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Evaluation of the step-feed biological nitrogen removal process at Kungsängen wastewater treatment plant

Utvärdering av kväverening med stegbeskickning vid Kungsängsverket

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Linda Åmand

Abstract

Evaluation of the step-feed biological nitrogen removal process at Kungsängen wastewater treatment plant

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The step-feed biological nitrogen removal process at Kungsängen wastewater treatment plant in Uppsala has been in operation since 1999. This process configuration in the activated sludge process has since its introduction into full-scale operation in the early 1990's proved to offer many benefits compared to an ordinary predenitrification process. The implemented process configuration at block C is quite unique in the Nordic countries and considering the high nitrogen load it has proven to manage and the robust operation it is of interest to further evaluate the nitrogen removal.

The evaluation in this study comprised of (i) static process calculations, (ii) on-site sampling to create a mass balance analysis and nitrogen profile, (iii) principal component analysis and (iv) modelling with ASM1 in the JAVA Activated Sludge Simulator (JASS). The results were compared with literature on nitrogen removal and step-feed, data from the process dimensioning and plants with predenitrification in Sweden.

The average nitrogen removal in block C was 78 % during the evaluation period with nitrification and denitrification rates of 1.7 and 2.2 mg N/g VSS, h respectively at 15 °C. Nitrification is suffering from alkalinity deficiency after the introduction of new water plants since pH in secondary effluent reaches as low as 6.5. Denitrification is carbon limited due to the low BOD to N ratio in primary effluent of about 2. The plant is recommended to continue with by-pass of influent wastewater and hydrolysis in primary clarification since it has a positive effect on denitrification. Further optimization of the step-feed process or increase of the hydrolysis in primary clarification might be needed to improve the denitrification potential.

Key words: Activated sludge, biological nitrogen removal, step-feed, Kungsängsverket, principal component analysis, JASS

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Referat

Utvärdering av kväverening med stegbeskickning vid Kungsängsverket

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Kaskadkvävereningen på block C på Kungsängserket i Uppsala har varit i drift sedan 1999. Processkonfigurationen visade sig ha flera fördelar jämfört med vanligt fördenitrifikation när den introducerades i full skala i början på 1990-talet. I Norden är processlösningen på Kungsängsverket så gott som unik och eftersom blocket har visat sig klara av höga kvävebelastningar och visat på robust drift är det av intresse att utvärdera driften mer ingående.

Utvärderingen består av (i) statiska processberäkningar (ii), provtagning för massbalans och kväveprofil, (iii) principalkomponentanalys och (iv) modellering med ASM1 i JAVA Activated Sludge Simulator. Resultaten jämfördes med litteratur kring kväverening och kaskadkväverening, data från processdimensioneringen och verk med fördenitrifikation i Sverige.

Reningseffekten av kväve uppgår i genomsnitt till 78 % under utvärderingsperioden med nitrifikations- och denitrifikationshastigheter på 1,7 och 2,2 mg N/g VSS, h. Nitrifikationen lider av alkalinitetsbrist efter att två nya vattenverk satts i drift i Uppsala vilket medfört att pH värdet efter kvävereningen är så lågt som 6.5. Denitrifikationen är kolbegränsad med en BOD/N kvot på 2 efter försedimenteringen. Verket rekommenderas fortsätta med by-pass flödet förbi förreningen och primärslamhydrolys för att gynna denitrifikationen. En mer noggrann optimering av kaskadkvävereningen alternativt utökad primärslamhydrolys kan behövas för att förbättra denitrifikationpotentialen.

Nyckelord: Aktivtslam processen, biologisk kväverening, kaskadkväverening, Kungsängsverket, principalkomponentanalys, JASS

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Svensk sammanfattning

Utvärdering av kväverening med stegbeskickning vid Kungsängsverket

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Kungsängsverket byggdes på 1940- och 50-talen i Uppsalas sydöstra delar. Verket är idag dimensionerat för 200 000 pe (personekvivalenter, baserat på 70 g BOD/person, d). Reningsverket använder sig av aktivslamprocessen, en process där mikroorganismer hålls i rörelse i luftningsbassänger och renar vattnet från kväve och organiskt material. Mikroorganismerna avskiljs från det renade vattnet med sedimentering. Block C, den senast påbyggda delen som driftsattes 1999, är beräknat att ta 58 % av det totala inflödet till verket. Blocket använder sig av kaskadkväverening vilket innebär att vatten stegbeskickas till flera punkter längs bassängerna. På så vis uppstår en koncentrationsgradient av slammet, och en större mängd slam kan hållas i bassängerna utan att öka belastningen på sedimenteringen. Block C har tre kaskader med en anoxisk och en aerob zon i varje. En tredjedel av inflödet går in till varje kaskad.

Två processer sker i reningsverkets biobassänger vid kväverening: nitrifikation och denitrifikation. Nitrifikationsbakterierna är strikt aeroba och omvandlar ammonium i inflödet till nitrat. De tillväxer relativt långsamt, vilket innebär att det krävs en hög slamålder i bassängerna och de är känsliga för låga pH-värden och annan inhibition av t.ex. metaller. Denitrifierarna är fakultativt anaeroba vilket innebär att de i syrefria miljöer kan använda sig av andra oxidationsmedel än syre. I de zoner i verket som endast har omrörning använder de nitrat för att oxidera organiskt material till koldioxid. Nitratet avgår som kvävgas. Denitrifikationen är därmed främst beroende av en syrefri miljö och att det finns tillgång till kolkälla, det vill säga organiskt material.

Denna studie har utvärderat kvävereningen på Kungsängsverkets C block. Meningen var att belysa den speciella processlösningens eventuella fördelar och få en bild av hur den fungerat för svenska förhållanden. På senare år har förutsättningarna för kväverening försämrats på Kungsängsverket. Den inkommande kolkällan som är viktig för denitrifikationen har minskat bland annat genom att flera livsmedelsproducenter stängt ner. Samtidigt har driftsättningen av de två nya vattenverken i Uppsala under hösten 2007 medfört en alkalinitetssänkning av inkommande vatten. Eftersom nitrifikationen konsumerar alkalinitet är det viktigt att det finns tillräckligt med alkalinitet i inkommande vatten för att hålla ett neutralt pH i bassängerna. Detta har inte lyckats efter vattenverkens introduktion. För att se till att behovet av kolkälla till denitrifikationen uppfylls har produktion av intern kolkälla med två olika metoder använts på block C. En delström av inkommande vatten har skickats förbi den primära reningen – så kallad by-pass – under perioder med vinterdrift sedan 2007. Primärslamhydrolys – då vatten pumpas flera gånger genom primärsedimenteringen för att bidra med mer lättillgängligt kol – har varit i drift i perioder under våren 2008. Undersökningen har även varit inriktad på hur dessa yttre faktorer påverkat kvävereningen men också vad produktionen av intern kolkälla bidragit med.

Genom statiska beräkningar på processen, användning av modellering i form av PCA (Principal Component Analysis) och modellering med ASM1 i JAVA Activated Sludge Simulator (JASS) samt provtagning på verket har en utvärdering av processens drift sedan våren 2007 utförts.

Block C tar förutom avloppsvatten emot rejektvatten från slamavvattningen vilket medför att verket handskas med höga halter kväve med svenska mått mätt. Ur ett internationellt

perspektiv är halterna in till block C normala. Processen lyckas i medeltal avskilja 78 % av allt inkommande kväve. Efter mellansedimenteringen kvarstår 12 mg/L, med lägre halter under våren 2008. Av dessa 12 mg/L utgörs 9,7 mg/L av nitrat och 1,3 mg/L av ammonium. Under perioden har en förskjutning skett mot lägre nitrathalter och högre ammoniumhalter.

Block C har under utvärderingsperioden haft en uppehållstid på 9 timmar och en slamålder på i medeltal 10 dagar. Slambelastningen till det biologiska steget är 0,07 kg BOD_7/ kg VSS, d. Returslamflödet har varit 180 % av inkommande flöde och medelslamhalten har varit hög; över 5100 mg/L i genomsnitt. Syrehalterna i de aeroba zonerna har legat mellan 2 och 3,5 mg/L fram till februari 2008 då börvärdena sänktes till mellan 1,5 och 2,5 mg/L.

Förutsättningarna för nitrifikation har mycket riktigt försämrats under utvärderingsperioden. Efter vattenverkens introduktion med mjukgörning av Uppsalas dricksvatten är pH ut från C-blocket 6,5, vilket medför att nitrifikationen arbetar långsammare än vid pH mellan 7 och 8, vilket var fallet innan driftsättningen. Det kan man även se på ammoniumhalterna som går upp i september 2007. Nitrifikationshastigheten är beräknad till 1,7 mg N/g VSS, h vid 15 ºC. Med utgående ammonium från block C på 1.3 mg/L är det rimligt att anta att hela den aeroba volymen inte utnyttjas fullt ut, och att nitrifikationshastigheten därmed underskattas. Den justerade nitrifikationshastigheten är uppskattad till 1.9 mg N/g VSS, h. I jämförelse med elva reningsverk med fördenitrifikation i Sverige har block C låga halter utgående ammonium i förhållande till den aeroba kvävebelastningen.

Denitrifikationen är beroende av den inkommande kolkällan till blocket. Ju högre COD/N kvoten är, desto lägre är nitrathalterna i utgående vatten. Efter försedimenteringen har C-blocket en BOD/N kvot på 2 (by-pass medräknat) vilket egentligen är för lågt för fullständig denitrifikation. Ett mer rimligt förhållande mellan BOD och N skulle vara 3-6:1. Intressant nog konsumerar processen relativt lite kol i relation till det kväve som renas i jämförelse till andra fördenitrifikationsprocesser, vilket indikerar ett effektivt kolutnyttjande. Av de två metoderna för produktion av intern kolkälla är primärslamhydrolysen det som ger störst effekt på BOD/N kvoten till biosteget, med en bidragande ökning på 28 %. Denitrifikationshastigheten beräknades till 2,2 mg N/g VSS, h vid 15 ºC. Även detta antas vara en underskattning framförallt under sommartid. En bättre uppskattning skulle vara 2,5 mg N/g VSS, h.

I den nationella jämförelsen grupperar sig block C tillsammans med de verk som doserar extern kolkälla, både vad gäller förutsättningar och reningsresultat, ammonium undantaget. Block C har de högsta nitrathalterna i jämförelsen, även om de verk som har lika låga C/N-kvoter som block C i regel behöver dosera externt kol. Om fosfor renas biologiskt, genom så kallad bio-P, tenderar verken att klara denitrifikationen bättre.

Det var svårt att få denitrifikationen i simulatorn JASS att överensstämma med verkligheten. Systemet var kolbegränsat och den kalibrerade modellen svår att verifiera för vinterdrift. Resultaten från simuleringen indikerade effektivare kolanvändning med kaskadkväverening i jämförelse med fördenitrifikation.

Sammanfattningsvis är det denitrifikationen som behöver åtgärdas på C-blocket. Eftersom det finns fler block på verket och ett av dessa väntar på ombyggnad, bör inga kostsamma investeringar, så som dosering av externt kol, göras innan denna ombyggnad är utförd. Det finns däremot möjlighet att optimera primärslamhydrolysen om kolbristen skulle bedömas för stor, samt att utnyttja möjligheten till ytterligare optimering av kaskadkvävereningen.

Preface

This project was written for Ramböll Sverige AB and is the Master Thesis project for the degree of M.Sc. in Environmental and Aquatic Engineering at Uppsala University.

In particular, I want to send my gratitude to Petter Björkman for supervision and moral support and to Peter Ek for his expertise and thorough knowledge of the subject. Bengt Carlsson has in his role of subject reviewer been as supportive as ever.

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Table of content

List of Symbols

Symbol	**Description**	**Swedish translation**
ASM	Activated sludge model	Aktivslammodell
BNR	Biological nutrient removal	Biologisk kväverening
BOD	Biological oxygen demand	Biologisk syreförbrukning
COD	Chemical oxygen demand	Kemisk syreförbrukning
DO	Dissolved oxygen	Löst syre
F/M	Food-to-microorganism ratio	Slambelastning
HRT	Hydraulic retention time	Hydraulisk uppehållstid
MLSS	Mixed liquid suspended solids	Slamhalt i luftningsbassängen
MLVSS	Mixed liquid volatile suspended solids	Volatil slamhalt i luftningsbassängen
$NH_4\text{-}N$	Ammonia nitrogen	Ammoniumkväve
$NO_3\text{-}N$	Nitrate nitrogen	Nitratkväve
PCA	Principle component analysis	Principalkomponentanalys
Q	Flow	Flöde
RAS	Return activated sludge	Returslam
R_d	Denitrification rate	Denitrifikationshastighet
R_n	Nitrification rate	Nitrifikationshastighet
S	Concentration of soluble substance	Koncentration löst ämne
SRT	Solids retention time/Sludge age	Slamålder
SS	Suspended solids	Suspenderad substans (slamhalt)
SS load	Solids loading	Slamytbelastning
SVI	Sludge volume index	Slamvolymindex
tot-N	Total nitrogen	Totalkväve
WAS	Waste activated sludge	Överskottslam
VL	Volumetric load	BOD-belastning
VSS	Volatile suspended solids	Volatil suspenderad substans
WWTP	Wastewater treatment plant	Avloppsreningsverk
X	Concentration of particulate substance	Koncentration fast ämne
Y	Yield	Utbyte
θc	See SRT	Se SRT

1 INTRODUCTION

When removing nutrients and organic matter from municipal and industrial wastewater, the activated sludge process is the most commonly used biological process. The name of the process refers to the active biomass which uses organic compounds and nutrients for their synthesis and generation of new cells.

In the 1940s, Uppsala municipality commissioned the first part of Kungsängsverket, the wastewater treatment plant of Uppsala city. There were to be several major rebuilds of the plant, the latest was taken in operation in the late 1990s. This expansion of the plant is referred to as the C-block and has to date been operating in nearly 10 years with in general very good performance according to the effluent standards.

However, the process solution of the C-block is quite unique in its form in the Nordic countries and hence it is of particular interest to evaluate the process further. The process incorporated in the C-block is a three stage step-feed biological nutrient removal (BNR) process and has earlier been evaluated in full-scale plants in Germany (Schlegel, 1992) and in the United States (Fillos *et al.*, 1996). The major advantages of the step-feed BNR process is a reduced need for recirculation of nitrate within the reactor and reduced loading to the secondary clarifier, leading to both an economic performance and a possibility to increase the solids inventory in the reactor compared to ordinary predenitrification.

An evaluation of the process performance can increase the knowledge of the operation of the plant in recent years in order to improve process performance in the future and perhaps contribute with knowledge if new plants are to be constructed according to the step-feed configuration. At the time being, Swedish communities await new regulations from the EU regarding criteria for effluent nitrogen standards. If the nitrogen effluent concentrations are to be reduced from Swedish wastewater treatment plants now operating without nitrogen removal, many plants will have to rebuild their process for nitrogen removal.

1.1 OBJECTIVES

The aim of this study is to perform an overall process evaluation of the nitrogen removal in the C-block at the Kungsängen wastewater treatment plant through verification of the previously made process calculations and through determination of several important process variables, such as detention times, loading and nitrification rate.

It is also of interest to verify the steady-state calculations with an activated sludge model to investigate how well the model describes the actual process and to simulate operational changes in the plant.

1.1.1 Limitations

The study is not concerned with the operation of block A and B at Kungsängen WWTP. Also, it does not cover inhibition on the biological processes by metals or other substances. The evaluation is focusing mostly on data with daily averages and is only to a limited extent discussing daily variations. The modelling with activated sludge model no. 1 does not include characterization of wastewater or calibration of model parameters through biological, physical or chemical methods.

2 THE ACTIVATED SLUDGE PROCESS

The main objective for treating wastewater was initially to remove pathogenic organisms and organic matter (Metcalf & Eddy, 1991). The enhanced treatment of wastewater in Sweden in 1900–1930 had a positive effect on the public health (Svenska kommunförbundet, 1996). Also, a high amount of organic matter in the effluent would reduce the dissolved oxygen (DO) concentration in the receiving waters. Later on in the 1960's, a deeper concern for the negative environmental impacts of nutrients in the effluent and the debate on eutrophication led to more stringent discharge standards for both nitrogen and phosphorus in the 1990s. The ability to remove these constituents from the wastewater became an additional objective for the treatment process and biological nutrient removal (BNR) was incorporated in the process solutions.

The activated sludge process is an over 90 year old biological wastewater treatment process. The process is the secondary treatment step in the overall wastewater treatment, the first being removal of particulate solids. Over the years, different process solutions have been developed and specific arrangements are supplied for removal of nitrogen and phosphorus. The basic principle for the process remains: an active biomass removes soluble organic matter, stabilizes insoluble organic matter and, if applicable, transforms soluble inorganic matter (Grady *et al.*, 1999).

In the ordinary activated sludge process, microorganisms are contained in an aerated basin where it oxidizes organic material. Entities of microorganisms, termed flocs, are kept in continuous flow in the basin together with the wastewater to be treated, and this slurry is referred to as mixed liquid (Grady *et al.*, 1999). The mixing provides contact time between microorganisms and soluble substrate and the flocs are essential due to their ability to gather and hence stabilize the insoluble nonsettleable solids. After the treatment the biomass is removed from the treated water through gravity clarification and the major part of the settled biomass, referred to as return activated sludge (RAS), is recycled to maintain a sufficient solids inventory in the reactor. The excess sludge (WAS) is wasted (Figure 1).

In a BNR system not all parts of the reactor are aerated, hence the biochemical environment changes in different zones (Grady *et al.*, 1999). If nitrogen is to be removed an aerobic and an anoxic zone is needed, while biological phosphorus removal requires aerobic, anaerobic and anoxic environments. The type of biochemical environment is determined by the electron acceptor in the oxidation-reduction reactions that provide the organisms with energy. In these reactions, an electron acceptor is reduced when the organic material is oxidized (Metcalf & Eddy, 1991). The most energy-efficient electron acceptor is oxygen and if this is available the reaction is aerobic. In an oxygen-free environment, other electron acceptors are needed, nitrates and iron being two examples. The term anoxic is used within the wastewater treatment field to distinguish the use of nitrate or nitrite from other anaerobic electron acceptors (Grady *et al.,* 1999).

The microbial growth not only needs an electron acceptor, but also a carbon source and nutrients to function properly (Metcalf & Eddy, 1991). In general, microorganisms can obtain their carbon source from organic compounds (the organism is heterotrophic) or from carbon dioxide (the organism is autotrophic). Autotrophs are more slow-growing than heterotrophs.

Nitrogen is present in the influent wastewater mainly as ammonium (NH_4) and to some extent in the form of organic nitrogen. In biological nutrient removal two major groups of microorganisms are responsible for the nitrogen removal; nitrifiers and denitrifiers.

Nitrifiers are aerobic bacteria and constitute two key genera: Nitrosomonas which oxidize ammonia to nitrite and Nitrobacter which oxidize nitrite to nitrate (Henze *et al.*, 2002). Due to the fact that nitrifiers are autotrophs with a slow growth rate, they require a long solids retention time (SRT) in the reactor. They are also sensible to low temperatures and inhibitors in the wastewater. The nitrification equation is described below.

$$NH_4^+ + 1.5\ O_2 \longrightarrow NO_2^- + 2H^+ + H_2O \qquad \text{(Nitrosomonas)}$$

$$NO_2^- + 0.5\ O_2 \longrightarrow NO_3^- \qquad \text{(Nitrobacter)}$$

Denitrification is a respiratory process performed in an anoxic environment and the process converts nitrate via nitrite to atmospheric nitrogen (Henze *et al.*, 2002). It is most important that no oxygen is available since this would mean oxygen will be the oxidizing agent instead of nitrate. There are more than 50 genera of denitrifying bacteria and they grow faster than their nitrifying relatives. Since these organisms are heterotrophs they need organic material to sustain their life processes. The denitrification process is described below.

$$2NO_3^- + H^+ + \text{organic matter} \longrightarrow N_2(g) + HCO_3^-$$

In the simplest form of a suspended growth BNR process there are two zones: one aerobic and one anoxic. Both predenitrification and postdenitrification is possible. With postdenitrification, an external carbon source might be needed in the anoxic zone while in the predenitrification internal recycle of nitrate is needed to accomplish denitrification. There is an overview of the predenitrification process in Figure 1. The physical configuration of the actual basins differs from the process configuration in Figure 1.

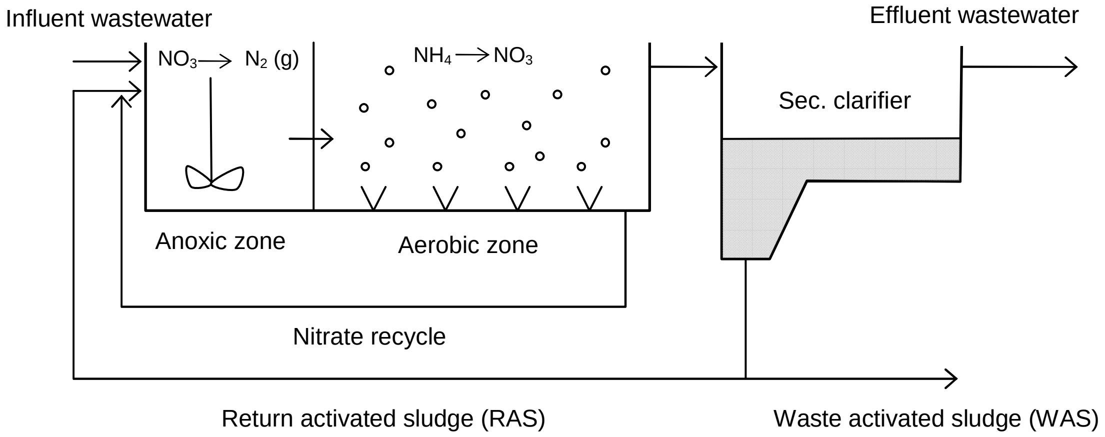


Figure 1. Overview of the predenitrification process.

2.1 COMMON WASTEWATER AND BIOMASS CHARACTERISTICS

There are many ways in which to characterize wastewater and biomass. It is essential to have a good overview of the wastewater contaminants in the design and operation of a wastewater treatment plant.

The *total solids* in the wastewater can either be dissolved or suspended (Henze *et al.*, 2002). The solids passing a filter with a pore size of 0.45 µm (in Sweden, pore size may vary) are dissolved or soluble, and the other particles are suspended or particulate. Soluble matter is denoted S with a suffix and for particulate solids, an X with a suffix is commonly used. Apart from solubility; the total solids in the water can be divided into settleable or nonsettleable solids. One of the main objectives in wastewater treatment is to reduce the *organic matter* in the wastewater. Organic matter can be present in many forms, such as carbohydrates, surfactants and proteins (Metcalf & Eddy, 1991). When analysing the organic content, all different types of organics should be detectable. Therefore, different collective analyses have been developed to be performed on wastewater (Henze *et al.*, 2002). The biochemical oxygen demand (BOD) is a measure of the biodegradable organic material in the wastewater. During a BOD test, the microorganisms oxidize organic material and ammonium and the resulting oxygen demand is measured. The test can be performed during five (BOD_5) or seven (BOD_7) days.

The chemical oxygen demand (COD) covers a larger fraction of the total organics and the test is conducted by adding a chemical oxidizing agent. The test is faster than the BOD test (1–2 h) and more suitable for mass-balance analyses (Section 2.2) (Henze *et al.*, 2002). COD is normally divided into different fractions depending on how it is formed and how it behaves. These are further discussed in Section 2.3.1.

The nitrogen content in wastewater is also divided into different fractions. The incoming wastewater mainly contains ammonia nitrogen (NH_4-N) and organic nitrogen (org-N). Through nitrification, ammonia is converted to nitrate nitrogen (NO_3-N).

2.2 PROCESS PARAMETERS

In Figure 2, flows and concentrations in a mass-balance over the aeration tank is presented, since these are common in several of the process parameters.

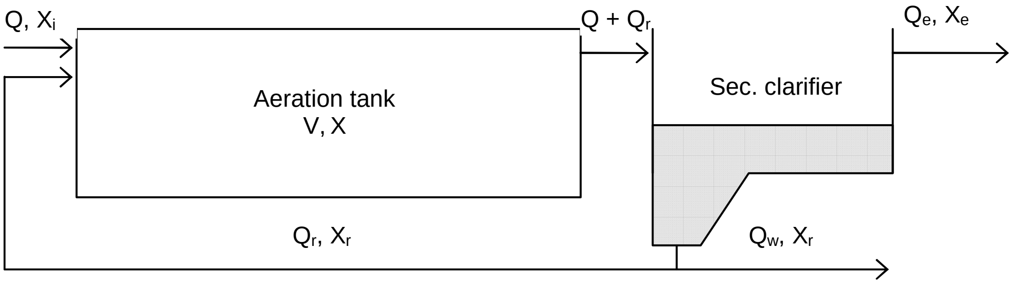


Figure 2. Aeration tank mass balance (Metcalf & Eddy, 1991).

The *suspended solids (SS)* or *mixed liquor suspended solids (MLSS)* is the concentration of suspended solid particles at any point in the system or in the aeration tank respectively. The volatile fraction of the suspended solids in the aeration tank (*MLVSS*) is the biomass concentration.

The *hydraulic retention time (HRT, θ) g*ives the average time it takes for the water to move from the start to the end of the aeration tank.

$$\theta = \frac{V}{Q} \tag{1}$$

θ= hydraulic retention time, d
V= aeration tank volume, m^3
Q= influent flowrate, m^3/d

The *solids retention time (SRT, θ_c)* is also referred to as the mean cell residence time or sludge age. The SRT describes the average time the solids remain in the system. SRT can be calculated for the whole reactor volume or only for the aerobic volume.

$$\theta_c = \frac{VX}{Q_w X_w + Q_e X_e} \tag{2}$$

θ_c= solids retention time, d
V= aeration tank volume, m^3
X= concentration VSS, mg/L
Q_w= waste sludge flowrate, m^3/d
X_w= VSS in waste sludge, mg/L
Q_e= effluent flowrate, m^3/d
X_e= VSS in effluent, mg/L

The *food-to-microorganism ratio (F/M)* gives a measure of how much substrate is entering the aeration tank in relation to the biomass concentration and for how long it remains in the system. In Sweden referred to as the sludge loading (slambelastning). F/M is together with SRT the most commonly used process parameter (Metcalf & Eddy, 1991).

$$F/M = \frac{S_{in}}{\theta X} \tag{3}$$

F/M= food-to-microorganism ratio, d^{-1}
S_{in} = influent BOD/COD concentration, mg/L
θ= hydraulic retention time, d
X= concentration VSS, mg/L

The *volumetric load (VL)* (BOD-belastning) determines the amount of food to the microorganisms entering the aeration tank in relation to the tank volume.

$$VL = \frac{QS_{in}}{1000 \cdot V} \tag{4}$$

VL= volumetric load, kg BOD_7/m^3, d
Q= inflow, m^3/d
S_{in}= BOD entering the aeration tank, mg/L
V= volume of aeration tank, m^3

The *yield coefficient (Y)* is the biomass produced in relation to the organic matter removed, given in BOD or COD.

$$Y = \frac{Q_w X_w + Q_e X_e - QX_i}{S_i Q - S_e Q_e} \qquad (5)$$

Y= yield coefficient, kg SS/kg BOD
Q_w= waste sludge flowrate, m^3/d
X_w= SS in waste sludge, mg/L
Q_e= effluent flowrate, m^3/d
X_e= SS in effluent, mg/L
Q= influent flowrate, m^3/d
X_i= SS in influent, mg/L
S_i= BOD or COD in influent, mg/L
S_e= BOD or COD in effluent, mg/L

In processes that remove nitrogen the *nitrification and denitrification rates (R_n, R_d)* are essential since they determine how efficient the nitrogen removal operates.

$$R_n = \frac{N_n}{X\theta_{aer}}, \text{ where } N_n = totNin - NO_3in - totNout + NO_3out - N_a \qquad (6)$$

R_n= nitrification rate, gNH_4-N/kg VSS, h
N_n= nitrified nitrogen, mg/L
N_a= particulate outgoing nitrogen, mg/L
θ_{aer}= aerobic hydraulic retention time, h
X= VSS in aeration tank, g/L

$$R_d = \frac{N_d}{X\theta_{an}}, \text{ where } N_d = totNin - totNout - N_a \qquad (7)$$

R_d= denitrification rate, g NO_3-N/kg VSS, h
N_d= denitrified nitrogen, mg/L
N_a= particulate outgoing nitrogen, mg/L
θ_{an}= anaerobic hydraulic retention time, h
X= VSS in aeration tank, g/L

Finally, two formulas describing clarification load are given. The hydraulic *surface load (SL)* is setting the limit for the allowable minimal settling velocity in the clarifier to achieve solids separation. The *solids loading (SS load)* is similar to the surface load, but describes the amount of solids entering the clarifier with respect to the surface area.

$$SL = \frac{Q}{A}, \qquad SSload = \frac{X(Q+Q_r)}{1000 \cdot A} \qquad (8)$$

SL= surface load, m/h
A= surface area of the clarifier, m^2
SS load= solids loading, kg SS/m^2, h
Q_r= RAS flow, m^3/h

2.3 MODELLING OF THE ACTIVATED SLUDGE PROCESS

The development of models describing the wastewater treatment process begun in the 1960's (Jeppsson, 1993). Due to the enhanced capability of computers and the reduced prize, the models became more sophisticated during the 1970's and 80's, incorporating among others different fractions of the organic material and the denitrification process.

A task group was formed in 1983 by the International Association on Water Pollution Research and Control (IAWPRC, today International Water Association, IWA). The aim was to find a model describing oxidation of carbon, nitrification and denitrification on the simplest form possible through revision of the models to date (Jeppsson, 1993). The task group presented the Activated Sludge Model no. 1 (ASM1) in 1987. Since then it has been considered a reference model since it was the first model gaining general acceptance (Gearney *et al.*, 2004).

There are several applications for WWTP modelling such as teaching, design and process optimisation. ASM1 is mainly applicable when municipal wastewater is to be treated and is still used in many contexts. In the 1990's, new versions of ASM1 were developed; ASM2 and ASM3 (Gearney *et al.*, 2004). ASM2 is extended with phosphorus removal and ASM3 is the preferred model when dealing with industrial wastewater.

In this project, ASM1 is used and the main purpose is to try to calibrate the model to the real system and if this succeeds evaluate different scenarios. The Java Activated Sludge Simulator (JASS), developed at the Department of Information Technology, Division of Systems and Control at Uppsala University, is based on ASM1 and has a version with an implementation of the process solution of block C at Kungsängen WWTP.

2.3.1 Activated Sludge Model no.1 (ASM1)

As mentioned above, ASM1 was the result of the work of the IAWPRC task group and it was presented by Henze *et al.* (1987). The basic unit for defining carbonaceous material was decided to be COD. When it comes to modelling of activated sludge processes, COD and nitrogen compounds are divided into several fractions as mentioned in Section 2.1. These fractions are state variables of the model. A state variable is an internal process variable that is updated at each time interval through integration of the corresponding state equations (Jeppsson, 1993).

The COD that is not active biomass can either be biodegradable or non-biodegradable. The biodegradable material can in turn be either readily (S_S) or slowly (X_S) biodegradable COD. Readily biodegradable material is believed to be small and simple molecules that can be transported through the cell walls of the biomass. Slowly biodegradable COD on the other hand needs to be enzymatically degraded through so called hydrolysis until it can be utilized by the microorganisms.

Non-biodegradable COD is not available for the microorganisms. The inert COD can both be soluble (S_I) and particulate (X_I). Some inert COD is produced when cells are decaying. The biomass is also a part of the total COD and can either be heterotrophs ($X_{B,H}$) or autotrophs ($X_{B,A}$), see Section 2.1. An overview of the COD characterization in ASM1 is found in Appendix A, Figure i.

Nitrogen can be divided into similar fractions as COD. Nitrate and nitrite nitrogen (S_{NO}), ammonia nitrogen (S_{NH}) are the fractions available for the biomass. A more detailed characterization of the nitrogen fractions in ASM1 is found in Appendix A, Figure ii.

The different ASM models all have their own way of describing substrate flows. The substrate flows of ASM1 are depicted in Figure 3.

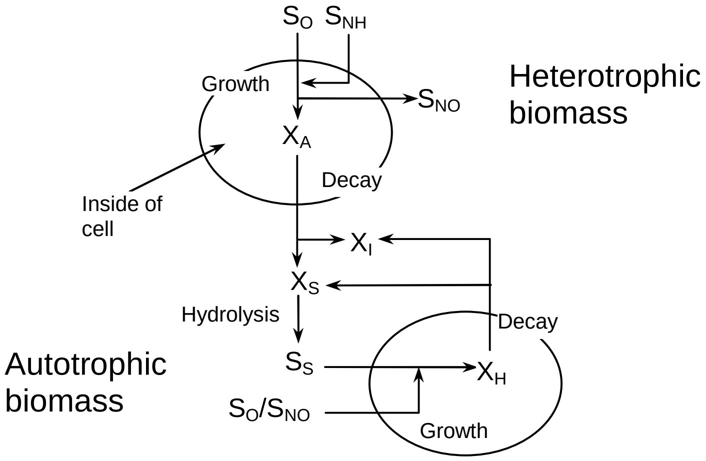


Figure 3. Substrate flows for autotrophic (nitrifying) and heterotrophic (denitrifying) biomass in ASM1. (Modified from Gujer *et al.* (1999)).

Apart from state variables and equations, the model parameters decide the model behaviour. There are 19 stoichiometric or kinetic parameters in ASM1, all described in Appendix B.

There are several constraints and limitations in ASM1, one example being the assumption that both temperature and pH are constant. For more information, see Jeppsson (1996) and Gearney *et al.* (2004).

3 STEP-FEED BIOLOGICAL NUTRIENT REMOVAL

As mentioned earlier, the activated sludge process can be configured in many ways; one example being the step feed activated sludge process with biological nutrient removal (BNR). The step-feed activated sludge process is a plug-flow process, i.e., a process where the RAS enters the head end of the reactor and the particles in the fluid are ideally leaving the reactor in the same order as they enter (Grady *et al.*, 1999). The opposite of the plug-flow reactor is the complete-mix reactor.

In the step-feed process the influent wastewater is fed at several stages in the biological reactor making readily biodegradable organic material available for the denitrifying bacteria in several compartments. If nutrient removal is to be incorporated, the basins are divided into several zones with different biochemical environments, similar to the ordinary activated sludge process with BNR. A step-feed predenitrification system with three stages is demonstrated in Figure 4. This process lacks the nitrate recycle present in the ordinary predenitrification process (Figure 1).

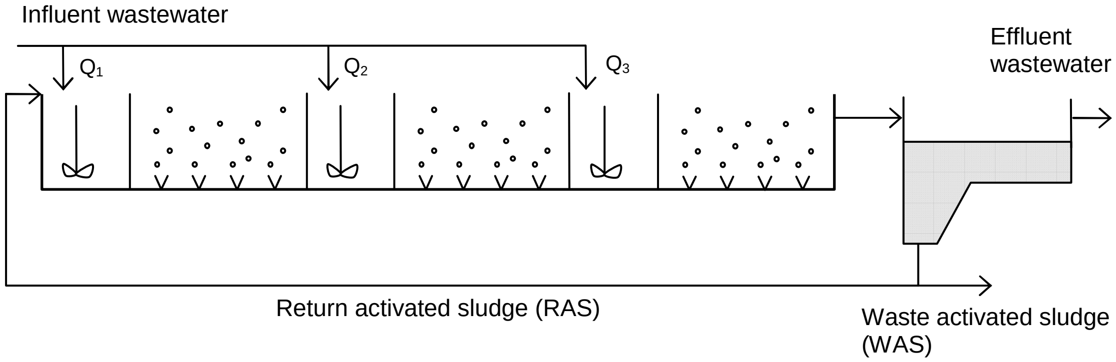


Figure 4. Overview of the step-feed denitrification process.

The step-feed BNR system was introduced in full-scale plants in Germany (Schlegel, 1992). The author concludes that successful nitrogen removal can be achieved without recirculation of nitrate since the nitrate generated in the denitrification can be oxidized in the next nitrification zone. The reduced recirculation of nitrate reduces the overall operating costs of the plant. This has also been evaluated on a theoretical basis by Miyaji *et al.* (1980).

Other studies with full-scale evaluations of the step-feed process have been conducted in Germany (Kayser *et al.*, 1992), in the United States (Fillos *et al.*, 1996; Johnson *et al.*, 2005; Daigger & Parker, 1999) and in Turkey (Görgün *et al.*, 1996). Several benefits with the step-feed process have been reported both in full-scale, pilot-scale and theoretical analyses.

Apart from reducing energy consumption through the reduced need of pumping, the step-feeding creates a concentration gradient of MLSS in the tank (Nolasco *et al.*, 1993). The return sludge will only be partially diluted with the incoming wastewater at the head end of the reactor and the MLSS will be higher in the early compartments. This implies that the average MLSS in the tank can be higher in the step-feed process – compared to an ordinary plug-flow system – without increasing the solids loading to the secondary clarifier and SRT is thus increased (deBarbadillo *et al.*, 2002).

Due to the MLSS gradient across the tank more wastewater can be treated within the same reactor volume, a 35–70 % increase has been reported by (Crawford *et al.*, 1999), or smaller plants can be constructed without affecting clarifier load (Daigger & Parker, 1999). Another effect of a reduced SS loading to the clarifier is more stable settling conditions since there is a smaller risk for bulking sludge (Jenkins *et al.*, 2003).

Since not all of the wastewater is entering the tank in the same zone, the F/M loading is lower compared to ordinary plug-flow, which lowers peak oxygen demand (Metcalf & Eddy, 1991), making the system better at handling high organic loads (Hegg, 1990). The division of incoming wastewater can also be beneficial for the microbial community since it reduces the negative effects of toxic substances.

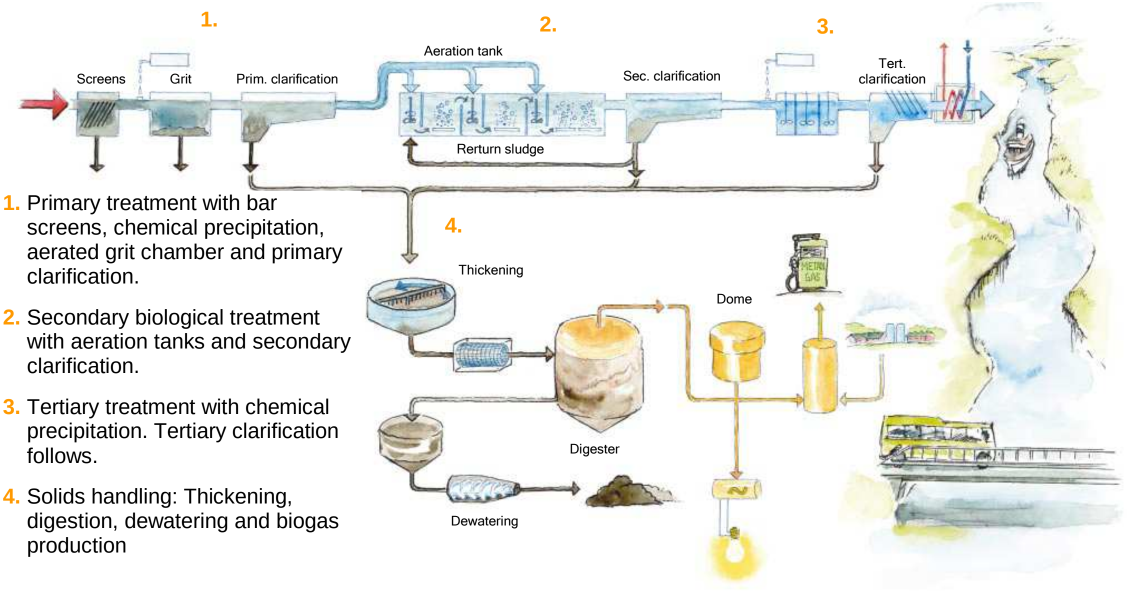
Another benefit from the step-feed configuration is the flexibility of operation (Metcalf & Eddy, 1991). One example is the possibility to shelter the biomass from wash-out during high flows through reduction of the influent flow to the first denitrification zone (Hegg, 1990; Nyberg *et al.*, 1996). As a result, there are plants which change the feeding point prior to wet weather flows.

The flexibility of operation is not only positive, since the many possibilities might make the operation more complicated if the operation is to be completely optimized. To tune the process and make proper use of the substrates and nutrients according to theoretical discussions within the field, a complete characterization of the incoming wastewater and subsequent implementation of new routines and operating conditions within the plant is required. In reality, this might be too time-consuming and requires high competence among the personnel at the plant.

4 KUNGSÄNGEN WASTEWATER TREATMENT PLANT

The Kungsängen wastewater treatment plant (WWTP) is situated in the south-eastern parts of Uppsala city. The first part of the plant, block A, was established in the 1940’s. Since then, several extensions have been added. The latest part, the C-block, was taken into operation in 1999 (Uppsala kommun, 2008). The present construction is dimensioned for 200 000 pe and the present load is 149 000 pe (based on 70 g BOD7/pe, d). Of the total flow to the plant, 7 % or 30 000 pe, is estimated to originate from industries in the area, the main actors being medical companies and food industry. For an overview of the process at Kungsängen WWTP, see Figure 5.

The plant separates visible objects, carbon, nitrogen and phosphorus from the wastewater. The treated wastewater is released into Fyrisån which enters the northern branch of Lake Mälaren. In the primary treatment step, physical objects are removed from the wastewater with the help of bar screens. Larger particles and sand with a diameter less than approximately 0.15 mm is separated in an aerated grit chamber. Before the wastewater enters the grit chamber, iron chloride is added to enhance the separation of primarily phosphorus. In the primary clarification, smaller particles are removed before the effluent continues to the biological treatment step. Block A and B have one common treatment system until the water enters primary clarification.



1. Primary treatment with bar screens, chemical precipitation, aerated grit chamber and primary clarification.
2. Secondary biological treatment with aeration tanks and secondary clarification.
3. Tertiary treatment with chemical precipitation. Tertiary clarification follows.
4. Solids handling: Thickening, digestion, dewatering and biogas production

Figure 5. Overview of Kungsängen WWTP (Drawing from Uppsala municipality).

The secondary treatment is a biological process with activated sludge. Ordinary predenitrification is operating in all of the three lines in block B and in line 1–2 in block A while step feed predenitrification with three stages is implemented in line 3–5 in block A and in block C. The influent wastewater is fed to the inlet with predenitrification and to the three anoxic zones with step-feed. Each block has its own secondary clarification tanks with a total area of 6250 m^2.

After the secondary clarification the wastewater from all three blocks is pumped to tertiary treatment where phosphorus and remaining bio solids are precipitated with small amounts of iron chloride. The flocs are separated through tertiary clarification. An overview of the plant is found in Appendix C.

Waste sludge from the chemical precipitation is pumped back to the primary clarification in block B and is further treated together with the primary sludge in the solids handling. The sludge is pumped to a gravity thickener and through drum thickeners to the digester for production of biogas. Through digestion, the sludge is stabilized and sent to the mechanical dewatering. Before dewatering a polyelectrolyte and a foam control agent is added. The stabilized and dewatered sludge is kept in large silos, awaiting transportation to the Hovgården industrial waste site and is presently used to cover closed landfill areas.

Block C is dimensioned for 2 800 m^3/h and block A and B for 1 000 m^3/h each. On average, the plant treated 51 100 m^3/d during 2007 out of which 62 % was treated in block C, which is equivalent to about 1 300 m^3/h. The effluent quality during 2007 and 2006 together with the required effluent concentrations is presented in Table 1. The total nitrogen concentration exceeds the limits in 2007.

Table 1. Effluent requirements and yearly averages during 2006 and 2007 for Kungsängen WWTP.

	BOD_7 [mg/L]	*Tot-N [mg/L]*	*Tot-P [mg/L]*
Required	10	15	0.3
2006	<3	14	0.11
2007	<3	16	0.12

4.1 BLOCK C

Block C was constructed in the mid 1990's in order to improve the nitrogen removal of Kungsängen WWTP and to prepare the plant for an increased load. When it was taken into operation in 1999, there was a significant reduction in nitrogen discharge from the plant. (Uppsala kommun, 2008).

The load of carbon, nitrogen and phosphorus was at Q_{dim} estimated to be 58 % of 14 000 kg/d, 24 000 kg/d and 550 kg/d (RUST VA-projekt AB, 1996). More details of the dimensioning data for block C are found in Appendix D, Table i. The reject water flow was supposed to be fed to block A, but is today fed to block C.

The process configuration of the biological treatment in block C is described in detail in Figure 6. Approximately one third of the incoming wastewater is fed to each stage. All three stages are divided into separate zones where equipment for mixing, aeration or both is installed (Ek, 2001). The combination of mixers and aerators in the same zone bring about a more flexible process and the zones can be operated differently during winter and summer periods to compensate for the effect of temperature changes on the growth rate of nitrifiers.

Three cases of operation have been used during the evaluation period, which are displayed in Table 2 together with the zone volumes. The two winter cases have a larger aerobic volume and are more suitable for a colder climate.

There is a possibility of adding a carbon source to zone 3:2 and 3:3 if there is a lack of easily biodegradable material for denitrification (RUST VA-projekt AB, 1996). This has not happened yet, but is a future possibility if needed.

In Figure 7, the physical configuration of one out of five process trains is demonstrated. Each process line has two sedimentation tanks. Line 3 is built as an experiment line. This implies that it has its own pumping station for RAS and WAS. There is one pumping station for line 1+2 and one for line 4+5.

Apart from incoming domestic wastewater, external sludge and reject water from the sludge dewatering has since a few years been fed to the inlet in block C. The reject water has a high concentration of ammonia and hence increases the nitrogen load to the block which handles higher nitrogen concentrations than block A and B.

Each process line has two sedimentation tanks in the secondary sedimentation step with a total area of 3600 m^2 and a depth of 5 m.

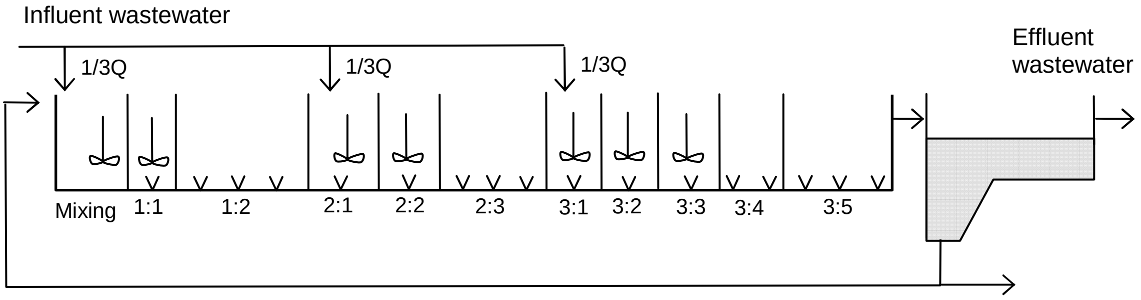


Figure 6. The process in block C. Several compartments in the activated sludge basin can be either aerobic or anoxic.

Table 2. Area and biochemical configuration of the zones in block C.

Zone	***Volume (m^3)***	***Winter 2007***	***Summer***	***Winter 2008***
Mixing	896	Mixing	Mixing	Mixing
1:1	896	Aerobic	Anoxic	Aerobic
1:2	1792	Aerobic	Aerobic	Aerobic
2:1	896	Anoxic	Anoxic	Anoxic
2:2	896	Aerobic	Anoxic	Aerobic
2:3	1878	Aerobic	Aerobic	Aerobic
3:1	936	Anoxic	Anoxic	Anoxic
3:2	936	Anoxic	Anoxic	Anoxic
3:3	936	Aerobic	Anoxic	Anoxic
3:4	936	Aerobic	Aerobic	Aerobic
3:5	2200	Aerobic	Aerobic	Aerobic
Total aerobic volume		9534 (72%)	6806 (52%)	8598 (65%)
Total anoxic volume		3664 (28%)	6392 (48%)	4600 (35%)

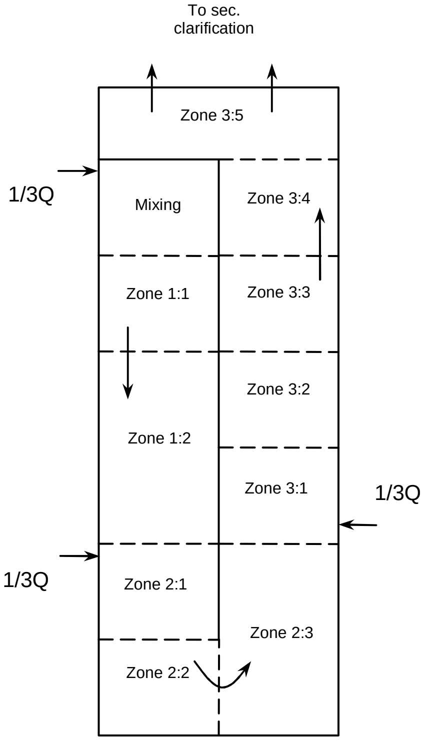


Figure 7. Physical process configuration of one of the lines in block C.

4.1.1 Operation of Block C

The wasted sludge flow is the control parameter to regulate the MLSS in the aeration tanks in block C, and hence affect the sludge age. The other major parameter that is available for control is the oxygen set-point. Apart from these possible modifications, several internal factors within the plant influences the operation of the nitrogen removal in block C, see list below (Holmström, 2008; Jidetorp, 2008).

- *Hydrolysis of sludge in the primary clarification* has been carried out through recirculation of sludge from the primary clarification tank back to the inlet of the same tank where it is mixed with incoming wastewater. This has been in operation in periods since December 2007 and enhances the carbon source into the biological treatment step through partial degradation of less available substrates.

- *By-pass over primary clarification* is the second strategy to increase the carbon level. During cold weather periods (December–May) a fraction of the raw wastewater is fed directly to the bio reactor and since it has not been treated in the primary clarification this fraction has a higher carbon concentration.

- *Reject water* from the dewatering of sludge is passed on to block C, which has already been mentioned. When the centrifuges are in operation they add a substantial amount of nitrogen load to the C-block.

- *External sludge* from the region is regularly transported to a tank at the WWTP where it is subsequently pumped to the inlet of block C. The external sludge originates from private and small scale wastewater treatment facilities or other actors in the water sector. Depending on the type of external sludge, this might add carbon to the process.

During the last years, external factors that are out of control of the plant itself though they have been predictable are (Holmström, 2008):

- Two new *water plants* in Uppsala in Bäcklösa and Gränby have been taken into operation during 2007, decreasing the influent alkalinity which consequently lowers the alkalinity in the water entering Kungsängen WWTP.

- *GE Healthcare* has improved their wastewater treatment in 2005. The effluent from their treatment is entering Kungsängen WWTP together with biological sludge. It is believed that the microorganisms in the sludge consume available substrates in the aerated grit chamber and in the primary clarification, leaving less carbon to the activated sludge process.

- *Scan* and *Slotts* are two provision industries that lately have closed down in Uppsala, further reducing the carbon source to the biological treatment.

Two major problems with the operation have been experienced on the block. There have been difficulties with filamentous bacteria in mainly line 4 and 5. The problem has been solved with dosage of the chemical PAX (poly aluminium chloride). Also, the blowers connected to block C have been stopping from time to time during the evaluation period. This has been remedied during summer 2008.

5 METHOD

In this project, an evaluation of the nitrogen removal in the C-block at Kungsängen WWTP is carried out. A process evaluation of this kind can be carried out in several ways depending on what is considered important. At the early beginning it was clear that an overall process evaluation, a comparison with dimensioned data and calculations of important process parameters together with simulations were the main objectives (Section 1.1) of the project. In order to meet these expectations the following evaluation has been performed:

- *Static calculations*
    - Calculations of process parameters
    - Brief analysis of flow patterns, aeration, MLSS variations and primary clarification
- *Complementary on-site sampling*
    - Mass balance calculations
    - Nitrogen profile in aeration tank
- *Principal Component Analysis (PCA)*
- *Simulations in JAVA Activated Sludge Simulator (JASS)*

When applicable, the results were compared with plant dimensioning data, other plants with predenitrification in Sweden and with literature on step-feed BNR. Changes in operation due to internal and external factors (Section 4.1.1) were also considered.

5.1 COLLECTION OF DATA MATERIAL AND STATIC CALCULATIONS

In order to evaluate the nitrogen removal in block C at Kungsängen WWTP, data from the supervision and control system Uni-View and from laboratory analyses performed on the wastewater was collected. The evaluation period reached from 2007-02-12 until 2008-04-23. This period was limited by the storage capacity of Uni-View, which only saves detailed data from the previous 14 months. The signals were collected through an Excel macro which can create reports with data from the Uni-View database.

There are many available signals in the supervision system of the Kungsängen WWTP. Not all of them are important to this project. The signals of interest that have been used in the evaluation are presented in full detail in Appendix E. Most of the information was collected as daily averages apart from flow through block C, precipitation, DO concentrations in aeration tank and MLSS where data also was present with hourly intervals.

The laboratory analyses are 24-hour composite samples taken on a random day each week. The available measurement points and respective parameters are found in Table 3. The influent water is sampled before the screens while the other sample points are just after the effluent of the primary and secondary clarification tanks.

The samples taken after the primary clarification do not include the by-pass stream. During the winter period of 2007 by-passed water was pumped at a rate of 100 m^3/h. During winter 2008, a gate was used for by-passing. The flow rate created by the partly open gate is larger than 100 m^3/h but otherwise unknown. Since it was physically impossible to measure the flow rate, it was estimated to be 120 m^3/h in the calculations.

Table 3. Measurement points and analyzed parameters during weekly composite sampling.

	Influent	*Prim. effluent*	*Sec. effluent*
Alkalinity	x	x	x
BOD_7	x	x	x
COD	x	x	x
N-tot	x	x	x
NH_4-N			x
NO_3-N			x
P-tot	x	x	x
SS	x	x	x

The MLVSS/MLSS ratio was regularly measured until 2001. The average of these measurements was 0.77. A sample was analysed within the frames of this project which resulted in a ration of 0.77, hence 0.77 was chosen in all calculations of MLVSS. When needed in the calculations, the ammonia concentration in the influent was estimated to be 70 % of the total nitrogen concentration. This was measured in May 22nd.

When correcting the nitrification and denitrification rates for temperature, the corresponding rate at 15 ºC is calculated by using the following formula, where T is temperature in ºC.

$$r(15) = r(T)e^{-0.1(T-15)} \qquad (9)$$

Higher sludge age is needed at colder climate to compensate for the decrease of nitrification rate. One way to see the effect of temperature is to calculate the temperature compensated sludge age which is lower when temperatures are below 15 ºC. Formula from Stake (2005).

$$\theta_{c,T} = \theta_c / 1.127^{(15-T)} \qquad (10)$$

Apart from parameters listed in Section 2.2, the following was estimated:

- *Removal efficiency* of nitrogen, phosphorus BOD_7 and COD over the primary clarification, aeration tank and both.
- *kWh/kg N*: The total effect (calculated as the sum of power consumed by pumps and aeration device) used to remove one kilogram of nitrogen. The power data had been collected by the plant personnel during the period 2008-02-29 to 2008-04-29 and is a part of an energy project at Kungsängen WWTP. The average total nitrogen treated, expressed in kg/d, was evaluated for the period 2008-02-27 to 2008-04-28.
- *COD/BOD*, *COD/N*, *BOD/N* and COD consumed to nitrogen denitrified ($\Delta COD/N_d$).

Since the step-feed process is a modified predenitrification process, process data from other Swedish plants with different predenitrification processes was collected and a comparison between these processes and the process at Kungsängen WWTP was performed.

5.1.1 Error estimation

To estimate the error of the above calculations, a probable error analysis was performed on all the formulas in the investigation, see equation 11.

$$\delta X = \sqrt{\left(\delta y_1 \frac{\partial X}{\partial y_1}\right)^2 + \left(\delta y_2 \frac{\partial X}{\partial y_2}\right)^2 + \ldots + \left(\delta y_n \frac{\partial X}{\partial y_n}\right)^2} \tag{11}$$

where δX denotes the probable error of function X that consists of the variables $y_1,\ldots, y_n$. δy_i denotes the error of the different variables respectively and $\partial X/\partial y_i$ is the derivative of X with respect to y_i.

The laboratory methods all have their own errors which were employed in equation 11. The equipment at Kungsängen WWTP on the other hand, also produces a measurement error. This error was estimated with the help of personnel at the plant for MLSS, flow, ammonia, nitrate, pH and temperature equipment (Eriksson, 2008, pers. comm.).

5.1.2 Limitations and missing data

During the evaluation period, not all data was available at all times. The weekly sampling data for the primary and secondary clarification at the C block is missing during May 2007 due to problems with the samplers.

Also, the temperature data is missing from September 27 to October 18 2007 and during 6 weeks from February to April 2008, the measurements are not considered accurate since they indicate a too high temperature. The inconsistent temperature measurements are due to rat attacks on the cables from the temperature sensor (Eriksson, 2008, pers. comm.). To be able to use the temperature data during periods without measurements, the temperature was interpolated for the missing periods.

The nitrate equipment that registers the outgoing nitrate concentration from block C did not function for 2.5 months during October to December 2007. The nitrate measurements from the 24-hour composite samples was however available during the same period. Unfortunately, the period with missing nitrate data was the period with very high nitrate in the effluent.

The flow distribution to the lines and to the zones has been changed during the evaluation period (Jidetorp, 2008, pers. comm.). In October 2007, there was an adjustment in order to have a more proportional flow division to the different aeration basins. In January 2008 the inflow to zones 1 and 2 was reduced by 10 %, and in the end of March the same year the flow was increased again. These changes have not been incorporated into the calculations above.

5.2 ON-SITE SAMPLING

5.2.1 Mass balance analysis

On May 22 2008, complementary sampling was carried out in order to be able to perform a mass balance analysis on the C block. Moreover, the aim was to calculate nitrification and denitrification rates.

May 22 was the ordinary sampling occasion for that specific week. Apart from the sampling positions in Table 3, a sample was collected after primary clarification after the by-passed stream was fed and at the three return sludge flows. All samples were analyzed for alkalinity, BOD_7, COD, tot-N, filtered tot-N, NH_4-N, NO_3-N, tot-P and suspended solids. Also, the volatile fraction of the suspended solids, which is equivalent to the concentration of microorganisms, was analyzed on one of the RAS samples at Alcontrol Laboratories. The filter size used was 1.6 µm.

A portable sampler was positioned after primary sedimentation and after the gate where by-passed water was added. The sampling was initialized at 7 am and continued for 24 hours in order to obtain a just picture of a whole twenty-four hour cycle.

The return sludge was sampled at the inflow points to line 1 and 2, line 3 and line 4 and 5 respectively. The return flow rate of each stream is fairly constant over time, although they are not comparable to each other. Ten samples were collected hourly between 7:30 am and 4:30 pm.

The four extra samples were treated and conserved together with the ordinary samples at the Water Laboratory in Uppsala.

5.2.2 Nitrogen profile

On July 3 2008, sampling was performed for the purpose of creating a nitrogen profile. The samples, taken in line 3 of block C, are depicted in Figure 8. Zone 1 was not sampled due to time constraints.

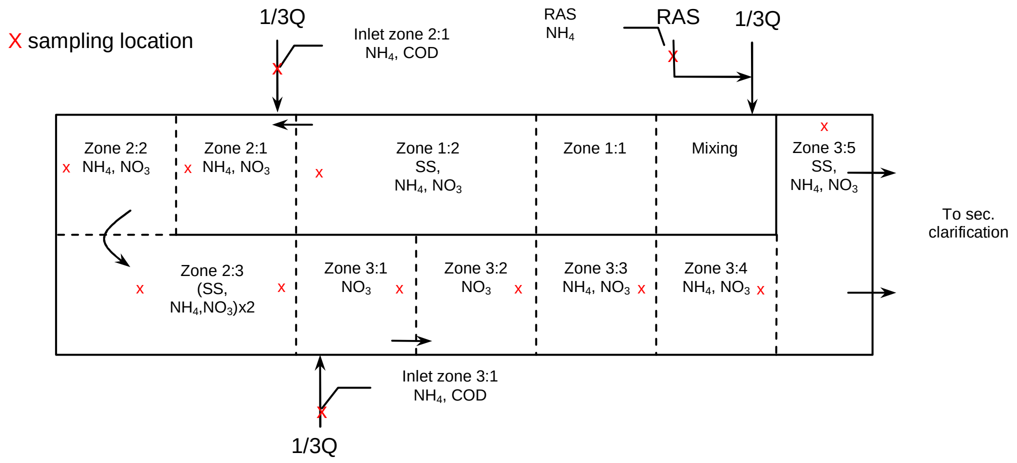


Figure 8. Sampling locations and analyzed parameters of the nitrogen profile. Zone 1:2, 2:3, 3:4 and 3:5 are aerobic.

The nitrogen sample was filtered at the sampling spot. The filtered sample was stored in refrigerator before analysis. Ammonia and nitrate was analyzed with a DR 2010 spectrophotometer during the course of the day. The methods used were the high range Test'N Tube methods from Hach-Lange. The SS and COD sample was analyzed at Uppsala Water Laboratory.

5.3 PRINCIPAL COMPONENT ANALYSIS

5.3.1 Theoretical background

Within the field of multivariate analysis, principal component analysis (PCA) is a projection technique used to reduce the dimensionality of data and find correlations between variables (Jolliffe, 1986). PCA is widely implemented in many different occasions and scientific applications. The technique has been reported valuable as a tool in wastewater treatment monitoring and evaluation (Nilsson *et al.*, 2007; Röttorp *et al.*, 1999).

In a large dataset where several variables are related to each other it could be hard if not impossible to consider every variable of its own and still keep an overview of the whole system. In PCA, the principal components (PCs) are introduced to replace the regular variables and reduce the dimensionality of data (Everitt & Dunn, 2001).

The first PC is a vector that describes as much variation in the data as possible and is created through ordinary least squares approximation. The second PC is orthogonal to the first one and it is directed towards the second largest variation possible. This is continued until there are several PCs which are all independent to each other and which describe the variance of the data. There can only be as many PCs as there are variables in the first place, though nothing is gained if too many PCs are considered. The PCA generates a PC model which is presented in two plots: the loadings and the scores.

Loadings display how the initial variables contribute to the PCs. Each PC has a loading vector with one element per initial variable. The higher the element in the loading vector is, the more does this variable contribute to the PC. This also means that when the loading plot is considered for two PCs, the correlations for the variables in the outer regions of the plot are most important since their dynamics have affected these PCs to a larger extent. If one variable is not contributing to one set of PCs it means that it does not vary in this particular direction. Still, it can have a high loading on another set of PCs.

The loadings are analyzed to find indications of which variables are negatively or positively correlated. If the variables are grouped in the loading plot, they might be correlated. If two variables have an opposite loading vector, they might be negatively correlated.

The scores are the projection of each observation onto the new space which is created by the PCs. If, for example, the original data is three-dimensional and two PCs are used in the model, the observations are projected onto a two-dimensional plane created by the PCs. There is an example of how the score plot is created in 3D in Figure 9.

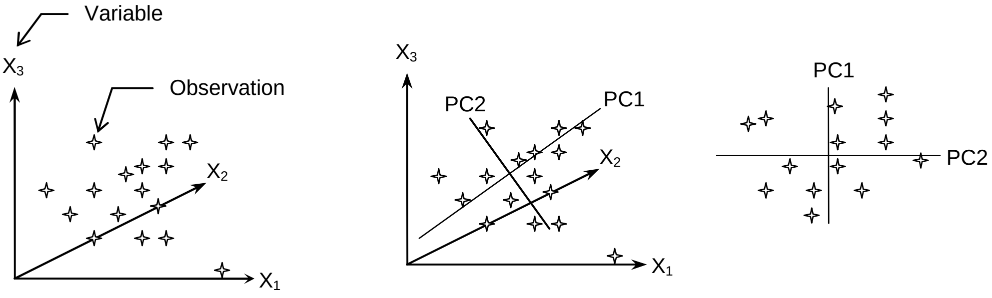


Figure 9. The creation of principal components in three dimensional space.

The score scatter can display trends and relations between observations but should not be analyzed on its own without considering the loading plot. If an observation is projected onto a region of the score scatter that according to the loading plot corresponds to one or several variables in a positive or negative way, this variable is also high or low in that particular observation. The loadings and scores are demonstrated in a biplot in Figure 10.

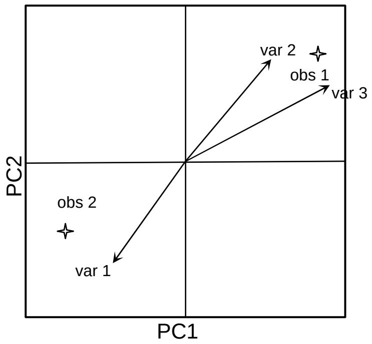


Figure 10. An example demonstrating a biplot of loadings and scores.

In Figure 10, variable 1 is negatively correlated to both variable 2 and 3. Variable 2 and 3 on the other hand are expected to be positively correlated. Observation 1 is high with respect to variable 2 and 3 while it is low with respect to variable 1. The opposite is true for observation 2.

5.3.2 Objective and procedures

In this study, a large amount of data over a long period of time has been available for analysis and several of the variables of interest are related to each other, which makes it suitable to perform a PCA on the dataset. The objective of the PCA is to create a number of PC models and through these discover if there are some unknown relations between the model parameters. The score plots will be employed to study the behaviour of the process over time.

There were two different time series available for PCA out of which three PC models were created: M1, M2 and M3. In M1 and M2, data is present as daily averages during the whole 14 month period. This data was either raw data directly from Uni-View or variables calculated from this data and include SRT, MLSS, and different flows in the biological treatment as well as the incoming flow and outgoing ammonia concentrations. M2 is a more sophisticated version of M1 where some variables are replaced by averages, and the effluent ammonia concentration added.

M3 has only 53 observations since it also incorporates the composite weekly samplings. This model is therefore dealing with the same data as M2 together with influent and effluent characteristics of block C and parameters that could be deduced from these. M3 is more focused on the nitrogen removal than the other two models. In Appendix F, table i, ii and iii, there is a detailed list of the parameters in M1, M2 and M3 with abbreviations.

The PCA was performed in MATLAB 2007a. Before the two datasets could be used in the modelling, autoscaling, which gives the data a standard deviation of one, and zeroing of mean values were performed. This is done so that all variables influence the model to the same extent.

The same limitations considering missing data apply in this evaluation as in the static calculations. Due to this lack of continuity in the data matrix, some parts of the data must be removed. All observations must be left out when data is missing for one or more variables. In software especially made for multivariate analysis, it is possible to have missing values in the dataset. This was not implemented in MATLAB. The nitrate concentration out from block C and the temperature data was not incorporated in M2, since that would have meant a too large period of autumn 2007 would have been left out. The nitrate concentrations during the missing period were very high during this period according to the composite samples. Also, the temperature measurements were left out in M3, since it did not contribute to the understanding and only confused the interpretation of the scores plot. The same applies to the reaction rates R_n and R_d.

The modelling process was iterative, starting with determination of included variables, removal of outliers and calculation of the cumulative coefficient of determination. The maximum number of PCs included in the models was determined through consideration of the eigenvalues of the PCs (the eigenvalue method, for more information see Hair *et al.* (2006)). The PCs not contributing with interesting information was then excluded.

5.4 SIMULATIONS IN JAVA ACTIVATED SLUDGE SIMULATOR

In this part of the project, focus was on gaining more understanding of the nitrogen removal process through calibration and simulation with the Java Activated Sludge Simulator (JASS). In order to use a model, calibration, which in this project is the process when model parameters are adjusted to fit a specific dataset, is the first thing to start with (Petersen *et al.*, 2003). When calibrating ASM1 to the full extent, a lot of information is needed. The purpose of the work and knowledge of the process decide the level of calibration (Petersen, 2000). The data quality from Kungsängen WWTP with mainly composite 24 hour samples served as input data for a static calibration. If a dynamic calibration is to be made, dynamic profiles of the plant is required (Gearney *et al.,* 2004), which is not covered in this study.

There are both physical-chemical and biological procedures for measuring the model parameters in ASM1 (Petersen, 2000). The attempt in this project was however to modify the default parameters until they fitted the calibration data since measurements would have been too time consuming. The model parameters are listed in Appendix B together with their default values. During the calibration procedure these default values were changed in order to fit the model to the plant characteristics, such as effluent concentrations and sludge balance.

Prior to the construction of block C at Kungsängen WWTP, a pilot study was performed on block A. Within the frames of this project, a complete characterization of the wastewater was carried out, as well as a calibration of ASM1 (Xu, 1996). Even though several years passed since then, the values from this calibration was fed into JASS after the attempt with the default values.

First, average values from the period May to December 2007 was used for calibration and spring 2007 was used for validation. The results from the calibration did not permit an extensive evaluation of the plant. However, a comparison was made between an ordinary pre-denitrification process and a process with step-feeding through modifications of the feeding points. Also, instead of a flow distribution of 0.33Q/0.33Q/0.33Q to zone 1 to 3, the distributions 0.4Q/0.4Q/0.2Q and 0.2Q/0.4Q/0.4Q were tested and the effect on the nitrogen removal was evaluated.

6 RESULTS

6.1 RESULTS FROM CALCULATIONS

Influent and effluent characteristics

The results from the static calculations are presented below. First, the surrounding physical factors concerning the influent wastewater to the plant together with the pH, temperature and alkalinity during the whole evaluation period are presented in Figure 11.

Secondly, the chemical characteristics of nitrogen and COD together with the SS concentration of the incoming and outgoing wastewater and the removal efficiency are presented in Figure 12. There is an overview of all the average influent and effluent characteristics together with process parameters in Appendix D, table i.

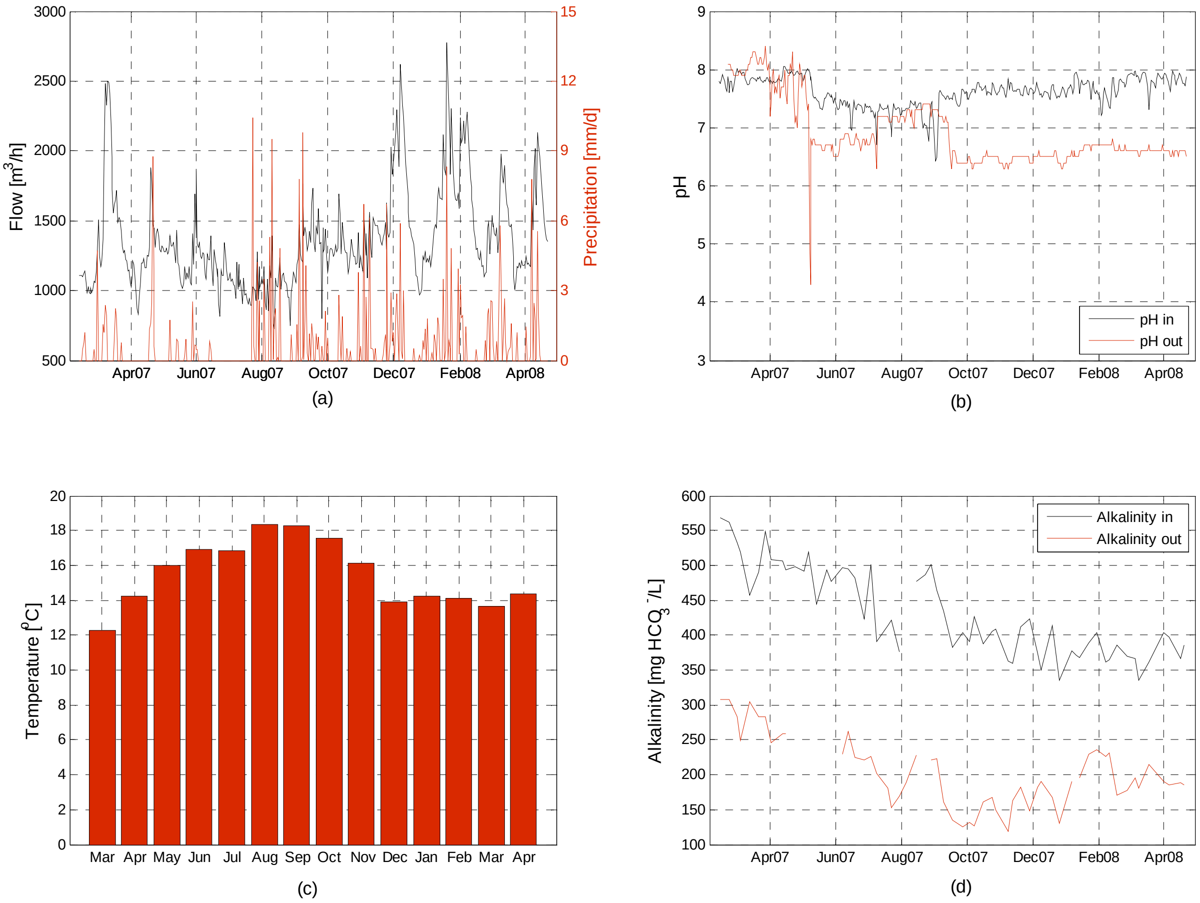


Figure 11. Physical and chemical characteristics of the incoming wastewater to block C during the evaluation period. (a) Flow from block C and precipitation at the plant. (b) Incoming pH to the plant and outgoing pH from block C. (c) Average temperatures of incoming wastewater. (d) Incoming and outgoing alkalinity.

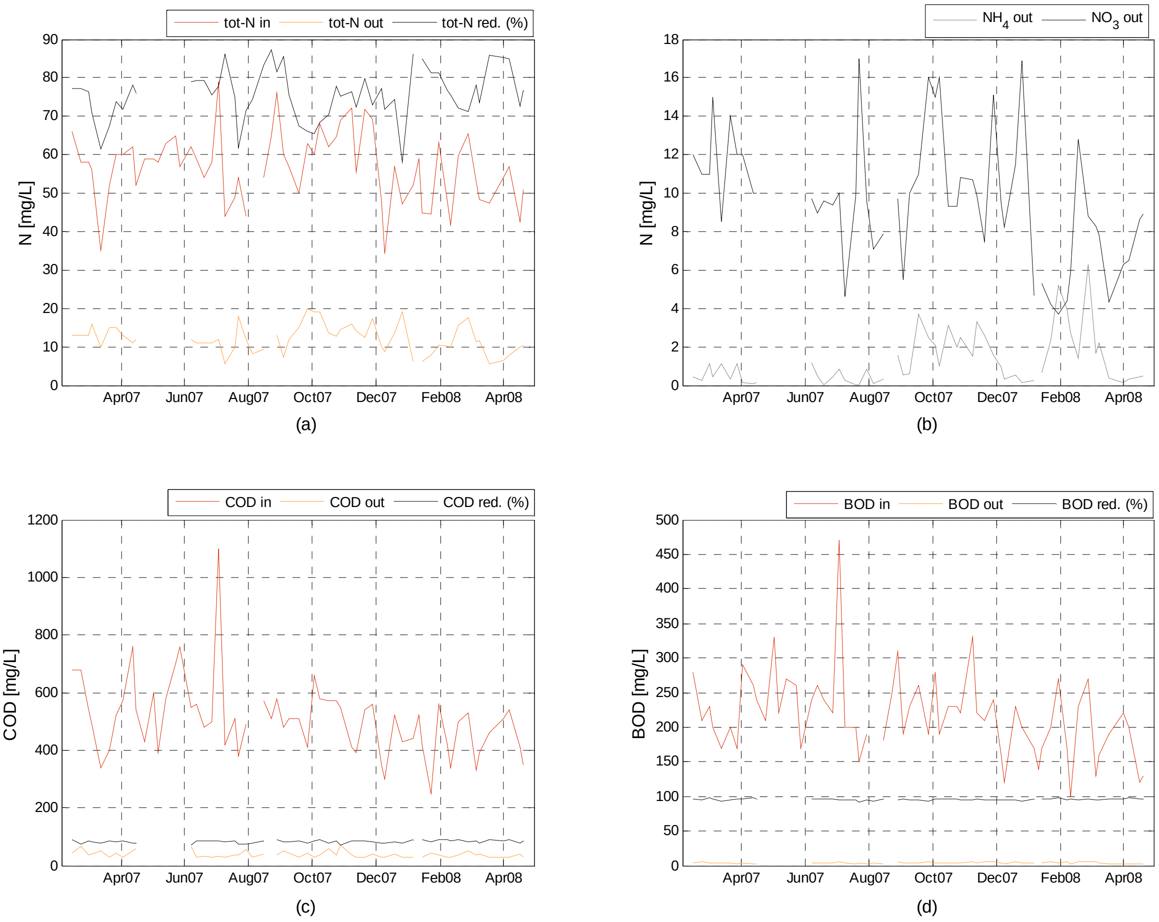


Figure 12. Concentrations of pollutants at block C during the evaluation period: (a) Incoming and outgoing concentrations and reduction of total nitrogen, (b) outgoing NO_3 and NH_4, (c) incoming and outgoing concentrations and reduction of COD and (d) incoming and outgoing concentrations and reduction of BOD.

Treatment effect

The average total treatment efficiency for nitrogen and organic matter is presented in Table 4, together with average outgoing concentrations of the same pollutants. There was a 12 % nitrogen removal through assimilation, incorporated in the total nitrogen removal of 78 %. In spring 2008 during hydrolysis, the same value was 17 %. The average COD consumed to nitrogen denitrified ($\Delta COD/N_d$) was 6.5.

Table 4. Incoming and outgoing concentrations of COD, BOD, tot-N, NH_4-N and NO_3-N together with total removal efficiency and removal efficiency over the biological step of block C for the period February 12 2007 to May 22 2008.

Substance	***Incoming (mg/L)***	***Primary effluent (mg/L)***	***Effluent (mg/L)***	***Total removal (%)***	***Bio step removal (%)***
COD	507	239	39	92	83
BOD	217	95	4	98	96
Tot-N	57	50	12	78	76
NH_4-N			1.3		
NO_3-N			9.7		

Process parameters

The different process parameters for block C are presented in Table 5 and, in more detail, in Appendix D, Table i.

Table 5. Average values of wastewater treatment process parameters during the evaluation period (February 12 2007 to May 22 2008). The volumetric loads are given for the biological step.

Process parameter	***Unit***	***Average***	***Max***	***Min***
HRT	h	8.9	15.3	4.4
Sludge age (SRT)	d	9.8	24	4.2
Temperature	℃	15.5	19.3	10.5
F/M	kg BOD_7/kg VSS, d	0.07	0.14	0.03
Volumetric load	kg BOD_7/m^3, d	0.27	0.55	0.09
Volumetric nitrogen load	kg N/m^3, d	0.14	0.20	0.06
Yield	kg SS/kg BOD_7	0.5	2.7	-0.4
Nitrification rate	mg N/g VSS, h	1.8	3.0	1.0
Denitrification rate	mg N/g VSS, h	2.2	3.6	0.8

The temperature compensated aerobic sludge age is about 4 d during a longer period in January and February 2008 (Figure 13). The highest temperature compensated aerobic sludge age of 25 d is measured during the larger part of August 2007.

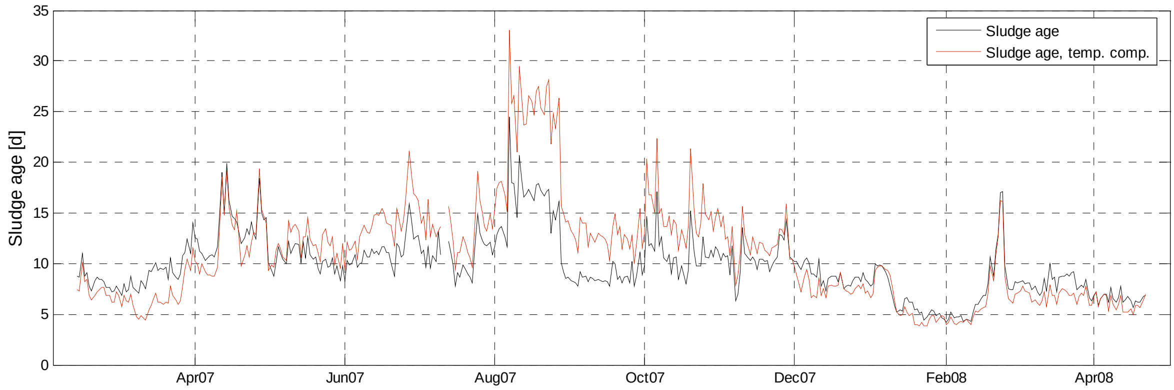


Figure 13. Aerobic sludge age and temperature compensated aerobic sludge age in block C during the evaluation period.

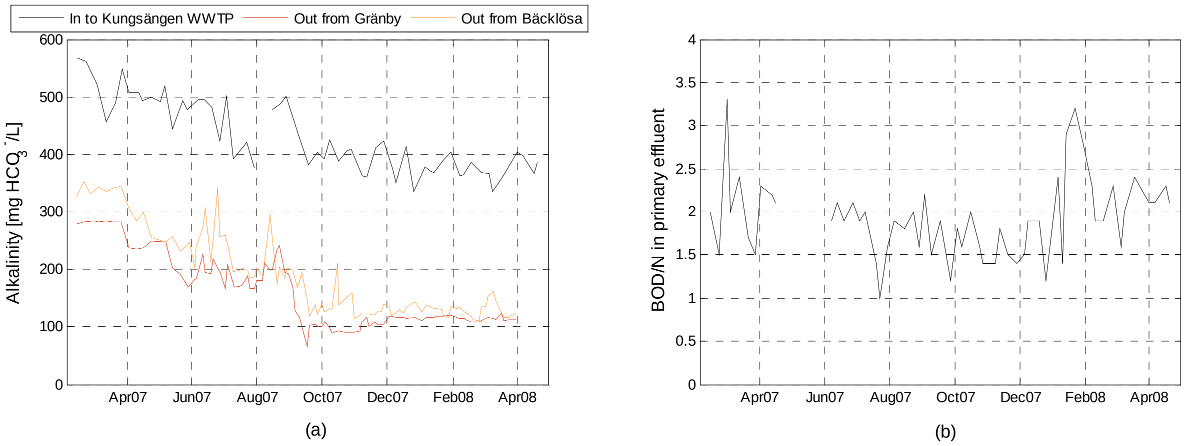
External and internal factors

During the period there have been changes in both the operation of block C and in the prerequisites for the plant operation connected to the incoming wastewater.

In Appendix D, Table ii, there is a list of different process parameters and removal rates for different periods of the evaluation period, covering the differences of the summer and winter operation, the reject water flow, the hydrolysis in the primary clarification and the values before and after the effect of the decreased alkalinity entering block C. The winter and summer periods (November–May and May–November) are the break points for different aeration regimes. The days used in the calculation when there was less reject water added to block C are days when reject water was added less than half of the time of the day.

The major difference concerning the incoming wastewater characteristics during the period is the change in alkalinity due to the introduction of two new water treatment facilities in Uppsala city. The incoming alkalinity to the block is again presented below, this time together with the alkalinity coming out from the water treatment plants Bäcklösa and Gränby (Figure 14(a)).

The change of the BOD/N in primary effluent is found in Figure 14(b), to display the effect of hydrolysis in primary clarification.



(a)

(b)

Figure 14. (a) Incoming alkalinity to block C and outgoing alkalinity from Bäcklösa and Gränby water plants. (b) BOD/N in primary effluent in block C.

The effect of the wastewater treatment plant at GE Healthcare that is expected to reduce the readily biodegradable material to the primary clarification, see Section 4.1.1, cannot be detected from this data since the change took place in 2005. The reduction of BOD over the primary clarification can be seen in Table 4.

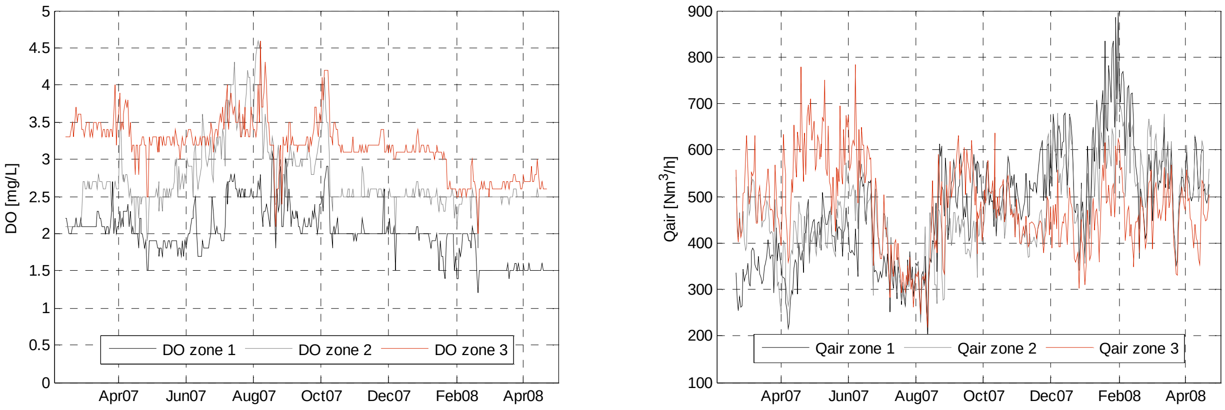
There is also a list of changed influent characteristics assuming that no by-passing was in operation. The first winter period in 2007 there was a pump in operation. During summer 2007, no by-pass was added to the aeration basins, and in winter 2008 there was an open gate feeding the by-pass water. This gate was open from December 27. Since the exact by-passed flow was unknown, the calculations for winter 2008 are based on the assumption of 120 m^3/h. A sensitivity analysis of the size of the flow shows a 1.5 % increase of BOD to the aeration basin with a 25 % increase in by-passed flow.

Flows and MLSS concentration

On average, the plant has treated 32 800 m^3/d and no overflow at the plant has occurred during the evaluation period. The water to block C enters the plant via Danmark pumping station. The inflow has a maximum during weekdays at 12 a.m. and a somewhat smaller peak at 8 p.m. In weekends, the corresponding flow peaks are found at 9 a.m. and 9 p.m. The daily variations of MLSS are inversely proportional to these daily variations in inflow. The average MLSS concentration during the whole evaluation period is 5100 mg/L with higher values in spring 2008. The average MLSS in zone one, two and three are 6000 mg/L, 5200 mg/L and 4500 mg/L respectively. The return activated sludge was in average 180 percent of the inflow and the waste activated sludge flow was 19 m^3/h, all zones included.

Aeration

The dissolved oxygen (DO) concentrations and air flow are found in Figure 15. The irregularities of the oxygen concentrations in spring 2007 in zone 2 were primarily pronounced in zone 5. The oxygen set point is higher in the later zones. The set point in zone 1 has ranged between 2.5 and 1.5 mg/L, in zone 2 between 2 and 3 mg/L and in zone 3 between 2.5 and 3 mg/L. The average concentrations have been 2 mg/L, 2.7 mg/L and 3.2 mg/L respectively. In 2007, it is unknown when the aeration changed to summer operation. In the calculations it is estimated to be May 1. The winter operation was initiated in 2007 in two steps: November 23 and 27 (Jidetorp, 2008, pers. comm.).



(a) (b)

Figure 15. (a) Average dissolved oxygen concentrations in zone 1 to 3 and (b) air flow rate during the evaluation period.

Reject water

The average total nitrogen concentration of the reject water during the evaluation period was 984 mg/L and the flow is estimated to be about 10 m^3/h (Jidetorp, 2008, pers. comm.), contributing with an average load of about 410 kg/d which is equivalent to about 13 % of the total load. The highest concentrations were found during spring 2008. One out of two centrifuges is active in periods of about 2–10 days with possible stops in between. When the centrifuges are active, the resulting reject water is pumped to Danmark pumping station. The most common situation during a 24-hour period was that the reject pumps were on 75 % of the time.

Pre-precipitation and primary clarification

On average, 83 L/h and 117 ml/m^3 of iron chloride was dosed in the pre-precipitation process. The later was somewhat lower from September 2007. The dose was lower during spring 2008 with 63 ml/m^3 iron chloride. Over the primary treatment step, including pre-precipitation with grit chamber and primary clarification, there was a 56 % removal of COD and a 52 % removal of BOD respectively. The nitrogen reduction was in average 11 %, although it was quite irregular, and there was a 72 % removal of phosphorus.

Secondary clarification

The secondary sedimentation step receives MLSS with a surface load of 0.4 m/h and a solids loading rate of 4.6 kg SS/m^2, h when calculated according to equation 8. If the solids loading rate is calculated with RAS excluded, the load is 1.7 kg SS/m^2, h. The solids loading rate to the clarifier has increased during the evaluation period, following the same trend as the MLSS concentration.

Other wastewater treatment plants

Data from the other Swedish treatment plants is found in Appendix G. All data concerning influent and effluent concentrations is from 2007. In Figure 16, different relations between effluent nitrogen and plant prerequisites are presented. The volumetric nitrogen load was used as a measure to quantify the nitrogen load to plants with different sizes. Block C at Kungsängen WWTP is depicted as Uppsala in Figure 16. The effluent concentrations for Uppsala are from the secondary effluent from block C.

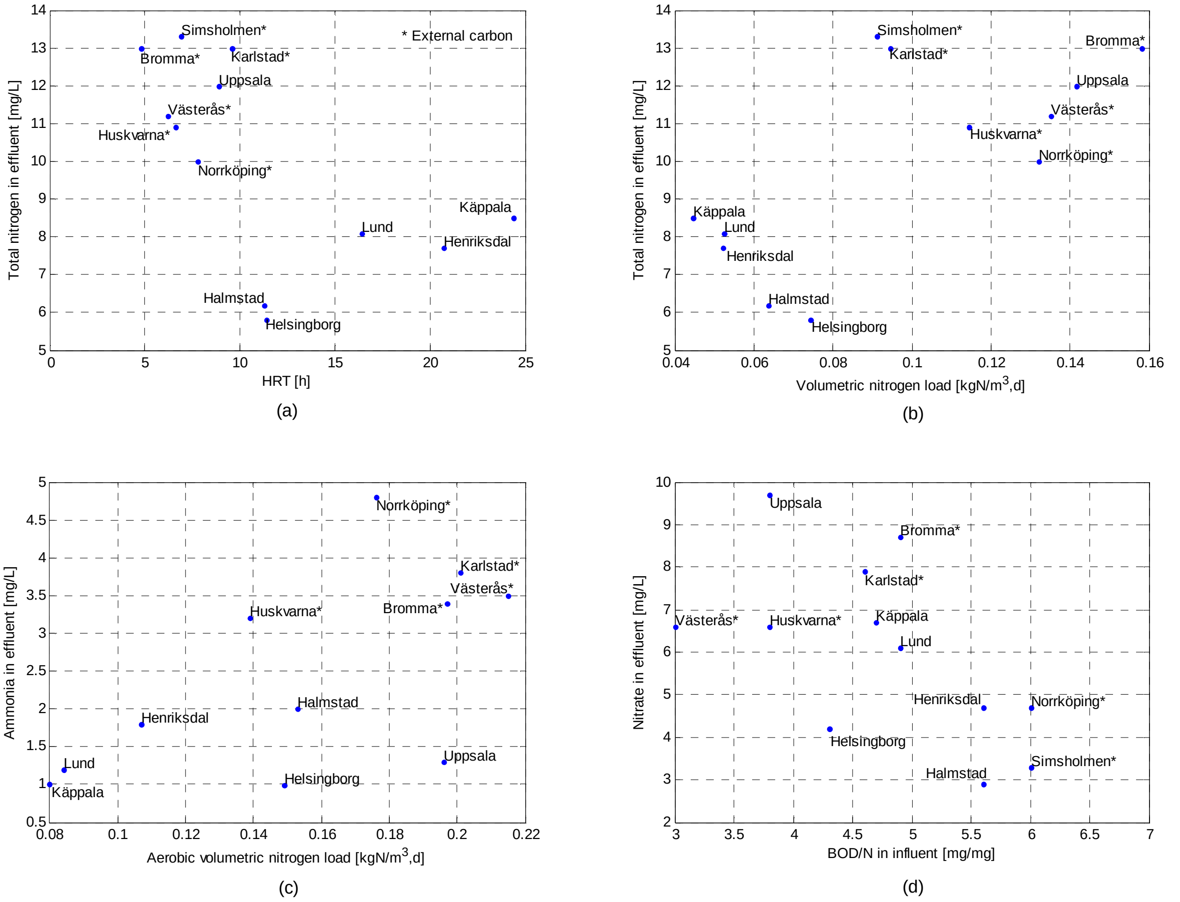


Figure 16. (a) Effluent total nitrogen in relation to the bio reactor HRT. (b) Effluent total nitrogen as a function of the incoming volumetric nitrogen load. (c) Effluent ammonia as a function of the incoming aerobic volumetric nitrogen load, based on the maximum aerobic volume. Simsholmen was an outlier. (d) BOD/N in influent and the nitrate in effluent. The star marks the plants using external carbon.

6.2 MASS BALANCE ANALYSIS

The overall mass balance analysis performed on May 22 2008 on the C-block is presented in Figure 17. The average temperature of the incoming wastewater was 17.7 ºC and the average inflow was 1049 m^3/h. The aerobic sludge age was particularly low this day, only 4 d, due to a high WAS flow. The average aerobic sludge age during the spring period was 11 d.

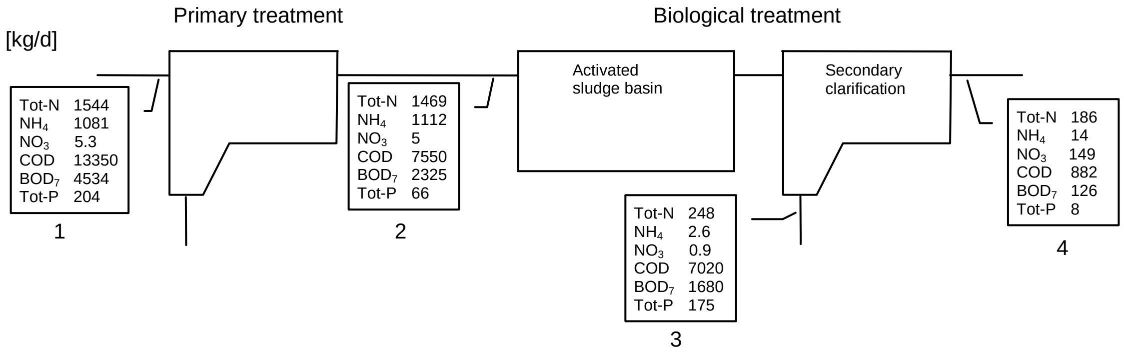


Figure 17. Mass balance over block C from May 22 2008. Unit kg/d.

The mass balance indicates that during this day, 1207 kg were nitrified from NH_4 to NO_3 and 1035 kg were denitrified from NO_3 to N_2(g). The balance with respect to organic matter should be positive since it is partly oxidized to carbon dioxide. When considering COD, a balance over the biological treatment results in a negative value of -352 kg and when considering BOD_7 it is 519 kg. However, there is a large error when combining all measurements, and hence it is not possible to draw any conclusions about the production of carbon dioxide.

The calculated nitrification rate was 1.8 mg N/g VSS, h and the denitrification rate was 1.6 mg N/g VSS, h at 17.7 °C, which after temperature correction with eq. 9 is equivalent to 1.4 and 1.3 mg N/g VSS, h at 15 °C respectively. These reaction rates are calculated based on the extra measurements performed for the mass balance. When calculating the rates with the information available from the ordinary composite samples, the rates are 1.7 and 1.5 mg N/g VSS, h respectively at 17.7 °C.

The COD/N ratio in influent was during this day 8.6 and the BOD_7/N and COD/BOD_7 ratios were both 2.9. The pH and alkalinity leaving block C was 6.6 and 187 mg HCO_3^-/L respectively, which is quite normal for the period. The reduction of nitrogen, BOD_7, COD and phosphorus over the biological treatment step was 87 %, 95 %, 88 % and 88 % respectively, while the total reduction including primary clarification was 88 %, 93 %, 97 % and 96 %.

The measurement that was supposed to measure the incoming flow to the aeration basin after by-passed water was added (sample 2) did not turn out well. The sampling indicated too high values in all substances compared to the sample after primary clarification. In the above measurements, the concentrations at the bio reactor inflow have been derived from the samples of the influent and of the primary effluent.

6.3 NITROGEN PROFILE

The result from the construction of a nitrogen profile is depicted in Figure 18.

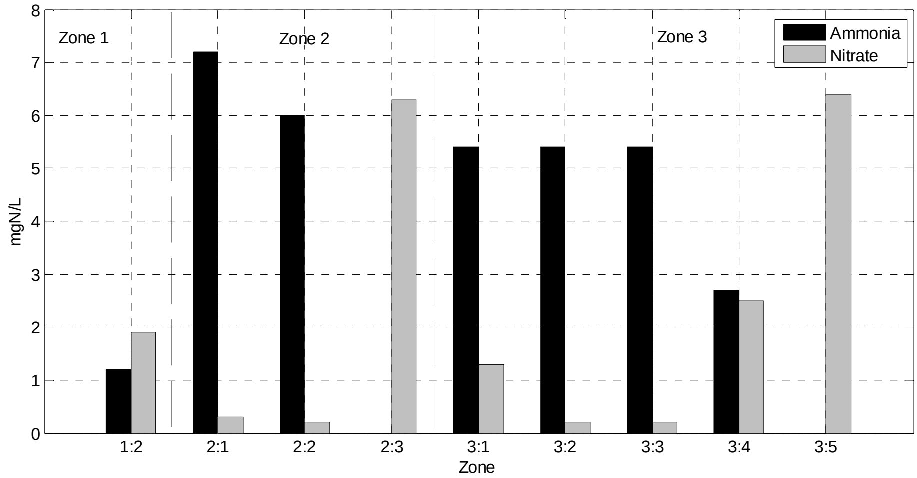


Figure 18. Nitrogen profile. Zone 1:2, 2:3, 3:4 and 3:5 are aerobic. The ammonia concentration in zone 3:1 and 3:2 are not measured but assumed to be the same as in zone 3:3. The first sample after half of zone 2:3 was 5.8 mg NH_4-N/L and 0 mg NO_3-N/L.

Unfortunately, the attempt to follow a plug through all the basins did not succeed. This was due to an error in the flow calculations underestimating the retention time of the zones. The first samples in zone 1:2 and zone 2:1 are hence expected to be underestimated compared to the other samples. The other samples can be considered relatively correct. From zone 2:2 and onward, the flow was stable between 1100 and 1200 m^3/h and the nitrogen load was almost constant during the rest of the sampling. The incoming ammonia concentration was about the same in the morning and afternoon (48.5 and 47.8 mg/L respectively. The effluent concentrations of nitrate and ammonia were also almost constant. The concentrations were 0.01–0.05 mg NH_4-N/L and 1.7–2.9 mg NO_3-N/L according to the instruments logging the effluent of block C.

The nitrification and denitrification rates in the different zones are presented in Table 6. The COD/N ratio was 8.7 in the morning and 6.3 in the afternoon. The average flow was 885 m^3/h and the aerobic sludge age was 7.2 d in line 3. The incoming and outgoing pH was 7.7 and 6.8 respectively.

Table 6. Nitrification and denitrification rates from the nitrogen profile. The rate in zone 2:3 and 3:5 are based on half of the actual zone volume. Unit mg N/g VSS, h.

Zone	*R_n at 15 (19) ℃*	*R_d at 15 (19) ℃*
2:1	-	0.6 (0.9)
2:2	-	0
2:3	3.0 (4.5)	-
3:1	-	2.3 (3.4)
3:2	-	0.6 (0.9)
3:3	-	0
3:4	1.3 (2.1)	-
3:5	1.1 (1.7)	-

6.4 PC MODELS

The resulting PC models are presented in Figure 19 and Figure 20. Model 1 (M1) is not presented graphically. The loadings of M1 indicated several correlated groups: MLSS in line 1 to 5, dissolved oxygen (DO) in zone 1 to 3, waste activated sludge (WAS) for all lines, return activated sludge for line 3, 4 and 5 and SS in RAS for all lines. The second model, M2, is presented with both loadings and scores in Figure 19.

When interpreting the loadings of M2, several correlations of importance are notable:

Positive correlations:

- NH_4 out, reject water flow to block C and precipitation from block C
- SS RAS, WAS and SS load

Negative correlations:

- SRT with WAS
- NH_4 out and pH out
- DO with air flow

Looking at the scores of M2, three major periods of operation can be distinguished. February to April 2007 is mainly characterized by the high pH in effluent, low ammonia in effluent, a small amount of precipitation and low flows. From May to October 2007 the observations at many times overlay each other, indicating similarity with respect to the parameters in the model. The same applies for the period November 2007 to April 2008, which during some periods shows the opposite characteristics than the period February to May 2007. Model M2 consists of principal components 1, 2 and 3 which all together describe 59 % of the variations of the data.

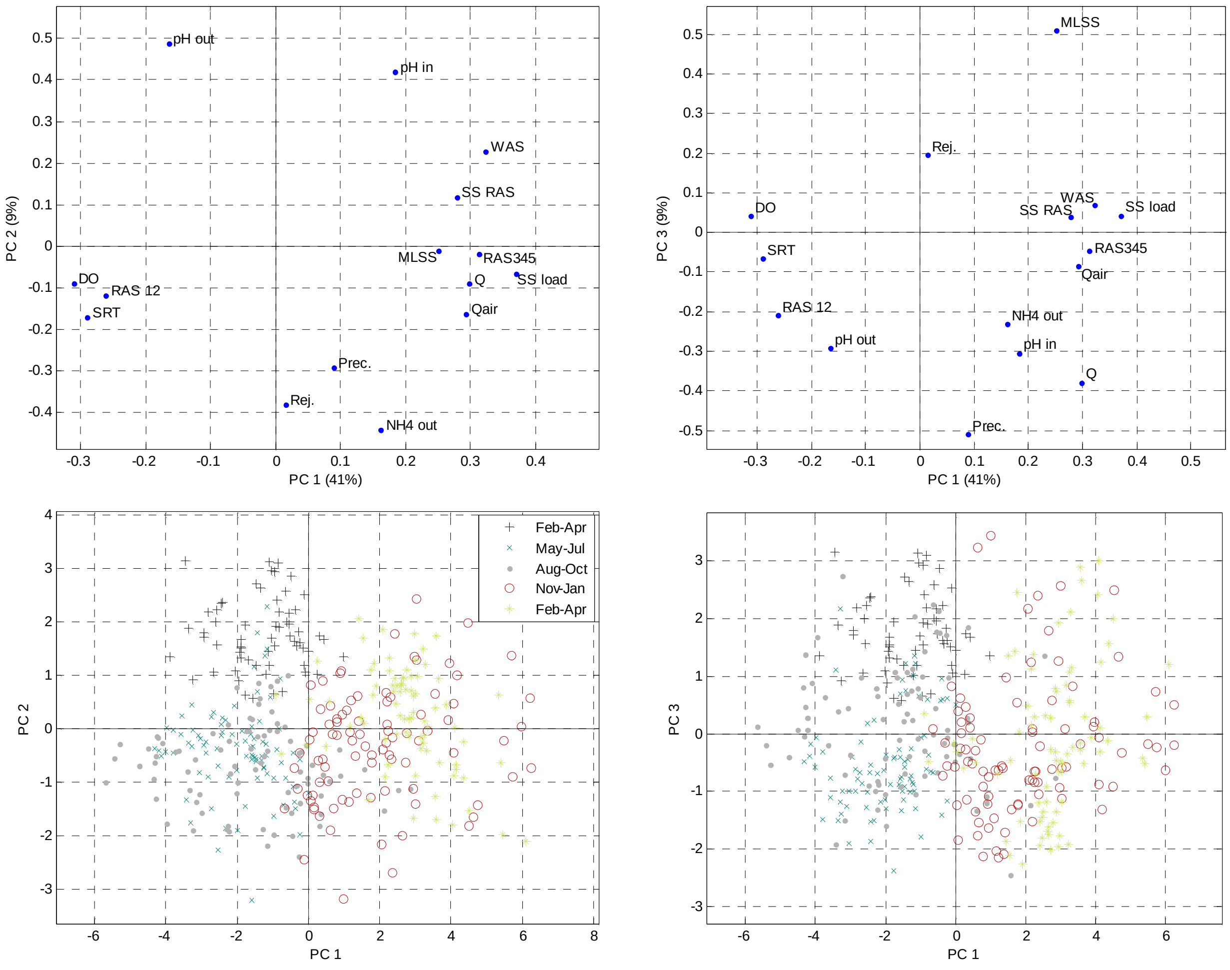


Figure 19. Loadings and scores for PC1, PC2 and PC3 for M2.

Figure 20 depicts the same plots as Figure 19, but for M3. M3 involves several more parameters and substantially less observations. The model describes 63 % of the variations in data.

Positive correlations:

- Tot-N out with tot-N in and Alkalinity in
- Air flow with F/M ratio and BOD in
- Nitrogen reduction, COD/N, MLSS and hydrolysis in prim. clarification
- Alkalinity in and pH out

Negative correlations:

- NO_3 out with F/M and BOD in
- NO_3 and tot-N out with COD/N and hydrolysis
- DO with air flow

The difference between different periods in time is not as pronounced in M3 as in M2 when the scores are considered, but are still recognizable. Again, the highest ammonia concentrations in the effluent are found during the period November 2007 to April 2008.

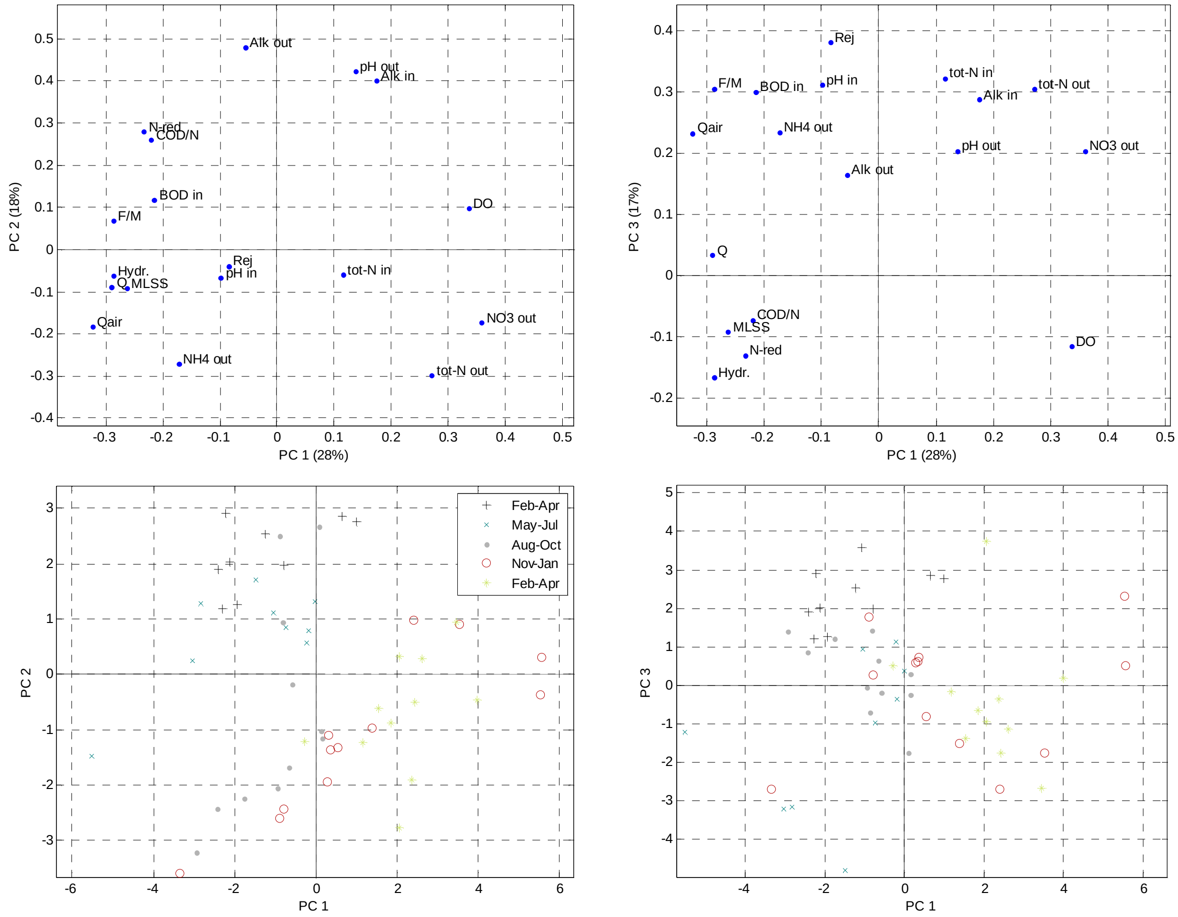


Figure 20. Loadings and scores for PC1, PC2 and PC3 for M3.

6.5 SIMULATIONS IN JASS

The default parameters and the calibrated values of the parameters are presented in Table 7. Without changing the model parameters, the outgoing nitrate concentrations were up to 30 mg/L in some simulations. The simulated system appeared very carbon-limited and could not denitrify properly. That is why the model parameters changed in the calibration procedure were mainly parameters affecting the denitrification.

When trying to boost performance of the model, some changes were made in the input data. The oxygen concentrations in the model were set 0.5 mg/L lower than the measured value in order to simulate incomplete aeration in the basins (Rosén, 2008). Also, the fraction of particulate biodegradable material, X_s, was increased compared to the values in (Xu, 1996), since this value is often underestimated in measurements. For the complete list of characteristics of the primary effluent incorporated in JASS is found in Table 8.

Table 7. Model parameters in ASM1: Default values and calibrated values at 20 °C.

Symbol	*Default (20 ºC)*	*Calibrated (20 ºC)*	*Unit*
Y_H	0.67		g cell COD formed/g COD oxidized
Y_A	0.24		g cell COD formed/g N oxidized
f_p	0.08		-
i_{XB}	0.086		g N/g COD in biomass
i_{XP}	0.06		g N/g COD in endogenous mass
μ_H	6.0		d^{-1}
b_H	0.62		d^{-1}
K_S	20		g COD/m^3
$K_{O,H}$	0.20	1.0	g O_2/m^3
K_{NO}	0.50		g NO_3-N/m^3
μ_A	0.80		d^{-1}
b_A	0.20		d^{-1}
$K_{O,A}$	0.4	0.5	g O_2/m^3
K_{NH}	1.0		g NH_3-N/m^3
η_g	0.8		-
k_a	0.08		m^3/g COD, d
k_h	3.0		g slowly biodeg. COD/g cell COD, d
K_X	0.03	0.01	g slowly biodeg. COD/g cell COD
η_h	0.4		-

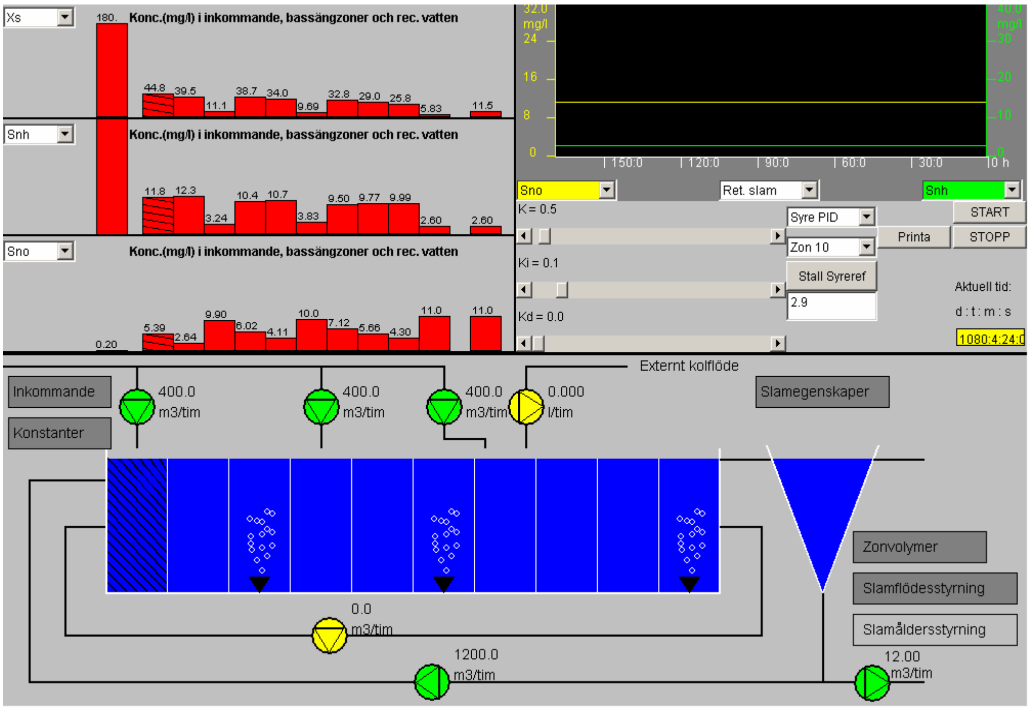
Table 8. Concentrations in primary effluent during model calibration in JASS.

Constituent	*Concentration (mg/L)*	*Fraction of total COD or N (%)*
Xbh	11,3	5,0
Xba	0	0
Xs	180	80
Xi	15,8	7,0
Xnd	8,8	15
Snh	40	77
Snd	3,6	7,0
Sno	0,2	0,4
Si	9,0	4,0
Ss	9,0	4,0
Tot-N	51,6	100
COD	226	100

The results from the simulation are presented in Figure 21. The wastewater characteristics and operation with respect to flows and aeration are that of summer 2007, see Appendix D. The only deviation from this data is the RAS and WAS flows. The very high RAS flow gave rise to a particularly high concentration of nitrate in the first reactor, which is not the case in reality. Therefore, the RAS flow was reduced to 100 % of Q_{in} and the WAS flow was somewhat lower to maintain a high MLSS concentration.

The scenario in Figure 21 was first evaluated without changing the default parameters. This resulted in nitrate in the effluent of 24.2 mg/L. To obtain nitrate values around 11 mg/L in the effluent, the influent nitrogen had to be reduced to about 25–30 mg/L. The slowly biodegradable organic material, X_S, could be increased to improve denitrification, but had to reach about 300 mg/L to obtain sensible nitrate concentrations.

The simulation below was very unstable with respect to denitrification. This made it very difficult to verify, since a small change in influent COD or ammonia had a direct effect on the effluent nitrate concentration. It was not possible to obtain results such as below when trying to simulate the operation during wintertime without changing the rate of hydrolysis, k_h.



K = 0.5
Ki = 0.1
Kd = 0.0
2.9
Aktuell tid:
d : t : m : s
1080:4:24:0

Figure 21. Results from simulations of the summer period 2007 in JASS. The first diagram in the upper left corner depicts the particulate organic matter, the second the ammonia concentration and the third the nitrate concentration.

When increasing the ammonia in the influent from 40 to 45 mg/L, the effluent nitrate concentration increased from 11 mg/L 18 mg/L. This could be compensated by addition of approximately 30 LCOD/h of external carbon source. When adding carbon source, the system was less sensitive to changes in the ammonia influent level.

The process was changed to a pre-denitrification process through feeding all influent to the first zone, maintaining the same total anoxic and aerobic volumes and keeping oxygen concentrations of 2 mg/L in aerobic zones. The outgoing nitrate concentration was 13.8 mg/L, the ammonia concentration was 2.8 mg/L and the X_S in the effluent was 6.7 mg/L. RAS flow was 100% of Q and the MLSS recycle was 200 % of Q. MLSS was about 4000 mg/L.

The results from the changes in flow distribution indicate that the optimum operation is when there is an equal flow splitting between the zones. When 0.4/0.4/0.2 was used, the outgoing nitrate concentration is increased from 11 to 12 mg/L, and the outgoing X_S concentration is reduced to 9.4 mg/L instead of 11.5 mg/L in Figure 21. A distribution to the zones of 0.2/0.4/0.4 increased the outgoing ammonia concentration from 2.6 to 4.6 mg/L.

7 DISCUSSION

In this section the plant is first described in relation to literature on plant operation and influent characteristics in general. Secondly, the nitrification and denitrification is covered in more detail. Thirdly, comments on the process design compared to the actual operation are made and block C is also compared to other plants with predenitrification in Sweden. The discussion is ended with comments on some of the methods in the study.

7.1 OVERALL OPERATING RESULTS

The influent total nitrogen concentration to block C is, based on typical contents in domestic wastewater, moderate to concentrated (Henze *et al.*, 2002). The reject water from the sludge dewatering is however included in the nitrogen measurements and the nitrogen content originating from the domestic wastewater can hence be considered moderate. The concentration of nitrogen in the reject water is according to Henze *et al.* (2002) high.

The carbon source entering the block is with respect to COD moderate but low in spring 2008. The biologically available BOD_7 concentration on the other hand is diluted to moderate, also decreasing in spring 2008. The fraction of COD to BOD is according to Appendix D, Table i, 2.4 which is quite typical even though it at times can reach above 4, indicating that the carbonaceous matter is not so easily biodegraded. The fraction of COD to total nitrogen of 8.9 is quite typical for domestic wastewaters, although the biologically available equivalent is low.

The suspended solids concentration in C, with an average of 5100 mg/L, is regarded concentrated (Henze *et al.*, 2002). Even the lowest concentration along the lines, which is due to the dilution with influent water found in zone three, is concentrated. Compared to design parameters for the step-feed activated sludge process listed in Metcalf & Eddy (1991), the process in block C has a higher MLSS (5000 compared to 2000–3500 mg/L). The process is also operated with a higher HRT and the recirculation ratio of RAS to influent is 1.8 compared to suggested 0.25 to 0.75.

The aerobic sludge age is within the range where it creates a stable effluent and a sludge with good settling characteristics (Metcalf & Eddy, 1991). The average temperature compensated value of 11 d is about what is needed not to wash out the nitrifying bacteria (Gerardi, 2002), see Section 7.2 below for more comments on the sludge age.

The F/M ratio, also referred to as the sludge load, is in average 0.07 kg BOD_7/kg VSS, d and within the range of a low loaded plant which is needed for nitrification (Metcalf & Eddy, 1991; Svenska kommunförbundet, 1996). Low loaded plants have a lower yield, i.e., a lower sludge production, which is often the intention. The yield in block C is calculated to be 0.5 kg SS/kg BOD which is within expected limits for this specific F/M according to literature (Henze *et al.*, 2002).

A high F/M ratio or higher incoming BOD_7 is causing a larger need for air flow and decreases the DO concentration according to PC model number 3 (M3, Figure 20). Also, it appears that the air flow is increased by the hydrolysis in primary clarification, which can be expected.

Considering the solids loading to the secondary clarifier, in average 4.6 kg SS/m^2, h, it is within the suggested design value of 4–6 kg SS/m^2, h (Metcalf & Eddy, 1991). The solids loading is quite low when RAS is excluded. According to VAI VA-Projekt AB (1999) the solids loading should be 3.4 kg SS/m^2, h at design flow if sedimentation depth is larger than 4 m. The surface load of 0.4 m/h is below the commonly mentioned value of 1 m/h for secondary clarifiers (Svenska kommunförbundet, 1996; Wanner, 1994).

The results at Kungsängen WWTP are comparable with earlier research on step-feed. Schlegel, (1992) reported an over 80 % nitrogen removal in a two-stage process. Kayser *et al.* (1992) describe a process that in many ways resemble the process in block C. It is a three-stage process with the same F/M ratio and nearly the same MLSS concentration. The influent water in the German plant is thicker in concentration and has a higher influent BOD/TKN ratio of 5. The nitrogen removal efficiency varies between 83–92 %.

7.2 NITRIFICATION

The most important environmental factors which determine the rate of nitrification are apart from available inorganic substrate the temperature, oxygen concentration, pH and toxic substances (Henze *et al.*, 2002).

Considering the temperature, the autotrophic nitrifiers have a nearly 10 % increase in growth rate, μ_a, per 1 ºC (Gerardi, 2002). The nitrification rate in block C is, without corrections for temperature, slightly higher during the summer season compared to the two winter seasons. However this increase during summer can be covered by the error in the calculation, and the nitrification rate calculated from measurements in plant operation is known to be misleading due to the unknown actual nitrification volume (Naturvårdsverket, 1991). Naturvårdsverket (1991) also states that the nitrification rate should not be calculated when the effluent ammonia is below 2 mg/L. This only applied to 14 out of 53 samples, and would lead to an average nitrification rate of 1.9 mg N/g VSS, instead of 1.7 mg N/g VSS at 15 ºC. Both the average of 1.7 or the modified value of 1.9 mg N/g VSS, h are low when comparing with Swedish plants in Naturvårdsverket (1991) and when comparing with plants with predenitrification in this study (Section 7.5). The nitrogen profile resulted in a maximum nitrification rate of 3.0 mg N/g VSS, h at 15 ºC, assuming only half of the aerobic volume was used for nitrification (see Section 6.3). Since the nitrogen removal was particularly high this day, this value might be considered the maximum rate at the block.

In order to nitrify efficiently during cold weather periods there has to be an increase in sludge age to compensate for the low growth rate. The sludge age has in general to be 2–3 times the generation time of nitrifiers which is 2–3 days (Gerardi, 2002). Because of this, a sludge age of 10 d and more is usually needed for nitrification which is fulfilled at block C apart from in spring 2007 and 2008 (Figure 13). There is no trend against higher sludge age in the winter period at block C. Instead, the shortest sludge age is measured during spring time when water temperatures are low. During 4 weeks with an aerobic temperature compensated sludge age of about 4 d in January and February 2008, there is a large increase of ammonia in secondary effluent from block C.

There is no seasonal variation of MLSS inventory at block C. Instead it is, as mentioned above, very high at all times. The goal is to keep a MLSS concentration in zone 3 of 4000–5000 mg/L (Jidetorp, 2008, pers. comm.). The MLSS favouring nitrifying bacteria are according to Gerardi (2002) 2000 mg/L. A high concentration of bacteria can offset the reduced growth rate when temperatures are low, and the sensitivity to toxic substances is lessened.

The oxygen concentration may not be lower than 0.5 mg/L, and should preferably be between 1.9–2.9 mg/L to allow nitrification (Gerardi, 2002). However, this may vary a lot between different plants depending on the specific oxygen uptake rate (Grady *et al.*, 1999). The personnel at Kungsängen WWTP reduced the oxygen set point in spring 2008 to 2.5 mg/L in zone 1 and 2 and to 1.5 mg/L in zone 1 (Figure 15(a)). Since they found this accomplishable without diminishing the treatment effect, this was considered a better choice because of the possible energy savings.

In spring 2008, there was a smaller aerobic volume in block C than the previous winters (Table 2). This can be one of the reasons for the higher effluent ammonia concentrations. Also, the increase of inflow to the last zone from January to March 2008 might be expected to have a negative effect on the ammonia removal, both according to the simulations in JASS and according to literature (deBarbadillo *et al.*, 2002; Tang *et al.*, 2007). The total nitrogen concentration is despite this lower and less aeration is as mentioned economically beneficial. It should also be mentioned that the temperature during spring 2008 was higher than during winter 2007 by about 1 ºC.

Low pH values are inhibiting to nitrifying bacteria and alkalinity is consumed in the nitrification process (Gerardi, 2002). The consumption of alkalinity is caused by destruction by nitrous oxide which is produced when ammonia is oxidized to nitrate. When there is an alkalinity deficiency, the pH will go down inhibiting the enzymatic activity of the nitrifiers. Below pH 6.7 there is a decrease in the nitrification activity. The nitrification rate is approximately constant at a pH range of 7.2–8, which is the interval for the outgoing pH from the water leaving block C in spring 2007. After the introduction of the new water plants the outgoing pH has declined a lot to a level around 6.5, which according to literature should be too low and contribute to a 40–60 % decrease in nitrification rate compared to pH at 7.5 (Henze *et al.*, 2002). In addition, the pH in the floc can be expected to be even lower than in the liquid, due to the consumption of alkalinity by nitrification. Figure 12 and Appendix D, Table ii indicates that there are lower ammonia concentrations out from block C when the outgoing pH value is high. This is also the case in M3, which also correlates the low ammonia values to the low outgoing pH, as seen in Figure 11.

It should be noted that the actual effect of the pH on the nitrification might not be reduced nitrification but, only a slower process (Henze *et al.*, 2002) which can be compensated for. If this is the case in block C it could be expected that a somewhat higher MLSS concentration after the decrease in pH has compensated for the decrease in nitrification rate. Also, nitrifying bacteria are able to adjust to a non-optimal pH (Gerardi, 2002), even though abrupt changes, such as the decrease in September 2007, is expected to be too fast for this adjustment to occur.

The ammonia concentrations have increased after the introduction of the new water plants, but from a very low level. Also, despite tough conditions with regard to pH, nitrification is maintained at a high level within a smaller volume than usual during the winter period of 2008. One reason for the relatively well-coping bacteria in block C might have to do with the configuration of the block. Since it is a step-feed process, there is an anoxic zone after the first two nitrifying compartments. The denitrification process adds alkalinity to the water, approximately half of what is consumed through nitrification (Gerardi, 2002), and it is only the last nitrifying zone that is experiencing the low pH values depicted in Figure 11. That the step-feeding mode reduces the need for alkalinity adjustment has been reported by Miyaji *et al.* (1980).

In summary, it appears that the introduction of the new water plants has had a measurable negative effect on the nitrification in block C. The pH values are according to literature low enough to have large negative impacts on the nitrification activity. It might be expected that the process is working on the lower limit of its functioning interval with respect to pH, and that a larger ammonia load which will consume even more alkalinity or an overall lower alkalinity in the influent might need to be compensated for by addition of alkalinity. The high MLSS appears to create a good environment for the nitrifying population, but the sludge age was too low during 4 weeks in spring 2008.

7.3 DENITRIFICATION

The denitrifying population in an aeration basin comprises 80 % of the total population (Gerardi, 2002). They are facultative anaerobes but prefer oxygen as oxidizing agent for their energy-yielding reactions. This is why one of the most important factors for denitrification is the absence of oxygen. It is only during such conditions that nitrate will be transformed to gaseous nitrogen.

Apart from the oxygen concentration, the availability of substrate is the most important environmental factor. Approximately 3–6 g BOD is required to reduce 1 g of nitrate (Kemira Kemi AB, 1990). Gerardi (2002) mentions a ratio of organic carbon to nitrate of 3:1 if adequate amounts are to be present. The measured BOD/N ratio of 1.9 in primary effluent indicates thereby carbon deficiency. That more incoming substrate has the positive effect on is indicated by the PCA. In M3, higher COD/N corresponds to lower outgoing nitrate concentrations.

The COD consumed to nitrogen denitrified ($\Delta COD/N_d$) can be used as an efficiency factor for the denitrifying system (Naturvårdsverket, 1991). The theoretical value is 2.86 and in the summary by Naturvårdsverket (1991), a comparison was made between plants with pre- and postdenitrification. This showed that postdenitrification was more efficient in using substrates than predenitrification. The $\Delta COD/N_d$ was in average 5 with postdenitrification and 10 with predenitrification. Hence, the value of 6.5 at block C suggests that the carbon is more efficiently utilized in the step-feed configuration compared to ordinary predenitrification.

When using carbon in the wastewater as the primary substrate source, the denitrification rate at 15 °C is expected to be about 1.4 mg N/g VSS, h (Henze *et al.*, 2002). This is very much alike the rate measured during the mass balance which after temperature correction is 1.3 mg N/g VSS, h, although this value is much lower than the calculated average of 2.2 mg N/g VSS, h at 15 °C. This is due to the same reasons as for the nitrification rate and it is important to recognize that all the anoxic volume is not expected to be employed for denitrification during summer operation. This is demonstrated by the nitrogen profile. Both in zone 2 and in zone 3, denitrification was measured at high rates only in the first out of two or three compartments. In zone 2 and zone 3, the total part of the volume utilized for denitrification was estimated to 50 % and 42 % respectively. These numbers are from a day with very good nitrogen removal, but still imply that there is an underestimation of the denitrification rate during summertime. The average anoxic volume utilization cannot be expected to be as low as 40-50 % during the whole summer period. Assuming an anoxic volume utilization during average summer operation of 65 %, the average denitrification rate for the evaluation period is calculated to 2.5 mg N/g VSS, h.

To summarize the discussion on reaction rates, the calculated values, the values from the mass balance and adjusted rates are found in Table 9. All rates are given at 15 °C.

Table 9. Summary of reaction rates. The adjusted nitrification rate is calculated from measurements when NH_4-N_{effl}>2 mg/L. The adjusted denitrification rate is corrected for inactive parts of the volumes during summer operation. Unit in mg N/g VSS, h. T= 15 °C.

	Calculated average	*Mass balance*	*Adjusted averages*	*Maximum*
R_n	1.7	1.4	1.9	3.0
R_d	2.2	1.3	2.5	4.3

It is interesting to determine the effect of hydrolysis in primary clarification on the carbon source for denitrification. That it is increasing the COD/N ratio and decreasing the nitrate in the effluent is indicated in M3 from the PCA (Figure 20). The winter season of 2008 is divided in periods with and without hydrolysis in Appendix D, Table ii. The incoming COD and BOD to block C are in general lower during the periods of hydrolysis. However, when studying the incoming carbon to the biological reactor, there is 10 % more COD and 17 % more BOD compared to the periods without hydrolysis. The BOD/N ratio is increased by 28 %. The reduction of COD and BOD over the primary treatment step is 39 % compared to about 50 % without hydrolysis. This extra carbon source is furthermore contributing to an outgoing nitrate concentration of 5.8 mg/L compared to 10.9 mg/L. The yield, i.e., the sludge production, has been higher during spring 2008. This can be expected to result from the hydrolysis.

The by-passed influent water to the aeration basins is the other operational change initiated in order to enhance the availability of substrate for denitrification. The effect of by-pass on the incoming water characteristics is demonstrated in Appendix D, Table ii, where the winter periods have been evaluated both with and without by-pass. The BOD concentration in the primary influent is increased by in average 12 % in 2007 and by 5 % 2008. The effect in 2008 is comparable with 2007 when considering the part of the winter period (23 November–April) that by-pass was in operation (27 December–April).

Both literature and simulations in JASS indicate that there should not be enough carbon for denitrification to function properly in block C. Despite this, and with the help of efforts to increase the carbon source in block C, effluent nitrate concentrations were decreased during spring 2008 to 8.1 mg/L. Crawford *et al.* (1999) declare that when using the step-feed BNR configuration, the actual hydraulic retention time for the wastewater is only reduced by the RAS flow, not by internal recirculation. This will increase the part of the substrate that is utilized. A high utilization of COD has been noted in block C. That the step-feed mode is beneficial for denitrification is also indicated by the simulations in JASS.

The effect of pH changes on denitrification rates is not consistent in literature. Henze *et al.* (2002) states that the effect of pH on denitrification is comparable with the effect of pH on other biological process, including nitrification. Gerardi (2002) on the other hand concludes that denitrification is less dependent on pH than nitrification and that pH values which are acceptable for proper floc formation (6.5–8.5) also is suitable for denitrification. Both authors agree that the optimal pH range is experienced at pH over neutrality.

7.4 COMPARISON WITH DIMENSIONING DATA

Compared to the dimensioning data in Appendix D, Table i, block C has operated with a very good performance for the given prerequisites. The flow to the block is far smaller than expected, but the nitrogen load to block C is 30 % higher than the prognosis. The incoming BOD load is 85 % of what was expected. Still, the average outgoing concentration of 12.2 mg/L is identical to the expected value but with a nitrogen reduction of 78 % instead of 57 %.

The plant is operating with a far higher MLSS in the bio reactor. The block was supposed to have an MLSS of 3500 mg/L, but in today's operation the MLSS is 46 % higher. The reaction rates are comparable to the design values of 2 mg N/g VSS, h, but with a higher adjusted denitrification rate. It can be considered fortunate that the MLSS in block C is higher than the design value. Otherwise, higher nitrification and denitrification rates than calculated in this study would have been needed to manage the higher nitrogen load.

The surface loading to the secondary clarifier is designed to be 2.7 kg SS/m^2, h, without including the RAS flow. This is higher than the actual load which without RAS is 1.7 kg SS/m^2, h.

7.5 COMPARISON WITH OTHER PLANTS

Compared to the data from other treatment plants shown in Appendix G, block C at Kungsängen WWTP has the highest incoming concentration of nitrogen to the plant and total nitrogen in the primary effluent is 55 % higher than average. The incoming COD/N ratio and the BOD/N entering the aeration tank is the lowest among the plants. That the step-feed mode contributes to a high MLSS is quite clear, since the MLSS above 5000 mg/L is not comparable to the other plants with an MLSS around 2000–4000 mg/L.

The reaction rates in Appendix G are uncertain in many senses. Several methods have most probably been used to calculate or measure the rates and the only plant where the temperature is known to be 15 ºC at the reported rate is at Slottshagen WWTP.

There is a trend towards higher nitrogen concentrations in effluent for plants with lower HRT and higher volumetric nitrogen load (Figure 16(a) and Figure 16(b)). It is also the plants with higher nitrogen loads and higher effluent nitrogen that make use of external carbon, apart from block C. The ammonia concentrations in effluent increase when the aerobic volumetric nitrogen load is higher (Figure 16(c)). Plants with higher outgoing nitrate concentrations have in general a lower BOD/N ratio in the influent (Figure 16(d)). The plants with external carbon are also found among the plants with relatively low BOD/N ratio. The BOD/N ratios are uncertain since nitrogen load from reject water and external carbon will contribute to the actual ratio in the biological step.

Västra Stranden and Öresund WWTP are two plants with low nitrogen in effluent compared to their prerequisites. Both are Bio-P plants and Västra Stranden uses contact stabilization with very good results since 2006 (Johansson, 2008, pers. comm.). Öresund WWTP uses hydrolysis in primary clarification to yield an internal carbon source for phosphorus removal (Jönsson, 2008, pers. comm.). This also has a positive effect on denitrification. The other plants with Bio-P (Käppala WWTP and Källby WWTP) are also performing well when considering denitrification compared to their BOD/N ratio and the fact that they do not use external carbon.

Uppsala has the highest effluent nitrate concentration but among the lowest ammonia concentrations. Judging from the effluent total nitrogen, block C is performing as well as other plants with similar HRT and volumetric nitrogen load, but without the use of external carbon. The ammonia in primary effluent is very low compared to the aerobic load, despite the low pH, indicating unreasonably large aerobic volumes at block C. Nitrate concentrations are high compared with plants with similar BOD/N, but not when considering that block C does not use external carbon.

In comparison, the energy required for nitrogen removal is higher for smaller plants than for larger facilities. The value for block C of 4.8 kWh/kg N is a medium value.

It is difficult to evaluate data from other plants on this level since much information about the processes is not taken into account. More information and data from a greater number of plants would have been needed to draw more far-reaching conclusions. Still, judging from this analysis it is the nitrate concentrations that are comparatively high at block C and should be attended.

7.6 SIGNIFICANCE OF PC MODELS

For a PCA to be trustworthy there should not be too few observations in comparison with the number of variables (Everitt & Dunn, 2001). The general rule is that there should be 5 times as many observations than variables. Also, there should not be less than 50 observations in a PC model. The general criterion is easily fulfilled for model M1 and M2 in this study. Model M3 is almost on the lowest limit possible with its 53 observations. The ability of M3 to contribute with true relationships among variables can, however, be considered reasonably high, since it describes correlations found in M2.

Another important factor is how high loading is needed for a correlation to be significant. Everitt & Dunn (2001) states that for practical significance, the minimal loading for interpretation of a structure should be above ±0.3. For a loading to be indicative, it must be above ±0.7. If the higher level of statistical significance should be reached the number of observations must be considered. If there are 350 observations, a loading of 0.3 is sufficient to reach a statistical significance, while 0.7 is needed when there are 60 observations.

The loadings criteria above indicate that M1 and M2, with factor loadings about or above ±0.3 in one or several components reach statistical significance. M3, with more PCs sharing the variations in data and with much fewer observations, can reach loadings of 0.4 at the highest, and has many variables in the range of 0.2–0.3. The model does not reach statistical significance but can be considered to have practical significance both because of the similarities between M1 and M2, but also because it confirms correlations which are expected, have a practical meaning and can be verified by the other calculations in this project. One such example is the negative correlation between the nitrate in the effluent and the COD/N ratio.

It is important to bear in mind that there might be parameters that should be displaced in time so as to compensate for retention times in the plant. However, since M1 and M2 only deals with flows with daily cycles together with incoming characteristics and M3 has too few measurements to be able to displace the data, this was not considered further in this study. If a more detailed PC model was to be made, a measurement series when time aspects are considered is preferred.

The PC model in this study had the purpose of being descriptive and a tool in the data evaluation. The data was not compensated for skew variables or low explanation levels, which should have been done in a more thorough model calibration. It is interesting to notice that ordinary measurements from an activated sludge process that has not been specifically collected for the purpose of creating a PC model can still be used in this application.

7.7 SIMULATIONS WITH ASM1 IN JASS

It was difficult for the model to describe the denitrification process taking place in reality. There appeared to be a lack of carbon source in the model, which was compensated by changing the parameters $K_{O,H}$ and K_X.

$K_{O,H}$ is the oxygen half-saturation constant for heterotrophs. In the ASM1 process equation describing the change of heterotrophic biomass concentration with time, $K_{O,H}$ determines at what oxygen concentration the denitrification will commence. Increasing the value of this parameter implies that denitrification starts at a higher oxygen concentration and thus that nitrification and denitrification is simultaneous. This also implies that the half-saturation for nitrification, $K_{O,A}$, must be increased (Larrea *et al.*, 2002). It is well known that different combinations of model parameters may produce similar results (Jeppsson, 1996; Petersen, 2000). Therefore it is advisable to choose combinations of parameter values, rather than changing one at a time. The combination of values for $K_{O,H}$ and $K_{O,A}$ in this project were the same as in an earlier simulation with ASM1 for a step-feed BNR system (Lesouef *et al.*, 1992).

In the study by Lesouef *et al.* (1992), the half-saturation constant for hydrolysis of slowly biodegradable material, K_X, is chosen to be 0.02 instead of the default value of 0.03. By decreasing this parameter, the available readily biodegradable material is increased and the denitrification is improved. The hydrolysis process is in reality quite complex and the rate is dependent on the enzymatic activity of the microorganisms. K_X and k_h are parameters that should be adjusted in the calibration procedure (Petersen, 2000). The reason for the difficulty to verify the model and the choice of model parameters with a winter period with a larger aerobic volume might be due to this change of the parameter K_X. Since the heterotrophic bacteria oxidize organic matter also in an aerobic environment, and the hydrolysis process is producing more easily degradable material, this material will be consumed in the aerobic volumes, leaving less food to the anoxic compartments.

The result of the flow-proportioning to the different zones also indicates that there is a lack of carbon. When the flow to the last compartment is reduced, there is not enough carbon to denitrify and the nitrate concentration is increased. If there is enough carbon for denitrification, reducing the flow to the last zone is one way to improve overall nitrogen removal (Kayser *et al.*, 1992; Kayser, 2006). A higher flow to zone 3 reduces nitrification performance due to a too shorter retention time.

ASM1 has been reported to be able to predict the step-feed process in earlier works (Lesouef *et al.*, 1992; Larrea *et al.*, 2001). The incoming nitrogen concentrations were in these studies comparable with the concentration needed to have good model performance without changing the model parameters in this study. It is expected that a more proper characterization of the model parameters would have generated better results from the model. In one sense, the difficulties with the denitrification in the model confirm what is suggested in the earlier discussion; that there ought to be a lack of carbon source in the system for proper denitrification.

8 FUTURE SUGGESTIONS

Both nitrification and denitrification could perform better if more alkalinity and more carbon were available for the respective process. Both these can be added chemically to the biological process; which will result in increased costs. Nitrification is, despite the abrupt pH change, managing well and – since the pH is on a low but constant level – might continue to do so. In fact, the ammonia concentration is despite the high nitrogen load low compared to the other plants in this study. Denitrification on the other hand is limited and directly dependent on the COD/N ratio in the wastewater. The effluent nitrate concentrations are high compared to other plants. If the trend in recent years is to continue, the nitrogen concentration will increase and the carbon source might be decreased.

It is from this perspective that the step-feed process configuration might have more to offer than it has in today’s operation. The process has up to date managed well without trimming and optimization for the given prerequisites. Still, there is a lot of literature on how the flexibility of the step-feed process can be employed in order to improve nitrogen removal and in the case of block C at Kungsängen WWTP meet the challenge of improving denitrification through optimal use of the carbon source. Some examples of studies addressing optimization of the step-feed BNR process are Lesouef *et al.* (1992), Fujii (1996), Görgün *et al.* (1996), Ayesa *et al.* (1998), Larrea *et al.* (2001), deBarbadillo *et al.* (2002), Zhu *et al.* (2006), Tang *et al.* (2007) and (Rivas *et al.*, 2008). All of these authors have used simulations or optimization algorithms in order to obtain their results and the method is on a theoretical rather than on a practical level. Since the introduction of the ASM models they have been recognized to offer many benefits when improving activated sludge process operation. The implementation of ASM1 in JASS used in this project is however not recommended to use for this purpose without improving its predictability through successful validation.

Improving the performance of block C should not be done without considering the other two blocks at the plant, since the requirements on the effluent standards are for the whole plant. In relation to block C, the nitrogen removal on the other two blocks is poorer. Since block B is awaiting a rebuild in the near future, expensive improvements on block C, such as addition of external carbon, is not recommended until the new block B has been taken into operation.

The two operational strategies used in later years to increase the carbon source to the aeration basin can be considered successful judging from the results in this study. This also applies to the reduced aeration in all the zones initiated in spring 2007 as long as the DO concentration is not limiting nitrification. If nitrate concentrations are to be decreased, increasing the hydrolysis in primary clarification to include not only weekends but longer periods should be the first thing to consider. It appears that hydrolysis has a larger positive effect on the BOD/N in primary effluent than by-pass. The two methods should be evaluated further to find a suitable level of operation, also considering the negative effect of a possible larger sludge production. Another alternative to increase the availability of substrate in the biological reactor is to decrease the primary precipitation of phosphorus to obtain more carbon in the primary effluent. All activities enhancing the substrate availability to denitrification should be considered with possible effects on the bio gas production in mind.

One particular aspect of the plant operation should be evaluated in the future, namely the RAS flow. A RAS flow of 180 % of the inflow is very high and a decrease will leave the process with a thicker return flow and reduce pumping requirements. Also, a reduced RAS flow will not re-circulate as much non-alkaline water to the influent, improving the environment for the nitrifiers which might fasten the nitrification process.

9 CONCLUSIONS

Block C at Kungsängen WWTP is operating with quite average influent concentrations of nitrogen in an international perspective. From a national point of view the influent nitrogen is high. The overall nitrogen reduction of block C is 78 % leaving an effluent with in average nitrogen concentration of 12.2 mg/L during the evaluation period. The nitrification and denitrification rates are in average 1.7 mg NH_4-N/g VSS, h and 2.2 mg NO_3-N/g VSS, h at 15 ° C respectively, though both are expected to be underestimations. Nitrification has suffered from the implementation of the water plants Gränby and Bäcklösa since the decrease in alkalinity has brought about a too low pH in the aeration basin. Denitrification in block C is carbon limited.

Block C is compared to other Swedish plants with predenitrification operating with a higher MLSS concentration. Effluent ammonia is low and effluent nitrate is high. Compared to other plants with low BOD/N ratios, block C is not using external carbon.

The simulations in JASS indicated carbon deficiency, and the model was hard to validate for winter operation.

The plant is operating with lower flows than expected but is managing a 30 % higher nitrogen load with 46 % higher MLSS. The denitrification rate is higher compared to the design value.

It is not recommended to use external carbon before block B has been rebuilt. Increasing hydrolysis in primary clarification or optimization of the step-feed process can be achieved to reduce effluent nitrate.

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APPENDIX A

COD fractions in ASM1:

S_S	Soluble biodegradable COD
X_S	Particulate biodegradable
S_I	Soluble non-biodegradable COD
X_I and X_P	Particulate non-biodegradable COD
$X_{B,H}$	Heterotrophic biomass
$X_{B,A}$	Autotrophic biomass

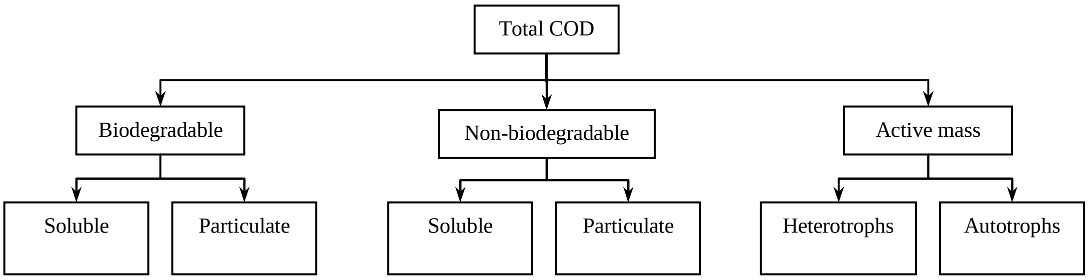

Figure i. Characterization of total COD in ASM1 (Modified from Jeppsson (1996)).

N fractions in ASM1:

TKN	Total Kjeldahl nitrogen
S_{NH}	Free and saline ammonia
S_{NO}	Nitrate and nitrite
S_{NI}	Soluble organically bound non-biodegradable nitrogen
S_{ND}	Soluble organically bound biodegradable nitrogen
X_{ND}	Particulate organically bound biodegradable nitrogen
X_{NI} and X_{NP}	Particulate organically bound non-biodegradable nitrogen
X_{NB}	Active biomass nitrogen

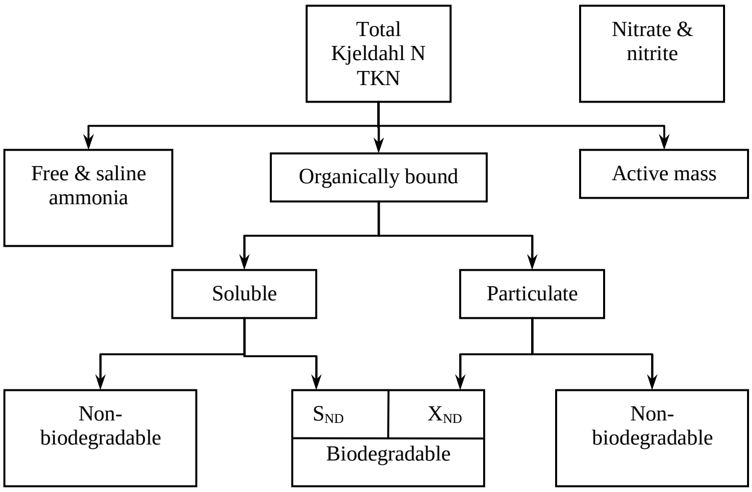

Figure ii. Characterization of total N in ASM1 (Modified from Jeppsson (1996)).

APPENDIX B

Table i. Model parameters of ASM1. From Jeppsson (1996).

	Stochiometric parameters	*Unit*	*Default at 20 °C*
Y_H	Heterotrophic yield	g cell COD formed/g COD oxidized	0.67
Y_A	Autotrophic yield	g cell COD formed/g N oxidized	0.24
f_p	Fraction of biomass yielding particulate products	-	0.08
i_{XB}	Mass N/mass COD in biomass	g N/g COD in biomass	0.086
i_{XP}	Mass N/mass COD in products from biomass	g N/g COD in endogenous mass	0.06
	Kinetic parameters		
μ_H	Heterotrophic max. sp. growth rate	d^{-1}	6.0
b_H	Heterotrophic decay rate	d^{-1}	0.62
K_S	Half-saturation coefficient (Hsc) for heterotrophs	g COD/m^3	20
$K_{O,H}$	Oxygen hsc for heterotrophs	g O_2/m^3	0.20
K_{NO}	Nitrate hsc for denitrifying heterotrophs	g NO_3-N/m^3	0.50
μ_A	Autotrophic max. sp. growth rate	d^{-1}	0.80
b_A	Autotrophic decay rate	d^{-1}	0.20
$K_{O,A}$	Oxygen hsc for autotrophs	g O_2/m^3	0.4
K_{NH}	Ammonia hsc for autotrophs	g NH_3-N/m^3	1.0
η_g	Correction factor for anoxic growth of heterotrophs	-	0.8
k_a	Ammonification rate	m^3/g COD, d	0.08
k_h	Max. sp. hydrolysis rate	g slowly biodeg. COD/g cell COD, d	3.0
K_X	Hsc for hydrolysis of slowly biodeg. substrate	g slowly biodeg. COD/g cell COD	0.03
η_h	Correction factor for anoxic hydrolysis	-	0.4

APPENDIX C

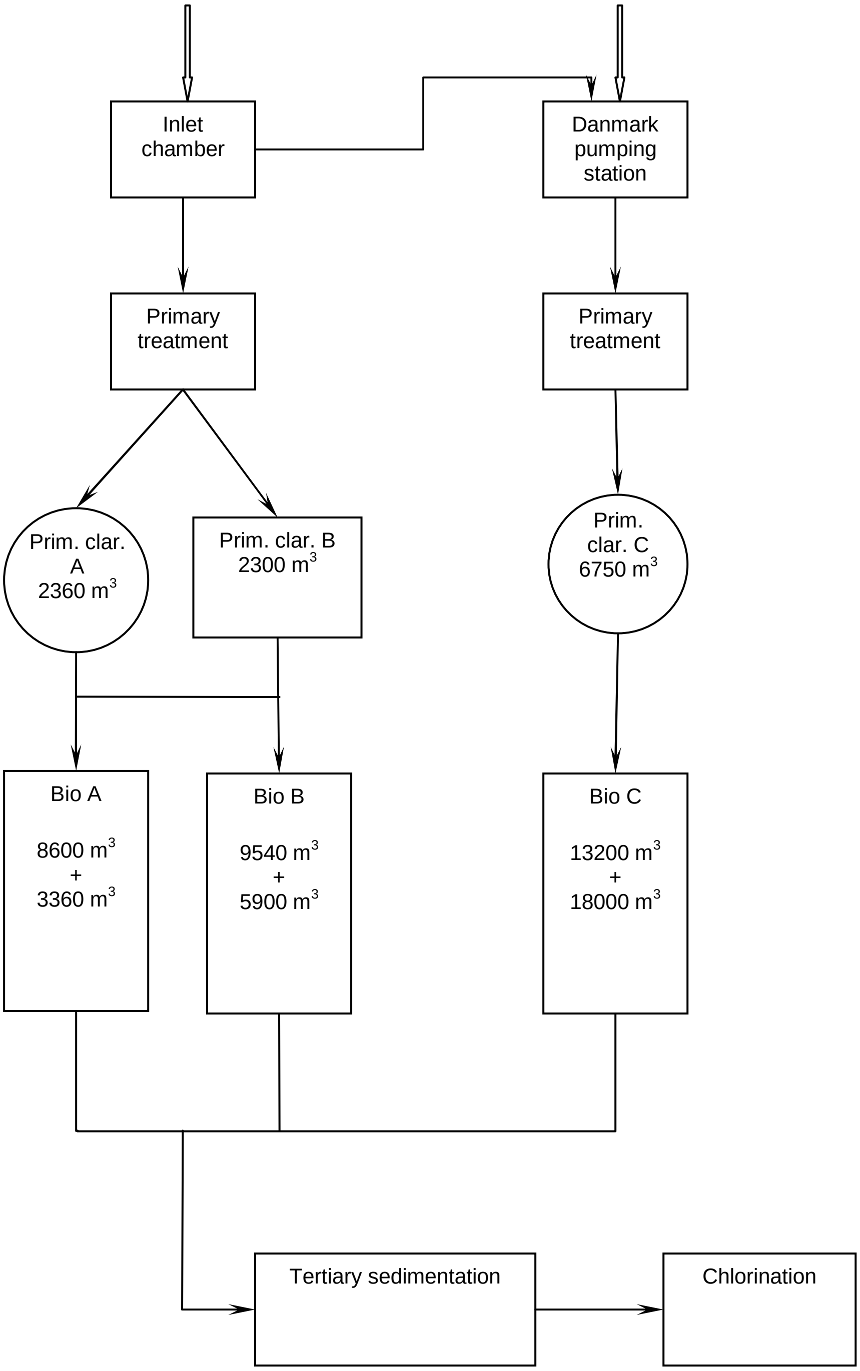

Figure i. Overview of Kungsängen WWTP.

APPENDIX D

Table i. Results from calculations.

			CALCULATIONS				DESIGN	% OF DESIGN	MASS BALANCE
		Unit	AVERAGE	±	MAX	MIN			
	Flow from block C	m^3/d	32772	3277	66540	17432	67200	49	
	Temperature	°C	15,5	0,2	19,3	10,5	14,7		17,7
	pH in	-	7,6	0,5	8,0	6,4			8,0
	pH out block C	-	6,9	0,2	8,4	4,3			
	Alkalinity in bio	mgHCO$_3^-$/L	379	27	528	114			
	Alkalinity out C	mgHCO$_3^-$/L	203	14	308	119			
	HRT	h	8,9	0,9	15,3	4,4	4,7	188	
	Surface load	m/h	0,4	0,04	0,8	0,2	0,8	47	
	SS load	kg/m^2,h	4,6	0,8	7,0	2,7	2,7^1	171	
	BOD load	kgBOD$_7$/d	6,9	1,8	12,0	3,2	8,1	85	
	N load	kgN/d	1,8	0,3	2,6	1,1	1,4	129	
tot	**COD in**	mg/L	507	81	1100	250			530
	BOD in	mg/L	217	54	470	100			180
	N-tot in	mg/L	56,7	0	79	34			61
bio	**COD in**	mg/L	239	38	389	130			300
	BOD in	mg/L	95,1	24	184	47			92
	N-tot in	mg/L	50,1	8	70	26			57
	COD out	mg/L	39	6	70	30			35
	BOD out	mg/L	4,0	1,0	6,0	3,0			5,0
	N-tot out	mg/L	12,2	2	20	6	12,2^2	100	7,4
	NH4-N out	mg/L	1,3	0,1	6,3	0,03			0,6
	NO3-N out	mg/L	9,7	1,2	17,0	3,7			5,9
tot	**COD red**	%	92	21	97	82			93
	BOD red	%	98	35	99	96	95	103	97
	N red	%	78	14	88	60	57^2	137	88
bio	**COD red**	%	83	19	90	69			88
	BOD red	%	96	34	98	91			95
	N red	%	76	13	87	58			87
tot	**COD/BOD**	mg/mg	2,4	0,7	4,5	1,2			2,9
	COD/N	mg/mg	8,9	1,6	13,9	5,6			8,6
	BOD/N	mg/mg	3,8	0,8	5,9	2,4			2,9
bio	**COD/BOD**	mg/mg	2,6	0,8	4,3	1,5			3,2
	COD/N	mg/mg	4,8	0,9	8,2	2,8			5,3
	BOD/N	mg/mg	1,9	0,5	3,3	1,0			1,6
	ΔCOD/N$_d$	mg/mg	6,5	2,0	13,5	3,6			6,8
	MLSS average	g/m^3	5117	512	6652	4173			5052
	MLSS zone 3	g/m^3	4950	495	5850	3786	3500	141	4590
	MLVSS/MLSS		0,77	0,12	-	-	0,65	118	0,77
	Rn	mgN/gVSS, h	1,8	0,3	3,0	1,0	2^3		1,8
	Rd	mgN/gVSS, h	2,2	0,4	3,6	0,8	2^3		1,6
corr.	**Rn**	mgN/gVSS, h	1,7	0,3	2,8	0,8	±10 %/°C		1,4
	Rd	mgN/gVSS, h	2,2	0,4	4,3	0,6			1,3
	Total SRT	d	17	1,7	48	6,2			
	Aerobic SRT	d	9,8	1,1	24	4,2			4,1
	Temp comp. aerob. SRT	d	11	1,3	33	3,8			5,7
	F/M	kgBOD$_7$/kgVSS,d	0,07	0,03	0,14	0,03			0,09
	Volumetric load	kgBOD$_7$/m^3,d	0,27	0,05	0,55	0,09			0,18
	Volumetric nitrogen load	kgN/m^3,d	0,14	0,03	0,20	0,06			0,11
	Yield	kgSS/kgBOD$_7$	0,52	0,27	2,7	-0,35			0,60
	Power/treated N	kWh/kgN	4,80						

Comments:

[1] Calculated without RAS

[2] At full load and no external carbon source

[3] At 15 °C

Table ii. Results from calculations, different periods. The water plants were taken into operation in September 2007.

				Aeration			Without bypass[4]		Reject water, hydrolysis				Water plants	
		Unit	AVERAGE	Winter 2007	Summer 2007	Winter 2008	Winter 2007	Winter 2008	Hydr.	No hydr.	Less rej.	Hydr.+less rej.	Before	After
	Flow from block C	m^3/d	32772	31633	28644	37966			39230	37273	30269	36523	28722	35942
	Temperature	°C	15,5	13,4	17,0	14,4			14,6	14,3	15,7	14,3	15,7	15,1
	pH in	-	7,6	7,9	7,4	7,7			7,7	7,7	7,5	7,6	7,5	7,7
	pH out block C	-	6,9	7,8	6,8	6,6			6,6	6,5	7,0	6,6	7,3	6,5
	Alkalinity in bio	mgHCO$_3^-$/L	379	474	382	332	474	332	340	322	373	335	430	340
	Alkalinity out C	mgHCO$_3^-$/L	203	278	181	191			206	172	202	205	236	176
	HRT	h	8,9	8,3	11,3	8,7			8,4	8,9	10,4	8,8	10,7	9,2
	Surface load	m/h	0,4	0,4	0,3	0,4			0,5	0,4	0,4	0,4	0,3	0,4
	SS load	kg/m^2,h	4,6	4,2	4,3	5,3			5,4	5,0	4,3	5,2	4,1	5,0
	BOD load	kgBOD$_7$/d	6,9	7,1	6,9	6,9			6,6	7,0	6,5	5,6	6,7	7,2
	N load	kgN/d	1,8	1,8	1,7	2,0			2,0	2,0	1,7	1,7	1,7	2,0
tot	**COD in**	mg/L	507	541	547	435			418	457	501	398	554	462
	BOD in	mg/L	217	224	239	182			169	187	215	152	235	199
	N-tot in	mg/L	56,7	56,2	60,4	51,7			50,8	53,0	54,7	47,7	57,8	55,6
bio	**COD in**	mg/L	239	253	235	238	226	226	255	231	234	244	238	240
	BOD in	mg/L	95,1	105	89,5	97,1	94,0	92,0	104	88,5	91,8	97,3	95,6	94,6
	N-tot in	mg/L	50,1	50,4	52,3	47,3	49,9	46,8	46,3	48,7	48,5	43,9	50,0	50,1
	COD out	mg/L	39	45	39	35			33	36	38	31	41	36
	BOD out	mg/L	4,0	3,8	4,1	4,1			4,0	4,1	3,8	3,8	3,9	4,1
	N-tot out	mg/L	12,2	13,1	12,7	10,9			9,2	13,2	11,5	7,9	12,0	12,4
	NH4-N out	mg/L	1,3	0,5	1,3	1,6			1,8	1,3	0,9	1,1	0,5	1,9
	NO3-N out	mg/L	9,7	11,6	10,2	8,1			5,8	10,9	9,3	5,7	10,2	9,2
tot	**COD red**	%	92	92	92	92			91	92	92	92	92	92
	BOD red	%	98	98	98	98			98	98	98	97	98	98
	N red	%	78	76	78	79			82	75	79	83	79	78
bio	**COD red**	%	83	82	83	85			86	83	83	87	82	84
	BOD red	%	96	96	95	96			96	95	96	96	96	95
	N red	%	76	73	75	77			80	73	77	82	76	75
tot	**COD/BOD**	mg/mg	2,4	2,4	2,3	2,5			2,6	2,5	2,4	2,7	2,4	2,4
	COD/N	mg/mg	8,9	9,6	9,1	8,4			8,2	8,7	9,2	8,4	9,4	8,2
	BOD/N	mg/mg	3,8	4,1	3,9	3,5			3,3	3,7	3,9	3,2	4,0	3,5
bio	**COD/BOD**	mg/mg	2,6	2,9	2,1	2,7	2,6	2,5	2,9	2,7	2,3	2,7	2,3	2,7
	COD/N	mg/mg	4,8	5,2	4,5	5,1	4,7	4,8	5,5	4,5	4,9	5,6	4,8	4,9
	BOD/N	mg/mg	1,9	2,1	1,7	2,1	1,9	2,0	2,3	1,8	1,9	2,2	1,9	2,0
	ΔCOD/N$_d$	mg/mg	6,5	7,1	5,7	7,2			7,8	6,4	6,4	7,6	6,2	6,9
	MLSS average	g/m^3	5117	4751	4929	5490			5604	5403	4967	5515	4865	5309
	MLSS zone 3	g/m^3	4950	4216	4413	4808			4888	4746	4424	4850	4353	4670
	MLVSS/MLSS		0,8	-	-	-			-	-	-	-	-	-
	Rn	mgN/gVSS, h	1,8	1,5	2,1	1,6			1,5	1,8	1,7	1,4	1,7	1,9
	Rd	mgN/gVSS, h	2,2	2,9	1,7	2,3			2,4	2,2	2,0	2,2	2,1	2,2
corr.	**Rn**	mgN/gVSS, h	1,7	1,8	1,7	1,8			1,6	1,9	1,6	1,5	1,6	1,8
	Rd	mgN/gVSS, h	2,2	3,4	1,4	2,5			2,5	2,4	2,0	2,4	2,1	2,2
	Total SRT	d	17	15	21	11			10	13	19	11	20	14
	Aerobic SRT	d	9,8	10,9	11,0	7,4			6,4	8,7	10,8	7,2	11,2	8,3
	Temp comp. aerob. SRT	d	11	9,3	14,2	6,9			6,0	8,1	12,4	6,6	12,9	8,8
	F/M	kgBOD$_7$/kgVSS,d	0,1	0,08	0,06	0,08	0,07	0,07	0,08	0,07	0,10	0,07	0,17	0,07
	Volumetric load	kgBOD$_7$/m^3,d	0,3	0,28	0,23	0,32	0,25	0,30	0,35	0,27	0,24	0,31	0,24	0,29
	Volumetric nitrogen load	kgN/m^3,d	0,1	0,13	0,13	0,15	0,13	0,15	0,15	0,15	0,13	0,14	0,12	0,15
	Yield	kgSS/kgBOD$_7$	0,5	0,39	0,46	0,66			0,57	0,57	0,52	0,55	0,50	0,54
	Power/treated N	kWh/kgN	4,8	-	-	-	-	-	-	-	-	-	-	-

[4] Calculated values based on the concentrations after primary clarification

APPENDIX E

Table i. Signals from Uni-View

Nr	Description	Unit	Signal
1	Q from bio C	m^3/h	MC-AFM-04
2	Q A+B	m^3/h	SF-AFM-01.ÄR
3	NH4 effluent bio C	mg/L	MC-ANH-01
4	NO3 effluent bio C	mg/L	MC-ANO-01
5	pH total in	pH	SF-PHM-01.ÄR
6	pH effluent bio C	pH	GC-APH-01
7	pH total effluent	pH	SS-APH-01
8	SS total effluent	mg/L	SS-ASS-02
9	NO3 total effluent	mg/L	SS-ANO-01
10	NH4 total effluent	mg/L	SS-ANH-01
11	Temperature total in	℃	SF-ATM-01.ÄR
12	Precipitation	mm	MA-REGNIDAG2
13	MLSS L1	mg/L	MC-ASS-01
14	MLSS L2	mg/l	MC-ASS-02
15	MLSS L3	mg/L	MC-ASS-03
16	MLSS L4	mg/L	MC-ASS-04
17	MLSS L5	mg/L	MC-ASS-05
18	SS RAS L1+L2	mg/L	MC-ASS-06
19	SS RAS L3	mg/L	MC-ASS-07
20	SS RAS L4+L5	mg/L	MC-ASS-08
21	SS after prim. clar. C	mg/L	MC-ASS-10
22	WAS L1+L2	m^3/h	MC-SFM-09
23	WAS L3	m^3/h	MC-SFM-10
24	WAS L4+L5	m^3/h	MC-SFM-11
25	RAS L1	m^3/h	MC-SFM-22-L1
26	RAS L2	m^3/h	MC-SFM-22-L2
27	RAS L3	m^3/h	MC-SFM-23
28	RAS L4	m^3/h	MC-SFM-24-L4
29	RAS L5	m^3/h	MC-SFM-24-L5
30	Sludge pump prim. clar.	m^3/h	FC-SFM-01
31	External sludge flow	m^3/h	ES-SFM-01
32	O_2 zone 1 L1-L5	mg/L	MC-AO2-101--501
33	O_2 zone 2 L1-L5	mg/L	MC-AO2-102--502
34	O_2 zone 3 L1-L5	mg/L	MC-AO2-103--503
35	O_2 flow zone 1-3, L1-L5	Nm^3/h	MC-LFM 101--503
36	FeCl dosing, grit chamber	mL/m^3	GC-JFM-FÖR
37	Tot. consumpt. FeCl	L/h	GC-JFM-01
38	FeCl dosing, chemcial prec.	mL/m^3	JK-JFM-EFTER
42	Centrifuge 1	digital	SB-SCF-01D
43	Centrifuge 2	digital	SB-SCF-02D

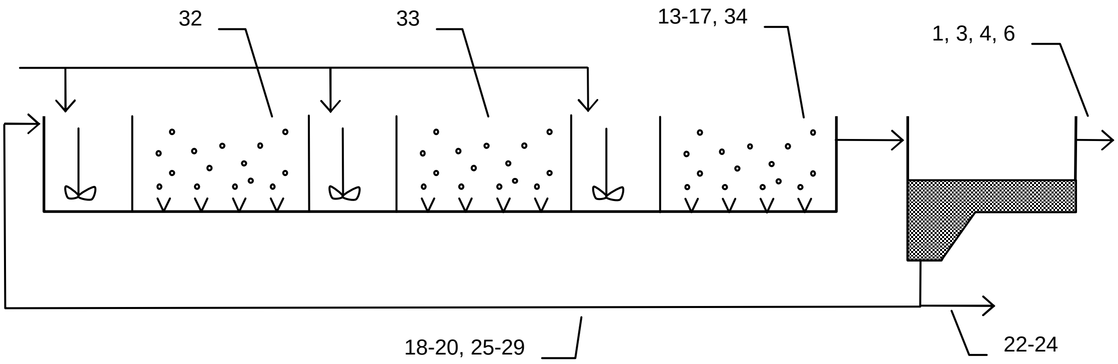

Figure i. Some of the signals from the biological step, an overview.

APPENDIX F

Table i. Model variables in M1.

Name	*Description*	*Unit*	*Comment*
Q	Flow from C	m^3/h	
NH4 out	NH4-N out from C	mg/L	
pH in	pH in incoming wastewater	-	
pH out	pH in outgoing wastewater from C	-	
Prec.	Precipitation at Kungsängen WWTP	mm	
MLSS1	MLSS concentration in zone 1, line 1	mg/L	
MLSS2	MLSS concentration in zone 1, line 2	mg/L	
MLSS3	MLSS concentration in zone 1, line 3	mg/L	
MLSS4	MLSS concentration in zone 1, line 4	mg/L	
MLSS5	MLSS concentration in zone 1, line 5	mg/L	
MLSS	Average MLSS concentration	mg/L	
SS RAS 12	SS in RAS line 1 and 2	mg/L	
SS RAS 3	SS in RAS line 3	mg/L	
SS RAS 45	SS in RAS line 4 and 5	mg/L	
SS RAS	Average SS RAS	mg/L	
WAS 12	WAS line 1 and 2	m^3/h	
WAS 3	WAS line 3	m^3/h	
WAS 45	WAS line 4 and 5	m^3/h	
RAS 12	RAS line 1 and 2	m^3/h	
RAS 3	RAS line 3	m^3/h	
RAS 45	RAS line 4 and 5	m^3/h	
SRT	Solids retention time (sludge age)	d	
Rej.	Reject water flow	-	Specified as 0-1
DO z1	Dissolved oxygen conc. in zone 1	mg/L	
DO z2	Dissolved oxygen conc. in zone 2	mg/L	
DO z3	Dissolved oxygen conc. in zone 3	mg/L	
Q O2	Total air flow rate	Nm^3/h	
Hydr.	Hydrolysis in primary clarification	-	Specified as 0-1

Table ii. Model variables in M2.

Name	*Description*	*Unit*	*Comment*
Q	Flow from C	m^3/h	
NH4 out	NH4-N out from C	mg/L	
pH in	pH in incoming wastewater	-	
pH out	pH in outgoing wastewater from C	-	
Prec.	Precipitation at Kungsängen WWTP	mm	
MLSS	Average MLSS concentration [mg/L]	mg/L	
SS RAS	Average SS RAS	mg/L	
WAS	Average WAS	m^3/h	
RAS 12	RAS line 1 and 2	m^3/h	
RAS 345	RAS line 3,4 and 5	m^3/h	
SRT	Solids retention time (sludge age)	d	
Rej.	Reject water flow	-	Specified as 0-1

DO	Average dissolved oxygen conc.	mg/L
Q O2	Total air flow rate	Nm^3/h
SS load	Solids loading to the sec. clarifier	kg/m^2, d

Table iii. Model variables in M3.

Name	***Description***	***Unit***	***Comment***
Q	Flow from C	m^3/h	
pH in	pH in incoming wastewater	-	
pH out	pH in outgoing wastewater from C	-	
Alk in	Alkalinity in to C	mg HCO_3^-	
Alk out	Alkalinity out from C	mg HCO_3^-	
BOD in	BOD in to C	mg/L	
totN in	Total nitrogen in to C	mg/L	
totN out	Total nitrogen out from C	mg/L	
NH4 out	NH_4-N out from C	mg/L	
NO3 out	NO_3-N out from C	mg/L	
N red.	Reduction of total nitrogen	%	
MLSS	Average MLSS	mg/L	
F/M	Food-to-microorganism ratio	d^{-1}	
COD/N	Ratio of COD to N in to C	-	
DO	Average dissolved oxygen	mg/L	
Rej.	Reject water flow	-	Specified as 0-1
Q O2	Total air flow rate	Nm^3/h	
Hydr.	Hydrolysis in primary clarification	-	Specified as 0-1

APPENDIX G

Table i. Data from other treatment plants.

		City	Uppsala	Stockholm	Stockholm	Stockholm	Helsingborg	Norrköping	Halmstad	Lund	Jönköping	Västerås	Karlstad	Jönköping
	N.A.= Not available	Plant	**Kungsängsverket (C)**	**Henriksdal**	**Käppala**	**Bromma**	**Öresundsverket**	**Slottshagen**	**Västra stranden**	**Källby**	**Simsholmen**	**Kungsängen**	**Sjöstadsverket**	**Huskvarna**
	Load	p.e.	149000 (total)	850000	501000	248000	139000	111 900	97000	84000	73742	73054	54700	30557
	Dim. load	p.e.	200000 (total)	N.A.	700000	270000	200000	200000	140000	90000	95000	125000	97000	43000
	Flow	m^3/d	32772	237000	141250	118000	69000	44300	40100	32000	33300	48700	24000	17800
	HRT	h	9	21	24	5	11	8	11	16	7	6	10	7
	External carbon (Y/N):		N	N	N	Y	N	Y	N	N	Y	Y	Y	Y
	Bio-P (Y/N):		N	N	Y	N	Y	N	N	Y	N	N	N	N
	Contact stabilization (Y/N):		N	N	N	N	N	N	N	Y	Y	Y	Y	Y
	Separate reject water treatment (Y/N):		N	N	N	N	N	Y (SBR)	N	N	N	N	N	N
	Vol bio basin	m^3	13200	204000	143850	23400	32796	15100	18925	21900	9600	12600	9642	4900
	Max. aerobic volume	m^3	9534	appr. 100000	80296	18800	16398	11325	7850	13680	7640	7920	4544	4044
	Vol sec. clar.	m^3	18000	58000	65120	9700	5280	15400	7260	9040	5800	9210	4400	2880
	BOD load	tonnes/d	6,9	59	30	17	10	12	6,7	5,7	5,2	5,1	4,2	2,2
	N load	tonnes/d	1,8	11	6,4	3,2	2,3	2,0	1,2	1,2	0,9	1,7	0,9	0,6
tot	**COD in**	mg/L	507	N.A.	561	N.A.	446	N.A.	500	N.A.	327	N.A.	414	N.A.
	BOD in	mg/L	217	250	211	154	141	270	167	177	157	105	173	121
	N-tot in	mg/L	57	45	45,3	31,4	33	45	30	36	26,3	35	38	31,5
	COD/N		8,9	N.A.	12,4	N.A.	13,5	N.A.	N.A.	N.A.	12,4	N.A.	10,9	N.A.
	BOD/N		3,8	5,6	4,7	4,9	4,3	6,0	5,6	4,9	6,0	3,0	4,6	3,8
bio	**COD in**	mg/L	239	N.A.	286	N.A.	210	N.A.	300	N.A.	172,3	95	228	N.A.
	BOD in	mg/L	95	114	N.A.	80	N.A.	90	N.A.	N.A.	70,8	N.A.	105	N.A.
	N-tot in	mg/L	50	35	34	32,5	28	38	27	N.A.	27,3	31	35	N.A.
	COD/N		4,8	N.A.	8,4	N.A.	7,5	N.A.	11	N.A.	6,3	3,1	6,5	N.A.
	BOD/N		1,9	3,3	N.A.	2,5	N.A.	2,4	N.A.	N.A.	2,6	N.A.	3,0	N.A.
	Prim. effl. data from period:		Feb 2007-Apr 2008	N.A.	2007	2007	2007	2007	2007	-	2007	Jan-Aug 2008	2007	-
	COD out	mg/L	39	N.A.	30,8	N.A.	23	N.A.	34	N.A.	24	N.A.	32	N.A.
	BOD out	mg/L	4	3	3	3	2,7	7	3,3	2,8	5,4	4,5	6,1	5,6
	N-tot out	mg/L	12	7,7	8,5	13	5,8	10	6,2	8,1	13,3	11,2	13	10,9
	NH4-N out	mg/L	1,3	1,8	<1	3,4	1,0	4,8	2	1,2	8,7	3,5	3,8	3,2
	NO3-N out*	mg/L	9,7	4,7	6,7*	8,7	4,2*	4,7	2,9	6,1*	3,3*	6,6*	7,9*	6,6*
	COD red. prim. cl.	%	53	N.A.	49	N.A.	53	N.A.	40	N.A.	47	N.A.	45	N.A.
	BOD red. prim. cl.	%	56	54	N.A.	48	N.A.	N.A.	N.A.	N.A.	55	N.A.	39	N.A.
	COD red.	%	92	N.A.	95	N.A.	95	N.A.	N.A.	N.A.	93	N.A.	92	N.A.
	BOD red.	%	98	99	99	98	98	97	98	98	97	97	96	95
	N red.	%	78	83	81	59	82	78	79	78	49	68	66	65
	Rn	mgN/gVSS, h	1,8	4,5, 1,5, 0,5	N.A.	3,4	N.A.	4,5	0,9	4,7	N.A.	N.A.	2,9	N.A.
	Rd	mgN/gVSS, h	2,2	2,5, 1, 0,5	N.A.	2	N.A.	N.A.	N.A.	N.A.	N.A.	N.A.	3,8	N.A.
	MLSS	g/m^3	5100	2400	1600-3300	2500-3200	3000-4000	3500	2500-2700	3200-4500	3400	2000-2800	2700	3500
	Aerobic SRT	d	10	15-20	6-7	5	10-15	12	5-6	9	20	6,1	7-9	10-11
tot	**F/M**	kgBOD/kgSS,d	0,11	0,12	0,09	0,27	0,08	0,23	0,14	0,07	0,16	0,16	0,16	0,13
	Vol. load	kgBOD/m^3,d	0,54	0,3	0,2	0,8	0,3	0,8	0,4	0,3	0,5	0,4	0,4	0,4
	Vol. N load	kgN/m^3,d	0,14	0,05	0,04	0,16	0,07	0,13	0,06	0,05	0,09	0,14	0,09	0,11
	Aer. vol. N load	kgN/m^3,d	0,20	0,11	0,08	0,20	0,14	0,18	0,15	0,08	0,11	0,22	0,20	0,14
bio	**F/M**	kgBOD/kgSS,d	0,05	0,06	N.A.	0,14	N.A.	0,08	N.A.	N.A.	0,07	N.A.	0,10	N.A.
	Vol. load	kgBOD/m^3,d	0,24	0,13	N.A.	0,40	N.A.	0,26	N.A.	N.A.	0,25	N.A.	0,26	N.A.
	Vol. N load	kgN/m^3,d	0,12	0,04	0,03	0,16	0,06	0,11	0,06	N.A.	0,09	0,12	0,09	N.A.
	Effect/kgN removed	kWh/kgN	4,8	2,7	5	3,9	N.A.	4	7,3	1,7	8,2	N.A.	N.A.	7

*NO3-N out is calculated from N-tot-NH4-N assuming org-N=0.1*tot-N

Rn and Rd calculated in three zones each

Rn based on SS

Prim. effl. higher due to int. flows

No aerobic cont. stab.

Rn, Rd average from 2007

References for Table i:
Reports: Stockholm Vatten, 2008; Käppalaförbundet, 2008a; Käppalaförbundet, 2008b; Jönsson, 2008a; Norrköping Vatten AB, 2008; Halmstads kommun, 2008; VA SYD, 2008; Jönköpings kommun, 2008a; MälarEnergi, 2008; Karlstads kommun, 2008; Jönköpings kommun, 2008b.

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