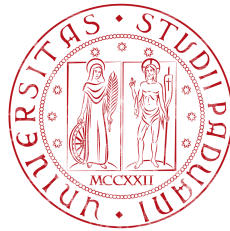


Università degli Studi di Padova  
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# Dynamic evaluation of a high loaded activated sludge plant as pretreatment for deammonification in the mainstream

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A mio padre e mia madre





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# Background

Over the last decade new technologies have been discovered and studied as innovative solutions for nitrogen removal in wastewater treatment plants. Among the biological treatments, the discovery of anaerobic ammonium oxidation (Anammox) can be acknowledged as one of the most important, since it has led to the development of new cost-effective and sustainable solutions. However, its main full-scale applications has been in sidestream treatments and its implementation in municipal wastewater mainstream is at the forefront of research. Within this study, the challenging implementation of the novel deammonification process in the wastewater mainstream is considered. Modelling of activated sludge processes became an innovative advancement for the evaluation of different operational and control strategies, supporting full and experimental-scale testing. One of the main advantages of a model-based approach is that the number of scenarios that can be evaluated can be significantly increased (Amond, 2011; Vanrolleghen et al., 2007) and the comparative evaluation is basically inexpensive, provided that the model be properly calibrated. On the other hand, many limitations must also be faced, due to both intrinsic limits of a model, as such, and case-specific restrictions as data availability and reliability. Nonetheless, a model-based assessment can represent an important starting point and a useful supporting-decision tool for process choices in real wastewater treatment plants. In this study, the Benchmark Simulation Model number 1 (BSM1) was applied to the high-loaded activated sludge unit at Sjölanda WWTP (located in Malmö, Sweden), in order to evaluate the effects of operational changes on effluent quality. The calibration of the model was based on both historical data set and data collected in a one-day measuring campaign, carried out in January 2013. Respirometric analyses and settling column tests on activated sludge samples were also performed in order to estimate some of the model parameters. Sjölanda WWTP is oriented towards the future perspective of deammonifi-

cation process implementation in the wastewater main stream (Manammox: Mainstream Anammox); this process is intended to be the subsequent treatment after the high loaded activated sludge unit. The plant is already experiencing the deammonification process in a pilot-scale moving bed biofilm reactor (MBBR) on rejected water and, soon, on mainstream wastewater. The Manammox influent should comply with low biodegradable COD and high ammonium concentrations, i.e. low COD/N ratios. Therefore, the required performances of the high loaded activated sludge unit result in high organic removal without nitrification.

Three different scenarios on plant layout were simulated and their dynamic response was evaluated under average and critical weather conditions. According to the model outcomes, one of the scenarios was selected as the most suitable for the future Manammox full-scale implementation. A further analysis was conducted in order to assess the effects of dissolved oxygen concentration and sludge age changes on effluent quality.

# Chapter 1

## Introduction

### 1.1 Literature review on activated sludge models

#### 1.1.1 Activated Sludge Model number 1

The activated sludge model number 1 (ASM1) is considered the state-of-the-art model for activated sludge processes. It was published in 1987 by the International Association of Water Quality (IWAQ), aiming to reach an internationally accepted model for activated sludge systems carrying out biological carbon and nitrogen removal.

The carbonaceous material is divided into biodegradable COD, unbiodegradable COD and biomass. The biodegradable COD is, in turn, divided into two fractions: soluble  $S_S$  and particulate  $X_S$ . They roughly represent the readily and the slowly biodegradable COD, respectively, even though some of the slowly biodegradable COD can be actually be soluble and it is neglected in the model. Analogously, the un-biodegradable COD is divided into a soluble  $S_I$  and a particulate fraction  $X_I$ . The biomass COD is divided in heterotrophic  $X_{BH}$  and autotrophic  $X_{BA}$ . The first lumps groups together the ordinary heterotrophic organisms (OHOs), responsible both for the carbon removal under aerobic condition and the denitrification of nitrate into gaseous nitrogen under anoxic condition. The autotrophic biomass is, instead, responsible for the aerobic nitrification of ammonia. Finally, another COD fraction  $X_P$  accounts for the particulate products due to the biomass decay (Figure 1.1). The COD fractions are expressed in  $gCODm^{-3}$  or, equivalently,  $mgCODl^{-1}$ .

The nitrogen content is divided into four components, according to the

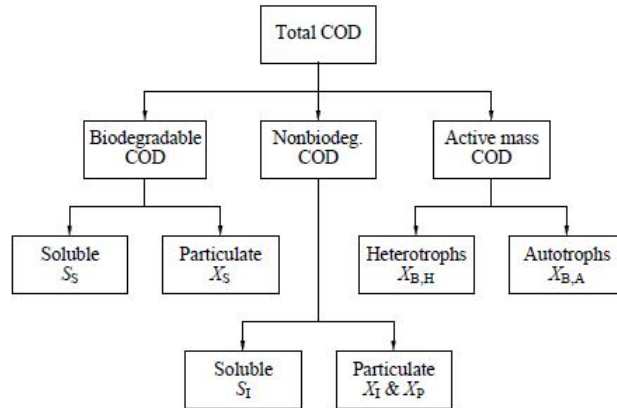


Figure 1.1: COD fractions in ASM1 model (figure from Jeppsson, 1996)

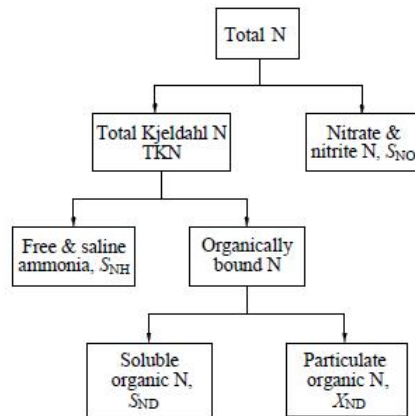


Figure 1.2: Nitrogen fractions in ASM1 model (figure modified from Jeppsson, 1996)

main nitrogen compounds found in wastewaters: ammonia, nitrate, nitrite and organic nitrogen (Figure 1.2). Free and saline ammonia is represented by the fraction  $S_{NH}$ , whereas nitrate and nitrite are grouped under the fraction  $S_{NO}$ . The organically bounded nitrogen is further divided into its soluble and particulate components:  $S_{ND}$  and  $X_{ND}$ , respectively. Nitrogen fractions are expressed in  $gNm^{-3}$ .

Finally, the last two state variables are the oxygen concentration, expressed as grams of negative COD per cubic meter ( $g-CODm^{-3}$ ), and the alkalinity (mol l<sup>-1</sup>).

This fractionation results in 13 state variables, listed in Table 1.1.

State Variable	Notation	Unit
Soluble inert organic matter	$S_I$	gCOD m <sup>-3</sup>
Readily biodegradable substrate	$S_S$	gCOD m <sup>-3</sup>
Particulate inert organic matter	$X_I$	gCOD m <sup>-3</sup>
Slowly biodegradable substrate	$X_S$	gCOD m <sup>-3</sup>
Active heterotrophic biomass	$X_{B,H}$	gCOD m <sup>-3</sup>
Active autotrophic biomass	$X_{B,A}$	gCOD m <sup>-3</sup>
Particulate products from biomass decay	$X_P$	gCOD m <sup>-3</sup>
Oxygen	$S_O$	g(-COD) m <sup>-3</sup>
Nitrate and nitrite nitrogen	$S_{NO}$	gN m <sup>-3</sup>
Free and saline ammonia nitrogen	$S_{NH}$	gN m <sup>-3</sup>
Soluble biodegradable organic nitrogen	$S_{ND}$	gN m <sup>-3</sup>
Particulate biodegradable organic nitrogen	$X_{ND}$	gN m <sup>-3</sup>
Alkalinity	$X_{ND}$	mol m <sup>-3</sup>

Table 1.1: ASM1 state variables

A relevant number of parameters is required in the model. The default values used in the BSM1 are listed in Table 1.2. Their values have to be changed in case of simulations implying temperature changes.

It has to be unlighted that such a detailed characterization of wastewater

components is never found in real plants. The collection of data is though compulsory for the main components in order to monitoring plant performances as well as effluent concentration in relation to reagation limits, but it could never cover the wide range of parameters considered in wastewater modelling. The benefit of having such a deep characterization would not overcome the required significant effort and cost, within a real plant management system. Thus, any attempt in the definition of all the model variables can be useful in order to gain model accuracy, but at the same time, it has to be kept in mind that many assumptions are made in extrapolating all the information usually not considered in a real system.

The processes implemented in the model are described by Jeppsson (1996) and briefly reported below, followed by their model mathematical formula-

Parameter	Unit	Value
$\mu_H$	$d^{-1}$	4.0
$K_S$	$gCODm^{-3}$	10.0
$K_{O,H}$	$g(-COD)m^{-3}$	0.2
$K_{NO}$	$g(NO_3-N)m^{-3}$	0.5
$b_H$	$d^{-1}$	0.3
$\eta_g$	–	0.8
$\eta_h$	–	0.8
$k_h$	$g \text{ slowly biodegradable COD. (g cell COD. d)}^{-1}$	3.0
$K_X$	$g \text{ slowly biodegradable COD. (g cell COD)}^{-1}$	0.1
$\mu_A$	$d^{-1}$	0.5
$K_{NH}$	$g(NH-N)m^{-3}$	1
$b_A$	$d^{-1}$	0.05
$K_{O,A}$	$g(-COD)m^{-3}$	0.4
$k_a$	$m^3(gCOD.d)^{-1}$	0.05

Table 1.2: ASM1 kinetic parameters at 15 °C

tion.

- *Aerobic growth of heterotrophic biomass.* The growth is modelled according to Monod kinetics. A maximum growth rate is defined and the limiting factors are the readily biodegradable substrate,  $S_S$ , and the oxygen concentration,  $S_O$ . The readily biodegradable substrate is used both for heterotrophs  $X_{BH}$  growth and energy uptake through respiration. The yield coefficient is the parameter defining the substrate fraction converted into new cell mass. The nitrogen sources for the cellular synthesis is ammonia. This is the main process responsible for the COD removal.

$$\rho_1 = \mu_H \left( \frac{S_S}{K_S + S_S} \right) \left( \frac{S_O}{K_{O,H} + S_O} \right) X_{BH}$$

- *Anoxic growth of heterotrophic biomass.* In case the oxygen is not available as final electron acceptor for heterotrophs growth and substrate removal (anoxic condition), nitrates are used instead. It results in nitrate reduction into nitrogen gas (denitrification) and in heterotrophs growth, together with alkalinity depletion. Ammonia is used as nitrogen source for cellular growth. The anoxic biomass growth is modelled through the same Monod formulation although a reducing factor  $\eta_g$  ( $<1$ ) is used, being its extent rather smaller than the aerobic one. The limiting factors are the readily biodegradable substrate  $S_S$  and nitrates



$S_{NO}$ . Oxygen is also taken into account as a limiting factor, meaning that its presence inhibits the process by inducing the more competitive aerobic growth.

$$\rho_2 = \mu_H \left( \frac{S_S}{K_S + S_S} \right) \left( \frac{K_O}{K_{O,H} + S_O} \right) \left( \frac{S_{NO}}{K_{NO} + S_{NO}} \right) \eta_g X_{BH}$$

- *Aerobic growth of autotrophic biomass.* In presence of ammonia and for long sludge age systems, autotrophic and heterotrophic biomass can thrive together. Autotrophic growth results in a higher oxygen requirement. Oxygen is used for the oxidation of ammonia into nitrate (nitrification). The process is modelled by Monod kinetic, being the limiting factors ammonia concentration and oxygen. The yield coefficient for this growth is significantly low, if compared with that used for heterotrophic growth. Ammonia is used for cellular growth.

$$\rho_3 = \mu_A \left( \frac{S_{NH}/2}{K_{NH} + S_{NH}} \right) \left( \frac{S_O}{K_{O,A} + S_O} \right) X_{BA}$$

- *Decay of heterotrophic biomass.* The death-regeneration hypothesis is used. It is assumed that the biomass dies at a certain rate (death rate,  $b_H$ ), assumed constant under both aerobic and anoxic conditions. The dead biomass partially turns into particulate matter partially unbiodegradable and partially biodegradable. The factor  $f_p$  accounts for the first portion and this fraction leads to the production of the particulate  $X_P$  component, whereas the balance fraction  $(1 - f_p)$  re-enters the system as new slowly biodegradable substrate,  $X_S$ . Also the organic nitrogen enmeshed with the particulate substrate  $X_S$  becomes available as  $X_{ND}$ .

$$\rho_4 = b_H X_{BH}$$

- *Decay of autotrophic biomass.* It is modelled with the same formulation used for the heterotrophic decay. A smaller decay rate is used, having

this process a smaller extent.

$$\rho_5 = b_A X_{BA}$$

- *Ammonification of soluble organic nitrogen.* Active heterotrophs are also responsible for the conversion of biodegradable soluble organic nitrogen  $S_{ND}$  into ammonia, through a first-order process.

$$\rho_6 = k_a S_{ND} X_{BH}$$

- *Hydrolysis of entrapped organic substrate.* The enzymatic extracellular lysis of slowly biodegradable substrate,  $X_S$ , leads to the production of readily biodegradable substrate,  $S_S$ . This process is dependent on the biomass content through surface kinetic reactions. It reaches a saturation phase when the amount of entrapped substrates become larger in proportion to the biomass. Substrate hydrolysis occurs both under aerobic and anaerobic conditions, being significantly faster in the first case. A reducing factor  $n_h$  is then used to account for this difference.

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

- *Hydrolysis of entrapped organic nitrogen.* Particulate organic nitrogen,  $X_{ND}$ , is hydrolyzed into soluble organic nitrogen,  $S_{ND}$ . The process is limited and modelled in the same way as for the particulate organics hydrolysis.

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

These processes may involve more than one state variable at the same time and, in turn, each state variable may be part of more than one process. The inert carbonaceous fractions  $S_I$  and  $X_I$  are the only two variables not involved in any of the biological processes described. The conversion processes referring to the other state variables are listed below as ordinary differential equations in Jepsson (1996).

### 1.1.2 Secondary settler model

Particles settling in secondary clarifiers is a complex process due to the flocculating nature of solid particles. Differently from primary settling and sand removal, the theory of discrete particle sedimentation cannot be directly applied and thickening as coalescing phenomena have to be considered. For this reason, former attempts in modelling secondary settling process failed and significant improvements were made when thickening phenomena started to be included. In general, four zones can be identified in a secondary clarifier (Takács,1991), according to their different settling characteristics. Starting from the top:

1. *Discrete particle zone*: thanks to very low solids concentration in the upper layers, particles settle as individual entities without any interaction with other particles. (Similarly to settling in grit and sand removal units).
2. *Flocculent particle zone*: particles aggregate as flocks and their settling velocity increases with their dimension.
3. *Hindered zone*: inter-particle forces result in settling hindering and suspension. The mass of particles settles as a unit.
4. *Compression zone*: at very high solid concentration settling velocity decreases dramatically and compression takes place due to the weight of particles. Compression entity is a function of many factors as: particle nature, depth of the sludge blanket, sedimentation time.

Most of the models used today are one-dimensional layer models. They were developed by implementing mainly the solid flux theory together with the conservation law (Vitasovic, 1985). The main assumptions of these models are the following:

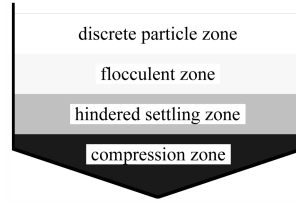


Figure 1.3: Secondary settler zones (figure from Jeppsson, 1996)

- Incoming solids are distributed instantaneously and homogeneously across the entire horizontal plane;
- Only vertical flow is considered;
- No significant biological reaction occurs;
- The mass flux in each layer cannot exceed the maximum solid flux the layer itself is capable of passing;
- The mass flux in each layer cannot exceed the maximum solid flux the receiving (immediately below or above) layer is capable of passing;
- The gravitational settling velocity is a function of solids concentrations.

The total solid flux in each layer, is considered as the sum of the solid flux due to the bulk movement of the liquid and the gravitational settling flux.

$$J_i = J_{BM,i} + J_{s,i}$$

The bulk movement flux can be upward or downward depending on the relative position of the layer with respect with the feeding one. Thereby, their expressions are:

$$J_{UP,} = X_i \frac{Q_e}{A}$$

$$J_{DOWN,} = X_i \frac{Q_u}{A}$$

being  $X_i$  the  $i$ -layer solids concentration,  $Q_e$  the settler effluent flow rate,  $Q_u$  the settler underflow flow rate and  $A$  the settler cross-sectional area.

The solid flux due to gravity is defined as:

$$J_s, = X_i v_{s,i}$$

where  $v_{s,i}$  is the gravitational settling velocity, that is, in turn, function of solids concentration. Many formulations have been proposed in literature for its assessment. The first well accepted one was proposed by Vesilind (1968):

$$v_s = v_0 e^{-\alpha X}$$

$v_0$  being the maximum settling velocity and  $\alpha$  a model parameter. According to this expression, the actual settling velocity exponentially decreases as solids concentration increases. It well models the settling process in the hindered zone, but it cannot be applied to the clarification zone, where it would predict the maximum settling velocity at very low solids concentrations. Because of this limitation, the double exponential equation proposed by Takacs (1991) is now widely used as one of the best model for gravitational settling flux assessment. Its formulation is the following:

$$v_s = \max [0, \min \{v'_0, v_0 e^{-r_h(X-X_{min})} - v_0 e^{-r_p(X-X_{min})}\}]$$

where  $v_0$  is the maximum settling velocity,  $v'_0$  is the maximum Vesilind settling velocity,  $r_h$  and  $r_p$  are model parameters referring to the hindered and the clarifying (low solids concentration) zone respectively, and  $X_{min}$  is the minim attainable solids concentration that in turn is given by:

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

being  $f_{ns}$  the non-settleable fraction of the influent mixed-liquor solids concentration,  $X_{in}$ .

According to this formulation, four zones are defined according to the solids concentration, as shown in Figure 2.3, and the predicted velocity is consistent with the discrete particle zone characteristics, unlike to Vesilind's model (dashed line). The fourth zone models the hindered zone and, in fact, the two curves overlap on each other.

The settler model implemented in the Benchmark simulation Model is a 10-layer one dimensional model, using the double exponential Takács equation for the gravitational settling velocity. Layers are equidistant and the cross-sectional area is constant along the settling depth. Layer 5 is considered as the feeding layer.

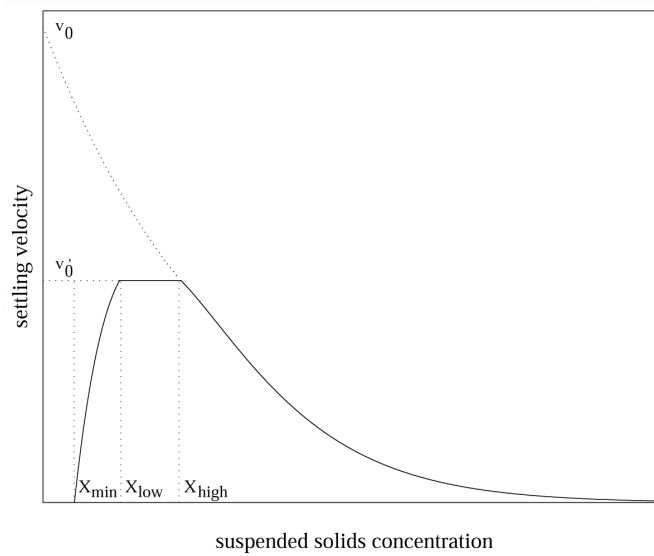


Figure 1.4: Settling velocity models proposed by Takács (continuous line) and Vesilind (dotted line). Figure modified from Jeppsson, 1996

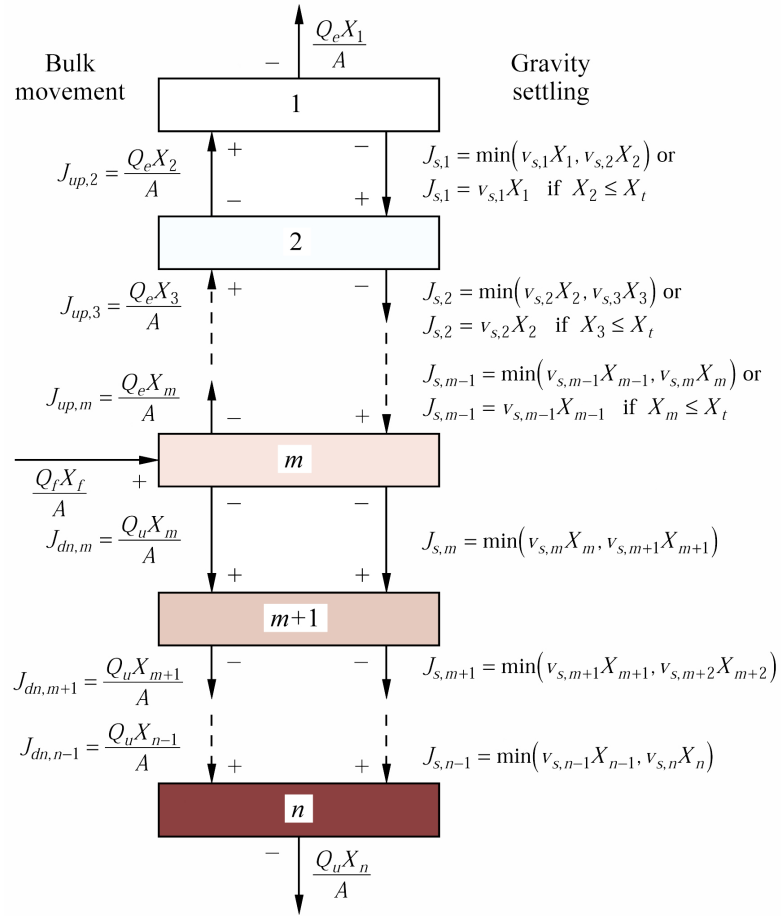


Figure 1.5: General description of the settler model (figure modified from Jeppsson, 1996)

## 1.2 Innovative processes for biological nitrogen removal

The conventional process of nitrification coupled with denitrification Nitrification/Denitrification process is the most widely used (method) for biological nitrogen removal in wastewater streams worldwide. Thanks to its well-established application, this process is characterised by high reliability and stability. Nevertheless, it showed to be not feasible for water streams with high nitrogen content ( $>100$  mgN/l) or low C/N ratios as: sludge liquor from dewatering of digested sludge; landfill leachate and some industrial wastewaters (Bertino, 2010). Moreover, the need for aeration and external carbon source (in case of post-denitrification configuration) dramatically increases the overall plant energy requirement and costs. This limitations have driven the research towards new solutions and new innovative processes have been developed over the past decades. Thanks to their full-scale application, some of them have been acknowledged as successful and cost-effective alternatives to conventional biological nitrogen removal systems, especially, for nitrogen-rich streams –for nitrogen concentrations up to 5000 mg/l (Van Hulle, 2005).

### 1.2.1 The Nitrataion-Denitrataion process

Analogously to Nitrification-Denitrification, this process allows oxidation of ammonia and subsequent reduction into nitrogen gas. Differently from the conventional process, aerobic oxidation of ammonia is arrested at its first stadium. Nitrite are therefore produced and consequently reduced into nitrogen gas. As showed in Figure 2.6 limiting the complete oxidation process yields to significant savings in terms of aeration energy and external carbon source requirements.

As well known, nitrification is a double-step reaction mediated by chemolithoautotrophic obligatory aerobic bacteria. The first step is carried out by ammonia oxidizing bacteria (AOB) –the main genera being *Nitrosomonas* and *Nitrospira*. They are responsible for ammonium oxidation into nitrite. The second step is the complete oxidation into nitrate, carried out by nitrite oxidizing bacteria (NOB)-as *Nitrobacter* and *Nitrospira*. In order to stop nitrification at the nitrataion step, AOB have to outcompete NOB. The main operating variables to keep partial nitrataion under control are: temperature, sludge retention time, dissolved oxygen, pH and substrate and inhibitors con-



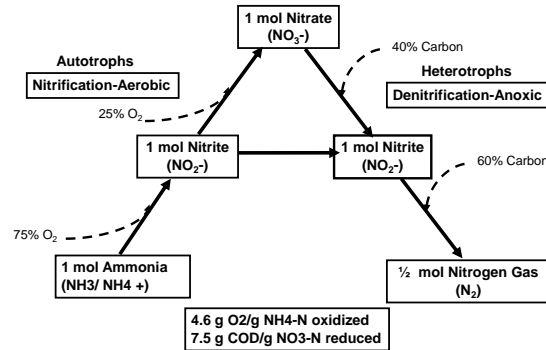


Figure 1.6: Nitratation-Denitritation process (from Stensel, 2006)

centration (Bertino, 2010). At temperatures above 25°C, AOB have higher growth rate than NOB, whereas lower temperatures yield an opposite condition. Therefore, at values of 25-35°C, shorter minimum sludge retention time is required by AOB rather than NOB (Figure 1.7) and sludge age can be manipulated in order to induce NOB washing out.

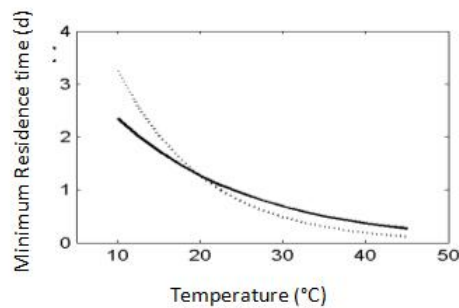


Figure 1.7: Qualitative effect of temperature on the minimum required residence time for AOB (continuous line) and NOB (dotted line)

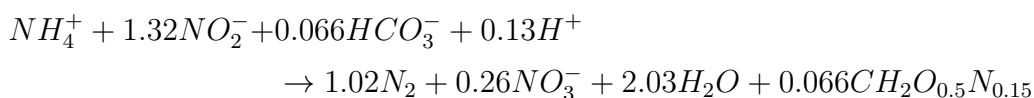
Since NOB have lower oxygen affinity than AOB, low DO concentrations limit nitratation and not nitritation rate –oxygen half-saturation constant ranges between 0.03-1.3 mgO<sub>2</sub>/l and 0.3-2.5 mgO<sub>2</sub>/l for AOB and NOB, respectively (van Hulle, 2005; Hunik et al., 1994; Guisasola et al., 2005). It was showed that the actual substrates for AOB and NOB are free ammonia (NH<sub>3</sub>) and free nitrous acid (HNO<sub>2</sub>) respectively (Suzuki at., 1974; Anthonisen et al., 1976 ), thereby high pH of 7.5-8 promote AOB activity

rather than NOB's. Finally, high concentrations of free ammonia and nitrous acid can have inhibitory effects on both the nitrifying bacteria, even though the threshold concentrations may vary according to bacteria adaptation (van Hulle et al., 2007; Bertino, 2010).

One of its most known applications is the so called SHARON process (standing for Single reactor High Activity Ammonia Removal Over Nitrite) in which aerobic autotrophic ammonia oxidizing bacteria (AOB) coexist with anoxic heterotrophic denitrifiers. It was developed at Delft University of Technology in the late 1990s (Hellinga et al., 1998). This process had its first full-scale application in Rotterdam-Dokhaven in 1999 and further plants were started in Netherlands, in the WWTPs of Amsterdam-Ost, Utrecht, Beverwijk, Gamerwolde and Zwolle.

### 1.2.2 Anammox process

Anaerobic ammonium oxidation ammonium (Anammox) is a promising process capable of oxidizing ammonium into nitrogen gas under anoxic condition. The reaction is biologically mediated by chemolithoautotrophic bacteria. Nitrite are used either as final electron acceptor for ammonia oxidation and as electron donor for carbon dioxide reduction (dissolved bicarbonate is used as inorganic carbon source). The whole reaction of ammonia oxidation is the following:



A nitrite/ammonium ratio of 1.32 is required and a slight amount of nitrate is produced (around 11% of the incoming nitrogen, as nitrite and ammonia). It has been proved that hydroxylamine (NH<sub>2</sub>OH) and hydrazine (N<sub>2</sub>H<sub>4</sub>) are produced as intermediates. The same factors as for Nitrification-Denitrification process are manipulated to keep Anammox process under control. Anammox bacteria have a slow maximum growth rate of 0.00648 d<sup>-1</sup> and a low biomass yield of 0.11-0.13 g VSS/g NH<sub>4</sub><sup>+</sup>-N (Strous et al., 1997, Jetten et al., 1999, Bertino 2010), resulting in a doubling time of up to 11 days (Strous et al., 1999, Jetten et al., 1999). Optimum ranges for pH and temperature are 6.5-8.8 and 25-45°C, respectively (Strous et al., 1999; Jetten et al., 1999). At low temperature of 20°C, decrease in nitrogen conversion rate was observed. In case of low temperature application, Anammox biomass must be produced in a separate reactor operating within the optimal temperature range and the required amount of biomass be picked from the optimum-temperature system, inoculated into the low-temperature one and let adapt gradually (Strous et al., 1999). Anaerobic oxidation of ammonium is strongly but reversibly inhibited by oxygen –Strous (1999) showed the inhibitory effects of oxygen at concentration between 0.25-2% air saturation, whereas a critical concentration of 63 μM has been indicated by Oshinki et al., (2011). Nitrite turns into inhibiting factor if Anammox bacteria undergo long exposure to high concentration Anammox activity is also sensitive to visible light, with a decrease in activity of 30-50% (Bertino, 2010). As a consequence of the low biomass yield, sludge production is low and long periods are required

for the startup of Anammox reactors. To operate Anammox system at low temperatures, the required amount of biomass must be produced in a separate reactor at a temperature close to the optimal temperature. Then the biomass can be gradually adapted to low temperatures in the same reactor and finally the low-temperature adapted biomass can be inoculated in the low-temperature reactor. However, optimum as well as inhibiting ranges of all the listed factors may differ case by case.

Wrinkler et al. (2012) showed that Anammox can also provide anaerobic oxidation of readily biodegradable substrates, using nitrate as final electron acceptor, in case of low C/N ratios down to 0.5. The observed organic removal did not lead to any new biomass production. These results might be of promising interest for future Anammox applications, beyond mere nitrogen removal targets. If compared with traditional nitrogen removal systems, anaerobic oxidation of ammonium has many promising advantages such as: significant reduction in aeration energy requirements (up to 60-90%); no need for external carbon source; smaller sludge production; smaller environmental footprint.

In order to provide the proper nitrite/ammonium ratio, Anammox process is usually coupled with other processes, as partial nitrification. The main two configurations for their combination are the single-reactor and two reactors partial nitrification-Anammox systems. In the former the two processes take place in the same reactor, while in the second, partial nitrification is carried out in the first reactor and Anammox in the subsequent second reactor.

Anaerobic ammonia oxidation potential was discovered around 15 years ago in Grist-Brocardes (Delft, Netherlands) and it found its main application in treating rejected water from dewatering of digested sludge (Bertino A., 2011). Indeed, it resulted to be particularly effective in treating highly nitrogen loaded streams with low organic carbon content, i.e. waters with low COD/N ratios, as rejected water is. Moreover, this sidestream have typically higher temperature if compared with the mainstream water, allowing process temperature closer to Anammox optimal range. The first full-scale deammonification plant started in 2001, treating rejected water at the WWTP of Hattingen, Germany. Other full-scale reject-water deammonification plants are currently in operation in Strass (Austria), Gralnerland-Zurich (Switzerland), Rotterdam Dokhaven (The Netherlands) and Himmerfjärden, Grödinge (Sweden). The next challenge for this promising process is now to succeed in its implementation also into the mainstream of wastewater treat-

ment.

### 1.2.3 The Deammonification process

The single-stage partial nitritation-Anammox process has several names, among with Deammonification and CANON (Completely Autotrophic Nitrogen removal Over Nitrite). Hereafter, this process will be referred to as Deammonification only. This combined treatment is also termed as autotrophic nitrogen removal (Van Hulle, 2005).

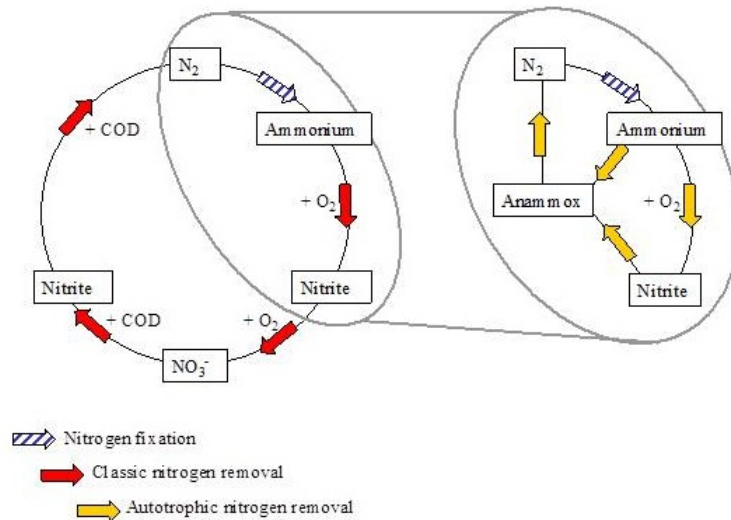


Figure 1.8: Nitrogen cycle in the Deammonification process

Such a system allows aerobic and anaerobic ammonium oxidizing bacteria (AOB and Anammox, respectively) to grow and co-operate together as the first provide the nitrite load required for the anaerobic ammonium oxidation. AOB and Anammox bacteria are actually competitors for ammonia nitrogen, whereas Anammox and NOB (still present under aerobic condition) compete for nitrite. According to the above mentioned observations, the main factors to keep the required bacteria population are: temperature, oxygen, sludge age, pH, substrate/inhibitors concentrations. At temperature lower than  $30^\circ C$ , oxygen is kept at low values in order to promote AOB over NOB and limit aerobic ammonia oxidation. Moreover low oxygen concentra-

tions are required for avoiding Anammox inhibition –Cui (2012) indicates a reference concentration of  $1 \text{ mgO}_2/\text{l}$ . With the same purpose and for promoting Anammox growth, the incoming nitrogen should be kept in the form of free and saline ammonia instead of nitrite or nitrate. In case biodegradable organic matter is present in the incoming flow, heterotrophic bacteria may thrive instead, either in aerobic or anaerobic conditions. In practice, DO and hydraulic retention time are the control factors, provided the required temperature and pH are achieved. In case the nitrate concentration in the reactor is too low, the DO concentration should be reduced and viceversa. If the ammonium concentration is too high, the hydraulic retention time should be increased instead (Cui, 2012).

The moving bed biofilm reactor (MBBR) system showed to be among the most feasible system for deammonification, together with granular sludge reactors and intermittently aerated sequencing batch reactors (SBR). In a MBBR system, the biomass grows as biofilm on free-flowing supports called carriers. Beyond the many advantages of such reactors (system capacity maximization; tank space minimization; high flexibility to load fluctuation) they seem to create good conditions for the cohesistence of the two different species of bacteria: aerobic nitrifying bacteria grow in the external layers, where oxygen diffusion is higher, while the anaerobic biomass is found instead in the internal part of the bio-carrier.

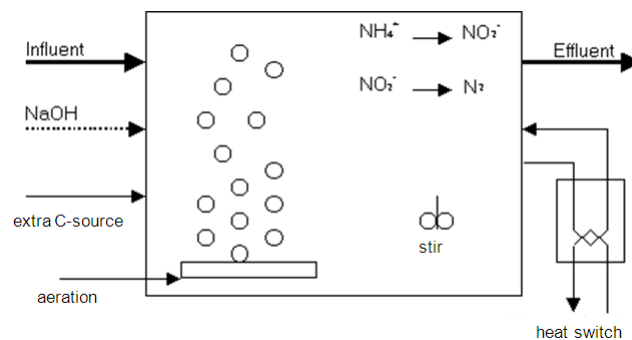


Figure 1.9: One-single reactor partial Nitrification-Anammox

The first full-scale deammonification plant started in 2001, treating rejected water at the WWTP of Hattingen, Germany. Other full-scale deammonification plants treating sludge liquor are currently in operation in Strass

(Austria), Gralnerland-Zurich (Switzerland), Rotterdam Dokhaven (The Netherlands) and Himmerfjärden, Grödinge (Sweden). All of these plants treat sludge liquor only. The next challenge for this innovative is to succeed in its implementation into the mainstream of wastewater treatments, at ambient temperatures.





# Chapter 2

## Material and methods

### 2.1 The Benchmark Simulation Model no. 1

The benchmark simulation model (BSM) is a simulation protocol, developed for the objective comparison and evaluation of wastewater treatment plant control strategies. The original simulation benchmark was first developed in the mid 90's by the International Water Association (IWA) Task Group on Respirometry-Based Control of the Activated Sludge Process, in order to provide a unified basis for evaluating the numerous control strategies proposed in the last decades and described in literature. The BSM1 and the subsequent BSM1\_LT and BSM2 resulted, instead, from the cooperation between the second IWA Respirometry Task Group and the European Co-operation in the field of Scientific and Technical Research (COST) in the Actions 682 and 624 (Copp, 2002), in the years running from the 1998 and 2004. The work is now continued under the umbrella of the IWA Task Group on Benchmarking of Control Strategies for WWTPs.

The BSM1 simulates the activated sludge unit of a benchmark wastewater treatment plant (WWTP), thereby including only the biological unit and the secondary clarifier.

The BSM1\_LT (BSM1\_Long Tern) is the BSM1 follow-up of BSM1 and it is devoted to long-term assessment of control strategies. For this purpose, the evaluation period is extended and the temperature is included in the simulation as well as sensors and actuators faults.

Finally, the BSM2 includes the BSM1 in a full treatment plant layout, considering both the wastewater and the sludge treatment lines. The units added in the second protocol are: primary settler, sludge thickener, anaerobic digester

and dewatering unit.

Other extensions are being considered so far, as the chemical precipitation, enhanced biological phosphorus removal and reactive settling for the future formulation of a BSM3 ([www.iwahq.org](http://www.iwahq.org)).

In order to increase their acceptability, all the BSMs include some of the most internationally accepted models for each process. None of them is linked to a particular simulation platform, so that different simulation software can be used for their implementation. The most widespread simulators used for the BSMs implementation are: BioWin<sup>TM</sup>, EFORTM, FORTRAN, GPS-XTM, MATLAB<sup>TM</sup>-Simulink<sup>TM</sup>, SIMBA<sup>®</sup>, STOATM and WEST<sup>®</sup>. The simulation models should be independent from the simulation software and using different software should not have any impacts on the modelling output, i.e. on the quality of the results. Though, this is not always the case and slight differences in the outcomes can be detected when using different simulators (Copp, 2002). A tuning procedure for the listed software is suggested in the COST Simulation Benchmark Manual, (Copp, 2002) so that users can compare their own results with the provided specifications.

### 2.1.1 General description

The BSM1 is property of the IWA Task Group and it is available for downloading on the website [www.benchmarkwwtp.org](http://www.benchmarkwwtp.org). The available version is implemented in MATLAB-Simulink and it was used in this study.

The plant layout comprises a biological reactor and a secondary settler. The biological reactor, in turn, comprises five completely mixed compartments: two anoxic followed by three aerobic. This configuration basically describes a pre-denitrification process layout, aimed to obtain biological Carbon and Nitrogen removal in the same AS unit. The oxygen concentration in the aerobic tanks is governed by the predefined maximum oxygen transfer coefficient,  $K_La$ , set equal to zero in the first two anoxic tanks. The secondary settler is simulated as a 10-layer tank, assuming all the layers have the same height and area. The feed is located at the sixth layer whereas the supernatant exits from the tenth. Two recirculation flows are provided: (i) the internal nitrate recycle from the fifth to the first biological reactor and (ii) the activated sludge recycle from the settler underflow to the first biological reactor.

The default control strategy is based on two independent controllers, aiming

to maintain the nitrate concentration in the second anoxic reactor and the dissolved oxygen concentration in the last aerobic reactor at certain set-point values. The first controller manipulates the internal recycle flow rate (for the nitrate recirculation); the second the oxygen transfer coefficient ( $K_{La}$ ) in the last aerobic reactor. Both the controllers are PI-controller type (Proportional Integrative controllers). A maximum value is defined for the two manipulated variables.

The plant layout is shown in Figure 2.1.

Figure 2.1: BSM1 Plant layout (Jeppsson, 1996)

The process models used in BSM1 are the Activated Sludge Model number 1 (ASM1) for the biological reactor and the 10-layer one-dimensional Takács double-exponential model for the secondary settler.

The BSM1 state variables are the same required by the ASM1 (listed in Table 1.1.) plus total suspended solids, TSS, and flow rate. The parameter used are those required by the two process models. The kinetic parameters, used in the ASM1 as default values for a reference temperature of 15°C are listed in Table 2.2. The stoichiometric and kinetic parameters, used in the ASM1 as default values for a reference temperature of 15°C are listed in Table 2.1 and Table 2.2. Table 2.3 presents, instead, the settling model parameters used in BSM1.

Note that the temperature is not considered neither as a variable nor a parameter. It is not directly included in the simulation, but its effect is implicitly accounted for in parameters' values. The default temperature is 15°C. Some of the parameters (namely the stoichiometric ones) are shown to be almost independent from the temperature, as the yield coefficient, the hy-

Parameter	Unit	Value
$Y_H$	g cell COD formed/(g COD oxidized)	0.67
$Y_A$	g cell COD formed/(g N oxidized)	0.24
$f_p$	-	0.08
$i_{X,B}$	gN/gCOD in biomass	0.08
$i_{X,P}$	gN/gCOD in particulate products	0.06

Table 2.1: Default ASM1 stoichiometric parameter in the BSM1

Parameter	Unit	Value
$\mu_H$	$d^{-1}$	4.0
$K_S$	gCODm <sup>-3</sup>	10.0
$K_{O,H}$	g(-COD)m <sup>-3</sup>	0.2
$K_{NO}$	g(NO <sub>3</sub> -N)m <sup>-3</sup>	0.5
$b_H$	$d^{-1}$	0.3
$\eta_g$	-	0.8
$\eta_h$	-	0.8
$k_h$	g slowly biodegradable COD.(g cell COD. d) <sup>-1</sup>	3.0
$K_X$	g slowly biodegradable COD.(g cell COD) <sup>-1</sup>	0.1
$\mu_A$	$d^{-1}$	0.5
$K_{NH}$	g(NH-N)m <sup>-3</sup>	1
$b_A$	$d^{-1}$	0.05
$K_{O,A}$	g(-COD)m <sup>-3</sup>	0.4
$k_a$	m <sup>3</sup> (gCOD.d) <sup>-1</sup>	0.05

Table 2.2: Default ASM1 kinetic parameters in the BSM1

Parameter	Unit	Value
$v'_0$	m/d	112.1
$r_p$	m <sup>3</sup> /g	2.70 x 10 <sup>-3</sup>
$r_h$	m <sup>3</sup> /g	2.93 x 10 <sup>-3</sup>
$f_{ns}$	-	2.59 x 10 <sup>-3</sup>

Table 2.3: Settler model parameters for high load plants (Takács, 1991)

drolysis coefficient (Petersen, 2000). Some of the kinetic parameters, instead, are dramatically affected by temperature. Their values have to be changed over time in the case of simulations implying temperature changes.

### 2.1.2 BSM1 implementation in MATLAB-Simulink

The simulation model is implemented in Matlab, using the user graphic interface of Simulink. In this MathWork tool, the simulation model is provided by block diagrams. Files created in this platform have the extension ".mdl" and algorithms used can be compiled both in C or MatLab programming languages. In the version used in this study, three Simulink models are available:

- Openloop.mdl;
- Benchmarkss.mdl;
- Benchmark.mdl.

The Openloop model is intended to simulate the plant behaviour without any active control. Then, the controlled variables (internal recirculation flow and  $K_La$  in reactor 5) have their constant default value. This model is usually used only for a preliminary assessment of the model behaviour on the user's software. The Benchmarkss.mdl is a model for a simulation with active control and ideal sensor. Finally, the Benchmark.mdl comprises both active control and real sensors: noise and delays are included for the sensors and actuators simulated in the model. The benchmarkss.mdl is used for preliminary long-term steady state simulations. The simulation period, at this step, is recommended to be around 100 and 150 days, at least. Benchmark.mdl is used for dynamic simulations instead. An example of the Benchmark.mdl layout is shown in Figure 2.2. The internal codes, used in the Simulink blocks refer to other C-files and m-files (already compiled and available online). The default models can be modified in order to fit with the real situation to be simulated. Concerning the influent characterization, four influent files are provided:

- Constant influent;
- Dry-weather influent;
- Rain-weather influent;
- Storm-weather influent.

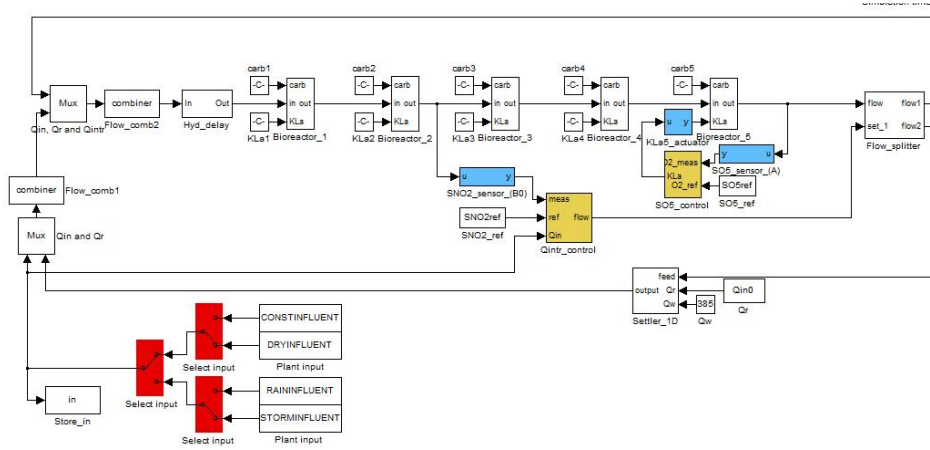


Figure 2.2: BSM1 implementation in MATLAB-Simulink

Three weather conditions are considered. During dry weather, the influent dynamic variation is simulated for a 2-week period without any precipitation: the maximum flowrate peaking factor is 1.74 (Copp, 2002) and daily as well as week-end variations are described. In the rain-weather file, it is simulated one week of dry weather followed by a long rain event in the second week; the maximum flowrate peak factor is 2.53 and no increase in pollutant load is simulated. The dynamic storm-weather data simulate one week of dry weather and two storm events in the second week. The flowrate peaking factor is the same as in the rain-weather file, but the flowrate distribution profiles two events of high intensity and short duration. As an effect of the first storm event, the flush-out of particulate matter in the sewer is simulated through a significant increase in inert and biodegradable suspended solids, whereas a modest increase is simulated after the second storm event, assuming that the sewers were cleared after the first (Copp, 2002). All these dynamic influent data are given at a time step of 15 minutes, for the 14-days period of weather simulation. The data include the 15 state variables and are given in the following order:

$$time S_I S_S X_S X_{B,H} X_{B,A} X_P S_O S_{NO} S_{NH} S_{ND} X_{ND} S_{ALK} TSS Q$$

Finally, in the constant influent file, the data are averaged from the dry-weather values, resulting in a 1-row 15-column file. The simulation procedure implies both a steady-state and a dynamic simulation. The constant influent file is used in this case and the running time period is set at 150 days. The dynamic simulation follows a preliminary steady state simulation in order to

have a consistent starting point and to limit the influences of dynamic starting conditions on the overall process (Copp, 2002). Dynamic simulations provide, instead, information on the short-time response of the process. The dynamic files (dry-weather, rain-weather and storm-weather influent) are used for this step. A 14-day dynamic simulation using the dry-weather file has to be performed prior to the simulation of each weather condition. Thereby, the actual simulation conditions will be: *dry-dry*; *dry-rain* and *dry-storm*, resulting in an overall running time of 28 days.

Out of the 28 days of dynamic simulation, the results of interest are those referring to the last week only (from day 22 to day 28 inclusive). Effluent concentrations and loads are calculated for the state variables and for other variables as total COD and total BOD. The BSM1 provides already several index for the process performance assessment, in order to help users in evaluating the huge amount of output data. An example of performance indexes are the effluent violations: they are calculated for five main pollutants (ammonia, total nitrogen, total BOD, total COD and total suspended solids), according to defined effluent limit concentrations –they are expressed both as number of violation and as percentage of time the plant is in violation. Operational variables as sludge production, pumping energy and aeration energy are calculated as well. Controller performance indexes are also provided as a different level analysis on the modelled plant performances. More detailed descriptions of performance assessment procedures are provided by Copp (2002) and Alex et al. (2008).

## 2.2 Sjölunda WWTP

The Sjölunda waste water treatment plant (WWTP) is located in the Northern part of the city of Malmö and it is one of the largest plants in Sweden. The receiving wastewater comes from at the greater extent from the city of Malmö, together with parts of other surrounding municipalities. Around 300000 residents are connected to the plant. Both combined and duplicated sewer systems are present in Malmö and the collected wastewater is pumped to the plant, thanks to several pumping stations located at the downstream part of the uptake areas. The plant is designed to serve a population of 550000 equivalents (considering an equivalent load of 70 gBOD<sub>7</sub>/d per person) corresponding to an average wastewater flow of 143000 m<sup>3</sup>/d (1650 l/s, according to the annual report 2011). In 1998-1999 the plant was upgraded to enhanced nutrient removal, due to the more stringent regulation standards requiring effluent limits of 10 mg BOD<sub>7</sub>/l, 0.3 mg total-P/l and 8 mg total-N/l. The upgraded scheme was designed to perform the biological carbon removal and the biological nitrogen removal separately. The first is accomplished by the high-loaded AS treatment, the latter by a high rate nitrifying trickling filter followed by a post-denitrification anoxic moving bed biofilm reactor (MBBR). Such a solution was selected for being compact (in terms of space requirements) and cost-effective (Hanner et al., 2003).

The overall wastewater treatment process comprises the following units:

- Flow equalization: it is provided by the trunk sewers between the large wastewater pumping station in the sewer network and the Sjölunda WWTP.
- Inlet pumping station: three pumps are used during dry-weather conditions whereas, in case of storm or rainy conditions, the wastewater exceeding the plant capacity is pumped in the adjacent overflow plant.
- Screenings separation: wastewater undergoes 3-mm screening. The separated screening is washed and dewatered prior to combustion for energy production.



- Grit removal: grit is removed in aerated removal tanks. Grit is removed and pumped to the grit treatment unit.
- Grit treatment: grit is washed and dewatered in order to wash out the organic matter. It is then transported to containers and used in soil construction.
- Pre-precipitation: ferrous sulphate is added at the inlet of the pre-aeration basin in order to remove phosphorus through chemical precipitation. The chemical addition also gives rise to better primary sedimentation performances.
- Primary sedimentation: primary sludge sinks in the primary sedimentation basins and is therefore pumped out to the sludge treatment line.
- Flow measuring devices: flowrate is measured in Parshal-flumes ahead the activated sludge unit.
- Activated sludge plant: biological carbon removal and partial nitrogen removal take place under high-loaded condition.
- Secondary sedimentation: activated sludge sank in rectangular clarifiers and is mainly returned ahead of the AS plant. The excess sludge is removed as wastage sludge and treated in the sludge treatment unit.
- Nitrifying trickling filters: after the upgrade in 1999 four trickling filters with a large surface area were built. Oxygen is added by natural draught through the filter, allowing nitrification of the incoming ammonia to occur.
- Post-denitrification in moving bed biofilm reactors: the second step of biological nitrogen removal is carried out by anaerobic heterotrophs growing in plastic carriers placed inside the MBBR basins. External readily biodegradable carbon source (methanol) is required and added,

being the wastewater at this stage, reach in nitrate and poor in biodegradable organic matter. Methanol addition is regulated by means of nitrate and flow on-line meters.

- Flotation plant: the last stage in the wastewater treatment is the removal of particles (mainly biological flocs), coming from the upstream nitrogen removal units. Dispersed water, achieved through the injection of very tiny air bubbles, lifts particle to the surface. The lifted sludge is then scraped and pumped to the sludge line.

The sludge treatment line comprises:

- Primary sludge thickening: the sludge from primary clarifiers is sent to gravity thickeners and the thickened sludge is further dewatered. The excess water is pumped back to the head of the plant.
- Surplus sludge thickening: surplus sludge comprises sludge from secondary settlers and the flotation unit. It is thickened mechanically through a filter press, with polymer addition.
- Anaerobic digestion: the dewatered primary sludge and the thickened surplus sludge undergo mesophilic anaerobic digestion (process temperature of 35-37°C). Three lines, with two digestors each, are present in the plant. The biogas is used for energy and heat production.
- Digested sludge dewatering: the digested sludge is dewatered in centrifuges, with polymer addition. The rejected water (sludge liquor) is further treated in the plant, while the dewatered sludge is sent for final disposal or utilization, according to its quality.
- Sludge liquor treatment: the ammonia-rich sludge liquor is treated in a sequencing batch reactor (SBR), where the partial nitrification process takes place, under oxygen-limited conditions. The effluent is then sent to the mainstream activated sludge unit.

### 2.2.1 The high-loaded activated sludge unit

In the current configuration, the biological removal of organic matter is carried out into four activated sludge (AS) plants:

- G1, G2 and G3 plants treat almost half of the total incoming flow. Each of them consists of two parallel lines with separate sludge recirculation, operating under high-loaded conditions. In year 2012, G3 plant was shut off and the wastewater diverted to G1 and G2;
- G4 plant treats the remaining wastewater flow in three parallel lines, with separate sludge recirculation.

G2-line2 is the high-loaded AS unit under investigation in this study. According to the online data on autumn 2012, it treated an average flow rate of almost 13200 m<sup>3</sup>/d. The biological reactor comprises five compartments: the first two have a volume of 206 m<sup>3</sup> each and operate under anoxic condition; the last three tanks have a volume of 412,5 m<sup>3</sup> each and are kept under aerobic condition (Table 2.4). The whole biological unit can be considered as a quasi plug-flow reactor and each tank as completely stirred-tank reactor (CSTR). An average nitrite load of 60 kgNO<sub>2</sub><sup>-</sup>-N/d is also received from the additional stream coming from the SBR, where partial nitrification takes place. This additional stream is pumped intermittently over the day and the nitrite load is removed through denitrification in the first two anaerobic compartments. The high load condition results in BOD<sub>7</sub> loads around 0.2-0.3 kgBOD<sub>7</sub>(kg aerobic MLSSd)<sup>-1</sup>, aerobic sludge age of 1-1.5 days and high suspended solids concentrations either in the reactor and in the recirculated flow. The excess sludge flowrate is set manually, in order to keep the MLSS concentration around 2200-2600 mgSS/l. No internal mixed liquor recirculation is provided as the high-loaded AS unit is intended to accomplish with the organic carbon removal only (biological nitrogen removal occurs in a small extent due to the nitrite load from SBR effluent, but it is not included in the removal of influent wastewater nitrogen load). Aeration is regulated by a cascade PI regulator, with feedback on the oxygen level in the last aerobic tank. G2 line 2 has its separated sludge recirculation system. A two-tank rectangular secondary clarifier, with a height of 3.8 m and an overall surface of 465 m<sup>2</sup>, follows the biological reactors. Hindered sludge is extracted and partially recirculated in the AS system.

### 2.2.2 Future perspective: the Manammox project

In autumn 2011, a two-year project was started at Sjölanda WWTP in collaboration with Lund University. It aims to gain insight into the deammonification process, with the future perspective of its full-scale implementation

	Notation	Value	unit
Tank characteristics			
Volume, reactors 1-2	Vol1, Vol2	206	$m^3$
Volume, reactors 3-4-5	Vol3, Vol4, Vol5	412,5	$m^3$
Volume, secondary settler		1650	$m^3$
Surface, secondary settler		465	$m^2$
Height, secondary settler		4	$m$
Hidraulic retention time	HRT	6	$h$

Table 2.4: ASM1 kinetic parameters at 15 C

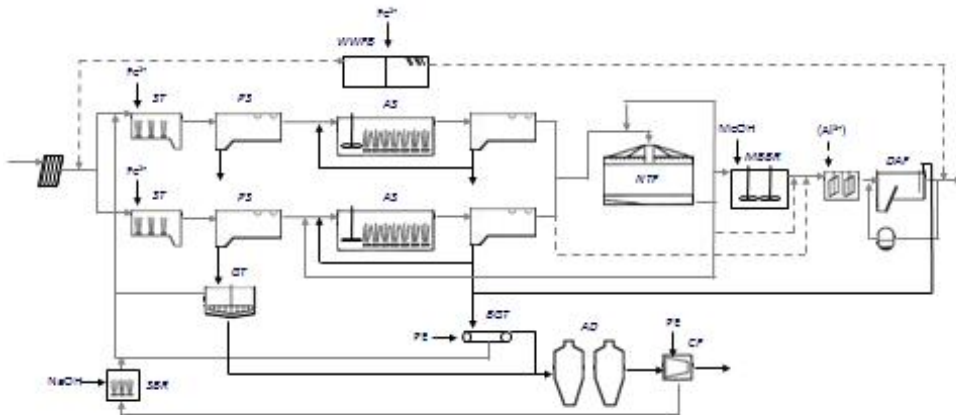


Figure 2.3: Sölunda wastewater treatment plant

in the wastewater mainstream, in place of the current nitrogen removal system (nitrifying trickling filter combined with denitrifying MBBR). A pilot plant was built and included reactors treating both sludge liquor from the dewatering of the digested and the effluent from the high-loaded activated sludge plant in the mainstream. Deammonification couples partial nitrification and anaerobic ammonia oxidation (anammox) processes. It is within the research front described in above that the pilot-plant at Sjölanda WWTP finds its place. Would the implementation on Sjölanda wastewater mainstream succeed, plant operating costs will be reduced significantly as well as plant environmental impact, in terms of energy requirements. This new process will follow the high-load AS unit. In case also the production of biogas from the activated sludge anaerobic digestion would be further boosted, a higher production of energy coupled with the lower deammonification energy requirement could turn the biological treatment plant into energy neutral or

even net-energy producer. According to this perspective, the oxygen requirement would be dramatically reduced since only partial nitrification is carried out by ammonia oxidizing bacteria (AOB) and the remaining ammonia is anaerobically oxidized by anammox bacteria, using nitrite as final electron acceptor. No external organic carbon source is required, being both AOB and anammox autotrophic bacteria. The main disadvantage of deammonification full-scale implementation lies in its extreme sensitivity to environmental and operational conditions. Factors as temperature, pH, oxygen content, sludge age as well as influent composition should be properly controlled in order to create the best condition for Anammox bacteria to thrive.

## 2.3 Historical data and measuring campaign

### 2.3.1 Historical data

The Sjölanda WWTP monitoring system provides continuous measurements of several parameters, thanks to meters placed in strategic points of the plant. For the purpose of this study, only data referred to the AS unit G2-line2 and the corresponding second clarifier were selected. All the following considerations refer to this specific line and not to the overall plant (for example, the influent and effluent flows are considered as those entering and exiting that specific line only). The five compartments are referred to as reactor 1 to 5, starting from the first anoxic tank.

Three sets of historical data are used. One refers to the entire year 2005; one to the autumn 2012 (comprising the months of September, October and November) and the last one on the months of January and February, 2013. This data set will be referred to as Year2005; Autumn2012 and January-February2013, respectively. They all comprise three different types of data, coming from:

- online measurements: by mean of probes and flow meters;
- laboratory measurements on daily samples: with a frequency of three days a week, on average;
- laboratory measurements on grab samples: with a frequency of three days a week, on average.

Online measurements from Year2005 give 6-minute data, whereas for Autumn2012 and January-February2013 1-minute data are available. Figure 2.4 summarises the available data from each data set and the type of measurement. The COD measurement is actually one of the most important ones, being crucial in the definition of ASM1 COD fractions. The Year2005 set was used only as a reference data source for an average plant behaviour assessment over the year, in term of effluent quality as well as influent fluctuations.

		Type of measurement		
		Year2005	Autumn2012	January-February2013
Influent flow (l/s)		Online measurement (every 6 minutes)	Online measurement (every minute)	Online measurement (every minute)
Excess sludge flow (l/s)				
Recirculated sludge flow (l/s)				
DO, reactor 3 (mg/l)		Online measurement (every 6 minutes)	Online measurement (every minute)	Online measurement (every minute)
DO, reactor 4(mg/l)				
DO, reactor 5 (mg/l)				
Airflow rate, reactor 3 (Nm <sup>3</sup> /h)		Online measurement (every 6 minutes)	Online measurement (every minute)	Online measurement (every minute)
Airflow rate, reactor 4 (Nm <sup>3</sup> /h) <sup>1</sup>				
Airflow rate, reactor 5 (Nm <sup>3</sup> /h)				
MLSS (mg/l)		N.A.	Online measurement (every minute)	Online measurement (every minute)
SS (mg/l)	Influent	Daily samples	Daily samples	Daily samples
	Effluent			
VSS (mg/l)	Influent	Daily samples	Daily samples	Daily samples
	Effluent			
COD and COD <sub>nitr</sub>	Influent	N.A.	N.A.	Daily samples
	Effluent			
BOD <sub>7</sub> and BOD <sub>7nitr</sub>	Influent	Daily samples	Daily samples	Daily samples
	Effluent			
N-NH <sub>3</sub> (mg/l)	Influent	Daily samples	Daily samples	Daily samples
	Effluent			
N-NO <sub>2</sub> <sup>-</sup> (mg/l)	Influent	Daily samples	Daily samples	Daily samples
	Effluent			
N-NO <sub>2</sub> <sup>-</sup> (mg/l) and N-NO <sub>3</sub> <sup>-</sup> (mg/l)	Influent	Daily samples	Daily samples	Daily samples
	Effluent			
N <sub>tot</sub> (mg/l)	Influent	Daily samples	Daily samples	Daily samples
	Effluent			
SS, reactor 5 (mg/l)		Grab samples	Grab samples	Grab samples
VSS, reactor 5 (mg/l)				
SS, excess sludge (mg/l)		N.A.	Grab samples	Grab samples
VSS, excess sludge (mg/l)				

Figure 2.4: List of available historical data on year 2005, autumn 2012 and January-February 2013

### 2.3.2 Measuring campaign

Measurement campaigns are usually carried out in order to provide dynamic data at a higher sampling frequency and accuracy of those routinely provided by the WWTP data collection system. Collected data can be used for dynamic assessment of both the model calibration and the validation step. The former is the case in this study. Sampling frequency can be defined according to the time constant of the process under examination. An important time constant in the activated sludge process is the hydraulic retention time. Some authors suggest to choose a sampling frequency at a time about 5 times faster than the HRT and a campaign duration of 3-4 times the HRT (Petersen et al., 2002). In this study, a hydraulic retention time of about 6 hours is calculated, if both the AS unit and the second clarifier volumes are considered as a whole system. It was decided to carry out a one-day measuring campaign, providing samples for every hour of the day. The above mentioned indications are, in fact, accomplished.

The 24-hour measuring campaign was conducted on Wednesday, 23 January 2013, starting from midnight to the following midnight. During the week before, the weather was stable, except for some moderate snow events. Air temperature was also stable at values below zero. (source: the Swedish Meteorological and hydrological Institute). Under these conditions the observed data can be considered as reliable and representative for Sjölanda WWTP average conditions, since the low temperature did not lead to any significant extra-flow due to snow melting. Two automatic samplers were installed. Sampling was time-proportional, taking around 70 ml of water every 10 minutes, for a hourly sample of around 420 ml. Sampling points were located at the inlet of G2 line 2 and at the outlet of the corresponding second clarifier. Analyses should be run as soon as possible after sampling, being time delay affecting the results (especially those on nitrogen). Unfortunately, it was not possible to run the analyses before 9 in the morning of Thursday, 24 January 2013. Therefore, samples referring to the first hours of the day were analysed with a relevant delay; nevertheless the very low temperature registered over the night (around 10°C minus) may have limited this effect. According to standards methods, the following analyses were conducted: TSS, VSS,  $N_{tot}$ ,  $NO_2^-$ -N,  $NO_3^-$ -N,  $NH_4^+$ -N, Alkalinity, COD and  $BOD_7$ . The latter two parameters were analysed both for filtered and unfiltered samples, in order to gain information on their soluble and particulate fractions. Alkalinity was



measured for every-second-hour samples only, whereas BOD measurements were conducted for 2-hour samples obtained by mixing two hourly sample. Filtration was carried out by mean of Munktell MGA glass fibre filters, pore size of 1.6 micrometers. Petersen et al. (2002) suggested to use a 0.45  $\mu\text{m}$  filter to better separate soluble and particulate fractions. Thus, the registered values for filtered samples could be slightly higher if compared with those presented in their or other case studies. Alkalinity was measured using Radiometer TIM 840 tritator (Bergman Labora, Danderyd, Sweden), according to EN-ISO 9963-2 standard method. COD and total nitrogen analyses were performed by using Hach LCK114 COD cuvettes and Hach LCK238LatoN Cuvette respectively, read in Hach Lange DR 5000 spectrophotometer. Nitrogen analyzer FOSS FIAStar 5000 (FOSS Analytical, Höganäs, Sweden) was used for measuring free and saline ammonia, nitrate and nitrite concentrations, as well as total nitrogen (standard methods: SS-EN ISO 11732:2005, SS-EN ISO 13395, SS-EN ISO 14403-1:2012). Total nitrogen was measured either through spectrometer and nitrogen analyzer. BOD<sub>7</sub> test was performed according to SS-EN ISO 1899-2 standard method, using Skalar Robotic Analyzer (Skalar, Breda, Netherlands). An accuracy of 10% was estimated for TSS, VSS, COD, Alkalinity and all nitrogen compounds, while for BOD<sub>7</sub> measurements it can be estimated as 20%.

## 2.4 Laboratory experiments

### 2.4.1 Oxygen Uptake Rate experiment

Respirometric tests were performed in order to estimate the main biological kinetic parameters. Respirometry has found a wide application in biological characterisation of either wastewater and activated sludge. It consists in the measurement and interpretation of oxygen depletion rate of activated sludge. (Petersen, 2000) In the context of activated sludge process models, it is recommended to support the traditional chemical-physical as well as biological analyses with respirometric tests, in order to gain information on the specific active biomass under investigation. In fact –differently from the traditional BOD analysis –the OUR test is performed using the sludge sampled from real activated sludge plants. Differently from conventional BOD tests, where oxygen depletion measurements refers to the start and the end of the experiment only, respirometric tests provide oxygen continuous monitoring, giving information on substrate degradation evolution throughout the experiment. Either wastewater or easily biodegradable compounds can be added as external substrate for the active biomass. The additional information that can be obtained refers to biomass activity, by defining kinetic and stoichiometric parameters and/or wastewater biodegradability. Several methods have been developed so far and they mainly differ in: (i) the phase in which the oxygen is measured (liquid or gaseous) and (ii) the flow regime of the experiment, i.e. continuous flow or batch system (Vanrolleghen et al., 2000).

The typical dissolved oxygen and OUR curves resulting from acetate addition in intermittently aerated activated sludge batch reactors are shown in Figure 2.5 and Figure 2.6, respectively. If the measurement of volatile suspended solids is available, the specific oxygen respiration rate (SOUR) can be obtained by dividing the measured OUR by VSS concentration. The SOUR is expressed in  $\text{mgO}_2(\text{gVSS h})^{-1}$  and may be more appropriate in case of respiration rate comparison among different sludge samples. In case of intermittently aerated, the oxygen uptake rate can be derived as the slope of the DO concentration curve during the non-aerated phase (Keskitalo et al., 2010, Hagman and Jansen, 2007), being the decrease in oxygen concentration typically linear (Hagman, 2007). It is generally accepted that the OUR is composed by two components: the endogenous and the exogenous respiration rate. As reported in Spanjers (1993), the endogenous respiration rate refers

to the oxygen consumption in absence of external substrate, even though authors do not totally agree about the biological meaning of this measurement (Patry and Chapman, 1989; Sushka and Ferreira, 1986; Bathla et al., 1966). The exogenous respiration rate, instead, refers to the (higher) oxygen consumption following external substrate addition. Easily biodegradable substrate leads to a rise in exogenous respiration rate, up to a maximum value at which all aerobic heterotrophic biomass is growing at the maximum specific growth rate (Hagman, 2007). Assuming that the endogenous term remains constant throughout short-term experiments (Gernaey et al., 2000) –as an intrinsic characteristic of the biomass in the experimental specific conditions –the exogenous respiration rate can be calculated as the difference between the measured (total) oxygen uptake rate and the endogenous respiration rate. As shown in Figure 2.5 and Figure 2.6, after substrate addition, the slope of the DO depletion in the non-aerated cycle is higher (meaning that the DO depletion is faster), resulting in a higher value of the total OUR. The plateau at the maximum exogenous respiration rate, indicates the time interval in which neither substrate nor oxygen limit the process. On the contrary, the descendent phase of the OUR curve accounts for the substrate-limited condition (due to its progressive depletion), causing a decrease in the actual growth rate. The added substrate can be considered completely degraded when the respiration rate levels off around the endogenous value again. According to the ASM1 formulation, the respiration rate – in case of activated sludge in the presence of nitrifying inhibitors –can be expressed by the following expression (Ciabini, 2005):

$$OUR = \frac{1 - Y_H}{Y_H} \mu_H \left( \frac{S_S}{K_S + S_S} \right) X_{B,H} + b_H X_{B,H}$$

#### *Experimental layout*

According to the classification given by Vanrolleghem et al. (2002), the procedure used in this study can be defined as a flowing gas-static liquid method. Acetate is used as external (readily biodegradable) substrate, instead of wastewater. Thereby, the OUR results are used for the active biomass characterization only and no further information on the wastewater biodegradability can be obtained.

Two experiments were performed on the 20<sup>th</sup> of December 2012 and on the

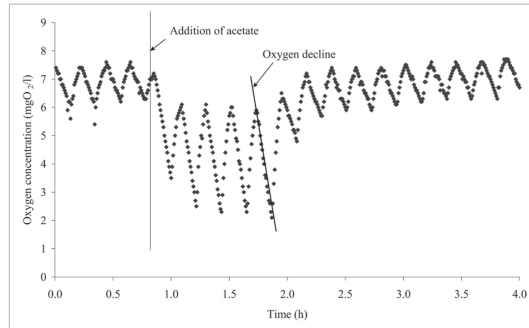


Figure 2.5: Dissolved oxygen concentration during respirometric experiment on activated sludge, with intermitted aeration and acetate addition (Hagman, 2007)

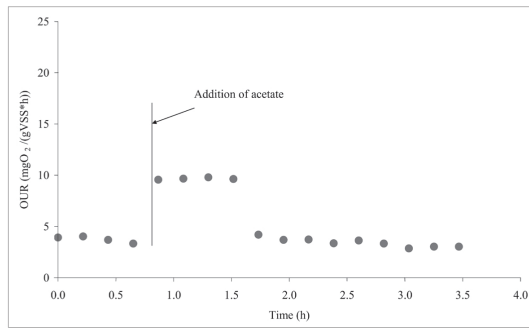


Figure 2.6: Oxygen uptake rate during respirometric experiment on activated sludge, with intermitted aeration and acetate addition (Hagman, 2007)

28<sup>th</sup> of January 2013 –they are referred to as Experiment1 and Experiment2 – according to the procedure described by Hagman (2007) and Keskitalo et al. (2010). In each experiment, two parallel reactors were run simultaneously, in order to improve results reliability. They are referred to as A and B data set. The experiment set-up consisted in a two-litre batch reactor, equipped with a lid, and continuously stirred by a mechanical stirring device (provided by IKA RW, set at a speed of 200/300 rev/min). The reactor was placed in a water-bath, in order to keep the temperature constant throughout the experiment, by means of a thermostat device, provided by LAUDA. Temperature was measured continuously and the selected values were set as close as possible to the one measured in the aeration basin at the sampling moment. Aeration was provided by a porous-stone diffuser connected to a STELLAR compressor through a silicon tube of 3 mm in diameter. Dissolved oxygen concentration was detected with immersed HQd HACH probe, and recorded every ten seconds. The aeration system was such that the DO concentration

in the reactor never reached values lower than 2 mg/l. This threshold concentration is usually considered sufficient for keeping the process not limited by oxygen. Reactants and probes were inserted through holes in the reactor's lid. One and a half litre of sampled activated sludge was poured in each reactor and nutrients and nitrification inhibitor were added: 15 ml of nutrient stock solution ( $\text{NH}_4\text{SO}_4$  0.236g/l,  $\text{KH}_2\text{PO}_4$  0.044g/l) and 4.5 ml of ATU stock solution (ATU 12 mg/l). First, aeration was provided continuously for at least one hour, in order to degrade any hydrolysis products that may have formed during sample transportation (Keskitalo et al., 2010) – it is recommended to carry out the experiment as soon as possible, after sludge sampling to limit particulate hydrolysis. Once Temperature and oxygen concentration reached constant values, intermittent aeration started. Alternate aeration was provided thanks to a timer connected to the air compressor and set at the required time interval. In Experiment1, ten-minute cycles were performed: aeration was kept on for five minutes and then off for other 5 minutes. In Experiment2, the cycle time was reduced to six minutes – resulting in three minutes of aeration followed by three minutes without aeration. Before substrate addition, the reactor was alternatively aerated for one or two hours. Once the respiration rate (i.e. the endogenous respiration rate) reached a constant value for several consecutive cycles, acetate was added. Acetate was added at the end of the aerobic phase, in order to clearly detect the increase in oxygen consumption due to its degradation. The acetate stock solution had a COD concentration of 20000 mg COD/l. In Experiment1 the amount of acetate added to the system was 6 ml of stock solution, in order to have 80 mg COD/l in the reactor. In the second experiment, the amount of acetate was doubled up to 12 ml (giving 160 mg COD/l in the reactor). In Experiment2, it was decided to increase the substrate concentration and decrease the cycle time in order to have a better understanding of the half saturation constant  $K_s$ . Samples of 15 ml were taken from the bulk activated sludge sample to measure TSS and VSS concentrations. pH was measured three times during the experiment: before nutrient addition, before acetate addition and after acetate degradation. Experimental conditions of both the experiments are summarised in Table 2.5.

	Experiment1	Experiment2
Activated sludge volume ( <i>l</i> )	1.5	1.5
Temperature ( $^{\circ}C$ )	15	15
Acetate solution addition ( <i>ml</i> )	6	12
Acetate COD addition ( <i>mgCOD/l</i> )	80	160
Cycle duration ( <i>min</i> )	10	6

Table 2.5: Experimental characteristics

## 2.4.2 Zone settling velocity test

Zone settling velocity (ZSV) tests were performed on January 29<sup>th</sup> to estimate the maximum Vesilind settling velocity. The apparatus described in literature (White, 1975; Catunda, 1992) comprises a transparent vertical stirred cylinder, 70-100 cm high and 10-20 cm in diameter, where the sludge is poured and let settling. After a short time (typically few minutes), a sharp solid-liquid interface is shaped and solid particles are considered to settle at the same velocity as the interface. After a high-rate phase of almost 10-20 minutes, ZSV decreases as the solid-liquid interface approaches the high concentration zone at the bottom of the cylinder. The displacement of the sludge blanket is recorded over time and the zone settling velocity is assessed as the slope of the first high-rate phase.

The test is performed at different sludge solids concentrations, usually obtained through dilution from the original activated sludge sample, in order to correlate solids concentration and zone settling velocity. In batch-wise operating device, no bulk movement is present and sludge settles under gravitational effect only. Vesilind gravitational settling model has been acknowledged by several authors to give the best description of empirical ZSV test data (Vanderhasselt et al., 1999). Its linear formulation describes a linear dependence between the settling velocity normal logarithm and sludge concentration:

$$\ln zZVS = \ln v_0 - \alpha X$$

The natural logarithm of Vesilind settling velocity,  $v_0$ , and parameter  $\alpha$  are obtained as the best-fitting values through the least square method. In this study, the apparatus was slightly different from those described in literature: the cylinder was 40 cm high and 10 cm in diameter; no stirring device was present.

The sludge was sampled from G2-line2 AS basin and diluted with clarified water from settler effluent; four dilution levels were evaluated. Almost one-litre sample was used for each test and two parallel columns were used simultaneously in order to have higher data reliability. Sludge interface height was recorded every minute during the high-rate phase and every five minutes at the slower one. The total test duration was around 50 minutes. Samples were taken at each diluted step and TSS concentration was measured in laboratory afterwards.

## 2.5 Laboratory-scale model

A sub-model was created to simulate the respirometric experiment as a lab-scale activated sludge system. It was intended to simulate the oxygen concentration and the OUR curves, with the purpose of estimating the main kinetic parameters: the maximum growth rate,  $\mu_H$ , the half-saturation coefficient,  $K_S$ , and the death coefficient,  $b_H$ . It was decided to model RED data set of both the experiments. A similar approach was implemented by Keskitalo et al. (2010) and Ciabini (2006). The lab-scale model was based on the same processes described in the ASM1, simulating a batch CSTR system with external substrate addition. A simplified equation set was used: out of the 13 state variables, only the following ones were taken into consideration:

- soluble biodegradable substrate,  $S_S$ ;
- particulate biodegradable substrate,  $X_S$ ;
- active heterotrophic biomass,  $X_{B,H}$ ;
- dissolved oxygen concentration,  $S_O$ .

The following assumptions were made:

- the external substrate entered the system at a defined time instant as soluble biodegradable COD, i.e. the  $S_S$  fraction;
- all processes that refer to nitrogen fractions were neglected;
- hydrolysis of particulate substrate accounted for the aerobic term only.

Since the external substrate was acetate, the first assumption is directly justified. The second assumption is due to the absence of nitrogen fractions characterization over the respirometric experiment. The last one implies that the term that refers to the hydrolysis under anoxic condition is neglected: according to the ASM1 formulation, it already contributes to the hydrolysis process at a lower extent compared with the aerobic term and its mathematical formulation requires the characterization of the soluble nitrate and nitrite fraction,  $S_{NO}$  –not available measurement. As a consequence, the modelled hydrolysis was probably underestimated and this has to be taken into account during the interpretation of the results. The process equations are the same used in ASM1, changed according with the above mentioned assumptions and solved through the finite differences method, forward in time.

The second term in  $S_O$  differential equation results from oxygen mass balance, to include external aeration in the process. In order to simulate the



alternate aeration, the oxygen transfer coefficient,  $K_{La}$ , was manipulated. For the aerated phase, its value was calibrated and set at  $0,7 \text{ min}^{-1}$  – it is reasonably higher than the one used in the full-scale model, being the oxygen transfer much more efficient in the batch lab-scale rather than in the full-scale –whereas in the non-aerated phase, its value was simply set at zero. Oxygen saturation concentration,  $S_{O,sat}$ , was set according to the highest oxygen concentration recorded during the aeration phase in the experiments. Acetate addition was simulated by setting the soluble substrate,  $S_S$ , at the injected concentration at the addition instant: 80 mgCOD/l for the first experiment and 160 mgCOD/l for the second. Except for the three kinetic parameters to be calibrated and the initial biomass concentration ( $X_{B,H}$  at the initial time), all other parameters involved in the model were set at their default ASM1 values (Table 1.1).

## 2.6 Dynamic evaluation of AS unit performances

A dynamic evaluation of the G2-line2 unit was carried out in order to assess the effect of plant operational changes on effluent quality, in the perspective of Manammox implementation. The AS unit would represent a pre-treatment for Manammox process, and it should achieve high organic removal, especially for readily biodegradable COD, and prevent nitrification. Three main scenarios were simulated in order to have an insight on possible new plant configurations capable to provide a proper influent for Manammox process. Their performances were assessed either under average and critical conditions. Higher attention was paid on average condition, being anammox bacteria more sensitive to average long-term conditions rather than short critical events. First, six conditions were analysed:

- Constant influent: for steady state analysis;
- Dry influent: for average dynamic state analysis;
- Rain and storm influent: for dynamic evaluation in case of high flow rate and high solids load;
- Temperature drop: for dynamic simulation of critical winter conditions;
- High temperature: for dynamic simulation of critical summer conditions.

### 2.6.1 Scenarios description

In sight of possible operational changes, three main scenarios and one sub-scenario were defined. In case Manammox process is implemented, the current plant configuration with two anoxic basins could be modified as denitrification will not be necessary any more. The three main scenarios are the following:

- Scenario 1: Current configuration: two non-aerated and three aerated compartments.
- Scenario 2: Five aerated compartments.
- Scenario3: Three aerated compartments only.

In order to better compare scenarios results, each scenario was modelled so that sludge age and organic removal efficiency were almost the same for all of them and close to the current plant condition. This was obtained by manipulating sludge age and dissolved oxygen concentration in aerated basins. A

sub-scenario 1-b was added in order to have the same DO set-point as scenarios 2 and 3, and to compare the effect of different aerobic volumes in the three main configurations. These changes resulted in the following characteristics:

- **Scenario 1:** wastage flow rate  $Q_W = Q_{W0} = 190 \text{ m}^3/\text{d}$ ; average sludge age of 1.2 d and DO set-point in aerated basins equal to measuring campaign values;
- **Scenario 1-b:** wastage flow rate  $Q_W = Q_{W0} = 190 \text{ m}^3/\text{d}$ ; average sludge age of 1.2 d and DO set-point in aerated basins equal to 1.5  $\text{mgO}_2/\text{l}$ ;
- **Scenario 2:** wastage flow rate  $Q_W = Q_{W0} = 190 \text{ m}^3/\text{d}$ ; average sludge age of 1.2 d and DO set-point in aerated basins equal to 1.5  $\text{mgO}_2/\text{l}$ ;
- **Scenario 3:** wastage flow rate  $Q_W = Q_{W0} = 140 \text{ m}^3/\text{d}$ ; average sludge age of 1.2 d and DO set-point in aerated basins equal to 1.5  $\text{mgO}_2/\text{l}$ ;

## 2.6.2 Observation on nitrification kinetics

Nitrification process is dramatically sensitive to temperature changes. Ekama and Wentzel (2008) defined the maximum autotrophic growth rate,  $\mu_A$ , and the ammonia half-saturation coefficient,  $K_{NH}$ , as high temperature sensitive parameters. In order to calculate their actual value at any temperature, using Arrhenius dependence equation, they suggested a coefficient of 1.123 instead of the usual values of 1.02-1.08. Death autotrophic coefficient,  $b_A$ , is instead considered of temperature sensitivity as low as for heterotrophic biomass (i.e. Arrhenius coefficient of 1.02). Salem et al. (2002) suggested alternative expressions for calculating  $\mu_A$  and  $K_{NH}$  at the desired temperature, as shown below:

$$\mu_{A,T} = 0.8 \left[ \exp^{-0.098(20-T)} \right]$$

$$b_{A,T} = 0.15 \left[ \exp^{-0.098(20-T)} \right]$$

For each temperature condition, the minimum sludge retention time required for nitrification can be calculated according to the following equation (Ekama and Wentzel, 2008; Salem et al., 2002):

$$SRT_{min} = \frac{1}{\mu_{A,MAX} \left( \frac{S_{NH}}{S_{NH}+K_{NH}} \right) - b_A}$$

This formulation results from autotrophic biomass mass balance under steady state condition. Clearly, temperature strongly affects its value as well, being it dependent on kinetic parameters. If the formulation presented by Ekama and Wentzel (2008) is applied, a temperature drop of 6°C would halve  $\mu_A$  and more than double SRT<sub>min</sub>. Sludge age values lower than this specific value are also referred to as "washout" sludge ages, in which cases the system is not be able to sustain nitrifying biomass (Ekama and Wentzel, 2008). The calculated minimum sludge age at different temperature conditions (average temperature, temperature drop and high temperature) was used in support of simulation outcomes interpretation.

### 2.6.3 Results interpretation procedure

A preliminary scenario analysis was made by comparing their performances over the six simulated conditions. Simulations focused mainly on the following effluent fractions and ratios:

- Readily biodegradable COD fraction,  $S_S$ ;
- Free and saline ammonia,  $S_{NH}$ ;
- Nitrite and nitrate fraction,  $S_{NO}$ ;
- Total and biodegradable COD;
- Solids;
- COD/N and bCOD/N ratios.

All values refer to average effluent concentrations based on loads.

In a second step, the best scenario was selected and further analyses on sludge age and DO variations were performed, in order to explore possible operational ranges and their effect on effluent quality. Several dynamic simulations were carried out to assess the influence of sludge age and oxygen separately:

- Analysis on DO concentration: sludge age was kept constant at the current value of 1.2-1.3 days. DO set-point in aerobic reactors ranged between 0.5 and 2.5 mgO<sub>2</sub>/l
- Analysis on sludge age: aerobic reactors DO set-point was kept constant at 1.5 mg/l . Sludge age ranged between 1 and 2.7 days.

## 2.7 BSM1model modifications

In order to make the simulation environment fit the real plant, some model adjustments were required. They mainly regarded plant operational characteristics, control strategy and influent characterization.

### 2.7.1 Plant operational characteristics

The default BSM1 five-compartment layout already fitted the current G2-line2 outline. Changes were required, instead, for the tank volumes, the recirculated and wastage sludge flow rates and the set-point oxygen concentration in the aerobic reactors. The physical dimensions of the biological reactor and the secondary clarifier are reported in Table 2.4.

### 2.7.2 Control strategy

At Sjölanda WWTP, the control strategy acts on the DO concentration in the aerobic compartments, whereas no devices are present for the nitrate concentration in the anoxic zone as no nitrate recirculation is provided in the system. A cascade PI regulator is used for the DO control. The monitored parameter is the DO concentration in both of the three aerobic compartments. The PI controller compares the measured DO in the last tank with the predefined set-point value and acts on the air flow rates of all the three aerobic reactors. The air flow rates injected in the three aerobic reactors are the actual manipulated variables: they are kept at values such that the DO concentration in the last compartment be constant around the set-point value. For simulating this control strategy some changes in the BSM1 are required. In the default BSM1 configuration, the DO control strategy provides only one PI controller in the last aerobic reactor, the manipulated parameter being the oxygen transfer coefficient,  $K_{La}$ . This coefficient describes the rate at which oxygen is transferred from the gaseous to the liquid phase, i.e. from the injected air flow to the activated sludge. The main factor affecting its value is the airflow rate, but in a real plant, many other factors should be considered as: type of aeration facilities (turbines, big or small bubble diffusers,...), wastewater composition, temperature, tank depth. Many relationships have been proposed by several authors in order to determine the transfer coefficient as a function of the airflow rate. Nevertheless, in this study, the  $K_{La}$  was used for the simulation of the real DO control, without

linking the coefficient to the actual airflow rate.

Two more PI controllers were added in the first two aerobic reactors (reactors 3 and 4). For each controller the DO set-point was set at the required value, whereas the maximum reference value for  $K_La$  coefficient was kept at  $360 \text{ d}^{-1}$ , the default BSM1 value. Concerning the nitrate controller, the default regulator was kept in the final simulation layout, but the maximum reference value for the internal nitrate recirculated flow (i.e. the manipulated variable) was set at the very low value of  $1 \text{ m}^3/\text{d}$ . Thereby, the absence of internal recirculation was successfully simulated without changing the model configuration. Equivalently, the whole block responsible for the nitrate control could have been removed.

## 2.8 Influent characterization

Data from the measuring campaign were used for creation of influent files. Either assumptions, mass balances or mathematical formulations were used.

### 2.8.1 COD and nitrogen fractions

On a first attempt, the fractions  $S_I$  and  $X_S$  were calculated according to mathematical formulations found in literature based on soluble and total BOD and COD measurements.  $S_S$  and  $X_I$  were obtained through mass balances of soluble and total COD, respectively. Below, the expressions used for these fractions.

$$S_I = 0.9COD_{out}$$

$$S_S = COD_{sol} - S_I$$

$$X_S = \frac{BOD}{1 - 0.2} - S_S$$

$$X_S = COD_{tot} - X_I - S_S - S_I$$

The expression referring to  $S_I$  estimation refers to the COD exiting the whole plant. It was used the concentration reported as annual average concentration in the annual report published by Sjölanda, in 2011 WWTP ("Sjölanda Avloppsreningsverk Malmö. Miljörapport Enligt Miljöbalken för

år 2011”, available at the plant website: [www.vasyd.se](http://www.vasyd.se)). The assumption on this fraction is that almost all the residual COD, found in the effluent comprises predominantly (90%) the inert soluble fraction, provided the biodegradable fraction has been degraded and the particulate matter has been removed in the secondary settler. Therefore,  $S_S$  fraction is straightforward to assess, through mass balance on inlet soluble COD. The expression for  $X_S$  was proposed by Stowa (1996) and reported by Petersen (2000) and Hellstedt (2005). In this case, the yield coefficient is lower than the one referred to COD and a value of 0.2 is proposed by the above mentioned authors. It has to be noticed that this expression should be applied to the long-term BOD, which was not measured in this study. It was decided to use this expression nonetheless, as a first rough fraction estimation. Finally, the particulate inert fraction was derived from total COD mass balance. The remaining COD fractions were assumed instead:  $X_{B,H}$ ,  $X_{B,A}$  and  $X_P$  were set to zero.

With regard to nitrogen characterization, the concentrations available from the measuring campaign were applied directly to the corresponding fractions, when possible. This was the case for  $S_{NO}$  and  $S_{NH}$  fractions. The total organic-bounded nitrogen was calculated as the total nitrogen subtracted by the above mentioned fractions, whereas its further differentiation in its particulate and soluble components was obtained by applying the same ratio found in the default BSM1 constant influent file.

## 2.8.2 Other variables

The dissolved oxygen concentration,  $S_O$ , was set to zero; the alkalinity and the flow rate to their measured values. Concerning the TSS, it has to be noticed that, in the benchmark model, its value is calculated as the 75% of the influent particulate matter, comprising:  $X_I$ ,  $X_S$ ,  $X_{B,H}$ ,  $X_{B,A}$ , (Copp, 2002). This percentage did not reflect the actual measured TSS concentration, if applied to the particulate matter estimated in this study (based on the assumptions explained above). Therefore, since the actual TSS concentration was available from the measuring campaign, a new specific percentage was calculated and used in all the other cases requiring the TSS estimation.

### 2.8.3 Constant and dry-weather influent files

The constant influent file requires only an average estimation of each variable. Therefore, the flow-weighted average measurements were applied. The values resulted from the first attempt in fractions estimation were used as starting point for model calibration.

The dry-weather file, instead, requires 15-minute data for each variable. Since the measurements from the measuring campaign had a frequency of one or two hours, a mathematical interpolation was done in order to get the required time-step values. It was based on average loads and flowrate. The concentrations were, therefore, calculated and used for the estimation of influent fractions. The obtained daily variation was then replicated for 14 days, as required by the dynamic input file. Thereby, it was assumed that no significant variations in terms of flow rate and loads were detected over the weekend. This assumption is partially supported by historical data, even though they did not fully justify it. Since more analyses would have been required to well model weekly variations, it was decided to keep the same daily variation all over the week. This choice does not represent a strong limitation in terms of qualitative assessment of dynamic behaviour of the plant (which is the main purpose of the study). Nevertheless, including weekly variations could clearly improve simulation quality, giving an influent characterization even closer to the real case.

Constant and dry weather influent files were also created for dynamic evaluation of future scenarios. In this case,  $S_{NO}$  fraction was set to zero, since SBR effluent –responsible for nitrate incoming –is thought to be diverted to other treatments, in case of manamox implementation.

### 2.8.4 Rain-weather and storm-weather files

The same flow rate and loads variations extrapolated from the BSM1 'RAIN-INFLUENT' and 'STORMINFLUENT' files were applied to the average values measured at Sjölanda WWTP. Rain-weather and storm-weather files were created for future scenario simulations only: the  $S_{NO}$  fraction was, therefore, set to zero. The simulated variations of flow rate and some other fractions are shown in Figure 3.2 *a)* and *b)*.



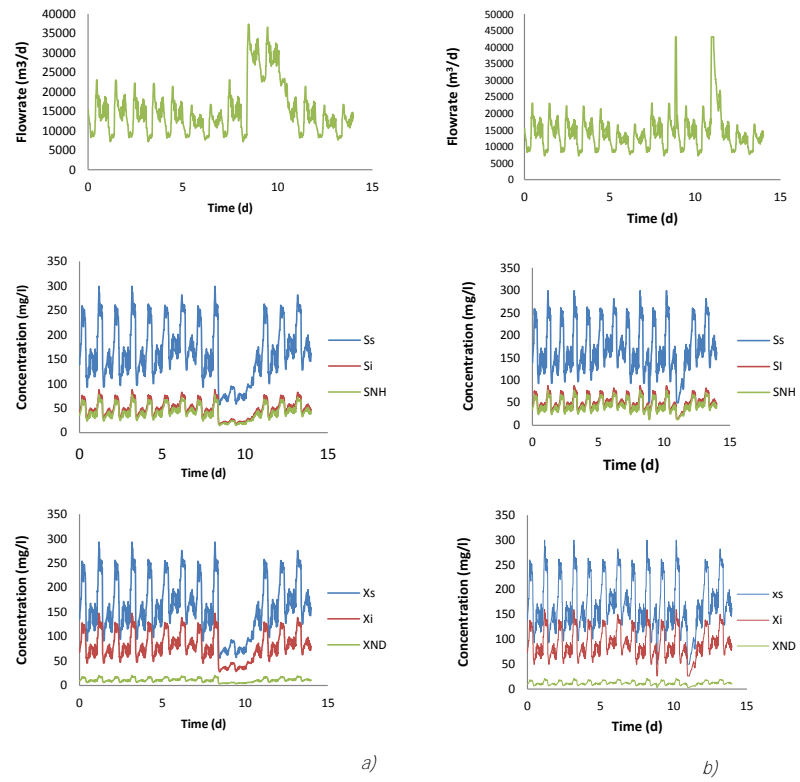


Figure 2.7: Influent flow rate and main concentrations for *a)* rain weather and *b)* storm-weather conditions

### 2.8.5 Model Calibration procedure

Irrespective of the wide application of ASM models over the past decade, a comprehensive calibration procedure has not been defined yet (Petersen et al., 2002; Keskitalo, 2011). Some authors as Petersen et al. (2002) and Langergraber et al. (2004) pointed out this lack of reference and tried to gather the information found in literature. Their work resulted in an overview on the calibration methods used so far, summarized into general guidelines. The calibration procedure used in this study was developed by taking into account the abovementioned guidelines applied to the specific case study of this work.

### 2.8.6 Lab-scale model calibration

The calibration of the three main kinetic parameters ( $\mu_H$ ,  $K_S$ ,  $b_H$ ) was conducted manually. Beyond a qualitative assessment of modelled curve fitting, the root medium square error (between the observed and the calibrated OUR values) was used for fitting quality assessment. Other than the OUR curve, also those on biodegradable substrate,  $S_S$ , and oxygen concentration,  $S_O$ , were taken into account. Different combinations of parameters' values were attempted, within literature ranges. For each of the two experimental batch systems, the initial concentration of active biomass ( $X_{B,H}$ ) was calibrated as well. A two-step calibration procedure was used:

- Step 1: the first part of the OUR curve was taken into account. This part is almost linear and interpolates the average value of endogenous respiration rate. Moreover, it is mainly affected by death coefficient and biomass concentration: as  $b_H$  and  $X_{B,H}$  values increase, the modelled endogenous OUR curve raises upward. The values of  $b_H$  and  $X_{B,H}$  were calibrated together to find their best-fitting combinations;
- Step 2: for each well-fitting combination found in step 1, the specific growth rate was calibrated along with the half-saturation coefficient, in order to model the OUR profile, under exogenous conditions. In particular, the maximum growth rate was selected to fit with the peak of the OUR curve (corresponding to the maximum exogenous respiration rate), whereas the half-saturation coefficient was assessed in order to model either the plateau, if any, and the descendent phase. The modelled  $S_S$  curve was also used while assessing  $K_S$ , in order to model its complete removal as two hour long, as observed during the experiment.

This calibration procedure was implemented for Experiment2 data and the best-fitting parameters were afterwards applied to Experiment1, as an independent data set, used for results validation. Finally, the validated parameters were used as starting values for the full-scale calibration.

### 2.8.7 BSM1 calibration

The constant dry-weather influent files, created using the assumptions on COD and nitrogen fractionations, were used as starting point for the steady and dynamic state calibration, respectively. The sets of kinetic parameters selected from the lab-scale model calibration were used as initial values and therefore calibrated, if needed. All the other model parameters were kept at their default Benchmark values (prior to temperature corrections to meet the actual condition). The average effluent total COD and total suspended solids, TSS, were first used as reference variables for a steady-state calibration assessment. Therefore, influent COD fractions were calibrated, according to their effect on effluent quality. Any change was always consistent with total and soluble COD mass balances. Once the average effluent COD and TSS matched the observed values, the dynamic-state calibration was performed. The percentage of the calibrated COD fractions, with respect to the total COD, were calculated and applied to dynamic influent files. The settling parameters for high-loaded plants, reported in Takács et al. (1991) were applied. Only the Vesilind settling velocity,  $v_0$ , was first estimated through settling tests and further tuned during dynamic calibration. Moreover, the wastage sludge flow rate was also calibrated. Its modifications were supported by an analysis, carried out by processing historical on-line flow rate data on sludge wastage and its effect on basin solids content. The calibrated value was therefore selected in order to be reasonably suitable to the registered data, observed the days before the measuring campaign.



# Chapter 3

## Results and discussion

### 3.1 Measuring campaign results

One-day results are shown in Figure 3.1, while average data are summarised in Tables 3.1, 3.2 and 3.3 .

An anomalous high value was observed in filtered COD effluent concentration, referring to hour 13. As shown in Fig 3.1 *a)* it does not fall within the observed range and was, therefore, excluded from the set of data. Figure 3.1 *c)* clearly shows the intermitted nitrite incoming, due to the intermitted SBR effluent immission in the main stream. The almost flat alkalinity influent and effluent curves almost overlap on each other, confirming that nitrification does not take place.

	Measuring campaign	January-February2013	Autumn2012	Unit
Influent flow rate	13450	15295	13245	m <sup>-3</sup> /d
Wastage sludge flow rate	130	230	235	m <sup>-3</sup> /d
Recirculated sludge flow rate	4350	4680	4348	m <sup>-3</sup> /d
Water temperature	13	13.2	17.9	-C
DO basin 3	0.5	0.6	0.4	mg/l
DO basin 4	0.9	1.0	0.9	mg/l
DO basin 5	1.7	1.8	2.0	mg/l

Table 3.1: Average operational data

If compared with historical data, operational condition measurements are within the average values (Table 3.1 ). Conversely, almost all influent and effluent concentrations have slightly higher values. This discrepancy may be due to the different sampling approaches used in the measuring campaign and routine data collection: the former was time-proportional, the latter flow-

	INFLUENT			EFFLUENT		
	Measuring campaign	January-February 2013*	Autumn 2012	Measuring campaign	January-February 2013	Autumn 2012
	Concentration (mg/l)	Concentration (mg/l)	Concentration (mg/l)	Concentration (mg/l)	Concentration (mg/l)	Concentration (mg/l)
COD	450	290	NA	155	79	NA
COD <sub>filt</sub>	210	169	NA	84.7	65	NA
BOD <sub>7</sub>	174	117.1	122.9	33	41.3	21.4
BOD <sub>7, filt</sub>	78	41.3	45.1	10	7.6	7.5
NH <sub>4</sub> <sup>+</sup> -N	42	29	30.9	37.6	27.9	29.1
NO <sub>2</sub> <sup>-</sup> -N	2.7	2.3	2.8	0.0	0.2	0.2
NO <sub>3</sub> <sup>-</sup> -N	0.0	0.2		0.01	0.7	0.8
N <sub>tot</sub>	63.4	45.4	43.4	46	34.9	30.5
TSS	152	129.7	156.8	66	24	31.9
VSS	118	118.5	123.7	50	25	26.9
Alkalinity (mol/m <sup>3</sup> )	6.44	NA	NA	6.1	79	NA

Table 3.2: Influent and effluent average concentrations on measuring campaign, January and February 2013, and autumn 2012 (\* concentrations of mixed G2 line 1 and line 2 samples)

	Loads		Removal efficiency
	Influent	Effluent	%
	(kg/d)	(kg/d)	
COD	6135	2112	65.6
COD <sub>filt</sub>	2825	1135	59.8
BOD <sub>7</sub>	2340	444	81
BOD <sub>7, filt</sub>	1050	135	87.1
NH <sub>4</sub> <sup>+</sup> -N	565	515	8.8
NO <sub>2</sub> <sup>-</sup> -N	31.3	0	100
N <sub>tot</sub>	853	619	27

Table 3.3: Main pollutants loads and removal efficiencies on measuring campaign day

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porportional. Furthermore, a significantly high concentration is observed for effluent TSS, VSS, and, consequently, total COD. It is believed that it is beacuse of sludge wastage changes. As mentioned above, at Sjölanda WWTP, wastage flow rate is changed manually, by mean of a sluice. Depending on several plant conditions, the transitional period between the sluice movement and the stabilization of solids –in the reactor as in the effluent –may range between one (or even less) day and a couple of days. The day before the measuring campaign, wastage flow rate was changed and this may justify the observed higher effluent solids content.

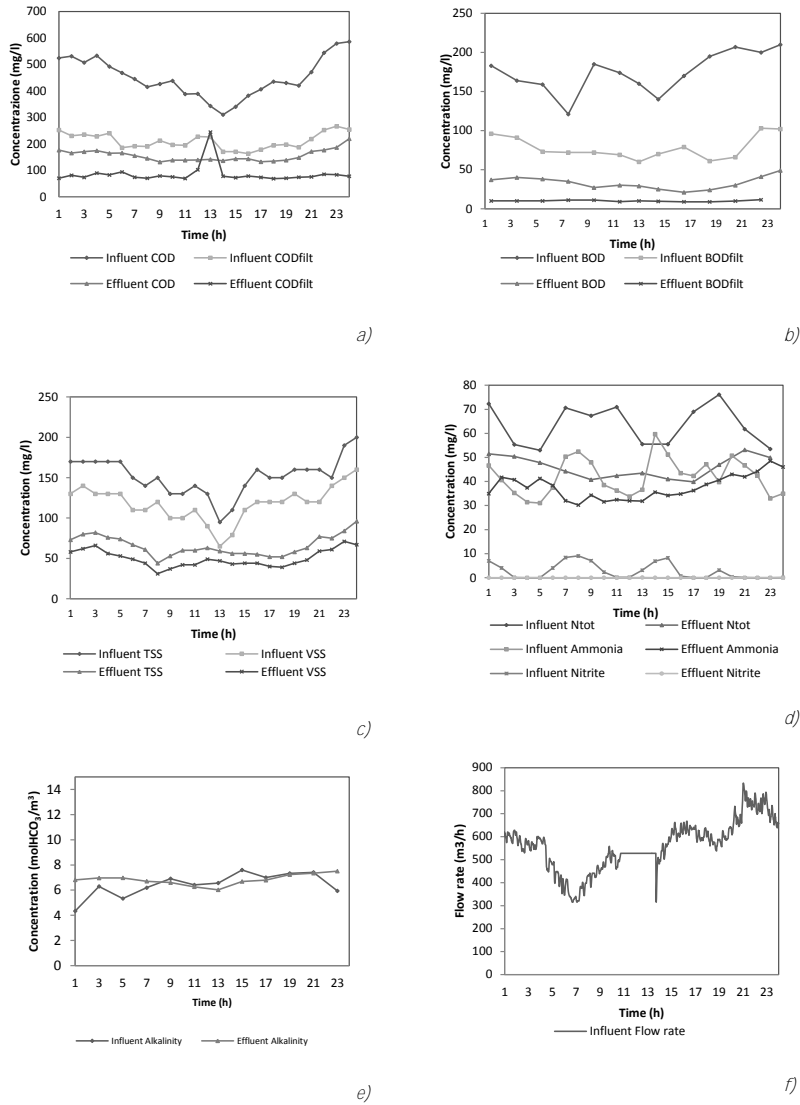


Figure 3.1: Measuring campaign results on: a) COD and COD<sub>filt</sub>; b) BOD and BOD<sub>filt</sub>; c) TSS and VSS; d) Nitrogen fractions; e) Alkalinity; f) Influent flow rate



## 3.2 OUR test results

The results from reactors A of Experiment1 and Experiment2 are showed in Figure 3.2 and Figure 3.3, respectively. The main findings in terms of respirometric rates and other operational measurements are presented in Table 3.4.

	Experiment1		Experiment1		Unit
	A	B	A	B	
Maximum respiration rate, $r_{o,MAX}$	68.5	62.5	66.6	58.0	$\text{mgO}_2\text{l}^{-1}\text{h}^{-1}$
Average endogenous respiration rate, $r_{o,end}$	16.9	15.4	13.7	11.9	$\text{mgO}_2\text{l}^{-1}\text{h}^{-1}$
Maximum exogenous respiration rate, $r_{o,ex,MAX}$	51.6	47.1	52.9	46.2	$\text{mgO}_2\text{l}^{-1}\text{h}^{-1}$
Maximum oxygen concentration	9.35	9.49	8.89	9.07	$\text{mgO}_2\text{l}^{-1}$
Total solids concentration	2.84	2.77	3.3	3.38	$\text{mgSSl}^{-1}$
Volatile suspended solids	2.29	2.25	2.86	2.9	

Table 3.4: Main results from respirometric experiments

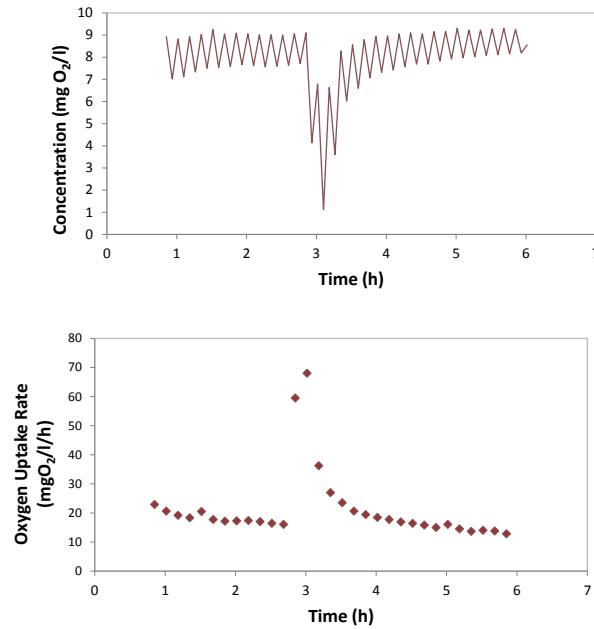


Figure 3.2: Dissolved oxygen and OUR curves, Experiment1.

Both the experiments gave similar findings, in terms of respiration rate values, indicating that the respirometric tests had good reliability. Slight differences are typical when dealing with biological and consequently very

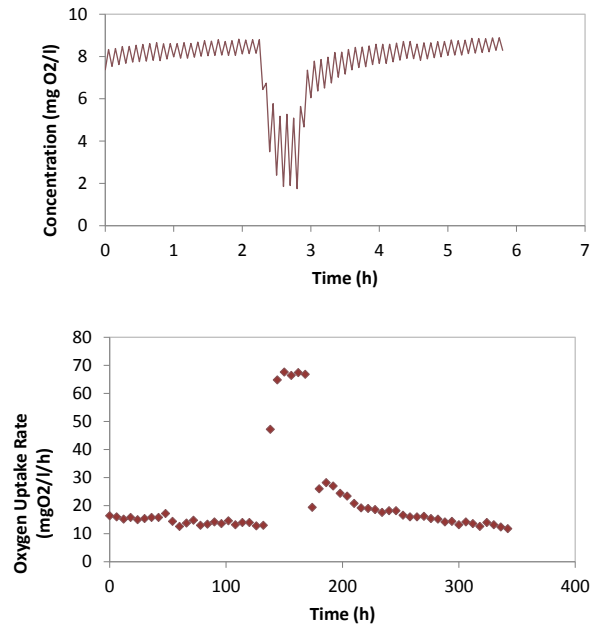


Figure 3.3: Dissolved oxygen and OUR curves, Experiment2.

sensitive systems. The endogenous respiration rate is well defined in the first part of the DO and OUR curves, in both the experiments, as well as the rapid increase in exogenous respiration rate due, following acetate addition. The exogenous respiration rate in Experiment1 reached its maximum value for one point only, whereas the OUR curve referring to Experiment2 showed the typical plateau at the maximum exogenous rate. The reason lies in the differences in the amount of acetate added in the system and in the time interval of intermitted aeration. Thanks to the higher amount of acetate and the shorter time interval in Experiment2, the substrate concentration was high enough for keeping the specific growth rate at its maximum value for a longer period of time. Moreover, an anomalous behaviour was shown in the descendent part of the OUR curve in Experiment2. It might be a result of temporary failures of either the aeration system or the oxygen meter. For these reason, those values were not considered during data processing.

### 3.3 Settling velocity test results

Zone settling curves are shown in Figure 3.7. Sludge TSS concentrations of the different diluted samples resulted in: (i) 0.76 g/l; (ii) 1.56 g/l; (iii) 2.08 g/l and (iv) 2.86 g/l. As expected, the zone settling velocity increased as the sludge concentration decreased. Vesilind settling velocity was assessed to be 126 m/d. This finding was used as starting point in BSM1 calibration.

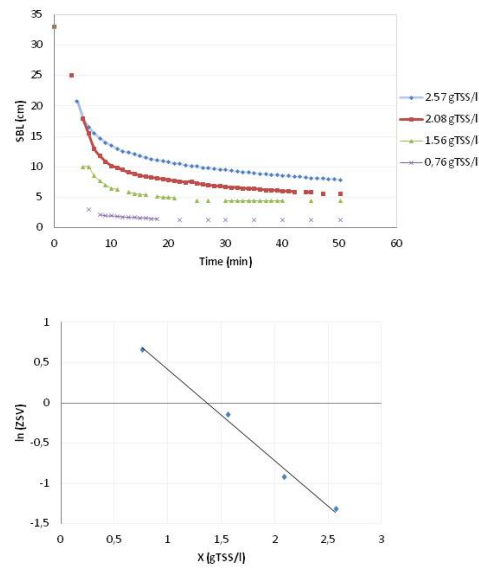


Figure 3.4: Sludge blanket level curves for different solids concentrations and zone settling velocity natural logarithm versus solids concentration

## 3.4 Model calibration results

### 3.4.1 Lab-scale model calibration results

Despite the validation on the independent data set of Experiment1, still some combinations of parameters gave similarly good results. They were eventually implemented in the full-scale model, as starting-point values and further calibrated. Table 3.5 shows two among the best-fitting combinations.

Parameter	Set 1	Set 2	Unit
$b_H$	0.259	0.37	$d_{-1}$
$\mu_H$	2.96	4.4	$d_{-1}$
$K_S$	22	22	mgCOD l <sup>-1</sup>

Table 3.5: Sets of calibrated parameters for the lab-scale model calibration

It has to be noted that the sub-model showed higher sensitivity to parameters as  $\mu_H$  and  $b_H$  and lower for  $K_S$ . Also the assumed biomass initial value affected the modelled OUR significantly. In the selected sets, the half-saturation coefficient was higher than the default BSM1 value of 10 mgCOD/l. Its value was increased in order to make the model describe the OUR plateau, in Experiment2. Provided the other two parameters are kept at the same values, the lower the  $K_S$  the shorter the OUR plateau, up to the single-peak profile. Moreover, since the kinetic parameters are strongly correlated, the effect of one parameter might be compensated by the effect of another one, in case of simultaneous modifications. In set 2, the higher value of  $\mu_H$  is, in fact, balanced by a higher value of  $b_H$ . Figure 3.9 shows model outcomes for Experiment2 data, using the values of the set 2. The validation step had good results for both the sets, listed in Table 3.5, as shown in Figure 3.10, presenting the model outcomes for Experiment1 and set 2 values. With the selected sets of parameters, the model well represented the expected biomass and substrate behaviour: biomass slightly increases after substrate addition and this latter is completely degraded within almost two hours. The OUR and DO curve showed good fitting in both the experiments, only the exogenous phase of Experiment 2 did not match the observed values very well. The modelled oxygen depletion during exogenous phase was, in fact, lower and longer than the observed one. This may be due to model limitations in describing the aeration batch system or to errors in the estimation of the initial biomass.

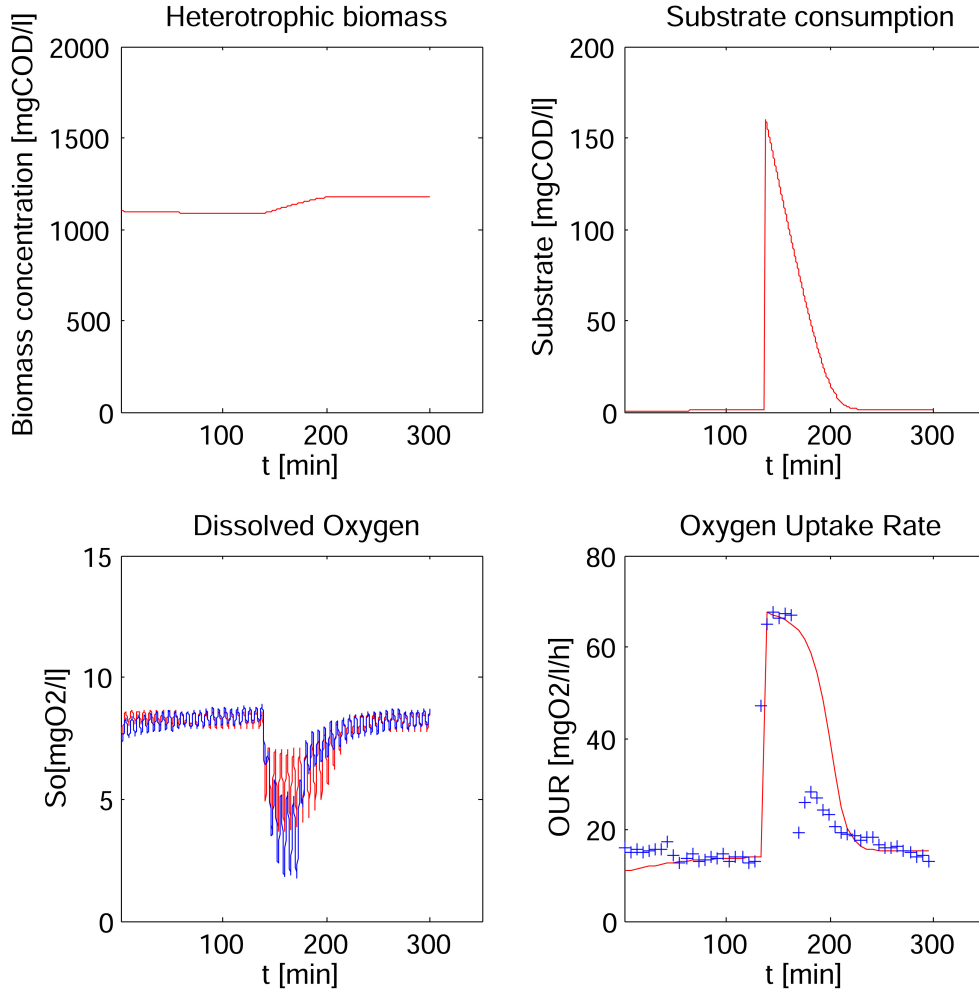


Figure 3.5: Model calibration outcomes using set 2 parameters and Experiment 2 data

### 3.4.2 BSM1 calibration results

The calibrated influent fractions resulted in the values reported in Table 3.6. The initial particulate inorganic COD fraction,  $X_I$ , was so high that its concentration in the settler underflow and in the reactor continuously increased, because of sludge recirculation, up to too high values, not consistent with the measured data. Therefore, it was significantly decreased in favour of the biodegradable particulate fraction,  $X_S$ . Influent autotrophic bacteria concentration was changed and set at 0.5 mgCOD/l for a better fit with effluent nitrogen. All the changes were consistent with mass balances.

Both of the sets resulted from the lab-scale model calibration were at-

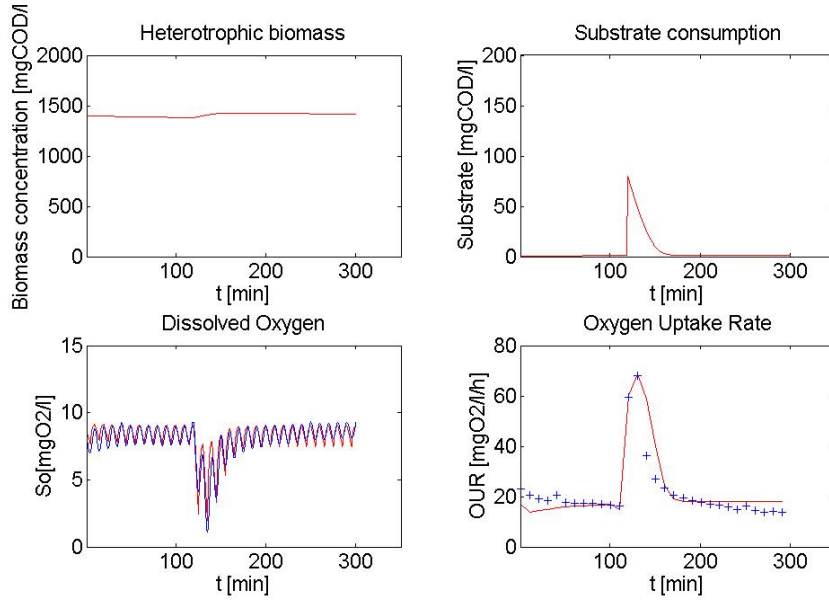


Figure 3.6: Model validation outcomes using set 2 parameters and Experiment 1 data

State Variable	Starting value	Calibrated value	Unit
$S_I$	47.7	47.7	$\text{gCOD m}^{-3}$
$S_S$	165.35	165.35	$\text{gCOD m}^{-3}$
$X_I$	166.16	80	$\text{gCOD m}^{-3}$
$X_S$	76.96	162.62	$\text{gCOD m}^{-3}$
$X_{B,H}$	0	0	$\text{gCOD m}^{-3}$
$X_{B,A}$	0	0.5	$\text{gCOD m}^{-3}$
$X_P$	0	0	$\text{gCOD m}^{-3}$
$S_O$	0	0	$\text{g(-COD) m}^{-3}$
$S_{NO}$	2.33	2.33	$\text{gN m}^{-3}$
$S_{NH}$	41.72	41.72	$\text{gN m}^{-3}$
$S_{ND}$	7.36	7.36	$\text{gN m}^{-3}$
$X_{ND}$	11.21	11.21	$\text{gN m}^{-3}$
$S_{ALK}$	6.4	6.4	$\text{mol l}^{-1}$
TSS	153.17	153.17	$\text{gSS m}^{-3}$
Q	13448.8	13448.8	$\text{m}^{-3}\text{d}^{-1}$

Table 3.6: Initial and calibrated values for steady-state influent fractions

tempted. The set 2 was finally selected. Only  $K_S$  was further increased up to a value of 40 mgCOD/l. This change was necessary in order to model the pretty high concentration of soluble BOD and COD of the real effluent of G2 line 2. Parameters were modified in order to achieve the plant characteristics listed in Table 3.7.

	Reference value	Unit
TSS in the reactor	ca 2200	$\text{g m}^{-3}$
TSS in the settler underflow	9000-10000	$\text{g m}^{-3}$
Sludge age	1.2-1.3	d
F/M ratio	ca 1	(kgBOD/d)/kgTSS
Aerobic F/M ratio	0.77	(kgBOD/d)/kgTSS

Table 3.7: Reference values for plant conditions

The final set of ASM1 parameters is reported in Table 3.8. Fig 3.5 shows the dynamic reponse of the calibrated model and its match with the measured data.

Parameter	Unit	Value
$\mu_H$	$\text{d}^{-1}$	4.4
$K_S$	$\text{gCODm}^{-3}$	40
$K_{O,H}$	$\text{g(-COD)m}^{-3}$	0.4
$K_{NO}$	$\text{g(NO}_3\text{-N)m}^{-3}$	0.5
$b_H$	$\text{d}^{-1}$	0.3
$\eta_g$	—	0.8
$\eta_h$	—	0.8
$k_h$	$\text{g slowly biodegradable COD. (g cell COD. d)}^{-1}$	3.0
$K_X$	$\text{g slowly biodegradable COD. (g cell COD)}^{-1}$	0.1
$\mu_A$	$\text{d}^{-1}$	0.5
$K_{NH}$	$\text{g(NH-N)m}^{-3}$	1
$b_A$	$\text{d}^{-1}$	0.05
$K_{O,A}$	$\text{g(-COD)m}^{-3}$	0.4
$k_a$	$\text{m}^3(\text{gCOD.d)}^{-1}$	0.05

Table 3.8: Calibrated values for ASM1 parameters, at 14 °C

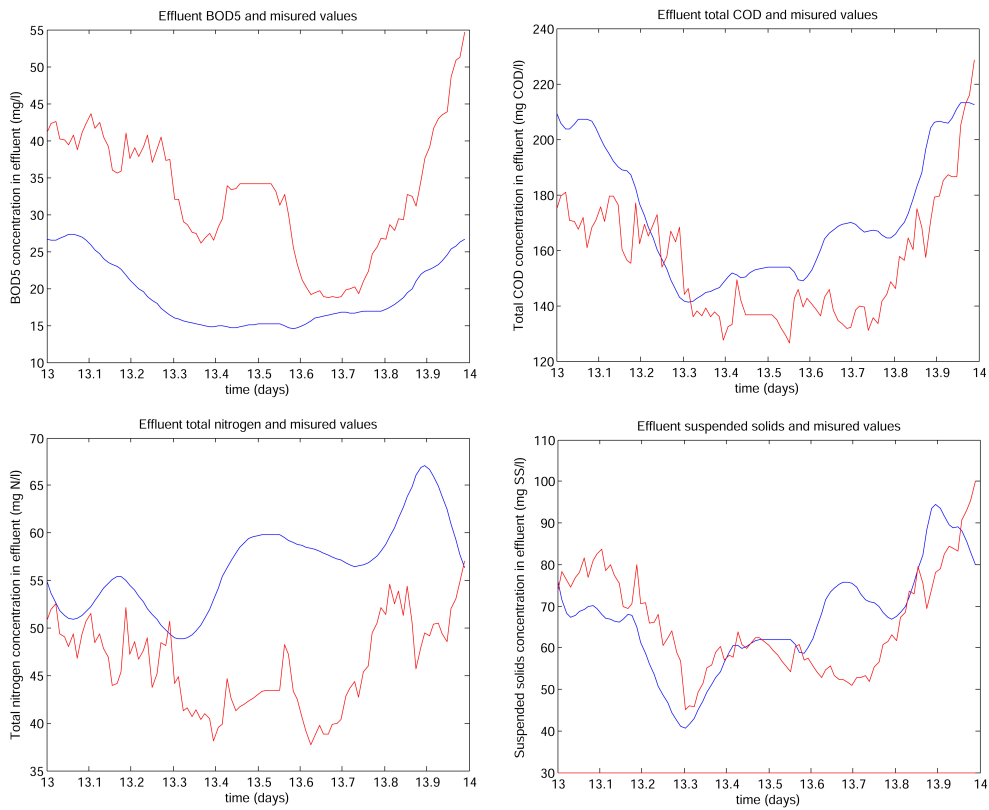


Figure 3.7: Modelled dynamic response and observed data



## 3.5 Results of the scenarios and discussion

### 3.5.1 Comparison of scenarios

The minimum sludge retention time was calculated at different temperature conditions according to either the expressions proposed by Salem et al., (2002) and those by Ekama and Wentzel (2008), using the default BSM1 values as parameters reference values at 15 °C. A concentration of 34 mgN/l was considered as  $S_{NH}$  concentration in activated sludge inlet, after mixing with recycle sludge flow. As shown in Table 3.9, the theoretically obtained values are higher than the current and modelled sludge ages (1-1.3 d) in all the simulated temperature conditions as well as for both the approaches. In this study the approach proposed by Ekama and Wentzel (2008) was used.

	14°C		20°C		6°C		Unit
	Salem, 2003	Ekama, 2008	Salem, 2004	Ekama, 2009	Salem, 2005	Ekama, 2010	
$b_A$	0.083	0.134	0.15	0.268	0.038	0.053	d <sup>-1</sup>
$\mu_{A,MAX}$	0.444	0.445	0.8	0.893	0.203	0.176	d <sup>-1</sup>
$K_{NH}$	1	0.89	1	1.786	1	0.352	mgN-NH <sub>3</sub> /l
SRT <sub>min</sub>	2.871	3.345	1.595	1.668	6.288	7.789	d

Table 3.9: Minimum SRT for nitrification at different temperatures, according to Salem (2003) and Ekama, (2008)

Figure 3.8 shows outcomes for the selected variables from the different scenarios.

Scenario 2 and scenario 3 showed the best performances concerning almost all the variables and under all influent conditions. In both scenarios,  $S_S$  concentration was significantly lower when compared with scenario 1 and 1-b, where a surprisingly high concentration was modelled under rain event. It may have been caused by model failure under those specific conditions. Scenario 2 and 3 seemed to be more stable even under high flow and loads conditions.

$S_{NH}$  fraction remained lower than 1 mgN/l in all the scenarios and in all the cases, with the exception of two peak values under high temperature condition in scenario 2 and 3. This finding is consistent with what expected, as sludge ages never reach the required minimum value. In a specular way,  $S_{NH}$  and total nitrogen keep almost constant values in all the scenarios and conditions. Scenario 2 and 3 show, again, the lowest values either for bCOD/N and bCOD/N ratio.

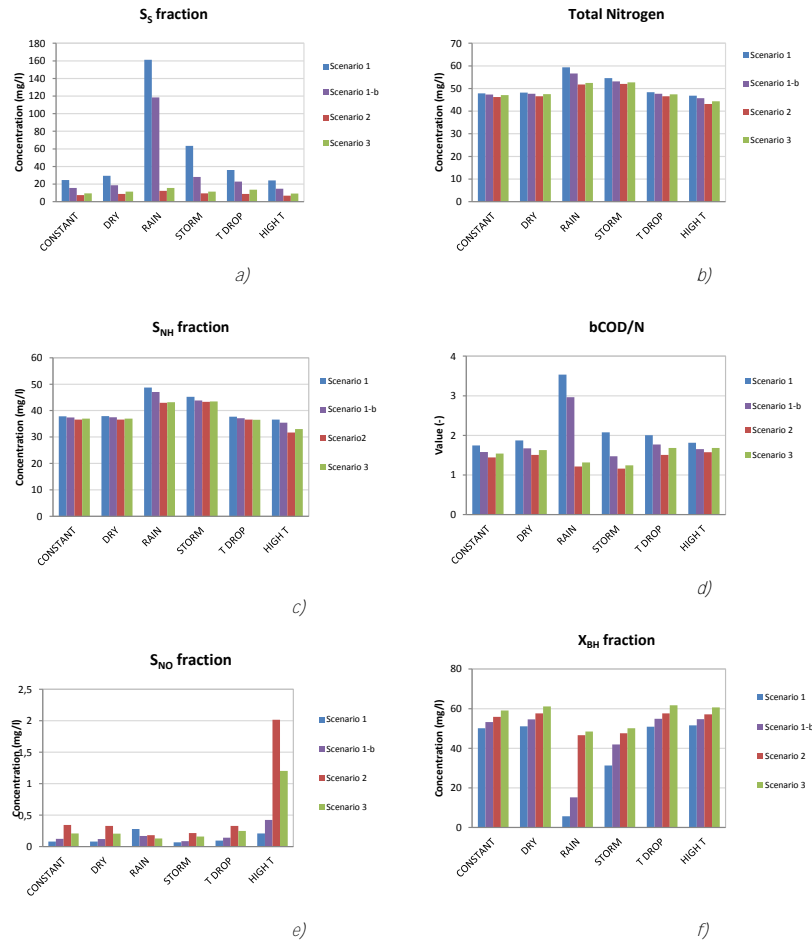


Figure 3.8: Outcomes and comparison of the main variables

Focusing on Scenario 2 and 3 only, scenario 2 seems to guarantee the lowest and the most stable values of  $S_5$  concentrations, whereas scenario 3 gives the lowest  $S_{NH}$  concentrations in all the cases.

Table 3.10 compares the main variables average concentration under dry-weather conditions and the standard deviation for the selected scenarios. De-

spite the comparable good organic removal efficiency,  $S_S$ , total and biodegradable COD concentration are higher in scenario 3, and also their fluctuations, over the cases, are wider. This is probably due to the higher aerobic volume in the fully-aerated five-compartment scenario, capable to bear different organic loads without any significant effect on effluent quality. Scenario3 shows instead a slight lower  $S_{NO}$  concentration. This might result from the low sludge age, always far below the minimum value for nitrification. Nevertheless the difference among the two scenarios is not relevant if dry-weather results are considered and special attention should be paid on scenario 2 in case of prolonged high temperature conditions.

It has to be noted that a slight difference in  $S_S$  effluent concentration would increase  $S_S$  load entering the Manammox reactor. This could have a direct effect in favour of heterotrophic biomass. As a result of its high yield coefficient ( $Y_H = 0.67$ ) sludge production might increase significantly, if compared with the average production due to autotrophic biomass only. In such a case, sludge age should be corrected by increasing wastage flow rate and, at the same time, it might be expected a higher risk of Anammox wash out as well as competition.

	$S_S$		COD		bCOD		$S_{NO}$	
	mg/l	s.d.	mg/l	s.d.	mg/l	s.d.	mg/l	s.d.
Scenario 2	8.8	1.86	146	2.83	70.2	4	0.33	1.2
Scenario 3	11.4	2.44	155	4.22	77.3	5.3	0.21	1.1

Table 3.10: Minimum SRT for nitrification at different temperatures, according to Salem, 2003 and Ekama, 2008

Regarding bCOD/N ratio, scenario 2 shows again the best outcomes, i.e. the lowest values.

On the other hand, higher aerobic volume results into two main disadvantages if compared with scenario 3:

- average sludge age is higher, increasing the risk of nitrification under high temperature conditions;
- new aeration systems and DO controllers are required for the first two basins, resulting in additional investment and operational costs.

In order to select the best among scenario 2 and 3, the idea was to consider possible effects that the different effluent characteristics might have on Manamox process. Again, it should be underlined that no reference values in terms of COD nor nitrate concentrations are defined in the pilot-plant project yet. Thereby, all the considerations resulted by reasoning on the known characteristics of the Deammonification process and full-scale as pilot-scale experiences reported in literature. In light of this, it was believed that the minimization of readily biodegradable COD in the effluent was the most important achievement, provided nitrate concentration remained at low concentrations ( $< 2$  mgN/l). It is therefore believed that the advantage of stable COD-poor influent would overbalance any additional cost. The risky situation of nitrification raising might be handled by properly manipulating the oxygen concentration, i.e. decreasing the basin DO set-point. Thus, scenario 2 was selected as the most suitable for Manamox implementation.

### 3.5.2 Sludge age and DO effects on nitrification

Figure 3.9 shows the results of steady-state DO analysis on scenario 2. The main effect of DO changes was observed on  $S_S$  fraction (Figure 3.9 a): either at 14 °C and 20°C, it gradually decreased for DO concentrations up to almost 1-1.5 mg/l and it almost leveled off at higher DO values. This finding confirms that, at DO higher than ca 1.5 mg/l, oxygen is not limiting the process anymore.  $S_{NO}$  and  $S_{NH}$  showed instead slight changes only: at temperature of 14 °C,  $S_{NO}$  showed a slight raise, always remaining below 1 mg/l even at high DO concentrations as 2.5 mgO<sub>2</sub>/l, whereas at high temperature, it increased up to a maximum of 4 mg/l at the highest DO value of 2.5 mg/l. Thereby, complete nitrification did not seem to take place, probably due to the low sludge age. Consistently, ammonia concentration behaved in an opposite way than  $S_{NO}$ , in both the temperature conditions. COD/N and bCOD/N ratios did not show any significant change for increasing DO concentration neither at 14°C nor at 20°C.

Figure 3.10 shows one-day plant dynamic behaviour for increasing DO concentration at high temperature conditions. Dynamic responses are plotted together with a *reference* condition. i.e. the dry-weather response of the calibrated plant at the measuring-campaign dry influent.

Steady-state and dynamic-state results on sludge age analysis are shown

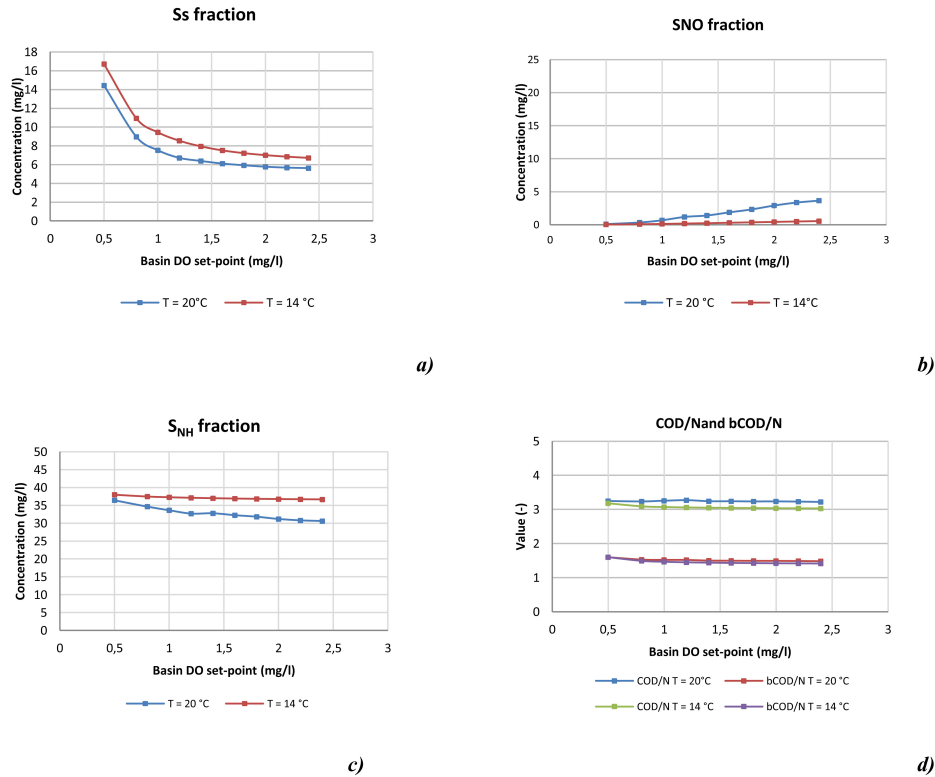


Figure 3.9: DO effect on: a)  $S_S$ ; b)  $S_{NO}$ ; c)  $S_{NH}$ ; d) COD/N and bCOD/N ratios. Steady state simulations on Scenario 2

in Figure 3.11 and Figure 3.12, respectively. Differently from the DO analysis, sludge age changes affected  $S_{NO}$  and  $S_{NH}$  mainly. At 20 °C, the model was able to simulate nitrification appearance. The modelled sludge age value required for nitrification was found to be slightly higher than those estimated and reported in Table 3.9. The sharp raise in  $S_{NH}$  concentration (after an initial moderate increase) and the consequent drop of  $S_{NH}$  confirm (may be a confirmation of) the switching role of minimum sludge retention time on nitrification. On the contrary, no significant effects are shown on  $S_S$  concentration, being its removal already at a high level at low sludge retention time values. COD/N and bCOD/N ratios are almost stable in average temperature conditions, whereas at high temperature, their value increase as nitrification starts. Sludge age higher than 2.7 days were not simulated, since it they would not fit with the operational condition of the high loaded activated sludge plant. Therefore, the theoretical minimum sludge age at 14°C of 3.34 days (Table 3.10) was not simulated and, in fact, results confirm that nitrification does not take place.

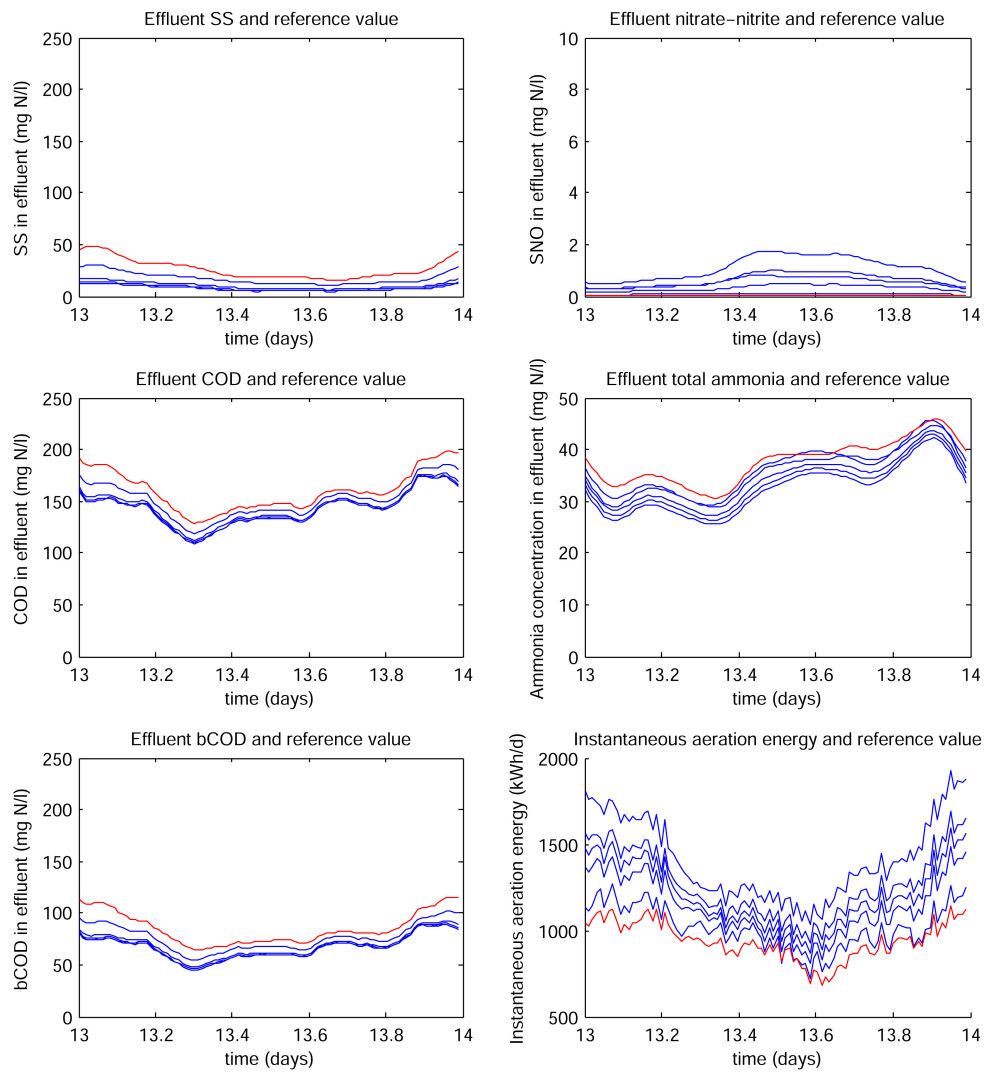
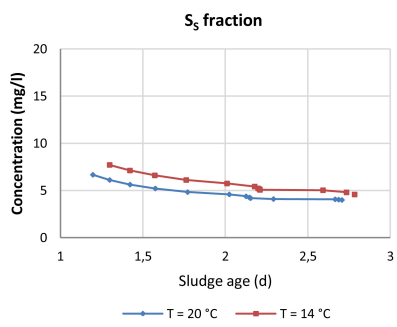
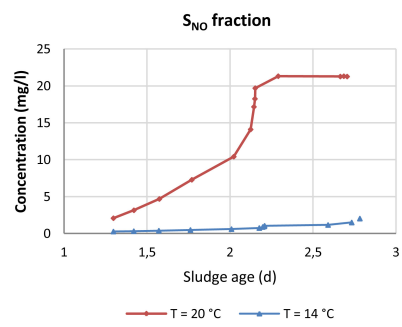


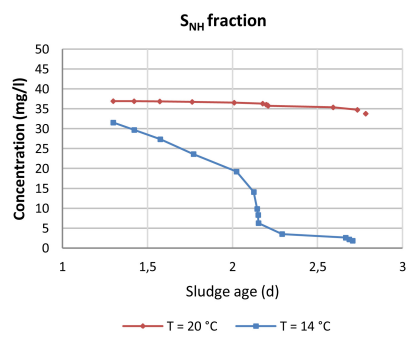
Figure 3.10: DO effect on dynamic plant response, Scenario 2 at 20 °C. Red line refers to reference conditions: dynamic behaviour on measuring campaign.



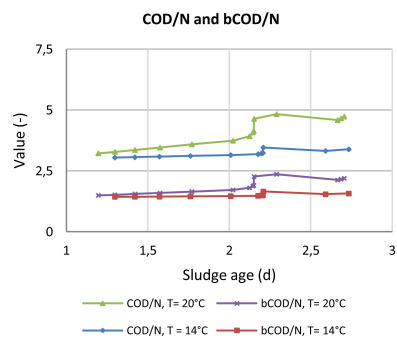
a)



b)



c)



d)

Figure 3.11: Sludge age effect on: a)  $S_S$ ; b)  $S_{NO}$ ; c)  $S_{NH}$ ; d) COD/N and bCOD/N ratios. Steady state simulations on Scenario 2

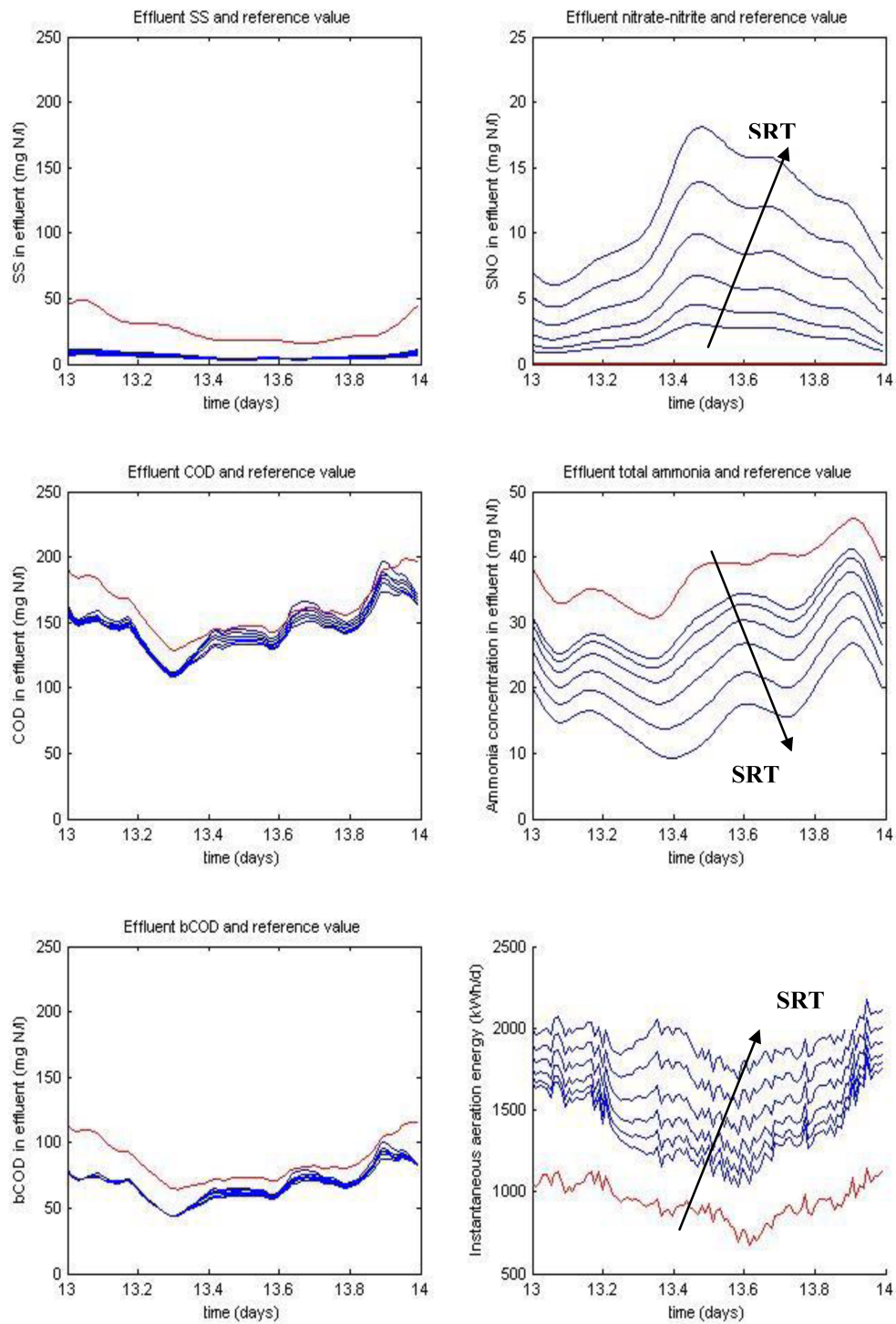


Figure 3.12: Sludge age effect on: a)  $S_S$ ; b)  $S_{NO}$ ; c)  $S_{NH}$ ; d) COD/N and bCOD/N ratios. Steady state simulations on Scenario 2



## Conclusions

This work was developed within the future perspective of deammonification implementation in the mainstream (Manammox) at Sjöulnda WWTP, in Malmö, Sweden. The plant is currently operating high-load activated sludge treatment for organic removal and Nitrification/Denitrification process in subsequent separate tanks. In case the innovative process is implemented, it will replace the current nitrogen removal treatment and the high loaded activated sludge plant will be intended/meant as a pre-treatment. Specifically, the innovative autotrophic nitrogen removal process requires an influent stream poor in biodegradable COD and rich in ammonia nitrogen. The high-load activated sludge plant is therefore required to maximise organic removal and prevent nitrification. The current activated sludge layout comprises two anoxic and three aerobic tanks. Three main scenarios were simulated by mean of the Benchmark Simulation Model number 1 for assessing the effect of aerobic volume changes on effluent quality. In particular: the first scenario simulated the current configuration, providing two anoxic and three aerobic compartments; the second a fully aerated five-compartment configuration; the third a fully-aerated three-compartment one; a sub-scenario 1-b, differing for the simulated DO set-point in aerated basins, was included as well. Scenarios 1-b, 2 and 3 had the same DO set-point of  $1.5 \text{ mgO}_2/\text{l}$ , whereas scenario1 kept the current DO concentrations of  $0.5 \text{ mgO}_2/\text{l}$ ,  $0.9 \text{ mgO}_2/\text{l}$  and  $1.7 \text{ mgO}_2/\text{l}$ , in the three aerobic compartments.

The calibration of the model was based on the results of a measuring campaign and of laboratory tests. The one-day measuring campaign provided an accurate wastewater characterization required for the estimation of model fractions. Respirometric tests were carried out in order to estimate the main kinetic parameters, referring to heterotrophic biomass activity. A lab-scale model was built in order to simulate Oxygen Uptake Rate experiments. The

calibration of the sub-model showed that more than one parameter values combinations gave similarly good results. This might be a result of their high correlation as well as of model limitations. The calibrated sets of parameters were then applied to the full-scale model (the BSM1) and further calibrated. Moreover, zone settling velocity tests were carried out to estimate the Vesilind settling velocity. The test showed good settling properties of the activated sludge and the zone settling curves were shaped as expected. Nonetheless, the test apparatus was slightly different from those proposed in literature and this might have affected the accuracy of the test.

The dynamic response of the different scenarios was assessed at average and critical weather conditions and their performances compared. Some effluent variables were selected as the most relevant for the desired characteristics: (i) the biodegradable COD concentration and, specifically, the soluble biodegradable model fraction,  $S_S$ ; (ii) the total and ammonium nitrogen and (iii) the COD/N and bCOD/N ratios. The minimum sludge retention time for nitrification was calculated according to formulations found in literature. All scenarios had nitrification sludge ages which were lower than the minimum required value, suggesting that all of them could be feasible for the required task of preventing nitrification. Relevant differences were observed, instead, in carbon removal efficiencies. The first scenario and its sub-scenario 1-b showed the worst performances resulting in the highest dry-weather  $S_S$  concentration of 29.4 and 18 mgCOD/l, respectively, with an anomalous behaviour under rain event. Scenario 2 and scenario 3 showed significantly lower dry-weather  $S_S$  concentrations of 8.8 and 11.4 mgCOD/l respectively. In particular, scenario 2 –simulating a five-aerated-compartment configuration –gave the lowest  $S_S$  concentration in all the simulated cases. Indeed, it is believed that, thanks to the high aerobic volume, such a plant could achieve the desired organic removal efficiencies, even in case of high load and high flow conditions as well as in case of extreme low temperature, being able to give stable effluent characteristics. Conversely, an increase of nitrate effluent content was showed under high temperature conditions, up to a value of almost 2 mgN/l. On the other hand, scenario 3 - simulating a three-aerobic-compartment layout - gave the lowest value of effluent  $S_{NO}$  in all the cases. The results suggest that this might be due to its lower sludge age which seems to suppress nitrification even at summer temperature. In any case, the dry-weather  $S_{NO}$  concentrations are low in both the scenarios: 0.33 and 0.15 mgN/l in scenario 2 and 3, respectively.

Considering the high sensitivity of the Deammonification process to sludge age and sludge production, it is believed that even a slight increase in readily biodegradable COD, rather than in nitrate, may represent a more critical condition for Manammox process. This might affect the sludge age, increasing the risk of Anammox wash out and likely affecting their competition with the coexisting bacteria.

For these reasons, scenario 2 was selected as the most suitable to be a pre-treatment for Manammox implementation at Sjöulunda WWTP, being able to provide a stable carbon-poor influent for the Manammox process, consistently with what desired. A further analysis was carried out in order to assess the effect of dissolved oxygen concentration and sludge age on nitrification raising. If the current sludge age is kept, the results suggest that DO concentration have a slight effect of the biodegradable COD fraction. Especially, concentrations higher than the set-point simulated value of 1.5 mgO<sub>2</sub>/l could be experienced in order to further decrease effluent  $S_S$  concentration.

According with model outcomes, sludge age had a more relevant effect on effluent quality. As a general effect, the higher the sludge age, the lower the effluent  $S_S$ . Under average temperature conditions, slight but not significant increases of  $S_{NO}$  were observed as the modelled sludge ages increased. Higher concentrations up to 20 mgN/l were showed, instead, under high temperatures. The findings profiled a sharp increase of effluent  $S_{NO}$  at sludge ages higher than 2 days, suggesting that the plant may experience nitrification raising during summer season. Nevertheless, the model outcomes suggest that this effect could be limited by controlling the DO concentration.

In conclusion, this work successfully integrates routine monitoring data,

measurements from water samples and results from standard tests collected ad-hoc by the author, as well as mathematical modelling of the physical, chemical and biological processes taking place in Sjöulunda wastewater treatment plant. The goal was to simulate the feasibility and sustainability of alternative plant management solutions.

According with the modelled outcomes, the fully aerated five-compartment layout appeared to be the best management scenario to achieve the goal of providing a proper influent for the Manammox process. In fact, it showed to be capable of giving a stable carbon-poor as well as ammonia-rich effluent, consistently with the desired characteristics. Despite the investment and op-

erational costs that would be required by the novel plant configuration, it is believed that they will be overbalanced by the advantages brought by such a solution. This work clearly demonstrates the usefulness of mathematical modelling and scenario-based approaches for the management of wastewater treatment facilities, especially in combination with case-specific experimental investigations.

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# Bibliography

- Alex J., Magdeburg V., Benedetti L., Copp J., Gernary K.V., Primodal, Jeppsson U., Nopens, Pons M.N., Rosen C., Steyer J.P., Vanrolleghem P. (2008). Benchmark Simulation Model no. 1 (BSM1) *IWA Taskgroup on Benchmarking of Control Strategies for WWTPs*.
- Andersson B., Asoegren H., Parker D.S., Lutz M. P., (1994). High rate nitrifying trickling filters. *Water Science Technologies Vol. 29, No. 10*, 47–52.
- Anthonisen, A.C., Loehr, R.C., Prakasam, T.B.S. and Srinath, E.G. (1976). Inhibition of nitrification by ammonia and nitrous acid. *Journal of Water Pollution Control Federation 48*, 835–852. Quoted in: Van Hulle S.W.H., (2005).
- Bertino A. (2010). Study on one-stage partial nitrification-anammox process in moving bed biofilm reactors: a sustainable nitrogen removal. *Doctoral thesis*/, Politechnical University of Turin
- Catunda P.F.C., Van Haandel A.C., (1992) Activated sludge settling Part 1: Experimental determination of activated sludge settleability. *Water SA/ Vol. 18. No. 3*, 165–172.
- Ciabini R. (2005). Combinazioni di tecniche respirometriche e titrimetriche per la valutazione del potenziale inibente di un refluo tessile. *Master Thesis*, University of Florence, Italy.
- Cui F. (2012). Cold CANON: Anammox at low temperatures. *Master Thesis*, University of Delft, The Netherlands.
- Ekama G. and Wentzel M. C., (2008). Nitrogen Removal. Biological wastewater treatment: principles, modelling and design. *IWA Publishing*, London, UK.
- Guisasola A., Jubany I., A Baeza J., Carrera J., Lafuente J., (2005). Respirometric estimation of the oxygen affinity constants for biological ammonium and nitrite oxidation, *Journal of Chemical Technology and Biotechnology 80*, 388–396.

- Guven D., Dapena A., Kartal B., Schmist C., Maas B., van de Pas-Schoonen K., Sozen S., Mendez R., Op den Camp H., Jetten M., Strous M., Schmidt I. (2004). Propionate Oxidation by and Methanol Inhibition of Anaerobic Ammonium-Oxidizing Bacteria. *Applied and Environmental Microbiology Vol. 71, No. 2*, 1066–1071.
- Hellstedt C. (2005). Calibration of a dynamic model for the activated sludge process at Henriksdal wastewater treatment plant. *Master thesis*, Uppsala University, Sweden.
- Hagman M., La Cour Jansen J., (2007). Oxygen uptake rate measurements for application at wastewater treatment plants. *Vatten 63*, 131–138.
- Hunik J. H., Tramper J., Wijffels R.H., (1994). A strategy to scale up nitrification processes with immobilized cells of nitrosomonas europaea and nitrobacter gilis. *Bioprocess Engineering 80*, 73–82.
- Jeppsson U. (1996). Modelling aspects of wastewater treatment processes. *Doctoral thesis*. Lund Institute of Technology, Sweden.
- Jetten M.S.M., Strous M., Van de Pas-Schoonen K.T., Shalk J., Van Dongen U.G.J.M., Van de Graaf A.A., Logemann S., Muyzer G., Van Loosdrecht M.C.M., Kuenen J.G., (1999). The anaerobic oxidation of ammonium. *FEMS Microbiology Reviews/ 22*, 421–437.
- Jetten M.S.M., Wagner M., Fuerst J., Van Loosdrecht M., Kuenen G., Strous M. (2001). Microbiology and application of the anaerobic ammonium oxidation ("anammox") process. *Current Opinion in Biotechnology 12*, 283–288.
- Keskitalo J., la Cour Jansen J. and LeiviskäK, (2010). Calibration and validation of a modified ASM1 using long-term simulation of a full-scale pulp mill wastewater treatment plant *EnvironmentalTechnology 31*, 555 – 566.
- Langergraber G., Rieger L., Winkler S., Alex J., Wiese J., Owerdieck C., Ahnert M., Simon J., and Maurer M. (2004). A guideline for simulation studies of wastewater treatment plants. *Water Science and Technology 50*, 131-138.
- Marsman E., Roeleveld P.J., Rensink J. H., (1997) High Nutrient removal in the three-sludge sewage treatment system: results and economic evaluation. *Water Science Technology Vol. 35, No. 10*, 129–136.
- Mulder A., Van de Graaf A.A., Robertson L.A., Kuenen J.G., (1995) Anaerobic ammonium oxidation discovering in denitrifying fluidized bed reactor. *FEMS Microbiology Ecology 16*, 177–184.



- Oshiki M., Shimokawa M., Fujii N., Satoh H., Okabe S., (2011). Physiological characteristics of the anaerobic ammonium-oxidizing bacterium "Candidatus Brocadia sinica". *Microbiology* 157, 1706–1713.
- Petersen B. (2000). Calibration, identifiability and optimal experimental design of activated sludge models. *Doctoral thesis*, University of Gent, Belgium.
- Petersen B., Gernaey K., Henze M. and Vanrolleghem P.A. (2002). Evaluation of an ASM1 model calibration procedure on a municipal-industrial wastewater treatment plant. *Journal of Hydroinformatics* 04.1, 15–38.
- S. Salem, D.H.J.G. Berends, J.J. Heijnen, M.C.M. Van Loosdrecht (200). Bio-augmentation by nitrification with return sludge. *Water Research* 37, 1794–1804.
- Spanjers H. (1993). Respirometry in Activated Sludge. *Doctoral Thesis*, Wageningen Agricultural University, The Netherlands.
- Stensel H. D., (2006). Sidestream treatment for nitrogen removal. *Technical report*.
- Strous M., Van Gerven E., Kuenen J.G., Jetten M.S.M., (1997). Effects of Aerobic and Microaerobic Conditions on Anaerobic Ammonium-Oxidizing (Anammox) Sludge. *Applied and Environmental Microbiology* 63, No. 6, 2446–2448.
- Strous M., Kuenen J.G., Jetten M.S.M., (1999). Key Physiology of Anaerobic Ammonium Oxidation. *Applied and Environmental Microbiology* 65, No. 7, 3248–3250.
- Suzuki, I., Dular, U. and Kwok, S.C. (1974). Ammonia or ammonium ion as substrate for oxidation by *Nitrosomonas europaea* cells and extracts. *Journal of Bacteriology* 120, 556–558. Quoted in: Van Hulle S.W.H., (2005).
- Takács I., Patry G. G., Nolasco D., (1991). A dynamic Model of the clarification-thickening process. *Water Resources Vol. 25, N., 10*, 1263–1271.
- Van Hulle S.W.H., Volcke E.I.P., Lopez Teruel J., Van Loosdrecht M.C.M., P.A. Vanrolleghem, (2007). Influence of temperature and pH on the kinetics of the Sharon nitrification process. *Journal of Chemical Technology and Biotechnology* 82, 471–480.
- Van Hulle S.W.H., (2005). Modelling, simulation and optimization of autotrophic nitrogen removal processes. *Doctoral Thesis*, University of Gent, Belgium.

- Vanderhasselt A., Van Rolleghe P.A. (1999) Estimation of sludge sedimentation parameters from single batchsettling curves. *Water Resources Vol.34, No. 2*, 395–406.
- Vanrolleghe P.A. (2002). Principles of Respirometry in Activated Sludge Wastewater Treatment. *Technical Report*, University of Gent, Belgium.
- Winkler M.K.H., Yang J., Kleerebezem R., Plaza E., Hultman B., Van Loosdrecht M.C.M., (2012). Nitrate reduction by organotrophic Anammox bacteria in a nitrification/anammox granular sludge and a moving bed biofilm reactor. *Biore-source Technology 114*, 217–223.
- Winkler M.K.H., Kleerebezem R., Van Loosdrecht M.C.M., (2012). Integration of anammox into the aerobic granular sludge process for main stream wastewater treatment at ambient temperatures. *Water Research 46*, 136–144.
- Winkler M.K.H., Kleerebezem R., Van Loosdrecht M.C.M., (2012). Integration of anammox into the aerobic granular sludge process for main stream wastewater treatment at ambient temperatures. *Water Research 46*, 136–144.