruvite recovery specifically to this plant. A further aim was to demonstrate the potentials of the process and to raise question regarding its applicability and improvement, in general.

Struvite recovery

Struvite is magnesium ammonium phosphate, with the formula of $MgNH_4PO_4 \cdot 6H_2O$ and its concentration by weight of the different elements are P 12,6%, N 5,7%, Mg 9,9% (Thelin, pers. and Linderholm et al., 2012). Next to P, it contains also nitrogen in a valuable concentration, which makes struvite a potential alternative for replacing N-fertilizer. Struvite can precipitate spontaneously in the sludge handling part of WWTPs, i.e. inside the pipes and pumps. This is an uncontrolled process, which can cause severe problems, in the pipes, decreasing their effective diameter and efficiency, resulting in high maintenance costs. On the other hand, the precipitation can be implemented in a controlled way, under certain physicochemical conditions, resulting in a fine mineral form of struvite, separated from the sludge liquor. P recovery as struvite has been reported in numerous studies in the literature (Parsons and Doyle, 2004; Münch and Barr, 2001; Stratful et al., 1999).

In WWTPs, the controlled crystallization of P as struvite is only efficient in plants operating with biological P removal, as the dissolved P concentration of the water used has to be high enough for the process (Linderholm et al., 2012). The proper conditions for struvite precipitation include range of pH 8-10, room temperature, and residence time in the reactor 10-40 minutes. Stoichiometric requirements include Mg:P molar ratio between 1-1,3 and Mg:Ca higher than 0,6 (in order to precipitate struvite instead of calcium phosphate) (Muster et al., 2013). Further, high concentration of phosphate and ammonium (supersaturation ratio greater than 1, optimal range is 2-6) in the influent at the reactor are also essential to facilitate efficient precipitation (Bhuiyan et al., 2008 and Seco et al., 2008). The chemical reaction for struvite formation is expressed in Equation (1), with n=0,1 and 2 being a function of pH (Bhuiyan et al., 2008).

$$H_nPO_4^{n-3} + NH_4^+ + Mg^{2+} + 6H_2O \rightarrow MgNH_4PO_4 \cdot 6H_2O + nH^+ \qquad (1)$$

Seco et al. (2008) and Bouzas et al. (2010) found that the configuration of the sludge treatment line, and the characteristics of the sludge liquors employed have great importance as the crystallization process is much affected by the influent stream. Struvite was found to be the most desirable precipitate for the recovery of P from wastewaters (Muster et al., 2013), and ideal for further utilization as fertilizer because of its relatively high P content next to the N, and its water soluble character, releasing P and the other mineral components slowly, and in plant available form (Linderholm et al., 2012; Shu et al., 2006). It has low content of pollutants, i.e. heavy metals, and is in a concentrated, crystalline form that is easy to handle. Several companies pack it and sell on a market. Recovering struvite from wastewater is not a novel process, but its commercial use has only been introduced recently. Large scale struvite production plants recovering struvite from sludge liquor are operating in numerous European countries, like UK, Germany, Belgium and the Netherlands, and also in the US, Canada and Japan (Maasß et al., 2014; P-REX, www).

MATERIALS AND METHODS

The two systems in comparison

Using life cycle assessment, two wastewater treatment methods were analyzed. The conventional biological treatment (‘Reference scenario’, shown in Fig. 1) is a known and prevalent process to treat wastewater in Sweden, usually combined with precipitation via metals (i.e. Fe, Al). The treatment results in water with low polluting capacity – below low thresholds, that can be discharged to water bodies, and also in sludge containing significant amounts of potential nutrients, and pollutants. The other system where struvite precipitation is integrated (Struvite scenario, shown in Fig. 2) is based on a biological treatment including biological P removal and struvite precipitation from the separated sludge liquor. The treatment plant analyzed in this study has used biological treatment as a core process and started to operate with the integrated struvite precipitation method on pilot scale, in 2013.

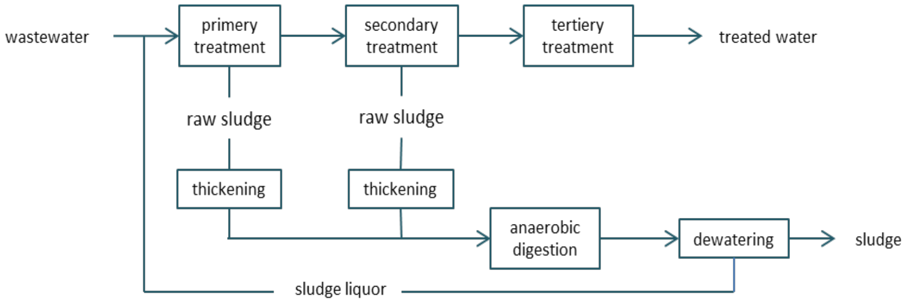

Fig. 1 Flow chart of the Reference scenario.

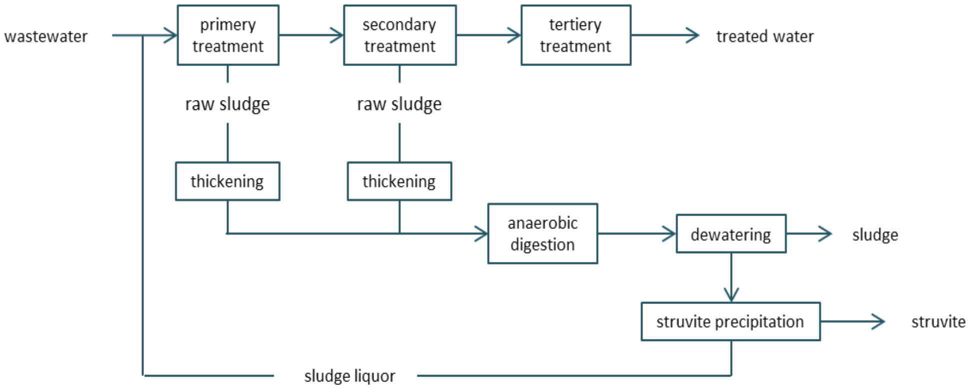

Fig. 2 Flow chart of the Struvite scenario.

Wastewater treatment – biological nutrient removal

In this section the conventional, enhanced biological nutrient removal (EBNR, Reference scenario) that was used in Öresundsverket before shifting to the complementary struvite recovery is described briefly. The core biological treatment process means the removal of organic matter, N and P using activated sludge (microorganisms), including polyphosphate-accumulating organisms (PAO), instead of chemical precipitation. The water treatment consisted of primary, secondary and tertiary treatment. In the primary treatment the water went through screens and grit separation. Heavy, readily settling and floating particles were removed in the aerated grit chamber. After that ferric chloride was added to the wastewater in small quantity for a chemical pre-precipitation that strengthens the primary sedimentation. The secondary, biological treatment followed it, where the water was treated biologically through activated sludge process under different oxygenic conditions in order to remove not only organic matter but also N and P. It was required since different microbial communities grow under different oxygen conditions. Anaerobic zones were required for microbes that remove P, while N removal happened under anoxic conditions. In the aerobic zone organic matter was degraded by microbial activity and the ammonium was oxidized to nitrate. At the end the suspended solids were separated from the effluent water in the secondary sedimentation basin. The majority of the activated sludge was recirculated for further use while the excess sludge was separated for treatment. The tertiary treatment consisted of a final particle removal in a two-media filter. The particles removed were led back to the inlet when the filter was back-flushed.

The primary sludge from the pre-sedimentation and the excess sludge from the biological process were thickened separately. Ferric chloride was dosed to the primary sludge in order to prevent hydrogen sulfide production in the following digestion process, and a polymer, polyacrylamide was dosed to the secondary sludge for better drainage. After the thickening, with decreased water content, the mixed sludge went through anaerobic digestion. This process stabilized the sludge as its organic matter content was partially degraded by anaerobic microbes. The decay of the organic matter resulted in a sludge enriched in inorganic ions, and biogas produced by certain microorganisms. Thereafter, the sludge was dewatered in centrifuges where polymer was added to enhance the process. The separated excess sludge liquor had pH=7-8, increased phosphate, ammonium and other inorganic ion concentration which could be potential nutrients, and also other organic and inorganic molecules which could act as polluters. These separated liquors from the centrifuges and thickeners were then transferred back to the WWTP intake for further treatment. The dried sludge was stored and tested before further utilization, such as on agricultural land or on reclamation-, or construction areas (Grady et al., 1999; Kárpáti and Vermes, 2011; NSVA, www).

Struvite precipitation – nutrient recovery

The alternative treatment technique (Struvite scenario) operated in a similar way to the conventional bio-P method regarding the water treatment chain. At the sludge treatment, thickening and stabilization of the sludge by anaerobic digestion was done, similarly to the Reference scenario. Before digestion most of the P was stored as polyphosphate in PAOs. They stored polyphosphate as an energy reserve, under aerobic conditions. During digestion, under anaerobic conditions PAOs released phosphate, and used the energy to accumulate simple organics (Strom, www). The polyphosphates hydrolyzed and the degradation of organic matter produced an additional release of dissolved P among other elements. The result was a high concentration of phosphate ions in the sludge - and, consequently, in the sludge liquor that facilitated P recovery by struvite precipitation (Seco et al., 2008). The sludge liquor coming from the thickeners and centrifuges after the digestion were in this scenario not returned directly to the plant intake, but instead led to a crystallization reactor. The pH of the liquor was high in the plant (7,8-8,1) and aeration to strip carbon dioxide from the sludge liquor took place before the reaction in order to increase pH even further (Fig. 3). In the reactor magnesium chloride was added to the liquor and also pH could be adjusted by NaOH in order to reach optimized conditions for precipitation. However, there was no need for NaOH in this case as the pH was already high enough for efficient precipitation. In the reactor, struvite precipitated and was separated in the settling zone right after. The effluent flowed out at the top of the settling zone. More than 90% of P of the sludge liquor was recovered as struvite with this process and ca. 10% of the N. This meant a ca. 20-25% recovery of P in form of struvite, from the incoming wastewater to the plants, similarly to the values given by Linderholm et al. (2012). The effluent water was then returned to the plant intake for further treatment, just as in the Reference scenario (Ekobalans, pers. com. and Thelin, pers. com.). The process of the precipitation of the struvite is shown in Fig. 3.

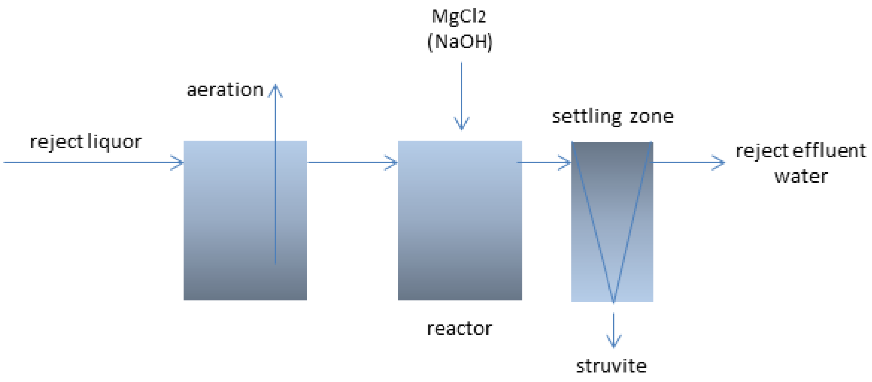

Fig. 3 Process of the struvite crystallization.

Öresundsverket, Helsingborg

The wastewater treatment plant Öresundsverket is located 1,5 km from the center of Helsingborg. The plant treats wastewater from approx. 119 000 people and produces 2500-

3000 tons of sludge in dry matter annually (NSVA, www). Table 1 shows the main characteristics of the inlet wastewater to the plant.

Table 1 Main characteristics of the inlet wastewater, and the average P and N concentrations of the treated wastewater (from NSVA).

Parameter	2012	2013	Unit
Amount treated	150730	143691	pe (70g BOD7/pe*d)
Flow	55575	65491	m3/d
Flow	2316	2729	m3/h
BOD7	190	154	mg/l
	3851	3671	ton/y
N-tot	36.3	29.6	mg/l
	736.6	707	ton/y
P-tot	4.5	3.4	mg/l
	90.9	81,4	ton/y
N (average concentration)	8.4	9.6	mg/l
P (average concentration)	0.26	0.36	mg/l

The LCA model

The inventory analysis included parameters describing energy and chemicals used, material flows and emissions. The flows of the systems were normalized to the functional unit (FU). Calculations regarding the flow models of the systems were carried out in MS Excel. The data for the impact assessment was acquired using Ecoinvent Database. Here the ‘CML 2001’ impact assessment model was chosen as a model for calculating the environmental impacts of each material, chemical and process used and included in the system investigated. For impact categories the following ones were selected to represent the most significant effects on the environment and human populations, supported by literature (Svanström et al., 2008; Corominas et al., 2013): (1) acidification potential, (2) climate change potential, (3) eutrophication potential and (4) human toxicity. Category 1, 2 and 3 have been applied the most in LCAs within the field of wastewater treatment according to Corominas et al. (2013). Regarding toxicity, ‘human toxicity’ and ‘terrestrial ecotoxicity’ have been used frequently in studies. ‘Human toxicity’ was chosen to be included in this assessment as ‘terrestrial ecotoxicity’ was found weakly supported by data and information from the plant. No data was found for the chemical, magnesium chloride. In this case it was decided to use data on calcium chloride, which production is quite similar to $MgCl_2$ (Doka, pers. com.).

System boundaries

The flow charts of the two studied treatment methods and their boundaries are shown below, the conventional wastewater treatment in Fig. 4 (the Reference scenario) and the one with struvite precipitation in Fig. 5 (the Struvite scenario). The models start with the wastewater entering the plant and include all the processes within them and the end materials (sludge, struvite) except the treated wastewater. Moreover, the ‘Avoided fertilizer production’ marks the amount of commercial fertilizer products and their environmental impacts of production that is avoided by the recovered P and N containing materials. Here not only P fertilizer was considered, but N fertilizer also, as the recovered sludge and struvite contain N as well, serving as an alternative for certain N fertilization. Both the wastewater treatment and the sludge treatment processes were analysed in this study because it was believed that the struvite precipitation affects not only the sludge treatment part (which it is basically connected to), but also the wastewater treatment. In both figures the blue frame indicates the wastewater treatment plant, including all activities and materials used or produced in the plant. The production of chemicals and energy used during the treatment processes were considered and accounted for in the model. The production of the potential fertilizer products (sludge, struvite and chemical fertilizers) was also included in the model, but their application on agricultural land was not. Transportation of the sludge and the stuvite and the plant availability of all the fertilizer products were accounted for, since it can differ considerably (Linderholm et al., 2012).

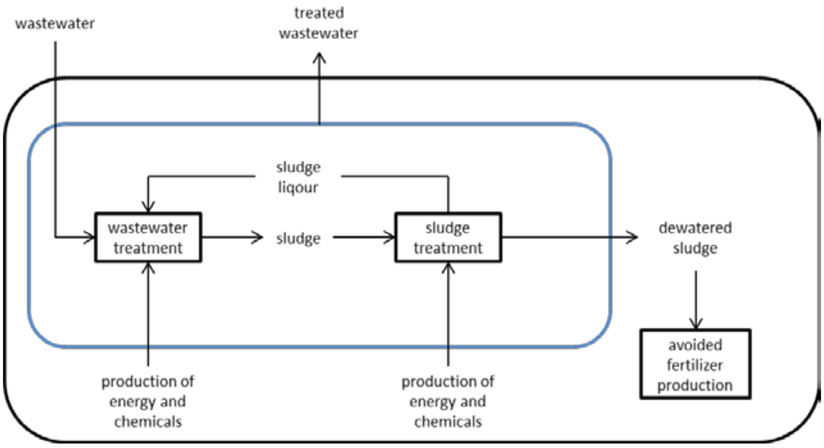

Fig. 4 Flow chart of the model for the Reference scenario.

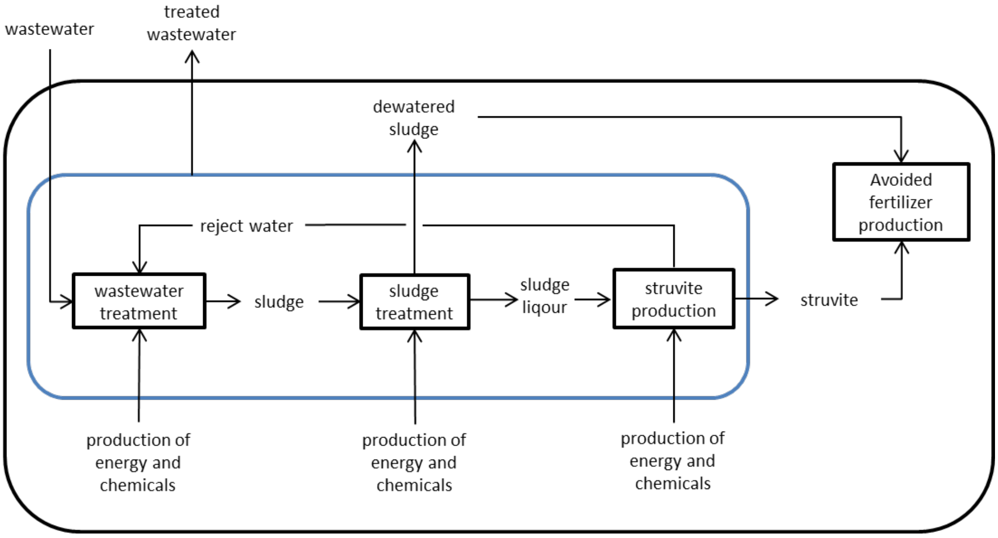

Fig. 5 Flow chart of the model for the Struvite scenario.

Functional unit

The functional unit of 1 kg recycled plant available P that was chosen in this study. Since P recovery is in the focus of this study and the amount of reclaimed P is considered to be one of the main tasks in this study, choosing a certain amount of recycled P serves as a suitable basis for comparison. The 2 scenarios were compared with respect of recycling of available P and this study wishes to enhance further development on recovery of available P from wastewater.

The plant availability of P of the two products, the sludge and the struvite are different. In case of the sludge, 60% of P was considered plant available, while 100% was assumed in case of the struvite (detailed below in ‘Data’).

Looking at the products per functional unit, 1 kg recycled plant available P, of the two scenarios, they were 56.06 kg of sludge for the Reference scenario, and 52 kg of sludge and 0.84 kg of struvite for the Struvite scenario (calculations are in Appendix).

Data

The data on wastewater and sludge treatment used for the analysis was provided by Nordvästra Skånes Vatten och Avlopp (NSVA), the operator of the WWTP Öresundsverket. For the Reference scenario data from 2012 was used, annual amounts. For the Struvite scenario data from 2013 was used, when the plant started to operate with the struvite recovery method on pilot scale. Data about the struvite production specifically was provided by Ekobalans, a Swedish innovative company focusing on nutrient recycling, and operating the project for struvite production. Table 2 and 3 present the input data for the Reference and the Struvite scenario respectively, and the calculations for the reference flow. Table 4 presents the data on the composition of the sludge and the struvite used in the study, while the index values of the selected impact categories used in the impact assessment are presented in Table 5. These indexes are the preset impact values for one unit of each product or activity, gathered from Ecoinvent database.

Table 2 Inventory data (annual) on wastewater and sludge and normalized amounts; Reference scenario (from NSVA and Ekobalans).

Wastewater treatment	Raw data		Normalized per activity; kg/kg sludge	Reference flow-normalized per FU; kg/kg FU
inflows				
energy	6518000	kWh	0.24	134.7
wastewater	17520000	m^3	650.50	362100
ferric chloride	125	m^3	0.0052	2.89
sludge liquor	24200	ton	0.90	500
outflows				
sludge	26900	ton	1	556.68
Sludge treatment			kg/kg dried sludge	kg/kg FU
inflows				
sludge	26900	ton	9.93	556.68
ferric chloride	125	m^3	0.050	2.80
polymer	32.2	ton	0.012	0.67
outflows				
dewatered sludge	2710	ton	1	**56.06** this contain: 1 kg plant av. P 1.7 kg plant av. N
sludge liquor	24200	ton		

Table 3 Inventory data (annual) on wastewater, sludge and struvite precipitation, and the normalized amounts; Struvite scenario (from NSVA and Ekobalans).

Wastewater treatment	Raw data		Normalized per activity kg/kg sludge	Normalized per FU; kg/kg FU
inflows				
energy	6789000	kWh	0.27 kWh	143 kWh
wastewater	23905000	m^3	970	501600
ferric chloride	125	m^3	0.0057	2.94
sludge liquor				
outflows				
sludge	24600	ton	1	516
Sludge treatment			kg/kg dried sludge	kg/kg FU
inflows				
sludge	24600	ton	9.93	516
ferric chloride	125	m^3	0.056	2.91
polymer	32.2	ton	0.013	0.68
outflows				
dewatered sludge	2480	ton	1	**52** this contain: 0.9 kg plant av. P 1.7 kg plant av. N

sludge liquor	22100	ton		
Struvite precipitation			kg/kg struvite	kg/kg FU
inflows				
energy	22100	kWh	0.55 kWh	0.46 kWh
sludge liquor	21100	ton	556	467
magnesium chloride	5.3	ton	1.33	1.12
citric acid	0.55	ton	0.014	0.012
outflows				
struvite	40	ton	1	**0.84** this contain: 0.1 kg plant av. P 0.05 kg plant av. N

Table 4 Data about the composition of the sludge and struvite (from NSVA and Ekobalans).

Parameter		Sludge 2012	Sludge 2013	Struvite
quantity (wet)	ton	10900	10500	40
quantity (TS)	ton	2710	2480	40
pH		8.1	8.2	
TS	%	24.9	23.7	100
NH4-N	mg/kg TS	12500	13600	2.2
N-tot	mg/kg TS	55000	58100	2.2
P-tot	mg/kg TS	29700	28700	5.04
Hg	mg/kg TS	0.68	0.6	< 0.4
Cd	mg/kg TS	0.84	0.79	< 0.2
Pb	mg/kg TS	21	20	< 2
Cu	mg/kg TS	442	411	8.4
Zn	mg/kg TS	614	600	12.2
PAH	mg/kg TS	1.2	1.1	
PCB	mg/kg TS	0.073	0.038	
Na	mg/kg TS	1180	1150	
S	mg/kg TS	15000	14800	

Table 5 Index values of the selected impact categories for the activities included (from Ecoinvent Database).

Activity	acidification pot. *European average* (kg SO2-Eq)	climate change pot. *GWP 100a* (kg CO2-Eq)	eutrophication pot. *European average* (kg Nox-Eq)	human toxicity *HTTP 100a* (kg 1,4 DCB-Eq)
electricity production *(production mix, Sweden; 1 kWh)*	0.00014	0.041	0.00014	0.017
ferric chloride production *(1 kg)*	0.00067	0.625	0.0017	1.072
polyacrylamide production *(1 kg)*	0.017	2.85	0.0073	0.9
citric acid production *(1 kg)*	0.017	3.2	0.013	1.2

calcium chloride production *(1 kg)*	0.0020	0.39	0.0019	0.41
avoided P fertilizer production *(TSP, as P2O5; 1 kg P2O5 or 2,08 kg TSP)*	0.022	1.47	0.0068	1.49
avoided N fertilizer production *(AN; 1 kg N or 2,86 kg AN)*	0.038	9.47	0.047	3.45
transport *(/metric ton*km; freight, lorry, 16-32 ton)*	0.00088	0.16	0.0014	0.059

The amount of ferric chloride and polymer used during the treatment process was associated with uncertainties, since there was a possibility that these numbers were not so accurate. These might have been dosed based on time and flow of wastewater rather than based on what was actually needed at each time. Furthermore, in certain years, according to the reports provided from NSVA, the amounts of the chemicals were equal. That can occur because of reporting the ordered amount instead of the exact, used amount of the chemicals.

The sludge liquor coming from the sludge dewatering is recirculated back for wastewater treatment in the Reference scenario, and thus it is shown in data as an *inflow* for the wastewater treatment. In the Struvite scenario this was also done in the same way after the liquor went through the reactor (struvite precipitation), but information on its exact manner was not provided. NaOH was not mentioned in the data, as it was in this case not needed for the process. Citric acid was instead on the list as it was used for cleaning certain parts of the equipment.

The avoided amount of commercial fertilizers applied on land was accounted for in the model. This means the amount of P in the form of commercial fertilizer produced from phosphorus rock that was substituted by the P product from the WWTP in form of struvite or sewage sludge. Triple superphosphate (TSP), a widely used fertilizer was assumed to be the fertilizer substituted, having 48% P_2O_5 content (21% P) in the form of monocalcium phosphate that is >90% water soluble (IPNI, 2016). Ammonium nitrate (AN) was decided to be the alternative N fertilizer that was substituted with the N from the sludge and the struvite. It has a 35% N content that is fully water soluble (IPNI, 2016).

Plant availability of P had to be assumed for both the sludge and the struvite. Since the effectiveness of the two treatments was evaluated in the study, in terms of what amount of P is being reclaimed and presented to the soil, plant availability has to be considered. Phosphate is readily available for plants and microbes. All of the P is present in form of phosphate in struvite, and its availability had been previously assumed to be >90%, similar to monocalcium phosphate (technically TSP) (Ganrot, 2005). Moreover, Johnston and Richards (2004) found no significant difference between monocalcium phosphate and struvite, as sources of P for plant growth, suggesting that P in struvite can fully replace P in monocalcium phosphate on the bases of effective P supply. Based on these, it was assumed that 100% of the P in struvite could replace P in TSP. In case of the sewage sludge 60% of P was assumed to able to replace TSP, as this fraction has been determined with citric acid solubility (Herter and Külling, 2001).

On the plant availability of the N, in case of the struvite 100% was assumed similarly to the AN, that is fully available (Jönsson, pers. com.). The availability of N in the sludge was assessed after Delin et al. (2012), who used the C/N ratio to determine how much of the mineral N that the sludge could replace. According to that study, 55% and 56% was assumed for the fraction of N in the sludge that could replace chemical fertilizer N, for the year 2012 and 2013 respectively. Table 6 compares the composition and relevant properties of the three P containing materials, sludge, struvite and TSP and of the AN, discussed in this assessment. Plant availability and the degree to which sludge and struvite can substitute commercial chemical fertilizers depend on soil properties and spreading technique among others, so uncertainty in the data can be significant. Availability of P varies also in time in the soil, as the less available organic forms goes through different forms of mineralization. When looking at plant availability, it has to be fulfilled in short time span. More complex, organic forms are not considered as plant available even though in longer timescale most of them presumably become available (Dawson and Hilton, 2011).

Table 6 Relevant characteristics of struvite, sewage sludge and TSP and AN.

Parameter	Struvite(a)	Sludge(b) (2012)	Sludge(b) (2013)	TSP(c)	AN(d)
P content (P)	12.6% of TS	3% of TS	2.9% of TS	21%	0%
P availability (replaceability)	100%	60% of tot. P	60% of tot. P	100%	-
N	5.5%	5.5% of TS	5.8% of TS	0%	35%
N availability (assumed)	100%	55%	56%	-	100%
Cd	< 0.2 mg/kg TS	0.84 mg/kg TS	0.79 mg/kg TS		
Cu	8.4 mg/kg TS	442 mg/kg TS	411 mg/kg TS		
Zn	12.2 mg/kg TS	614 mg/kg TS	600 mg/kg TS		

a: from Ekobalans
b: from NSVA
c: from IPNI
d: from Ecoinvent Database

Transport of the product from the plant to the field (place of utilization) was included in the model. Based on information from the WWTP the dewatered sludge was first stored for a longer timespan (roughly six months) required for chemical and hygiene tests. After that, it was transported to the fields once it had been proved safe. In the model the impacts of the storage and handling of sludge was not included because of the lack of data. According to the NSVA, in the years investigated the sludge utilized was transferred to nine locations in total in Southern Sweden, around Helsingborg with a total distance of approx. 700 km (estimated with Google Earth). In case of two of the locations sludge was transported twice in the year. Due to the lack of information on the exact proportions of sludge transferred to each location, an average distance of transport from Helsingborg was calculated and it was assumed that the amount of sludge and struvite per FU was transferred this average distance, 76 km from Öresundsverket. No information was provided on the transport of the struvite and therefore it was assumed it was transported in the same way and the same distance as the sludge. It was considered that the struvite and/or the sludge were transported one-way, via lorry with the capacity of 16-32 tons.

RESULTS AND DISCUSSION

Results of the assessment

During the impact assessment calculation of environmental impacts for the selected categories were carried out. The impacts of the two scenarios were calculated and compared per FU (Table 7). The results of the assessment for the two scenarios are compared and shown in Fig. 6. In order to simplify the comparison of the scenarios, the results are scaled so that the ones for the Struvite scenario are given relative to the Reference scenario.

Table 7 Calculated impacts of the two scenarios per FU

	acidification potential (kg SO2-Eq)	climate change potential (kg CO2-Eq)	eutrophication potential (kg Nox-Eq)	human toxicity (kg 1,4 DCB-Eq)
Reference scenario	-0.273	-20.7	-0.115	-13.1
Struvite scenario	-0.271	-20.1	-0.113	-12.4

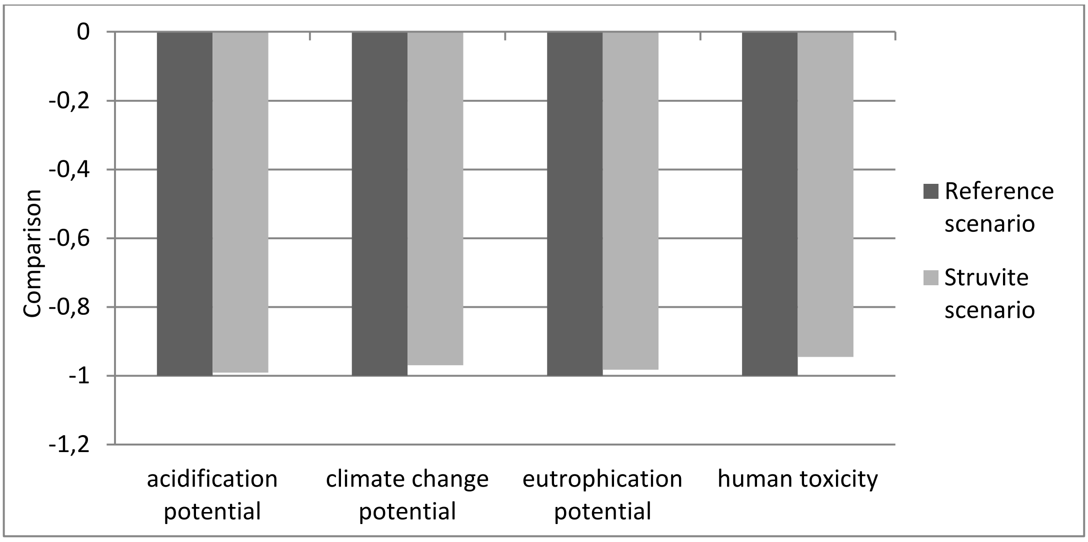

Fig 6 Results from the impact assessment scaled to the Reference scenario

The two scenarios have quite similar impacts regarding all the categories investigated. The impact values are negative, representing a reduction of impacts in comparison to using the replaced chemical fertilizers. These favorable impact values were a result of the reducing effect of the ‘avoided fertilizer production’. Regarding ‘acidification potential’ both scenarios helped to avoid the impacts almost equally, the difference is 1% as the Reference scenario helped to avoid a little bit more. In case of ‘climate change potential’, both scenarios decreased the climate change but the Struvite scenario 3% less than the Reference scenario. Both scenarios also decreased the ‘eutrophication potential’ and the Reference scenario decreased it 2% more than the Struvite scenario. Looking at ‘human toxicity’ the Struvite scenario means a smaller reduction of impacts again, by 5%. From Table 7 and Fig 6 it can be seen that the difference between the two scenarios regarding the impact categories investigated is consistent. The Struvite scenario proves to be less favorable than the Reference scenario according to the assessment, even though the differences are not considerable.

Fig. 7 and 8 show the contribution of different life cycle steps to the total impacts of the four categories, for the Reference and the Struvite scenario respectively. Looking at the figures, there were only small differences between the two scenarios. The avoided production of the fertilizers AN and TSP contributed the most for the categories investigated, having a reducing effect on the impacts in both scenarios. Replacing TSP had the largest effect on the scenarios, contributed to a great reduction of the impacts in all the four categories. Replacing AN seemed also important, especialy for 'climate change potential ' and 'eutrophication potential'. The use of electricity contributed most to the impacts 'acidification potential' and 'eutrophication potential' in both scenarios. Looking at the category 'human toxicity', ferric chloride contributed the most to the impacts, while in case of 'climate change potential' the shares of the impacts of energy use, polymer and ferric chloride consumption were not different particularly, and trasportation had the lowest contribution.

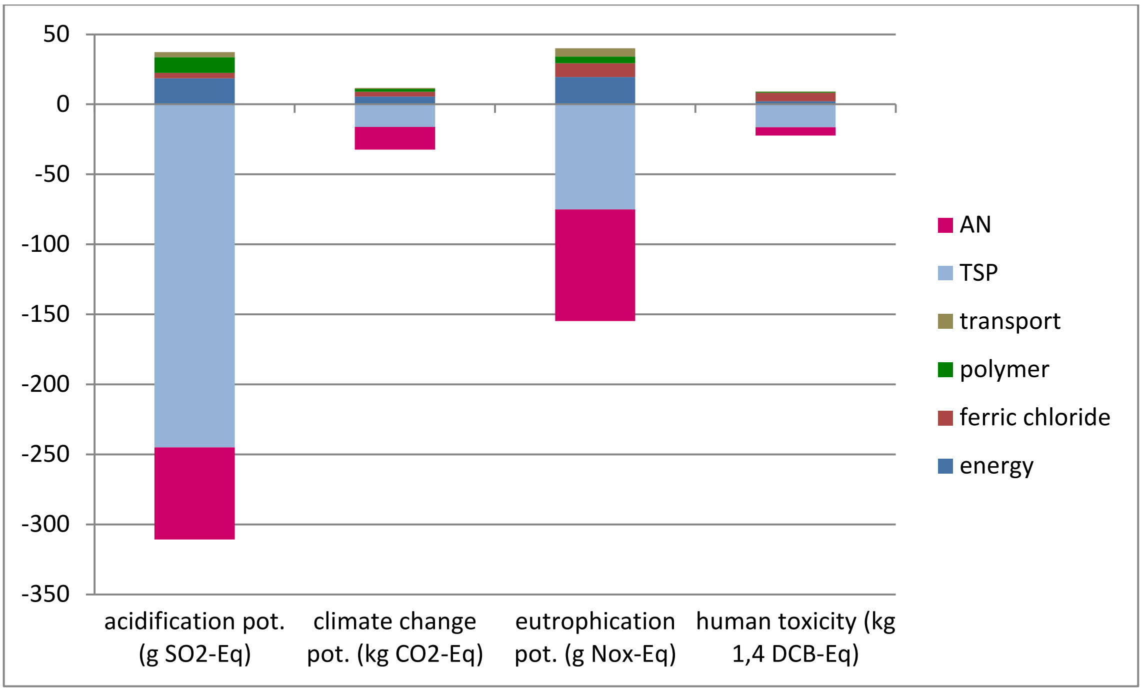

Fig. 7 Detailed contribution of activities to the impacts for the Reference scenario.

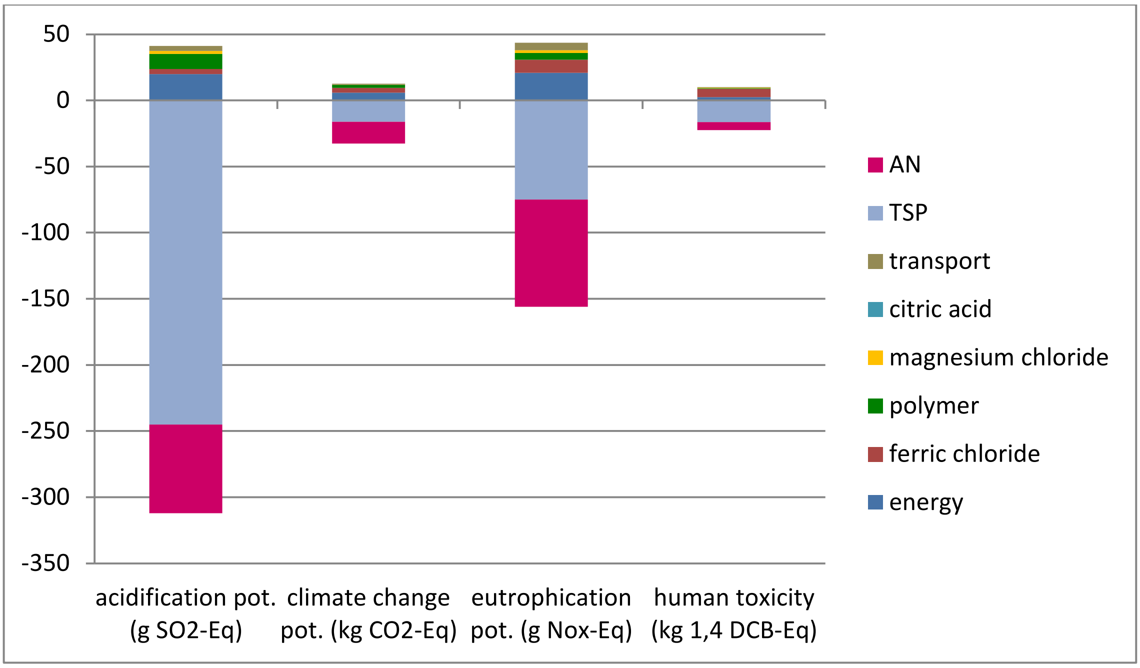

Fig. 8 Detailed contribution of activities to the impacts for the Struvite scenario. Data for magnesium chloride was lacking and instead data for calcium chloride was used.

As the effect of the avoided production of AN and TSP combined proved to be considerable for both scenarios, in Table 8 the impact values for the categories investigated are shown where the ’avoided fertilizer production’ was not included in the assessment. In this case only the impacts that were not compensated with the avoided production of the P and N fertilizers were accounted for. Svanström et al. (2008) also showed the results excluding the avoided fertilizer production separately, in order to give a clear understanding of its importance. All the impact values are positive (Table 8), and the difference between the two scenarios are larger in this case. Struvite scenario has larger impacts than the Reference scenario in all the four impact categories, by 8-10%. The difference is the largest for the ’acidification potential’ (10%), the smallest for the ’climate change potential’ (8%). Impacts for both the ’eutrophication potential’ and the ’human toxicity’ were increased by 9% in case of the Struvite scenario. From Table 7 and Table 8 it is visible that the avoided production of the P and N fertilizer decreased the impacts per FU considerably so that all impact categories had negative values. It also decreased the difference between the two scenarios. Excluding the avoided fertilizer production, the Struvite scenario had increased impacts by 8-10% (Table 8), and this difference went down to 1-5% when the avoided production of TSP and AN was accounted for (Table 7). The marked impact of the avoided fertilizer production was highlighted also in Svanström et al. (2008).

Table 8 Calculated impacts of the two scenarios per FU, ‘Avoided fertilizer production’ excluded

Scenario	acidification potential (kg SO2-Eq)	climate change potential (kg CO2-Eq)	eutrophication potential (kg Nox-Eq)	human toxicity (kg 1,4 DCB-Eq)
Reference scenario	0.037	11.70	0.040	9.22
Struvite scenario	0.041	12.66	0.044	10.06

Fig. 9 shows that the Struvite scenario used relatively more electricity per FU than the Reference scenario. The use of energy contributed notably to the impacts in the category ’climate change potential’ in both scenarios, but mainly in the Struvite scenario. The increase is a result of the extra process of struvite precipitation and its energy consumption.

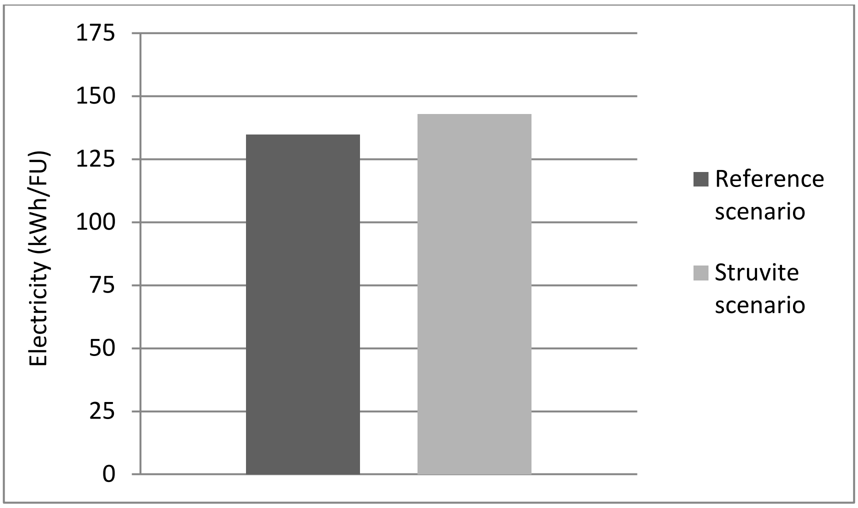

Fig. 9 Electricity use of the two scenarios per FU.

Sensitivity analysis

For analyzing the sensitivity of the model used in this study, the aim was to identify critical parameters where small change can lead to large differences in the final results. The sensitivity of parameters was analyzed as the difference in the final results of impacts resulted from a 10% change in the input variables (amount of chemicals, energy and distance of transport per FU) and is shown in Table 9. According to the results, a 10% increase in the amount of polyacrylamide, calcium chloride, citric acid used, and in the transport distance increased the impacts only a minor level in all categories. The change in the energy used showed similar results, in case of ‘climate change potential’ it reached 3% increase in the impact as the highest result. Ferric chloride made a 5% increase in the impact category ‘human toxicity’ in response to the 10% increase in its amount used in the model. In case of the avoided TSP and AN production, they reached more considerable results. The increase in the amount of avoided TSP was able to decrease the impacts by almost 10% in the categories ‘acidification potential’ and ‘climate change potential’, by 13% in case of ‘human toxicity’ and by more than 6% in ‘eutrophication potential’. The avoided AN managed to decrease the impacts of ‘climate change potential’ and ‘eutrophication potential’ by 8% and 7% respectively, when its amount was increased. In case of ‘human toxicity’ it reached 5% and 2.5% in the ‘acidification potential’. Clear from the results was that the model was not sensitive to the citric acid and the calcium chloride, the chemicals used for the struvite precipitation. Energy consumption did not seem very important either, having small effect on the final impacts. Ferric chloride seemed more important than the previous parameters but only in case of ‘human toxicity’. The model was the most sensitive to the avoided fertilizer production, namely that these parameters can affect the impacts most, having considerable decreasing potential.

Table 9 Results of the sensitivity analysis. The results for the parameters 'avoided TSP prod.' and 'avoided AN prod.' are in blue, marking a decrease.

	acidification pot.	*climate change pot.*	*eutrophication pot.*	*human toxicity*
enegy prod.	<1%	3%	1.8%	2%
ferric chloride prod.	<1%	1.8%	<1%	5%
polyacrylamide prod.	<1%	1%	<1%	<1%
transport	<1%	<1%	<1%	<1%
magnesium chloride prod.*	<1%	<1%	<1%	<1%
citric acid prod.	<1%	2%	1.5%	1.2%
avoided TSP prod.	9%	8%	6.7%	13%
avoided AN prod.	2.5%	8%	7%	5%

*Data for calcium chloride was used for magnesium chloride, for which no data was found.

Discussion of the impact assessment results

According to the results of the assessment, the Reference scenario proved to be more favorable regarding the impact categories investigated, and it produced only sludge as a potential P recovery product. The Struvite scenario produced slightly smaller amount of sludge and certain amount of struvite with more favorable P recovery properties such as its low content of heavy metals but high content of P. Further it has higher plant availability of P and N than the sewage sludge, and concentrated crystalline form. Despite of these benefits, the production of struvite at the WWTP Öresundsverket increased the environmental impacts of the system because of the extra chemicals and energy used for the process. On the other hand, it lowered the impacts of transport and increased the avoided impacts of the production of commercial fertilizers. However, the result was an overall decrease of the avoided impacts regarding all impact categories, but this decrease was rather small. The data for the struvite scenario was mainly gathered from a pilot-scale scale process operated on approx. 50% of the time a full scale plant could, and with approx. 5 m^3/h flow rate instead of the 15-20 m^3/h. This most likely affected the results.

By expanding the system investigated, the boundaries of the model included the replaced chemical fertilizer production. From this respect, the use of chemical fertilizers was compared with P and N recycling at the treatment plant, and the benefits were clear from the results. The effect of the avoided fertilizer production was considerable, as impacts were avoided rather than produced in all categories, in both scenarios. Without recycling P and N, and using conventional chemical fertilizers the model had increased environmental impacts in both scenarios. The positive effects of recycling P and N in the form of sewage sludge and struvite was seen in this study, when their potential use for agricultural purposes was considered.

Uncertainties and incompleteness of the model

The amounts of chemicals ferric chloride and polyacrylamide, used for the treatment of both the wastewater and the sludge were initially expected to decrease in the Struvite scenario compared to the Reference scenario. This expectation was based on that the reject water from struvite precipitation turned back to the wastewater inlet should need less treatment and less chemicals as the internal flow of P is decreased in this way in the WWTP. This was not proved in this study. The reason might be that inventory data regarding chemicals had too large uncertainties due to poor documentation and the fact that 2013 was the first year for the plant running with this method. The amount of the sludge produced was also expected to decrease in case of the precipitation of struvite. From 2012 to 2013 there was a 9% reduction in the volume sludge (TS) at the WWTP. In Woods et al. (1999), where the implementation of P recovery processes in municipal wastewater treatment applications were modeled, a 20-30% possible reduction in sludge volumes is reported in treatment plants with EBNR. The amount of sludge produced was mainly based on the volume and properties of the incoming wastewater to the plant. As the inner flow of P and N was decreased in the plant in case of the Struvite scenario, the sludge volume reduction was reasonable. A possible explanation to the smaller reduction in the sludge volume than reported by Woods et al. (1999) can be the pilot-scale operation, which decreased the reduction compared to full-scale operation.

Impacts of the storage and handling of sludge, as well as spreading and emission after spreading were not included in this study even though their contribution can be of great importance according to Svanström et al. (2008). That study emphasized that biogeochemical emissions, i.e. nitrous oxides, heavy metals, methane and ammonia emissions, after spreading of the fertilizers can considerably affect the impact categories climate change potential, acidification potential and eutrophication potential.

If other impact categories had been taken into account, the results could have looked differently. Including for example ‘resource depletion’ could certainly provide new information about the two scenarios. Both of the scenarios recycle P and N to a certain level, contributing to the deceleration of the depletion of the deposits of these elements. By analyzing resource depletion the extent of recycling in both scenarios could be addressed. The potentials of struvite to recycle P and N and thereby to the preservation of these compounds could be analyzed more thoroughly with such information.

Further it would be interesting if the assessment were carried out from the perspective of the treatment plant. Changing the FU of the model to a certain period of operation at the plant Öresundsverket, the results would show the plant’s capacity of P and N recovery in the two scenarios. The amount of P and N recovered could be seen in each scenario at the WWTP under the given period, also the amount of chemical fertilizer being replaced. From the results the environmental impacts of electricity and chemicals used in the two scenarios could be compared, and also the avoided impacts from the chemical fertilizers replaced under that given period. The plant could be evaluated as a nutrient recycling spot that also treats wastewater.

About the products – sludge and struvite

Looking at the Struvite scenario, the difference in the amounts of the two products, sludge and struvite, is momentous, as the sludge:struvite is over 60:1 regarding dry mass. Struvite is a more concentrated material than sewage sludge in that it has a 6.4 times greater plant available P content (considering equivalent dry mass), and so it contributes to a larger amount of avoided fertilizer – 6.4 times more, based on mass. Furthermore, recovery of struvite reduces the impacts of transportation, as struvite has less than 15% (14.7%) of the sludge’ impact (considering equivalent dry mass). The difference between the heavy metal concentration of the sludge and of the struvite (and of the chemical fertilizers also) is also considerable. The concentration of Cd in sludge is multiple compared to chemical fertilizers, and looking at zinc and copper it is higher by more than one digit. The application of sludge means higher amount of heavy metals spread on land, considering equivalent dry mass, but sludge has to be applied in larger quantity because of its lower plant availability of P and N. The relatively much smaller amount of struvite produced could not contribute significantly to reduce the impacts of the system per FU, even though its properties are suitable for that.

Future perspectives

The avoided fertilizer production proved to be a highly important factor, since it was the one the model was most sensitive to in both scenarios. It decreased the impacts in all categories and would possibly decrease them further in case of a larger volume of struvite produced per FU. It was clear that avoided fertilizer production also decreased the difference between the impacts of the two scenarios. Operating on full scale could potentially decrease that difference further, making the scenario as favorable (or more) than the Refernce scenario. The increased use of electricity caused by the full-scale struvite production would increase the impacts in most of the categories, but it is assumed that not considerably as the model proved to have low sensitivity to this factor. The chemicals used for the precipitation of struvite ($MgCl_2$, citric acid) did not have considerable effect on the impacts, i.e. the model was not sensitive to these elements. In case of full-scale precipitation, they would presumably not contribute much to the impacts either. Ferric chloride, the chemical used in largest amount also seemed important, as its production contributed to the impacts in the category ‘human toxicity’ in both scenarios. Even though a decrease in the used amount was not experienced in this study as discussed earlier, it is likely that it would occure, specifically when the operation goes on full-scale. Further, the increased struvite precipitation would decrease the volume of sludge produced at the WWTP. This could mean a decreased impact of transportation. Along with the volume, the P concentration of the sludge would also lower in case operating on full-scale, as an increased amount of P would be present in the form of struvite. This increased amount of struvite would mean more plant available P and N, and more chemical fertilizer replaced.

In order to enhance the efficiency of P recovery in form of struvite, Ekobalans has developed another concept of the method (Thelin, pers. com.). It included dewatering the sludge directly after the biological treatment, before mixing it with the primary sludge and hence recovering the P from that reject water and also from the liquor coming from the original sludge dewatering (Fig. 12). This separate dewatering can enhance the proportion of P recovered as struvite to up to 60% of incoming P to the plant (instead of the 20-25% with the current scheme), according to Ekobalans. However, this would mean a decreased amount of P remaining in the sludge.

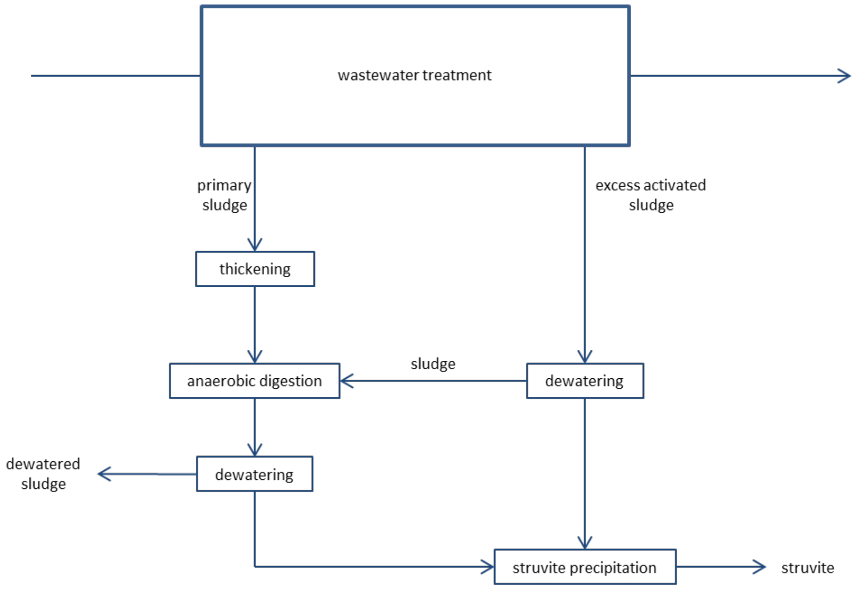

Fig. 10 Flow chart of the separate dewatering concept for increased struvite precipitation, according to Ekobalans

In Sweden, just like in the whole EU there are requirements for the use of sewage sludge on land. These regulations concern the content of heavy metals and pathogens in the sludge, as well as the organic content. Limits set for these criteria assure the quality of the sludge, and also its use for agricultural purposes. There are certain surfaces on which use of sludge is prohibited, i.e. where vegetables and fruits are grown, grazing and pastureland (European Commission, www). In cases where sewage sludge is not allowed to use, because of its quality or the type of surface, struvite has an advantage. This because of its low concentration of pollutants, like heavy metals, and the lack of pathogens and organics it can be applied on these areas. Due to its slow release of nutrients, it can potentially be used near surface waters, where sludge is prohibited in some countries. Though struvite is an ideal soil amendment considering its nutrient concentration and plant availability, it lacks providing organics that some farmers interested in. From this respect sewage sludge can have benefits, as its organic matter content is approx. 70% of TS in the treatment plant Öresundsverket (NSVA, www). Use of sludge is ideal when organic matter is the desired component, but it also has higher concentration of heavy metals and other potential polluters than struvite, which has to be considered.

In Sweden, REVAQ certification of the sewage sludge is important. REVAQ is a quality assurance certification system, established by the Swedish Water & Wastewater Association. REVAQ-certified sludge must have Cd:P ratio below 30 mg Cd/kg P (SWWA, www). This criterion is different from the one stated in the legislation, which determines the quality of sludge according to its Cd:total solids ratio (Cd/kg TS). When P content of the sludge is decreased, as a result of part of the P being precipitated as struvite, it results in an increase in the Cd/P ratio, which poses a risk that the sludge cannot be certified and hence used for agricultural purposes. This is a major reason for the treatment plant Öresundsverket, and many other WWTPs in Sweden, neither being willing to operate with the concept of separate dewatering of sludge and enhanced struvite precipitation, nor to proceed on full-scale instead of pilot-scale struvite precipitation. Determining Cd content relative to P is based on the concept that farmers who accept sludge for spreading are primarily interested in the P content of the sludge. This might not be general, because some farmers are more interested in the organic matter- and/or the nitrogen- and/or micro nutrient contents of the sludge (Thelin, pers. com.).

CONCLUSIONS

Two scenarios of wastewater and sludge treatment were compared in the assessment with respect of struvite recovery, at a WWTP in Sweden. It was seen from the results that the Struvite scenario was less favorable per kg of plant available phosphorus recycled as it had lower avoided impacts in all categories investigated than the Reference scenario. The impacts of the two scenarios did not differ considerably in any of the categories, considering that the operation of the struvite precipitation was a small pilot, and the uncertainties were also large due to unsure inventory results. This small difference was possibly due to that the level of operation in case of the struvite precipitation was not sufficient for reaching a marked decrease or increase in the impacts. These results were case-specific, and to some extent unexpected, considering the high P concentration and plant availability, and concentrated form of the struvite. Running the precipitation on full-scale instead of pilot-scale would increase the amount of struvite recovered, and would presumably increase the avoided impacts, due to the more avoided fertilizer production and the decrease of transport. In this case the impacts of chemicals and energy used for the precipitation would also increase, however the sensitivity results show that these factors did not prove to be important for the results.

The avoided fertilizer production had the biggest influence on the impacts. The combined effects of replacing TSP and AN resulted in a decrease in environmental impact in all the categories in both scenarios, i.e. the recovery of sludge and struvite was beneficial in an environmental perspective compared to the production of chemical fertilizers for all the assessed impact categories. It demonstrated that this form of system expansion is of high importance. Another crucial factor in the model was the ferric chloride used by both scenarios in the greatest volume among the chemicals. It made one of the greatest contributions to the impact category ‘human toxicity’. The possible reduction in the volume of ferric chloride at the Struvite scenario was not reflected in this study, as it had been expected beforehand, but further analysis are worth doing since uncertainties were major regarding this topic in this study. The chemicals magnesium chloride and citric acid used specifically for the struvite precipitation did not have any large effect on the impacts according to the assessment, neither proved to be crucial in the sensitivity analysis. The results and findings mentioned demonstrated that struvte precipitation can increase the environmental impacts of a WWTP, but holds potential also to be as favorable (or more) than operating conventionally (excluding struvite) in case of full-scale precipitation. The pilot-scale production of struvite proved to increase the environmental impacts to some extent, and that increase was not compensated fully. According to the study, the chemicals used for the struvite precipitation are not considerable, as they are increasing the environmental impacts of the plant slightly. The results can be valuable for future research on struvite recovery in treatment plants.

Among the clear benefits of struvite recovery, it must be mentioned that it helps to avoid the spontaneous precipitation of struvite in the pipes, reducing the maintenance cost at WWTPs operating with biological nutrient removal. Precipitating struvite recovers nutrients in an utilizable form in treatment plants as a byproduct of the treated wastewater. The extra chemicals used for the precipitation do not mean considerable environmental impacts as this study demonstrates. The high concentration of P and high plant availability of P and N of the struvite, next to its low concentration of pollutants like heavy metals makes it an ideal fertilizer product that has a compact, crystalline form. This form of the struvite facilitates a slow release of nutrients that avoids leaching from the fields, and also helps to reduce the costs and impacts of transport. It is ready for market, does not require further treatment after precipitation. Because of these properties struvite helps to ensure P security on a community level, and decrease the dependence on this finite resource. The drawbacks of struvite recovery are its extra use of electricity and environmental impacts, and the lack of organics in the crystalline product, organics which can be valuable for farmers.

The uncertainties were large in the model regarding inventory data and possibly poor documentation. As the first year of operation of the struvite recovery process was analyzed, these uncertainties and deficiencies had to be dealt with in the analysis. The boundaries of the model were expanded, but factors still important were left out of the system. Impacts regarding the handling of sludge, the spreading, and the emissions after spreading of the sludge and struvite have proved to be highly important in other studies, these impacts were not included in the present study.

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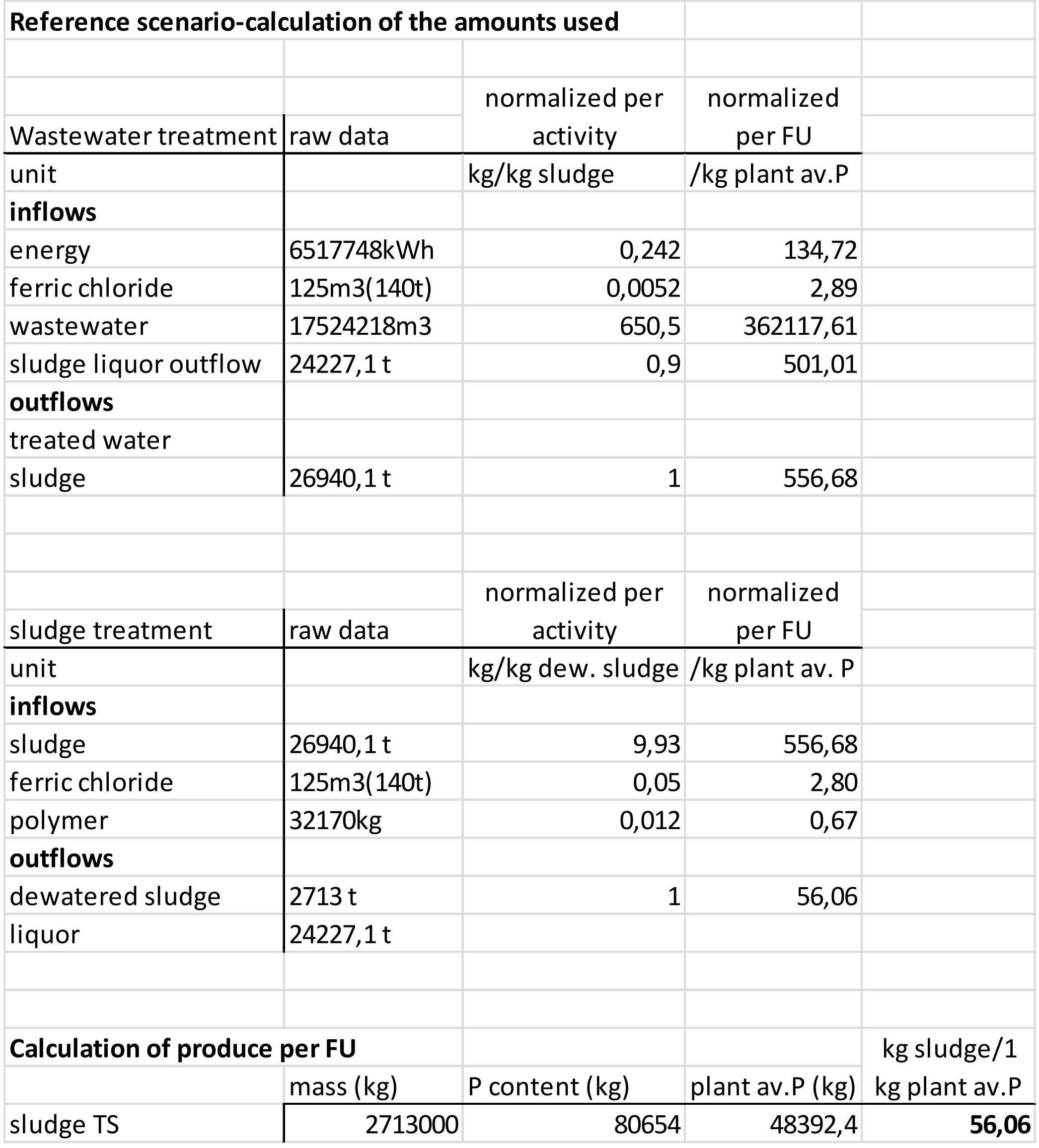
APPENDIX



Reference scenario-calculation of the amounts used				
Wastewater treatment	raw data	normalized per activity	normalized per FU	
unit		kg/kg sludge	/kg plant av.P	
inflows				
energy	6517748kWh	0,242	134,72	
ferric chloride	125m3(140t)	0,0052	2,89	
wastewater	17524218m3	650,5	362117,61	
sludge liquor outflow	24227,1 t	0,9	501,01	
outflows				
treated water				
sludge	26940,1 t	1	556,68	
sludge treatment	raw data	normalized per activity	normalized per FU	
unit		kg/kg dew. sludge	/kg plant av. P	
inflows				
sludge	26940,1 t	9,93	556,68	
ferric chloride	125m3(140t)	0,05	2,80	
polymer	32170kg	0,012	0,67	
outflows				
dewatered sludge	2713 t	1	56,06	
liquor	24227,1 t			
Calculation of produce per FU	mass (kg)	P content (kg)	plant av.P (kg)	kg sludge/1 kg plant av.P
sludge TS	2713000	80654	48392,4	**56,06**

Struvite scenario-calculation of the amounts used

Wastewater treatment	raw data	normalized per activity	normalized per FU
unit		kg/kg sludge	/kg plant av.P
inflows			
energy	6789206kWh	0,276	142,52
ferric chloride	125m3(140t)	0,0057	2,94
wastewater	23904554m3	971,47	501628,25
outflows			
treated water			
sludge	24606,5 t	1	516,36

sludge treatment	raw data	normalized per activity	normalized per FU
unit		kg/kg dew. sludge	/kg plant av.P
inflows			
sludge	24606,5 t	9,93	516,36
ferric chloride	125m3(140t)	0,056	2,91
polymer	32170kg	0,013	0,68
outflows			
dewatered sludge	2478 t	1	52,00
liquor	22128,5 t		

struvite precipitation	raw data	normalized per activity	normalized per FU
units		kg/kg struvite	kg/kg plant av.P
inflows			
energy	22128,5 kWh	0,55	0,462
liquor	22128,5 t	555,57	466,6788
MgCl2	53108,4 kg	1,33	1,1172
citric acid	553,2 kg	0,01389	0,0116676
outflows			
struvite	39,83 t	1	0,84

Calculation of produce per FU

	mass (kg)	P content (kg)	plant av.P (kg)	kg/1 kg plant av.P		
struvite	39830	4978,75	4978,75	**0,84**	→	mass struvite/ sum plant av. P
sludge TS	2478000	71138	42682,8	**52**	→	mass sludge TS/ sum plant av.P
		sum:	47661,55			

In this case, calculating how much sludge and struvite needed for 1 kg plant av. P, the allocation had to be managed. Certain amounts of the two produce combined needed to present this 1 kg plant av. P, so they were divided according to their mass. Doing so, 52 kg of the sludge and 0,84 kg of the struvite present 1 kg plant av. P, the FU.

Reference scenario-calculation of the impacts per FU

activity	used amount/FU	calculated impact/FU
energy prod.	134,7 kWh	
acidification pot.		0,0187
climate change pot.		5,54
eutrophication pot.		0,0195
human toxicity		2,26
ferric chloride prod.	5,7 kg	
acidification pot.		0,00380
climate change pot.		3,56
eutrophication pot.		0,00973
human toxicity		6,11
polyacrylamide prod.	0,67 kg	
acidification pot.		0,0111
climate change pot.		1,91
eutrophication pot.		0,00489
human toxicity		0,604
transport	76 km	
acidification pot.		0,00375
climate change pot.		0,682
eutrophication pot.		0,00591
human toxicity		0,250
avoided TSP prod.	4,8 kg	
acidification pot.		0,245
climate change pot.		16,20
eutrophication pot.		0,0753
human toxicity		16,43
avoided AN prod.	4,9 kg	
acidification pot.		0,0658
climate change pot.		16,22
eutrophication pot.		0,0797
human toxicity		5,91

Struvite scenario-calculation of the impacts per FU

activity	used amount/FU	calculated impact/FU
energy prod.	143,7 kWh	
acidification pot.		0,0199
climate change pot.		5,91
eutrophication pot.		0,0208
human toxicity		2,41
ferric chloride prod.	5,9 kg	
acidification pot.		0,00393
climate change pot.		3,69
eutrophication pot.		0,0101
human toxicity		6,32
polyacrylamide prod.	0,68 kg	
acidification pot.		0,0113
climate change pot.		1,94
eutrophication pot.		0,00496
human toxicity		0,613
transport	700 km	
acidification pot.		0,00354
climate change pot.		0,645
eutrophication pot.		0,00559
human toxicity		0,237
calcium chloride prod.	1,12 kg	
acidification pot.		0,00232
climate change pot.		0,437
eutrophication pot.		0,00211
human toxicity		0,460
citric acid prod.	0,012 kg	
acidification pot.		0,00021
climate change pot.		0,0383
eutrophication pot.		0,00016
human toxicity		0,0144
avoided TSP prod.	4,8 kg	
acidification pot.		0,245
climate change pot.		16,20
eutrophication pot.		0,0753
human toxicity		16,43
avoided AN prod.	5,0 kg	
acidification pot.		0,067
climate change pot.		16,56
eutrophication pot.		0,081
human toxicity		6,03

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