

Calibration of a dynamic model for the activated sludge process at Henriksdal wastewater treatment plant

Cajsa Hellstedt

Abstract

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To simulate the activated sludge process at a wastewater treatment plant a dynamic model that describes the process is needed. In 1987 IWA, International Water Association presented *ASM1, Activated Sludge Model No.1* which still is the most widely used model for this process. In this thesis the ASM1 has been used to describe the activated sludge process. The work is a part of a European project, HIPCON (Holistic Integrated Process CONTROL) at IVL, Swedish Environmental Research Institute.

The main objective of the work was to calibrate a model that realistically describes the activated sludge process and secondary sedimentation at Henriksdal wastewater treatment plant in Stockholm. A benchmark model in MATLAB/Simulink was used as a base and rebuilt and extended to fit the process of Henriksdal. In the model ASM1 is used to describe the activated sludge process. The settler is modelled with a mass balance model where the settling velocity is described by a double exponential function. The parameters used in both models have to be calibrated to fit the wastewater treatment plant. To find information about Henriksdal two measuring campaigns were performed to provide data for calibration and validation and to gather information about the composition of the incoming wastewater. From this data a model was developed and calibrated for the process at Henriksdal. After calibration the obtained model worked very well for modelling average values but did not adjust quite as well to fast dynamic changes.

Keyword: activated sludge process, ASM1, calibration, secondary sedimentation, wastewater treatment

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Referat

Kalibrering av en dynamisk modell för aktivslamprocessen på Henriksdals reningsverk

Cajsa Hellstedt

För att simulera aktivslamprocessen på ett reningsverk krävs en dynamisk modell som realistiskt beskriver processen. 1987 kom IWA, International Water Association med *ASM1, Activated Sludge model no. 1* som fortfarande är den mest använda modellen för att beskriva denna process. I detta examensarbete har ASM1 används för att beskriva aktivslamprocessen på Henriksdals reningsverk i Stockholm. Arbetet har utförts som en del i ett europeiskt projekt, HIPCON (Holistic Integrated Process CONtrol) på IVL, Svenska Miljöinstitutet AB.

Arbetet har gått ut på att ta fram en modell som realistiskt beskriver aktivslamprocessen och eftersedimenteringen. För att göra detta har en referensmodell i MATLAB/Simulink använts som grund och byggts om för att likna processen vid Henriksdal. Denna modell i Simulink använder ASM1 för att beskriva aktivslamprocessen. Eftersedimenteringen modelleras med en massbalansmodell där sedimenteringshastigheten beskrivs av en dubbelexponentiell sedimenteringsfunktion. Både ASM1 och sedimenteringsfunktionen använder en mängd olika parametrar för att beskriva processerna och dessa måste kalibreras fram för den process som skall modelleras. Aktivslamprocessen är en biologisk process som beror på en mängd yttre och inre faktorer och är unik för varje reningsverk. Därför finns det inte något enkelt sätt att kalibrera en modell på och information för det enskilda reningsverket i fråga måste tas fram. I detta arbete har två mätkampanjer utförts på Henriksdal för att få mätserier till kalibrering och validering samt information om avloppsvattnets sammansättning. Litteraturstudier har också genomförts för att få information om vilka parametervärden som är av störst intresse för modellen samt i vilket område varje parameter kan förväntas finnas.

Arbetet har sedan gått ut på att efter riktlinjer för kalibrering funna i litteraturen ta fram en modell som så realistiskt som möjligt beskriver processen på Henriksdal. Först genomfördes en kalibrering med medelvärden för att hitta jämviktstillstånd och därmed en stabil modell på länge sikt. Utifrån den modellen utfördes sedan en dynamisk kalibrering för att få en modell som beskriver även kortsiktiga och snabba förändringar. Till sist utfördes en validering för att kontrollera om modellen fungerar även för en dataserie som ej använts vid kalibrering. Den framtagna modellen fungerade mycket bra för att modellera medelvärden på lång sikt. För snabba förändringar verkade modellen ligga fel i tiden och troligtvis var den reella uppehållstiden kortare än den teoretiska och bidrar till sämre modellenpassning.

Nyckelord: aktivslamprocess, ASM1, avloppsvattenrening, eftersedimentering, kalibrering

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Preface

The aim of this thesis was to calibrate and validate a dynamic model for the activated sludge process at Henriksdal wastewater treatment plant. The project was initiated by IVL, Swedish Environmental Research Institute as a part in the HIPCON (Holistic Integrated Process CONtrol) project. Magnus Rahmberg at IVL has been the supervisor of this work at IVL and Bengt Carlsson at Uppsala University, Department of Information Technology the supervisor in Uppsala.

First of all I would like to thank Magnus Rahmberg at IVL for helping me out with everything and never getting tired of my questions. I also would like to thank Dr. Ulf Jeppsson, LTH, for letting me use his Simulink implementation of ASM1 and Bengt Carlson and Mats Ekman at Uppsala University for providing me with the benchmark model and answers to my questions. A big thank you to Michael Medoc and all the other people at Stockholm Vatten for giving me access to Henriksdal and helping me out during my measuring campaigns and sharing information about their wastewater treatment plant.

Stockholm June 2005

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1 INTRODUCTION

This Master of Science thesis work is part of the HIPCON- (Holistic Integrated Process CONtrol) project at IVL, Swedish Environmental Research Institute. The main objective of the work is to adjust an already existing model for the activated sludge process, ASM1, to Henriksdal wastewater treatment plant in Stockholm, Sweden. The dynamic model will later be part of a base for optimising the process according to economic- and environmental parameters as a part in the HIPCON project.

1.1 BACKGROUND

1.1.1 HIPCON

HIPCON is a European project aiming to develop methodology and technology for holistic process management for process industries. The project is co-ordinated by IVL and there are five other partners involved in the project; London School of Economics and Political Science (LSE), Computer Technology Institute (CTI), Uppsala University Division of Systems and Control, IMCG Ltd, SSAB Oxelösund AB and Stockholm Vatten AB. The project started in January 2004 and will run for three years. There are two case study industries in the project and one of them is the wastewater treatment at Stockholm Vatten. One of the purposes with the project is to develop a modelling method for the process that links economic, environmental and quality performance together (hipcon.org, 2005).

1.1.2 Henriksdal wastewater treatment plant

The Henriksdal wastewater treatment plant is blasted into a rock and is the worlds largest wastewater treatment plant under ground. It covers about 300 000 m² and has 18 km of tunnels. Henriksdal wastewater treatment plant treats sewage from about 700 000 persons. The load arises mostly from physical persons and a large number of factories but with small pollutant factors. The flow rate is about 2.8 m³/s in average but varies a lot over time due to different weather conditions such as rain (stockholmvatten.se, 2004).

The wastewater enters the treatment plant from two directions, the central city of Stockholm (Henriksdal) and the southern part of town (Sickla) and there are separate pre-treatment stations. First the water passes through a coarse screen (3 mm between bars) and a grit chamber to remove big and heavy particles. Then the water enters an aerated step where a solution of ferrous sulphate is added to precipitate phosphate and some organic material, the ferrous ions is oxidised to ferric ions and precipitates to ferric phosphate. After this pre-sedimentation step the water from the two streams merge into a block of 13 primary sedimentation basins. Water from Henriksdal inlet goes to basin 1-3 and water from Sickla inlet to basin 11-13 and basin 4-10 receives mixed wastewater. The water phase is then pumped into the biological stage which consists of seven parallel basins with a total volume of about 205 000 m³. The basins

are divided into eight zones; one anoxic mixing zone, six aerated zones and one de-aeration zone.

- anoxic mixing zone: denitrification occurs, nitrate is reduced to form nitrogen gas, and some organic material is oxidised
- aerated zones: remaining organic compounds and ammonium are oxidised, carbon dioxide and nitrate is formed
- de-aeration zone: the remaining oxygen is consumed

Some of the water from the de-aeration zone is re-circulated to the anoxic mixing zone and the remaining water enters the secondary sedimentation basin and then into the last step, the sand filter, before it is let out in Saltsjön (Ek, 2004). The process line at Henriksdal is described in figure 1.

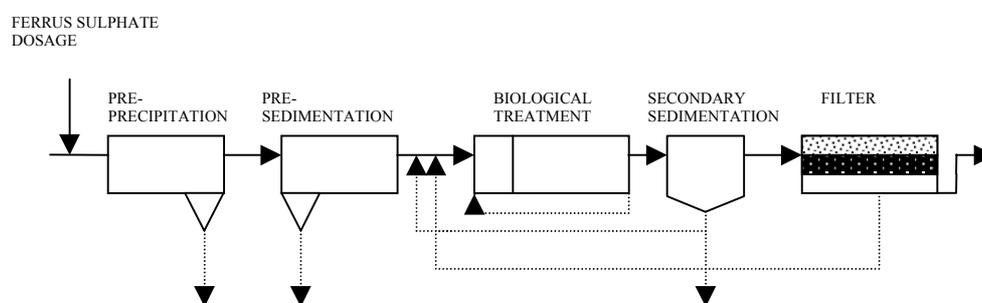


Figure 1. The process line at Henriksdal wastewater treatment plant.

1.1.3 Biological wastewater treatment

The activated sludge process is the most common biological wastewater treatment around. The idea is to maintain suspended material in the wastewater by stirring or aeration. The suspended material contains bacteria, micro-organisms, organic and inorganic particles. The organic material is used as the energy source for the biomass and in that way organic material is removed when biomass is produced. From the beginning the process was mainly used for removal of organic carbon substrates from the wastewater which was easily obtained by rather simple methods. During the last two decades many activated sludge plants have been modified to more advanced treatment to remove biological nitrogen and phosphorus as well (Svenska Kommunförbundet, 1996).

1.1.4 Secondary settler

One of the most important physical processes in a WWTP is the separation of solids from water by gravity and density difference between solids and liquid. This is done in settler basins prior the biological reactor in the primary clarifier and after the biological process in a secondary clarifier, settler (Jeppsson, 1991). In this thesis focus will be on the secondary settler because this process will be a part of the model.

In activated sludge plants soluble organic matter transforms into biomass (sludge) and this process requires the biomass to be separated from the liquid before discharge in the recipient. This removal is done in a secondary settler followed the biological reactor. Some of the sludge is removed from the process but a large part is returned to the activated sludge reactor. This makes the separation a very important process. There are four different types of settling characteristics in a WWTP (Takács et al., 1991):

- Discrete particle settling: solids settle individually with little interactions with other particles.
- Flocculent particle settling: flocculation of solid particles through the water column that occurs in the upper layers of the secondary settler.
- Hindered settling: the particles settle as a unit by suspension.
- Compression settling: The weight of the particles adds to the system by compression.

To model the secondary settler in this thesis a mass balance model is used where the settling velocity is described by a double exponential function by Takács et al. (1991), (see section 2.2).

1.1.5 ASM1

In 1983 the International Water Association (IWA), formerly IAWQ, formed a task group to promote development and facilitate the application of practical models for design and operation of biological wastewater treatment systems. The goals were to review existing models and to find the simplest model realistically describing the activated sludge process. In 1987 the work resulted in *the IAWQ Activated Sludge Model no. 1 (ASM1)*, (Jeppsson, 1996). The model has been extended since then but the original model is probably still the most widely used for describing wastewater treatment all over the world and it can be considered as a “state-of-the-art” model for biological wastewater treatment when phosphorus removal is not considered.

1.2 OBJECTIVES

The aim of this thesis was to calibrate and validate the ASM1 to Henriksdals wastewater treatment plant. By extending and rebuilding a simulation benchmark from COST (European Co-operation in the field of Scientific and Technical Research) in MATLAB/Simulink adjustments to Henriksdal was made. Two measuring campaigns at Henriksdal were also performed to gather information needed for the calibration and validation of the model.

2 LITERATURE REVIEW

2.1 THE ASM1 STRUCTURE

The structure of the ASM1 is based on ordinary differential equations. These equations describe the dynamic changes in COD (Chemical Oxygen Demand) and nitrogen throughout the process. To define the different components in the model, COD and nitrogen are divided into fractions that are represented by state variables. There are also kinetic and stoichiometric parameters to describe the dynamic changes in the model.

2.1.1 State variables

COD is selected as the most suitable parameter for defining the carbon substrates because it is a link between electron equivalents in the organic substrate, the biomass and oxygen utilised. There are seven COD fractions in the ASM1 based on solubility, biodegradability, biodegradation rate and viability (biomass), see figure 2.

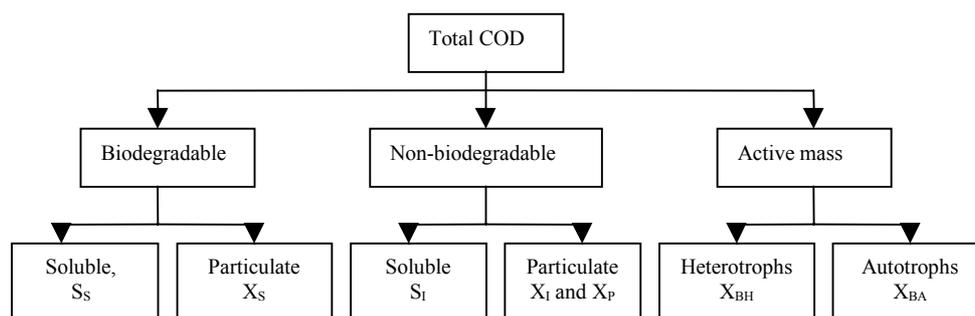


Figure 2. COD fraction in ASM1 (figure modified from Petersen 2000).

These variables are denoted as follows (Jeppsson, 1996):

- Readily biodegradable substrate, S_s
- Slowly biodegradable substrate, X_s
- Soluble inert organic material, S_i
- Particulate inert organic material, X_i
- Particulate products arising from biomass decay, X_p
- Active heterotrophic biomass, X_{BH}
- Active autotrophic biomass, X_{BA}

As the COD, the nitrogenous material in the wastewater is divided into fractions, see figure 3.

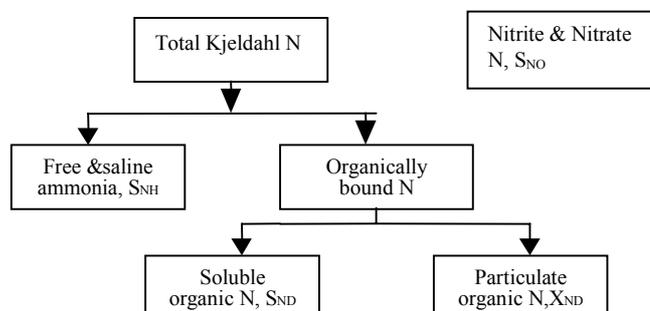


Figure 3. Nitrogen fractions in ASM1 (figure modified from Jeppsson, 1996).

There are four nitrogen fractions based on solubility, biodegradability and biodegradation rate (Petersen, 2000):

- Ammonia nitrogen, S_{NH}
- Nitrate + nitrite nitrogen, S_{NO}
- Soluble organic nitrogen, S_{ND}
- Particulate organic nitrogen, X_{ND}

There are two more components described in the ASM1 as state variables, oxygen concentration, S_O and alkalinity, S_{ALK} .

2.1.2 Model parameters

There are two different kinds of parameters in the ASM1, stoichiometric parameters and kinetic parameters. Both kinds have to be determined to make an accurate model.

The stoichiometric parameters are:

- Heterotrophic yield, Y_H (g X_{BH} COD formed (g COD utilised)⁻¹)
- Autotrophic yield, Y_A (g X_{BA} COD formed (g N utilised)⁻¹)
- Fraction of biomass yielding particulate products, f_P (dimensionless)
- Mass N/mass COD in biomass, i_{XB} (g N (g COD)⁻¹ in biomass (X_{BA} and X_{BH}))
- Mass N/mass COD in products from biomass (i_{XP} , g N (g COD)⁻¹ in X_P)

The kinetic parameters are:

- Heterotrophic maximum specific growth rate, μ_H (day^{-1})
- Heterotrophic decay rate, b_H (day^{-1})
- Half-saturation coefficient for heterotrophs, K_S (g COD m^{-3})
- Oxygen half-saturation coefficient for heterotrophs, K_{OH} ($\text{g O}_2 \text{ m}^{-3}$)
- Nitrate half-saturation coefficient for denitrifying heterotrophs, K_{NO} ($\text{g NO}_3\text{-Nm}^{-3}$)
- Autotrophic maximum specific growth rate, μ_A (day^{-1})
- Autotrophic decay rate, b_A (day^{-1})
- Oxygen half-saturation coefficient for autotrophs, K_{OA} ($\text{g O}_2 \text{ m}^{-3}$)
- Ammonia half-saturation coefficient for autotrophs, K_{NH} ($\text{g NH}_3\text{-N m}^{-3}$)
- Correction factor for anoxic growth of heterotrophs, η_g (dimensionless)
- Ammonification rate, k_a ($\text{m}^3 (\text{g COD day})^{-1}$)
- Maximum specific hydrolysis rate, K_h ($\text{g X}_S (\text{g X}_{BH} \text{ COD day})^{-1}$)
- Half-saturation coefficient for hydrolysis of slowly biodegradable substrate, K_X ($\text{g X}_S (\text{g X}_{BH} \text{ COD})^{-1}$)
- Correction factor for anoxic hydrolysis, η_h (dimensionless)

The parameters μ_H , b_H , b_A , K_h , K_X , k_a and μ_A are temperature dependent Jeppsson, 1996).

2.1.3 Dynamic processes

There are four different types of dynamic processes in ASM1 (Jeppsson, 1996). The growth rates in these processes are described by generalised monod kinetics. The basic monod kinetic is given by:

$$\mu = \mu_{\max} \frac{S}{S + K_S} \quad (1)$$

where

μ = specific growth rate

μ_{\max} = maximum specific growth rate

S = substrate concentration

K_S = half saturation constant

2.1.3.1 Growth of biomass

Aerobic growth of heterotrophic biomass

The degradation of soluble readily biodegradable substrate under consumption of oxygen leads to growth of heterotrophic biomass. Ammonia is used as nitrogen source and incorporated into cell mass. The process is generally the main contributor to the production of biomass and removal of COD. The process is described with the monod function (COST, 1998):

$$\rho_1 = \mu_H \left(\frac{S_S}{K_S + S_S} \right) \left(\frac{S_O}{K_{OH} + S_O} \right) X_{BH} \quad (2)$$

Anoxic growth of heterotrophic biomass

In the absence of oxygen the heterotrophic organisms use S_S as substrate and nitrate as their electron acceptor. The result is growth of biomass and production of nitrogen gas due to denitrification. The process is also described with the monod function and the kinetic rate expression is multiplied by a factor $\eta_g (<1)$ (COST, 1998):

$$\rho_2 = \mu_H \left(\frac{S_S}{K_S + S_S} \right) \left(\frac{K_{OH}}{K_{OH} + S_O} \right) \left(\frac{S_{NO}}{K_{NO} + S_{NO}} \right) \eta_g X_{BH} \quad (3)$$

Aerobic growth of autotrophic biomass

Ammonia nitrogen is oxidised to nitrate via nitrification. This results in growth of autotrophic biomass and a high oxygen demand. Ammonia is also the nitrogen source for synthesis and incorporated into the cell mass. This process as well is described with a monod function (COST, 1998):

$$\rho_3 = \mu_A \left(\frac{S_{NH}}{K_{NH} + S_{NH}} \right) \left(\frac{S_O}{K_{OA} + S_O} \right) X_{BA} \quad (4)$$

2.1.3.2 Decay of biomass

Decay of heterotrophic biomass

The organisms are assumed to die at a certain rate and part of the decay results in release of slowly biodegradable substrate. The rest is considered non-biodegradable and adds to the X_p fraction. The process is assumed to have the same rate during aerobic, anoxic and anaerobic conditions and is described by (COST, 1998):

$$\rho_4 = b_H X_{BH} \quad (5)$$

Decay of autotrophic biomass

This process is described in a similar way as the decay of heterotrophs (COST, 1998):

$$\rho_5 = b_A X_{BA} \quad (6)$$

2.1.3.3 Ammonification of soluble organic nitrogen

Biodegradable soluble organic nitrogen is converted to free and saline ammonia in a first order process (COST, 1998):

$$\rho_6 = k_a S_{ND} X_{BH} \quad (7)$$

2.1.3.4 Hydrolyses

Hydrolyses of entrapped organic matter

Slowly biodegradable substrate entrapped in the sludge breaks down extracellularly. This process produces readily biodegradable substrate that is available for the organisms for growth. It occurs both under aerobic and anoxic conditions and is modelled on surface reaction kinetics. Under anoxic conditions the rate is reduced with a factor η_h (<1) (COST, 1998):

$$\rho_7 = k_h \frac{X_S / X_{BH}}{K_X + (X_S / X_{BH})} \left[\left(\frac{S_O}{K_{OH} + S_O} \right) + \eta_h \left(\frac{K_{OH}}{K_{OH} + S_O} \right) \left(\frac{S_{NO}}{K_{NO} + S_{NO}} \right) \right] X_{BH} \quad (8)$$

Hydrolysis of entrapped organic nitrogen

This process of biodegradable particulate organic nitrogen breaking down to soluble organic nitrogen is described as the rate for the entrapped organics (COST, 1998):

$$\rho_8 = k_h \frac{X_S / X_{BH}}{K_X + (X_S / X_{BH})} \left[\left(\frac{S_O}{K_{OH} + S_O} \right) + \eta_h \left(\frac{K_{OH}}{K_{OH} + S_O} \right) \left(\frac{S_{NO}}{K_{NO} + S_{NO}} \right) \right] X_{BH} \frac{X_{ND}}{X_S} \quad (9)$$

2.1.4 Model formulation

The final description of the model is based on the dynamic processes and builds a set of ordinary differential equations.

The first one represents the behaviour of the heterotrophic biomass concentrations and shows how it is affected by the three processes; aerobic growth, anoxic growth and decay (Jeppsson, 1996):

$$\frac{dX_{BH}}{dt} = \rho_1 + \rho_2 - \rho_4 \quad (10)$$

The second one describes the autotrophic biomass concentration and is slightly simpler because the autotrophs do not grow in anoxic environment (Jeppsson, 1996):

$$\frac{dX_{BA}}{dt} = \rho_3 - \rho_5 \quad (11)$$

The third differential equation describes how the readily biodegradable substrate behaves. The concentration is reduced by growth of heterotrophic bacteria and increased by hydrolysis of slowly biodegradable substrate (Jeppsson, 1996):

$$\frac{dS_s}{dt} = \rho_7 - (\rho_1 + \rho_2) \quad (12)$$

The next differential equation is describing how the slowly biodegradable substrate is increasing by the recycling of dead bacteria and decreasing by the hydrolysis process (Jeppsson, 1996):

$$\frac{dX_s}{dt} = (1 - f_p)(\rho_4 + \rho_5) - \rho_7 \quad (13)$$

The fifth model equation describes how the concentration of inert particulate products is arising from biomass decay (Jeppsson, 1996):

$$\frac{dX_p}{dt} = f_p(\rho_4 + \rho_5) \quad (14)$$

The sixth equation is describing how the concentration of particulate organic nitrogen behaves similar to the slowly biodegradable substrate (Jeppsson, 1996):

$$\frac{dX_{ND}}{dt} = (i_{XB} - f_p i_{XP})(\rho_4 + \rho_5) - \rho_7 \quad (15)$$

Soluble organic nitrogen is affected by ammonification and hydrolysis which shows in the seventh equation (Jeppsson, 1996):

$$\frac{dS_{ND}}{dt} = \rho_7 - \rho_6 \quad (16)$$

Ammonia is used as the nitrogen source in all growth processes of micro-organisms and the ammonia concentration is affected of all these processes. The fact that ammonia concentration also is decreased by nitrification and increased by ammonification leads to a very complex equation (Jeppsson, 1996):

$$\frac{dS_{NH}}{dt} = -i_{XB}(\rho_1 + \rho_2) + \rho_6 - \left(i_{XB} + \frac{1}{Y_A}\right)\rho_3 \quad (17)$$

The concentration of nitrate is increased by nitrification and decreased by denitrification and this process is described by the ninth equation (Jeppsson, 1996):

$$\frac{dS_{NO}}{dt} = \frac{1}{Y_A}\rho_3 - \left(\frac{1 - Y_H}{2.86Y_H}\right)\rho_2 \quad (18)$$

(The number 2.86 represent the oxygen equivalent for conversion of nitrate nitrogen to nitrogen gas.)

The next model equation describes the oxygen concentration in the wastewater. The concentration is decreased by aerobic growth of biomass (Jeppsson, 1996):

$$\frac{dS_O}{dt} = -\left(\frac{1 - Y_H}{Y_H}\right)\rho_1 - \left(\frac{4.57 - Y_A}{Y_A}\right)\rho_3 \quad (19)$$

(The number 4.57 represent the theoretical oxygen consumption for oxidation of ammonium nitrogen to nitrate nitrogen.)

The last model equation describes the dynamics of the alkalinity change. The inclusion of alkalinity in the model is to detect problems with changes in pH without including pH in the model (Petersen, 2000).

$$\frac{dS_{ALK}}{dt} = -\frac{i_{XB}}{14} \rho_1 + \left(\frac{1-Y_H}{14.286Y_H} - \frac{i_{XB}}{14} \right) \rho_2 - \left(\frac{2}{14Y_A} + \frac{i_{XB}}{14} \right) \rho_3 + \frac{1}{14} \rho_6 \quad (20)$$

In appendix A the model equations are described in a process matrix.

2.1.5 Model restrictions

To be able to make an accurate model of the activated sludge process some simplifications and assumptions must be made, both associated with the physical system and with the mathematical model itself. A number of restrictions are made within the ASM1. Some examples are that the system is assumed to operate at constant temperature and that the pH is constant and near neutrally. The limitation of inorganic compounds for removal of organic substrate is not considered, so care must be taken to make sure that sufficient quantities are present. There are many more restrictions within the model which simplifies it from reality (Jeppsson, 1996).

2.2 SETTLER MODEL STRUCTURE

The model of the secondary settler is based on a one- dimensional ten-layer mass balance model based on the continuity equation presented by Vitasovic in 1986 (Jeppsson, 1996). The concentration profile of the solids is predicted by dividing the settler into ten layers of equal thickness (fig. 4).

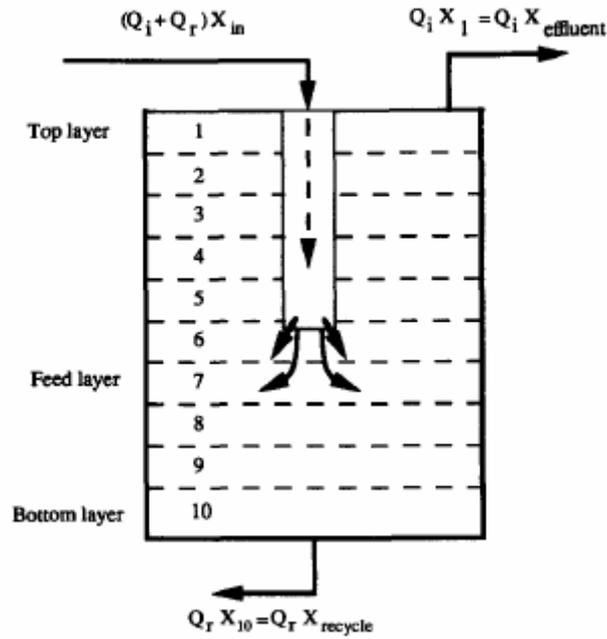


Figure 4. Layered settler model (Takács et al., 1991).

The layers are divided into five different groups according to their position relative to the feed layer:

- Top layer
- Layers above feed layer
- Feed layer
- Layers below feed layer
- Bottom layer

To calculate the settling velocities in the layers a double-exponential settling velocity function by Takács et al. (1991) is used (eq. 21). This function is applicable to both hindered and flocculent settling conditions (COST, 1998). The first part of the equation calculates the settling velocity for the large, well flocculating particles. The other part is a velocity correction factor to account for the smaller, slower particles.

$$v_{sj} = v_0 e^{-r_h X_j^*} - v_0 e^{-r_p X_j^*} \quad 0 \leq v_{sj} \leq v_0' \quad (21)$$

where

v_{sj} = settling velocity in layer j

v_0 = maximum settling velocity

v_0' = limit of v_0

r_h = settling parameter characteristic of the hindered settling zone

r_p = settling parameter characteristic of low solids concentration

$$X_j^* = X_j - X_{\min}$$

X_j = suspended solids concentration in layer j

X_{\min} = minimum attainable suspended solids concentration

$$X_{\min} = f_{ns} X_{in}$$

X_{in} = mixed- liquor suspended solids entering the settler

f_{ns} = non- settable fraction of X_{in}

2.3 MODEL CALIBRATION

Model calibration is the adaptation of the model to fit a certain set of information obtained from the wastewater treatment plant. Even though the ASM1 has been around for many years there is no fully developed model calibration procedure to be found in literature. One typical problem related to the calibration of the ASM1 is that more than one combination of parameters can give the same description of the collected data. This indicates that a calibration procedure based on changing the parameters by trial and error is not advisable. It becomes important to gather as much information as possible to find the realistic parameter combination (Petersen, 2000).

To obtain a good model calibration a general procedure can be followed. Petersen (2000) presents some steps that can be used as guideline to perform a model calibration. At first there are some suggestions of what data has to be obtained such as measurements of the concentrations of the wastewater and other design data such as volumes and flows. Information about the hydraulics of the treatment plant is also necessary to collect. Such information could be the result of a tracer test. The state variables and the model parameters can then be calculated and calibrated.

2.3.1 ASM1, state variables

One of the most difficult parts of calibrating the ASM1 is to characterise the wastewater fractions. Wastewater can be characterised either with physical-chemical methods or with biological methods. To be able to characterise all the fractions the two methods must be combined (Petersen, 2000). The physical-chemical method is build on the fact that the wastewater can be separated into different components via separation methods and the biological methods are based on the rates of degradation.

2.3.1.1 COD components

The total COD consists of different components according to:

$$\text{COD}_{\text{tot}} = S_I + S_S + X_I + X_S (+ X_{\text{BH}} + X_{\text{BA}} + X_P)$$

The biomass concentrations X_{BH} and X_{BA} is often negligible and X_P is assumed to be zero in the influent (Petersen, 2000).

Inert soluble organic matter, S_I

The inert organic matter is present in the influent but also produced during the activated sludge process. Influent S_I could be estimated as 90- 95 % of effluent COD. It could also be determined as the soluble fraction of COD, COD_{sol} , remaining after long-term BOD (Biological Oxygen Demand) test with the influent wastewater (Petersen, 2000). According to Xu (1996) the inert organic matter in the influent is equal to the filtered COD in the effluent.

Readily biodegradable substrate, S_S

The soluble COD fraction excluding S_I is mostly considered representing the readily biodegradable substrate S_S :

$$S_S = \text{COD}_{\text{sol}} - S_I$$

According to Petersen (2000) the soluble fractions passes through a 0,1 μm filter.

Inert suspended organic matter, X_I

The X_I concentration could be determined as the particulate COD remaining after long-term BOD test. The assumption that no X_I is produced during the test has to be made to use this method. The assumption may be questionable since X_I will be produced due to decay during the BOD test and corrections for this must be considered (Petersen, 2000).

In 1996 STOWA made guidelines for characterisation of wastewater and defined X_I as the difference between particulate COD and X_S (Petersen, 2000).

Slowly biodegradable substrate, X_S

According to the study by STOWA the X_S fraction can be estimated based on a long-term BOD test based on the difference between BOD and S_S by the following equation:

$$X_S = \frac{BOD}{1 - Y_H} - S_S \quad (22)$$

where Y_H is 0.2.

Otherwise if the biomass concentrations are negligible the X_S concentration can be calculated from the mass balance if the other fractions are known.

2.3.1.2 Nitrogen compounds

All the nitrogen compounds can be determined by physical-chemical methods according to figure 5. The total nitrogen also consists of different fractions:

$$N_{\text{tot}} = X_{\text{ND}} + S_{\text{ND}} + S_{\text{NH}} + S_{\text{NO}}$$

Nitrate- nitrite nitrogen, S_{NO}

S_{NO} is simply the nitrate and nitrite concentrations, which are easily measured.

Ammonium, S_{NH}

S_{NH} is the ammonium concentration in the wastewater and is also easy to measure.

Particulate biodegradable organic nitrogen, X_{ND}

X_{ND} is the particulate fraction of the Kjeldahl nitrogen and is also possible to find through chemical analyses.

Soluble biodegradable organic nitrogen, S_{ND}

S_{ND} is found by subtracting the other nitrogen fractions from the total nitrogen in the nitrogen mass balance.

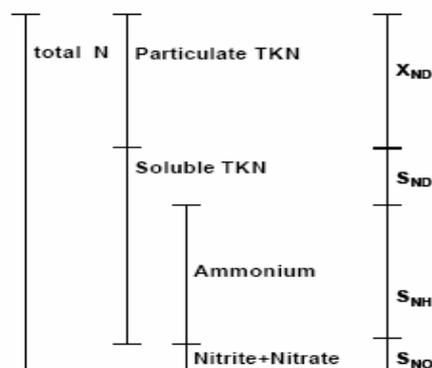


Figure 5. Nitrogen fractions in ASM1 (Petersen 2000).

2.3.2 ASM1, model parameters

It is possible to determine the model parameters using physical- and biological methods but this is often time consuming and expensive (Petersen, 2000). A literature review has been made to find the values others have used for the different parameters to be aware of in what range it can be expected to find them (table 1). This will make it easier to find the appropriate values for Henriksdal during the calibration of the model. Many of the parameters are temperature dependent and care has to be taken to this fact.

Table 1. Values of the different parameters from literature review.

	1*	2*	3*	4*	5*	6*	7*	8*	9*	10*
Y_H	0.67	0.67	0.67	-	0.67	0.67	0.67	0.62	0.6	-
Y_A	0.24	0.24	0.24	-	0.24	0.24	0.24	0.24	0.24	-
f_p	0.08	0.08	-	-	-	-	0.08	0.08	0.08	-
i_{XB}	0.086	0.086	-	-	-	-	0.08	0.08	0.08	-
i_{XP}	0.06	0.06	-	-	-	-	0.06	0.06	0.06	-
μ_H	6	3	6	6	6	6	6	4.59	6	2.5
b_H	0.62	0.2	0.62	0.62	0.62	0.62	0.3	0.635	0.5	-
K_S	20	20	3	5	3	5	10	20	20	11
K_{OH}	0.2	0.2	0.2	0.4	1	0.2	0.2	0.33	1	-
K_{NO}	0.5	0.5	0.5	0.5	0.5	0.1	0.5	0.5	0.5	-
μ_A	0.8	0.3	0.86	1.4	0.8	0.9	0.5	0.657	0.65	0.43
B_A	0.2	0.1	0.13	0.15	0.15	0.13	0.05	0.098	0.15	-
K_{OA}	0.4	0.4	0.1	0.4	0.4	0.4	0.4	0.4	0.5	-
K_{NH}	1	1	0.4	1	0.8	0.8	1	1	1	-
η_g	0.8	0.8	0.6	0.6	0.6	-	0.8	1	0.6	-
k_a	0.08	0.04	-	-	-	0.05	0.05	0.092	0.08	-
K_h	3	3	-	-	-	2	3	1.72	2	-
K_X	0.03	0.01	-	-	-	0.02	0.01	0.02	0.02	-
η_h	0.4	0.4	-	-	-	0.5	0.8	0.32	0.4	-

1* ASM1 default values for 20°C, Jeppsson, 1996, 2* ASM1 default values for 10°C, Jeppsson, 1996, 3* Dupont, Sinkjaer, 1994, 4* Pedersen, Sinkjaer, 1992, 5* Xu, Hultman, 1996, 6* Sota et al., 1994, 7* COST, 1998, 8* Abusam, 2000, 9* Lesouef et al. 1992, 10* Stokes et al., 1993

Investigations have been made by Jeppsson, 1996 and by Petersen, 2000 for the ASM1 to find which parameters are the most important ones to determine accurately. This has been done by looking at the sensitivity, variability and uncertainty of the parameters. The parameters that were found to be most important are: b_H , μ_H , k_h , K_X , η_h and μ_A according to Jeppsson (1996). In the report by Petersen (2000) Y_H , f_p , b_H and $X_{I, \text{influent}}$ are the most important parameters for long-term predictions and μ_H , μ_A , η_g , η_h , K_S , K_{NH} , K_{OH} and K_{OA} for short-term predictions. Some parameters are correlated which means that different sets of parameters gives about the same model behaviour. These parameters can not be changed separately but must be tuned simultaneously. The growth and decay rates correlated with oxygen demand is one example; if both growth- and decay rates are increased it will lead to identical net

growth but the oxygen demand will increase. Another example is yield and growth rates, if both are increased they might outbalance each other but the oxygen consumption will increase. The third example is that a high yield and low concentration of heterotrophs in the influent wastewater is showing the same results as a low yield and a high concentration of heterotrophs (Jeppsson, 1996).

2.3.3 Settler model parameters

The settler parameters needed to be determined in the model are v_0 , v_0' , r_h , r_p , f_{ns} , X_{in} and the position of the feedlayer. It is possible to determine the parameters by laboratory experiments and non-linear optimisation techniques (Takács et al., 1991). Settling column analyses can be used to measure the minimum effluent suspended solids, f_{ns} . By diluting the wastewater to a concentration of 1-2 g/l and measure the settling velocity of the large individual floc particles the maximum practical settling velocity, v_0' can be obtained. There is also possible to measure the hindered settling velocity parameters, v_0 and r_h through a series of column settling tests (Kennedy, 1991). In the study by Takács et al. (1991) a non-linear dynamic optimisation package, SIMUSOLV, was used to obtain the settling parameter r_p . According to Petersen (2000) the settler model parameters can be found by trial and error in the steady-state calibration.

2.3.4 Steady state model calibration

As a first step in the model calibration average data can be used to represent a steady-state for the model to determine the parameters responsible for long-term behaviour. The data used in this step is the average of the data obtained from the wastewater treatment plant. The relevant parameters that can be determined are; Y_H , f_p , b_H and X_I in the influent (Petersen, 2000). These parameters are somehow correlated and successful studies have been made where Y_H and f_p are fixed and X_I and b_H determined from the steady-state data (Petersen, 2000). The steady-state calibration is not enough to calibrate the entire model though the real variations in the input are much faster than the slow process of the average input. It is however very useful for determination of the initial conditions for dynamic calibration.

2.3.5 Dynamic model calibration

A dynamic model calibration is needed to describe and predict fast and dynamic situations. According to Petersen (2000) the most important function of the dynamic calibration is to estimate the parameters that predict dynamic situations. These parameters are the specific growth rates, μ_H and μ_A . Data series are needed for the dynamic calibration and samples have to be collected from the wastewater treatment plant. Often a measuring campaign has to be carried out to get data with the right sampling frequency. The needed frequency should be chosen to at least five times faster than the hydraulic retention time and the minimum duration time of the campaign should be 3-4 times the hydraulic retention time. There might also not be enough with the data collected from the full-scale WWTP and lab-scale experiments must be performed to obtain the data needed.

2.4 TRACER STUDIES

In order to get information about the process dynamics of a WWTP a tracer study can be carried out. A tracer study will provide information about the degree of mixing in the aerated tanks and the overall hydraulic retention time (HRT). The basic theory is that a slug of tracer is injected at the inlet and the concentration of the tracer is measured at the outlet as a function of time (Shilton, 2001). The tracer has to be a conservative substance so the mass will not change throughout the process in the tanks and the whole injected amount can be recovered at the outlet.

One possible tracer is Rhodamine B that is a conservative, not reactive, strongly fluorescent substance. A small amount of Rhodamine B turns the activated sludge reactor red and as small concentrations as 0.01 ppb can be detected (Shilton, 2001). The outlet concentration is measured with a fluorometer connected to a computer, which makes it possible to obtain data with short time interval. The strong red colour makes it also possible to visually observe the spreading of the injected tracer in the reactor. Another alternative tracer is lithium chloride, LiCl. It is also conservative and normally only exists in very low concentration in a wastewater treatment plant (Borglund, 2004). The analyses can be made at Stockholm Vatten and concentrations of 20 µg/l can be detected with relatively good accuracy. The samples have to be collected manually at the outlet and analysed for lithium.

Most activated sludge reactors are not totally mixed and the entire volumes of the reactors are not used. To model this a series of smaller total mixed reactors are modelled. System identification can be used to estimate the number of reactors. A simulation program e.g. GPS-X can be used for this identification and the measured result from the tracer test is compared to a modelled tracer test. One problem with this method is that different models can give the same result because it is a non-linear system (Borglund, 2004).

2.5 COST BENCHMARK

A benchmark for evaluating control strategies for activated sludge plants has been developed by the European CO-operating in the field of Scientific and Technical Research, COST 682 Working Group No.2 and later overtaken by the COST 624 Working Groups. The benchmark is a simulation model describing and defining a plant layout with influent loads, test procedures and evaluation criteria. The benchmark can be used in different platforms and is independent of the software being used. The simulation in different systems should give the same results (COST, 1998). In this thesis MATLAB/Simulink is being used.

2.5.1 Benchmark plant layout

The original benchmark plant describes five biological reactors in series with one 10-layer secondary-settling tank. Tank one and two are unaerated and fully mixed and tank three, four and five are aerated. There is a recirculation flow from tank five to tank one and a return sludge flow from the bottom of the settler to tank one. The IAWQ's ASM1 is used for the biological process and the settling velocity function by Takács et al. (1991) is used to represent the settling process. The incoming wastewater is connected to the model through a m-file with the different fractions in columns as follows: time (days), S_I , S_S , X_I , X_S , X_{BH} , X_{BA} , X_P , S_O , S_{NO} , S_{NH} , S_{ND} , X_{ND} , S_{ALK} (moles/m³), T_{SS} (suspended material), Q (m³/day). All concentrations are in mg/l.

3 METHOD

3.1 MEASURING CAMPAIGNES

Two measuring campaigns have been carried out at Henriksdal to collect data needed to calibrate and validate the model. The sample points for each campaign were at the inlet to the activated sludge tank at line four and at the outlet from one of the secondary settler tanks also at lane four (point A and B in figure 6). Even though the seven lines have the same layout they run somehow differently. Line one gives better results than line seven. Line four was chosen because it is the middle one and probably best represents a mean value of the system. It would have been desirable to take samples from all lines but this was decided too time consuming and expensive. Line four also had aeration discs, which was in the middle of their lifetime. Line four seemed to represent the overall mean value of the basins best.

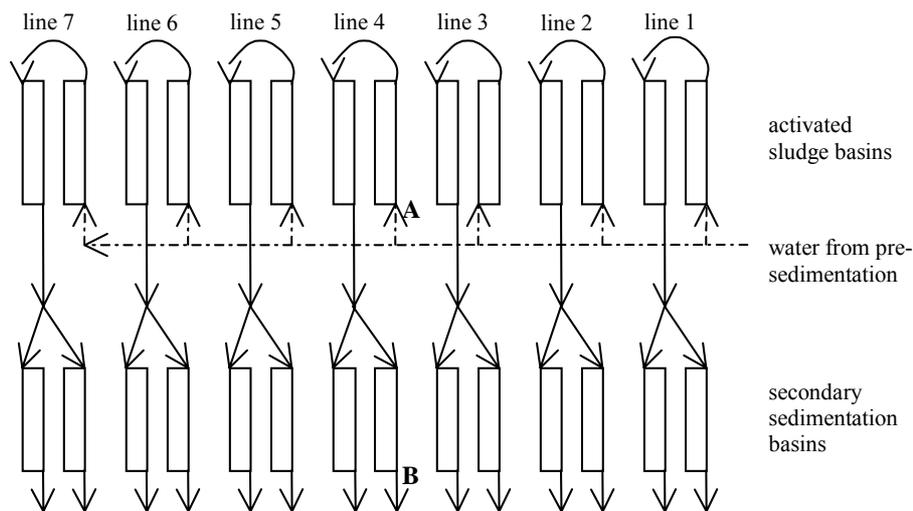


Figure 6. Layout over the biological step and the secondary sedimentation at Henriksdal WWTP. The sample points are marked A, inlet and B, outlet.

3.1.1 First measuring campaign

During the 7th to 10th of December 2004 the first measuring campaign was carried out. The theoretical hydraulic retention time was roughly estimated to about 15 hours and the sample frequency set to two hours during 72 hours to follow the criteria given by Petersen, 2000 (see section 2.3.5). Samples of 125 ml was collected every 15 minutes and added to a sample of one litre collected after two hours. The samples were then analysed for total nitrogen, soluble nitrogen, total COD, soluble COD, ammonium, nitrite and nitrate. The temperature and pH was also checked to see if they were about constant and neutral to fit the model restrictions. The data can be found in figure 7 and appendix B.

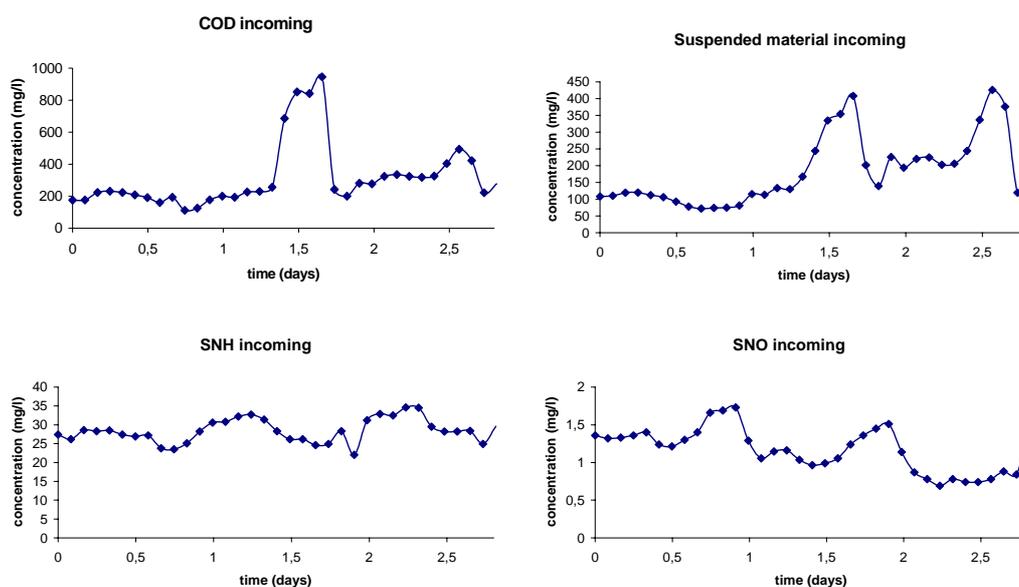


Figure 7. Measured incoming concentrations from measuring campaign one.

The week before and during the time period of the campaign the weather was snowy/rainy and mild, which lead to snow melting and very high flows into the wastewater treatment plant. In these somehow extreme conditions the plant is not working as well as during normal conditions, which leads to data that are not representative for the general performance of the plant. The high peaks of incoming COD and suspended material (fig. 7) might be because of very high increased load of COD and suspended material in to the treatment plant or a failure in the pre-sedimentation process during this period.

3.1.2 Second measuring campaign

The second measuring campaign was carried out the 24th to 28th of April 2005. The same sample volume and frequency were used as in campaign one. This time analyses were made for total-nitrogen, total-COD, ammonium, nitrate and suspended material for both sample points, see figure 8 and appendix B. During this time the weather was dry and stable which made the data more reliable and representative for the WWTP in the long run.

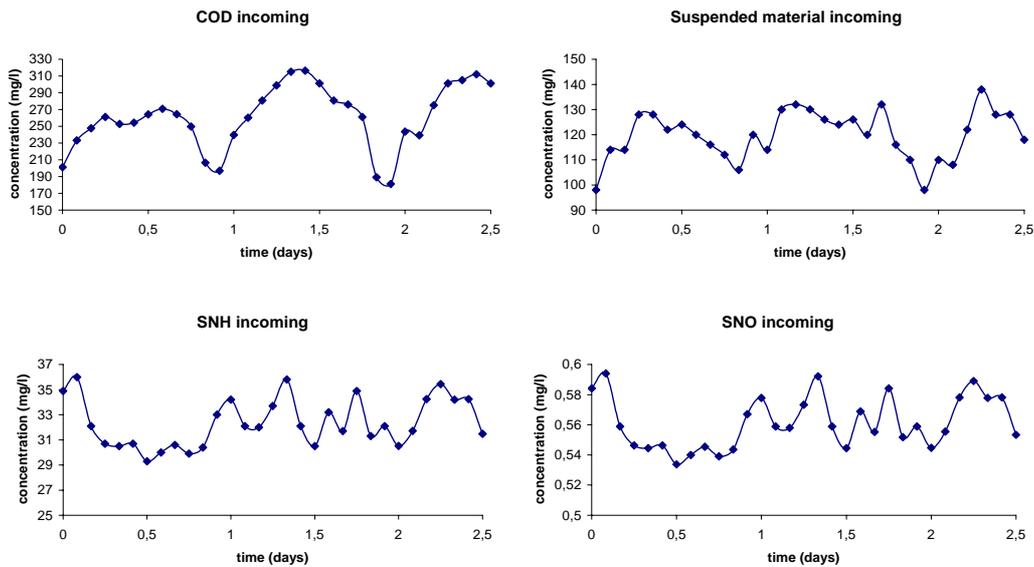


Figure 8. Measured incoming concentrations from measuring campaign two.

3.1.3 Analyses

The samples collected during each campaign were analysed as soon as possible. The total nitrogen, ammonium, nitrite, nitrate and COD were analysed with cuvette tests from HACH – LANGE and the suspended material by filtration of the water through a 1.6 μm glass microfibre filter. Of the samples from the incoming wastewater 50 ml were filtrated and 100 ml from the outgoing wastewater. The suspended material was calculated from the weight of the particles remaining in the filter. Because of the small amount of water passed through the filter the measuring error for the suspended material concentrations are very high. The analyses of the soluble fractions, ammonium, nitrite and nitrate were made after the filtration. Since the pore size of the filter are larger than recommended by Petersen (2000) the soluble fractions of COD and nitrogen might be too large and may contribute to the margin of error. The cuvettes were read in a photometer, LASA 30 which have a measure value accuracy of 2%. Because of not having access to the WWTP between four p.m. and seven a.m. the samples collected during this hours were not analysed until the morning after and the time delay might have affected the results especially the nitrogen tests. Some of the samples also had to be diluted to fit the restrictions of the cuvette tests and this might also affect the accuracy of some results. It is hard to estimate the accuracy of the measured concentrations, but a guess is that each value might have a relative error of $\pm 10\%$. Due to trouble with the analyses of total nitrogen and ammonium during the first campaign some of the samples were frozen and analysed after a couple of weeks and the defrost of these samples might have affected the results.

3.2 MODEL ADJUSTMENTS

The Benchmark model in Simulink was changed to fit the layout of Henriksdal, line four. Three more biological reactors were created and put in line with the others so

there were eight reactors in line. The first reactor is a mixing zone, volume 1400 m³. The second and third reactors are anoxic zones with a volume of 4500 m³ each. Reactors four, five and six are aerated zones with a volume of 4500 m³ each. Reactor seven was not aerated at the time of the measuring campaigns so it was modelled without aeration with a volume of 4500 m³. The last reactor is a deaeration zone with a volume of 710 m³. The aeration to the three aerated reactors was regulated by measuring the oxygen in the tank to set a oxygen setpoint that regulates the air into the reactor. This is done by an air-regulator that calculates a new K_{La} value as input to the reactor. K_{La} is an oxygen mass transfer coefficient for the rate of oxygen transferred per unit volume wastewater. From the last reactor the water is divided into a recirculation flow back to reactor one and to two different settling tanks. The flow is equal to both the settling tanks and both have the same design. They are 80 m long, 10 m wide and 5 m deep. From the bottom of each tank there is a return sludge flow back to reactor one and output sludge flow. There is also a flow out from the top of the settling tanks through a sand filter out of the model to the recipient.

Table 2. Average flows, theoretic HRT and temperatures from the two measuring campaigns.

	Campaign 1	Campaign 2
Average inflow	56 424 m ³ /day	33 357 m ³ /day
Average recirculation flow	154 656 m ³ /day	154 656 m ³ /day
Average total output sludge flow, Q _w	768 m ³ /day	576 m ³ /day
Average total return sludge flow, Q _r	18 144 m ³ /day	17 280 m ³ /day
Theoretical hydraulic retention time, HRT	15,7 h	26,7 h
Average temperature	14.0 °C	14.7 °C

As in the Benchmark model the ASM1 was used for modelling the biological reactors and the settling function from Takács et al. (1991) used for modelling the settling tanks. The layout of final Simulink model can be found in appendix C. The original programming code from Benchmark for the processes in the model were also adjusted to fit the new Simulink model and can be found at the HIPCON homepage, www.hipcon.org.

The concentrations of the incoming wastewater was calculated into fractions and put into a m-file according to COST (section 2.5.1.) which was connected as input to the model. To start with the fractions were calculated as follows:

- $S_I = 90\%$ of COD out
- $S_S = \text{COD}_{\text{sol}} - S_I$
- $X_S = \text{BOD}/(1-0,2) - S_S$, according to equation 22 where Y_H is 0,2
- $X_I = \text{COD}_{\text{tot}} - S_I - S_S - X_S$
- $S_{\text{NO}} = \text{nitrite} + \text{nitrate}$
- $S_{\text{NH}} = N_{\text{sol}} - S_{\text{NO}}$. This relation was used since the measured values of ammonium where sometimes higher then the total- nitrogen and did not seem reliable

- $X_{ND} = N_{tot} - S_{NO} - S_{NH}$

The other fractions were assumed to be zero in the influent and the alkalinity 7 moles/m³ according to Petersen, 2000. The flow rates and BOD value for the period were collected from the Henriksdal data system, WASTE. The output from the model is the ammonium-, nitrite + nitrate-, COD- and suspended material concentrations.

3.3 CALIBRATION AND VALIDATION FOR HENRIKSDAL

To calibrate the model a steady- state calibration and a dynamic calibration was performed. To check how the model fit for other data a validation was carried out after the calibration. Because of the unstable weather during the first measuring campaign the data from the second campaign was used for calibration, to develop a model that will fit the normal run of Henriksdal as good as possible. The first calculation of the fractions in the incoming wastewater was made based on the data from measuring campaign one since there was not enough information collected from the second measuring campaign to perform these calculations. The calculated percentages of total COD were then used for the second data set as initial values before the calibration. It would be desirable to make another campaign for validation data when the weather is more stable than during the first campaign. But due to lack of time the first data set will have to do for validation of the model.

3.3.1 Steady- state calibration

In the steady- state calibration the data from the inlet from the measuring campaign was averaged and used as input to the model. The default values of the model parameters were used and the temperature dependent ones was calculated as if they were linear dependent from 10 to 20 degrees. The temperature used in the model was 14.7 °C, which was the measured temperature at Henriksdal during the second measuring campaign. The different flows in the model were also adjusted according to data from the WASTE system during this time period. The first runs were then made and the settler parameters were changed to adjust the output of the model to the measured data of COD and suspended material. Also the fractions for calculating the amount of sludge composed in the tanks were changed to get the right amount of sludge out. When the settler parameters were adjusted the X_I and the X_S fractions were changed in the incoming wastewater to make a better fit of the model. This change adjusted the outgoing S_{NH} and S_{NO} . The final step of the steady- state calibration was to adjust b_H which also lead to changes in S_{NH} and S_{NO} . When this did not make the model fit the outgoing data desirable, K_{NH} and K_{OH} were also changed which lead to changes in S_{NH} and S_{NO} . When the model had reached steady- state the concentrations in each zone and settler were used as initial concentrations for the dynamic calibration.

3.3.2 Dynamic model calibration

For the dynamic calibration the data from the measuring campaign two was used as input to the model. The fractions were calculated according to the percentages from the steady- state calibration and the initial values in the basins were changed to the values they had reached after steady- state was achieved. The theoretical hydraulic retention time was roughly calculated to 26 h and the modelled output data compared visually to the measured output data. The first idea was to use the Simulink Parameter estimation toolbox to estimate the important parameters for the dynamics of the model. Unfortunately this attempt never worked out so a small program was created to perform loops to test different parameter values of two parameters at the time. Some of the parameters were changed by trial and error. This procedure was repeated until the best fit was obtained. Focus was on the outgoing ammonium and nitrite + nitrate since these parameters are among the most important ones for the WWTP but also suspended material and COD were considered in the calibration.

3.3.3 Validation

In order to perform the validation the flows in the model were changed to fit the first measuring campaign and the data from the first campaign used. Otherwise the parameters and concentrations obtained after the dynamic calibration were used during the validation. The model output was compared to the measured output data from campaign one and the theoretical HRT was 15 h.

4 RESULTS

4.1 RESULTS OF THE STEADY- STATE CALIBRATION

The steady- state calibration was run for 30 days until steady- state was almost reached. It was not possible to get all four outputs to reach steady values within this time range but the changes in output were so small after 30 days that they were assumed to make little difference.

In table 3 the values of incoming COD fractions are shown. The values are calculated according to the list on page 23 and calibrated during the s-s calibration.

Table 3. Calculated and calibrated fractions of incoming COD.

COD fractions in incoming	Percentages of total COD
S_I	8
S_S	21.8
X_I	8.2
X_S	62

The final parameter values after s-s calibration are shown in table 4.

Table 4. Values after steady- state calibration.

ASMI parameter	Values after s-s calibration	Settler parameters	Values after s-s calibration
Y_H	0.5	$V0_max$	250
Y_A	0.24	$\bar{V}0$	474
f_p	0.08	r_h	0.00028
i_{XB}	0.086	r_p	0.006
i_{XP}	0.06	f_ns	0.00228
μ_H	4.7	\bar{X}_t	3000
b_H	0.04	feedlayer	4
K_S	20		
K_{OH}	1	X_ITSS	0.85
K_{NO}	0.5	X_STSS	0.85
μ_A	0.65	X_BHTSS	0.85
B_A	0.147	X_BATSS	0.85
K_{OA}	0.4	X_PTSS	0.85
K_{NH}	0.5		
η_g	0.8		
k_a	0.0588		
K_h	3		
K_X	0.01		
η_h	0.2		

The concentrations reached after 30 days are shown in table 5 and compared to measured average values. COD, suspended material and S_{NO} reached very good steady-state concentrations compared to the measured average concentrations. S_{NH} increased little too much and reach a slightly too high steady-state concentration.

Table 5. Concentrations reached after steady-state calibration and measured concentrations from campaign two.

	Average measured concentration (mg/l)	Modelled steady-state concentration (mg/l)
COD	32.3	32.7
Susp	10.0	10.2
SNH	0.95	1.23
SNO	4.8	4.9

The ammonium factor, S_{NH} increased to desired value and steady- state very slowly but was almost constant at the end of the time period. The nitrite + nitrate factor, S_{NO} was decreasing slowly during the whole period but so little that it was considered as steady. The COD and suspended material reached desired values rapidly and were steady during the 30 days modelling period, see figure 9.

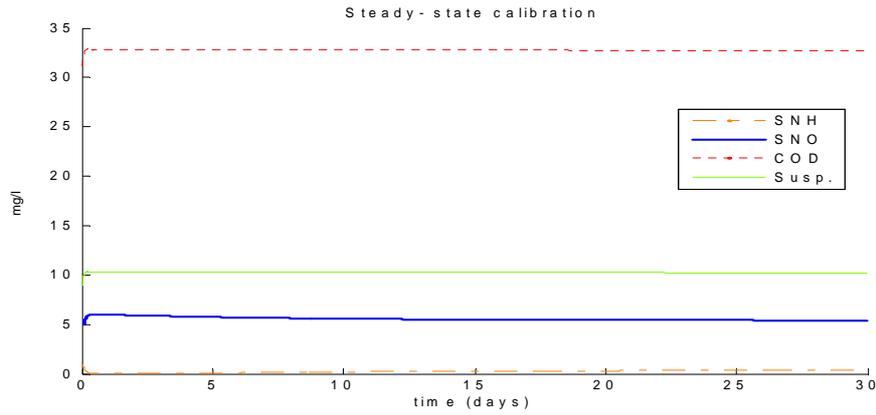


Figure 9. Results from the steady-state calibration.

4.2 RESULTS OF THE DYNAMIC MODEL CALIBRATION

The outputs from the dynamic calibration were compared to measured outputs from measuring campaign two. The parameter values obtained from the dynamic calibration are shown in table 6.

Table 6. Values after dynamic calibration.

ASM1 parameter	Values after dynamic calibration
Y_H	0.5
Y_A	0.26
f_P	0.08
i_{XB}	0.086
i_{XP}	0.06
μ_H	4.7
b_H	0.04
k_S	30
K_{OH}	0.9
K_{NO}	0.3
μ_A	0.57
b_A	0.2
K_{OA}	0.2
K_{NH}	0.3
η_g	0.8
k_a	0.006
K_h	3.5
K_X	0.05
η_h	0.3

The results for S_{NH} , S_{NO} , COD and suspended material are shown in figure 10- 13.

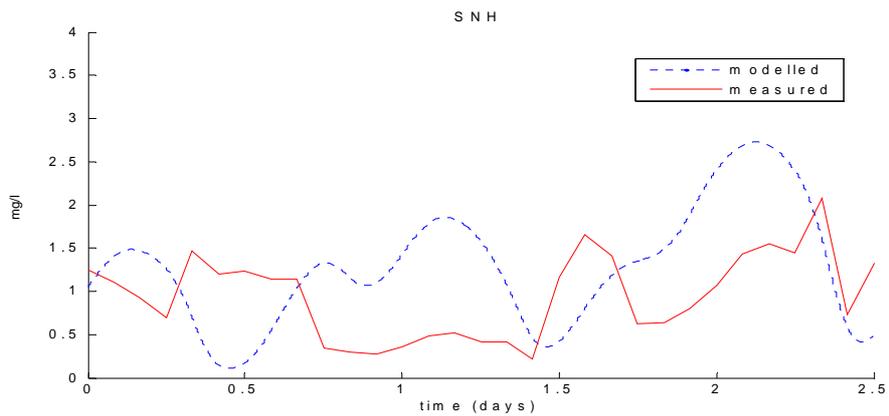


Figure 10. Modelled and measured S_{NH} after dynamic calibration.

In figure 10 the results of the modelled and measured ammonium concentrations are shown. The curves start out at about the same values and have about the same amplitudes. The curves shows daily variations but are not entirely in phase with each other.

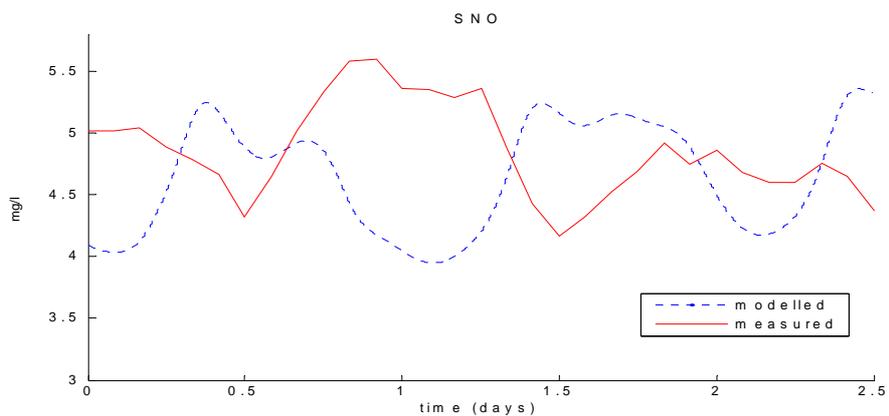


Figure 11. Modelled and measured S_{NO} after dynamic calibration.

The modelled and measured S_{NO} curves in figure 11 do not follow each other very well but have about the same frequency. The amplitudes of the curves are about the same size and in the same interval. Both curves have the same frequency in the daily variations but are not in phase with each other which implies that the theoretical hydraulic retention time are not the same as the real retention time.

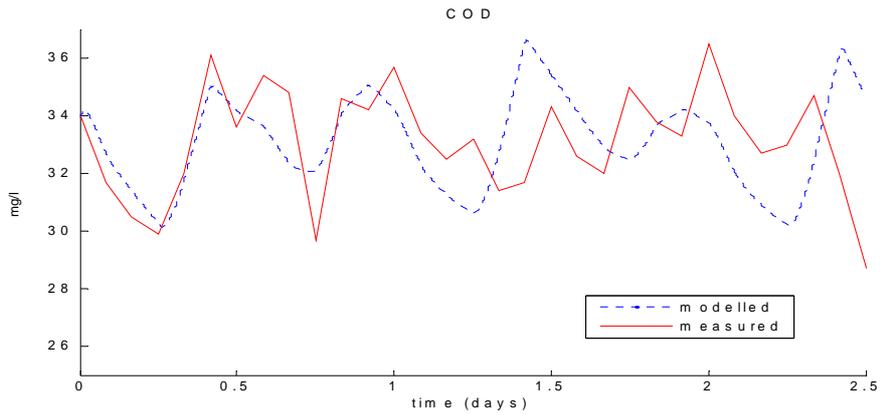


Figure 12. Modelled and measured COD after dynamic calibration.

The measured COD curve in figure 12 varies fast over time but within a small range. These changes might be within the relative error range and focus was on the average value. The modelled curve stays in the same range as the measured curve and has about the same average value.

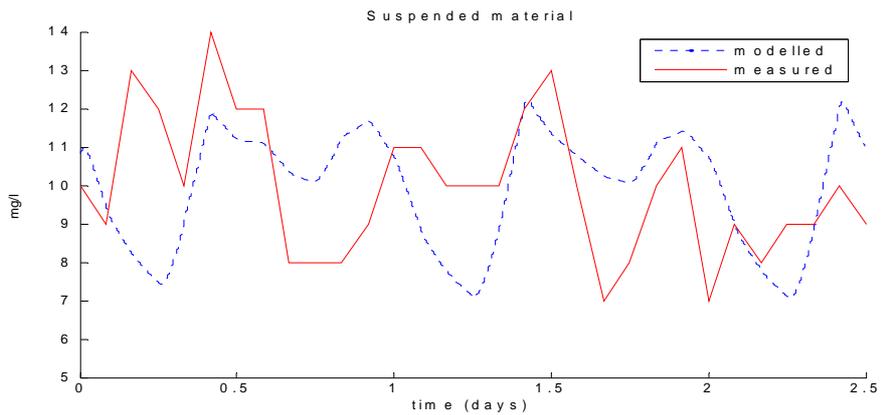


Figure 13. Modelled and measured suspended material after dynamic calibration.

The measured suspended material in figure 13 varies a lot over time but stays within a range from 7-14 mg/l. Since the relative error is quite large for suspended material only the mean value is of interest. The modelled curve does not vary as much and stays in the same range as the measured curve. The mean value of the two curves is about the same.

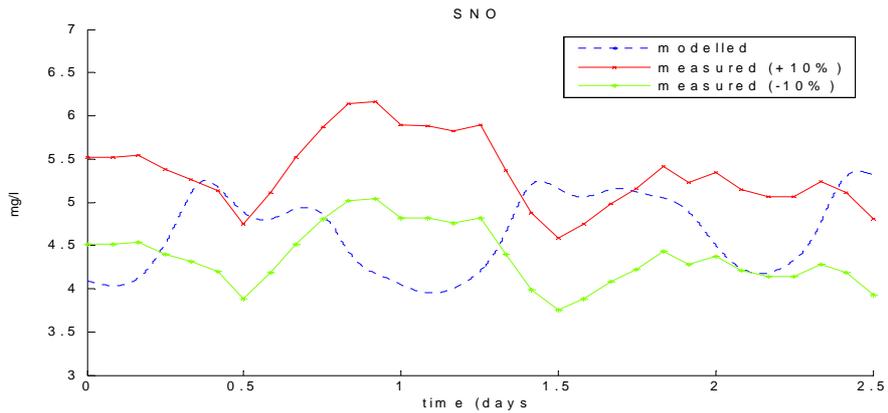


Figure 14. Modelled S_{NO} after dynamic calibration with $\pm 10\%$ relative error of the measured values included.

In figure 14 the same modelled values of S_{NO} are shown as in figure 11 but in figure 14 curves for the margin of error of the measured values are included. The upper curve is the measured values of S_{NO} plus ten percent and the lower curve is the measured values minus ten percent. This to see if the modelled values is within this range. As seen in figure 14 the modelled curve does not stay within this range the whole time. A displacement of the measured curves to the left, shorter HRT, would make the modelled curve stay within the range of the error of margin for the measured curve. This implies that the real HRT is shorter than the theoretical HRT.

4.3 RESULTS OF VALIDATION

The modelled concentrations were compared to measured outputs from campaign one.

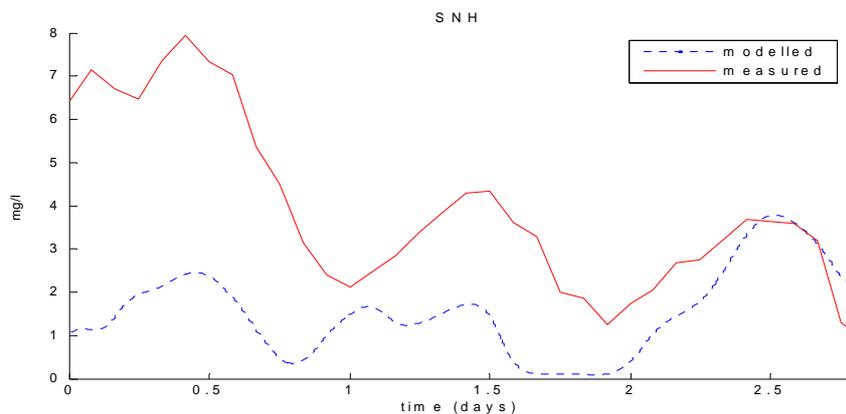


Figure 15. Modelled and measured S_{NH} after validation.

In figure 15 it is seen that the modelled and measured ammonium curves are start at very different concentrations because of the initial values in the reactors. The

modelled reactors are calibrated with data from the second campaign and the initial states in the validation are the same as during the calibration. During the validation the modelled values became negative for a period between 1.6 and 2 days. Since this is not possible in reality a command was put in the programming code to make S_{NH} values below zero become zero. This only changed the output between 1.6 and 2 days, the rest of the modelled curve stayed the same.

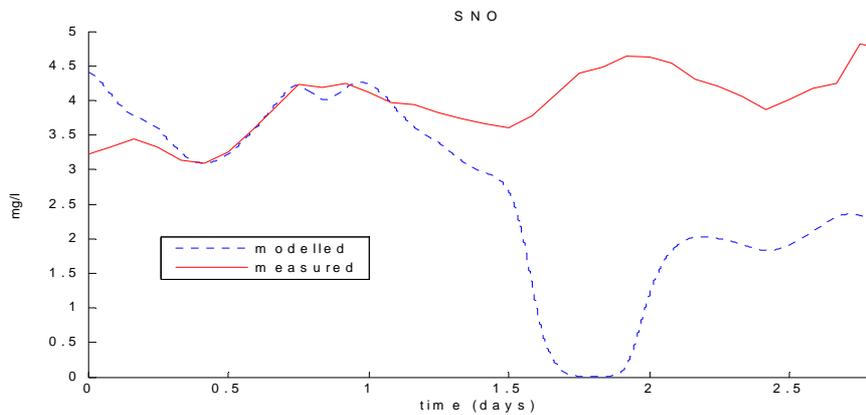


Figure 16. Modelled and measured S_{NO} after validation.

In figure 16 the modelled and measured curves of S_{NO} are shown. The curves starts at different concentrations because of the initial values but after that follow about the same mean value for 1.3 days. After about 1.5 days the modelled curve drops rapidly probably because of a large increased load of incoming COD where a too high amount is modelled as soluble COD (fig. 7).

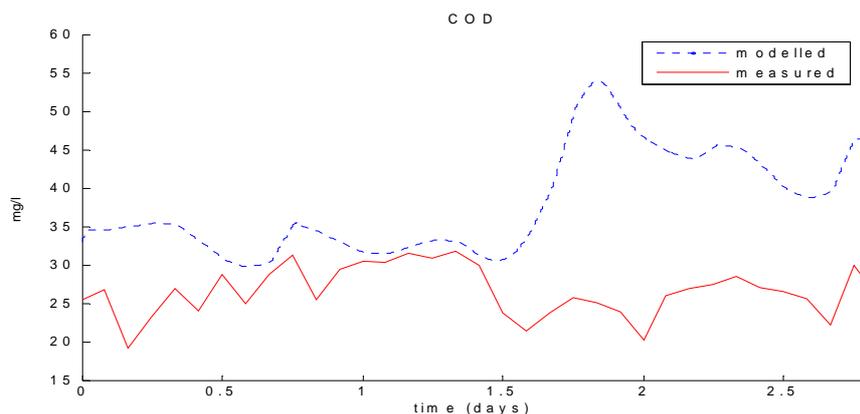


Figure 17. Modelled and measured COD after validation.

Figure 17 shows the modelled and measured COD. The modelled values are higher than the measured and show not much variation in time until after 1.5 days when the increasing load of incoming COD (fig. 7) is shown as a peak in the modelled values. This peak is not shown in the measured curve.

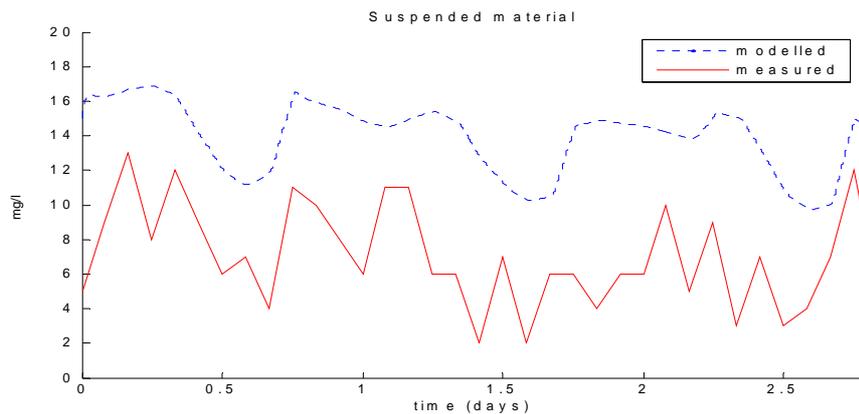


Figure 18. Modelled and measured suspended material after validation.

The measured concentrations of suspended material varies a lot over time and the modelled values show two quite large dips, figure 18. The large variations in measured values are probably due to errors in the analysis process. The mean value of the modelled curve is almost twice the mean value of the measured curve.

5 DISCUSSION

The ASM1 together with the settler model provides a good base for simulation of the biological wastewater treatment. To obtain a good model it is very important to gather as much information about the plant as possible. After doing the literature study it seems clear that there is not one easy and simple way to do this. There are many factors that affect the process and all biological treatment plants are different and run in various ways. External parameters such as weather, basic composition of the load, time of year and inflow to the wastewater treatment plant are very important for the process and differs a lot over time and from plant to plant. All these factors make every biological process unique and to simulate the processes all models have to be different. This is probably why it is not possible to provide one way to calibrate the ASM1 and only guidelines can be found in literature. If these guidelines are followed and a lot of information collected from the treatment plant it should be possible to obtain an ASM1 that will simulate the biological process quite well.

As shown in the result section the calibration of the model never managed to make the model fit the measured data very well. This can be because of a number of different reasons. One large factor is probably the problems during the measuring campaigns. During the first campaign the weather was unstable and during these conditions the wastewater treatment plant does not work very well. From the beginning this campaign was supposed to provide data for calibration and the second one for validation data. This is why more analyses were made in the first measuring campaign to provide for information to calculate the different COD and nitrogen fractions according to theory presented in the literature review section. The original idea was to

use the same percentages of the fractions calculated and calibrated from the first campaign when the data from campaign two was used for validation. After the second campaign the results were stable and showed expected daily variations, which the first campaign did not. It was then decided to use the second campaign for calibration data to hopefully produce a model that will better fit the average run of the wastewater treatment plant. Because of not enough data collected in the second campaign to calculate the COD and nitrogen fractions the same percentages calculated from campaign one data had to be used for the input data from campaign two. More analyses of the incoming wastewater would probably have made the calculations of the fractions better and they would not have to be adjusted during the calibration. That might have lead to a more accurate model. Since the theoretical hydraulic retention time for the second campaign was assumed to be about the same as for the first campaign (about 15h) it was assumed that a campaign of three days would be enough for the second campaign as well according to the criteria by Petersen, 2000. Since the flows turned out to be much lower during the second campaign the theoretical hydraulic retention time was longer (26 h) and a campaign longer than three days would have been preferred to make a better model.

In section 3.1.3 the accuracy of the analyses are mentioned. The measuring error in analyses might have a large impact of both incoming and outgoing concentrations in the wastewater. An error in the incoming concentration will result in miscalculations in the model and lead to a larger error in the output. Since the range of the measured output data is quite small the error factor in analyses will make the variations in concentration quite uncertain. This is shown in figure 14 where the curves for $\pm 10\%$ of the measured values are plotted with the modelled values of S_{NO} from the dynamic calibration.

A large problem during the steady- state calibration was to determine the settler parameters. This was done by trial and error and it was hard to find a good fit. A better way of finding the parameters would have been to perform lab-scale experiments to find the velocities, v_0 , v_0' and r_h and the minimum suspended solids, f_{ns} according to section 2.3.3. Then there would only have been a few parameters test manually, a better fit would probably have been easier to find, and a more reliable model obtained. Due to lack of time no lab-scale experiments were performed. Overall more experiments should have been performed before the calibration started to gather more information about the composition of the wastewater and the hydraulics of the settler.

In order to gather information of the hydraulics of the activated sludge reactors a tracer study was planned from the start. The study was never carried out during the time period of this work due to different reasons. There will be a tracer study performed later on and the results will be used to improve the model further. The most important information from a tracer study is probably the real hydraulic retention time. The real time is probably shorter than the theoretical time. By looking at the results of S_{NH} and S_{NO} from the dynamic calibration (fig. 10-11) it is seen that the modelled- and measured curves seem to be opposite each other which implies that the estimated hydraulic retention time for the measured values does not agree with the

modelled values. By displacing the modelled curve to the left about 0.5 days the modelled curve would have a better fit. This implies that the real hydraulic retention time is about 12 h shorter than estimated for the second measuring campaign.

The validation of the model was made with data from the first campaign. The parameters were the same as after the dynamic calibration and also the initial states in the reactors. Since the weather was unstable during this period the actual states in the reactors were far from the calibrated ones used in the model. This can be seen in figure 15- 18 as the curves start at very different values. The measured output concentrations during this time period are much higher than during the second campaign. This supports the theory that the WWTP is not working very well during these extreme conditions. After about one day there is a very high load of COD in to the reactors (fig. 7). Since soluble COD was measured in this campaign it can be seen that the soluble fraction of COD does not rise during this period and the COD top must consist mainly of particulate COD. This is not considered in the model where the same percentages for the COD fractions are used the entire time and a too high input load of soluble COD is modelled. In figure 17 the model shows a rise of COD, in figure 16 a large reduction of S_{NO} and reduction of S_{NH} in figure 15 after about 1,7 days. The measured output values do not show this COD load noticeably since it was mainly particulate COD and those fractions are reduced in the sedimentation. In the model the too high load of soluble COD provides the micro- organisms with larger energy source and the S_{NO} and S_{NH} can be reduced. During the validation the S_{NH} curve dropped below zero which of course is not possible in reality and some changes in the c- code were made to prevent this. This reduction was probably caused by the large increase in soluble COD that lead to increased nitrification, the nitrifying bacteria used ammonium that did not exist, and the values became negative. During the second measuring campaign the COD and suspended material loads are quite low and the model is calibrated for these conditions. This is probably why the modelled COD and suspended material concentrations in the validation are too high. The temperature during the first measuring campaign was about 0.7° C lower than during the second campaign but since the temperature dependent parameters were changed during the calibration it was not possible to change them linearly to fit the new temperature in the validation. This problem probably has an impact on the results from the validation.

During the calibration μ_A and b_H showed to be the parameters affecting the model output most. K_{OH} , K_{NH} , K_{OA} , k_a , η_h , Y_H and Y_A were also important. This was expected according to Petersen (2000) and Jeppsson (1996) except for the parameter k_a . It was also expected to find μ_H and K_S as important parameters but these two did not change the output of the model at all. The difficult part in the calibration of the model parameters is that they are correlated and a change of one parameter changes the effect of changes of other parameters. To obtain a better result all parameters should be tuned simultaneously.

There is still a lot of work that can be done to improve the model for Henriksdal. The major project is to perform the tracer study to gather more information about the hydraulics. It would also be desirable to perform another measuring campaign when

the weather is similar to the weather during the second measuring campaign and use the data for validation. In order to make the model fit the overall run of all the lines at Henriksdal it is needed to perform measuring campaigns at the other six lines as well. Since the run of Henriksdal is different in wintertime than the rest of the year it would be desirable to calibrate two different models, one for the winter and one for the rest of the year. For use of the model in HIPCON it is needed to get information about the energy usage during the aeration process. In this stage the model only provides information about $K_L a$ values at each aerated reactor but a function that calculates the energy use from the $K_L a$ is needed. This can be obtained by measurements of $K_L a$ at different energy effects at Henriksdal. The model also needs to be linearised before implemented in the HIPCON-prototype. This however will be done by another partner in the HIPCON project.

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Appendix A

Table 1. The ASM1 process matrix (Henze *et al.*, 1987)

Component (i) → ↓ Process (j)	1	2	3	4	5	6	7	8	9	10	11	12	13	Process rate (q _j)
	S _I	S _S	X _I	X _S	X _{BH}	X _{BA}	X _P	S _O	S _{NO}	S _{NH}	S _{ND}	X _{ND}	SALK	
1 Aerobic growth of heterotrophic biomass		$\frac{1}{-Y_H}$			1			$-\frac{1-Y_H}{Y_H}$		-i _{XB}			$-\frac{i_{XB}}{14}$	$\mu_{maxH} \frac{S_S}{K_S + S_S} \frac{S_O}{K_{OH} + S_O} X_{BH}$
2 Anoxic growth of heterotrophic biomass		$\frac{1}{-Y_H}$			1					-i _{XB}			$\frac{1-Y_H}{14 \cdot 2.86 Y_H}$ $-\frac{i_{XB}}{14}$	$\eta_e \mu_{maxH} \frac{S_S}{K_S + S_S} \frac{K_{OH}}{K_{OH} + S_O}$ $\frac{S_{NO}}{K_{NO} + S_{NO}} X_{BH}$
3 Aerobic growth of autotrophic biomass						1		$-\frac{4.57 - Y_A}{Y_A}$	$\frac{1}{-Y_A}$	$-\frac{1}{-i_{XB} Y_A}$			$-\frac{2}{14 Y_A} \frac{i_{XB}}{14}$	$\mu_{maxA} \frac{S_{NH}}{K_{NH} + S_{NH}} \frac{S_O}{K_{OA} + S_O} X_{BA}$
4 Decay of heterotrophic biomass				1-f _p	-1		f _p					i _{XB} -f _p i _{XB}		b _H X _{BH}
5 Decay of autotrophic biomass				1-f _p		-1	f _p					i _{XB} -f _p i _{XB}		b _A X _{BA}
6 Ammonification of soluble organic nitrogen										1	-1		$\frac{1}{14}$	$k_a S_{ND} X_{BH}$
7 Hydrolysis of slowly biodegradable substrate		1		-1										$k_h \frac{X_S / X_{BH}}{K_X + X_{BH}} \frac{S_O}{K_{OH} + S_O}$ $+ \eta_h \frac{K_{OH}}{K_{OH} + S_O} \frac{S_{NO}}{K_{NO} + S_{NO}} X_{BH}$
8 Hydrolysis of organic nitrogen											1	-1		$p_7 (X_{ND} / X_S)$

Appendix B

Table 1. Results from measuring campaign one, point A, inlet.

Sample	Kond.	pH	N-tot	COD-tot	NH4-N	Nitrate	Nitrite	N-filtr	COD-sol	Susp.
A1	767	7.22	30.3	160.4	28.2	1.8	0.26	28.1	70.9	154
A2	756	7.15	27.8	176	27.4	1.11	0.25	27.3	57.4	108
A3	767	7.13	29.1	176	26.2	1.05	0.27	30.3	59.8	110
A4	796	7.11	30.1	222.8	28.6	1.06	0.27	27.9	93.6	119
A5	815	7.16	31.8	231.2	28.3	1.08	0.28	29.7	98.8	120
A6	805	7.12	31.4	223.2	28.5	1.09	0.31	30.1	91.6	112
A7	771	7.14	30.2	208.4	27.4	0.929	0.31	27.7	102	106
A8	769	7.12		191.2	26.9	0.914	0.3	25.4	92.4	93
A9	774	7.05	27.4	161.2	27.2	1.00	0.3	24.8	86.8	78
A10	767	7.01	27.6	193.6	23.8	1.12	0.28	24	68.4	72
A11	760	7.37	26.1	112	23.5	1.36	0.3	23.7	42.9	74
A12	777	7.3	28.5	126	25.1	1.39	0.3	25.6	106	75
A13	782	7.15	32.2	178	28.2	1.41	0.32	26	75.5	81
A14	796	7.2	22.2	200	30.6	0.991	0.3	23.68	74.1	115
A15	811	7.2	24.3	194.2	30.8	0.746	0.31	27.44	77.2	113
A16	844	7.1	31.4	226	32.2	0.835	0.31	27.8	84.6	133
A17	842	7.13	32.2	230	32.7	0.833	0.33	27.76	85.7	130
A18	821	7.16	34.3	256	31.4	0.756	0.28	27	70.9	167
A19	797	7.13	29.9	686	28.3	0.756	0.21	23.32	60.9	244
A20	779	7.14	37.1	852	26.2	0.789	0.2	24.72	50.2	335
A21	778	7.14	32.56	842	26.2	0.865	0.19	23.52	43.4	354
A22	775	6.94	31.7	946	24.6	1.04	0.2	22.2	39.2	408
A23	787	7.09	25.6	242	24.9	1.13	0.23	21.88	36.6	202
A24	773	7.3	25.3	201	28.3	1.16	0.29	24.8	37.5	139
A25	788	7.28	35.68	281	22.00	1.25	0.26	29.8	49.9	226
A26	793	7.15	33.28	277	31.2	0.89	0.25	28.48	46.8	194
A27	807	7.08	38.4	324	32.9	0.62		28.16	67.3	220
A28	836	7.09	41.6	335	32.5	0.54	0.24	30.12	65.9	225
A29	841	7.1	39.76	324	34.6	0.44	0.25	30.48	76.9	203
A30	814	7.13	39.56	318	34.5	0.54		29.56	64.4	206
A31	787	7.14	37.8	326	29.5	0.5	0.24	25.32	58.2	244
A32	772	7.07	46.8	404	28.2	0.51	0.23	23.8	70.9	337
A33	772	6.97	42.4	494	28.2	0.55		24.64	44.6	426
A34	778	7.03	46.4	422	28.4	0.66	0.22	23.96	41.9	376
A35	781	7.15	34.84	222	24.9	0.56		21.72	39.7	119
A36	782	7.19	37.68	277	29.6	1.18	0.3	24.88	41.5	135

Table 2. Results from measuring campaign one, point B, outlet.

Sample	Kond.	pH	N-tot	COD-tot	NH4-N	Nitrate	Nitrite	N-sol	COD-sol	Susp.
B1	653	6.64	8.55	25.5	6.42	3.23	0.13	9.44	16.2	5
B2	660	6.67	8.72	26.8	7.14	3.32	0.13	8.46	23.9	9
B3	665	6.61	9.39	19.2	6.7	3.45	0.13	8.63	18.4	13
B4	670	6.64	8.93	23.2	6.47	3.33	0.13	8.55	20.7	8
B5	677	6.76	10.5	26.9	7.35	3.13	0.13	8.89	22.7	12
B6	682	6.7	11.4	24	7.95	3.09	0.13	9.09	20.7	9
B7	683	6.72	11.4	28.7	7.33	3.26	0.13	9.36	20.7	6
B8	680	6.63	10.7	24.9	7.04	3.56	0.13	9.13	25.1	7
B9	675	6.64	8.38	28.8	5.38	3.9	0.12		22.8	4
B10	668	6.96	7.46	31.3	4.5	4.23	0.13	7.96	26	11
B11	664	6.87	6.69	25.4	3.14	4.19	0.11	6.45	23.1	10
B12	656	6.7	7.12	29.4	2.4	4.24	0.16	7	26.8	8
B13	667	6.81	8.64	30.5	2.115	4.11	0.14	6.41	24.2	6
B14	669	6.72	9.31	30.3	2.46	3.97	0.15	6.95	19.9	11
B15	671	6.12	9.28	31.5	2.835	3.94	0.15	7.71	20.1	11
B16	678	6.33	10.1	30.8	3.39	3.83	0.14	8.33	20.2	6
B17	685	7.28	8.73	31.8	3.855	3.73	0.14	8.14	15.9	6
B18	690	6.84	8.12	29.9	4.29	3.66	0.13	8.3	22.7	2
B19	690	6.74	8.77	23.7	4.34	3.6	0.13	8.58	20.6	7
B20	688	6.6	8.18	21.4	3.61	3.78	0.12	8.39	21.1	2
B21	685	6.74	6.03	23.8	3.275	4.08	0.12	7.98	19.1	6
B22	680	6.67	4.78	25.7	9	4.4	0.11	7.23	15.2	6
B23	671	6.59	5.95	25.1	1.85	4.48	0.12	7.06	23.3	4
B24	666	6.63	5.59	23.9	1.25	4.64	0.12	6.27	20	6
B25	668	6.88	7.25	20.2	1.75	4.63	0.19	7.21	19.3	6
B26	667	6.87	7.74	26	2.05	4.54	0.13	7.31	18.5	10
B27	672	6.77	7.85	26.9	2.67	4.31		8.33	18.2	5
B28	675	6.77	8.16	27.5	2.74	4.21	0.13	8.13	18.7	9
B29	682	6.76	8.95	28.5	3.22	4.06		8.24	18.7	3
B30	683	6.78	9.56	27.1	3.68	3.87	0.12	8.59	18.2	7
B31	683	6.75	8.44	26.5	3.64	4.02		8.5	21.4	3
B32	681	6.75	9.48	25.6	3.58	4.17	0.11	8.54	22.5	4
B33			8.51	22.2	3.2	4.25		8.28	20.9	7
B34			7.75	29.9	1.3	4.82	0.11	6.49	19.8	12
B35			7.23	25.9	0.9	4.76		6.39	23.3	5
B36			7.11	22.3	0.95	4.52	0.09	6.61	25	10

Table 3. Results from measuring campaign two, point A, inlet.

Sample	COD	N-tot	NH4	Nitrat	Susp.
A1	201.2	47	34.9	0.584	98
A2	233.2	46.8	36		114
A3	247.6	44	32.1	0.262	114
A4	261.2	42.4	30.7		128
A5	252.8	43.2	30.5		128
A6	254.4	38.8	30.7		122
A7	178	39.2	29.3		124
A8	270.8	39	30		120
A9	264.4	39.4	30.6		116
A10	249.6	40	29.9		112
A11	206.4	40	30.4		106
A12	196.8	44.2	33		120
A13	239.6	44.6	34.2	0.537	114
A14	378.4	45.2	32.1		130
A15	280.8	45.6	32		132
A16	298.8	47.4	33.7		130
A17	315.2	47.4	35.8	0.322	126
A18	316.4	46.8	32.1		124
A19	301.2	42.6	30.5		126
A20	280.8	56.2	33.2		120
A21	276	39.6	31.7	0.26	132
A22	261.2	41.2	34.9		116
A23	189.2	37.4	31.3		110
A24	181.2	42	32.1	0.605	98
A25	243.6	44.6	30.52		110
A26	239.2	45.6	31.72		108
A27	275.2	55.6	34.24	0.36	122
A28	301.2	47.4	35.44		138
A29	305.2	47.2	34.2	0.348	128
A30	312	44.8	34.24		128
A31	301.2	44.4	31.48	0.256	118
A32	295.2	42.4	32.52		122
A33	280.4	43.8	32.2	0.259	110
A34	231.6	42.8	28.92		96
A35	178.8	43	31.2		92
A36	184.8	40.8	30.2		90

Table 4. Results from measuring campaign two, point B, outlet.

Sample	COD	N-tot	NH4	Nitrat	Susp.
B1	34	7.96	1.25	5.02	10
B2	31.7	7.76	1.1	5.02	9
B3	30.5	7.2	0.93	5.04	13
B4	29.9	7.02	0.7	4.89	12
B5	32	6.97	1.47	4.79	10
B6	36.1	6.54	1.2	4.67	14
B7	33.6	6.3	1.23	4.32	12
B8	35.4	7.04	1.14	4.65	12
B9	34.8	6.85	1.14	5.02	8
B10	29.7	7.29	0.35	5.34	8
B11	34.6	7.24	0.3	5.58	8
B12	34.2	7.3	0.27	5.6	9
B13	35.7	7.72	0.36	5.36	11
B14	33.4	7.52	0.48	5.35	11
B15	32.5	7.89	0.52	5.29	10
B16	33.2	7.02	0.42	5.36	10
B17	31.4	7.18	0.41	4.88	10
B18	31.7	6.35	0.219	4.43	12
B19	34.3	6.19	1.16	4.17	13
B20	32.6	7.72	1.66	4.32	17
B21	32	6.74	1.41	4.53	7
B22	35	7.26	0.622	4.69	8
B23	33.8	7.4	0.635	4.92	10
B24	33.3	7.67	0.803	4.75	11
B25	36.5	8.06	1.07	4.86	7
B26	34	7.92	1.43	4.68	9
B27	32.7	8.29	1.55	4.6	8
B28	33	8.65	1.45	4.6	9
B29	34.7	8.38	2.08	4.76	5
B30	32	6.99	0.73	4.65	10
B31	28.7	6.68	1.33	4.37	9
B32	29.6	6.68	0.387	4.73	15
B33	30.3	7.17	0.468	4.7	8
B34	27.7	7.41	0.704	4.78	10
B35	29.4	7.78	0.795	4.85	12
B36	28.8	8.05	0.99	4.87	10
B37	27.4	8.72	1.38	4.71	9
B38	30	9.23	1.78	4.61	8
B39	30.1	8.98	2		7

