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A technique for sanitizing sewage sludge
Urea treatment with dual advantages

En metod för hygienisering av avloppsslam
Ureabehandling med dubbla fördelar

Ida Sylwan

ABSTRACT

A technique for sanitizing sewage sludge - Urea treatment with dual advantages

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Sewage sludge contains valuable plant nutrients, which to a higher degree than the current could be utilized in agriculture. First and foremost the phosphate contents in sewage sludge could be an important contribution in the highly productive and resource demanding agriculture of today. Only around 15 % of the sewage sludge produced in Sweden is currently recycled to productive land.

There is in Sweden a political goal to recycle at least 60 % of the phosphorus in wastewater to productive land by 2015, as stated in the Swedish Environmental Objectives. In the future access to the currently most common phosphorus source in agriculture, easily mined phosphate minerals, is thought to decrease.

Agricultural application of sewage sludge is however not an uncontroversial issue because of the contaminants, mainly heavy metals, contained in sludge. Pathogens are another concern.

Upcoming legislation suggests sanitization requirements for sewage sludge used in agriculture. This has previously not been required and might thereby create a market for new solutions for sanitizing sludge.

The sanitization technique studied is addition of urea to sewage sludge. At sufficiently high concentrations ammonia becomes toxic to microorganisms. Achieving sanitization through urea treatment has previously been studied for various organic materials; such as source separated dry material, co-compost and single use biodegradable toilets. The method has in these studies been found effective.

This study was designed for evaluating application of urea at the inlet of decanter centrifuges at sewage treatment plants, for the purpose of sanitization of the dewatered sludge. The sanitizing effect after application of urea to dewatered sludge was studied separately.

The results indicate that addition of urea at the inlet of decanter centrifuges would cause a too high ammonia concentration in the reject water. More than one fifth of the urea added ended up in the liquid effluent which corresponded to reject water. The expected increase in nitrogen load for sewage treatment plants was calculated to between 4 and 8 %. Addition of urea to dewatered sludge caused a clear decrease in studied pathogens. Approximations of the costs for sanitization of sludge by urea addition indicated that urea sanitization would be a low cost method compared to liming, pasteurization and thermophilic anaerobic digestion.

Keywords: sanitization, urea, ammonia, sewage sludge, wastewater sludge, sludge treatment

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REFERAT

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Slam från reningsverk innehåller värdefulla växtnäringsämnen som i högre utsträckning skulle kunna utnyttjas i jordbruket. Först och främst är fosforresursen i slam värdefull att ta till vara i dagens högproduktiva och resurskrävande jordbruk. Idag används endast kring 15 % av det avloppsslam som produceras i Sverige i jordbruket.

Genom Miljömålen har Sverige som politiskt mål att senast 2015 återföra minst 60 % av fosforn i avlopp till produktiv mark. I framtiden tros tillgången till dagens största fosforkälla, fosfatmineral av god kvalitet, komma att bli knapp.

Slamanvändning i jordbruk är dock ingen okontroversiell fråga på grund av de föroreningar, främst tungmetaller, som avloppsslam innehåller. Patogener är ett annat bekymmer.

I en ny förordning som föreslagits av Naturvårdsverket finns krav på att hygienisera avloppsslam innan användning i jordbruk, något som för närvarande inte krävs. Detta gör att marknaden för hygieniseringslösningar kan komma att växa framöver.

Hygieniseringstekniken som studeras här är tillsats av urea till slam. Urea bryts efter inblandning i slam ner till ammoniak, som i tillräckligt höga koncentrationer är giftigt för mikroorganismer. Hygienisering genom ureatillsats har tidigare studerats för flera typer av organiska material, bland annat för källsorterade avloppsfraktioner, blandad kompost och för nedbrytbara engångstoaletter. I dessa fall har metoden visat sig vara effektiv.

I denna studie utvärderades tillsats av urea vid inloppet till dekantercentrifuger, med syfte att hygienisera det avvattnade slammet. Effekten av tillsats av urea till ett avvattnat slam undersöktes också separat.

Resultaten indikerar att tillsats av urea vid inloppet till dekantercentrifuger skulle orsaka en för hög ammoniakkoncentration i rejektvattnet då mer än en femtedel av den tillsatta urean hamnade i vätskefasen efter separeringen. Kvävebelastningen på reningsverket förväntas enligt beräkningar öka med mellan 4 och 8 %. Tillsats av urea till avvattnat slam gav en klar minskning av patogena mikroorganismer. Beräkningar pekade på att hygienisering av avvattnat slam genom tillsats av urea skulle vara en billig hygieniseringsmetod jämfört med kalkning, pasteurisering och termofil anaerob rötning.

Nyckelord: hygienisering, urea, ammoniak, avloppsslam, slambehandling

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PREFACE

This master thesis was performed at the Department of Energy and Technology at the Swedish University of Agricultural Sciences, Uppsala. It was funded by a cluster group for research in the water and wastewater sector based in the Mälardalen region. Supervising the work was associate professor Björn Vinnerås and subject reviewer was Professor Håkan Jönsson, both at the Department of Energy and Technology at the Swedish University of Agricultural Sciences, Uppsala.

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POPULÄRVETENSKAPLIG SAMMANFATTNING

Hygienisering av avloppsslam syftar till att minimera riskerna för smittspridning då slammet används i produktion av livsmedel eller som anläggningsjord där människor kan komma i kontakt med jorden. I dagsläget finns flera beprövade metoder för hygienisering av avloppsslam, bland annat termofil rötning, kompostering, långtidslagring och kalkning. Ureabehandling är en ny metod i dessa sammanhang, men bygger på en princip som länge varit känd; urea omvandlas till ammoniak vid kontakt med slam och den ammoniak som bildas har en negativ verkan på patogena mikroorganismer.

Urea, också kallat urinämne, är världens vanligaste konstgödselmedel. Av den urea som produceras globalt används 90 % i jordbruket, men det finns också många andra användningsområden, som till exempel i hudvårdsprodukter, ämnet kallas då karbamid, och i brandsläckare. Produktionen av urea utgår från naturgas, vilken omvandlas först genom Haber-Boschprocessen till ammoniak och sen genom Bosch-Meisers process till urea. Alternativt skulle biogas kunna användas för framställning av urea. Även om ureabehandling kräver utnyttjande av en fossil resurs, så kan det ses som att den endast kräver omdirigering av ett redan existerande flöde. Genom att den urea som direkt kunde ha använts för gödsling istället går via avloppsslammet, för hygienisering innan återföring till jordbruk, kan även de näringsämnen som finns i slammet utnyttjas.

Examensarbetets rubrik, där det hävdas att dubbla fördelar föreligger vid ureabehandling, syftar dels på den billiga hygienisering som kan uppnås och dels på det ökade gödselvärde som slammet uppnår genom tillsats av urea. Tillsats av urea till avloppsslam skulle därför utöver den hygieniserande effekten vara positivt ur gödselvärdessynpunkt. Den ökade mängden kväve i slammet skulle komplettera den höga halten fosfor i slam. Slammet skulle därigenom kunna bli mer attraktivt för jordbrukare.

Urea är ett billigt ämne och det är lätthanterligt i sin fasta form. Den hygiensiserande verkan uppkommer då urean brutits ned. Det är enzymet ureas som driver nedbrytningen och produkterna som bildas är ammoniak och koldioxid. Ammoniak är i vatten i jämvikt med ammonium, och ammonium kan i sin tur nitrifieras, omvandlas till nitrat. I form av ammonium och nitrat är kväve tillgängligt för upptag av växter. Men ammnoniak är i höga koncentrationer också toxiskt för mikroorganismer. Den nämnda jämvikten mellan ammoniak och ammonium förskjuts vid höga pH mot ammoniak, vilket gör att hygiensieringen blir mer effektiv vid högt pH. Studier har visat att hygiensiering med ammoniak är mer effektivt vid en högre temperatur.

Med fördel skulle ureahygienisering kunna användas för slam som är tänkt att appliceras på produktiv mark. Debatten kring slamanvändning i jordbruket pågår och frågan är inte okontroversiell. Sedan 2008 finns ett system för certifiering av slam för återföring till jordbruk, REVAQ. Genom detta verkar man för att slam som används i jordbruket ska hålla en god kvalité och att kvalitén samtidigt kontinuerligt ska förbättras. Sverige har också genom miljömålen bestämt att verka för att senast år 2015 ha minst 60 % återföring av fosforn ur avlopp till produktiv mark. Just nu finns också förslag på en ny förordning som kommer att skärpa reglerna för återföring av slam till jordbruket genom att krav på hygienisering läggs till, något som tidigare inte funnits i Svensk lag.

Den idé som testats genom detta examensarbete är att föra in urea i behandlingen av avloppsslam. Närmare bestämt vid inloppet till dekantercentrifuger. Dekantercentrifuger används ofta för slamavvattning och avvattning är oftast det sista steget i slambehandlingen. Hypotesen i denna studie var att genom införsel av urea i slammet under centrifugering skulle en god inblandning åstadkommas, medan det helst inte skulle ske några större förluster till rejektvattnet som bildas i avvattningen. Försök gjordes på liten skala, där slam med tillsats av urea avvattnades. Resultaten från analys av avvattnat slam och av vattnet som bildades vid avvattningen visade att en stor del av den tillsatta urean hamnade där det inte önskades, i vattenfasen. Därför kunde konstateras att applicering av urea vid inloppet av dekantercentrifuger förmodligen inte är något lämpligt alternativ då urean löser sig för snabbt.

Även de hygieniserande effekterna vid tillsats av urea till redan avvattnat slam studerades och denna studie utmärker sig här genom att hygienisering testades för en högre torrsubstanshalt än i tidigare studier. En mycket god hygiensiering kunde påvisas när det gällde salmonella. Kostnadsberäkningar för ureahygienisering av avvattnat avloppsslam indikerade att metoden skulle vara billig jämfört med andra vanliga hygieniseringsmetoder.

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GLOSSARY

AD - anaerobic digestion

Decimal reduction - reduction to a tenth of the initial amount

DM - dry mass

Grey water - Water used in households, for washing and doing dishes and laundry

Infection dose - the number of organism required to cause an infection

Log-unit reduction - see decimal reduction

Logarithmic reduction - the reduction observed studying log-values

LRF - Association of Swedish farmers

N-tot - total nitrogen

Pe - person equivalent

SEPA - Swedish Environment Protection Agency

Storm water - precipitation and runoff

STP - sewage treatment plant

SWWA - Swedish Water and Wastewater Association

TS - total solids

Urea - organic substance with the chemical formula $(NH_2)_2CO$

1 INTRODUCTION

Urea was, in 1828, the first organic substance to be synthesised by man. It is produced from natural gas and in the year 2000 more than 100 million tons of urea were produced in the world. Of the urea produced 90 % is used for fertilization (Kemikalieinspektionen:1, 2010-02-25), which makes urea one of the most frequently used nitrogen fertilizers in the world (Ottoson et al., 2008).

When applied to soil urea degrades into ammonia (NH_3) and carbon dioxide (CO_2). A reaction controlled by the enzyme urease which is naturally present soils (Winker et al., 2009). The same reaction occurs when urea is applied to sewage sludge. Urea has the chemical formula $(NH_2)_2CO$.

Urea addition, and the following degradation into ammonia, has been shown to give an efficient die off for pathogenic microorganisms (Vinnerås et al., 2009; Nordin et al., 2009; Ottoson et al., 2008). The reason is that the ammonia produced after degradation is toxic to microorganisms in sufficiently high concentration. Application of urea could therefore be a future alternative for sanitization of sewage sludge. The method is particularly interesting for sludge that is to be utilized for fertilization since addition of urea will increase the fertilizer value of sludge.

The sustainability of applying urea, which today is the product of a not renewable energy source, can be discussed. However, the application of urea to sewage sludge can be seen as redirecting an already existing flow and thereby better utilizing the resources available, as long as the treated sludge is used for fertilizing. Alternative production of urea, from biogas instead of natural gas, might be a possibility in the future.

Weather recycling of sewage sludge to agricultural land is appropriate is subject to an ongoing debate. A larger recycling of sewage sludge would enable a decreased consumption of mineral phosphorus, which is taken from reserves that are thought to become small within the coming century. Arguments for not recycling sludge are that the contaminations contained could have negative influence on soil quality and in extension affect human health and have negative environmental impacts. Heavy metals, organic contaminants and pharmaceutical residues are all present in sludge in various concentrations. However, heavy metals and organic contaminants can be reduced by source control, i.e. dealing with contaminations before they enter the wastewater stream. Regarding pharmaceutical residues, their biodegradability plays a great role. To keep track of the contaminant contents in sludge used in agriculture both laws and a certification system exists.

Since pathogens are present in sludge it means that there is a risk of disease transmission and the risk will prevail if the material is not properly sanitized. It will therefore always be important to minimize pathogen amounts before making use of the sludge resource. New legislation coming into effect in the near future gives an additional incentive to improve sanitization of sludge. Swedish laws have previously only regulated heavy metals and organic contaminants but a new bylaw suggestion also include limiting values for the amounts of pathogenic organisms allowed when recycling sewage sludge.

Sludge from sewage treatment plants can be seen as waste or as a resource. In this era of recycling the more compelling alternative is to view it as a resource for use in agriculture or on other productive land and Sweden has an environmental goal concerning recycling of the phosphorus in sewage sludge to productive land.

1.2 OBJECTIVES

The aim of this study was to evaluate urea application to sewage sludge, for sanitization of the dewatered sludge, in connection with dewatering through centrifugation at sewage treatment plants.

The subobjectives were, 1) to investigate the flow of nitrogen during dewatering when adding urea to thickened sludge at the inlet of decantercentrifuges, 2) to investigate the sanitization process when adding urea to dewatered sludge.

1.3 LAYOUT OF THE STUDY

In order to reach the objectives a literature study, a survey to sewage treatment plants and two lab studies were performed.

A literature study was performed with the aim of gathering information on how sludge is used in Sweden, which laws that govern the use, what the pathogen contents in sewage sludge are and which the possible methods for sanitizing sewage sludge are, with emphasis on urea treatment. The following questions were formulated.

- Why sanitize sewage sludge and in which ways can this be done?
- What has been found in previous studies on sanitization?

A survey was sent to sewage treatment plants in the Mälardalen region, Sweden. The purpose of the survey was to find common conditions for dewatering of sewage sludge, in order to set as realistic conditions as possible for lab studies. A further purpose was to learn about the current handling of sewage sludge at the studied sewage treatment plants, to enable the evaluation of the possibility of applying urea treatment in the Mälardalen region.

A lab study was performed to find whether urea addition at the inlet of decantercentrifuges could be used for sanitization of the dewatered sludge, without an excessive increase in the nitrogen load on the sewage treatment plant by increased ammonia concentration in the reject water.

Another lab study was performed to evaluate the sanitization after addition of urea to dewatered sludge.

2 BACKGROUND

2.1 CURRENT HANDLING OF SEWAGE SLUDGE IN SWEDEN

In Sweden approximately 207 000 tons dry mass of potentially utilisable sludge is produced annually. For 66 % of the sludge produced in 2006 the usage was declared in the report Utsläpp till vatten och slamproduktion 2006 (Naturvårdsverket & Statistiska centralbyrån, 2008). Of the known use 31,000 tons was used in agriculture. That means a minimum of 15 % of the total production was recycled to serve as fertilizers on agricultural land. In the Swedish province Skåne as much as 48 % of wastewater sludge was used in agriculture. The most common use for Sweden as a whole, 46,000 tons, was "Other land use" half of which was used for covering landfills (Naturvårdsverket & Statistiska Centralbyrån, 2008). The distribution between different uses is shown in Figure 1.

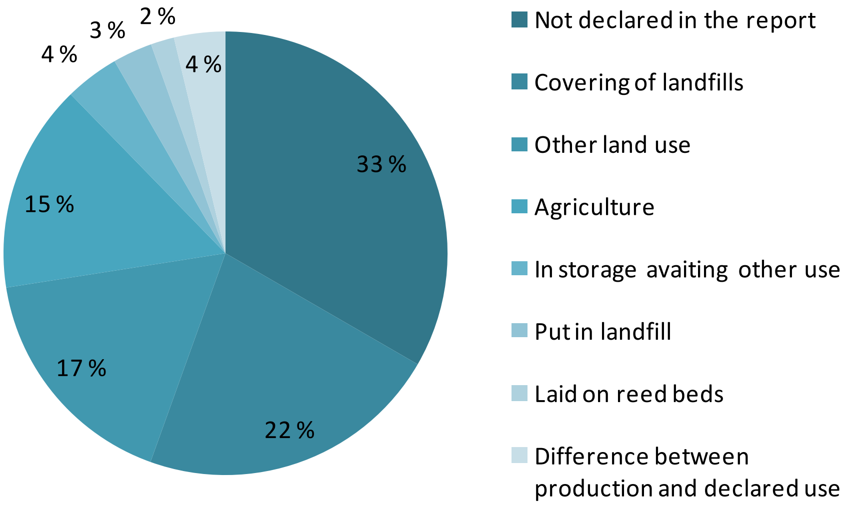

Figure 1. Sewage sludge disposal in Sweden (according to figures from Naturvårdsverket & Statistiska Centralbyrån, 2008)

Comparing the Swedish situation to that in other European countries it becomes evident that Swedish farmers have a quite cautious approach to using sludge in agriculture. The agricultural usage in our neighbouring countries Denmark and Norway is much higher in relation to the population (Statistiska Centralbyrån et al., 2007).

In the context of utilizing sewage sludge in agriculture there are, contrary to the common perception among the public, in most cases not very high heavy metal contents in sewage sludge and the concentrations generally do not exceed the limiting values. In 2006, less than 8 % of the sewage treatment plants declared concentrations of heavy metals or organic substances in the sludge exceeding the regulated values. Of the sludge accounted for, 50 % had concentrations not exceeding the set limits or guidelines for any of the regulated substances. In 42 % of the cases analyzes were lacking for some compounds, but for the contaminants of known concentration the sludge met the standards (Naturvårdsverket & Statistiska centralbyrån, 2008). As a reservation, it could be claimed that the limiting values are set too high or too low, that is however a different issue. According to Vinnerås et al. (2008), the contents of nutrients in human excreta could potentially replace about 20 % of the mineral fertilizers used.

2.1.1 Swedish environmental objective on sewage sludge

When revising the Swedish Environmental Objectives, “Miljömålen”, in 2005, the Swedish government added a new formulation under the interim targets for waste which are subordinate to environmental objective number 15, “A good built environment”. The goal, which addresses the urgency to appreciate the phosphorus resource in sewage sludge, is there stated as (Miljömålsportalen, 2009-11-18):

> “By 2015 at least 60 percent of phosphorus compounds present in wastewater will be recovered for use on productive land. At least half of this amount should be returned to arable land.”

The judgement made by SEPA, the Swedish Environmental Protection Agency (Naturvårdsverket), from analysing the present situation, is that this goal will be hard to reach in time. In 2003 the proportion of recycled phosphorus was about 10 % of the total incoming amounts of phosphorus to sewage treatment plants (Statistiska Centralbyrån et al., 2007).

2.1.2 The sludge debate

Technically and in a cost perspective the easiest way to recycle phosphorus is by returning the sludge to agricultural land (Laturnus et al., 2007; Bengtsson & Tillman, 2004). Opposition against sludge use in agriculture exists, both from environmental organisations and from the food industry which is depending on the trust of consumers and therefore takes caution concerning this issue (Bengtsson & Tillman, 2004). Another major stakeholder is the Federation of Swedish Farmers, LRF, which had a large impact on the use of sludge in agriculture when they in 1999 recommended their members not to use sludge. This event is the main reason why the levels of sludge use in agriculture today are down to 15 %, while they in the 1990s were around 25-30 % (Bengtsson & Tillman, 2004).

Hazards feared are increasing contents of heavy metals and organic contaminants, which might in long-term lead to degradation of soils and might also, if contaminants are transferred to crops or water bodies, have effects on human health and animals. Another concern is pathogens leading to disease transmission (Laturnus et al., 2007; Vinnerås et al., 2008). But the biggest obstacle might be the cultural context where the thought of using human excreta in association with food production does not have mainly positive connotations for most people.

Since 2008 the Swedish Water & Wastewater Association, SWWA (Svenskt Vatten), has been running the project REVAQ (Svensktvatten, 2009-10-30), a certifying program for attaining safe recycling of sewage sludge. Today LRF has a more positive attitude towards using sewage sludge as a fertilizer, but their recommendation is to use only sludge certified by REVAQ (LRF:1, 2009-10-30). Hitherto 22 sewage treatment plants and water corporations have been certified and additionally around ten plants are waiting for their applications to be handled (Svensktvatten, 2009-10-30), the certified sludge comprises 40 % of the total amount of sludge produced in Sweden (Naturvårdsverket, 2009). With a reliable certifying system the issue of trust in the producer-consumer link might now be met.

2.2 REGULATIONS FOR AND CERTIFICATION OF SLUDGE USED IN AGRICULTURE

Swedish regulations for the usage of sewage sludge in agriculture are found in paragraph 20 (20§) in the bylaw 1998:944. The regulation gives maximum concentrations of a number of heavy metals when sludge is used in agriculture. The regulated metals are lead, cadmium, copper, chromium, mercury, nickel and zinc.

There is also a precept on sludge use in agriculture, SNFS 1994:2, that regulates the following:

- The maximum amount of phosphorus and ammonium that may be applied through application of sewage sludge
- The allowed soil contents of heavy metals when applying sewage sludge
- The maximum amount of heavy metals that may be added trough sludge application

The maximum amount of phosphorus that may be added through application of sewage sludge is 22 kg/(ha·yr) or 35 kg/(ha·yr) depending on soil phosphorus concentration. For 85% of the Swedish farmland the lower limit applies (Eriksson, 1997). Regarding allowed soil contents of heavy metals and the maximum heavy metals addition allowed the strictest rules have been set for the non-essential heavy metals mercury and cadmium.

Concerning organic contaminants there is a national agreement, made in 1994, between LRF, SEPA and SWWA. The agreement is called “Slamöverenskommelsen” and it includes non-binding guidelines for maximum contents of four types of organic contaminants; PAHs (polycyclic aromatic hydrocarbon), PCB (polychlorinated biphenyls), Nonylphenols and Toluene. Toluene was however dismissed from the guidelines in 1999 since it was found that it might be formed during wastewater treatment (Naturvårdsverket & Statistiska Centralbyrån, 2008).

The laws of EU are superior to Swedish legislation on sludge. EU laws on sludge are found in EG directive (86/278/EEG). Similar to Swedish legislation it regulates the maximum addition of heavy metals to soils, the maximum contents of heavy metals in soil where sludge is applied and the maximum heavy metal contents in the sludge applied. The directive was (according to the directive itself) set for the purpose of regulating the usage of sludge so that effects on soil, vegetation, animals and man would be prevented. The directive is a minimum directive and several European countries have set stricter regulations than those of the EU (Naturvårdsverket, 2002). The limiting values are 10-100 times higher in the EG directive compared to Swedish legislation.

In addition to the regulations around sludge use in agriculture there is in Sweden a bylaw on landfills that affects the possibilities of sludge disposal i.e. bylaw 2001:512, the 10th paragraph, which prohibits disposing of material of high organic content in landfills.

There currently are no existing regulations concerning pathogen reduction required before applying sludge in Swedish agriculture.

2.2.1 Upcoming regulations

In 2009 the Swedish government commissioned SEPA to work out a new proposition for a bylaw concerning the usage and disposal of sewage sludge. The proposed bylaw is a revision of one that was suggested in Aktionsplan för återföring av fosfor ur avlopp (Naturvårdsverket, 2002). It was handed to the Ministry of the Environment on the 16th of November 2009. If accepted the bylaw is planned to take effect on the 1st of January 2012.

Regulations for heavy metal concentrations in sludge applied in agriculture are proposed to remain the same with the exception of cadmium and mercury, for which lower limiting values are proposed. A new metal, silver, is also added in the proposal. Similarly the limits for total addition of cadmium and mercury are proposed to be lowered, and also here a value is given for silver. It is also suggested that the limiting values for tin are dismissed. Regarding allowed soil concentrations of heavy metals when applying sewage sludge, the proposition is that they remain the same (Naturvårdsverket, 2009).

A new aspect in the suggested bylaw is that it contains requirements regarding sanitization. A number of treatments are recommended for sludge which is to be used in agriculture. The recommended treatments include drying supported by heat, pasteurisation, thermophilic anaerobic digestion (AD) with liquid composting, liming, decomposing the sludge in a reed bed, drying sludge in a drying bed and storage. The treatments and conditions required are listed in Table 1.

Two categories, class A and B, are suggested for the sanitization level achieved through the respective treatments. Class A sludge stands for the higher sanitization level. The possible uses of B-classified sludge will be more restricted as the risk of pathogen transmission is judged to be higher.

In the bylaw proposal it is also stated that other methods than the suggested ones can be used after they have been found equally effective and approved by SEPA. The sanitization requirements proposed are that no salmonella should be found in 25 grams wet weight and that the number of *Escherichia coli* (*E. Coli*) should be less than 1000 per gram of dry matter. Furthermore, there is an additional condition made for the sludge to be classified in the A-category, viz that the sludge should contain less than 1000 enterococci per gram of dry matter.

Table 1. Recommendations on sanitization treatment according to proposed bylaw (translation of “Tabell 1” in the bylaw proposal)

Class	Treatment method	Conditions required	Terms
A	Thermal drying	A temperature of 80°C for 10 min	All material shall reach the given temperature. The moisture content shall be less than 10 %
A	Pasteurization	A temperature of 70°C for 60 min	All material shall reach the given temperature.
A	Thermophilic AD and liquid composting	a. A temperature of 52°C for 10 h b. A temperature of 55°C for 6 h c. A temperature of 60°C for 2.5 h	All material shall reach the given temperature. The hydraulic retention time shall be 7 days at minimum with minimum temperature 52 °C
A	Lime treatment (quicklime)	A pH of 12 and a temperature of 55°C, for 2.5 h	All material shall reach the given temperature and pH.
B	Lime treatment (hydrated lime)	A pH of 12 for a period of 3 months	All material shall reach the given pH.
B	Treatment in reed bed or drying bed	1 year when no new sludge is added	-
B	Storing	1 year when no new sludge is added	-

2.2.2 REVAQ

REVAQ is a sludge certifying program that was started in 2008 and is run by SWWA. The demands of the certification are in others that the accumulation in soils to which the sludge is applied should be less than 0.2 % per year for 60 trace elements (REVAQ, 2010). To become certified sewage treatment plants have to set up a plan for improvements of sludge quality, for example by working actively with upstream emissions. The goals of REVAQ are among others to make information on the production of sludge easily accessible and to attain a sludge quality that fulfils the set demands.

The REVAQ rules also include demands for sanitization of sludge. The methods for sanitization proposed include storage for at least 6 months, thermophilic anaerobic digestion, liming and pasteurisation. Other methods can be approved if the producer of sludge can prove their efficiency. The efficiency of the sanitization is proved by taking a sample contaminated with salmonella, treating it, and showing that it after treatment is free from salmonella. This has to be done at three separate occasions.

REVAQ recommends immediate incorporation into the soil in connection with application of sludge.

2.3 SEWAGE SLUDGE AS A FERTILIZER

Sludge from sewage treatment plants can contain human excreta; faeces and urine, flushing water, grey water, storm water and water from industries in varying fractions. Additionally, at the end of the treatment process there will be a large amount of biomass which has grown during biological treatment.

Nitrogen (N) and phosphorus (P) are the plant nutrients of greatest importance. Sewage treatment has the function of reducing the concentrations of nitrogen and phosphorus in the treated wastewater to avoid eutrophication in the recipient. A large proportion of the nitrogen in wastewater will be removed through biological treatment, which is common in Sweden and was introduced in the late 80’s (Vattenportalen, 2009-12-14). Biological nitrogen reduction comprises of nitrification and subsequent de-nitrification which means the nitrogen is transferred to the atmosphere. Phosphorus is in sewage treatment often bound in ferrous or aluminium compounds through precipitation. Succeeding precipitation is sedimentation, which means most of the phosphorus in sewage ends up in the sludge.

Much of the nitrogen contained in sewage is lost to the atmosphere during sewage treatment but in spite of this the nitrogen contents in the resulting sludge are higher than the phosphorus contents. Even though nitrogen concentrations in sludge are higher than phosphorus concentrations phosphorus is the plant nutrient which has the highest importance in sewage sludge, since plants require more nitrogen than phosphorus for their growth. The total quantity of phosphorus in Swedish sewage sludge, about 6,000 tons per year, corresponds to more than 40 % of the contents in inorganic fertilizers used in Sweden today. In the case of nitrogen, this figure is only 9,000 tons, which corresponds to 5 % (REVAQ, 2010-03-12).

2.3.1 Recycling of phosphorus

Recycling phosphorus from sewage sludge back to productive land can be done either by direct application of a processed sludge or by, prior to recycling, extracting phosphorus from sludge by some means. The potential of the latter is currently small. Techniques of extracting phosphorus from sewage sludge or incinerated ashes exist, but these are still at the research stadium and none have been tried out on large scale in Sweden. There are though a few larger projects being tried in for example Germany and the US. These are connected to sewage treatment plants with biological phosphorus reduction. The methods cannot be used together with chemical treatment, which is the most common method of phosphorus reduction in Sweden (Naturvårdsverket, 2009). In the report Uppdatering av "Aktionsplan för återföring av fosfor ur avlopp" (Naturvårdsverket, 2009) a method being developed in Sweden is described. It is called EasyMining and aims to extract phosphorus from incinerated sludge. The method is now said to be under commercialization (Dagens Samhälle, 2010-01-18). Apart from the fact that techniques are not yet fully developed the construction of facilities for phosphorus extraction has been considered too costly (Lundin et al., 2004).

Another alternative for recycling phosphorus, which would completely alter the substrate dealt with, is source separating sewage into fractions; grey-, brown-, yellow- and storm water. The fractions could more easily be handled in an appropriate way since most contaminants are found in grey water and storm water with the exception for medical residues which has the highest concentration in urine and faeces.

A finite resource

The majority of phosphorus applied in agriculture is phosphate mineral. In Sweden 40 % of the agricultural land is supplied only with phosphate from mineral resources. The phosphorus fertilization in Sweden has however decreased largely since the 1970s, since phosphorus over the years has accumulated in soils (Statistiska Centralbyrån et al., 2007).

Worldwide about 85 % of the phosphorus mined is used in agriculture. Mining primarily takes place in China, US and Morocco. Easily mined phosphorus reserves of high quality are however limited (Naturvårdsverket, 2009). The quality of the reserves depends on their cadmium contents, which in many cases are too high. Currently there are no commercial methods for extracting cadmium from the mineral (Naturvårdsverket, 2009). Phosphorus in such is not a finite resource, since it is the 11th most common element on in the earth’s crust (Naturvårdsverket, 2002), the easily mined mineral phosphorus resources can however be said to be finite. The approximation of how long the resource will last varies among authors. According to Kroiss (2004), with the expected trend of increasing fertilizer usage, the phosphorus reserves of the world will be exhausted within 150 years. Expectations for if the usage of commercial fertilizer phosphorus remains at the level of today are that the resource could last more than 1000 years (Naturvårdsverket, 2009).

2.3.2 Other advantages of making use of sludge in agriculture

In addition to phosphorus and nitrogen sewage sludge also contains micronutrients, such as iron, copper, cobalt, zinc and selenium. Another positive effect is that sludge contains organic material and thereby can improve quality of soils as an effect of increased levels of soil organic matter (REVAQ, 2010-03-12). The soil quality improvements include improved texture features which favour root growth, and enhanced water holding capacity which makes the tolerance for drought higher (Düring & Gäth, 2002).

2.3.3 Hazardous substances in sewage sludge

Since sewage sludge is a mixture, emanating from all which is flushed down toilets, wash basins etc., plus storm water and industrial sewage, it will contain most of the substances used in our society. Source control for polluting substances is therefore of high importance. Heavy metal and organic contaminant contents can thereby be decreased. For reduction of pharmaceutical residues the key should be attaining an appropriate biodegradability of the substances used.

In general it can be said that there are many different views on the hazards of applying sewage sludge to agricultural land. However, according to a Swedish review on the matter, Uppdatering av “Aktionsplan för återföring av fosfor i avlopp” (Naturvårdsverket, 2009), there should be no negative effects on the soils microorganisms or soil fauna, and no significant risks that organic contaminants or pharmaceuticals would affect waters or the food produced negatively.

Heavy metal contaminations

Heavy metals tend to accumulate in soils; only a very small proportion will leave soils through leakage or plant uptake. This means plant uptake of heavy metals is small but nevertheless it could have large effect on the growth and quality of crops and may also affect human health (Defra, 2005). However, the toxicity of a metal mainly derives from the fraction that is easily soluble (Häni et al., 1996). Applying the same sludge batch to different soils would give varying effects depending on the soil characteristics which will influence solubility, and sorption (Häni et al., 1996; Düring & Gäth, 2002).

The effects on soil microorganisms from heavy metal contaminations in sewage sludge are largely unknown. But effects on soil respiration rates, soil microbial biomass and fixation of atmospheric nitrogen have all been shown. Heavy metals can also cause antibiotic resistance in microorganisms found in soils (Defra, 2005).

In many experiments on the effects on soil biology from sludge application, the sludge used have been spiked with high concentrations of heavy metals to be able to determine the effects more easily. The results can therefore be hard to translate to real conditions (Häni et al., 1996).

The highest concern for applying sewage sludge in agriculture has been a fear of cadmium contamination (Kroiss, 2004). This fear has arisen after increasing soil cadmium concentrations has been observed in many countries. Increasing cadmium concentrations can be derived from the application of fertilizers, soil amendments and bio-solids and from contamination through atmospheric deposition (McLaughlin et al., 1999).

Since it is more mobile than most other contaminating heavy metals cadmium is the most relevant heavy metal concerning food chain contamination (McLaughlin et al., 1999). The effects of low cadmium intake on human health have been debated. In areas of very high contamination, and with rice as the basic diet, actual impacts have been evident (McLaughlin et al., 1999). In epidemiological studies, negative effects have also been shown concerning many human body systems (renal dysfunction to the kidneys, bone demineralisation and cancer to lung and breast etc.) and therefore the European Food Safety Agency (EFSA) in 2009 recommended lowering the tolerable weekly intake to 2.5 µg/kg body weight, down from 7 µg/kg body weight previously provisionally recommended by FAO/WHO Expert committee on Food Additives (EFSA, 2009).

Eriksson (1997) investigated the heavy metal contents of sludge samples from 50 Swedish sewage treatment plants and found that cadmium in general was the limiting element regarding the amount of sewage sludge that can be applied to agricultural land under the current Swedish legislation. As of today, the biggest source of cadmium in agriculture is atmospheric deposition (Statistiska centralbyrån & Jordbruksverket, 2009). The situation might change if use of sewage sludge in agriculture was to become extensive.

Swedish sewage sludge has higher cadmium contents per kg phosphorus (on average 37 mg/kg P) than does manure (8 mg/kg P for pig and 20 mg/kg P for bovine) or commercial fertilizers (12 mg/kg P). On the other hand, it has lower cadmium content per kg phosphorus than the average values for commercial fertilizers on the world market (140 mg/kg P) (REVAQ, 2010-03-12).

Organic contaminants

The destruction of organic contaminants is typically 15-35 % in sewage treatment combined with sludge treatment (Defra, 2005). This means that a large part of the organic contaminants in sewage passes through sewage treatment plants and ends up both in the treated water and in the sludge produced.

In soils the toxicity of an organic compound, as of heavy metals, mainly derives from the fraction that is easily soluble (Häni et al., 1996), this since plant uptake goes via soil water (Erhardt & Prüeß, 2001). Solubility of an organic contaminant will vary between soils, depending on parameters as soil pH and contents of organic matter, clay and iron-oxides. This makes it difficult to predict the general outcome of application of a certain sludge batch although the sludge composition might be well known (Häni et al., 1996). However, many organic contaminants in sewage sludge are hydrophobic, i.e. not soluble in water (Erhardt & Prüeß, 2001), as hydrophilic organics will rather stay in the wastewater.

Studies have shown that effects on human health caused by organic pollutants from sewage sludge are generally minimal because of the low uptake in crops (Smith, 2009; Erhardt & Prüeß, 2001).

Many organic compounds are easily volatilized or quickly biodegraded (Smith, 2009) and thereby not of great environmental concern. However, a group of organic pollutants that have been pointed out as a probable hazard to human health and the environment are persistent organic pollutants, POPs. As the name suggests, these persist in the environment/degrade slowly. POPs may be found in places far from where they were produced or released. Some examples on POPs are PAHs, PCBs and PCDD/Fs (Smith, 2009).

The continuous introduction of new chemicals is a cause of concern for sludge quality since their environmental effects can be hard to foresee. This causes uncertainty regarding the effects of applying sludge in agriculture (Lundin et al., 2004).

Pharmaceutical residues

A subgroup of organic contaminants contained in sewage sludge is pharmaceutical residues. The hazards of which are largely dependent on the time required for their degradation. When the degradation is quick the probability of impacts on soil, crops or water is small.

Concerns exist about the difficulty in predicting the synergetic effect between different compounds. Another concern is that measurement of pharmaceutical residues after sludge application is complicated by uncertainties around which degradation products might have been formed. A degradation product might be well as toxic as the parent compound (Onesios et al., 2008).

That pharmaceutical residues contained in the wastewater stream end up in sludge could actually be seen as positive since the residues will then be subject to a higher pressure from microorganisms than they would be in the recipient water (Defra, 2005). The amount of micro-organisms in soil is much higher than that in water; one cubic meter of soil contains about as many microorganisms as one cubic kilometer of water (LRF:2, 2010-01-18).

2.3.4 Other potential drawbacks of applying sludge in agriculture

Beyond concerns for potential effects on soil quality and human health, the type of sludge distribution chosen affect for example energy consumption and euthrophication.

Eutrophication

According to Hållbarhet i svenskt jordbruk 2007 (Statistiska centralbyrån et al., 2007), agricultural land contributes with 45 % of the phosphorus load to water. Applying sludge in farmland can cause leakage of nutrients which might lead to eutrophication in surrounding water bodies if the amounts of phosphorus are high compared to the coinciding uptake of crops. The application of sludge in agriculture however has to be compared to the effects of other alternatives of disposal.

Energy consumption

Energy usage is for example connected to the transport from sewage treatment plant to field. This will be energy consuming and contribute emissions (and costs) which are not directly connected to the issue of nutrient recycling. Recycling could be compared to incineration, in which case energy can be produced instead of consumed. According to Lundin et al. (2004) the energy consumption would be large for returning sludge to agriculture, as compared to incineration with extraction of phosphorus. They considered production of district heating in connection with the incineration as “negative” energy consumption and the transport of sludge to farmland is in their calculations a big contributor to the high energy consumption. They did not though consider the transport of extracted phosphorus to farm land.

A report by Balmér et al. (2002) arrived at a different conclusion. Their calculations showed that the energy consumed for recycling sludge was lower than that of methods for extraction of phosphorus. And that the energy consumption was similar to that of incineration. The phosphorus extraction techniques considered by Lundin et al. (2004) and SEPA were the same two techniques (Bio-Con and Cambi-KREPRO). Thus, it can be concluded that the energy issue is complex, and there are numerous factors to take into account. The results depend mainly on approximations, of parameters such as the transport distance and the utilisation of the energy produced for some of the options. For example the average distance to agricultural land application of sludge was by SEPA approximated to 30 km and by Lundin et al. (2004) approximated to 80 km. This demonstrates how the setting of system parameters and boundaries is critical to the outcome.

2.3.5 General comment on recycling of sludge

In dealing with phosphorus recycling and sludge handling it should be stressed that, logically, there is no one optimal solution, but many different handling options (Campbell, 2000). The local conditions should therefore be thoroughly considered and even then the decision cannot be made entirely on scientific grounds since there are uncertainties in the future (Campbell, 2000). Environmentally it might even be a good idea to have several different ways of handling sewage within a municipality or town; however it will probably be a more costly solution.

2.4 SANITIZATION ASPECTS WHEN APPLYING SEWAGE SLUDGE IN AGRICULTURE

The amounts of non pathogenic microorganisms in faeces are normally high, typically around 10^{11}-10^{13} microorganisms per gram (Schönning & Stenström, 2004). Because it is common that infections occur without clinical symptoms, both sick and apparently healthy individuals can excrete pathogenic organisms (Stenström, 1996).The danger of pathogens in excreta is among other that they can cause enteric infections (Nordin, 2007).

Typical concentrations of pathogens in untreated sewage sludge are listed in Table 2. Large variations can be seen. *Enterococcus faecalis* are non pathogenic bacteria and are usually present in faeces in a somewhat lower concentration than *Escherichia coli* (Stenström, 1996).

Table 2. Typical concentrations of pathogenic and indicator microorganisms in sewage sludge according to different authors (Sahlström et al. (2004) sampled sludge from Swedish sewage treatment plants. Carrington (2001) listed typical values for the US.)

Organism	Carrington, 2001 (CFU/g)	Sahlström et al., 2004 (CFU/g)
Salmonella spp.	10^2-10^3	-*
Escherichia coli	10^6	10^5-10^6
Enterococcus faecalis	-***	10^4
Clostridiae spp.	-***	10^6
Enteroviruses	10^2-10^4	-***
Ascaris lumbricoides	10^2-10^3	-***
Giardia lamblia	10^2-10^3	-***

*No mean values declared.

**No organisms found.

***Not sampled for/not listed.

2.4.1 Viruses

More than 120 types of viruses are excreted in faeces (Schönning & Stenström, 2004). Some commonly mentioned ones in connection to disease transmission are (Schönning & Stenström, 2004; Vinnerås et al., 2008):

- *Astroviruses*
- *Caliciviruses*
- *Enteric adenoviruses*
- *Enteroviruses*
- *Hepatitis A*
- *Hepatitis E*
- *Noroviruses*
- *Parvoviruses*
- *Rotaviruses*

Among these, Svensson (2000) emphasized the importance of *Caliciviruses*, which is the cause of epidemic viral diarrhoea (Nordin, 2007). *Hepatitis A* is also commonly occurring and is in Europe one of the most important food borne pathogens (Schönning & Stenström, 2004; Nordin, 2007). The infection dose, i.e. the number of organism required to cause an infection, for viral disease is in many cases only one virus (Vinnerås et al., 2008). Since viruses cannot reproduce independently their numbers will decrease, or might be relatively stable, when outside of a host (Nordin, 2007).

2.4.2 Bacteria

Salmonella, *Campylobacter* and *Enterohemorrhagic E. coli* all have importance as disease transmitting bacteria found in faeces, both in industrialized and in developing countries (Schönning & Stenström, 2004).

Sahlström et al. (2004) investigated pathogen contents in sludge from eight Swedish sewage treatment plants. The organisms tested for were *Salmonella* spp., *Listeria monocytogenes*, *Campylobacter coli*, *Campylobacter jejuni*, *Escherichia coli O157* and a number of indicator bacteria. Out of these the outstandingly most occurring pathogen was *Salmonella*. It was found that 55 % of the samples were contaminated with salmonella. They also discussed the possibility that salmonella becomes a part of the in-house flora at some sewage treatment plants, but could draw no conclusion about this.

The infection dose for salmonella is anywhere between 20 cells and 10^6 cells, but most often in the higher range (Vinnerås et al., 2008).

2.4.3 Parasites

The parasites found in faeces can be divided into two types: Helminths (or parasitic worms) and Protozoa. Parasites are however of greater concern in developing countries than in industrialized countries (Nordin, 2007).

Ascaris lumbricoides is a helminth that strikes “particularly consumers of uncooked vegetables and fruits grown in or near soil fertilized with sewage” (FDA, 2010-03-11).

2.4.4 Indicator and model organisms

Indicator organisms are used either for studying the inactivation of pathogenic organisms over time or for detecting fecal contamination.

In the study of pathogen inactivation the requirements for a good indicator organism is that it has a high concentration in the material studied. It should also be similar to the pathogens under interest, in the way that it has the same sensitivity to the inactivation technique under investigation (Nordin, 2007). The reason not to directly study the pathogen under interest is that the pathogens sought might be hard to analyse. For example, many viruses are not easily cultivated in a laboratory. The cultivation might also be time consuming and resource demanding.

For detection of faecal contamination the indicator organism chosen is preferably one that is naturally present in faeces, but not pathogenic although in high numbers. The organism’s survival time in the environment should be sufficiently long to be detected after varying time periods and it should not be able to grow in the environment under normal conditions (Stenström, 1996).

Model organisms are used for studying inactivation of pathogens in the case where no appropriate indicator organism is present in sufficiently high concentrations (Nordin, 2007). As opposed to indicator organisms which are originally present in the material, model organisms are separately cultivated and then added. The advantage of using model organisms is the possibility to have a high CFU (colony forming unit) count at the start of an experiment. This gives a clearer picture of how the pathogens decrease over time. The assumption is made that the relative rate of decrease will be the same regardless of the initial concentration.

Organisms used as indicator and model organisms

Organisms commonly used as indicator and model organisms are (Nordin, 2007; Stenström, 1996):

- *Enterococcus faecalis*

  A gram-positive coccus, which is the most commonly occurring bacteria clinically, mainly occurring after causing infections of the urinal tract and in wounds (Bakteriologi, 2010-01-15).

- *Escherichia coli*

  A gram-negative bacteria commonly occurring in the intestines of humans and animals. Although normally present in the intestines, there are some subspecies that produce toxins, e.g. Verotoxine producing *E coli* (*E coli* O157). *E. coli* is often used as an indicator of faecal contamination in soil or water (Stenström, 1996).

- *Salmonella* spp.

  A group of gram-negative rod shaped pathogenic bacteria, which can occur in the intestines of humans and warm- or cold-blooded animals (Stenström, 1996).

2.5 METHODS FOR SANITIZATION OF SEWAGE SLUDGE

The traditional methods for sanitization of sewage sludge, and biowastes in general, can be divided into three categories; heat inactivation, storing and chemical treatment.

There are four alternatives for **heat inactivation** of pathogens, namely pasteurisation, composting, thermophilic anaerobic digestion or incineration. Pasteurisation requires energy to be added for heating the sludge. Different temperatures require different treatment time. Pasteurisation can for example be achieved through keeping the sludge at 70°C for 30 minutes (Stenström, 1996). Both composting and incineration utilize and consume the organic contents in sludge (Vinnerås, 2007). This is negative in the aspect that the soil improving capacity is decreased. Moreover, in incinerated sludge the phosphorus resource is more difficult to utilize. To achieve satisfying sanitisation the temperature has to be sufficiently high throughout the material, composting therefore requires mixing (Vinnerås, 2007) and the temperature must reach at least 55°C for the sanitization to be effective (Stenström, 1996).

Pathogen inactivation through **storing** requires a minimum temperature. In a study by Vinnerås (2007), the inactivation after a period of 50 days was found to be small at a storage temperature of 20°C. This is though largely dependent on the substrate and pathogens studied.

Sanitization through **chemical treatment** can be attained through addition of lime or ash. This will cause the pH to rise. For an effective sanitization to occur pH has to be raised to between 11 and 12.

2.5.1 Urea application for sanitization of sewage sludge

Urea is when applied to soil or sludge degraded into carbon dioxide and ammonia. The enzyme urease acts as a catalyst in the degradation (Equation 2.1).

$$(NH_2)_2CO + H_2O + urease \leftrightarrow CO_2 + 2NH_3 + urease \tag{2.1}$$

Sanitization occurs at sufficiently high urea addition since uncharged ammonia (NH_3) is at high concentration toxic to all organisms. The reason for its toxicity is not well understood, except for in viruses where it passes into the organism and destroys its genome (Vinnerås et al., 2008).

Ammonia is a weak base (Zumdahl, 2005). In solution it is in equilibrium with its conjugate acid, the ammonium ion (NH_4^+) (Equation 2.2). The concentration of uncharged ammonia attained in sludge will be correlated to temperature and pH. For increasing pH and temperature the equilibrium between the two shifts towards the formation of uncharged ammonia. Thus, for achieving an efficient sanitization, a high temperature and pH is favourable. There are other methods of sanitization that function because of the high pH. However, for ammonia sanitization the concentration of ammonia has been showed to be of higher importance than the pH achieved (Allievi et al., 1994; Pecson et al., 2007).

$$NH_3(aq) + H_2O(l) \leftrightarrow NH_4^+(aq) + OH^-(aq) \tag{2.2}$$

Total ammonium nitrogen (TAN) is the term used to describe the sum of uncharged ammonia and ammonium ion concentrations. Emerson et al. (1975) empirically found an equation for estimation of the fraction of ammonium nitrogen that will be present in the form of uncharged ammonia at a given time (Equation 2.3).

$$f = 1/(10^{pK_a - pH} + 1) \tag{2.3}$$

Where f is the sought fraction and pK_a is the dissociation constant of ammonia in logarithmic form. pK_a can be calculated as a function of temperature according to Equation 2.4 (Emerson et al., 1975).

$$pK_a = 0.09018 + 2729.92 / T \tag{2.4}$$

Where T denotes the temperature in degrees Kelvin.

The addition of urea to sewage sludge will enhance its value as a fertilizer. Especially since the composition of sewage sludge is normally such that there is a surplus of phosphorus in comparison with the nitrogen contained and in relation to the amounts needed by plants. For sludge which is to be returned to arable land, the beneficial effect of using urea for sanitization will be two sided: there will be no disease transmission and the need of complementary fertilizers will decrease.

Another advantage of using urea is that it will not be consumed in the process. The ammonia concentrations will prevail, provided that the ventilation losses are small, and as a consequence recontamination during transport will not be an issue as it might be in the case of for example using lime or after composting (Winkler et al., 2009; Vinnerås et al., 2003).

After application of ammonia to sludge it is thus important to cover containers or other storage since uncharged ammonia at relevant temperatures will be in gaseous form and therefore might be lost to the atmosphere if the storage is not properly covered (Vinnerås, 2007).

Application of aqueous ammonia

An alternative to urea addition is to add ammonia in aqueous form. The solubility of ammonia in water is ~35 % (Kemikalieinspektionen:2, 2010-04-05). Aqueous ammonia is according to Ottoson et al. (2008) easier to add on a large scale than is urea. The cost of aqueous ammonia and urea is same per kg of nitrogen (Ottoson et al., 2008) but aqueous ammonia has been found to be more effective than urea when added to the same proportion concerning the total nitrogen concentration (Ottoson et al., 2008; Adamtey et al., 2009). The reason is that aqueous ammonia addition results in higher pH and thereby a larger fraction of the ammonium nitrogen will be present in the form of uncharged ammonia.

The advantage of using urea instead of aqueous ammonia is that while ammonia is toxic and highly volatile urea is a harmless compound, commonly used in for example cosmetics and toothpaste (Nordin, 2007).

2.5.2 Previous research on urea/ammonia treatment

Ammonia treatment has been studied and found effective for several types of biowaste. Recent studies have investigated sanitization of various substrates, such as manure (Ottoson et al. 2008), faeces from source separated dry toilets (Vinnerås et al., 2003; Nordin et al., 2008 & 2009), co-compost (Adamtey et al., 2009), single use biodegradable toilets (Vinnerås et al., 2009) and sewage sludge (Pecson et al., 2007). These studies have shown that the inactivation rates of pathogens increase with increasing ammonia concentration (NH_3 + NH_4^+), and with increasing pH and temperature when ammonia has been added. Decimal reduction (the time required for a 90 % reduction, also called 1 log-unit reduction) of colony forming units (CFU) can be fitted to first order kinetics (Ottoson et al., 2008).

Ottoson et al. (2008) investigated the sanitizing effect when adding urea to farmyard manure. The study was made at two different temperatures, 4ºC and 14ºC. The organisms studied were salmonella phages (*Salmonella* Typhimurium phage type 178) and *Enterococcus faecalis*. The dry substance of the samples was 12 % DM (dry mass). The reduction of salmonella phages was much faster than that of enterococcus. For a urea addition of 2 % (based on wet weight of the material) and a temperature of 14ºC the time for one decimal reduction was 2.0 days for the salmonella phages and 15 days for enterococcus.

Nordin et al. (2009) studied sanitization of feces from source separating dry toilets. The organisms studied were salmonella (*Salmonella* Typhimurium), enterococcus (*Enterococcus faecalis*) and *Salmonella* Typhimurium bacteriophage 28B. The tests were made at temperatures 14, 24 and 34°C at a dry matter content of 17 % and in two different batches. The time required for decimal reduction at 14°C and with addition of 2 % urea (based on the wet weight of the material) was found to be 1.2 or 1.7 days for salmonella and 38 or 46 days for enterococcus (the bacteriophage was not tested for at this temperature). At 24°C and 2 % urea addition decimal reduction was found to take 0.3 or 0.4 days for salmonella, 12 or 14 days for enterococcus and 33 or 37 days for the bacteriophage. The times for decimal reduction were shorter at 34°C than at the other temperatures studied.

2.5.3 Urea concentration required to achieve sanitization of sewage sludge

With the proposed new bylaw (see section 2.2.1) three different bacteria might become regulated when sewage sludge is used in agriculture; salmonella, *E. Coli* and enterococcus.

Salmonella typically has a count of 10^2-10^3 CFU/g in raw sewage sludge (Carrington, 2001). According to the suggested bylaw a sanitization method must, to be approved, eliminate salmonella from sludge. Based on the study by Nordin et al. (2009) a decimal reduction of salmonella at 24°C with 2 % urea addition can be expected to take less than half a day. To achieve full sanitization, a 2 to 3 log-unit decrease (a decrease by a 100 to a 1,000 times) with respect to salmonella would then take less than 1.5 days after an addition of 2 % urea.

The count of *Escherichia coli* (*E. Coli*) in untreated sewage sludge is often higher than the salmonella count, typically around 10^6-10^7 CFU/g (Carrington, 2001; Stenström 1996). The bylaw suggestion sets the limit of *E. coli* in treated sludge at 1000 CFU/g dry matter. This means a reduction of between 3 and 4 log-units, or between 1000 and 10,000 times, would be required. The inactivation rate of *E. coli* is similar to that of *Salmonella* (Nordin et al., 2009), which means the time required for sanitization after addition of 2 % urea would be around 1.5-2 days.

For enterococcus the bylaw suggestion sets the limit in treated sludge at 1000 CFU/g dry matter. This is required for an A-classified sludge. The reduction of enterococcus after ammonia treatment has been shown to take considerably longer time than reduction of salmonella (Ottoson et al., 2008; Nordin et al., 2009). The amounts of enterococcus found in raw sewage sludge are often slightly lower than the amounts of *E. coli* (Stenström, 1996), which means a reduction of between 3 and 4 log-units should be sufficient also for enterococcus. A 4 log-unit reduction of enterococcus will take no longer than 45 days at temperatures above 14°C and urea addition of 2 %, based on the findings by Ottoson et al. (2008) and Nordin et al. (2009).

3 METHODS AND MATERIAL

3.1 SURVEY OF THE SLUDGE HANDLING AT SEWAGE TREATMENT PLANTS IN MÄLARDALEN

A survey of the present handling of sewage sludge in the Mälardalen region was made by submitting a questionnaire to six people representing one or more mid size or large scale sewage treatment plants in the region. The questionnaire can be found in Appendix 1. Four of the six answered the questionnaire and a summary of their answers is found in the results section.

The purpose was to get an idea of how the sludge separation process usually looks, to be able to conduct the lab study in a way that would make its results practically relevant. Questions about destination for the sludge after leaving the plant and if any sanitization method was currently in place were also included since this is of interest in the long run, for future implementation of the method studied. Furthermore the storage of a treated sludge is important since the increased nitrogen contents could cause both losses to the atmosphere and via effluents if the sludge was stored in an improper way.

3.2 DEWATERING OF SLUDGE

3.2.1 Experimental design

Dewatering technique

The goal of the experiments was to test the possibility to utilize decanter centrifuges for the application and mixing in of urea into sewage sludge, while avoiding excessively increased ammonia concentration in the liquid effluent.

A decanter centrifuge is comprised of a rotating drum and inside of the drum there is a rotating screw which rotates independently of the drum. The screw has the function of pushing the sludge towards one end of the drum, towards an outlet, while the rotating drum is what drives the separation; the sludge is separated due to the centrifugal force created and due to the fact that the solids in the sludge have a higher density than the liquids. Polymer is in most cases added to aid the sludge separation. Figure 2 shows a decanter centrifuge in cross section.

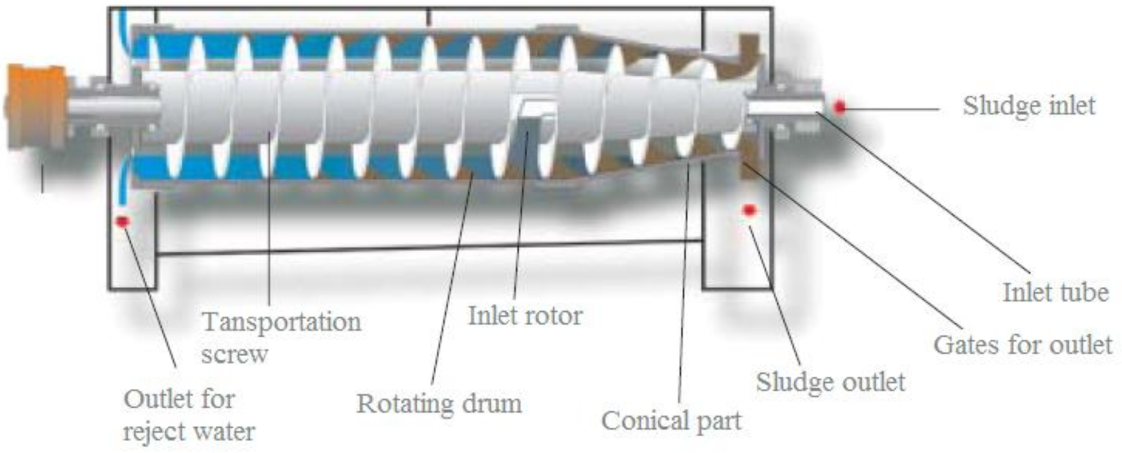

Figure 2. Decanter centrifuge in cross section (Modification based on Alfa Laval, 2010-03-11)

To mimic the dewatering of a decanter centrifuge two alternative approaches were applied; filtration in a Büchner funnel and centrifugation in a small table centrifuge at Kungsängsverket STP (sewage treatment plant), Uppsala. The methods of separation were chosen because of their simplicity and because they enabled working with small sample volumes, which was convenient for making a large number of separations.

Urea additions

Urea additions were made in form of urea pellets of different sizes to investigate if the pellet size would have any effect on the fraction of ammonia ending up in the liquid effluent after dewatering. As 2 % urea addition was enough for eliminating pathogens according to previous studies urea was added both as 2 % of the original wet weight of the sludge and as 0.6 % of the original wet weight, which corresponded to ~2 % of the wet weight of the dewatered sludge. At one occasion urea was added to 1.3 % (based on the initial wet weight of the sludge sample).

The types of urea pellets were: big pellets, small pellets and YARA pellets. Size and weight data for the pellets are found in Table 3. The small and big pellets were prepared from urea powder using a pellet press. The YARA pellets were ordered from YARA and originally intended for fertilization.

Table 3. Pellets used in the experiments

Pellet	Size (cm^3)	Weight (g)
Big	0.11	~0.250
Small	0.23	~0.125
YARA	0.02	~0.006

3.2.2 Treatments

The treatment times were four minutes and one minute. The four minute treatment was chosen after an approximation of the time of centrifugation at Kungsängsverket STP where the sludge was collected. According to the manufacturer of the decanter centrifuges (Noxon DC 40FC) used at Kungsängsverket STP the retention time for sludge in the centrifuges is approximately 4 to 5 minutes (Gustafson, pers. com.). The shorter filtration time of 1 minute was chosen as a comparison to investigate if a shorter time of filtration would cause less ammonia to end up in the liquid effluent. Furthermore, a control was made where sludge was filtered without any addition of urea.

Filtrations were made in a Büchner funnel with diameter 260 mm. The centrifuge used was a Sorvall General Laboratory Centrifuge. The speed during centrifugation was 6000 RPM (revolutions per minute).

Controls were made for one and four minute filtration and for four minute centrifugation. Urea treatments were; one minute filtrations for small pellets and big pellets; four minute filtrations for small pellets, big pellets and YARA pellets; centrifugation for four minutes with big pellets.

All treatments are listed in Table 4. Some treatments were made in several batches and each batch has its own denomination, given in Table 4.

Table 4. Treatments, denominations and number of samples in each batch. Sample denomination has a logical background, F/C stands for filtration/centrifugation, S/B//Y stands for small pellets/big pellets/YARA pellets, the numbers 1 or 4 stands for the treatment time and K means the samples were analyzed by Kjeldahl analysis.

Treatment	Treatment time (min)	Urea addition (%, in "raw" sludge)	Type of urea	Denomination (for each batch)	No. of samples
Filtration	1	-	-	F1	4
Filtration	4	-	-	F4	5
				F4K	3
Filtration	1	2.0	Small pellets	SF1	4
Filtration	1	2.0	Big pellets	BF1i	5
				BF1ii	5
Filtration	4	2.0	Small pellets	SF4i	5
Filtration	4	2.0	Big pellets	BF4i	5
				BF4ii	5
				BF4K	3
Filtration	4	0.6	YARA pellets	YF4	5
Centrifugation	4	-	-	C4	2
Centrifugation	4	1.3	Big pellets	BC4i	5
Centrifugation	4	2.0	Big pellets	BC4ii	5

3.2.3 Preparation of samples

Preparation of a sample was initiated by weighing 40 g of sludge and 6.9 g of polymer solution. Urea was weighed for the samples where it was to be added. The polymer had a TS (total solids) of ~0.15 % and the addition corresponded to 7 kg dry polymer/ton TS in sludge. Since mixing would occur in a centrifuge the samples were mixed by shaking them for ten seconds just before application to the filter or insertion into the centrifuge. The samples filtrated were spread to cover most of the filter surface. After treatment the solid and liquid phases were collected and incubated to allow the added urea to degrade before analysis of ammonia concentrations (NH_3 + NH_4^+). The solid phase was diluted with water so that the degradation of urea would occur faster.

3.2.4 Sludge and polymer used in experiments

All sewage sludge used in the experiments was collected at Kungsängsverket STP, which is located in Uppsala and handles the sewage from 157,600 persons (survey results). Treatment at the plant includes mechanical treatment, chemical treatment (pre-precipitation and secondary precipitation to reduce phosphorus) and biological treatment (with nitrogen reduction). The sludge produced in the sewage treatment is anaerobically digested and then dewatered in decanter centrifuges. Collection of samples was made directly prior to centrifugation, which is the last step in the sludge treatment. TS of the sludge collected was measured to 3.3±0.3 %. Polymer solution, of the type Nordfloc C101, was also collected at Kungsängsverket STP.

3.2.5 TS measurements

All samples were sub-sampled and the sub-samples were dried for determination of TS according to Equation 3.1. The sludge was dried at 100°C for 18±4 h.

$$TS = 100 \cdot \frac{w_d}{w_w} \qquad (3.1)$$

Where w_d is the dry weight of the material and w_w is the wet weight. TS is the total solids in percent.

3.2.6 Measurements of ammonia concentration in sludge and liquid discharge

Ammonia concentration (NH_3 + NH_4^+) in the sludge was measured by Spectroquant® Ammonium Test in an Aquamate spectrophotometer. To allow the urea to decompose the samples were before ammonia measurement incubated at either room temperature or 37°C, for a time period of between 5 and 29 days. A higher temperature required less time for incubation since the urea degraded faster.

The samples F4K and BF4K were analyzed for total organic nitrogen by Kjeldahl analysis. The analysis was made at SLU Kungsängen.

3.2.7 pH measurements

The pH was measured for both liquid effluent and solid phase in connection with ammonia measurements.

3.2.8 Mass balance calculations

Mass balance calculations were made on the basis of the amount of ammonia in samples before and after treatment. The ammonia amount before treatment included both the added urea, which according to stoichiometry was translated into an ammonia amount (1 gram of urea corresponds to 0.568 grams of ammonia) and the ammonia originally present in the sample, which was found through ammonia measurements. The amount of ammonia after treatment was, as described above, also found by ammonia measurements. The amount or mass of ammonia was calculated as the product of concentration and volume for the samples.

The recovery was calculated as the quota of the amount of ammonia in samples after treatment and that before treatment. The losses, below also termed apparent losses, were calculated as the difference between the two.

3.3 SANITIZATION OF SLUDGE

The original idea for the sanitization study was to evaluate of the inactivation of pathogens after dewatering urea amended sludge. Since dewatering urea amended sludge did not give a satisfying result, regarding ammonia concentrations in the liquid discharge, the approach changed and urea was added to already dewatered sludge. The study comprises of a 2 % urea treatment and a control.

Using model organisms made it possible to study the logarithmic decrease in bacteria numbers with time. The organisms added were salmonella (*Salmonella* Typhimurium, isolated from sewage sludge) and enterococcus (*Enterococcus faecalis*, ATCC 29212). These were chosen as they together with *E. Coli* are the organisms that might become regulated by the new bylaw on sludge use in agriculture.

3.3.1 Sludge used in experiments

The sludge used in the sanitization study was collected at Kungsängsverket STP. The collection was made subsequent to dewatering and the sludge had a TS of ~28 %.

3.3.2 Sample preparation

The model organisms, salmonella and enterococcus, were applied to sludge by addition of bacteria solutions at an amount of 1 % of the sludge weight. For the bacteria solutions to mix in homogenously the solutions were first added to a small amount of sludge which was then "diluted". The spiked sludge was divided into two portions. Urea (YARA pellets) was added, at an amount of 2 % (based on weight), to one of the portions. Mixing was done by hand and continued until the pellets appeared to be distributed throughout the material. The other portion of the spiked sludge functioned as a control.

The control was sub-sampled (5 samples) for determination of initial bacteria counts.

For determination of bacteria counts on day 6 both urea amended sludge and control were sub-sampled (5 samples each).

Sub-samples were also taken for determination of bacteria counts on day 20, both urea amended sludge and control was sub-sampled (5 samples each).

Incubation of samples took place at room temperature, in a cabinet holding 20°C. The amount of sludge in each sample was about 20 grams.

3.3.3 Quantification of bacteria amounts

The sludge was analysed by standard methods. The samples were serially diluted in saline solution (0.9 % NaCl). The urea treated samples were diluted in phosphate buffer to stop the sanitizing effect of ammonia before plating. 0.1 ml sample was spread on agar plates, selective of the bacteria studied (xylose lysine deoxycholate with novobiocin for salmonella and Slanetz-Bartley for enterococcus). Enumeration of colony forming units, CFU, was for salmonella made after 21±3 h incubation at 37°C and for enterococcus after 44±4 h incubation at 44°C.

3.3.4 pH measurements

pH of the urea amended sludge and control was measured on day 0 and after incubation (day 6 and day 20). The samples were diluted around fivefold with deionized water and left to stand for 1 h before pH measurement.

3.3.5 Statistical analysis

Student's t-test was used to find weather there was a significant difference in bacteria counts for the respective bacteria studied and for the different occasions of sampling. The test performed was a 2-sample t-test in Minitab 15. With null hypothesis that the two sample/population means are equal the alpha level was set at 0.05 and a p-value below this then suggested that the means were not equal.

4 RESULTS

4.1 SLUDGE HANDLING IN MÄLARDALEN REGION – SURVEY RESULTS

Five sewage treatment plants are represented with results in the survey given in Appendix 2 (one person answered for two plants). The five are: Kungsängsverket (Uppsala), Ekebyverket (Eskilstuna), Bromma and Henriksdal (both in Stockholm) and Himmerfjärdsverket (Grödinge).

4.1.1 General information on the Mälardalen STPs

General information on size of the sewage treatment plants and their yearly production of sludge can be found in Table 5.

Table 5. Information on number of people or pe (person equivalents) connected, mean incoming flow and yearly production of sludge, in dry mass (DM), for the sewage treatment plants included in the survey

Plant	Number connected	Mean flow [m3/day]	Yearly production of sludge [tons DM]
Bromma	301,900 persons	131,000	5,210
Ekebyverket	130,000 pe	45,000	2,200
Henriksdal	721,700 persons	259,000	14,500
Himmerfjärdsverket	280,000 persons + 50,000 pe industrial	100,000	5,500-6,000
Kungsängsverket	157,600 persons	55,100	3,250

The final destination for the sludge differed largely between plants. For one plant, Bromma, the main utilization was in agriculture. Sludge from Himmerfjärdsverket is used both in agriculture, for soil manufacturing, in forestry and for covering landfills. Coverage of landfills and such is what happens with sludge from the remaining plants.

The sludge which is distributed in agriculture is sanitized through long time storing. The sludge produced at Kungsängsverket STP is also stored for a longer period of time, but only awaiting a way of disposal.

Directly after centrifugation the sludge in all cases stay shortly (between 0 and 3 days) at the respective plants and is thereafter transported away for either storage or disposal. The storage at the sewage treatment plants takes place in tanks/silos/sludge towers.

4.1.2 Sludge dewatering at the studied STPs

Four of the five plants in the survey employ centrifuges for sludge dewatering. The exception is Ekebyverket where belt filter presses are used (retention time approx. 1 min). The time of centrifugation was only given for Himmerfjärdsverket, where it was said to be around 10 minutes. For the other three plants there was no direct answer to this question.

Temperature, as previously discussed, affects the rate of sanitization when adding urea. The temperature while centrifuging is for Himmerfjärdsverket and Henriksdal between 20 and 35ºC. At Kungsängsverket STP it is between 25 and 30ºC, and at Bromma between 30 and 35ºC. At all plants polymer is added before centrifugation to attain a quicker and better separation.

The dry mass contents before and after centrifugation are similar in all four plants that employ centrifuges for dewatering. The dry mass ranges between 2.3 % and 3.5 % with a mean of 2.6 % DM before centrifugation. After centrifugation the mean dry mass is 28.2 % with range from 25.0 to 30.3 % DM.

4.2 FILTRATION AND CENTRIFUGATION OF SLUDGE SAMPLES

4.2.1 TS measurements

The mean total solids (TS) before and after treatment are gives in Table 6. The TS for BF1i deviated largely from that of the other batches. A probable reason is an undiscovered air leakage in the filtration device. The average TS of filtered samples is 12.0 %, excluding BF1i, and the centrifuged samples have an average TS of 11.3 %. The average TS of the “raw” sludge was 3.32 %.

Table 6. Mean total solids of solid phase after filtration

Treatment	Batch	TS before treatment (%)	TS after treatment (%)
One minute filtration, no urea addition	F1	3.58	11.3
Four minute filtration, no urea addition	F4	2.98	11.0
One minute filtration, 2 % urea addition, small pellets	SF1	3.59	9.8
One minute filtration, 2 % urea addition, big pellets	BF1i	2.98	5.4
	BF1ii	3.50	14.4
Four minute filtration, 2 % urea addition, small pellets	SF4	3.50	12.6
Four minute filtration, 2 % urea addition, big pellets	BF4i	3.50	12.8
	BF4ii	2.98	12.3
Four minute filtration, 0.6 % urea addition, YARA pellets	YF4	3.11	11.6
Four minute centrifugation, no urea addition	C4	3.36	11.8
Four minute centrifugation, 1.3 % urea addition, big pellets	BC4i	3.36	10.8
Four minute centrifugation, 2 % urea addition, big pellets	BC4ii	3.36	11.2

4.2.2 Ammonia concentration in sludge and liquid discharge after treatment

The results from ammonia analysis made after filtrations of sludge showed that the ammonia concentrations (the sum of NH_3 and NH_4^+) in the liquid effluent were substantially higher for samples with urea addition then for samples where no urea had been added (Table 7). With the exception of samples BF1i the ammonia concentration in the liquid phase was around four to eight times higher for the samples with urea addition compared to those without. The reason for deviation in the results for BF1i is most probably that the TS achieved in this treatment was much lower than that for the other treatments.

Also after centrifugation there was a higher ammonia concentration in the liquid effluent for samples where ammonia had been added. The difference in ammonia concentration was however not as large as for the filtered samples, partly due to the fact that ammonia concentrations in the liquid effluent were higher for samples centrifuged without urea addition then for the corresponding filtered samples.

Table 7 sums up the results for ammonia analysis of the treated sludge.

Table 7. Mean values on ammonia recovery, measured concentrations of ammonia in the solid and liquid effluent after the different treatments and calculated total amounts of ammonia in the solid phase and liquid effluent

Batch	Ammonia recovery (%)	Ammonia loss (g)	Ammonia concentration in solid phase (g/kg)	Ammonia concentration in liquid effluent (g/kg)	Total amount of ammonia in solid phase (g)	Total amount of ammonia in liquid effluent (g)	Fraction of ammonia in solid phase present in the form of uncharged ammonia
F1	50.4	0.030	1.04	0.52	0.012	0.018	0.02
F4	63.6	0.023	1.90	0.58	0.018	0.022	-**
SF1*	60.4	0.199	16.90	1.96	0.243	0.060	0.43
BF1i*	72.6	0.140	16.97	0.51	0.359	0.014	0.06
BF1ii	80.4	0.099	26.66	3.76	0.252	0.153	0.31
SF4	72.9	0.141	20.54	4.58	0.198	0.182	0.45
BF4i	83.7	0.082	26.78	3.72	0.278	0.144	0.35
BF4ii*	64.2	0.181	27.03	1.87	0.247	0.078	0.35
YF4	91.7	0.016	6.65	2.67	0.068	0.106	0.34
C4	137.0	-0.016	1.85	1.27	0.019	0.042	-**
BC4i	77.0	0.078	18.12	1.55	0.209	0.054	-**
BC4ii	64.5	0.170	21.83	1.67	0.250	0.059	0.44

*For SF1and BF1i one of the samples were excluded from the data set. For BF4ii two of the samples were excluded from the data set. In all cases samples were excluded because a deviating ammonia concentration was measured in the liquid effluent. If the samples had not been excluded mean values for ammonia concentration in liquid effluent would have been 13.63/3.3 g/kg, 16.92/0.92 g/kg, and 24.53/1.43, for solid phase/liquid effluent, for SF1, BF1 and BF4ii respectively.
**No pH measurement was made.

4.2.3 Kjeldahl analysis

The results from the Kjeldahl analyses are displayed in Table 8. The nitrogen analysed was, based on stoichiometry, translated into the corresponding amount of ammonia, to have a clearer comparison of the Kjeldahl analysis and the ammonia analyzes made. The result of main interest was the recovery, where analysis of samples where no urea was added gave near 100 % recovery and samples with urea addition gave a recovery of just over 80 %.

Table 8. Results from N-tot (Kjeldahl method) analysis of filtered samples, where mean nitrogen concentration, according to stoichiometry, has been translated into ammonia concentration

Treatment	Batch	Ammonia recovery (%)	Ammonia loss (g)	Ammonia concentr-ation in solid phase (g/kg)	Ammonia concentr-ation in liquid effluent (g/kg)	Total amount of ammonia in solid phase (g)	Total amount of ammonia in liquid effluent (g)
Four minute filtration, no urea addition	F4K	99.4	0.001	8.57	0.69	0.083	0.027
Four minute filtration, 2 % urea addition, big pellets	BF4K	83.1	0.094	32.6	3.69	0.316	0.148

Analysis of raw sludge

Sludge sampled at a single occasion at the Kungsängsverket STP had an ammonia concentration and total organic nitrogen concentration according to Table 9. (The sampling was made 19th of January 2010.)

Table 9. Ammonia concentration and total organic nitrogen concentration (Kjeldahl analysis) in "raw" sludge

Ammonia concentration (g/kg)	Total organic nitrogen, translated into ammonia concentration (g/kg)
1.22	2.80

4.3 SANITIZATION STUDY

A 2-sample t-test showed significant differences, in both time and between control and urea treatment, for all sample series at alpha level 0.05 (except for salmonella at day 6 and day 20 as these values were all zero). The urea treatment can therefore be said to have had effect. Urea additions gave a clear decrease of the bacteria studied, especially for salmonella, which was completely eliminated on day six. Enterococcus were however still present in all of the urea treated samples on day six, and in some of them enterococcus were still present at the final sampling at day 20. Results from CFU counts on day zero, day 6 and day 20, for both urea treatment and control, are showed in Table 10.

Table 10. Mean bacteria counts; initially, after 6 days of incubation and after 20 days of incubation at room temperature. In parenthesis the standard deviation (5 samples for each mean) is given

Bacteria analyzed	Initially (log (CFU/g sample))	6 days incubation (log (CFU/g sample))		20 days incubation (log (CFU /g sample))	
		2 % urea addition	Control	2 % urea addition	Control
Salmonella	6.63 (0.22)	0	7.70 (0.20)	0	6.98 (0.43)
Enterococcus	6.27 (0.18)	5.02 (0.38)	5.59 (0.27)	2.14 (2.11)	4.75 (0.36)

Figures 3 and 4 show the logarithmic decrease with time for salmonella and enterococcus respectively. Curves for both sludge amended with urea and control are shown. As an average over time decimal reduction for salmonella took less than 0.9 days, while the time for decimal reduction of enterococcus was 4.8 days. As it appears from Figure 3 the logarithmic reduction rate of enterococcus was quite constant.

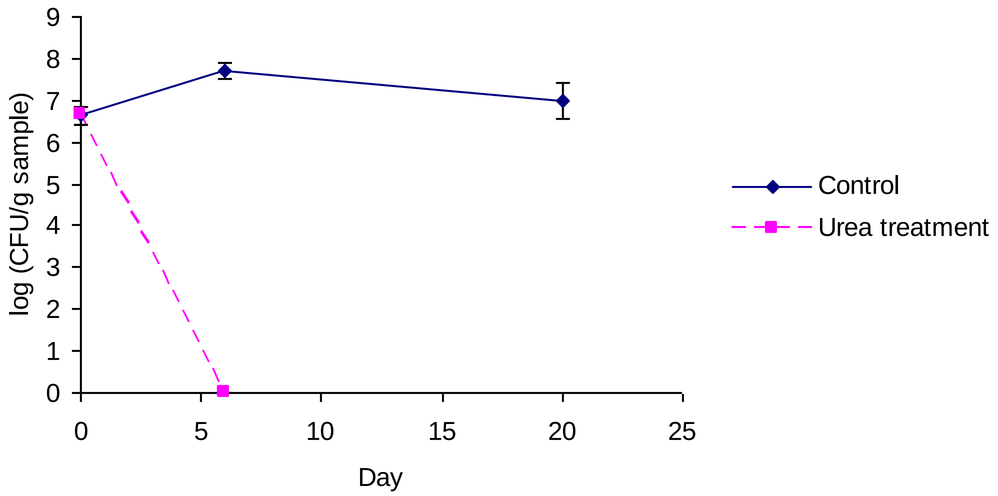

Figure 3. Salmonella in control and urea treated samples, with bars showing the standard deviation (the salmonella count in the urea treated samples was down to zero on day 6, although the log-scale does not allow the expression of zero)

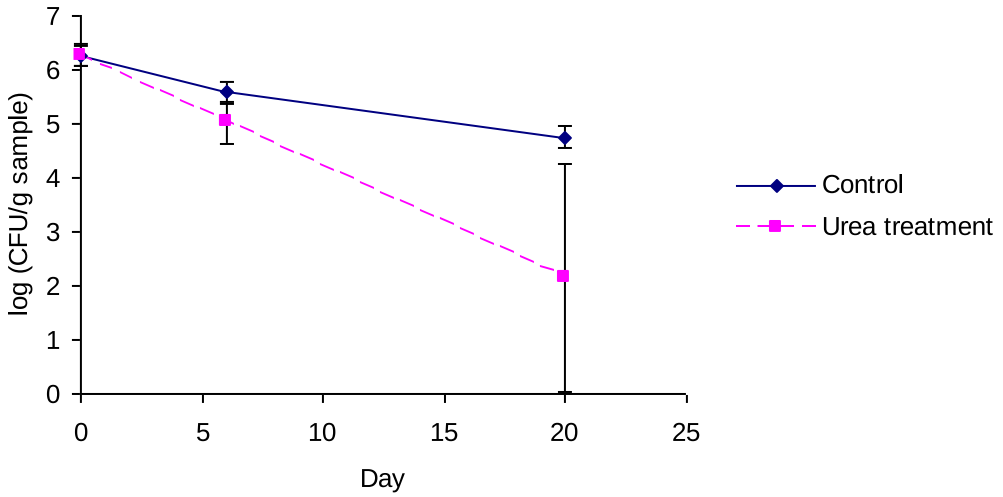

Figure 4. Enterococcus in control and urea treated samples, with bars showing the standard deviation

4.3.1 Measured pH and calculated concentrations of uncharged ammonia

The average pH on day 6 and day 20, for the 5 replicas, is displayed in Table 11.

Table 11. Average pH on day 6 and 20

Day	pH, control	pH, urea amended batches
6	7.54	8.78
20	7.96	8.52

Assuming all added urea (20 g/kg) was degraded and no ammonia was lost from the samples, the concentration of uncharged ammonia (NH_3) could be calculated (based on Equation 2.3 and 2.4). The calculation was made for a temperature of 20°C and the calculated average concentrations of uncharged ammonia are shown in Table 12. The calculation only takes into consideration the added ammonia and not that which was originally present in the samples.

The ammonia concentration in the original sample (the dewatered sludge collected) was measured as 0.711 g/kg, while the pH was measured to 8.49. The concentration of uncharged ammonia in the original sample could be calculated to 0.08 g/kg or 5 mmol/kg.

Table 12. Calculated average concentrations of uncharged ammonia on day 6 and day 20, for samples with urea addition

Day	Uncharged ammonia (g/kg)	Uncharged ammonia (mmol/kg)
6	2.22	130
20	1.33	78

Studying the ammonia concentrations on day 20 more closely, the concentration of uncharged ammonia was calculated to 94.8 and 98.6 mmol/kg for the two samples where enterococcus were completely eliminated and between 61.8 and 70.0 mmol/kg for the samples where enterococcus were not eliminated.

5 ANALYSIS AND CALCULATIONS

The results from sludge dewatering required further calculations to be fully interpreted. The distribution of urea/ammonia between solid phase and liquid effluent after dewatering was the main parameter for evaluation of urea addition at the inlet of decanter centrifuges. The distribution was calculated based on the mass of ammonia in the different fractions after dewatering.

After calculation of the distribution of ammonia, two batches were picked out for further analysis through calculations. The following calculations were made:

- The urea addition required to "raw" sludge to achieve 2 % urea concentration in the dewatered sludge.
- Increase in nitrogen load following from urea application at the inlet of decanter centrifuges, for Kungsängsverket STP.
- The cost for urea addition to dewatered sludge for the sewage treatment plants included in the survey made. For Kungsängsverket STP the approximate cost for urea addition at the inlet of decanter centrifuges was also calculated.

5.1 DISTRIBUTION OF AMMONIA BETWEEN SOLID PHASE AND LIQUID EFFLUENT AFTER TREATMENT

The ammonia analysed was distributed between solid phase and liquid effluent according to Figure 5. For all samples, with exception of BF1i, at least close to a fifth of the ammonia analyzed was found in the liquid phase.

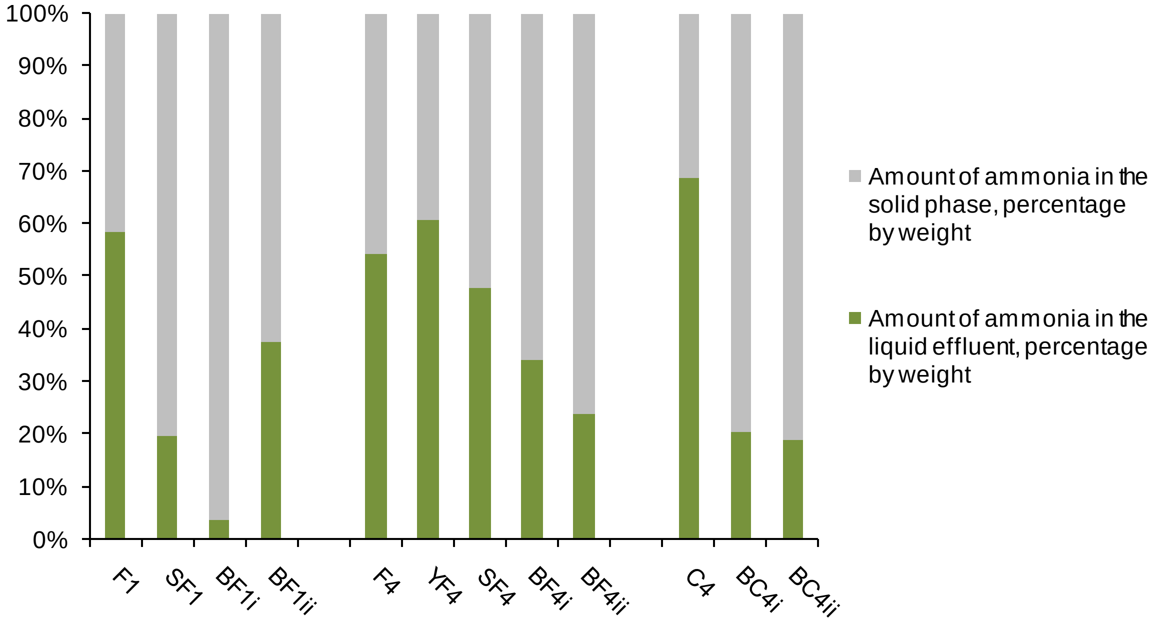

Figure 5. Distribution of totally analyzed ammonia between solid and liquid phase after treatment. Abbreviations used for the different batches: F=filtered, C=centrifuged, Y=YARA pellets, S=small pellets, B=big pellets, 1 or 4 stands for the time of treatment

To reveal the distribution of added ammonia, instead of total ammonia contents in the analyzed samples, the ammonia amount in the control was subtracted from the ammonia amount in samples amended with urea. The resulting distribution of ammonia is shown in Figure 6.

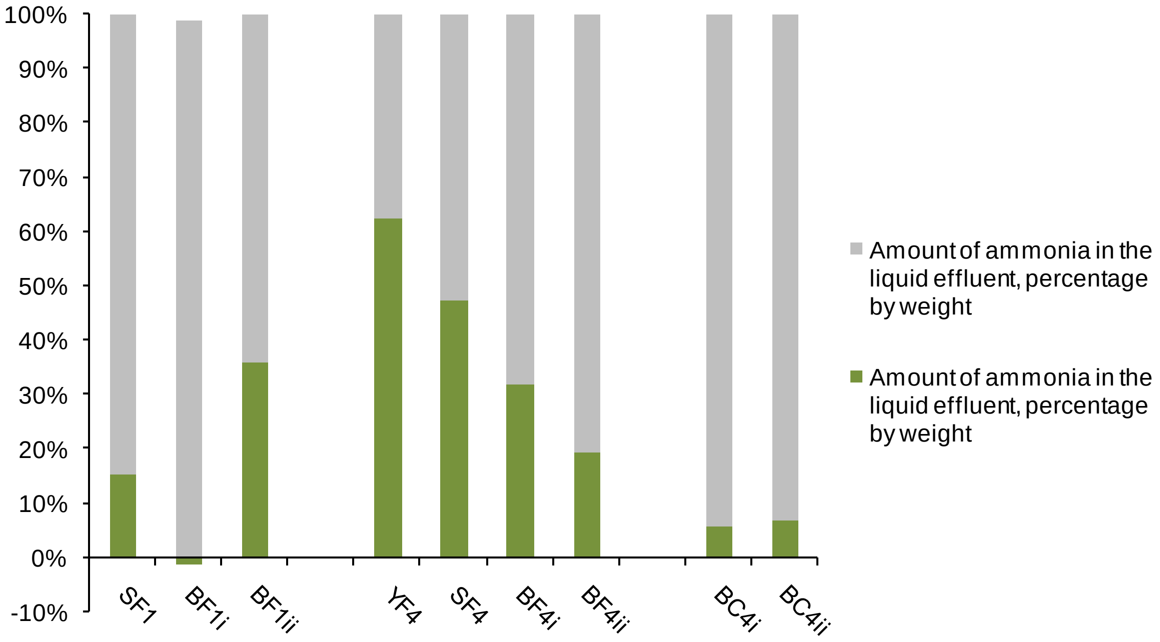

Figure 6. Distribution of "added" ammonia after treatment

5.1.1 Losses of urea/ammonia

Mass balance calculations made it obvious that losses of ammonia occurred somewhere between sample preparation and ammonia analysis. The mass of ammonia was in all cases, except for in batch C4, smaller after treatment than before and the loss was calculated as the difference between the two.

In Figure 7 the losses in the different treatments are compared to the final contents of ammonia in the solid phase and in the liquid effluent. The figure demonstrates how the apparent losses in most cases were high in relation to the amounts of ammonia finally contained in the samples. Furthermore, the apparent losses for urea amended samples were larger than the total ammonia contents (solid phase plus liquid effluent) in the respective control (with exception of batch YF4).

The high losses leads to the conclusion that, subtracting the ammonia contained in control samples, as in Figure 6, does not give a higher accuracy with regards to how the ammonia was really distributed. It could therefore be discussed which of Figure 5 or 6 (Section 5.1) gives the more accurate picture of the ammonia distribution.

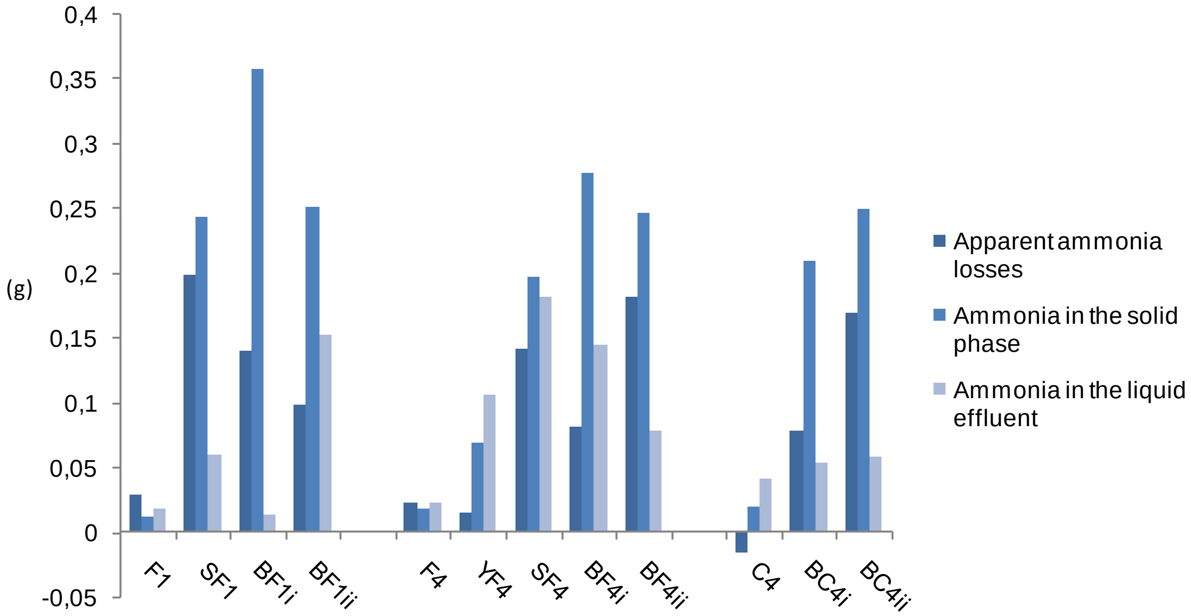

Figure 7. Ammonia losses and analyzed ammonia contents in solid and liquid phase after treatment. Abbreviations used for the different batches: F=filtered, C=centrufiged, Y=YARA pellets, S=small pellets, B=big pellets, 1 or 4 stands for the time of treatment. (Negative loss for batch C4 follows from its more than 100 % recovery.)

5.1.2 Distribution of ammonia expected at higher TS

The TS achieved through filtration and centrifugation was between 5.4 and 12.8 %. This is low compared to the TS achieved in a decanter centrifuge (~28 %). Approximations on how large fraction of the added ammonia would end up in the liquid effluent at a higher TS therefore had to be made.

Modeling of the ammonia fraction ending up in the liquid effluent after treatment, as a function of the TS achieved, the size of the pellets added and the treatment time, was attempted. Data were though too clustered and too few to satisfyingly test the connection of these parameters.

Instead a simple assumption was made for the following calculations, namely that the ammonia concentration in the liquid effluent would be independent of the TS achieved under a constant treatment time. This would likely be the case since the fraction of urea dissolving during treatment should be dependent only on the contact time between urea and sludge.

Assuming a constant concentration of ammonia in the liquid effluent, regardless of the volume of the liquid effluent, makes it possible to calculate the fraction of ammonia in the liquid effluent if higher TS had been achieved. At Kungsängsverket STP the average TS of dewatered sludge is 27.9 %. From the amount of sample in the experiments, 47.0 mg (40 ml sludge, 6.9 ml polymer and 0.8 mg urea), the mass of solid phase if a TS of 27.9 % had been achieved could be calculated. The calculation was made starting off with the concept of TS (Equation 5.1).

$$TS = 100 \cdot \frac{w_d}{w_w} \qquad (5.1)$$

Which can be rewritten as in Equation 5.2.

$$w_d = w_w \cdot \frac{TS}{100} \tag{5.2}$$

As w_d will be constant regardless of how much water is extracted from the sample. The following is true when drying a sample to different degrees, Equation 5.3.

$$w_{w1} \cdot \frac{TS1}{100} = w_{w2} \cdot \frac{TS2}{100} \Leftrightarrow w_{w2} = w_{w1} \cdot \frac{TS1}{TS2} \tag{5.3}$$

Hence, it is possible to calculate the mass of the solid phase at a TS of 27.9 % when having an original TS of 3.32 % in "raw" sludge, and 2.83 % TS after sample preparation. The mass of the solid phase would be 4.8 g (w_{w2} = 47.0·2.83/27.9 = 4.8 g). Correspondingly the weight of the liquid effluent would have been 42.2 grams (47.0-4.8 = 42.2). To calculate the theoretical mass of ammonia in the liquid effluent the ammonia concentration measured was then multiplied with the theoretical volume of effluent, Equation 5.4.

$$\mathrm{M} = \mathrm{C} \cdot \mathrm{V} \tag{5.4}$$

Where M is the mass of ammonia, C is the concentration of ammonia and V is the volume of the sample. In Table 13 the theoretical amount of total ammonia in the solid and liquid phase can be found for this case (27.9 % TS). The theoretical amount of ammonia in the solid phase was found by subtracting the theoretical ammonia amount in the liquid effluent from the true ammonia amount in solid phase plus liquid effluent after filtration.

Table 13. Theoretical ammonia amount in liquid and solid phase (according to Equations 5.3 and 5.4) if TS 27.9 % had been achieved

Samples	Theoretical ammonia amount in liquid phase, if 27.9 % TS (g)	Theoretical ammonia amount in solid phase, if 27.9 % TS (g)
F1	0.008	0.022
SF1	0.220	0.083
BF1i	0.351	0.022
BF1ii	0.247	0.159
F4	0.016	0.024
YF4	0.062	0.113
SF4	0.187	0.193
BF4i	0.265	0.157
BF4ii	0.246	0.079
C4	0.007	0.054
BC4i	0.198	0.066
BC4ii	0.239	0.070

From the values in Table 13 the theoretical fractions of ammonia in liquid and solid phase at a TS of 27.9 % could be calculated, Figure 8. Compared to ammonia fractions in the liquid effluent at a lower TS (see Figure 5) the fraction would become a few percent larger at TS 27.9 %.

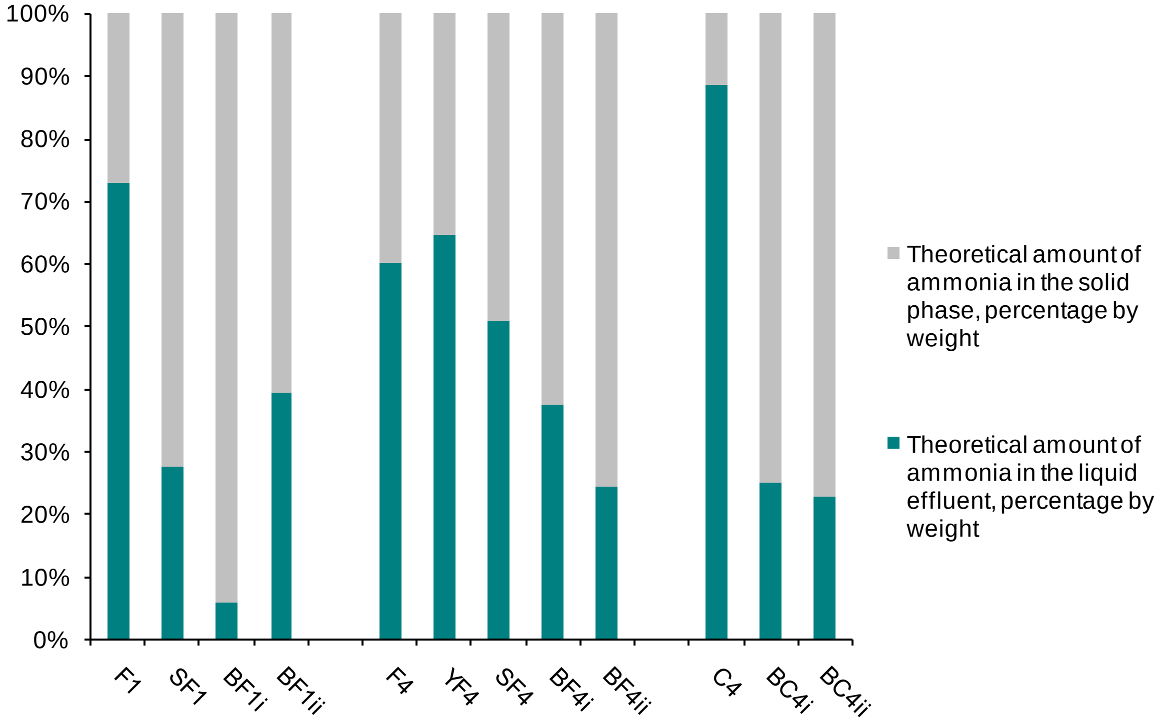

Figure 8. Theoretical distribution of ammonia between solid phase and liquid effluent at TS 27.9 %. Abbreviations used for the different batches: F=filtered, C=centrufiged, Y=YARA pellets, S=small pellets, B=big pellets, 1 or 4 stands for the time of treatment

5.2 SELECTION OF BATCHES FOR FURTHER CALCULATIONS

Before performing calculations on increased nitrogen load and costs for implementing urea addition two batches were chosen for these calculations, one batch among the filtrations and one batch among the centrifugations, namely BF4i and BC4i. The batches were chosen because they had low losses and at the same time a low fraction of ammonia in the liquid effluent compared to other batches (see Figures 7 and 8).

The reason that a low loss was desired was that a high loss would cause larger uncertainties on how the added urea became distributed. A low fraction of ammonia in the liquid effluent would be favorable both concerning sanitization of the dewatered sludge and concerning increase in nitrogen loads.

5.3 REQUIRED UREA ADDITION IN “RAW” SLUDGE TO ACHIEVE 2 % UREA CONCENTRATION IN DEWATERED SLUDGE

What would be the required urea addition to “raw” sludge to achieve 2 % urea in the dewatered sludge? With the assumption that the original mass of sludge was X kg and that the sludge had a TS of 3.5 %, the mass of the dewatered sludge could be calculated according to Equation 5.3 (Section 5.1.1). The mass of the dewatered sludge would be X/8 kg (if in Equation 5.3 TS1 is 3.5 % and TS2 is 27.9 %).

A requirement of 2 % urea concentration in the dewatered sludge corresponds to $0.02 \cdot X/8$ kg of urea. Dividing the mass of urea in the solid phase by the fraction of ammonia in the solid phase gives the total urea addition required in “raw” sludge. Thus, the required addition would be (0.02·X/8)/0.63 kg of urea for BF4i and (0.02·X/8)/0.75 kg for BC4i.

The urea addition to “raw” sludge in percentage could then be calculated by dividing this mass of urea by the mass of the sludge, X. The urea addition in the “raw” sludge would in other words have to be at least (0.02/8)/0.63 or 0.40 % for BF4i and at least (0.02/8)/0.75 or 0.33 % for BC4i.

5.4 INCREASE IN NITROGEN LOAD FOR KUNGSÄNGSVERKET STP

Reject water from the centrifuges is normally led back to the inlet of the STP. It is therefore of interest to know how much the nitrogen load would increase in the reject water due to urea addition at the inlet of decanter centrifuges. The question can be formulated: Would the amount of nitrogen/ammonia in the reject water be significant in relation to the load from incoming water?

The increase in nitrogen load was calculated in mass nitrogen, which could be calculated according to Equation 5.4 (Section 5.1.1). But first, the concentration and volume of the reject water had to be calculated.

The **volume of reject water** at Kungsängsverket STP was calculated from the volume of sludge produced per year, 11,670 m^3, and the TS of this sludge, 27.9 % (survey results). The approximation was made that sludge has the same density as water. The wet weight for the produced amount of sludge at a TS of 3.5 % (the TS of sludge before dewatering at Kungsängsverket STP) could then be found through Equation 5.3 (Section 5.1.1). The wet weight would be 93,000 m^3 (w_{w2}=11,670·27.9/3.5=93,027 m^3). With an incoming sludge volume of 93,000 m^3 the reject water would have a volume of about 81,000 m^3/year (with subtraction of the amount of dewatered sludge, 93,027-11,670=81,357 m^3/year).

The **concentration of ammonia/urea** in the reject water was calculated by dividing the mass of ammonia in the liquid effluent by the volume of the liquid effluent. The volume of the dewatered sludge would, according to Section 5.3, be one eight of the original volume (if TS1 is 3.5 % and TS2 is 27.9 %). This means that the volume of the reject water would be 7/8 of the original volume. The urea addition required, in terms of mass, was above (Section 5.3) calculated to (0.02·X/8)/0.63 kg and (0.02·X/8)/0.75 kg for BF4i and BC4i respectively. If 37 % of the added urea went into the liquid phase for BF4i and 25 % went into the liquid effluent for BC4i (see Figure 8) it would correspond to an amount of (0.02·X/8)/0.63·0.37 kg and (0.02·X/8)/0.75·0.25 kg respectively. The concentration in the liquid effluent would thus be 0.17 % and 0.097 % of urea for BF4i and BC4i respectively (((0.02·X/8)/0.63·0.37 kg)/(7/8·X kg) and ((0.02·X/8)/0.75·0.25 kg)/(7/8·X kg)).

Multiplying the urea concentrations with the volume of reject water estimated gives the urea load. This would be about 140 ton/year and 79 ton/year for BF4i and BC4i respectively, according to Equation 5.5 and 5.6.

$$1.7 \cdot 10^3 \,\mathrm{g/m^3} \cdot 81{,}000\ \mathrm{m^3/year} = 1.4 \cdot 10^8\ g/year = 140\ ton/year \tag{5.5}$$

$$0.97 \cdot 10^3 \,\mathrm{g/m^3} \cdot 81{,}000\ \mathrm{m^3/year} = 7.9 \cdot 10^7\ g/year = 79\ ton/year \tag{5.6}$$

Translated into nitrogen amount, 140 tons of urea corresponds to 66 tons of nitrogen and 79 tons of urea corresponds to 37 tons of nitrogen (1 g urea contains 0.47 g nitrogen). The total volume of incoming water at Kungsängsverket STP is according to the environmental report 20,155,400 m^3/year (Uppsala Kommun, 2008). The average concentration of total nitrogen in incoming water, 43 mg/l, was likewise found in the environmental report. The amount of total nitrogen in the incoming water can be approximated through multiplying the average concentration with the volume of water, which has been done in Equation 5.7.

$$43 \cdot 10^3 mg/m^3 \cdot 20{,}155{,}400\ m^3/year = 8.7 \cdot 10^{11}\ mg/year = 870\ ton/year \qquad (5.7)$$

An increase of 66 or 37 tons compared to an original 870 tons is a quite large increase. In percentage this would be an increase of 8 or 4 % respectively.

5.5 ECONOMIC CALCULATIONS

To evaluate the possibility of applying urea sanitization at the STPs included in the survey made the major costs were approximated.

5.5.1 Cost for urea addition to dewatered sludge

The following calculation only takes into account the cost for purchasing urea. The current price on urea is, according to fertilizerworks.com, 323.73 USD per ton (fertilizerworks, 20100212), which with the current rate of exchange translates to 2363 SEK per ton.

A simple calculation for the cost of 2 % urea addition to the dewatered sludge was made according to Equation 5.8. The cost was calculated on the basis of the yearly production of sludge in tons dry mass (DM) at the STPs, since for some of the plants the production was known only as dry mass. In the calculations the TS of dewatered sludge was assumed to be 28 % for all STPs.

$$C = tons\ sludge \cdot percentage of\ urea \cdot price = DM \cdot \frac{100}{28} \cdot 2\ \% \cdot 2{,}363 \qquad (5.8)$$

Where C is the sought cost. The amount of sludge (tons sludge) is calculated based on Equation 5.3 (Section 5.1.1). Table 14 shows the estimated costs for urea purchases.

Table 14. Costs for 2 % urea addition to dewatered sludge

STP	Production of sludge (tons DM/year)	Yearly cost on plant basis (SEK/year)	Cost per pe (SEK/year)
Bromma	5,210	870,000	2.9
Ekebyverket	2,200	370,000	2.9
Henriksdal	14,500	2,440,000	3.4
Himmerfjärdsverket	5,500-6,000	920,000-1,010,000	2.8-3.1
Kungsängsverket	3,250	540,000	3.5

5.5.2 Cost for urea addition at the inlet of decanter centrifuges

If urea instead was added in a centrifuge the addition would have to be larger to achieve 2 % urea concentration in the dewatered sludge, since part of the added urea would end up in the reject water. The total urea addition required was calculated based on samples BF4i and BC4i Since 63 % and 75 % of the urea added for BF4i and BC4i respectively would end up in the solid phase (Figure 8, Section 5.1.1). The urea addition in relation to the amount of dewatered sludge would for BF4i have to be 3.2 % (2 % / 0.63 %) and for BC4i 2.7 % (2 % / 0.75 %).

The cost for 3 % urea addition, in relation to the amount of dewatered sludge, is displayed in Table 15.

Table 15. Approximated costs for achieving 2 % urea concentration in the dewatered sludge if urea was added at the inlet of decanter centrifuges

STP	Production of sludge (tons DM/year)	Yearly cost on plant basis (SEK/year)	Cost per pe (SEK/year)
Bromma	5,210	1,310,000	4.4
Ekebyverket	2,200	550,000	4.3
Henriksdal	14,500	3,670,000	5.1
Himmerfjärdsverket	5,500-6,000	1,390,000-1,510,000	4.2-4.6
Kungsängsverket	3,250	820,000	5.2

The cost that would arise from the increased nitrogen load in reject water when applying urea at the inlet of decanter centrifuges was calculated for Kungsängsverket STP. A price for increased removal of nitrogen in Uppsala sewage was given by Olshammar et al. (2003). Assuming more carbon source would have to be added in the biological treatment they approximated the cost for nitrogen reduction to 20 SEK/kg N. As calculated in Section 5.4 the yearly nitrogen load could increase by 66 tons. This means the yearly treatment cost for biological nitrogen reduction could be approximated to increase by 1,300,000 SEK when applying urea at the inlet of decanter centrifuges (20 SEK/kg N· 66,000 kg N). For Kungsängsverket STP this cost would be larger than the cost for purchasing urea. The cost for urea sanitization would be 13 SEK/pe/year (5.2 SEK/pe/year + 1,300,000 SEK/year/157600 pe).

6 DISCUSSION

6.1 RECOVERY AND LOSSES OF AMMONIA

To have a clear picture of how the urea added became distributed between solid phase and liquid effluent was important for evaluating if addition of urea at the inlet of decanter centrifuges would be appropriate with regards to the ammonia concentrations in the reject water. Therefore the recovery should ideally have been 100 percent for all samples or in other words, there should have been no losses. This was however not the case.

An incomplete recovery means that the distribution of ammonia between solid phase and liquid effluent after treatment cannot be fully known and that is why samples with low losses were chosen as an estimate of how large fraction of the added ammonia would end up in the liquid effluent when adding urea at the inlet of decanter centrifuges.

The ammonia losses/low recoveries in samples where urea had been added could have had several causes and could be divided into actual losses and apparent losses.

Actual losses

Actual losses could have been caused by vaporization of ammonia during filtration, urea could have been trapped in the filter or ammonia could have been vaporized in connection with analysis. The last process seems the most probable as the ammonia concentration in the samples would have become high at the time of analysis. What also points in this direction is that there often was a distinct smell of ammonia when opening the sample containers (which had been sealed during incubation).

After Kjeldahl analysis it became clear that the actual ammonia losses differed depending on whether urea had been added or not. Since Kjeldahl analysis gave the total organic nitrogen content the losses calculated after Kjeldahl analysis could only include actual losses; nitrogen contained in other forms than ammonia and adsorbed ammonia and urea would also be included in the results (while this could lead to apparent losses in ammonia analysis, see the section below). For the samples where no urea was added the losses seemed to be negligible, with a recovery of near 100 %. The Kjeldahl analysis for samples with urea addition gave a mean recovery of 83.1 %, which was taken as an indication on how large the actual loss after urea addition was; around 17 %. Kjeldahl analysis was only made for samples with 2 % urea addition. The amount of urea added might influence the fraction lost. Since the actual losses for samples without urea addition was negligible, the losses would likely be smaller for a smaller urea addition.

Apparent losses

The apparent losses are assumed to, beyond actual losses, derive from adsorption of ammonia to organic material and from the fact that some of the added urea was still not degraded at the time of analysis and could therefore not be detected in the ammonia analysis. A few percent of the losses can be derived from losses of sample during treatment. In terms of mass the loss of sample was generally a couple of grams.

The ammonia loss in YF4 is much lower than in the other samples where urea has been added. This might be an effect of degradation of nitrogenous organic compounds originally contained in the samples. It could also be explained by the fact that the urea addition was smaller than for the other batches.

Losses in the control

Even though the Kjeldahl analysis showed that the actual losses were negligible for the control, the ammonia analysis did not give a satisfying recovery for neither the filtered nor the centrifuged control. The recovery of ammonia in the samples filtered without urea addition was low, only 50.4 and 63.6 % for F1 and F4 respectively. For the samples which were centrifuged, C4, the ammonia recovery was on the other hand very high. The recovery was well over 100 %, which in reality is impossible.

A reason for the high recovery in the centrifuged samples (batch C4) could be that degradation of organic nitrogen originally contained in the samples occurred. The fact that the centrifuged samples were incubated at a higher temperature than the filtered ones supports this theory, as a higher temperature would favor microbial activity and thereby the degradation of organic material. However, it is harder to find a reason to why the recovery in the filtered control samples should be so far below 100 %. Nordin et al. (2009) mentions adsorption of ammonia to organic material as a possible cause of low recoveries, but it is hard to see a reason to adsorption in this case since all the ammonia was originally present in the samples. Adsorption should therefore occur to the same degree after separation of sludge as before. A hypothesis is that ammonia was adsorbed to the polymer that was added.

According to Kjeldahl analyzes the total organic nitrogen concentration in raw sludge was originally more than double the ammonia concentration analyzed (see Table 9). If translating the nitrogen concentration into ammonia concentration the difference in concentration is 1.58 g/kg. For a mean sample amount of 40 mg this amounts to an additional 0.063 g of ammonia per sample (1.58 g/kg ·0.040 kg), if the organic nitrogen contained in the sample had been degraded into ammonia. Thus, it can be established that there initially was enough organic nitrogen contained in the sludge to account for the surplus in recovery for the samples in batch C4 (Figure 7).

6.2 DISTRIBUTION OF AMMONIA BETWEEN SOLID PHASE AND LIQUID EFFLUENT AFTER TREATMENT

A hypothesis was that a shorter treatment time would cause a smaller fraction of the ammonia to end up in the liquid effluent. This hypothesis can however not be confirmed by the collected data. For small pellets (SF1 and SF4) it would seem as the hypothesis was confirmed, though not for big pellets (BF1ii and BF4i/ii).

Another hypothesis was that larger urea pellets would cause a smaller fraction of the added urea/ammonia to end up in the liquid effluent. This hypothesis seems to be confirmed if observing the results for the 4 minute filtrations, where a trend seems to be present. The same trend is however not seen for the 1 minute filtrations, and the reason is unclear.

From ammonia measurements it could be estimated that at least a fifth of the added urea would end up in the liquid effluent (Figure 5). This indicates that urea addition at the inlet of decanter centrifuges would cause a large loss to the reject water. According to calculations, the increased nitrogen concentration in reject water would lead to an increased nitrogen load of around 4-8 % for Kungsängsverket STP. In the economic calculations the costs for urea addition at the inlet of decanter centrifuges was approximated to more than double the cost of urea addition to dewatered sludge. The reason for this is the expenses for removing the increased nitrogen concentrations in reject water.

6.2.1 Methods of sludge dewatering used in the lab study

It could be questioned whether the methods of sludge dewatering used in the lab study were comparable to dewatering sludge in a decanter centrifuge.

No mixing occurred in neither the filtrations nor the centrifugations, though it would have occurred in a decanter centrifuge. The samples were however shortly shaken before the respective dewatering treatment. It could be expected that if mixing had continued throughout the treatment the fraction of urea that had ended up in the liquid phase could have been larger, because of the mechanical strain that this would have exercised on the urea pellets.

Comparison of the ammonia concentrations in the liquid effluents for control samples and the ammonia concentration in reject water at Kungsängsverket STP, where the sludge was collected, indicated that the methods of sludge dewatering used (filtration and centrifugation) showed resemblance to what would happen in a decanter centrifuge. The ammonia concentration in reject water at Kungsängsverket STP is in average 1.03 g/l, according to six spot-checks made in the course of 2009 (Olsson, pers. Com.). The spot-checks were made on N-tot concentration in the reject water, measured by the Kjeldahl method. During 2009 no analyzes were made on the ammonia concentration in the reject water, but measurements of ammonia and N-tot concentrations from 2006 show that the mean difference between the two was less than 5 %. Therefore the ammonia concentration in the reject water could be approximated by the N-tot concentration. Though the ammonia concentration in the liquid effluent after filtration, 0.52/0.58 g/kg, and centrifugation, 1.27 g/kg, differed from the ammonia concentration found in reject water at Kungsängsverket STP, 1.03 g/l, the concentrations could be regarded to be quite similar if taking into consideration the recoveries calculated for the control samples.

In summary, since no mixing occured during dewatering, the methods of sludge dewatering used in this study might have underestimated the amount of urea that would end up in the reject water if urea was added at the inlet of decanter centrifuges.

6.3 FULFILLMENT OF SANITATION REQUIREMENTS

As previously discussed (section 2.5.3) the requirement on logarithmic reduction should be at least a 3 log-unit reduction for salmonella and an at least 4 log-unit reduction for enterococcus. The said reductions were in this study achieved in less than six days for salmonella and, regarding the mean reduction, in less than 20 days for enterococcus. Thus, the results of the sanitization study indicate that an addition of 2 % urea, a temperature of 20ºC and a treatment time of 20 days would be sufficient for reaching the standards of A-classification of sludge according to the proposed bylaw. The treated sludge could also meet the demands of REVAQ.

The reduction of salmonella was very quick, with a time of decimal reduction shorter than 0.9 days. This finding agrees with previous findings on reduction rate of salmonella. This study was though performed for sludge of higher TS than that in the studies in comparison (Nordin et al., 2009; Ottoson et al., 2008). Comparing the observed reduction to previous observations on decimal reduction for enterococcus, 12-14 days at 24ºC and 15 days at 14ºC (Nordin et al., 2009; Ottoson et al., 2008), the decimal reduction for enterococcus was in this study quite quick, 4.8 days at 20 ºC. A theory expressed by Ottoson et al. (2008) is that the reduction rate would be lower in material of higher TS because of the larger buffering capacity of the material, which would cause ammonia to rather prevail in the form of ammonium ions than in the form of uncharged ammonia, which is the disinfecting agent. The study at hand does not confirm this theory. A hypothesis is that the sanitization in materials of higher TS could instead be faster because more ammonia would be present in gaseous form and thereby have a larger contact area.

In the results for enterococcus large uncertainties exist because of a large variation on day 20, Table 10. The standard deviation in CFU count for enterococcus after urea treatment was large, i.e. some samples had a high CFU count after 20 days while some where completely free of enterococcus. It is believed that a more efficient reduction of enterococcus would have occurred if there had been a better homogenization after adding the urea. Since the urea addition took place before sub-sampling the urea concentration probably differed somewhat between samples. The concentrations of uncharged ammonia in the samples were estimated based on pH measurements and on urea addition (see Table 12 for mean concentrations), assuming that all samples contained 2 % urea after preparation. The estimation indicated that the concentrations of uncharged ammonia were higher in those samples where enterococcus were eliminated on day 20, since the pH (see Table 11 for average pH) was higher in those samples. However, the high pH might have been induced by a higher concentration of urea in the mentioned samples, which would support the theory that the urea added did not become evenly distributed between samples.

At higher treatment temperature the treatment time could be decreased. It might be possible that merely the temperature of the sludge after dewatering is enough to achieve a treatment time shorter than that found in this study, provided that this temperature can be kept. At STPs in the Mälardalen region the temperature in decanter centrifuges was between 20 and 35°C. With an isolated tank at collection, it could be possible to conserve the heat. A higher temperature could also be achieved through heating. This would though decrease the simplicity of the method.

6.4 INCREASE OF NITROGEN LOAD

Calculations showed that addition of urea at the inlet of a decanter centrifuge would most probably cause a significantly increased nitrogen load for Kungsängsverket STP. This would probably also be the result if adding urea at the inlet of decanter centrifuges at other sewage treatment plants since the conditions are similar; similar temperatures of the sludge when it enters the decanter centrifuges, similar TS of both "raw" sludge and dewatered sludge and polymer is added before centrifugation.

6.5 SLUDGE HANDLING

Urea treatment for sanitizing sewage sludge is a very logical choice for sludge that is to be utilized in agriculture. It is a less logical choice for sludge that not is to be applied to productive land since the nitrogen resource added would be wasted.

When using urea treatment for sanitization, including farmers/land owners in the planning process will be of great importance. It is important to assure that the increased nitrogen content in the sludge is utilized. The farmers' interest in applying sludge for fertilization should increase with the increased fertilizer value of sludge. At the same time it might be appropriate to look into REVAQ certification to ensure that there is trust in the relationship between STPs and farmers and credibility in relation to consumers. When using urea treatment for sanitization of sludge the sludge should have a good enough quality to enable agricultural application, meaning that it should at least live up to the statutory demands.

The logistics around sludge handling is thought to be of great importance since the sludge will be of much higher nitrogen content after urea addition. The storage of urea treated sludge is of great weight. Since the storage time at the STP or in other storage facilities could be as short as 15 days for this treatment not only STPs would be responsible for storing the sludge in an appropriate way, a large responsibility would also lie on the receiver of sludge to handle it in a way that minimizes ammonia leakage and volatilization. Most important would be to cover the sludge storage with a material that hinders volatilization. Ammonia losses after application of sewage sludge should be avoided through incorporating the sludge into the soil as soon as possible. Recommendations from REVAQ are immediate incorporation into the soil when applying sludge in agriculture, and this could help minimize losses. What is positive from a sludge handling perspective is that the risks of recontamination are minimized when the ammonia losses are small.

A possible challenge is that sludge of high nitrogen content preferably should be spread during spring time, which is already a busy time for farmers. STPs might also have to make agreements with sludge receivers which include financing for storage and spreading of sludge, to guarantee that the ammonia losses are minimized.

The survey results showed that there is a potential market for the method in the Mälardalen region. The two plants that produced sludge that was used in agriculture, currently used storing for sanitizing sludge. With urea treatment the sanitization could be sped up and costs for storage could thereby be decreased. The three remaining plants need to find an appropriate sanitization solution with the upcoming law on sanitization. After further studies urea treatment could probably become one viable option.

6.6 ECONOMY

The economic calculations were made to enable evaluation of the economic conditions for implementation of urea sanitization of sewage sludge. The estimated costs for adding urea to dewatered sludge only took into account the costs for the urea added, which also might vary somewhat depending on the safety margins required. A calculation of the costs when urea is added at the inlet of decanter centrifuges was made for Kungsängsverket STP, where both the cost for the added urea and the costs for removing the increased nitrogen load were included. In addition to the costs for purchasing urea, the costs for urea treatment might include; modification of sludge storage facilities, transportation of sludge and in the case where urea is added to dewatered sludge - a mechanism for mixing urea into the sludge. The calculations made here are thus approximations.

According to an approximation by Balmér (Naturvårdsverket, 2009) the yearly cost for sanitization would be between 10 and 26 SEK per person when using pasteurization or thermophilic anaerobic digestion at large plants and liming at smaller plants. This cost is around 3 to 8 times the approximated cost for urea addition to dewatered sludge (see Table 14). This indicates that there would be a large investment margin, for improved storage etc., if urea treatment was applied.

Based on the economic calculations, urea addition at the inlet of decanter centrifuges could also be a cost efficient sanitization solution compared to the figures calculated by Balmér (Naturvårdsverket, 2009). It would though probably be cheaper and certainly more appropriate from an environmental perspective to find a way of applying urea to already dewatered sludge.

Instead of adding urea, aqueous ammonia could be added to dewatered sludge. Accounting for the amount of nitrogen the price of aqueous ammonia is equal to the price of urea. It has been shown that aqueous ammonia can sanitize biowastes at lower ammonia concentration than urea can. It might thus be possible to further lower the costs by adding aqueous ammonia instead of urea, since aqueous ammonia could be added at a lower dose. However, as urea “contains” about 50 % ammonia and aqueous ammonia can have a maximum concentration of 35 % the sludge weight would be slightly more affected by addition of aqueous ammonia than by addition of urea. The weight of sludge in turn affects the transport costs involved. Based on the costs for urea and ammonia respectively and the increase in transport costs that would follow, it could be calculated which option would be the most cost efficient for a specific STP.

6.7 FURTHER STUDIES

For the implementation of the method to be possible it is crucial to find an appropriate technique of mixing urea into dewatered sludge. Mixing urea into decanter centrifuges has in this study been found inappropriate with regards to the increased costs and environmental drawbacks. When mixing urea into dewatered sludge it could be assumed that the most appropriate form of urea to add would be urea powder. At most STPs the sludge is ejected from the centrifuges and falls directly into a large container placed just below the centrifuges. This can be seen as a problem since the outlet of the centrifuges is normally the last step in the sludge treatment. An easy solution would though be to construct a simple mixing mechanism which the dewatered sludge would fall through. According to the survey dewatered sludge had similar TS at all studied STPs, which means that once an appropriate solution is found for mixing urea into dewatered sludge, it should be possible to use it as a standard method.

An idea for mixing urea into decanter centrifuges was to find coated urea pellets, which would not dissolve as quickly as pure urea pellets, to decrease the ammonia concentrations in the reject water. The YARA pellets tested in this study had a hardened surface, which was designed to make them dissolve slower, but no positive effect was seen. On the contrary they caused a larger ammonia concentration in the liquid effluent, probably because they were smaller than the other pellets applied.

Further studies should be made to confirm the findings on reduction for enterococcus since the sanitization results in this study showed a big variation (Table 10, Figure 4). It would also be of interest to study the effects of different ammonia addition and different treatment temperatures at a sludge TS of around 28 %. The sanitization effect of urea and its dependence on temperature and concentration has been studied before, but not at as high TS.

Currently the absolutely largest contributor to anthropogenic ammonia emissions, 83 %, is the agricultural sector and around 30 % of the total nitrogen contained in manure is lost through ammonia losses (Statistiska Centralbyrån et al., 2007). Therefore an important aspect to study is the losses of ammonia that might occur while handling, storing and spreading urea/ammonia treated sludge. As fertilization is bound to seasons, storage times for sludge that is to be used in agriculture could become long even though the treatment time for achieving sanitization is short. On the other hand this can be used for decreasing the urea dose in relation to the storage time available.

Green house gas emissions could potentially decrease because of the stabilization of sludge achieved through urea addition. It would therefore be of interest to study e.g. methane and nitrous oxide emissions form urea amended sludge.

It would also be interesting to investigate a farmer’s view on the benefits of applying urea treated sludge for fertilization, with its enhanced fertilizer value. Would it make farmers in general keener on accepting sludge from STPs?

7 CONCLUSIONS

The study showed that urea, in the form of pellets of three different sizes (0.02/0.11/0.23 cm^3), is dissolved too quickly to be appropriate to add at the inlet of a decanter centrifuge. The result would most likely be an increased amount of ammonia for the sewage treatment plant to handle, as the effluent from sludge dewatering is re-circulated to the inlet of the plant. An increased nitrogen load would lead to increased costs for reducing the nitrogen concentrations in the sewage, through increased aeration and in some cases increased addition of carbon source in the biological treatment. The increased energy consumption that would follow would also be negative from an environmental perspective.

The conclusions from this study are in brief:

- Urea addition at the inlet of decanter centrifuges would probably cause a significantly increased (4-8 %) nitrogen load for STPs and can therefore not be seen as appropriate.
- Urea addition has the possibility of effectively sanitizing dewatered sewage sludge.
- An appropriate way of mixing urea into dewatered sewage sludge needs to be found.
- Further studies should be made to confirm the time required for sanitization of sewage sludge using urea.
- The method is low cost and at the same time enhances the fertilizer value of sewage sludge. It therefore has a large potential of competing with more established methods of sanitization.

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Appendix 1 – Survey questionnaire

Undersökning av slamhantering hos reningsverk i Mälardalen

1. Vilka reningsverk med ett medelflöde på mer än 5 000 kubikmeter/dygn ansvarar organisationen för?

För detta/dessa reningsverk:

2. Verkets namn.
3. Hur många personer är anslutna, och hur många personekvivalenter är verket dimensionerat för?
4. Vilket medelflöde renas i verket?
5. Hur mycket avloppsslam produceras?
6. Hur hanteras slam i verket? (förtjockning, centrifugering osv.)
7. Vilka tillsatser (så som polymer) görs i samband med slamhanteringen, när och vilka volymer?
8. Hur lång är uppehållstiden för slam i slamcentrifugerna?
9. Vilken temperatur håller slammet under centrifugeringen?
10. Vilken TS-halt har inkommande och utgående slam vid centrifugerna?
11. Hur ser slamhanteringskedjan ut efter centrifugeringen?
    a. Vid uppsamling, direkt efter centrifug (ge gärna ungefärlig uppehållstid och en uppskattning för temperaturen)
    b. Sker lagring? (ge gärna ungefärlig uppehållstid och en uppskattning för temperaturen samt ange om lagring sker utomhus, finns tak/täckning etc.)
    c. Distribution?
12. På vilket sätt hygieniseras slammet?
13. Hur används slammet efter att det lämnat reningsverket? (Om det finns, gärna med siffror på hur stor andel som används för jordförbättring/i jordbruk/deponitäckning osv.)

Appendix 2 – Answers to the survey

The answers to the survey are here presented in alphabetical order according to the name of the sewage treatment plant.

Bromma

2. *Antal anslutna 2008: 301900 p. Finns ingen dimensionerat pe. Verket har byggts ut i omgångar sedan 1934. Kraven för verksamheten är gemensamma för Bromma och Henriksdal och baseras på utsläppt mängd P och N.*
3. *Medelflöde 2008: 131 000 m3/d*
4. *17 200 ton slam med TS 30,3%*
5. *Primärslam pumpas direkt till rötkammare. Överskottslam förtjockas i centrifuger. Slammet rötas och avvattnas i centrifuger.*
6. Polymer vid avvattning 50 ton under 2008
7. *Kapaciteten är 30 m3/h*
8. *30-35 grader.*
9. *Ca 2,3% in och 30,3% ut*
10. -
    a. -
    b. *Det avvattnade slammet samlas upp i en på 600 m3.*
    c. *2008: Slammet hämtas med lastbil i täckta containrar. Containrarna körs till Jordbro ca 20 km där de lastas på järnväg för transport till Gällivare. Järnvägstransporten är returfrakt för koppar som transporteras i andra riktningen.*
    2009: Slammet hämtas med lastbil och körs till ett mellanlager, ca 20km från Bromma, där det lagras ca 6 månader innan det sprids på jordbruk.
11. *Vid jordbruksspridning lagras slammet ca 6 månader.*
12. *Slammet användes 2008 vid återställning av sandmagasin och gråbergsupplag. Slammet ersätter andra resurser som morän och fosfor.*
    Under 2009 har huvuddelen av slammet gått till jordbruk.

Ekebyverket

1. Ekebyverket
2. 80 000pe, 130 000pe
3. 45 000m3/d
4. Ca 2200ton TS
5. Silbandspressar
6. Polymer ca 8 kg/kgTS
7. Ca 1min
8. 10-20C
9. 2,5% resp. 24%
10. -
    a. -
    b. Lagras i ett slamtorn uppehållstid ca 1dygn. Temp ca 10C inomhus
    c. Slammet transporteras per bil 1mil till sluttäckning av deponi
11. Ingen hyg.
12. Allt går till deponitäckning

Henriksdal

2. *Antal anslutna 2008: 721700 p. Finns ingen dimensionerat pe. Verket har byggts ut i omgångar sedan 1942. Kraven för verksamheten är gemensamma för Bromma och Henriksdal och baseras på utsläppt mängd P och N.*
3. *Medelflöde 2008: 259 000 m3/d*
4. *52 100 ton slam med TS 27,9%*
5. *Primärslam pumpas direkt till rötkammare. Överskottslam förtjockas i centrifuger. Slammet rötas och avvattnas i centrifuger.*
6. Polymer vid avvattning 100 ton under 2008
7. *Kapaciteten är 30 m3/h*
8. *20-35 grader beroende på om värmeväxling sker.*
9. *Ca 2,3% in och 27,9% ut*
10. -
    a. -
    b. *Det avvattnade slammet samlas upp i en på 800 m3.*
    c. *Slammet hämtas med lastbil i täckta containrar. Containrarna körs till Jordbro ca 12 km där de lastas på järnväg för transport till Gällivare. Järnvägstransporten är returfrakt för koppar som transporteras i andra riktningen.*
11. *Ingen hygienisering sker*
12. *Slammet används vid återställning av sandmagasin och gråbergsupplag. Slammet ersätter andra resurser som morän och fosfor.*

Himmerfjärdsverket

1. Himmerfjärdsverket
2. Till verket är f.n. 280 000 personer anslutna och den industriella anslutningen motsvarar ca 50 000 pe. Verket har tillstånd för behandling av 130 000 m3 avloppsvatten/dygn (som årsmedelvärde). Dagens anslutning ger upphov till 100 000 m3 avloppsvatten per dygn.
3. 100 000 m3 per dygn.
4. 5500 till 6000 ton TS
5. Slammet från reningsprocesserna förtjockas och rötas för att sedan avvattnas i centrifuger. En tredjedel av det avvattnade slammet torkas till 95 % TS, i första hand för gödsling i skogsbruk.
6. Vid centrifugering tillsätts polymerer. Förbrukningen uppgår till 35-50 ton per år.
7. Uppehållstiden är ca 10 minuter vid centrifugering och 4 timmar vid torkning.
8. Temperaturen vid centrifugering är 20-35 C. Vid torkningsprocessen uppgår temperaturen till 80-90 C.
9. Inkommande och utgående TS-halt under centrifugering är 2,6 % och 25-28 %.
10. Centrifugerat slam samlas upp i silo under dagen. Silorna töms kontinuerligt och slam avsett för jordbruket mellanlagars av entreprenör.
11. Slammet hygieniseras genom långtidslagring.
12. Fördelningen av slamanvändningen uppgår till 35 % lantbruksspridning (avvattnat slam), 25 % deponiåterställning (avvattnat slam), 5 % jordtillverkning (avvattnat slam) och 30 % skogsbruksspridning (torkat slam).

Kungsängsverket

2. *157 600 pe anslutna 143 000 i föroreningsbelastning vad gäller BOD*$_7$*. Verket är dim. för 200 000 pe*
3. *Inget medelflöde 2008 55 100 m*^{3}*/dygn genom Kungsängsverket. Alltså ca 2300 m*^{3}*/h*

4. *11 670 ton torrsubstans 27,9 %*
5. *Förtjockas gravimetriskt, förtjockning av bioslam, delar av primärslam och kemslam med en slask. Inpumning av ca 5 % slam till mesofilrötkammare. Rötat slam centrifugeras.*
6. *En polymer tillsätts före slaskarna 13,2 ton/år (2008) och en typ av polymer före centrifugerna 17,2 ton/år (2008)*
7. *Svårt att svara på. Beror helt på moment som körs osv samt flöde genom centrifugen*
8. *Ca 25-30 ºC*
9. *In ca 3-3,5 % ut se 4*
10. -
    - a. Slamsilos på vardera 100 ton. *Ungefärlig uppehållstid kan variera mellan 0-3 dygn. En silos fylls och skiftar sedan. Tiden mellan silon är full och lastbil kommer och hämtar kan variera mellan 0-2 dygn.*
    - b. *Mellanlagring sker ute på slamcellerna på Hovgårdens avfallsanläggning Mellanlagring sker enligt uppgift i 3 år.*
    - c. *Används för tipptäckning*
11. *Idag ingen hygienisering*
12. *100 % till deponitäckning. Före 1999 användes det till jordbruksmark men på grund av höga kopparhalter används det nu till deponitäckning*