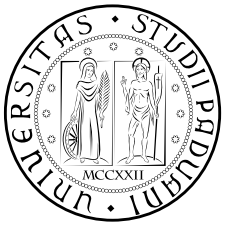

University of Padova

DEPARTMENT OF INDUSTRIAL ENGINEERING

Master Thesis in Environmental Engineering

Integrating experimental analyses and a dynamic model for enhancing the energy efficiency of a high-loaded activated sludge plant

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We never know the worth of water till the well is dry
Thomas Fuller

Contents

Abstract

This thesis presents a research concerning the energy efficiency optimization of a high-loaded activated sludge treatment plant coupled with anaerobic digestion. The study, implemented on Sjölunda wastewater treatment plant in Malmö, Sweden, started by asking whether there was room for energy efficiency improvement in the current process operating conditions.

Efforts were directed towards enhancing energy and economical balance at a minimum initial investment cost, while assuring wastewater effluent quality.

Two specific aspects catalysed the attention and were selected for improvement: biogas production and pre-denitrification potential.

Experimental work and dynamic simulation were combined together to create one single tool for investigation.

A measuring campaign (determination of diurnal wastewater quality variations), a respirometry test, and a Zone Settling Velocity test was performed on wastewater and activated sludge; information was obtained about wastewater quality trend over the day, pollutants removal efficiencies, bacteria kinetics, and sludge settling properties.

Knowledge obtained was integrated with the employment of the Benchmark Simulation Model 1. The Activated Sludge Model 1, for the biological reactor, and the 1-D Takács model for the secondary settler were adjusted and calibrated on the full-scale plant, thus resulting capable of dynamically simulating its performances.

The answer appears clear: the denitrified nitrate load in pre-denitrification may be increased by 4.7 times; methanol in post-denitrification and energy aeration in the aerobic reactor may be considerably reduced; biogas production may be improved through proper operating conditions changes.

These promising optimization solutions are proposed for full-scale plant testing and implementation.

KEY WORDS:
Energy efficiency · High-loaded wastewater treatment plant · Benchmark Simulation Model 1 (BSM1) · Activated Sludge Model 1 (ASM1) · Respirometry test · Biogas

1 Introduction

1.1 Aim of the present study

The aim of the study presented in this thesis is to investigate and determine how the energy balance efficiency of an activated sludge treatment plant could be improved, by acting on the operating conditions. The requirement of a high quality wastewater treatment is not of secondary importance either. The link between water quality and energy efficiency is reflected in the biological activity and corresponding energy consumption or production. The synergy of these two aspects of wastewater treatment was constantly questioned throughout the research.

Wastewater treatment plants (WWTPs) are usually very energy-intensive and expensive to operate (Rojas and Zhelev, 2011 and Yonkin et al. 2008) and their energy consumption is expected to increase by 30-40% in the next 20-30 years (Metcalf and Eddy, 2003). This fact cannot be neglected any more, especially considering that: i) fossil fuel depletion and the current economical crisis are leading to volatile and rising energy costs (Rojas and Zhelev, 2011); ii) environmental pressure on aquatic resources is becoming more severe, with a potential increase of energy required for keeping their ecological quality acceptable (Descoins et al., 2011).

The thesis was born out of this context. Improving energy efficiency is a sustainable measure (Commission of the European Communities, 2006) that allows energy requirement to be reduced often with low investment costs (Rojas and Zhelev, 2011). Surprisingly, there are few articles in the available literature devoted to energy efficiency optimisation of wastewater treatment plants. Descoins et al. (2011) pointed out that so far wastewater experts have focused mainly on wastewater quality issues and models have been developed in that direction. Energy aspects do not usually play a relevant role in treatment plant design, where the major consideration is reserved to effluent requirement satisfaction.

The case study was a high-loaded activated sludge system incorporated in Sjölunda Wastewater Treatment Plant located in the oil port of Malmö, south Sweden. Designed for a Population Equivalent of 550,000 inhabitants it represents one of the biggest WWTPs in Sweden. It is a high-loaded plant for carbon removal only, characterised by i) F/M ratio between 0.5 and 1 $kgBOD(kgTSSm^3)^{-1}$; ii) sludge age lower than 2 days; iii) no nitrification occurring; iv) low denitrification occurring (around 30 $kgNO_3$-N d^{-1} removed); v) carbon-based pollutants removal efficiency around 85%. It is

hydraulically followed by Nitrifying Trickling Filters (NTFs) for nitrification and Moving Bed Biofilm Reactors (MBBR), with external carbon addition, for post-denitrification. The activated sludge process is coupled with anaerobic digestion of primary and biological wastage sludge, resulting in biogas production.

In order to enhance the energy efficiency of this activated sludge plant, two optimisation strategies were selected for investigation for their presumed promising potential on energy and economic savings:

1. Optimization of biogas production through anaerobic digestion of the biological sludge

2. Optimization of pre-denitrification

The former aims to improve biogas production by increasing the biodegradable organic substrate load sent to the anaerobic digestion; sludge age was the key factor manipulated and the effect it has on biological processes, on sludge content, and on effluent quality was investigated. Energy balance was selected as comparison tool for detecting the best operating conditions.

The latter aims to improve pre-denitrification capacity with consequent reduction of the amount of external carbon added in post-denitrification and of the aeration energy supplied in the aerobic compartment (due to partly degradation of biodegradable COD in pre-denitrification). The idea explored for enhancing pre-denitrification, was to recycle part of the nitrate-rich NTF effluent to the head of the biological tank, thus providing available nitrate to facultative heterotrophic bacteria. Effluent quality acted as the key aspect for detecting the maximum amount of recyclable water, limited by the capacity of either pre-denitrification or secondary settler. The energy balance in economic terms was employed for comparing three different scenarios, mainly differing for the way the nitrate-rich flow was recycled. The possibility of extend the settling capacity by increasing the secondary settler volume was evaluated, and in this new condition the three scenarios were assessed again. Conclusions were drawn to select the most promising scenario for full-scale plant testing.

In order to explore these new control strategies and evaluate plant performances under different conditions without affecting the full-scale plant, the use of a steady-state and dynamic modelling tool was chosen. According to Descoins et al. (2011), mathematical models bring a deep understanding of the interactions between physical and biological mechanisms, allowing more reliable predictions to be made. An important step was the model output interpretation; trends given by the model, rather than exact values of pollutants concentration, are the key factors from which strategy conclusions are drawn.

The simulation environment selected for this thesis is the Benchmark Simulation Model 1 (BSM1), developed between 1998 and 2004 by Working Groups of COST Action 682 and 624 (Alex et al., 1999). It couples two theoretical mathematical models for achieving a thorough description of the whole activated sludge system: one for the activated sludge process—the Activated Sludge Model 1 (ASM1) advocated by International Water As-

sociation (Henze et al., 1986)—and the other for the secondary settle—the one-dimensional Takács model (1991).

In order to improve model results reliability, the model needs to be characterized and adjusted to the study case. Design data, process operating conditions, major kinetic and settling parameters, influent and recycling flows, aeration system, daily averaged and dynamic trajectories of influent end effluent wastewater quality were investigated deeply.

A monitoring campaign was designed and carried out for 24 hours, sampling influent and effluent wastewater every hour and analyzing each sample for COD, BOD, TSS, VSS, Nitrate, Nitrite, Ammonia, Total Nitrogen, and Alkalinity. Trajectories of influent and effluent water quality, for each compound, were obtained allowing a better understanding of process performances and the creation of a dynamic input file for the model calibration.

A respirometry test was performed twice on mixed liquor samples to gain information about the quality of biomass in the biological reactor; a submodel, simulating the two tests conditions and performance, was created in Matlab allowing the main kinetic parameters characterizing the growth of heterotrophic bacteria to be detected.

A Zone Settling Column Test was performed on the activated sludge to investigate its settling capability and determine a specific settling velocity parameter. In addition to all these experimental data, historical data from routine laboratory analyses and on-line sensors were collected together to obtain a thorough process understanding and to calibrate and validate the dynamic model.

The final aim of this study was to investigate how the energy efficiency of the plant may be improved and to what degree, offering to the engineers managing the Plant promising optimization solutions that may be tested in the full-scale plant. Besides pointing out further potential research objectives and highlighting the additional experimental work necessary, the thesis leaves open the possibility of extending the research.

1.1.1 Modelling aspects of the wastewater treatment processes

Simplifying reality and identifying the essential internal cause-effect relationships within a phenomena is what we do when we develop a model. Computers now enable mathematical models to be solved numerically. When we want to investigate a process, test a new hypothesis, experiment extreme conditions, predict static or dynamic behaviour or convey the knowledge to other people, we may need a model. In many cases it can replace practical experiments, when these appear too expensive, dangerous, or time-consuming (Finnson, 1994).

Models have extensive application in wastewater treatment as well (Petersen, 2000; Henze *et al.*, 2008): i) design of the plant, balancing treatment performance and costs and assisting in identifying the major parameters that influence the system response and thereby guiding in the establishment of design criteria; ii) process control, investigating new control strategies without affecting the full-scale plant and assisting in identifying possible causes for system malfunction or failure; iii) research, formulating and testing new hypotheses, allowing potentially feasible solutions to be explored and also guiding in the selection of the most promising ones for experimental testing; iv) forecasts, predicting plant performance when the influent wastewater or other conditions change; v) education, exploring the plant behaviour improving the learning process.

An example of wastewater treatment modelling is the application of a model to an activated sludge process. Any activated sludge treatment demands to achieve good treatment performance, at minimum costs, while minimising energy consumption and sludge production. A treatment process control strategy is clearly required. How do you choose the best control strategy? The answer is the need of a standardised procedure capable of evaluating and comparing different types of strategies. The IAWQ Task Group on Benchmarking of Control Strategies for WWTPs is developing benchmark tools for simulation-based evaluation of control strategies for activated sludge plants, work that was started in 1998 by Working Groups of COST Action 682 and 624 (Alex *et al.*, 1999 and 2008).
They finally came up with the standardised Benchmark simulation protocol implemented in this thesis.

1.1.2 Sjölunda wastewater treatment plant

The Sjölunda wastewater treatment plant is located in the oil port of Malmö, South Sweden. It was designed for a Population Equivalent of 550,000 and for an organic load of 70 g $BOD_7(PE{\cdot}d)^{-1}$, thus representing one of the biggest WWTP plant in Sweden. At the moment, it treats an average flow of 4600 m^3h^{-1}. It has been in operation since 1963 and since than it has been upgraded several times. In 1998, a novel process concept was introduced to respond to the new more stringent outlet standards: 10 mg BOD_7 l^{-1}, 0.3 mg total P l^{-1}, and 8 mg total N l^{-1} (according to VA-verket Malmö, 2001). This new concept aimed to achieve nitrogen and phosphorous removal. Some existing structure were modified and adapted for accomplishing new functions (the existing trickling filters originally designed for carbon removal were modified to achieve nitrification) and two new treatment units were built (I: moving bed biofilm reactors for the denitrification and II: a sequencing batch reactor for nitritation of supernatant from the sludge dewatering plant, prior to sending it back to the plant inlet) (Hanner *et al.*, 2003). In addition in 2008, a wet weather overflow plant was completed in order to reduce the number of untreated overflows.

The present Waste Water Treatment Plant (WWTP) structure can be summarised as follows:

Water Treatment Line

Primary treatment

- Flow equalisation
- Inlet pumping station
- 4 Coarse screens (3 cm)
- 4 Grit chambers (volume 1140 m^3) and grit treatment
- 4 Pre-precipitations (volume 1489 m^3)
- 8 Primary clarifiers (Area 5600 m^2, volume 7900 m^3)

Biological treatment

- Flow measurement (Parshall-flumes)
- Activated sludge plant (12 Pre-denitrification basins and 12 Oxidation basins for carbon removal)
- 12 Secondary clarifiers (Area 3270 m^2, volume 11670 m^3)
- 4 Nitrifying trickling filters, NTF (Area 2400 m^2, volume 8640 m^3)
- 6 Post-Denitrification basins with moving bed biofilm reactors, MBBR (volume 6230 m^3, present filling degree 50

Post-treatment

- 16 Flotation basins (Area 2000 m^2)

Other structure

- Outlet pumping station
- Outlet sewers
- Wet weather overflow plant (3 screens with 6 mm mesh, 2 basins (volume 12000 m^3)

Sludge treatment Line

- 3 Primary sludge thickeners (area 390 m^2, volume 1275 m^3)
- 2 Surplus sludge filter belt thickeners (2*200 m^3/h)
- 6 Anaerobic digesters (volume 16000 m^3)
- Reception station for organic material (2 tanks)
- Gas holders
- Gas motors / gas boiler
- Vehicle fuel upgrade
- Buffer tank for digested sludge
- Digested sludge liquor treatment
- Digested sludge storage prior of utilisation

The following lines describe the main important units of the plant and in Section 1.1.3 a more detailed focus on the activated sludge unit under investigation is offered.

Water Line

Inlet pumping station.

The way the influent wastewater is pumped into the WWTP varies according to the conditions of the weather. During dry weather, three pumps located in an inlet pumping station transport the wastewater into the plant. During wet weather, the wastewater is pumped directly into the plant by the pumping stations present in the sewer network. When the incoming flow rate exceeds the plant treatment capacity, the wastewater is pumped by overflow pumps into an adjacent overflow plant, which aims to avoid overflow of untreated wastewater.

Grit removal and treatment.

The grit removal is performed in an aerated chamber and the grit collected is further treated, by washing out organic material, prior to be used for soil construction.

Pre-precipitation.

For the purpose of improving primary clarification and removing phosphorous from the water stream, ferric-based chemicals are added in a pre-aeration basin.

Activated sludge

(See Section 1.1.3 for a more detailed description). This biological treatment consists of an anoxic pre-denitrification basin followed by an aeration basin.
The pre-denitrification does not aim to treat nitrate and nitrite recirculated from the subsequent aeration basin (in fact there is no internal mixed liquor re-cycling from the aerated unit), but its goal now is to denitrify nitrite and nitrate (mainly nitrite) found in the effluent of the SBR plant (where nitritation of supernatant from the sludge dewatering plant occurs). Re-circulation of sludge from the secondary clarifier is provided.
The aerobic basin has the purpose of removing organic load only.
The activated sludge plant, being center of interest of this work, was further discussed in Section 1.1.3.

Secondary sedimentation.

Most of the sludge is recirculated back to the pre-denitrification basin, while the surplus sludge is transferred to the sludge treatment line.

Nitrifying trickling filters (NTF).

In this unit nitrification of ammonium takes place. It is composed of four aerated reactors, operated in parallel, packed with a folded plastic material characterised by a large surface area. The total volume is 8,640 m^3 and the total effective area is of approximately 1,200,000 m^2 (Hanner *et al.*, 2003). The wastewater is distributed over the filters by rotating spreaders. Recycling the water once/twice more over the trickling filters improves the ammonium oxidation. The NTFs operation condition is characterised by a nitrification rate of 1.75 g $NH_4(m^2 \cdot d)^{-1}$ necessary to treat the entire ammonium influent in the plant, an average hydraulic load around 2.35 $m^3(m^2 \cdot h)^{-1}$, and a flushing intensity varying between 11 and 22 mm/pass during average load. Its effluent is characterised in Section 2.7.2, since it will be used for the implementation of the model in the second optimisation solution proposed.

Post-denitrification with moving bed biofilm reactor, MBBR.

In this unit denitrification of nitrate, produced in the trickling filters, occurs. It is operated in 6 parallel lines maintained in anoxic condition and each is filled with plastic element kept completely mixed by mechanical mixers. The media utilised is from Kaldnes Miljoteknologi (Hanner *et al.*, 2003) with a filling degree of 50%. The design denitrification rate is of 1.2 g NO_3-$N(m^2 \cdot d)^{-1}$. An easily degradable carbon source, methanol, is added at the inlet section of the basins as a carbon and energy source for the heterotrophic bacteria. It is stored in large storage tanks and its dosage is controlled by on-line nitrate and flow meters.

Flotation plant.

The intent of this unit is the removal of particulate material, mostly formed in the nitrogen removal stages. Tiny air bubbles (dispersion water) adhere to the flocs, lifting them to the surface and forming a sludge layer. This is scraped away with scrapers and pumped to the sludge treatment plant. Sometimes, when the removal of particulate material and phosphorus has to be enhanced, a coagulant is added to improve flotation.

Outlet pumping station.

The effluent wastewater is normally transported to Öresund, the strait between the Swedish and the Danish coast, by gravity through two large concrete pipes.

Wet weather overflow plant.

During wet weather condition, wastewater exceeding the WWTP capacity is pumped into an overflow pant, built in 2008. It consists of a screening separation and a basin (volume 12,000 m^3) divided into two parts. At the inlet of the second unit, ferric chloride and polymer are added in order to precipitate phosphorus and coagulate particular material. To improve sedimentation, the basin is filled with lamellas, which increase the effective clarifier area.
In case of short precipitation event, the hydraulic load on the plant decreases before the basin is filled, and in this case, the overflow plant acts as a temporary storage and the wastewater is pumped back to the inlet pumping station. In case of long precipitation event, when the hydraulic flow is persistently high and the basin is filled, the treated wastewater flows to the outlet sewer by gravity. When the influent flow to the overflow plant has stopped, the wastewater is pumped back to the inlet pumping station and the overflow tank is emptied. The remaining sludge on the bottom of the tank is washed down and pumped back into the inlet pumping station for being separated in the primary clarifiers.

Sludge line.

Primary sludge thickening.

These gravitation thickeners have the intent of dewatering and reducing the volume of the sludge derived from only the primary clarifiers, before sending it to the anaerobic digestions. The water phase is pumped back into the inlet section of the plant. Before the sludge reaches the thickeners it is passed through screens. The screenings are washed and pressed in preparation for combustion at the nearby combustion plant.

Surplus sludge thickening. These mechanical thickeners aim to dewater two fluxes: the surplus sludge separated from the return sludge of the secondary clarifiers and the flotation sludge. Polymer is added to improve the process. The water phase is pumped back into the inlet section of the plant.

Anaerobic Digesters.

This unit is made of three parallel lines, each includes two digesters that are charged consecutively. They are operated in mesophilic condition keeping the temperature around 35-37 degrees Celsius. Co-digestion of primary and secondary sludge, with sludge coming from grease removal tanks from

restaurants, occurs. The digested sludge is temporarily stored in a large storage tank, before sludge dewatering. The biogas produced is collected in gas holders prior to being conducted to two cogenerators, which produce both heat and electricity used for operating the WWTP. The remaining heat is delivered to the central heating network. If needed, a gas boiler may convert biogas into heat. If the gas treatment units is out of order, the biogas is burned in a gas torch. Besides electricity and heat, the biogas can also be refined into vehicle fuel after removal of carbon dioxide, particles and other unwanted substances.

Digested sludge dewatering.

Centrifuges aim to dewater digested sludge with addition of polymer to improve the separation process. The dewatered sludge, reduced in volume and increased in solid and nutrient concentration, is transported by truck to a storage facility. It is controlled by inspection bodies so that the certified product can be applied optimally in agriculture. A soil product can also be produced by adding sand and other construction material to the sludge. The sludge liquor is pumped into a sequencing batch reactor for further treatment.

Sludge liquor treatment, SBR.

This unit aims to reduce the ammonium concentration of the digested sludge liquor from the dewatering plant, prior to send it back to the water treatment Line. This step is necessary for not overloading the nitrifying trickling filters, which nitrifyes the ammonium coming with the influent wastewater. It consists of a sequencing batch reactor, SBR, where nitritation of ammonium to nitrite occurs. The nitrite produced will be further denitrified in the pre-denitrification section of the activated sludge basins. The reactor is operated in 4 cycles per day. In some sequences, air is blown in and sodium hydroxide is added, in order to maintain the alkalinity value in the reactor. It is designed to treat 700 kg NH_4-N d^{-1} (Hanner *et al.*, 2003).

1.1.3 Focus on the activated sludge section

The main aim of this work is the energy balance optimisation of the activated sludge (AS) unit. For this reason, this Section will provide a more detailed description of its structure and its operation. In Section 2.5, a complete characterization of influent, effluent, process operation is offered. The activated sludge plant consists of a biological step—further divided into an anoxic sector followed by an aerobic one—and a secondary clarification step. It is operated in 4 lines: 3 smaller lines designed with the same capacity, treating almost 50% of the flowrate, and 1 bigger line treating the rest. Each of the 3 equal smaller lines is composed of two further parallel basins, leading to a total of 6 equal basins, as shown in Figure 1.1. In August 2012, one of these three lines was shut down. This study is focused on one single basin, specifically on the second basin of the second line (circled in green in Figure 1.1).

The design structure

The biological basin under study, like the other equal basins, is made of two compartments operated in two different ways and is followed by a sec-

ondary settler.
The pre-denitrification compartment, hydraulically preceding the aerobic compartment, is operated in absence of dissolved oxygen and is maintained in movement through opportune mixers. It is divided into two equal sectors in series and each of them has a volume of 206.25 m^3 considered completely mixed.
The aeration compartment, kept continuously aerated, is divided into three equal sectors in series, each of them has a volume of 412.5 m^3 considered completely mixed.
The depth of the entire basin is 3.8 m, leading to a total volume of 1,625 m^3.
The secondary clarifier has a volume of 1650 m^3, is 3.8 m deep, and is divided in two equal compartments.

The process

The activated sludge section is operated as a high loaded activated sludge plant (further information about high loaded activated sludge plant are found in Section 2.4). Despite the biological basin includes the anoxic and aerobic compartments, it now aims to remove only the carbon fractions, whereas it had been designed for nitrogen removal as well. No mixed liquor recycle is provided, so very little pre-denitrification is present. Nitrification and post-denitrification occur in the two treatment sections that follow the activated sludge plant: the nitrifying trickling filters and the post-denitrifying moving bed biofilm reactors. Considering the values of Autumn 2012 for the specific activated sludge basin under study (described in detail in Section 2.5.1.3), the average total suspended solids concentration in the tank is kept around 2600 mg TSS l^{-1}, leading to an average F/M (food to microorganism ratio) of 0.5 kg BOD (kg TSS·d)$^{-1}$. The average hydraulic retention time for the biological reactor is 2.94 h and 5.94 h for the entire section (biological reactor plus secondary sedimentation). The solid retention time is low and is around 2 days (average values of autumn 2012). Recycling of activated sludge from the secondary clarifier is provided, with an average flowrate of around one third the influent wastewater. The TSS concentration in the return sludge is around 9300 mg TSS l^{-1}, but this value is highly variable. A daily average amount of 185 m^3d^{-1} of surplus sludge is sent to the sludge treatment line (only from the basin under investigation).

The aeration strategy

The aerobic compartment is divided into three completely mixed reactors operated in series. The aeration power supply varies through out the three reactors. The strategy aims to keep the oxygen concentration equal to 2 mg l^{-1} in the last aerobic reactor. This is achieved by regulating the aeration flux to the two previous aerobic sectors; this strategy is called the "cascade aeration". In this way, three different oxygen concentration levels are obtained in the three reactors: 0.4 mg O_2 l^{-1}, 0.9 mg O_2 l^{-1}, 2 mg O_2 l^{-1} repectively in the first, in the second, and in the third reactor (average values of Autumn 2012).

Characterisation of wastewater influent and effluent from the activated sludge section

Two fluxes are conveyed to the activated sludge plant: the influent wastewater and the SBR effluent, which accounts for only 5% of the total influent flow rate.
The influent wastewater undergoes grit removal and primary clarification prior to reaching the activated sludge section.
The SBR effluent is rich in nitrite, result of the nitritation process carried out in the reactor. Even if the nitrate concentration is high (around 16 mg l^{-1}, in Autumn 2012), it is diluted in the bigger flow of the mainstream, leading to median concentration of 1 mg($NO_{2,3}$-N) l^{-1}.
To characterise the influent and effluent flows of the basin, for the model calibration and validation, two sources of data were employed: for the calibration, a measuring campaign was performed to collect more information to dynamically characterize the wastewater; for the validation, historical laboratory analyses carried out regularly were available. All data are reported in Section 2.5.

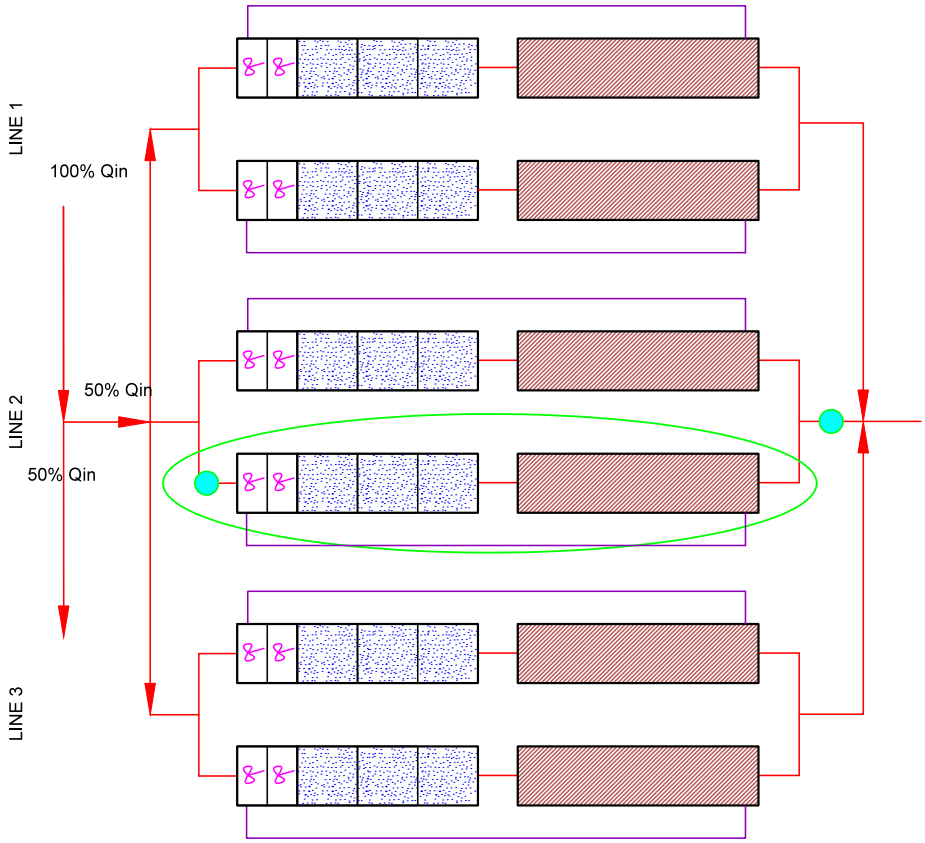

Figure 1.1: Scheme of the 3 equal activated sludge lines treating half of the influent flow rate. The studied section is circled in green. The green dots represent the places of sampling for the routine laboratory work.

2 Materials and Methods

In order to organise the thesis work, the guidelines by Langergraber *et al.* (2004) were followed and adapted to the specific goals of this study. The work was divided in seven major stages (shown in Figure 2.1):

1. **Definition of the objectives**

   The objectives of this study were discussed in cooperation with the process engineer of Sjölunda wastewater treatment plant (David Gustavsson).

2. **Data collection and model selection**

   Collection of plant routine data. Particular attention was given to the identification of the set of information necessary to characterise the model. Operational data (SRT, set-points) and performance data (daily mean values of influent/effluent, flow rates, mixed liquor quality) were provided by the Sjölunda process engineer (See Section 2.5.1). Definition of model boundaries and model selection. A specific activated sludge line-treating $\frac{1}{8}$ of the whole influent flow rate-was selected for this study, assuming to be representative of the behaviour of the other parallel lines.

   The model selected was the Benchmark Simulation Model 1, able to describe a whole activated sludge system: biological reactor + secondary settler. It includes two mathematical models: the IWA Activated Sludge Model 1 (Henze *et al.*, 2000) for describing the biochemical transformation and degradation processes in the biological rector; and the dynamic model (Takács *et al.*, 1991) for the secondary clarifier (See Sections 2.1 and 2.2).

3. **Data quality control**

   Data evaluation. The available data were evaluated and missing data individualized.
   Data quality assurance. Data were assessed by means of COD and total nitrogen mass balance calculation (suggested by Nowak *et al.*, 1999; Meijer *et al.*, 2001.). Langergraber *et al.* (2004) strongly recommend checking mass balances before performing the monitoring campaign.

4. **Evaluation of other information sources and experimental design**

Evaluation of other information sources. The idea of obtaining dynamic data of influent COD starting from data of airflow blown into aerobic reactors (continuously recorded) has been taken into account. This idea was abandoned when the measuring campaign was projected. It will not be discussed any further in this thesis.

Setting up a monitoring campaign and other experiments. The experimental work was designed with respect to collected data, to the missing data, to the main goals defined and to budget considerations.

The measuring campaign. According to Ljung (1987), it should have a duration of 3-4 times the hydraulic retention time, which in the case in point is of 6 hours, leading to a monitoring time of 24 hours. Only those parameters considered essential for deriving the values of the 16 ASM1 variables were analyzed. The frequency of the sampling—each hour—was decided as a compromise between the time-step used in the BMS1 (15 minutes) and practical/economical considerations. The day of the monitoring was chosen in a dry week, so that no dilution due to rain had to be taken into account.

Respirometry and Settling Column test. These two experiments were performed for gathering information about some kinetic parameters of the heterotrophic metabolism and settling parameters. The OUR test was performed two times to increase the results reliability.

5. **Data collection for simulation study**

   Data for model calibration were experimentally retrieved. Data quality evaluation. Like in phase 3, the monitoring campaign results were evaluated. Some data obtained (e.g. TSS concentration in the effluent) was found to be outside the ranges of the data gathered from routine laboratory analyses. Therefore, some problems in the interpretation of the results occurred.

   Data elaboration and creation of the sub-model. All collected data were analyzed and elaborated. The elaboration of the Respirometry test results required the creation of a sub-model in Matlab, in order to estimate kinetic parameters.

6. **Calibration and validation**

   Initial conditions. The set of parameters selected derives partly from literature, partly from the default BSM1 values, and partly from experimental work.

   The input files (steady state and dynamic) were initially created starting from theoretical relationships, taken from literature (Petersen, 2000), that link lab analyses to the 16 state variables of the model.

   Calibration and parameterization. The parameter values not selected from literature and the influent characterizing fractions needed a calibration stage. The kinetic parameters were first calibrated through a sub-model and afterwards calibrated in the full-scale model.

The overall procedure was divided into steps and during each step only one sensible parameter was adjusted (Langergraber *et al.*, 2004). Validation. The validation is done to verify the model under an independent set of data. Unfortunately, the validation did not give the expected results.

7. **Development of scenarios and evaluation of success.**

   Once the model was calibrated the selected scenarios could be simulated and assessed. The model offered the possibility of identifying the presence of energetic or economic saving potential and of evaluating the possibility to run the plant dynamically considering the diurnal variations.

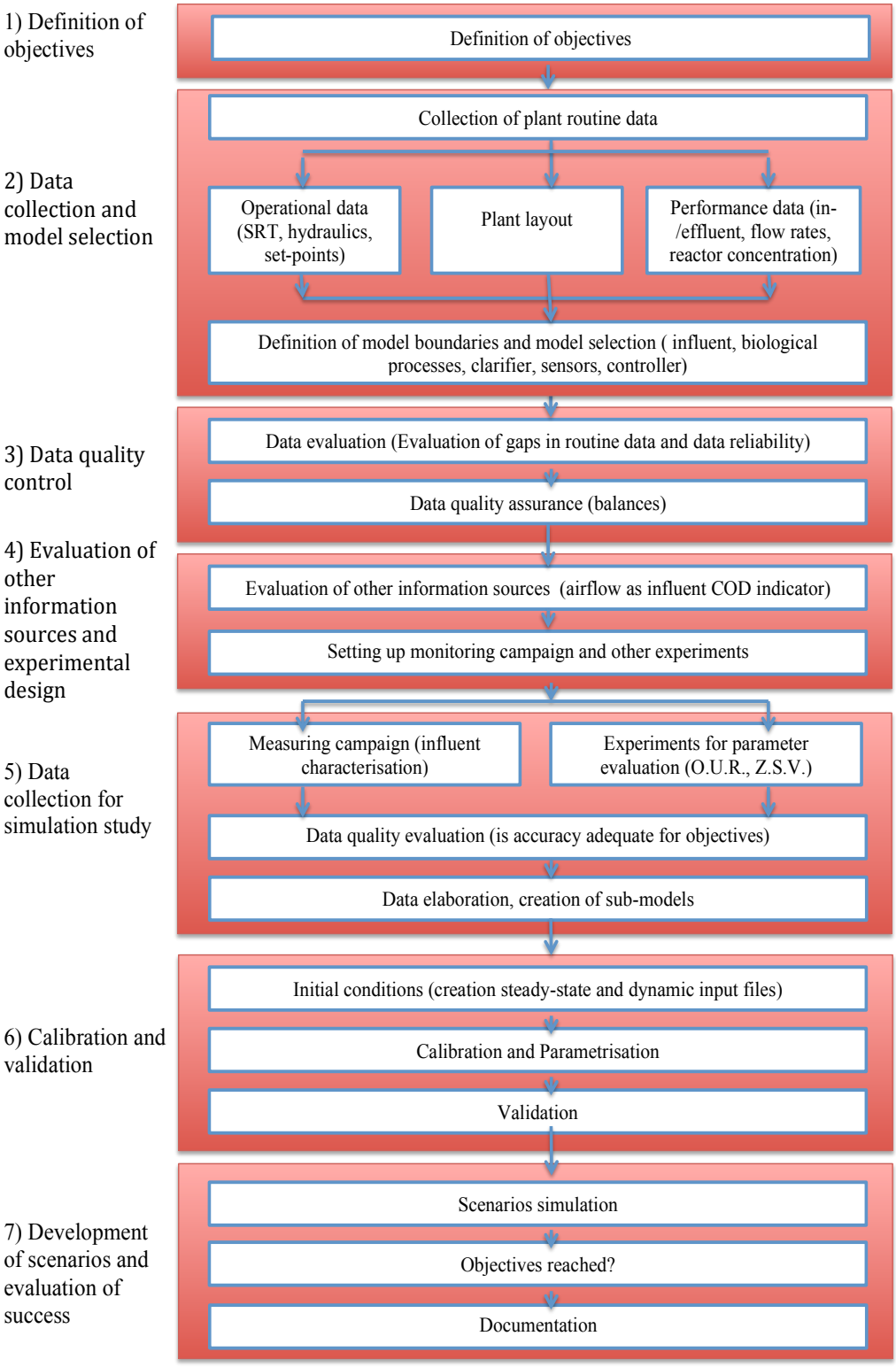

Figure 2.1: Steps of the thesis, readapted from Langergraber et al. (2004)

2.1 Activated Sludge Model 1 (ASM1)

Activated Sludge Model 1 is a theoretical mathematical model depicting the biological processes occurring in the activated sludge section of a wastewater treatment plant. It represents an useful tool for the design and operation of a plant. It was developed in 1986 (Henze *et al.*, 1986) by the task group formed from the International Association on Water Quality (IAWQ, formerly IAWPRC). The primary aim was to set out a standardisation of biological WWTP design by building a mathematical model able to realistically describe carbon oxidation, nitrification and denitrification.
The more detailed and close to reality the model equations are, the more complicated the computational solutions are likely to be. Therefore, the modellers focused on finding the best balance between these two conflicting needs, depicting only those processes considered essential to a realistic prediction and selecting the easiest rate expressions consistent with them. Eight processes were chosen resulting in eight rate expressions.
According to the task group, the main aspect the model should be able to carefully predict is not the effluent concentration, which usually does not vary considerably from plant to plant (especially considering that most WWTP adopt a long solid retention time and a low specific growth rate). Two other aspects were picked out for their importance to be accurately predicted: the solids concentration of the activated sludge and the electron acceptor requirements. A good appraisal of these two phenomena is important, since large differences from plant to plant are usually encountered. Thus, stoichiometric expressions were selected to better describe the activated sludge concentration and rate equations to better define electron acceptor requirements.

2.1.1 State variables and model parameters

Chemical Oxygen demand was chosen as the proper measurement unit for describing those model variables that are related to the process of carbon removal. In fact, it provides a link between electron equivalents in organic matter, biomass and oxygen consumption and assures units consistency throughout the model. Furthermore, it offers the possibility to easily carry out mass balances in terms of COD unit.
The model incorporates thirteen variables necessary to depict carbon-based, nitrogen-based pollutants, together with biomass, oxygen and alkalinity. Each chemical compound is described by a stoichiometric formula. Carbonaceous and nitrogenous matter are fractionated in several components, according to Dold *et al.*, 1980, as shown in Figures 2.2 and 2.3 and described in the following lines. Organic matter is subdivided into several components—all of them expressed as COD units—adopting the bisubstrate hypothesis by Dold *et al.*, 1980. The total COD is partitioned according to biodegradability (readily biodegradable, slowly biodegradable, and non-biodegradable) and physical state (soluble and particulate). Variables nomenclature conforms with IAWPRC, where soluble component are denoted S, particulate are denoted X, biodegradable are subscribed s, non-biodegradable are subscribed i.

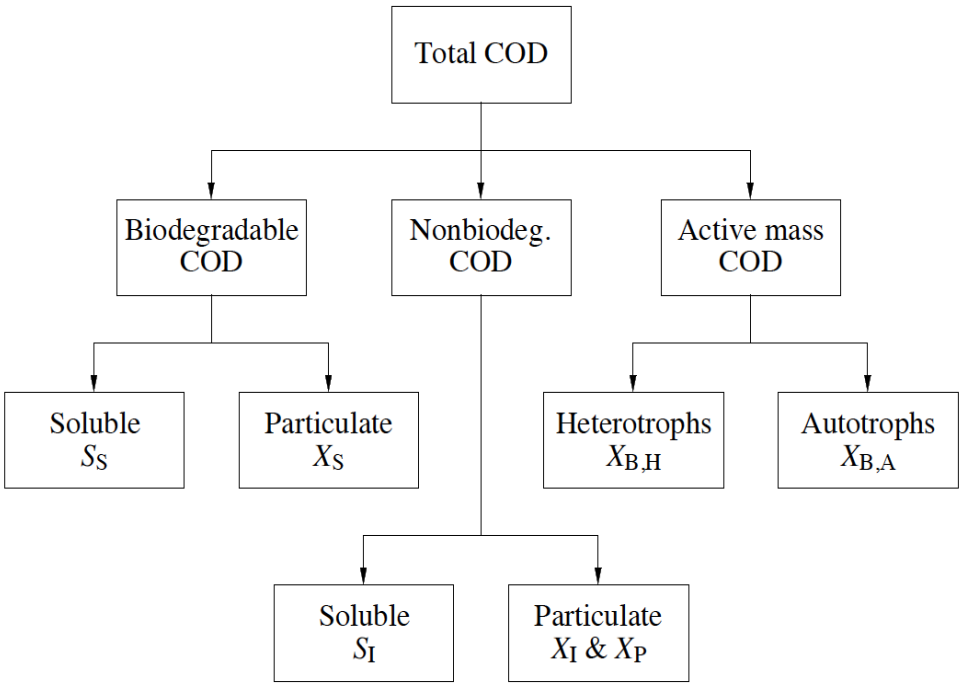

Figure 2.2: COD fractionation.

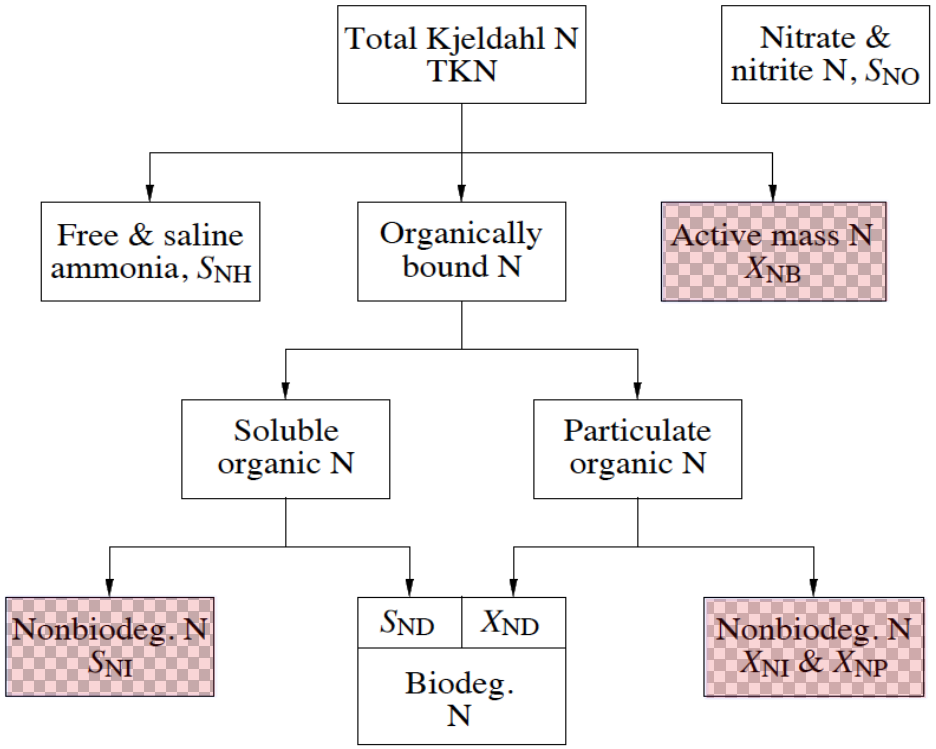

Figure 2.3: Nitrogen fractionation. The coloured boxes represent fraction neglected into the ASM1.

The thirteen variables are described below.
Biodegradable carbon is divided into two fractions: readily biodegradable, S_s, assumed as if it were all soluble, and slowly biodegradable, X_s, assumed as if it were all particulate. These are assumptions that have a merely modelling purpose—helping in the prediction of electron acceptor requirement—, but it is known that some slowly biodegradable material could actually be soluble.

- *Readily biodegradable fraction,* S_s: it is consumed by growth of heterotrophic bacteria (both under aerobic and anoxic conditions); it is the hydrolysis result of slowly biodegradable matter entrapped in the biofloc, besides being introduced through the influent wastewater.

- *Slowly biodegradable fraction,* X_s: it is presumed to be instantaneously entrapped in the biofloc, from where it is transformed by hydrolysis into S_s. Its formation occurs through decay of heterotrophic and autotrophic biomass.

Non-biodegradable carbon is partitioned into soluble and particulate fractions: both are considered not to be involved in any biological conversion process.

- *Soluble non-biodegradable fraction,* S_i: it strongly contributes to the effluent COD, since its influent amount in the wastewater is considered to leave unchanged the system.

- *Particulate non-biodegradable fraction,* X_i: it is enmeshed in the sludge and subtracted from the system by removal of the excess sludge; it constitutes a part of the volatile suspended solids of the activated sludge.

The active biomass in the system is partitioned into heterotrophs, X_{bh}, and autotrophs, X_{ba}, still expressed through COD units.

- *Heterotrophic biomass,* X_{bh}: it grows under both aerobic and anoxic condition and is destroyed by decay.

- *Autotrophic biomass,* X_{ba}: it is assumed to grow under only aerobic condition and is destroyed by decay.

- There is an additional variable, Xp, which is an inert particulate result of biomass decay. As it will be explained in more detail in the next Section, decay of both heterotrophic and autotrophic bacteria is assumed to generate two fractions: X_s and Xp. X_s ri-enters the cycle of hydrolysis, conversely, X_p is not further transformed and accumulates in the system as inert particulate. This assumption may not reflect reality, however it is introduced in the model to take into account the fact that not all biomass is active.

The sum of $S_i + S_s + X_s + X_i + X_{bh} + X_{ba} + X_p$ builds up the total COD. The sum of $X_s + X_i + X_p + X_{bh} + X_{ba}$ constitutes the Volatile Solids.

- *Oxygen concentration* in the reactor, denoted with S_o, is expressed as negative COD unit. In the Activated Sludge Model 1, oxygen concentration is consumed by aerobic growth of both heterotrophic and autotrophic bacteria; no addition of oxygen in the reactor is modelled. The Benchmark Simulation Model 1 integrates the contribution of the aeration system by incorporating in the rate expression the oxygen transfer coefficient, $K_l a$, which regulates S_o dissolution in the system.

Nitrogenous material, like carbon material, is assumed to be subdivided into various fractions, according to Figure. However, only four of those fractions are incorporated into the model and are: nitrate nitrogen S_{no}, soluble ammonia nitrogen S_{nh}, soluble biodegradable nitrogen S_{nd} and particulate biodegradable nitrogen X_{nd}.

- *Nitrate nitrogen fraction,* S_{no}: it is the second electron acceptor, after oxygen, present in the model. It is consumed for energy by growth of facultative heterotrophic bacteria and it is formed as a result of autotrophs growth under aerobic condition. For sake of simplicity, the model assumes that nitrification of ammonium nitrogen is one single step process and that nitrate nitrogen is the only oxidized form of nitrogen.

- *Soluble ammonium nitrogen,* S_{nh}: it is assumed to include both ionized and un-ionized forms of ammonium. It derives from ammonification of soluble organic nitrogen and it is used for energy by growth of autotrophic nitrifying bacteria. In addition, it is integrated into new cells during both heterotrophic and autotrophic cell synthesis.

- *Soluble biodegradable nitrogen fraction,* S_{nd}: it is converted to ammonia by ammonification and is the result of hydrolysis of particulate organic nitrogen.

- *Particulate biodegradable nitrogen fraction,* X_{nd}: it is hydrolysed to S_{nd}, together with the hydrolysis of slowly biodegradable COD. It increases in parallel with the decay of heterotrophic and autotrophic biomass.

- The thirteenth variable is *total alkalinity,* S_{alk}; it is not indispensable in the description of substrate removal and neither affects other processes in the model, but it may help for checking the variation of pH. Processes involving addition or removal of protons have impact on alkalinity. The model considers the influence on alkalinity of these processes:

    - ammonification and conversion of ammonia to amino acids;
    - denitrification that produces an increase of alkalinity;
    - nitrification that has the greatest impact with the net release of two protons, thus decreasing alkalinity.

Kinetic and stoichiometric coefficients are incorporated into the model. They are considered to be constant at a fixed temperature. However, in reality

most of them vary over time for several reasons, like variations of pH, temperature, operating conditions, bacteria dynamics, influent wastewater composition etc... Some of them have more impact on the prediction of plant performance, requiring a more accurate calibration. This subject will be examined in more detail in Section 2.6.5 where calibration of some kinetic parameters for the specific wastewater treatment plant under study.

2.1.2 Mathematical formulation and dynamic process equations

Four kind of processes are included into the model: growth of biomass, decay of biomass, ammonification of soluble organic nitrogen, and hydrolysis of particulate biodegradable carbon. A total of eight processes are present, each of them mathematically expressed as a differential equation. In this Section, model processes, equations, and assumptions are described.
Model equations do not pretend to exactly describe reality, but to realistically simulate the major processes effects. Therefore, the provided equations need to adapt and change their behaviour in relation to environmental condition under which occur. This is performed by incorporating in the equations the so called "switching function", which are able to turn the process on and off whenever it is needed. This is particularly useful in expression rates involving phenomena dependent upon the electron acceptor present. Mathematically, this switching function is modelled with a saturation function. Examples of its use is given in the following lines, where all processes, together with the rate expressions are described.

With respect to growth of biomass, three different processes are taken into account; essentially all of them are mathematically described as $\frac{dX}{dt} = \mu \cdot X$, where μ is the specific growth rate and X represents a generic biomass. Further, stoichiometry is used to relate biomass to substrate (S_s) through the yield coefficient, Y_h: $\frac{dX}{dt} = -Y_h \cdot \frac{dS}{dt}$. The three different microbial growth expressions are described below.

- *Growth of heterotrophic biomass under aerobic condition.*

$$\begin{aligned} \rho(1) &= (-\frac{1}{Y_h}S_s + X_{bh} - \frac{1-Y_h}{Y_h}S_o - i_{xb}S_{nh} - \frac{i_{xb}}{14}S_{alk}) \cdot \\ &\cdot \quad \hat{\mu}_h(\frac{S_s}{K_s+S_s})(\frac{S_o}{K_{oh}+S_o})X_{bh} \end{aligned} \tag{2.1}$$

  This process occurs at the expense of only readily biodegradable carbon ($-\frac{1}{Y_h}S_s$), which is used both as energy and as carbon source, and results in the production of heterotrophic biomass, X_{bh}. In parallel, oxygen is consumed ($-\frac{1-Y_h}{Y_h}S_o$). All these three variables (S_s, X_{bh}, S_o) are expressed in terms of COD, allowing to check for continuity. Being oxygen concentration denoted as negative COD, its utilization balances net COD consumption, which is biodegradable carbon consumed minus amount of biomass grown (S_o consumed = S_s consumed - X_{bh} produced, $-\frac{1-Y_h}{Y_h}S_o = -\frac{1}{Y_h}S_s - X_{bh}$). During growth of heterotrophic biomass, ammonium nitrogen is incorporated into new cells ($i_{xb}S_{nh}$). Two limiting

functions are present to express that the aerobic heterotrophic growth is subjected to two limitations: presence of readily biodegradable substrate and oxygen. The second of these functions, $(\frac{S_o}{K_{oh}+S_o})$, acts as a switching function; for this reason, employs a low value of the half saturation coefficient K_{oh}, which permits the aerobic growth to stop under low oxygen concentration conditions.

- *Growth of heterotrophic biomass under anoxic condition.*

$$\begin{aligned}\rho(2) &= (-\frac{1}{Y_h}S_s + X_{bh} - \frac{1-Y_h}{2.86Y_h}S_{no} - i_{xb}S_{nh} + \frac{1-Y_h}{14\cdot 2.86Y_h - \frac{i_{xb}}{14}}S_{alk})\cdot \\ &\cdot \quad \hat{\mu}_h(\frac{S_s}{K_s+S_s})(\frac{K_{oh}}{K_{oh}+S_o})(\frac{S_{no}}{K_{no}+S_{no}})\eta_g X_{bh} \qquad (2.2)\end{aligned}$$

This process requires the presence of readily biodegradable carbon for carbon needs and nitrate as electron acceptor allowing denitrification to occur. Similarly to the case of oxygen in the previous process, nitrate consumption $(\frac{1-Y_h}{2.86Y_h}S_{no})$ equals the net COD removal (readily biodegradable substrate,S_s, consumed minus new cells, X_{bh}, formed). The factor 2.86 is needed to convert nitrate nitrogen to nitrogen gas, in terms of oxygen equivalence.
During growth of biomass, ammonia nitrogen is incorporated in the new cells $(i_{xb}S_{nh})$ exactly like happens in aerobic growth. Alkalinity increases during denitrification, since the reduction of nitrate involves a net uptake of a proton. It is known that the rate at which substrate is removed in anoxic condition is lower in respect to aerobic condition. The modeller chose to introduce this difference by adding an empirical coefficient η_g, modelling technique considered to be the easiest.
Three limiting functions are included in the equations, since anoxic growth is considered to be limited by concentration of oxygen, nitrate and readily biodegradable carbon. The dependence on nitrate nitrogen, $(\frac{S_{no}}{K_{no}+S_{no}})$, is analogous to the relationship between aerobic growth and oxygen concentration (explained in the previous process). Conversely, with respect to oxygen concentration, its presence inhibits anoxic growth; for this reason, the switching function, $(\frac{K_{oh}}{K_{oh}+S_o})$, uses the same K_{oh} included in the expression of aerobic growth, allowing anoxic growth to develop when aerobic growth declines.

- *Growth of autotrophic biomass under aerobic condition.*

$$\begin{aligned}\rho(3) = &\quad (X_{ba} - \frac{4.57-Y_a}{Y_a}S_o + \frac{1}{Y_a}S_{no} + (-i_{xb} - \frac{1}{Y_a})S_{nh} + \\ + &\quad (-\frac{i_{xb}}{14} - \frac{1}{7Y_a})S_{alk})\cdot \hat{\mu}_a(\frac{S_{nh}}{K_{nh}+S_{nh}})(\frac{S_o}{K_{oa}+S_o})X_{ba} \qquad (2.3)\end{aligned}$$

Autotrophic biomass grows at the expense of soluble ammonium, which being used as electron donor is oxidised to nitrate. Besides, ammonium nitrogen will also be incorporated in the new autotrophic cells. Oxygen is utilized in proportion to the amount of ammonium consumed, $\frac{4.57-Y_a}{Y_a}S_o$.

The effect of pH is not included in the model, although having a relevant impact upon nitrification process. Alkalinity should be checked instead, since it decreases due to a net release of two protons during ammonium oxidation. Two limiting functions are incorporated in the equation, to express the dependency upon soluble ammonium concentration and oxygen concentration. The latter acts as a switching function, $(\frac{S_o}{K_{oa}+S_o})$.

Microbial decay is modelled according to the *death-regeneration* concept by Dold *et al.*, 1980. It is assumed that the decayed cell is released by lysis, resulting in two particulate fractions: an inert component X_p, which is not further subjected to biological attack and a slowly biodegradable component X_s, which enters the cycle of hydrolyses for being transformed into readily biodegradable substrate S_s and becoming thus again available for biomass growth. In the model, it is hypothesised that the decay rate keeps the same magnitude regardless of the type of electron acceptor present and does not involve any electron utilization. The death-regeneration concept is able to predict well the loss of biomass occurring in an activated sludge reactor, but there is no evidence that it reflects the real biological mechanism taking place. It was adopted for a pragmatic reason.
Two decay rate expressions are included in the model: decay of heterotrophs and autotrophs.

- *Decay of heterotrophic biomass.*

$$\rho(4) \quad = \quad ((1-f_b)X_s - X_{bh} + f_p X_p + (i_{xb} - f_p i_{xp})X_{nd})) \cdot b_h X_{bh} \tag{2.4}$$

  According to the death-regeneration model, the disappearance of one unit of biomass, X_{bh}, generates a fraction of inert biomass, $f_p X_p$, and a balancing fraction, $(1-f_b)X_s$, which is slowly biodegradable carbon. Besides, particulate organic nitrogen is also released, $(i_{xb} - f_p i_{xp})X_{nd}$, and it is then hydrolysed into soluble organic nitrogen, thus becoming available for ammonification. The expression rate describing microbial decay is a first order equation with respect to heterotrophs concentration, X_{bh}, through a decay coefficient, b_h. The magnitude of the decay coefficient encountered in this model is larger than the usually used rate constant. This is because it includes the recycling of carbon substrate.

- *Decay of autotrophic biomass.*

$$\rho(5) \quad = \quad ((1-f_b)X_s - X_{ba} + f_p X_p + (i_{xb} - f_p i_{xp})X_{nd})) \cdot b_a X_{ba} \tag{2.5}$$

  This process is handled in the same way as the decay of heterotrophs, considering however a smaller decay coefficient, b_a.

- *Ammonification of soluble organic nitrogen.*

$$\rho(6) \quad = \quad (S_{nh} - S_{nd} + \frac{1}{14}S_{alk}) \cdot k_a S_{nd} X_{bh} \tag{2.6}$$

  The ammonification of biodegradable soluble organic nitrogen generates free and saline ammonia. This relationship is expressed through a first-order equation mediated by heterotrophic biomass.

The last two processes incorporated in the model are the hydrolysis of slowly biodegradable carbon and particulate organic nitrogen. These phenomena play a significant role, since they allow a realistic estimation of the electron acceptor profile in time and space.

- *Hydrolysis of slowly biodegradable substrate.*

$$\begin{aligned} \rho(7) &= (S_s - X_s) \cdot k_h \frac{\frac{X_s}{X_{bh}}}{K_x + \frac{X_s}{X_{bh}}} [(\frac{S_o}{K_{oh} + S_o}) + \\ &+ \eta_h (\frac{K_{oh}}{K_{oh} + S_o})(\frac{S_{no}}{K_{no} + S_{no}})] X_{bh} \end{aligned} \tag{2.7}$$

  Organic matter is broken down extracellularly into readily biodegradable carbon, which enters again in the cycle of biomass growth. It is assumed to occur only under aerobic and anoxic environment. Mathematically, it is modelled on the basis of surface reaction kinetics. The rate of hydrolysis is lower under anoxic condition compared with aerobic condition and is reduced in the equation by the addiction of a factor η_h <1. It results in a first-order equation with respect to heterotrophic biomass, but it saturates as the amount of entrapped substrate becomes large in respect to the biomass.

- *Hydrolysis of particulate organic nitrogen.*

$$\rho(8) = (S_{nd} - X_{nd}) \cdot \rho_7 \frac{X_{nd}}{X_s} \tag{2.8}$$

  Particulate organic nitrogen, X_{nd}, is degraded to soluble organic nitrogen, S_{nd}. Nitrogen is assumed to be uniformly distributed throughout the slowly biodegradable carbon, so that the rate of hydrolysis of particulate organic nitrogen could be proportional to the rate of hydrolysis of slowly biodegradable substrate.

2.1.3 Assumptions and restrictions

Several assumptions and restrictions have been made with the purpose of simplifying the model equations and reducing the computational effort. These hypothesis are summarised and listed below.

- Temperature (15°C) and pH (neutral) are assumed to be constant. The influence of pH is not included in the model but its control is made by checking alkalinity.
- Stoichiometric and kinetic coefficients are assumed to be constant.
- Nutrient limitations are not taken into account.
- The heterotrophic biomass is considered to be homogeneous and not to change is species diversity.

- Hydrolysis of organic matter and particulate nitrogen occur simultaneously with the same rate. Nitrogen is homogeneously distributed throughout the slowly biodegradable carbon, in order to allow the particulate organic nitrogen hydrolysis rate to be proportional to the slowly biodegradable carbon hydrolysis rate.
- Slowly biodegradable substrate is entrapped instantaneously in the biofloc.
- Decay of biomass does not depend upon the type of electron acceptor present.

2.2 Settler Model

In this Section, the dynamic model for the secondary clarifier is outlined. It reproduces the clarification-thickening processes and describes the solids profile throughout the settling column, including the underflow and effluent suspended solids concentrations (Takács, 1991). It is coupled with the Activated Sludge Model 1 (outlined in Section 2.1) for a thorough description of the whole activated sludge treatment system.
It is a one-dimensional model, where only processes on the vertical dimension are described, whereas horizontal solids gradients and horizontal velocity contributions are neglected (Vitasovic, 1985). The secondary settler is idealized as a settling cylinder with a constant cross sectional area A. The model is based on the flux theory (Kynch, 1952) and takes into consideration two distinct contributions:

1. the bulk flow flux, which may be distinguished into downward flow—towards the underflow exit—and upward flow—towards the effluent exit;

2. the solids settling flux, relative to the water.

In mathematical terms, the two factors may be combined together in the following equation:

$$J_{tot} = v \cdot X + v_s \cdot X \tag{2.9}$$

expressed as differential equation:

$$-\frac{\partial X}{\partial t} = v\frac{\partial X}{\partial z} + \frac{v_s \partial X}{\partial z} \tag{2.10}$$

The bulk flow flux contribution, $J_B = v \cdot X$, consists of the vertical bulk velocity, v, depending weather the observed cross section is in the overflow region over the inlet position or in the underflow region, and the solids concentration throughout the depth of the settler.
The solids settling flux contribution, $J_S = v_s \cdot X$, consists of the settling velocity of the sludge, v_s, and the solids concentration, both function of the settler depth.
The flux theory is implemented in the Benchmark Simulation Model 1 and its mathematical framework derives by dividing the settler in n layers (n=10) and discretizing the differential conservation equation on these layers: the change of total amount of particles in a layer is equal to the inward net flux across the horizontal section of the settler, considering that no sources or sinks are present (Jeppsson, 1996).
The settling velocity of the sludge has been empirically defined by many authors; the BSM1 implements the Takács function based on the exponential function, which simulates the settling velocity of dilute and more concentrated suspensions. The main factors having a major role in the model are the settling velocity and the sludge solids concentration.

The Sections below describe the mathematical set of equations derived from the flow flux theory application and the Takács (1991) settling velocity model, together with the main assumptions.

2.2.1 Mathematical formulation of the settler model

Geometrically, the settler is assumed to be divided into n horizontal layers (counting from the top to the bottom) with equal depth and each considered completely mixed. The effluent and the thickened sludge are withdrawn from the first (the top) and last (the bottom) layers, respectively. The inlet is located at layer m, assuming that the feed is instantaneously and completely distributed throughout the inlet layer. The other layers are grouped in layers standing above and below the feed layer, according to their respective position.
The set of equations presented below constitutes the secondary clarifier model and derives from the application of the mass conservation law to each layer (the letter i indicates the layer taken into account and the letter m is the feed layer):

- For the top layer ($i = 1$)

$$\frac{\partial X_1}{\partial t} = \frac{J_{up,2} - J_{up,1} - J_{clar,1}}{z_1} \tag{2.11}$$

  where
  X_1 is the effluent solid concentration;
  z_1 is the height of the top layer;
  J_{clar} is the flux of TSS of the clarification zone of the settler;
  J_{up} is the upward flux of TSS due to upward bulk flow and is defined as

$$J_{up,i} = v_{up} \cdot X_i$$

  where v_{up} is the upward bulk fluid velocity expressed as

$$v_{up} = \frac{Q_e}{A}$$

  assuming A being the cross-sectional area of the clarifier.
  This equation represents the first boundary condition.

- For the layers above the feed layer ($m < i < n-1$) (which build up the clarification zone)

$$\frac{dX_1}{dt} = \frac{J_{up,i+1} + J_{clar,i-1} - J_{up,i} - J_{clar,i}}{z_1} \tag{2.12}$$

  where the solids flux in the clarification zone is defined as

$$J_{clar,i} = \begin{cases} J_{s,i} & if \quad X_{i+1} \leq X_t \\ min(J_{s,i}, J_{s,i+1}) & if \quad X_{i+1} > X_t \end{cases} \tag{2.13}$$

where J_s is the settling flux defined in eq. (2.9). In this region, the gravitational settling velocity of the particles is considered to be stronger than the upward movement. If the solid concentration in a layer is higher than an empirical threshold solid concentration, Xt, the settling flux will affect the rate of settling within adjacent layers.

- For the feed layer ($i = m$)

$$\frac{dX_m}{dt} = \frac{\frac{Q_f X_f}{A} + J_{clar,m-1} - (v_{up} + v_{dn})X_m - min(J_{s,m}, J_{s,m+1})}{z_m} \quad (2.14)$$

where Q_f and X_f represents feed flowrate and concentration, respectively. In this layer the bulk flow is considered to have both directions: upward at velocity v_{up} and downward at velocity v_{dn}. Fluid flows upward from the feed layer at the rate determined by the overflow and downward at the rate at which the thickened underflow is removed.

- For layers below the feed layer ($n - 1 < i < m + 1$)

$$\frac{dX_m}{dt} = \frac{\frac{Q_f X_f}{A} + J_{clar,m-1} - (v_{up} + v_{dn})X_m - min(J_{s,m}, J_{s,m+1})}{z_m} \quad (2.15)$$

the fluid is assumed to flow downward with a speed dependent upon the rate at which the sludge is removed. In fact, v_{dn} is the downward bulk fluid velocity that is equal to $\frac{Q_u}{A}$, being Q_u the sludge flowrate ($Q_u = Q_r + Q_w$).

- For the bottom layer ($i = n$)

$$\frac{dX_n}{dt} = \frac{v_{dn}(X_{n-1} - X_n) - min(J_{s,n-1}, J_{s,n})}{z_n} \quad (2.16)$$

where X_n is the concentration characterising the withdrawn sludge. This equation represents the second boundary condition (after the boundary condition on the top layer).

2.2.2 Mathematical formulation of the settling velocity

The *settling velocity* function used in this model is defined by a double-exponential equation proposed by Takács *et al.* (1991).

$$v_s = max(0, min(v_0', v_0(e^{-r_h(X - X_{min})} - e^{-r_p(X - X_{min})}))) \quad (2.17)$$

where
v_0 is the maximum theoretical settling velocity proposed by Vesilind (1968);
v_0' is the maximum practical settling velocity;
r_h characterises the hindered settling zone;
r_p characterises the settling behaviour at low solids concentration;

X_{min} is the minimum suspended solid concentration in the effluent and it is in turn related to the settler influent concentration in this way

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

where
f_{ns} is the non-seatleable fraction;
X_f is the solids concentration in the settler influent.
The Vesilind's velocity v_0 is one of the main parameter in the settler model that needs calibration. As it is described in Section 2.5.2.3 and 2.6.3, its value was estimated with the help of column test experiments performed on the activated sludge. The settling velocity is function of the different fractions of the sludge. Three sludge fractions are taken into consideration in the equation: unsettleable, slowly settling, rapidly settling fraction. Figure 2.4 shows the settling velocity how described by Takács *et al.* (1991), where four main regions may be identified (Holenda, 2006).

1. For $X < X_{min}$, the settling velocity is zero since in this case the concentration is under minimum achievable effluent SS concentration .

2. For $X_{min} < X < X_{low}$, the slowly settling particles dominate the settling velocity. As the solids concentration in the free settling zone of the settler gets higher the mean particle diameter also increases, resulting in a higher settling velocity (Patry *et al.*, 1992).

3. For $X_{low} < X < X_{high}$, the settling velocity is assumed indipendent of the solids concentration since the flocs reach their maximum size.

4. For $X > X_{high}$, the hindered zone is defined and the settling velocity becomes expressed by the Vesilind (1968) exponential equation.

2.2.3 Assumptions

The settler model is based on some assumptions outlined below (Stenstrom, 1975):

- every layer is assumed to be completely mixed;
- the concentration is considered uniform in any horizontal plane;
- no vertical dispersion is included;
- no biological reaction occurs;
- the solid flux is zero at the bottom of the settler;
- the mass flux into a differential volume cannot exceed the mass flux the volume is capable of passing, nor can it exceed the mass flux which the volume immediately below is capable of passing.

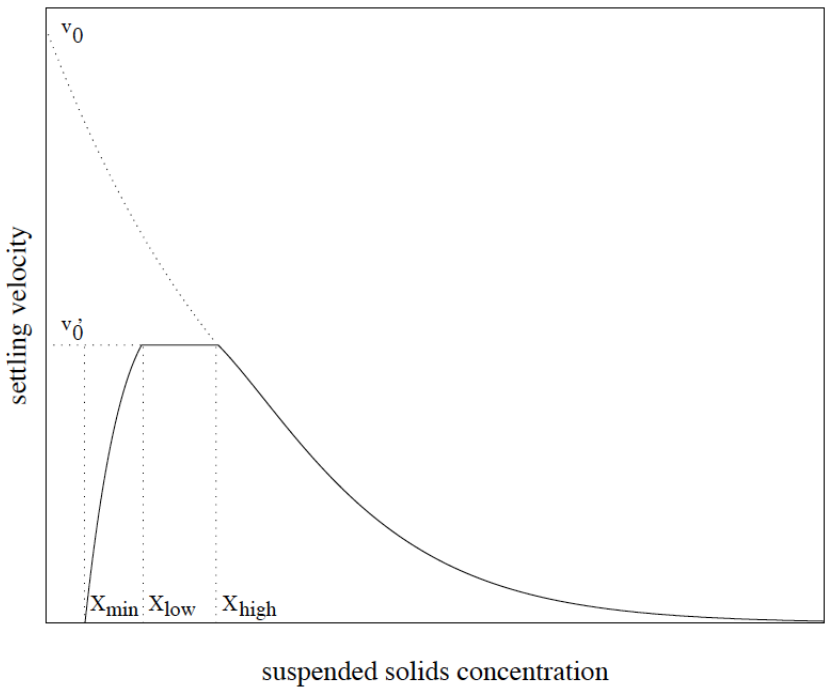

Figure 2.4: Double-exponential settling velocity model suggested by Takács al. *et al.* (1991).

2.3 The Benchmark Model Simulation 1 (BMS1)

After having outlined the Activated Sludge Model 1 and the Settler model, a description of how these two models are combined and implemented in a single simulation environment is given below, according to Copp (2002).
This simulation environment, the Benchmark simulation model 1 (BSM1), was developed between 1998 and 2004 by Working Groups of COST Action 682 and 624 (Alex *et al.*, 1999). This tool was built in response to the need to define a standard model implementation capable of helping design the wastewater treatment plant and evaluate the control strategies.
It is not linked to any particular simulation platform (Pons *et al.*, 1999).
It includes a plant layout, a simulation model, influent files, test procedures and evaluation criteria (Pons *et al.*, 1999).

2.3.1 Plant layout and simulations set-up

The design structure. A biological section and a secondary settler compose the model layout. Two internationally accepted models are used for the mathematical description of this plant set-up: the Activated Sludge Model 1 for the biological reactor and the 1-dimensional double-exponential settling velocity model for the secondary settler. All the biological and physical processes, the parameters, and variables having a role in the Benchmark environment were already defined before (see Sections 2.1 and 2.2). The biological section consists of 5 tanks in series, the first two are kept in anoxic condition (but fully mixed), whereas the other three in aerobic condition. The final settler is divided into 10 horizontal layers; the upper layer and the bottom layer have the same water quality as the effluent wastewater and as the recycled sludge, respectively. The 5th layer (placed at 2.2 m from the bottom) is the feed layer. The BSM1 in its default condition (before any adaptation to the Sjolunda wastewater treatment plant under study) fully characterises the layout features as follows (Alex *et al.*, 1999):

- total biological section volume of 5999 m^3 (tanks 1 and 2 each 1000 m^3 and tanks 3,4 and 5 each 1333 m^3);
- K_La 0f 10 hr^{-1} in tanks 3 and 4, and 3.5 hr^{-1} in tank 5;
- dissolved oxygen saturation of 8 gO_2m^-3 in all three aerobic tanks;
- a non reactive secondary settler with a volume of 6000 m^3 (depth of 4m);
- 2 internal recycles:
- nitrate internal recycle, from the 5th to the 1st biological tank (flow rate of 55338 m^3d^{-1})
- sludge recycle from the bottom of the secondary settling tank to the 1st biological reactor (flow rate of 18446 m^3d^{-1})
- wastage sludge flow rate of 385 m^3d^{-1}.

These features together with the influent characteristics and some parameters have to be modified and adapted to the specific real plant in question.

Control strategies

A basic control strategy is implemented in the BMS1 and reflects the most common control loops and sensing elements adopted in real plants. It aims to control two process features: the dissolved oxygen level in the final compartment of the reactor by manipulation of the oxygen transfer coefficient and the nitrate level in the last anoxic tank by manipulation of the internal mixed liquor recirculation.

Simulation procedure

The simulation procedure consists of two steps, a steady-state simulation followed by a dynamic simulation (Copp, 2002). For the different kind of simulations, three Simulink models are available:

- openloop.mdl simulates the plant without active controllers; it is used in order to check the simulation software;
- benchmarkss.mdl simulates the plant without noise, but with active controllers; it is used for the steady-state simulation;
- benchmark.mdl simulates the plant with active controllers and noise; it is used for the dynamic simulation.

The *steady-state* step describes the model in a long-term perspective. It is used before the dynamic simulation to reach constant conditions and eliminate the influence of the starting stage on the generated output. The input data file for the steady-state simulation is run for 100 days (or 10 times the sludge age). It describes the plant influent wastewater and is composed of daily values averaged over time.

The next step is the *dynamic simulation*. It is used to evaluate the plant performance in a short-term perspective, considering the expected diurnal and weekly variation (with a 15-minutes time step), in both quality and quantity. The dynamic influent file describes the wastewater composition variation during 14 days.

Influent file

The influent files are used in the model for characterizing the quality and quantity of the wastewater entering the plant. In Section 2.6.5, a description of the way the two files were constructed is reported. Each influent file contains a vector of the 16 components of the Activated Sludge Model 1 (described in Section 2.1):

$$t, S_i, S_s, X_i, X_s, X_{bh}, X_{ba}, X_p, S_o, S_{no}, S_{nh}, S_{nd}, X_{nd}, S_{alk}, TSS, Q_i$$

The time is given in days, the flow rate in $m^3 d^{-1}$, and the concentrations in $g\ m^{-3}$.
The Benchmark environment offers one input steady-state file and three dynamic files, but all of them need to be characterized with data deriving from the full-scale plant under study.

The *steady-state simulation* employs an input file—called constinfluent.mat—that describes the average wastewater composition; therefore, its values do not change over time.
The *dynamic simulation* employs a dynamic input file that describes the wastewater variation during the day and during the week with a default time step of 15 minutes.

2.3.2 Output offered

The Benchmark environment provides several kinds of outputs and performance assessments (Benedetti *et al.*, 2006):

- quality of effluent and other streams (recirculation flows, all five reactors, settler layers): concentrations and loads of all the pollutants considered in the input;
- Effluent Quality Index and Influent Quality Index: they are calculated from the effluent and influent compounds that have a major effect on the quality of the water;
- cost factors for operation:
  - sludge production to be disposed;
  - total sludge production: sludge to be disposed and sludge lost in the effluent;
  - aeration energy and pumping energy;
  - external carbon source;
  - mixing energy;
- controller output variations: offers the maximum and the variance of the variable variations
- overall cost index

2.4 High loaded activated sludge plant

Sjölunda wastewater treatment plant is operated in high-loaded condition. Indeed, it shows all thee typical features distinguishing the high-loaded plant operating conditions:

- high F/M, from 0.5 kgBOD(kgTSS-·m^3)$^{-1}$ in Autumn 2012, to 1 kg kgBOD(kgTSS·m^3)$^{-1}$ on the day of the measuring campaign.
- low solids concentration in the biological reactor, around 2600 mgTSS l^{-1} in Autumn 2012 and 2200 mgTSS l^{-1} in the week of the measuring campaign.
- very low sludge age, always lower than 2 days.
- low (compared to low-loaded plants) efficiency for carbon-based pollutants removal, around 85
- almost no nitrification.

According to Berta et al. (2003), the classification between different kinds of activated sludge plant operating conditions is made according to their F/M ratio and hydraulic retention time, as reported in Table 2.5.

Figure 2.5: Classification of activated sludge plant operating conditions according to Berta et al. (2003).

	F/M (kgBOD(kgMLVSS·m^3)$^{-1}$)	HRT (h)
Extended aeration	< 0.15	> 72
Low-loaded	0.2-0.3	10-72
Medium-loaded	0.3-0.5	5-10
High-loaded	> 0.5	< 5

Regarding high-loaded plants, an interesting case is the Hyperion Treatment Plant (HTP) located in the City of Los Angeles USA, which has implemented different levels of high-loading conditions, from 0.3 to 4 kg kgBOD (kgMLVSS·m^3)$^{-1}$ in its long history .

Four different operational conditions were experimented in HTP (summarised in Table 2.1 , according to Shao et al. (1992)).

Three of them are of particular interest and are reported here together with the related problem:

1. From year 1951 to 1960, the F/M ratio was kept between 2-4 kg kgBOD (kgMLVSS·m^3)$^{-1}$; the result was: i) high concentration of SS and BOD in the effluent, averaging around 60 and 50 mg l^{-1}, respectively; ii) strong Nocardia foaming problem on the aeration basin (1-2 m).
2. From year 1961 to 1986, the F/M ratio was lowered to around 0.5 kg kgBOD (kgMLVSS·m^3)$^{-1}$ for solving Nocardia foaming problems. As a consequence, the influent flow rate to the biological treatment facility had to be drastically reduced from 11.4 to 4.4 m^3 s^{-1}.

Table 2.1: Comparison between high-loaded activated sludge treatment plants.

	Los Angeles Increasing high-loading				Sjölunda
F/M [kgBOD(kgMLVSS·m^3)$^{-1}$]	0.3-0.4	0.5	1	2.0-4.0	0.5-1
MLSS [mgTSS l^{-1}]	1500-2500	1500	1000	300-500	2200
SRT [d]	4,8	3	1.5	0.2-0.5	¡2
Hydraulic loading rate on settler [$m^3m^2h)^{-1}$]	0.9	1.3	1.7	2.2-2.8	1.2
Solid loading rate on settler [kgTSS($m^2h)^{-1}$]	3.1-4	3	2	0.8	2.6
Effluent BOD [mgBOD l^{-1}]	5.-15	15	15	50	20-35
Effluent TSS [mgTSS l^{-1}]	5.-10	-	6	60	30-66
SVI [ml/g]	100-150	147	210	50	360

3. Around 1988, the operational conditions were changed in order to treat a higher flow rate and still meet the effluent quality limits; the F/M ratio was increased to 1 kg kgBOD (kgMLVSS·m^3)$^{-1}$. The results were: i) the disappearance of the foaming problems: Nocardia were washed out (even if a small amount of Nocardia was still observed microscopically) as a result of a longer MCRT for Nocardia than for mixed liquor (MCRT= mean cell retention time); ii) the BOD effluent concentration and the amount of sludge produced were equal to the case before; iii) SVI became higher, but no problems of sludge settling were detected thanks to the low solid loading rate on the secondary settler; iv) partial nitrification was still taking place; v) phosphorus removal was reduced.

2.5 Collection of data

A relevant step in the implementation of a model is the collection of information from the full-scale WWTP under study. Information is needed to gain a better understanding of the plant and its processes, to characterise the model, calibrate it, and if possible validate it. The set of information selected to be collected has been extracted from Peterson (2000) and adjusted according to the specific case of this thesis:

1. Design data: reactor and settler volumes and sizes
2. Operational data:
   - 2.1 Flow rates as averages and dynamic trajectories of influent, effluent, recycle and waste flows
   - 2.2 pH, aeration and temperatures
3. Characterisation of the settler model (e.g. zone settling velocities at different suspended solids concentrations)
4. Characterisation of the biological model (ASM1):
   - 4.1 Wastewater concentrations of full-scale WWTP influent and effluent (as well as some intermediate streams between the WWTPs unit processes), both as average values and as diurnal trajectories (e.g. TSS, VSS, COD, BOD, NH_4-N, NO_3-N, NO_2-N, Alkalinity, etc.)
   - 4.1 Sludge composition (e.g. TSS, VSS content)
   - 4.1 Reaction kinetics (e.g. growth and decay rates)

This information was obtained from three main sources: available historical data, previous literature, and most importantly experimental work.
This chapter describes how these data have been collected in practice and distinction is made between data already available and data gathered experimentally.

2.5.1 Available data

Besides data derived from literature (mainly for stoichiometric and kinetic parameters), the rest of data already available was provided by David Gustavsson, one of the process engineers at Sjölunda Wastewater Treatment Plant.
These include: design data, characterising the structure of the activated sludge plant (volumes and sizes of rectors) and historical data, defining the activated sludge treatment process (quality and quantity of wastewater, of mixed liquor, and of sludge).
The design data are reported in Table2.2 in the next Section.

The historical data are available from two kinds of sources: periodical laboratory analyses on grab and flow proportional samples and on-line sensors' recordings.
Laboratory analyses of the influent and effluent wastewater are conducted on flow proportional samples at regular times twice or three times a week, whereas analyses of the mixed liquor and sludge recycle are conducted on grab samples once a week.
These available laboratory analyses concern the standard pollutants:

- COD (filtered and non-filtered);
- BOD (filtered and non-filtered);
- TSS, Total Suspended Solids;
- VSS, Volatile Suspended Solids;
- NH_4-N, ammonium;
- $NO_{2,3}$-N, sum of nitrate and nitrite;
- total nitrogen.

Furthermore, on-line sensors provide information about:

- influent flowrate, every minute $[ls^{-1}]$;
- wastage sludge, every minute $[ls^{-1}]$;
- recycle sludge, every minute $[ls^{-1}]$;
- oxygen concentration in each of the three aerobic tanks, every minute $[mg\ l^{-1}]$;
- airflow, every minute $[m^3h^{-1}]$.

All these data have been employed to study the plant performance and the degradation processes occurring within the plant. Some were included into the model, some other were employed to attempt a model validation.
Instead, for the model calibration, these available data were not enough; for this reason, a measuring campaign and other laboratory experiments were necessary and were therefore carried out (see Section 2.5.2).

Out of all the historical information potentially available, two sets of data, related to two different time periods, have been used: autumn 2012 and the whole year 2005.
The first set (referred to as Autumn 2012) better reflects the current plant configuration, because it describes the plant in its present layout, which was modified before autumn 2012. Indeed, one of the original parallel equal treatment lines has been shut down, with direct consequences on the biological basin under study. Unfortunately, this set of data does not provide COD values, which are essential for the implementation of the model. Last time COD analyses were periodically carried out in routine laboratory work was in 2005, and that is why the second set of data is also taken into account.

Table 2.2: Design data of the biological reactors and secondary settler of the treatment unit under study

	Height m	Area m^2	Volume m^3
$1^{st} - 2^{nd}$ anoxic reactors	3.8	54.3	206.25
$1^{st} - 3^{rd}$ aerobic reactors	3.8	108.6	412.5
Total biolological reactor	3.8	434.2	1650
Secondary settler	3.8	467	1783

2.5.1.1 Design data

The design data required are volumes, surface and height of the biological reactors and secondary settler (reported in Table 2.2).

2.5.1.2 Operational data and characterization of the activated sludge section

In order to characterise the treatment process, information about mixed liquor, recycled flows, wastage sludge, oxygen concentration, and airflow blown are required. Figures illustrated below show data over Autumn 2012 and on the day and the week of the measuring campaign. Both scatter and box plot are shown. Box plots were chosen as a convenient way to graphically depict the great number of data available. They include the median, the first quartile, the third quartile, the mean (the yellow point) and the maximum and minimum values drawn as whiskers.

Mixed liquor quality

Total solids concentration in the biological basin is monitored once a week. Data are reported in Figure 2.6.

Mixed liquor recycle

The mixed liquor recycle is absent. The activated sludge section under study performs only carbon removal, while other downstream sections treat the nitrogen compounds; for this reason, there is no need to recycle the mixed liquor.

Sludge recycle quantity and quality

Sludge from the secondary settler is recycled back to the head of the biological basin. An on-line sensor continuously quantifies the flow, offering data every minute (Figure 2.9). Besides data from Autumn 2012, data from the measuring campaign week are illustrated as well (Figure 3.1). Laboratory analyses that assess the quality of the sludge recycle—in particular the total solids concentration—are carried out twice a week. Figure 2.6 shows TSS concentrations regarding data from Autumn 2012.

Wastage sludge

An on-line sensor measures the wastage flow and measurements are shown in Figure 2.11. Again, besides Autumn 2012, data from the measuring campaign week are illustrated as well (Figure 2.12). The reason for showing the whole week of the measuring campaign is linked to the calculation of the sludge age.

The sludge age reflects the quantity of sludge wasted, but changes with delay after the wastage sludge varies. The wastage sludge quality is the same as the one of the recycled sludge, which is reported in Figure 2.6 (data Autumn 2012).

Oxygen concentration

Oxygen concentration is measured by on-line sensors in all three aerobic sectors (Figure 2.7). Airflow is also continuously recorded (Figure 2.8).

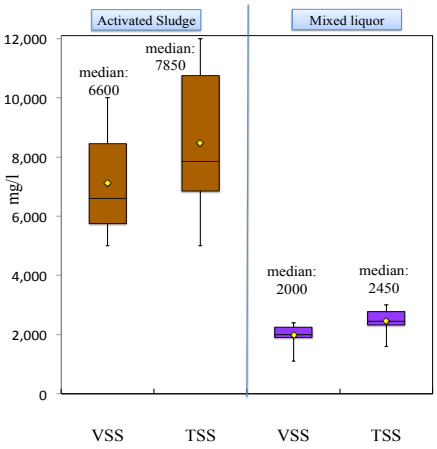

Figure 2.6: TSS and VSS concentrations in activated sludge and mixed liquor (Autumn 2012)

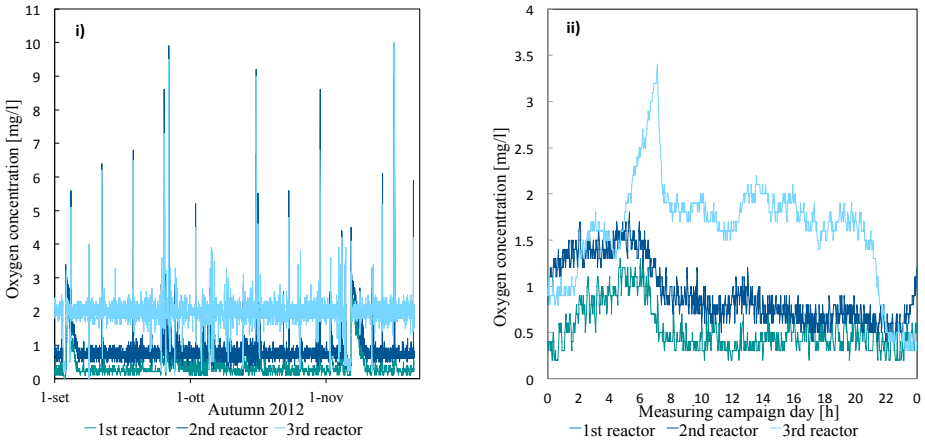

Figure 2.7: Oxygen concentrations in the three aerobic reactors. i)Autumn 2012; ii)Measuring campaign day

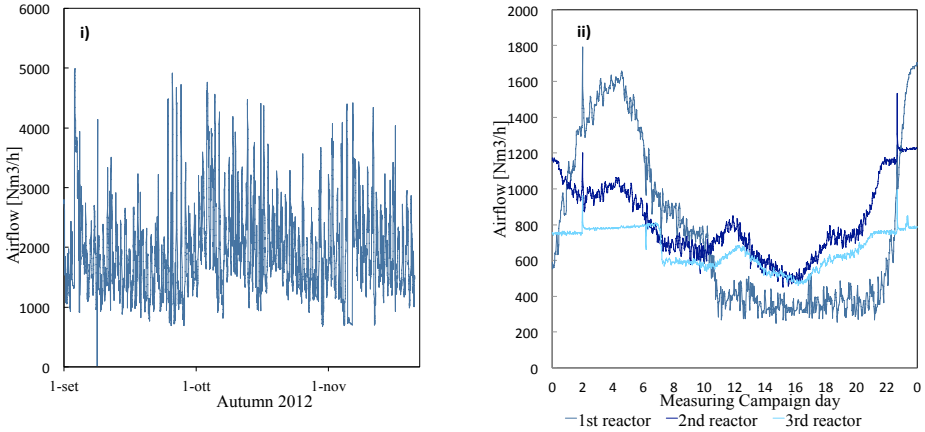

Figure 2.8: Airflow. i)Autumn 2012; ii)Measuring campaign day)

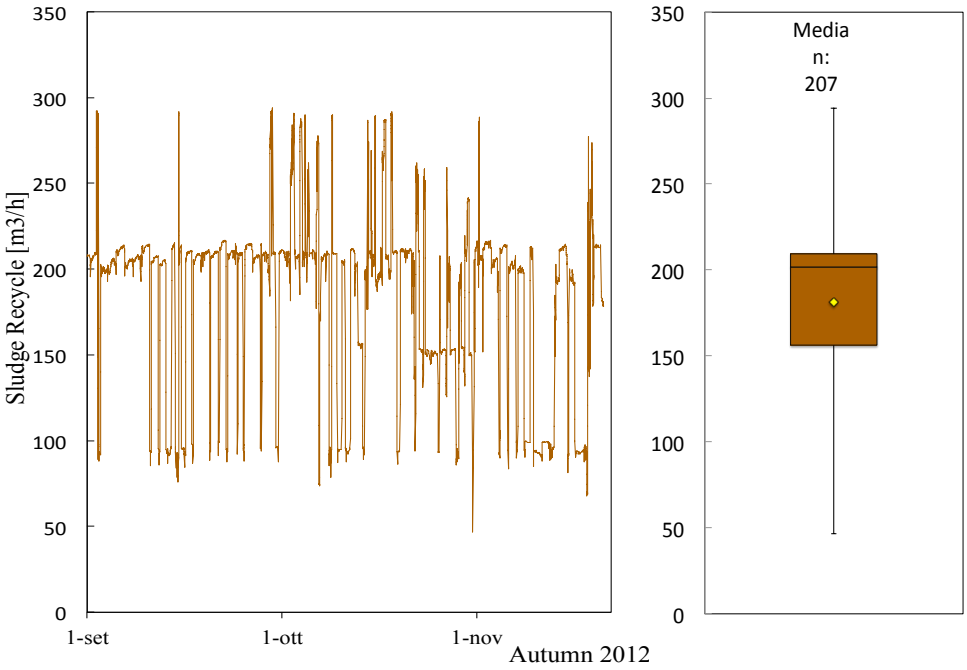

Figure 2.9: Sludge recycle flow trend during Autumn 2012

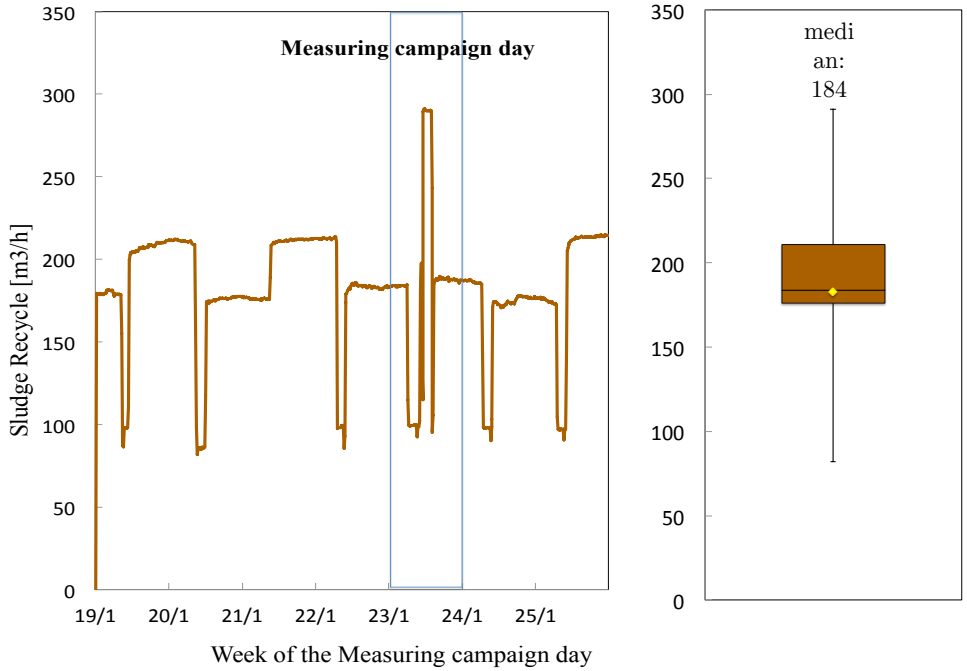

Figure 2.10: Sludge recycle flow trend during the week of the measuring campaign day

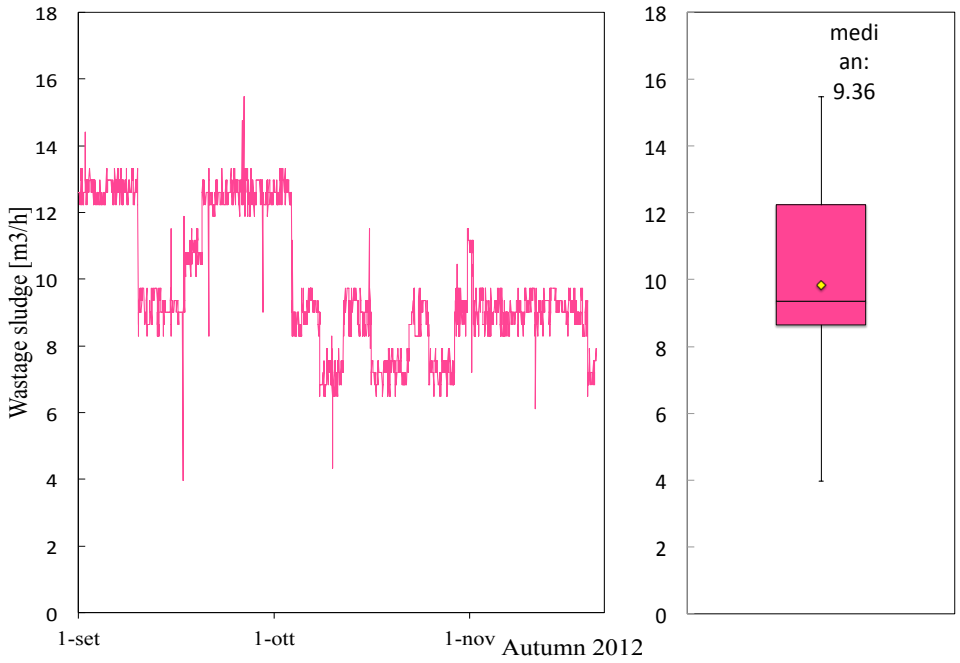

Figure 2.11: Wastage sludge flow trend during Autumn 2012

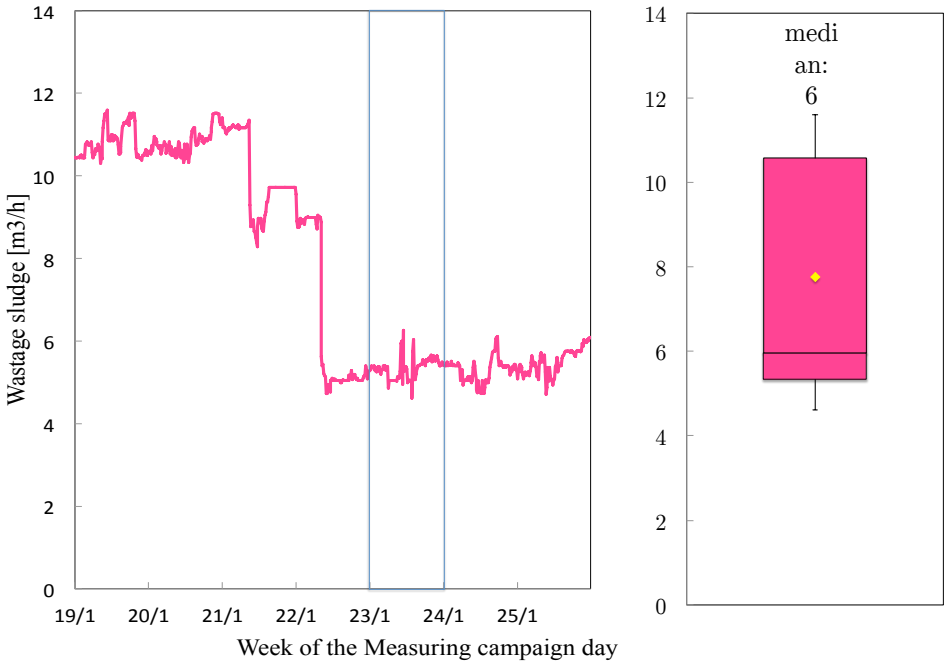

Figure 2.12: Wastage sludge flow trend during the week of the measuring campaign day

2.5.1.3 Influent and effluent data

As far as the quality of the wastewater is concerned, routine laboratory analyses provide only daily average values. In Figures 2.15 and 2.16 box plots for concentrations of the main pollutants in the influent and effluent are reported. The laboratory stopped performing COD analyses after 2005, therefore they are absent in the Autumn 2012 data.
Instead, the influent flowrate is continually measured by an on-line flow meter (shown in Figures 2.13 and 2.14).
These data are not enough for the calibration of the model, where the activated sludge treatment process is simulated in both steady-state and dynamic conditions. The two simulations require two different kinds of influent data. The steady-state simulation needs only a daily average characterisation of influent and effluent wastewater, whereas the dynamic simulation needs a dynamic characterisation, which implicates the definition of dynamic trajectories of the most common pollutants.
For this reason, it was chosen to carry out a measuring campaign (described in Section 2.5.2.1) and to employ those data for the whole model calibration phase.

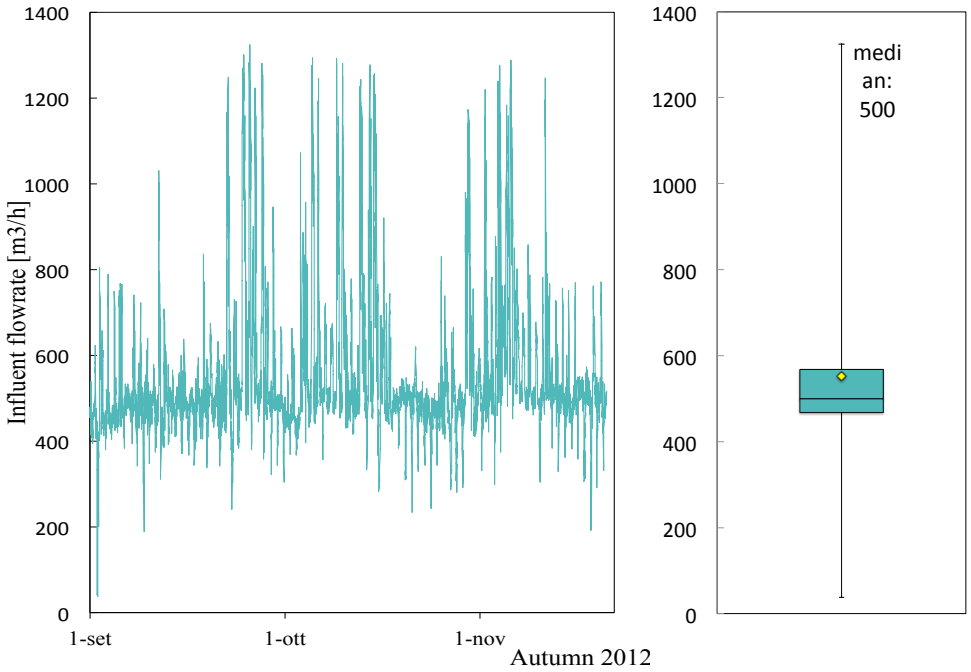

Figure 2.13: Influent flowrate trend during Autumn 2012

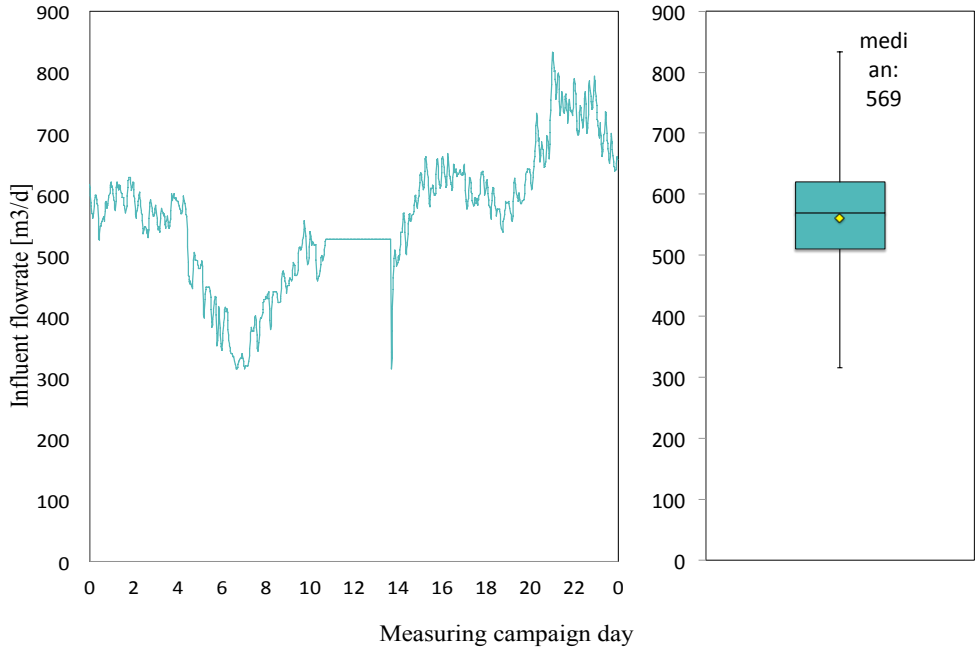

Figure 2.14: Influent flowrate trend on the measuring campaign day

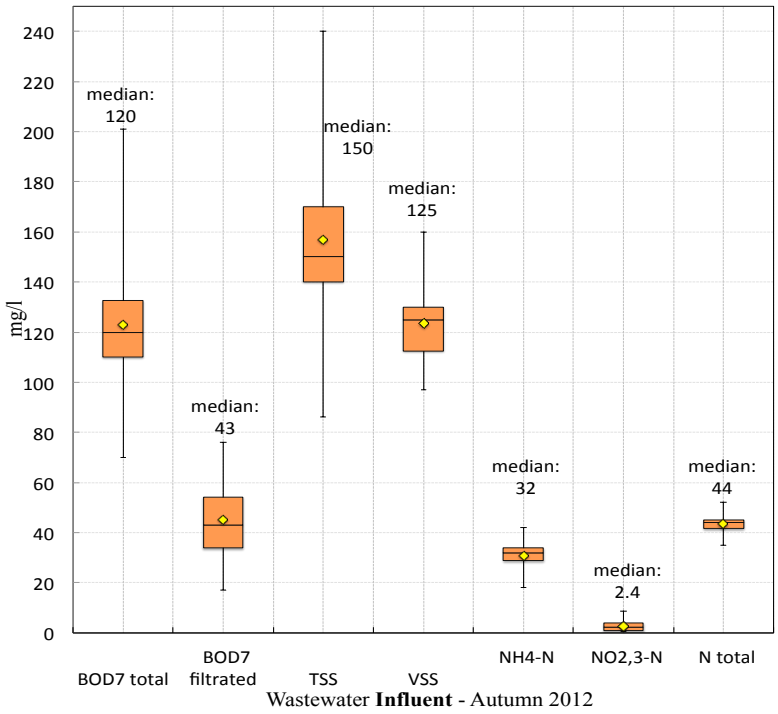

Figure 2.15: Influent wastewater quality, during Autumn 2012

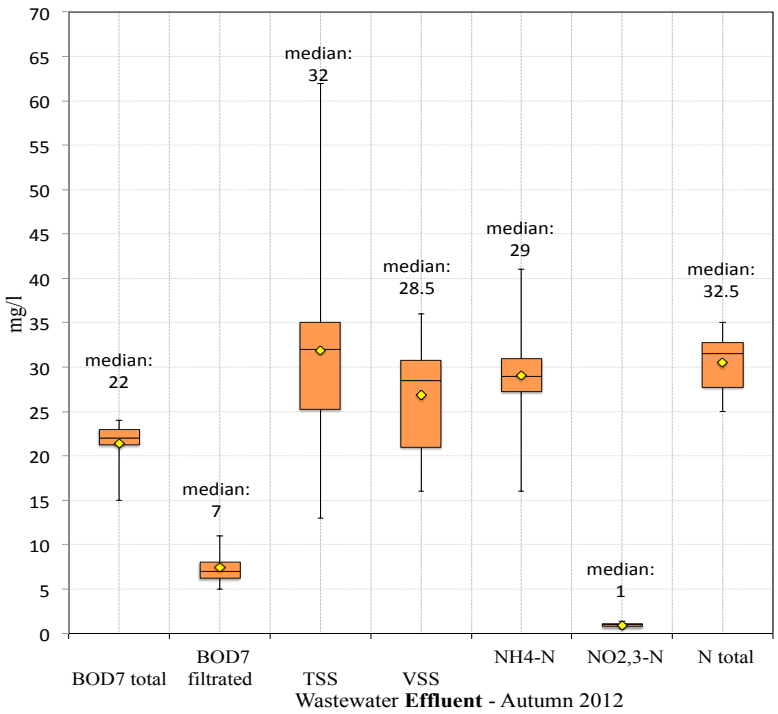

Figure 2.16: Effluent wastewater quality, during Autumn 2012

2.5.1.4 Process parameters

Three kinds of parameters are used in the Benchmark model: dynamic and stoichiometric parameters for the Activated Sludge Model 1, and settling parameters for the settler model.
Dynamic parameters
The values of dynamic parameters were obtained from different sources: some from literature, others from Benchmark default values, and further ones through laboratory experiments. Table 2.17 reports the chosen sources for each parameter. The laboratory experiments, from which some parameters were indirectly derived, are described in Section 2.5.2. The procedure used to elaborate the experimental results is explained in Section 2.6.2, and the final values, obtained after the calibration phase, are illustrated in Section 3.2.2.
Stoichiometric parameters
The stoichiometric parameters used in the model are the default Benchmark values, and are reported in Table 2.17.
Settling parameters
One of these parameters, v_o, has been gathered from the calibration procedure and the others from the article by Takács *et al.* (1991). This article reports some reference settling parameters suitable for a high loaded activated sludge plant. The laboratory experiment, from which the settling parameter v_o was indirectly derived, is described in Section 2.5.2.3. The procedure used to process the experimental results is explained in Section 2.6.3, whereas the final values, obtained after the model calibration phase, are illustrated in Section 3.2.2.

2.5.2 Data collected from experimental work

A series of data required for the correct implementation of the model was not available, thus it was necessary to carry out some experimental work in order to retrieve that information. Laboratory tests on wastewater, mixed liquor, and sludge recycle, taken from Sjölunda Activated Sludge Plant, were performed by the author of this thesis to collect as much experimental data as possible. In the model calibration phase, the experimental origin of parameter values has an important role in the reliability of the calibration; the higher number of parameter values of an experimental origin there are, the more reliable the calibration becomes. The experimental work carried out consisted of:

- a *Measuring campaign* to assess the trajectory of the influent and effluent wastewater composition during the day; data collected were employed for creating the input files required from the calibration of the model, for both the steady-state and dynamic simulation;

- the *Oxygen Uptake Rate test* (run two times) to gather information about the activated sludge; data collected were used to obtain three kinetic parameters: the maximum specific growth rate, μ_{maxH}; the decay rate coefficient, b_h; and the half-saturation concentration, K_s;

- the *Zone Settling Rate test* to assess some settling properties of the activated sludge and to obtain a first estimation of one specific parameter—the Vesilind's maximum theoretical velocity, v_o—required in the settler model.

Figure 2.17: Parameters required from The Benchmark Simulation Model 1.

Parameter		Source[a]	Value
Stoichiometric parameters			
Y_a [g cell COD formed.(g N oxidized)$^{-1}$]		BSM1	0.24
Y_h [g cell COD formed.(g COD oxidized)$^{-1}$]		BSM1	0.67
f_p [-]		BSM1	0.08
i_{xb} [g N.(g COD)$^{-1}$ in biomass]		BSM1	0.08
i_{xp} [g N.(g COD)$^{-1}$ in particulate products]		BSM1	0.06
Kinetic parameters			
μ_h [d^{-1}]		OUR	*
K_S [g COD.m^{-3}]		OUR	*
$K_{O,H}$ [g (-COD).m^{-3}]		BSM1	0.2
K_{NO} [g NO_3-N.m^{-3}]		BSM1	0.5
b_h [d^{-1}]		OUR	*
η_g [-]		BSM1	0.8
η_h [-]		BSM1	0.8
k_h [g slowly biodegradable COD.(g cell COD.d)$^{-1}$]		BSM1	3
K_X [g slowly biodegradable COD.(g cell COD)$^{-1}$] 6	BSM1	0.1	
μ_A [d-1]		BSM1	0.5
K_{NH} [g NH3-N].m^{-3}		BSM1	1
b_a [d-1]		BSM1	0.05
$K_{O,A}$ [g (-COD).m^{-3}]		BSM1	0.4
k_a [m3 (g COD.d)$^{-1}$]		BSM1	0.05
Settler parameters			
v'_o, maximum settling velocity [m.d^{-1}]		Takács	112.1
v_o, maximum Vesilind settling velocity [m.d^{-1}]		Z.S.V.	*
rh, Hindered zone settling parameter [m^3.(g SS)$^{-1}$]		Takács	0.000293
rp, Flocculant zone settling parameter [m^3.(g SS)$^{-1}$]		Takács	0.0027
fns, Non-settleable fraction [-]		Takács	0.00259

[a] Four different sources are possible= i) BSM1, it refers to the Benchmark Simulation Model 1 default values; ii) OUR, it refers to those parameter that have been indirectly estimated with the OUR Test, and subsequently calibrated through a sub-model; iii) Takács, it refers to the article by Takács et al. (1991), where settler parameters values for high-loaded plant are suggested; iv) Z.S.V., it refers to the Zone Settling Velocity Test, through which was obtained a first estimation of Vo.
* it stands for those parameter values that were determined experimentally (either through the OUR test or Zone Settling Velocity test) and will be reported in the Results chapter (see Section 3.2.2)

2.5.2.1 Measuring campaigns and consequent analyses

A measuring campaign was necessary in order to perform a dynamic simulation of the treatment process, since it requires the knowledge of the diurnal variation of the influent and effluent wastewater quality. The same set of data was used in the steady-state calibration, by averaging loads over the day.
The measuring campaign was carried out on a dry-weather day, on January 23^{rd} 2013, and lasted 24 hours. According to Ljung (1987), it should have a duration of 3-4 times the hydraulic retention time, which in the case under study is of 6 hours, leading to a monitoring time of 24 hours (Hulsbeek*et al.,* 2002 affirms that 24-h of sampling may be sufficient for most WWTPs).
Two samplers, each containing 24 sample bottles, were positioned at the inlet and outlet of the activated sludge section under study. The samplers were programmed to automatically collect 10 cl of wastewater every 6 minutes and fill one litre sample every hour.
A total of 48 samples, 24 for the influent and 24 for the effluent, were collected and then analysed. Each sample was analysed in the laboratory as soon as possible. Since there was no access to the WWTP between five p.m. and seven a.m., the samples taken during these times were not analysed until the morning after. The results of the measuring campaign are illustrated in Section 3.1.1.

Laboratory analyses
The following analyses were performed: COD filtered and non-filtered, BOD filtered and non-filtered, Total Suspended Solids, Volatile Suspended Solids, Nitrate, Nitrite, Ammonium, total Nitrogen, and Alkalinity.
Some of these were analysed every hour (COD filtered and non-filtered, TSS, VSS, Nitrite, Nitrate, and Ammonium), while others were analysed every two hours (BOD, total Nitrogen, and Alkalinity) for economical and practical reasons.
The filters used for the filtration are Munktell MGA Glass Fibre filters, with pore size of 1.6 micrometres.
COD analyses were performed using LCK 114 cuvettes, manufactured by Hach Lange in Dusseldorf, Germany. The results were read through a spectrophotometer, called Hach Lange DR 5000.
The BOD machine used was a Skalar Robotic Analyzer, mademanufactured by Skalar in Breda, Netherlands. The standard method used was the SS-EN 1899-2.
The Nitrogen analyzer was a FOSS FIAStar 5000, manufactured by FOSS Analytical in Höganäs, Sweden. The analyses of the soluble fractions, ammonium, nitrite and nitrate were performed after filtration. The following methods were adopted: SS-EN ISO 13395 for nitrate and nitrite; SS-EN ISO 11732: 2005 for ammonium; SS-EN ISO 14403-1:2012 for total nitrogen.
The alkalinity titrator was a TIM 840 Radiometer, produced by Bergman Labora in Danderyd, Sweden. The standard method for alkalinity is called EN-ISO 9963-2.

With respect to the measuring accuracy of the methods employed, an estimation may be: 10% for TSS, VSS, COD, all nitrogen analyses, and

alkalinity and 20% for BOD.

2.5.2.2 Oxygen Uptake Rate Test

This Section discusses why and how the Oxygen Uptake Rate test—also called Respirometric test—was carried.

The OUR test was performed because it offers the possibility of indirectly estimating three kinetic parameters of the heterotrophic metabolism required from the Activated Sludge Model 1: the maximum specific growth rate, μ_{maxH}; the decay rate coefficient, b_h; and the half-saturation coefficient, K_s. The values of these parameters are not found directly in the Test results; therefore, it was decided by the author to create a mathematical sub-model able to reproduce the process of the Test performed, including in the equations the three sought parameters.
From the calibration of the sub-model, some possible sets of values of the three parameters were obtained. These were then inserted in the full-scale model and further calibrated; the details of this procedure are explained in Section 2.6.2. A similar approach was implemented by Keskitalo *et al.* (2010) and Ciabini (2006).

The test is considered to be a valid modelling tool: a substantial indicator of activated sludge condition and of wastewater characteristics, finding wide application in studies that investigate and model activated sludge plants. It is a tool that measures and interprets the rate at which biomass consumes dissolved oxygen, the so-called respiration rate (Petersen, 2000). This important variable, the respiration rate, is measured using respirometers. Unlike other tests such as COD or BOD, the OUR test measures the evolution of oxygen consumption in time. Because oxygen consumption is directly associated with both biomass growth and substrate removal, respirometry is a useful technique for monitoring, modelling and controlling the activated sludge process (Vanrolleghem, 2002). Therefore, it is not a BOD or COD test alternative: it is employed to obtain biokinetic characteristics—as is done in this thesis—and it is considered one of the most important information sources in activated sludge process modelling (Vanrolleghem, 2002).
It is performed on samples of mixed liquor taken from the activated sludge reactor under study. The continuous oxygen monitoring offers information about the speed at which the biodegradable carbon is removed and consequently about the activity of the biomass present in the sample (since the oxygen consumption is directly linked to biomass growth). The biodegradable substrate, added to the sample to make it available to the biomass, might be either easily biodegradable carbon (e.g. methanol), or wastewater of the plant under investigation itself. With the latter option, the OUR Test is also used to gather information about the potential ability of the activated sludge plant to treat a defined wastewater influent (Xu and Hasselblad, 1996). Some authors affirm that this Test may also be used for plant performance control (Witteborg *et al.*, 1996).

In order to understand how the three selected kinetic parameters are linked to the OUR test, the oxygen uptake rate may be divided into two

components, according to Spansjers (1993):

- Exogenous respiration rate. It is associated to the oxygen used for the synthesis of new cellular material and it is linked to the oxygen consumption in presence of external carbon (it is zero when no substrate is present).

$$\frac{d_{ex}}{dt} = -\frac{(1-Y_H}{Y_H}\mu_{maxH}\frac{S(t)}{K_s+S(t)}X_{bh}(t) \tag{2.18}$$

- Endogenous respiration rate. It is associated to the oxygen consumed to ensure vital functions and the energy required and according to Spanjers (1993) it is linked to the oxygen consumption in absence of external carbon.

$$\frac{d_{end}}{dt} = -(1-f_p)b_h X_{bh}(t) \tag{2.19}$$

Experiment procedure and OUR respirograms interpretation
There are several types of respirometry apparatus (described by Spanjers *et al.*., 1998) and all of them are based on some technique for measuring the rate at which biomass takes up dissolved oxygen (DO) from the liquid. According to Vanrolleghem (2002) the different respirometres may be distinguished on the base of the phase in which the oxygen is measured (liquid or gaseous), and of the flow regime (batch or continuous flow). The equipment adopted in the laboratory at Lund University is based on the measurement of dissolved oxygen by an electrochemical sensor and is called the gas-static liquid method. The components of the experiment set-up used are:

- a glass reactor (2-litres capacity) covered with a lid equipped with several holes used for inserting: probes (for measuring dissolved oxygen, temperature and pH); a tube for the diffusion of air; reagents and carbon substrate dosage;

- a compressor, provided by Stellar, that insufflates air into the reactor through a silicone tube (3 *mm* diameter), which ends with a porous stone;

- a mechanical mixer, provided by IKA RW, that assures the constant mixing of the sample; it was adjusted to a speed of 200-300 rev / min and connected to an immersed stirrer;

- a probe that measures dissolved oxygen (HQd Probe series field kit provided by HACH Company);

- a probe that measures pH (WTW);

- a thermostat, provided by Lauda, that keeps the temperature at the chosen value, 15°C.

The OUR test was run two times, on two different days (20^{th} December 2012 and 28^{th} January 2013), and each time with two parallel tests to improve the results reliability.
The tests were conducted according to the guidelines in literature (Hagman and la Cour Jansen, 2007; Keskitalo *et al.*, 2010). The first experimental trial helped improve the final design: in the second experimental trial, some changes were made on the basis of the results of the previous test (how explained later). The mixed liquor samples (5 litre) were taken from the oxidation tank at Sjölunda plant, and transported to the laboratory, where pH, TSS, and VSS were determined.
The sludge samples did not need any dilution, since the solids concentration in the oxidation tank is around 2-3 gTSS l^{-1}, which is the range most frequently adopted for this kind of test in the consulted references. 15 ml of nutrient stock solution (NH_42SO_4 0.236 $g\ l^{-1}$, KH_2PO_4 0.044 $g\ l^{-1}$) and 4.5 ml of ATU, for inhibiting nitrification, were added to the respirometer reactors (each reactor contained a sample of 1.5 litre).
During the whole test, dissolved oxygen was constantly measured with a measuring frequency of 10 seconds.
Before starting the test, the activated sludge samples were agitated and aerated for about 2 hours. This pre-aeration phase eliminates the existing organic substrate in the sample—mainly produced by hydrolysis occurring during the transport stage (Keskitalo *et al.*, 2010)—and ensures the achievement of the endogenous phase. The two respirometers were kept in a water bath equipped with a thermostat, in order to maintain a constant temperature of 15°C; the temperature selected was the one measured in the Sjölunda activated sludge plant. Once the temperature and the oxygen concentration became constant, the test started and the aeration of the sample was no longer carried out continuously but intermittently, through cyclical activation and deactivation of the air compressor. In the first experimental trial each cycle lasted 10 minutes-5 minutes of aeration followed by 5 minutes of non-aeration—whereas, in the second trial each cycle was reduced to 6 minutes—3 minutes of aeration followed by 3 minutes of non-aeration.
In this way, the pattern of dissolved oxygen concentration became alternate: oxygen concentration increased during the aeration phase and decreased as soon as the aeration was switched off. In this last phase, the decrease of the oxygen concentration is almost linear (Hagman and la Cour Jansen, 2007); the slope of the oxygen curve was calculated and this value represents the OUR. value (Jukka *et al.*., 2010; Hagman and la Cour Jansen, 2007). The sequence of the OUR. values is called "respirogram" and describes the metabolic activity of the activated sludge at the defined conditions. The dissolved oxygen concentration was preferably maintained between an upper threshold, represented by the saturation conditions (variable between 8 mgl^{-1} and 9 mgl^{-1}), and a lower threshold fixed at 2 $mg\ l^{-1}$, in order to always operate under non-oxygen limiting conditions.
After 1 or 2 hours of alternate aeration, the endogenous respiration rate achieved constant values and the next part of the experiment could start. A known amount of carbon source (acetate) was added. The direct conse-

Table 2.3: Experimental conditions for the OUR Tests carried out.

	1^{st} Exp.[a]	2^{nd} Exp.[b]
Activated sludge volume (l)	1.5	1.5
Temperature (°C)	15	15
Acetate solution addition (ml)	6	12
Acetate COD addition (mgCOD l^{-1})	80	160
Maximum oxygen concentration recorded (mg l^{-1})	9.5	9

[a] The 1^{st} experiment was performed on 20/12/2012
[b] The 2^{nd} experiment was performed on 28/01/2013

quence is a greater and faster oxygen consumption, which suddenly produces an OUR increase. The acetate was added at a specific moment-which is the end of the aeration phase-in order to be able to detect the moment when the oxygen rate reached the greatest speed that is when the aerobic heterotrophic biomass is growing at the maximum specific growth rate (Hagman, 2007). The maximum oxygen uptake rate was so calculated. The acetate stock solution used as external carbon source had a COD concentration of 20000 $mg\ l^{-1}$. The amount of stock solution added to the respirometers was different in the two experimental trials: 6 ml, in the first (leading to a COD concentration of 80 $mg\ l^{-1}$); 12 ml, in the second (leading to a COD concentration of 160 $mg\ l^{-1}$).

These two changes—a double amount of acetate and a reduced cycle of aeration and non-aeration—were implemented in order to be sure that the maximum oxygen uptake rate was reached. Indeed, in the first experimental trial, the part of the curve describing the maximum oxygen uptake rate was not well defined (see the first graph of Figure 3.10 in Section 3.1.2).

After the maximum oxygen uptake rate was reached, the OUR returned to the endogenous respiration rate, as soon as all the organics were consumed. According to Gernaey *et al.,* (2001), the endogenous terms may be considered constant; this assumption permits the calculation of the exogenous respiration rate as the difference between the measured oxygen uptake rate and the endogenous respiration rate.
Table 2.3 summarises the experimental conditions of the two trials carried out. In Section 3.1.2, the respirograms obtained are shown.

2.5.2.3 Settling Column Test

The Benchmark Simulation Model 1, as explained in Section 2.3, implements two mathematical models: one for the biological reactor and the other for the secondary settler. The latter is based on the solid flux theory (Daigger, 1995) and contains the double exponential equation by Takács *et al.,* (1991), which includes several parameters. Among these parameters, the Vasilind's maximum theoretical settling velocity, v_o, may be experimentally determined through the Zone Settling Rate test (Vanderhasselt and Vanrolleghem, 2000), outlined in this Section.
The Zone Settling Rate test is generally performed on an activated sludge

sample and consists of monitoring the interface level displacement speed, thus obtaining the zone settling velocity (ZSV) of the activated sludge (Catunda and Vanhaandel, 1992).

The Vasilind theoretical settling velocity, v_o, needed for the model, cannot be directly defined though this test. This Section only describes the Zone Settling Rate Test, whereas the entire procedure to calculate v_o is outlined in Section 2.6.3. Briefly put,v_oo appears in the theoretical model provided by Vesilind, which links the zone settling velocity (ZSV)—determined by this test—to the sludge solids concentration. In order to determine the parameter v_o, according to Vanderhasselt and Vanrolleghem (2000), the dilution experiment was carried out: by running the test on several activated sludge samples, each with different sludge solids concentration, the Vesilind equation was experimentally investigated.

Experiment procedure

The experimental apparatus consisted of a transparent vertical cylinder, 60cm high and 10 cm in diameter, on the outside of which a calibrated millimetre tape was attached. The settling vessel is not equipped with a stirrer. A thermometer and a stopwatch are required.

As explained before, the test was performed several times—5 in total—for different sludge concentrations. In order to obtain the different solids concentrations, the 5 sludge samples—picked from the activated sludge reactor and from the sludge recycle channel—were diluted with treated effluent wastewater. The tests were run in loco, as soon as the samples were ready. For each sample, the temperature was monitored and the suspended solids concentrations analyzed. The sludge was vigorously mixed and poured into the settling vessel.

The high solids concentration, characterizing the activated sludge, allow the suspensions to settle in the zone-settling regime. After the sample was poured into the vessel, the suspension first agglomerated, forming a coarse structure with visible fluid channels, and then quickly delineated a sharp interface, made from sludge particles that settle at the same rate. This interface separates the clear supernatant liquor, in the upper part, from the concentrated sludge, accumulated in the lower part.

Using the stopwatch, the interface height was measured over time. When the interface approached the region of the concentrated sludge, its settling velocity decreased.

In the Results chapter (Section 3.1.4), the experiment results are illustrated.

To calculate the Zone Settling Velocity (ZSV) the position of the interface is plotted as a function of time at different solids concentrations. For each curve, the part considered as linear is selected and a best-fit straight line is drawn through those data points, whereas the starting and ending tails are excluded. The zone settling velocity (ZSV) is the gradient of each straight line.

2.6 Characterisation, parameterization and calibration of Benchmark Simulation Model 1

A fully developed calibration procedure for the Activated Sludge Model 1 is not available in literature. Therefore, various sources were checked to obtain an overview and plan the calibration steps to be followed in this study. It was decided to gather as much information from the full-scale plant as possible in order to help the framing of a realistic model parameter combination. The starting point of the characterisation of the model was the insertion of the information collected through historical data, laboratory work and literature. In this way, the model was initially characterised with the sizes of the plant structure, the process operation features, the quality and quantity of the influent wastewater, and the starting estimates parameters required by the mathematical model. Afterwards, some of these factors—mainly parameters, input wastewater quality fractions, and the wastage flow extraction value—were changed using the trial and error method until a good description of the measured data was reached.

In the following Sections, all the steps are described: initial insertion of the available data; creation of the input files to start the simulation with; processing the experimental results to obtain useful information to be included in the model; definition of realistic ranges of values of the calibrated parameters; identification of the most important process performances that the model should be able to reliably predict (the calibration phase will then focus on these aspects); and trial and error method application.

After having included the structural sizes of the biological reactors and of the secondary settler in the model, the model was characterised by including data describing the whole activated sludge treatment process. The set of data used derives mainly from the measuring campaign.
The *mixed liquor recycle* was removed from the Simulink model to reflect its absence in the real plant.
The *return sludge* flow was set to 25% of the main stream flow rate, which derives from the measuring campaign average value offered by the on-line flow meter.
The *wastage sludge* flow selection represented a problem: as Figure 2.12 (in Section 2.5.1.2) shows, the wastage sludge had been around 9 m^3h^{-1} till the day before the measuring campaign, which was carried out on January 23^{st} 2013. In the morning of the day before, the wastage sludge was decreased to values around 5 m^3h^{-1}, for process management needs. Therefore, the choice of the correct wastage sludge value to be inserted into the model and to represent the process on the measuring campaign day was uncertain. This variable has a great impact on the model outputs, since it reflects the sludge age. The sludge age does not change immediately with the variation of the wastage sludge, and for this reason, it was decided to select a value of Qw in between the range of 5-9 m^3h^{-1}.
The default *aeration system* of the Benchmark model was modified in order to meet the configuration at Sjölunda plant, which works differently (as de-

scribed in Section 1.1.3).
The control of oxygen concentration in the Benchmark model occurs only in the last biological reactor, unlike the Sjöulnda plant's aeration system, which monitors and controls oxygen concentration in all three aerobic reactors; to imitate the full-scale plant, the Simulink model was modified and the oxygen control was extended to all the three aerobic reactors.
Afterwards, three oxygen concentration set-points needed to be fixed, so that the aeration system model would keep the oxygen concentration in each reactor constant around the defined set-point values. Looking at the data measured by the three on-line oxygen sensors and selecting the average numbers, the three set-points become: 0.3, 0.8, 1.7 $mgO_2\ l^{-1}$. The model maintains the chosen set-points concentrations by changing the amount of pumped air.

2.6.1 Creation of the input file data

The input files characterise the wastewater entering the activated sludge plant. Through experimental analyses, the concentrations curves of the main pollutants over the day and the amount of wastewater were determined. To these values, theoretical relationships from literature were applied, in order to assign the right values to the variable building the model; this way, the first input files were created. Afterwards, they were calibrated using the trial and error method, enabling the model output to meet the effluent concentrations measured in the full-scale plant.

Two kinds of input files were developed. As explained in Section 2.3, *constinfluent.mat* is used for the steady-state simulation and *dryinfluent.mat* for the dynamic simulation in dry weather conditions. Both data files contain 16 columns referring to the 16 state variables that characterise the wastewater in the Activated Sludge Model 1. These state variables are not divided between filtered, colloidal and settleable wastewater fractions, which would have made their empirical determination easier.
Therefore, the first difficulty was to convert wastewater laboratory analyses into a data set that can be used as input for the model; of the 16 ASM1 state variables, the COD fractionation turned out to be especially problematic. Literature offers several theoretical relationships, but no standard procedure has yet been defined. The procedure chosen in this thesis and outlined below derives in part from Petersen (2000) and in part from reasonable assumptions made from the author and it combines physical-chemical and biological methods. Through this method, for characterising the COD fractions, the knowledge of total COD, COD filtered, total BOD, and total COD in the final Sjölunda effluent becomes necessary. This physical separation (filtered and non-filtered) may not be accurate since does not perfectly reflect the distinction between readily and slowly biodegradable (required from the ASM1). But according to Hulsbeek *et al.* (2002), the full-scale simulation model is not too sensitive for the division between slowly and readily degradable COD.

Conversion of wastewater data into model components
The 16 state variables, already defined in Section 2.1, are:

$$t, S_i, S_s, X_i, X_s, X_{bh}, X_{ba}, X_p, S_o, S_{no}, S_{nh}, S_{nd}, X_{nd}, S_{alk}, TSS, Q_i$$

COD components
The total COD balance is COD tot=$S_i + S_s + X_i + X_s + X_{bh} + X_{ba} + X_p$

- *S_i, Inert soluble non-biodegradable fraction*
  It leaves the system at the same concentration as it enters, and it is assumed not to be entrapped in the activated sludge system. For this reason, the COD found in the final Sjölunda plant effluent, after the whole treatment series, was considered to be composed for the 90% of inert soluble COD.
  S_i= 90% COD tot eff

- *S_s, Readily biodegradable fraction*
  It was considered to coincide with the filtered COD after having subtracted the inert soluble fraction. Levin *et al.* (1985) affirm that the right filtration pore size is 0.1 μm. The filter pore size used in the laboratory for this thesis is 1.6 μm, which can lead to an overestimation of the soluble readily biodegradable substrate concentration, assuming the definition of Levin *et al.* (1985) holds.
  S_s=COD inf filtr - S_i

- *X_i, Particulate non-biodegradable fraction*
  It was calculated from the COD mass balance, subtracting all the other COD fractions from the total (non-filtrated) influent COD.
  X_i= COD inf non-filtr - S_i - S_s - X_s - X_{bh} - X_{ba} - X_p.

- *X_s, Slowly biodegradable fraction*
  According to the study by STOWA (1996), the X_s fraction was estimated starting from the influent BOD and subtracting the S_s fraction.
  X_s= $BOD_7/(1\text{-}Y_h)$-S_s, where Y_h is 0.2.

- *X_{bh}, Heterotrophic biomass*
  It was assumed to be zero; according to Henze *et al.* (1995), the determination of the biomass could be difficult, and for this reason it may be included into X_s.

- *X_{ba}, Autotrophic biomass*
  This fraction was put equal to 0.5 mg l^{-1}. In literature, it is usually considered negligible. In the specific case in point, 5-10%, in terms of volume, of the influent of the activated sludge section is composed of the SBR treatment effluent, which contains X_{ba}.

- *X_p, Inert particulate*
  It was assumed to be zero, as suggested by Petersen (2000).

Nitrogen components
The total nitrogen balance is $N_{tot} = S_{no} + S_{nh} + S_{nd} + X_{nd}$

- S_{no}, *Nitrate+Nitrite nitrogen fraction* It coincides with the analysis of the sum of nitrate and nitrite
- S_{nh}, *Soluble ammonia nitrogen fraction* It coincides with the analysis of ammonium
- S_{nd}, *Soluble biodegradable nitrogen fraction* The estimation of this fraction is derived from the relation found in the Benchmark default input file. Considering that Snd + Xnd is equal to Ntot- Sno Snh and that in the Benchmark model Xnd = 1.524*Snd, Snd becomes Snd= (Ntot-Sno-Snh)/2.524
- X_{nd}, *Particulate biodegradable nitrogen fraction* It was calculated from the total nitrogen mass balance
- S_{alk}, *Alkalinity* It coincides with the alkalinity analysis.

This set of equations was selected before planning the measuring campaign, in order to direct the choice of which compounds of the wastewater needed to be analysed to allow the definition of the 16 state variables. Once all the measuring campaign analyses became available, the two input files were created.

Creation of the initial constinfluent file for the steady-state simulation
The constinfluent represents the input file for the steady-state simulation. It was obtained by applying the previous relationships to the measuring campaign analyses averaged over the day, thereby assuming that these averages represent a steady state. Afterwards, all the COD fraction were manually adjusted (see Section 2.6.4) in order to meet the measurements, whereas the nitrogen fractions were not modified.

Creation of the dryinfluent file for the dynamic simulation
The dryinfluent represents the input file for the dynamic simulation. It still contains the 16 columns related to the state variables, but each row, instead of daily average values, contains time-varying values according to the wastewater quality and quantity dynamic trajectory. The time step is 15 minutes; therefore, for each day, the number of rows describing the wastewater is 96 ($4 \cdot 24 = 96$). Replicating the daily file fourteen times, the dryinfluent file is created and it represents a two-week wastewater characterisation.

As just explained, for creating the dryinfluent, the 16 state variables need to be defined every 15 minutes, over the 24 hours. Clearly, from a practical point of view, analysing the wastewater for each pollutant every 15 minutes was not an option. To by-pass this difficulty, during the measuring campaign, wastewater was analysed every hour and through a linear interpolation, a 15-minute time step data were developed.
The pollutants were analysed with the following frequency: COD (filtrated and non-filtrated), TSS, VSS, Nitrite, Nitrate, and Ammonium every hour, while for economical and practical reasons, BOD, total Nitrogen, and Alkalinity every two hours.

This dryinfluent file was created once the constinfluent was already calibrated.

For nitrate, ammonia, and alkalinity, the measured data—and interpolated every 15 minutes—were used.
For the COD fractions, the calibrated fractions of the constinfluent file were calculated as percentage of total COD (percentage on loads). These percentage were applied to the COD (loads) dynamic data measured and interpolated on 15 minutes time-steps.

2.6.2 Initial estimation of dynamic parameters from OUR test

As explained in Section 2.5.2.2, the OUR test was performed because it offers the possibility of indirectly estimating three kinetic parameters of the heterotrophic metabolism required by the Activated Sludge Model 1: the maximum specific growth rate, μ_{maxH}; the decay rate coefficient, b_h; and the half-saturation concentration, K_s. In this Section, the procedure for obtaining an initial estimation of these parameters starting from the Test results is outlined.

It was decided to create a mathematical sub-model capable of reproducing the main biological processes occurring in the Test for simulating its overall behaviour. The equations building up the model contain the three sought parameters. The best-fitting parameters' values were determined through the calibration of this sub-model, as explained below. These values were then inserted in the full-scale BSM1 model as a starting point for a further calibration. A similar approach was adopted by Keskitalo *et al.* (2010) and Ciabini (2006).
The respirometer was modelled as a batch CSTR. The oxygen concentration curve and the oxygen uptake rate curves were the goal of the simulation and of the calibration. Therefore, the sub-model must include the alternate aeration and the addition of external biodegradable substrate.

For reproducing the main Test processes, the following set of differential equations have been included and solved through the forward finite differences method:

- The consumption of readily biodegradable substrate

$$\begin{aligned} \frac{dS_s}{dt} &= -\frac{1}{Y_h}\mu_H \frac{S_s}{K_s + S_s}\frac{S_o}{K_{O,H} + S_o}X_{bh} + \\ &+ k_h \frac{\frac{X_s}{X_{bh}}}{K_x + \frac{X_s}{X_{bh}}}\frac{S_o}{K_{oh} + S_o}X_{bh} \end{aligned} \tag{2.20}$$

- The growth of heterotrophic biomass

$$\frac{dX_{bh}}{dt} = \mu_H \frac{S_s}{K_s + S_s}\frac{S_o}{K_{oh} + S_o}X_{bh} \tag{2.21}$$

- The consumption of slowly biodegradable substrate

$$\frac{X_s}{dt} = (1 - f_p)b_h X_{bh} - k_h \frac{\frac{X_s}{X_{bh}}}{K_x + \frac{X_s}{X_{bh}}}\frac{S_o}{K_{oh} + S_o}X_{bh} \tag{2.22}$$

- The trajectory of dissolved oxygen

$$\begin{aligned} \frac{dS_o}{dt} &= \frac{1-Y_h}{Y_h}\mu_H \frac{S_s}{K_s+S_s}\frac{S_o}{K_{oh}+S_o}X_{bh} + (1-f_p)b_h X_{bh} + \\ &+ X_L a(S_{oSAT}-S_o) \end{aligned} \tag{2.23}$$

These four equations were extracted from the 13 equations of the Activated Sludge Model 1 (Henze *et al.,* 2000). In order to simulate the two OUR experiments carried out in the laboratory, the experimental conditions had to be included in the model:

- *The alternate aeration.* In the OUR Test, the aeration was provided intermittently, through cyclical activation and deactivation of the air compressor. In order to simulate this oxygen behaviour, the oxygen transfer coefficient, K_La—included in the last equation (2.23)—was selected as the key parameter. Its value was manipulated according to the presence or absence of aeration in the respirometer, allowing the cycles of aeration and non-aeration to be reproduced. The choice of the two K_La values derived from the calibration of the sub-model: 0.7 $min{-}1$, for the aerated phase; 0 min^{-1}, for the non-aerated phase. The value for the aerated phase is much higher than the default value found in the BMS1, because in the batch-laboratory experiment the oxygen transfer efficiency is greater compared to the full-scale plant. This differential equation also requires knowledge of the oxygen saturation concentration, S_{oSAT} . It was set equal to the maximum oxygen concentration recorded in the experiment during the aeration phases (slightly different for the two trials: 9.5 $mgO_2\ l^{-1}$, for the first; 9 $mgO_2\ l^{-1}$, for the second).

- *The acetate addition.* The acetate was included in the model as readily biodegradable COD fraction, S_s. S_s was set equal to 80 mgCOD l^{-1} when simulating the first experimental trial, and 160 mgCOD l^{-1} for the second trial (reflecting the two different concentrations characterising the two experiments). These values were assigned at the time instant corresponding to the acetate addition, while they were assumed to be zero at t=0.

- *The initial biomass concentration.* The biomass concentration and the death parameter were calibrated together in order to match the value of the endogenous respiration, as expressed by the following equation:

$$\frac{d_{end}}{dt} = -(1-f_p)b_h X_{bh}(t) \tag{2.24}$$

- *The other kinetic and stoichiometric parameters.* All the kinetic and stoichiometric parameters, except for the three parameters to be estimated, were set to their default values as in the Benchmark Simulation Model 1.

- *The inhibition of nitrification.* During the OUR Test, ATU was added to the respirometers, in order to inhibit nitrification. Therefore, the contribution of nitrification was not included in the model.

Some simplifications were made:

- the anoxic contribution to the hydrolysis of organic matter was neglected. First, its estimation requires knowledge of the nitrate fraction, Sno, which was not analysed; second, according to the Activated Sludge Model 1, the rate of hydrolysis under anoxic condition is lower compared to the one under aerobic condition.

- all the processes that refer to nitrogen fractions are neglected, because no nitrogen fractions were analysed throughout the experiment.

Sub-model calibration procedure

The sub-model was both calibrated and validated (the results are shown in Section 3.1.3).

The 2^{nd} experiment was used for the sub-model calibration, whereas the 1^{st} experiment for the sub-model validation. This decision started from the consideration that the 2^{nd} experiment may offer more information useful for the calibration, due to the presence of the plateau of maximum OUR values (as explained in Section 3.1.2).

The calibration of the sub-model was divided in two steps:

1. First, the endogenous part of the respirogram was calibrated, through the manual adjustment of the death coefficient, b_h, and the biomass concentration, X_{bh}. Once the sub-model output fit the endogenous respiration rate, the rest of the respirogram was evaluated.

2. Second, the exogenous respiration part of the respirogram was calibrated, through the manipulation of the maximum specific growth rate, μ_{maxH}, and the half saturation coefficient, K_s. μ_{maxH} was changed in order to meet the maximum height of the respirogram, where K_s was neglected, since the external substrate is not limiting at all. Afterwards, K_s was changed in order to fit two different aspects: the length of the plateau and the descendent curve of the respirogram, where the external substrate becomes a limiting factor for the heterotrophs growth. Simultaneously, the substrate degradation curve was also monitored, since the availability of external carbon had to be consumed before the end of the exogenous respiration, in a length of time of around 2 hours.

The storage phenomenon was neglected during the calibration.

To assess the simulation output, the Nesh efficiency coefficient—between the observed and the calibrated values for the OUR—was used.

2.6.3 Initial estimation of settling parameters from settling column test

The Vesilind maximum theoretical settling velocity, v_o, is one of the parameters included in the double exponential equation by Takács *et al.* (1991) used

for describing the settling velocity of the activated sludge in the secondary settler (and delineated in Section 2.2). As explained below, v_o was indirectly retrieved by employing the zone settling rate test (the test is described in Section 2.5.2.3).

The starting point was the theoretical model provided by Vesilind (1968), which links the zone settling velocity (ZSV)—determined by the test—to the sludge solids concentration and to v_o, the maximum theoretical settling velocity. To determine v_o experimentally, this relationship was developed by running the test on several activated sludge samples, each with different sludge solids concentrations, according to Catunda and Van Haandel (1992).

The Vesilind model (1968) is defined as follows:

$$ZSV = v_o \cdot e^{-kX} \tag{2.25}$$

where:
ZSV is the zone settling velocity obtained from the zone settling rate test;
v_o is the maximum theoretical settling velocity, which is the parameter to be defined;
k is a sludge settleabilty parameter;
X is the activated sludge suspended solids concentration.

The test was performed five times, each for different sludge solids concentration. The values of the natural logarithm of the zone settling velocities—obtained through the five tests—were plotted as a function of the sludge concentration and the best-fit straight line through the data points were drawn. This straight line approximates the Vesilind equation. Therefore, the two constants were finally determined: k is the straight line gradient and vo is equal to the intersection of the straight line with the vertical axis. Section 3.1.4 illustrates the experimental Vesilind relationship obtained and the calculated value v_o.

2.6.4 Calibration of the model

As soon as all required information was gathered and inserted into the Benchmark model, the trial and error method was implemented in order to achieve a better calibration. The main goal of this phase was to enable the model to produce outputs comparable to those measured in the full-scale plant. An essential step was to identify the goal of the calibration, that is the selection of the most relevant set of performance compounds of the full-scale plant that the model outputs needed to meet. It was decided to focus the calibration on four compounds considered as the most representative for plant performance: *TSS, COD, and BOD in the effluent and TSS in the biological reactor.* No nitrogen fraction were included in this selection, since the activated sludge plant does not perform nitrogen removal and consequently the impact on the influent nitrogen fractions is slight. During the calibration process, the effect produced by tuning the component of the model was assessed by monitoring the following:

- the four major performance compounds (TSS, COD, and BOD in the effluent and TSS in the biological reactor) that are the goal of the calibration;
- all the COD fractions and—to a slighter extent—nitrogen fractions, in the sludge recycle, and in the 5 biologic reactors, to understand the behaviour of each fraction through the different sectors;
- TSS in the various settler layers, to check the sludge blanket and the settling performance.

Four kinds of components were tuned through manual adjustment and their final calibrated values are reported in chapter 3:

1. *COD fractions of the influent file*
   In Section 2.6.1, the creation of the constinfluent input file starting from laboratory analyses and theoretical relationships was discussed. Once this file was created, some of its fractions—namely the COD fractions—were further calibrated. The total influent COD balance was kept, but its fractions were adjusted one by one. *The Particulate non-biodegradable fraction,* X_i*,* was reduced considerably to meet the solids concentration in the sludge. *The slowly biodegradable fraction,* X_s, was consequently increased to meet the balance of the non-filtered COD (X_s+X_i=COD non-filtered) Only the inert soluble non-biodegradable fraction, Si, was considered reliable enough and was kept the same as the first theoretical calculation: equal to the 90% of the total COD remaining in the final Sjölunda plant effluent, after having been through the whole series of treatments and consequently after most of the rest of COD had been degraded. The readily biodegradable fraction, S_s, was adjusted in order to meet the balance of the filtered COD (S_i+S_s=COD filtered) X_{ba}*, autotrophic biomass,* was set equal to 0.5 $mg\ l^1$ to take into consideration the SBR effluent that entering in the activated sludge section contributes with a small amount of autotrophic bacteria. X_p*, inert particulate, and* X_{bh}*, heterotrophic biomass,* were kept equal to zero.

2. *Kinetic parameters*
   While calibrating the kinetic parameters, the identifiability problem needed to be considered: identifiability is the ability to delineate a unique set of parameters capable of describing the behaviour of a system (Petersen, 2002). To avoid the problem of unidentifibility, it was necessary to define a range of possible values for each parameter. The ranges of the parameters proposed by Petersen (2002) were adopted as reference and are reported in Table—[I didn't include the table here]-.

   Not all the parameters were calibrated: most were default values, see Table 2.17; three (the maximum specific growth rate, μ_{maxH}; the decay rate coefficient, b_h; and the half-saturation concentration, K_s) were determined experimentally and then further calibrated. All of them were

adjusted to the temperature of 14°C, which is the average temperature of the measuring campaign day. For this purpose, the traditional Van'Hoff-Arrhenius formula was adopted, i.e.

$$k_{T^\circ C} = k_{20^\circ C} \cdot \theta e^{T-20} \tag{2.26}$$

where:
k is the parameter;
$k_{20^\circ C}$ is the parameter at 20°C;
$k_{T^\circ C}$ is the parameter value at the chosen temperature;
θ is the temperature coefficient (dimensionless), set equal to 1.03 (Bonomo, 2008);
T is the temperature, set to 14°C.

The calibrating procedure of the three kinetic parameters (μ_{maxH}, b_h, K_s) started from the OUR test and from the sub-model that was developed to reproduce the test (see Section 2.6.2). Different sets of values were proposed from that sub-model and all of them were tried in the full-scale model and finally the best-fit set was selected (and reported in Section 3.2.2).

3. *Settling parameters*
The final set of settling parameters was obtained by calibrating those offered by Takács *et al.* (1991) and the Vesilind maximum theoretical settling velocity, v_o, obtained experimentally.

4. *Wastage sludge flows*
As explained in Section 2.6.5, the choice of the wastage sludge was difficult, because its flow was reduced drastically the day before the measuring campaign (decreasing from an average of 9 to 5 m^3h^{-1}), leading to changes in the operational conditions. Since it is linked to the sludge age, its value was accurately selected. The sludge age does not change immediately with the variation of the wastage sludge, which choice became therefore critical. It was decided to select a value of Q_w in between the range of 5-9 m^3h^{-1}; through the trial and error method, the final value of Q_w was set to 7.9 m^3h^{-1}.

2.6.5 Validation of the model

A validation (Jørgensen and Bendoricchio, 2001) attempt of the model simulations was performed with a different data set from that of the measuring campaign (which were used for the calibration). Unfortunately, only daily average data were available, since no other measuring campaign could be performed. Historical data from the year 2005 and Autumn 2012 were employed. As explained in Section 2.5.1.3, Autumn 2012 data better represent the current plant configuration, but do not include COD analyses, which are necessary for the model implementation. To overcome this problem and still employ Autumn 2012 data, the ratio COD/BOD was taken from the average data of the year 2005 and applied to Autumn 2012 data, assuming that

the influent wastewater kept the same ratio over the years. The same ratio value was found in the measuring campaign analyses too, which confirmed the assumption.

The theoretical relationships outlined in Section 2.6.1, were applied to this new set of data and a new constant input file data was created and used for the validation. Results of the validation are reported in Section 3.3.

2.7 Model implementation: enhancement of the energy efficiency of the high-loaded plant at Sjölunda WWTP

To investigate different scenarios and study their effects on the plant performance, the use of the model becomes fundamental.

In this thesis, the application of the calibrated model has two main purposes: the first is the identification of the presence of energy and economic saving potential in the proposed configuration; the second is the evaluation of the actual possibility to run dynamically the plant, considering the diurnal variations.

The following optimization solutions investigated were selected as the most promising strategies to improve energy balance efficiency at a minimum initial investment cost.

2.7.1 Optimization solution I: Improvement of biogas production through anaerobic digestion of the biological sludge

The main goal of this Optimisation solution is the improvement of the energy balance of Sjölunda plant by increasing the biogas production. This aim was reached by increasing the load of biodegradable substrate extracted from the secondary settler, which is sent to the anaerobic digestion plant. The sludge age was manipulated and its effect on the quality of the wastage sludge investigated. Furthermore, this purpose has to be pursued without deteriorating the quality effluent of the activated sludge plant. Therefore, both aspects were taken into account, choosing the plant configuration that maximises the biogas production and minimises the effect upon the quality of the effluent. The following components of sludge and wastewater were selected for representing the biogas production and the effluent quality:

- *The sum of loads of the X_s (slowly biodegradable substrate) and the X_{bh} (active heterotrophic biomass) fractions in the wastage sludge*, for evaluating the potential biogas production. These two COD fractions were considered responsible of the biogas production, accounting for the biodegradable COD present in the biological sludge sent to the digester. The biodegradability of the two compounds was assumed to be equal, even if it is slightly different. According to the Activated Sludge Model 1 (Henze *et al.*, 1986), the X_{bh} fraction is almost completely hydrolysed to X_s, except for a minor part that instead is transformed in X_p (inert fraction). Since X_p is usually very small, it was neglected in this study; therefore, X_{bh} was considered to be totally transformed in X_s in the digester and to equally contribute to the biogas production.

- *The effluent COD concentration*, for evaluating the effluent wastewater quality.

In order to study the behaviour of these components, the model was dynamically run several times, each time with different sludge ages. Since the sludge

age is not a state variable of the model, in order to affect it, the *wastage sludge extracted* (Q_w) was manipulated instead.

The energy balance

For assessing the impact of the sludge age on the selected components, an energy balance was developed and used as comparison tool. The following components of the energy balance were evaluated:

- Estimation of energy produced from biogas. The biogas amount produced was quantified starting from the sum of loads of X_s and X_{bh} in the wastage sludge; the biogas produced was so calculated:

$$\text{Biogas production } [\frac{Nm^3}{d}] = \frac{kg(X_s + X_{bh})}{dx} \cdot \frac{0.42 Nm^3 biogas}{kg(X_s + X_{bh})} \quad (2.27)$$

  assuming that one kg of X_s and X_{bh} produces 0.42 Nm^3 of biogas. No value was found in literature that directly links the X_s and X_{bh} fractions to biogas production. Appels *et al.* (2008) reports an average specific gas production of 0.5-0.75 Nm^3/kgVSS loading (here, an average value of 0.6 Nm^3/kgVSS was chosen); Takács and Vanrolleghem (2006) estimate a X_s/VSS ratio of 1.42; according to these two aspects and considering that the same ratio is valid for X_{bh}/VSS as well (since it was assumed that X_{bh} is completely hydrolysed to X_s in the anaerobic digester), the specific gas production becomes 0.42 Nm^3biogas/kg(X_s+X_{bh}). Considering a calorific value of 6.25 kWh/m^3 and an efficiency of 85% (50% for heat and 35% for electricity production) according to Banks (2009), the energy becoming available is:

$$\text{Energy produced } [\frac{kWh}{d}] = \frac{Nm^3 biogas}{dx} \cdot \frac{6.25 kWh}{d} \cdot 0.85 \quad (2.28)$$

- Consideration of the amount of aeration energy consumed. This is an output value from the Benchmark model.

- Estimation of additional energy consumed for treating the increased effluent COD. This energy consumption was included in the balance because it accounts for the amount of COD that exceeds the current effluent COD concentration and consequently affects the quality of the water. Therefore, it was supposed that the increased amount of COD would need to be further treated in an activated sludge system and thus requiring additional energy. The electricity consumed for providing the right amount of oxygen, to degrade the increased COD, was calculated employing the Benchmark model.

Simulation procedure

The model was run dynamically 23 times, each time changing the wastage sludge extracted. For each dynamic simulation the following model output were monitored:

- X_s+X_{bh} loads in the sludge;

- COD concentration in the effluent;
- Aeration energy consumed;
- Aeration energy required for meeting the CODeff of the reference concentration;
- Other model outputs (BOD, TSS, COD fractions (S_s, X_s, X_i, X_{bh}) in the effluent, and TSS in the reactor).

The following outputs were plotted in two different kinds of graphs:

1. Daily average values for each parameter plotted against the sludge age;
2. Dynamic values for each parameter plotted over the day.

The first type of graphs allows evaluating plant average behaviour and the energy balance. The second type of graphs was needed to assess if the plant could actually run dynamically over the day, when the diurnal variation were considered.

2.7.2 Optimization solution II: Enhancement of pre-denitrification

The main goal of this optimisation solution is to improve the energy balance at Sjölunda Plant by enhancing the pre-denitrification. In the current Plant layout, nitrification and denitrification of the influent ammonium do not occur in the activated sludge system; consequently the existing pre-denitrification basin is not exploited. Instead, the activated sludge system is followed by Nitrifying trickling filters (NTFs) for nitrification and moving bed biofilm reactors (MBBR) for post-denitrification. In post-denitrification, methanol is added in order to provide biodegradable COD to the heterotrophic denitrifying bacteria. The methanol addition contributes to the plant management expenses. This is why, this Scenario investigates the possibility of enhancing the pre-denitrification, *by recycling a part of the Nitrifying trickling filter's nitrate-rich effluent (referred to as Qr_{NO_3})*. Therefore, the first purpose is to estimate the pre-denitrification potential.
Expected overall positive effects from implementing this Scenario:

- saved methanol addition in post-denitrification;
- saved aeration energy in the activated sludge aerobic reactors, due to degradation of part of biodegradable COD in pre-denitrification.

Expected overall negative effects from implementing the Scenario:

- pumping costs for recycling Qr_{NO_3};
- reduced hydraulic retention time in biological reactor and in secondary settler, which may lead to a deterioration of the settler effluent.

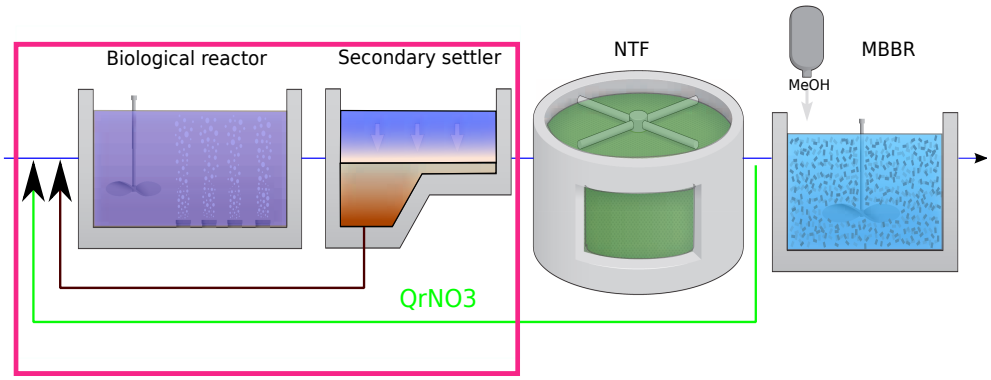

Figure 2.18: Layout configuration considered for recycling NTF effluent

The employment of the model aims firstly, to evaluate whether the implementation of this solution in the full-scale plant, could lead to an economic saving, without deteriorating the effluent quality; secondly, to investigate whether the potential of either the pre-denitrification or the settler represent the limiting factor to the amount of recycled wastewater. Indeed, not only the pre-denitrification, but also the secondary settler could be the critical point: if the settler is not able to handle the amount of recycled water, the effluent will be deteriorated.

Plant configuration considered for this solution In Section 1.1.3, the

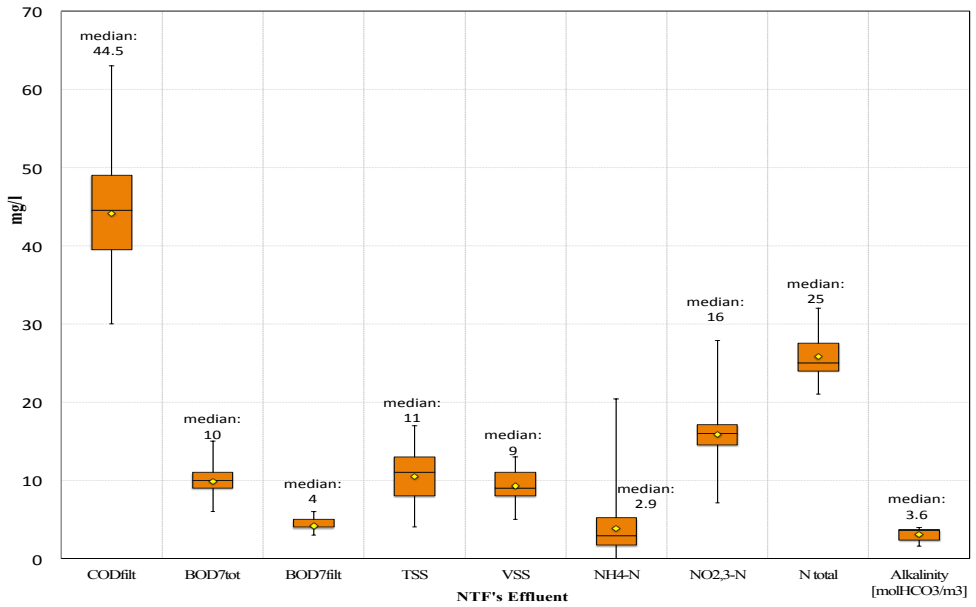

Figure 2.19: NTF effluent quality in January and February 2013.

current layout of the entire activated sludge system is described. In Autumn 2012, one treating line was shut down. This includes a settler volume (divided into 4 compartments) double the one in the section under study.

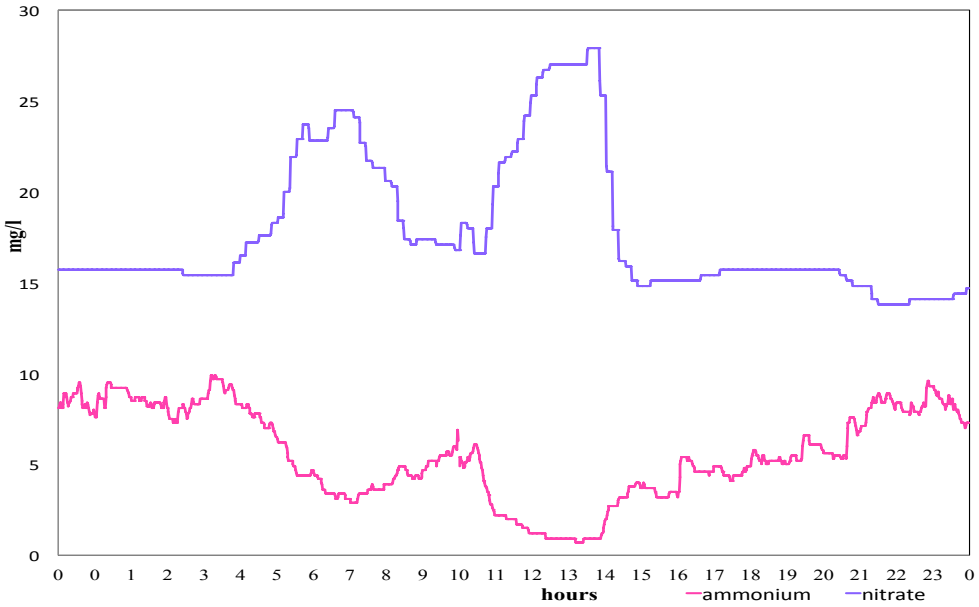

Figure 2.20: NTF effluent nitrate and ammonium on the day of the measuring campaign.

If the model simulation confirms the expected deterioration of the effluent, due to the hydraulic overloading of the secondary settler, the settler volume may be increased by employing one or more of the four settler compartments that are currently not in use. This new configuration would require only one new pump: the activated sludge is diverted in the new settler by gravity, but a pump is needed to provide the recycle of the settled sludge to the head of the biological tank.

From Figure 3.17 it is seen that the activated sludge plant is followed by Nitrifying trickling filters (NTFs) for nitrification and moving bed biofilm reactors (MBBR) for post-denitrification. Nowadays, all the wastewater treated in the activated sludge lines is collected and mixed before being distributed through the four Nitrifying Trickling Filters (NTFs). The effluent from the activated sludge section under study is therefore mixed with the rest.

The system considered in this study is different from the current Plant layout. It includes the activated sludge basin in point followed by a smaller NTF (which does not exist in the real plant), with the right volume to treat the amount of wastewater corresponding to a single activated sludge basin (the volume is around $\frac{1}{2}$ that of a single real NTF).

At the end of this Section, the idea of recycling the NTF effluent will be extended to the other three similar activated sludge basins in use, in order to estimate the total saving potential.

Current operation conditions of the plant

- NTF effluent. The daily mean quality of the NTF's effluent is shown in Figure 2.19. Nitrate and ammonium were continuously recorded by two on-line sensors positioned on the effluent of the NTF (data are shown in Fig.2.20)

- Mainstream nitrate

The mainstream is composed—in terms of volume—of 5-10% of SBR treatment effluent, which is rich in nitrite and nitrate. During the measuring campaign the trajectory of nitrite and nitrate over the day was monitored and is illustrated in Section 3.1.1.

Scenarios evaluated

Three different scenarios were investigated deferring in the way Qr_{NO_3} is pumped back to the head of the biological reactor. This aspect was taken into account for its impact on the activated sludge influent (referred to as A.S. influent), which assumes different characteristics according to the relative contribution of Qr_{NO_3}.
The three scenarios are the following:

- *Constant recycle scenario:* the influent flow rate to the Activated Sludge keeps the same variation as the mainstream over the day; Qr_{NO_3} is continuously pumped at a constant value (see Fig. 2.21).

- *Flow equalisation scenario*: the influent flow rate to the Activated Sludge is equalised (in terms of quantity). Therefore, Qr_{NO_3} is not pumped at a constant value; it is pumped according to the mainstream flow, in order to obtain a constant flow rate entering the activated sludge section (see Fig. 2.22).

- *Nitrate load equalisation scenario*: the influent nitrate/nitrite load to the Activated Sludge is equalised. Qr_{NO_3} is pumped according to the SBR effluent, rich in nitrate, in order to obtain a constant load of nitrate entering the activated sludge section. In order to create the influent file for the model capable of equalising the nitrate load, dynamic trajectory of nitrate concentrations had to be known, both in the mainstream and in the NTF effluent. Data of the mainstream were obtained during the measuring campaign; whereas data regarding the NTF effluent are continuously offered from a nitrate on-line sensor (see Fig. 2.23).

These three scenarios were not chosen only for their potential convenience from an operational point of view, but also for their concrete possibility of realistically implementing them in the full-scale plant:

- in the constant recycle scenario, the pump would work continuously at the same flow;

- in the flow-equalisation scenario, the pump would be automatically controlled by the flow meter already existing on the mainstream;

- in the nitrate load equalisation scenario, the pump would be automatically controlled by two nitrate on-line sensors already present in the plant: one on the NTF effluent, and one on the SBR effluent.

Procedure of scenarios investigation and of results evaluation

In order to study the behaviour of the activated sludge plant with the implementation of the nitrate recycle and to investigate the three selected scenarios, the model was dynamically run several times.

The investigation procedure consisted of gradually increasing the daily recycle flow rate Qr_{NO_3}, starting from a minimum value, and for each flow rate simulating dynamically the three scenarios.

As suggested by Copp (2002), all dynamic simulations followed a steady state simulation. This ensures a consistent starting point and should eliminate the influence of starting conditions on the generated dynamic output. For each recycle flow, and for each scenario, several outputs of the dynamic simulation were analysed, interpreted and compared with the reference case (which is the condition corresponding to the current operational plant conditions).
The output considered are:

- the concentration and load of nitrate in the effluent, for monitoring the pre-denitrification potential;
- the concentration and load of total suspended solids in the effluent, for monitoring the potential of the settler to handle the increased amount of water;
- the overall effluent quality for all the main pollutant parameters and their removed loads;
- the aeration energy consumed;
- the energy balance.

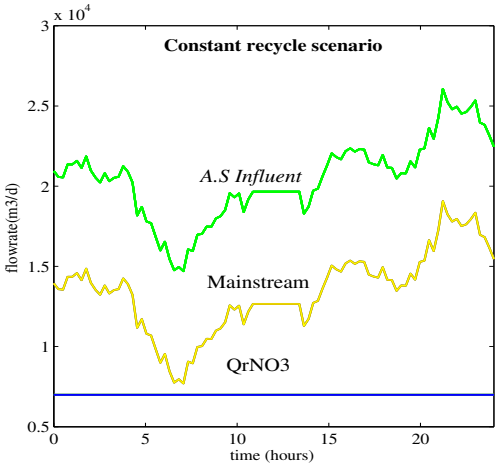

Figure 2.21: Constant recycle scenario. Qr_{NO_3} and mainstream are combined together before entering the activated sludge unit.

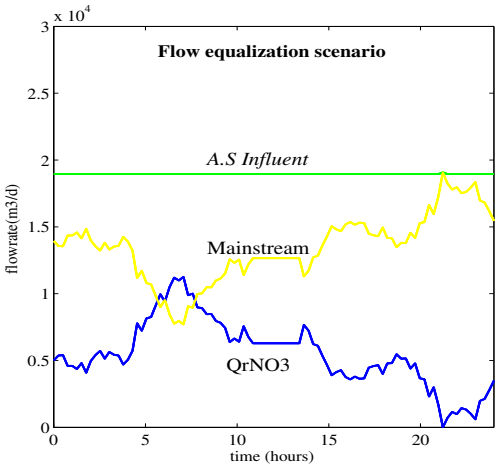

Figure 2.22: Flow equalization scenario. Qr_{NO_3} and mainstream are combined together before entering the activated sludge unit.

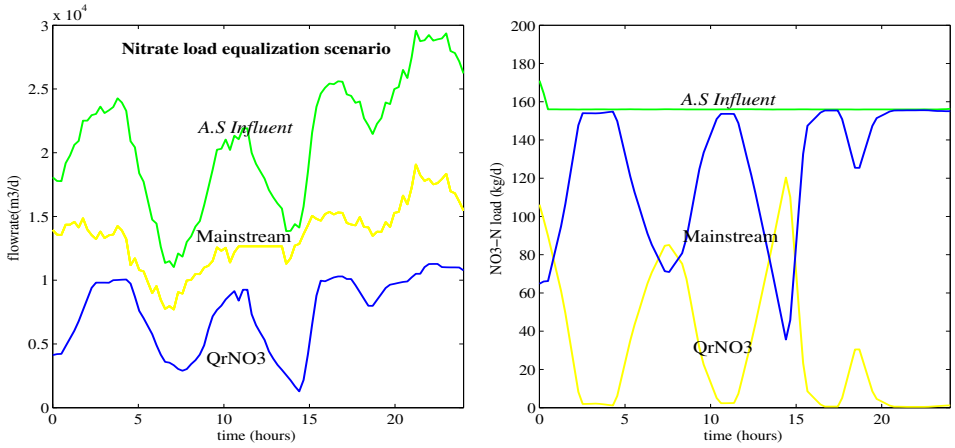

Figure 2.23: Nitrate load equalization scenario. Firs graph shows flowrates, the second nitrate loads of Qr_{NO_3} and mainstream, which are combined together before entering the activated sludge unit.

The energy balance

The energy balance was carried out in economic terms between the main contributors. Two savings components:

- Reduced methanol consumption. Once the maximum load of nitrate that the activated sludge system is able to treat was defined, the difference with the nitrate removed in the reference case gives the increased nitrate removed. The methanol saving was calculated assuming a dose of 4 kilos of methanol per kilo of increased nitrate removed and a cost of 0.44 € per kilo of methanol (as indicated by the process engineer at Sjölunda WWTP, David Gustavsson).

- Reduced aeration energy consumption. The amount of aeration energy consumed is calculated by the model in kWh over the day. The difference between the energy consumed in each scenario and the energy consumed in the reference case is the energy saved. The saved money is calculated applying an electricity cost of 0.12 € per kWh (as indicated by the process engineer at Sjölunda WWTP, David Gustavsson). The expense components:

- Additional energy consumption to pump the recycle flow. To estimate the pump cost, a pump was selected according to the total head and the flow rate Qr_{NO_3}; from the data of energy consumed per cubic meter [kWh/m^3], characteristic of the specific pump model selected, the daily expenditure was calculated (assuming an electricity cost of 0.12 € per kWh).

- Additional energy consumption in the event of increased settler volume.

If the settler appears to be hydraulically overloaded, the settler volume will be increased by pumping a part of the flow rate in the settler currently not in use. In this configuration the activated sludge can flow by gravity towards the new settler, whereas one additional pump is required for recycling the sludge to the head of the biological reactor. The pump cost was calculated as in the case before, selecting the right pump and calculating its energetic cost.

Assumptions on the quality of the recycle flow

An important step in the implementation of this optimisation solution is the definition of the quality of the recycle flow and the creation of the input file data to insert into the model. Several considerations and assumptions are necessary.

Determination of possible effects on the operational behaviour of the nitrifying trickling filter (NTF)

Lab data derived from the analyses of the current NTF effluent are available. Do these data describe also the recycle flow quality? It should be considered that when the new recycle is implemented, the overall system behaviour changes, with possible effects on the NTF effluent. Without a proper NTF model capable of predicting its behaviour, some assumptions and simplifications need to be made. The following lines explain the reasons to assume

that the NTF process and performance are not affected extensively, in the range of conditions taken under study.
Considering that:

1. the ammonium concentration in the NTF effluent is much lower than the one in the main stream (6 $mg\ l^{-1}$ compared to 38 $mg\ l^{-1}$), and it is also lower during summer;

2. in the current Plant process, the influent of the NTF is already recycled 2 times on the NTF itself (which means that a great amount of water enters and is diluted);

3. the main stream has a larger flow rate than recycle flow; then the NTF process would not be extensively affected and its effluent quality would not present relevant alteration. Therefore, the already available data characterising the current quality of the NTF effluent were used for creating the model input files. This assumption was confirmed by the expert judgment of Professor Jes la Cour Jansen, Chemical Department, Lund, Sweden.

Development of the input file data

Input data for the ***steady-state simulation***

Starting from the median values of the available laboratory data about the NTF effluent quality graphically showed in Fig.–, the input file data was created. It contains 16 values, according to the state variables of the Activated Sludge Model 1, which represents the average quality during the day:

$$t, S_i, S_s, X_i, X_s, X_{bh}, X_{ba}, X_p, S_o, S_{no}, S_{nh}, S_{nd}, X_{nd}, S_{alk}, TSS, Q_i$$

According to Petersen (2000), the following theoretical relationships were applied:

COD components

The total COD balance is COD tot=$S_i + S_s + X_i + X_s + X_{bh} + X_{ba} + X_p$

- S_i, *Inert soluble non-biodegradable fraction*
  It leaves the system at the same concentration as it enters, and it is assumed not to be entrapped in the activated sludge system. For this reason, the COD found in the final Sjölunda plant effluent, after the whole treatment series, was considered to be composed for the 90% of inert soluble COD.
  S_i= 90% COD tot eff

- S_s, *Readily biodegradable fraction*
  It was considered to coincide with the filtered BOD divided by (1-Y_h), where Y_h was set at 0.2, Benchmark default value. $S_s = \frac{BOD_7 filt}{1-Y_h}$

- X_i, *Particulate non-biodegradable fraction*
  Considering that X_{bh}, X_{ba}, and X_p were set to zero, this fraction was

estimated from the TSS mass balance, subtracting the Xs fraction from to the Total Suspended Solids. previously converted from TSS units to COD units.
$X_i = \frac{TSS}{0.75 - X_s}$

- X_s, *Slowly biodegradable fraction*
  According to the study by STOWA (1996), the X_s fraction was estimated starting from the influent BOD and subtracting the S_s fraction. X_s= BOD_7/(1-Y_h)-S_s, where Y_h is 0.2.

- X_{bh}, *Heterotrophic biomass*
  It was assumed to be zero, considering that the biomass is entrapped in the Trickling Filters.

- X_{ba}, *Autotrophic biomass*
  It was assumed to be zero, considering that the biomass is entrapped in the Trickling Filters.

- X_p, *Inert particulate*
  It was assumed to be zero.

Nitrogen components
The total nitrogen balance is $N_{tot} = S_{no} + S_{nh} + S_{nd} + X_{nd}$

- S_{no}, *Nitrate+Nitrite nitrogen fraction* This value was changed for every simulation. It derived from the weighted average calculated on the nitrate loads in the dynamic input file (which changes any time the recycle flow rate changes).

- S_{nh}, *Soluble ammonia nitrogen fraction* This value was changed for every simulation. It derived from the weighted average calculated on the ammonium loads in the dynamic input file (which changes any time the recycle flow rate changes).

- S_{nd}, *Soluble biodegradable nitrogen fraction* The estimation of this fraction is derived from the relation found in the Benchmark default input file. Considering that S_{nd} + X_{nd} is equal to $N_{tot} - S_{no}S_{nh}$ and that in the Benchmark model $X_{nd} = \frac{1.524}{S_{nd}}$, S_{nd} becomes $S_{nd} = \frac{N_{tot} - S_{no} - S_{nh}}{2.524}$

- X_{nd}, *Particulate biodegradable nitrogen fraction* It was calculated from the total nitrogen mass balance

- S_{alk}, *Alkalinity* It coincides with the alkalinity analysis.

Input data for the ***dynamic simulation***

For evaluating the three Scenarios the dynamic simulation results were compared. Every dynamic simulation requires the creation of a dynamic input file. As explained in Section 2.3, the dynamic file reproduces the variation of influent characteristics over the day. For the NTF effluent, only nitrate and ammonium concentration were monitored over the day by two on-line sensors, consequently only the dynamic trajectory of these two variables were inserted into the input file.
The file was created as follows:

- all the 16 variables, except for the nitrate and ammonia variables, were kept constant over the day, equal to the values calculated for the creation of the file for the steady-state simulation.

- the variation of nitrate and ammonia concentrations were inserted, according to the sensor recordings on the day of the measuring campaign;

- the flow rate was changed for every simulation.

Different dynamic input files were created for the three scenarios and were changed for each daily amount of recycle flow Qr_{NO_3} investigated. The three types of scenario input files mainly differ in the flow rate trajectory over the day.

- For the Constant recycle scenario, the flow rate values are set equal to a constant value that is repeated for all the 15-minutes time-steps along the day. And for each flow rate evaluated, this number is changed.

- In the Flow equalisation scenario, Qr_{NO_3} is pumped in a way that equalises the influent flow rate to the activated sludge section. The Qr_{NO_3} flow rate values are calculated as the difference between the total amount of flow rate that is selected to enter in the activated sludge section (which is a constant value) and the mainstream flow (which varies over the day). This difference is made for each time-step. The total amount of flow rate that enters the activated sludge section is changed on each simulation trial.

- In the Nitrate load equalisation scenario, Qr_{NO_3} is pumped in a way that equalises the influent nitrate/nitrite load to the activated sludge section. This means that Qr_{NO_3} is pumped according to the SBR effluent, rich in nitrate. Therefore, in each simulation the constant value of nitrate load that enters in the activated sludge is selected; for each time-step, the nitrate load already contained in the main stream is calculated; the difference between the nitrate load selected and the nitrate load in the mainstream, gives the nitrate load that the recycle flow Qr_{NO_3} has to contain in each time-step. Consequently the Qr_{NO_3} flow rate values are obtained by dividing each time-step load for the concentration of nitrate in the same time-step.

3 Results and Discussions

3.1 Data collected from the experimental work

3.1.1 Measuring campaigns analyses

In this Section, the results of the measuring campaign analyses are outlined. Figures 3.1 and 3.2 picture the dynamic concentration trajectories of the influent and effluent pollutants. Figures 3.3 and 3.4 summarise with box plots data of each pollutant, through median, average, 1^{st} and 3^{rd} quartiles, maximum and minimum values.

The influent trends are used for the creation of the dynamic input file necessary for the dynamic simulation. The effluent trends are employed for the evaluation of the model accuracy, comparing them with those obtained from the dynamic model simulation; instead, their average daily values are used during the steady-state calibration.

It is interesting to take a look at the nitrate influent trend: its pattern reflects the behaviour of the SBR plant, which provides the main contribution to the nitrate load in the mainstream, and operates three cycles per day. As reported in Table 3.2, almost the whole influent nitrate load (30 kg NO_3-N d^{-1}) is denitrified in pre-denitrification; indeed, its effluent concentration is zero. This removal capacity is investigated for improvement in the optimization solution II, in Section 2.7.

The measuring campaign was carried out on 23^{rd} January 2013 and some of its results are outside the ranges characterising January and February wastewater quality.

This is illustrated in Figures 3.5, 3.6, and 3.7, where blue dots represent the median values of the measuring campaign results, whereas the box plots show the ranges of January and February laboratory analyses performed on mean daily flow proportional samples.

Above all, the median value of total suspended solids concentration in the effluent wastewater on the day of the measuring campaign is much higher (66 mg TSS l^{-1}) than the whole range of data characterising January and February (whose median value is 22 mgTSS l^{-1}); whereas its influent value is comprised in the average ranges. This means that low TSS removal efficiency characterizes that specific day, differently from the rest of the month.

A first possible justification may be that the day before the monitoring campaign, the wastage sludge extraction was consistently reduced by the process engineers of the plant, possibly provoking a sludge loss from the settler.

Table 3.1: Average operational conditions on the measuring campaign day

Influent flow rate	$m^3 d^{-1}$	13449
Temperature	°C	14
Mixed liquor solids concentration	mg SS l^{-1}	2200
Sludge solids concentration	mgTSS l^{-1}	9000-10000
Sludge age	d	1.3-1.4
Sludge load (anox+aerobic)	kgBOD(kgSS d)$^{-1}$	0.72
Sludge load (aerobic only)	kgBOD(kgSS d)$^{-1}$	0.96
D.O. 1^{st} reactor	mg l^{-1}	0.5
D.O. 2^{nd} reactor	mg l^{-1}	0.8
D.O. 3^{rd} reactor	mg l^{-1}	1.7

Table 3.2: Influent, effluent and removed loads (kg d^{-1}) on the day of the measuring campaign. The last row presents the removal efficiency for each pollutant parameter.

Parameters	CODtot	CODfilt	BODtot	BODfilt	TSS	VSS	NH_4-N	NO_3-N	Ntot
Influent	6135	2865	2607	1164	2063	1624	561	31	842
Effluent	2112	1135	485	148	899	686	515	0.4	620
Removed	4023	1730	2122	1016	1163	938	46	31	223
Percentage	0.66	0.6	0.81	0.87	0.56	0.58	0.08	0.99	0.26

Indeed, Fig. 3.7 shows how the wastage extraction on the monitoring day is lower than the average values.

A second reason could be the different way of sampling wastewater: the monitoring campaign was carried out on time proportional samples, whereas the routine laboratory work is always performed on flow proportional samples, which may lead to a reduced pollutant concentration as a consequence of a bigger flow dilution.

Table 3.1 reports the average operational conditions measured and calculated on the day of the measuring campaign. All of them confirm the high-loaded condition of the activated sludge plant: i) the sludge age is low, between 1.3 and 1.4 days; the solids concentration in mixed liquor is also low, around 2200 mg l^{-1}, leading to iii) a high F/M ratio: 0.72 kgBOD(kgSS d)$^{-1}$, if considering the whole biological reaction, and 0.96 kgBOD(kgSS d)$^{-1}$, when only the aerobic sectors are taken into consideration. The solids concentration in the sludge varies considerably on a day-to-day basis and on the measuring campaign day it is around 9000-10000 mgTSS l^{-1}.

The removal percentage for each pollutant parameter was calculated over the day and their average values are reported in Table 3.2, together with influent and effluent loads.

It appears clear that nitrification does not occur, how expected from a high-loaded activated sludge plant with a sludge age lower than 2 days. Indeed, the ammonium removal efficiency is very low: only 46 kg out of a influent load of 561 kg d^{-1} are removed.

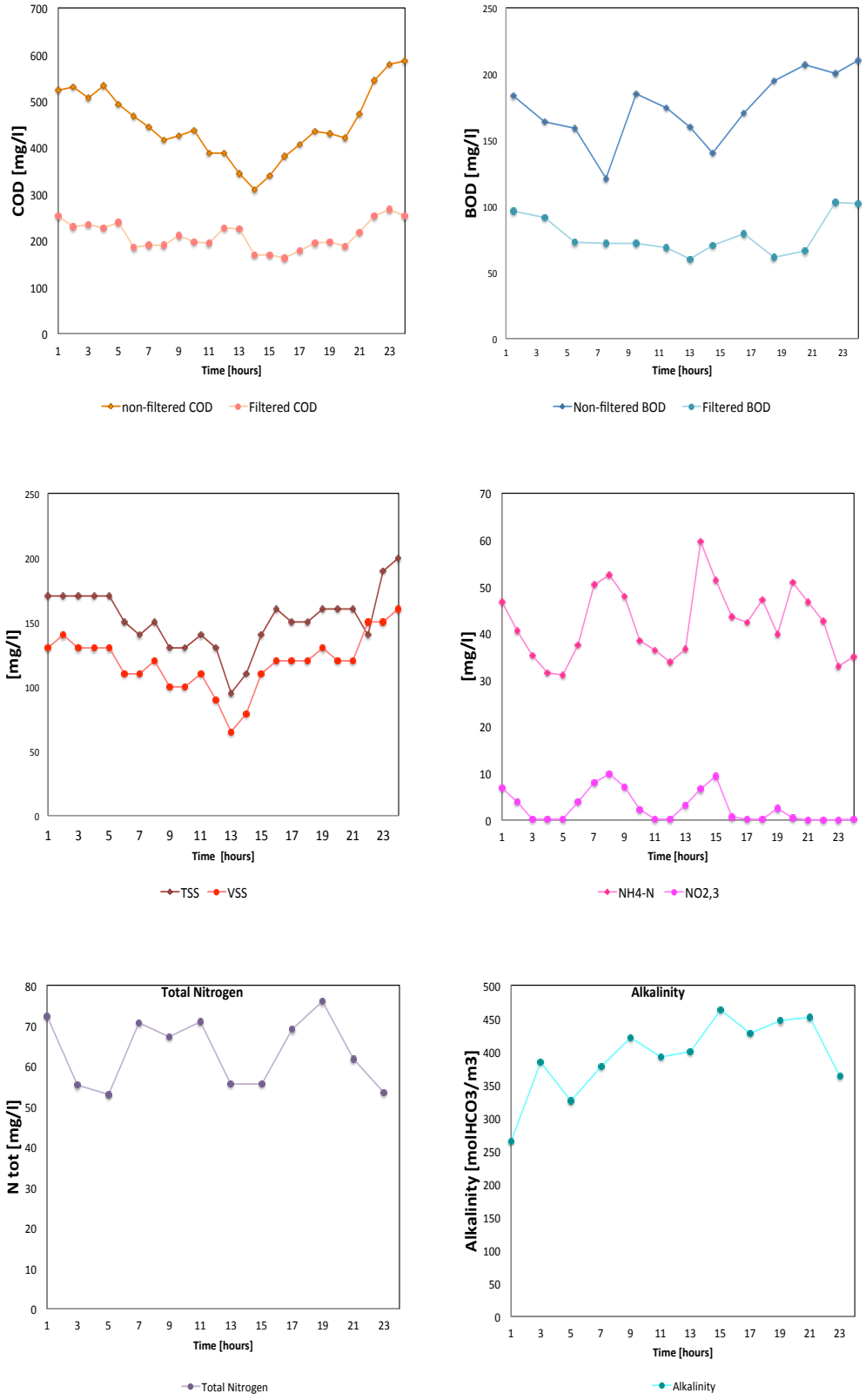

Figure 3.1: Influent wastewater quality trends measured during the measuring campaign

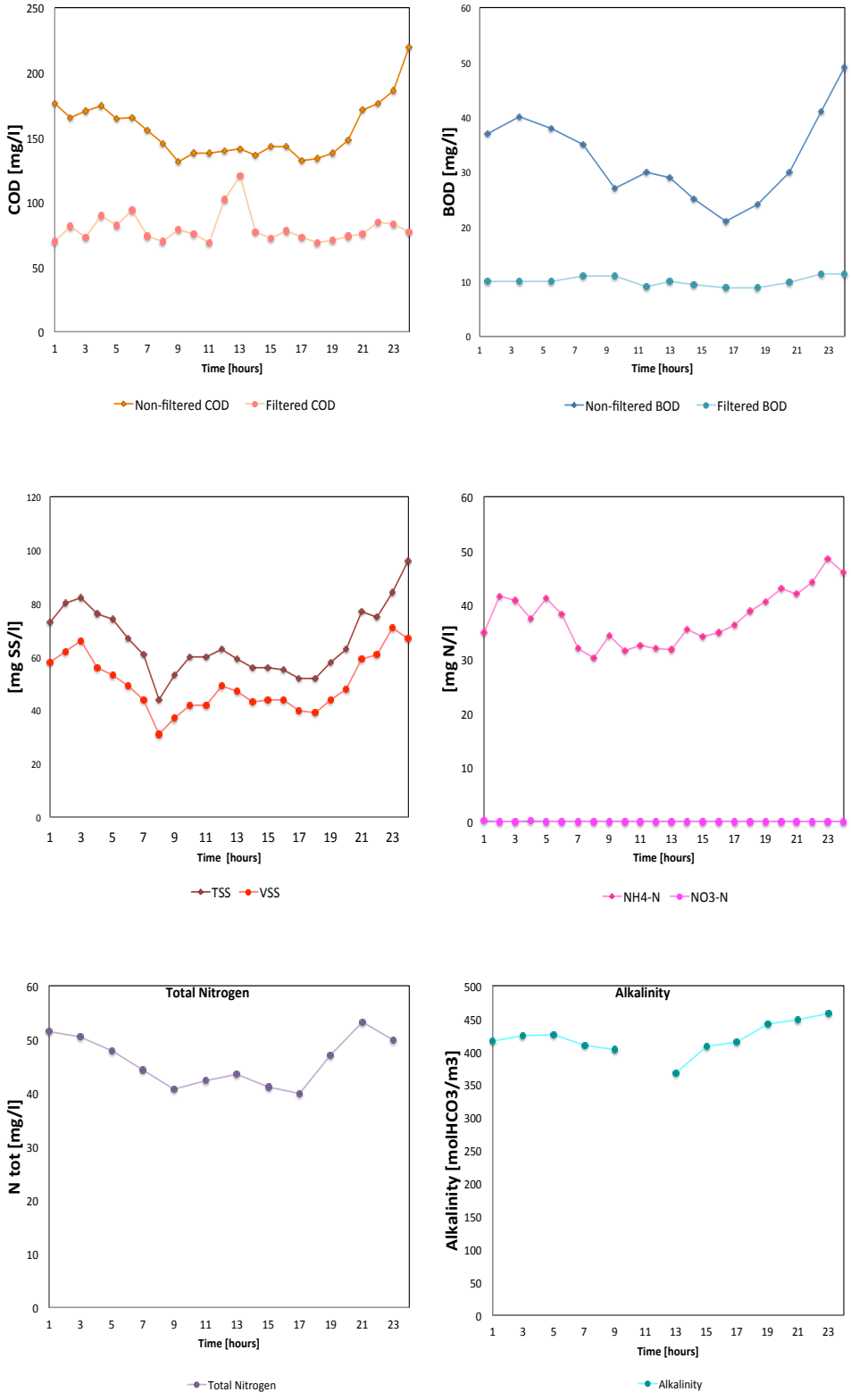

Figure 3.2: Effluent wastewater quality trends measured during the measuring campaign

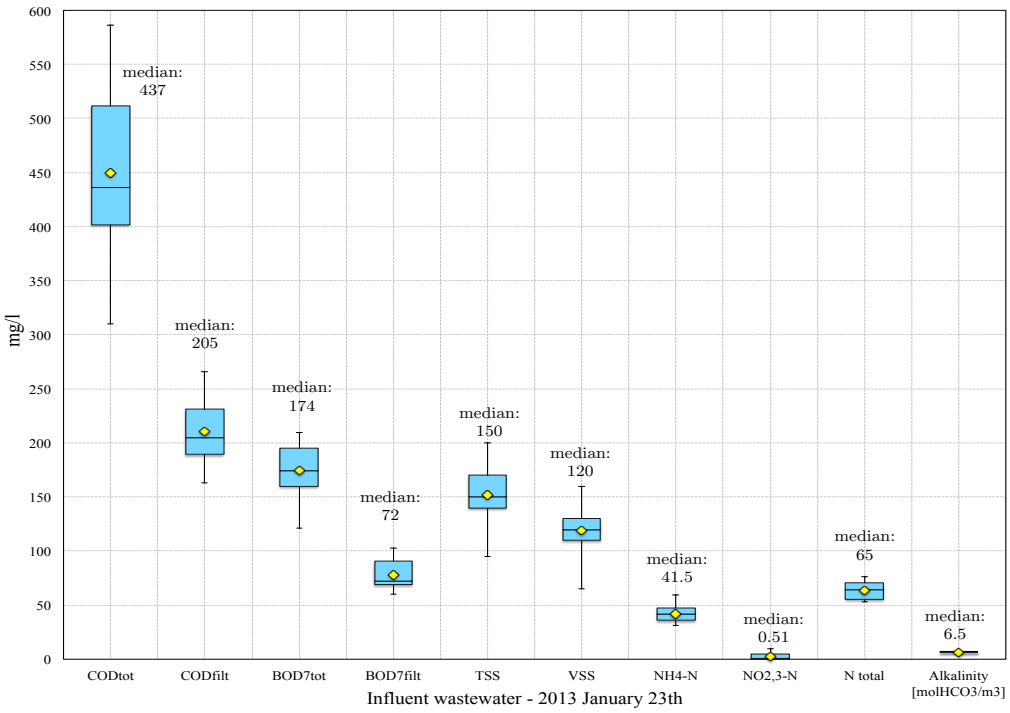

Figure 3.3: Influent wastewater quality ranges measured during the measuring campaign

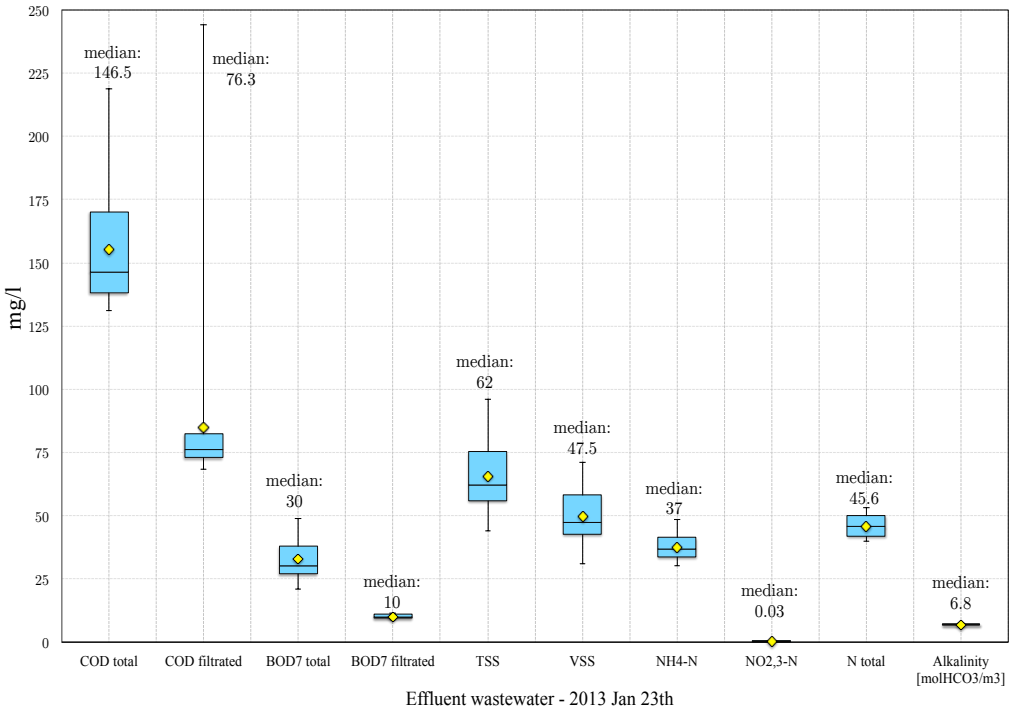

Figure 3.4: Effluent wastewater quality ranges measured during the measuring campaign

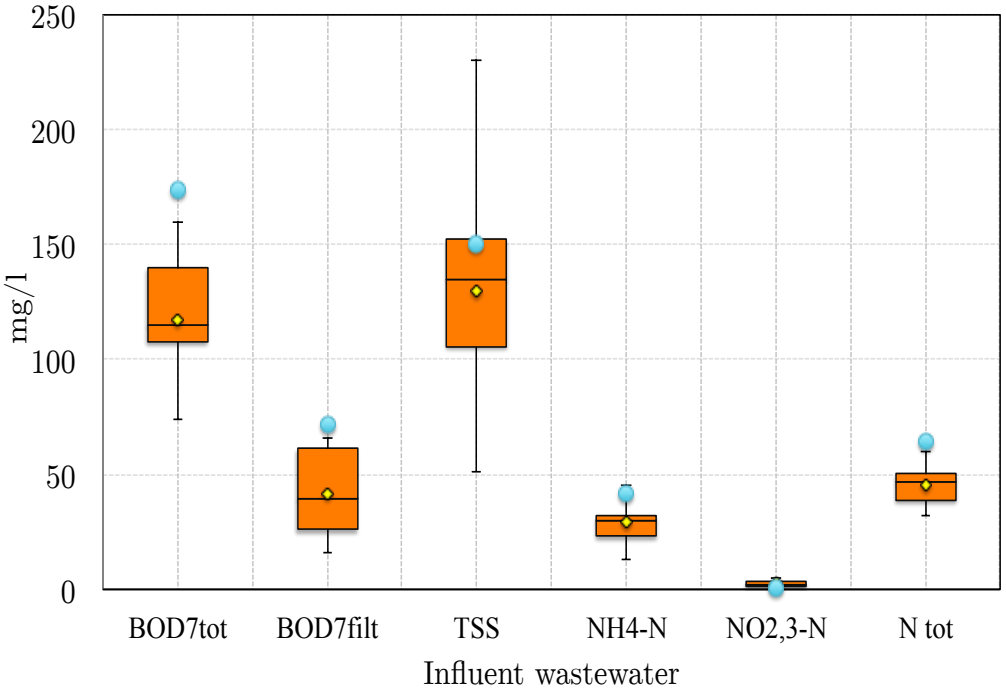

Figure 3.5: Influent wastewater. Comparison between routine laboratory data from January and February 2013—graph boxes—and daily average data obtained from the measuring campaign—blue dots.

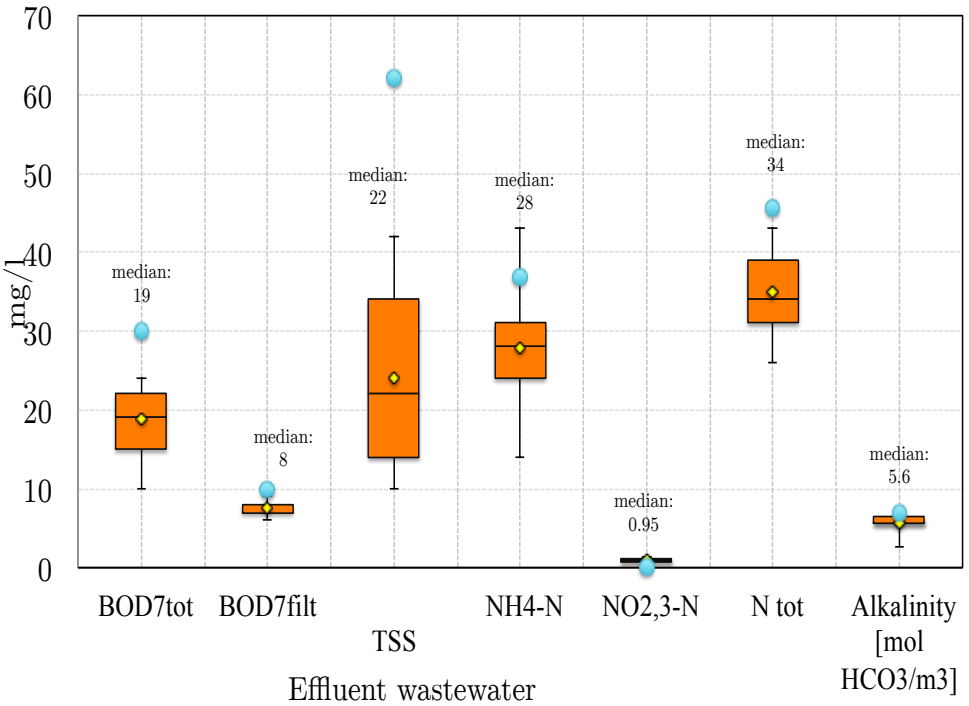

Figure 3.6: Effluent wastewater. Comparison between routine laboratory data from January and February 2013—graph boxes—and daily average data obtained from the measuring campaign—blue dots.

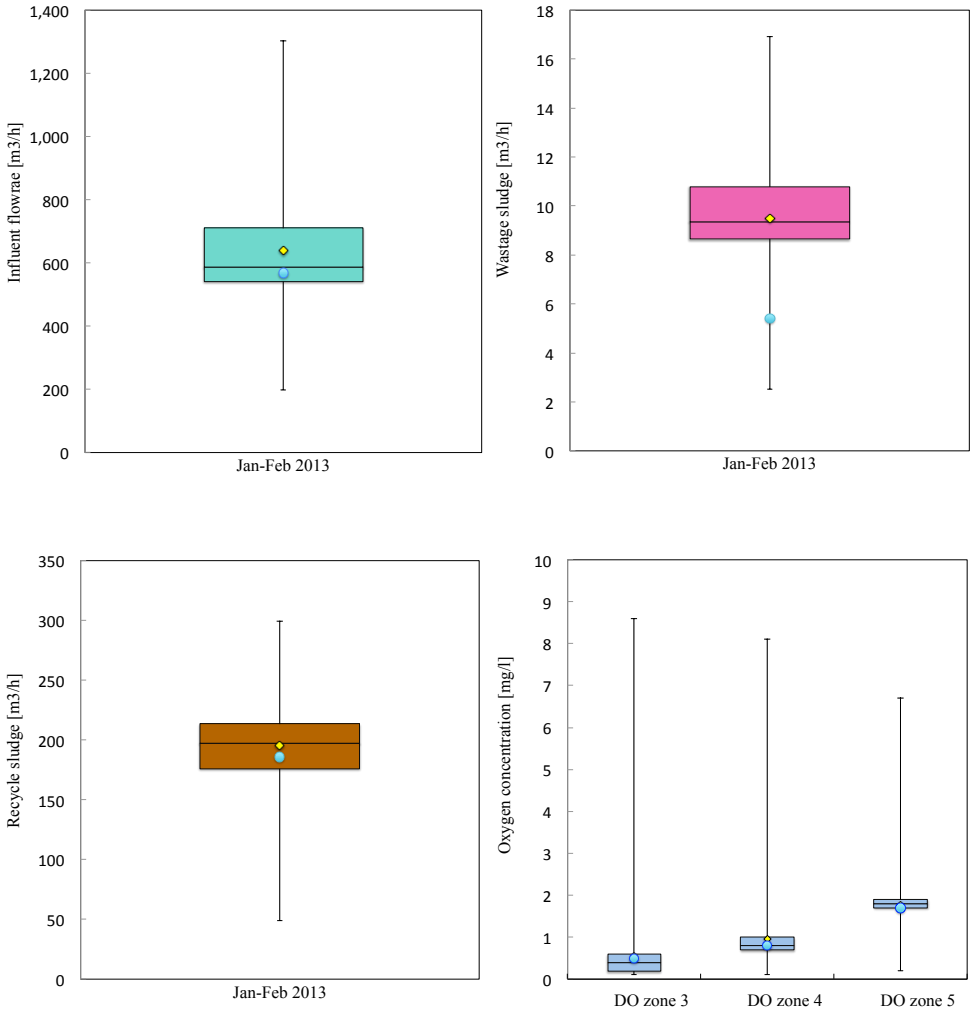

Figure 3.7: Influent flowrate, wastage sludge, sludge recycle, oxygen concentration in the three aerobic reactors. Comparison between routine laboratory data from January and February 2013—graph boxes—and daily average data obtained from the measuring campaign—blue dots.

3.1.2 Oxygen Uptake Rate Test

Two experiments were performed on 20^{th} December 2012 and on 28^{th} January 2013 (referred to as 1^{st} experiment and 2^{nd} experiment). In each experiment, two parallel reactors were run simultaneously (referred to as A and B), in order to improve results reliability. Furthermore, the 1^{st} experiment consisted of two trials in series. Figures 3.8 and 3.10 present the results of the two experimental trials (with both the two parallel experiments, A and B).

Each Figure includes several graphs depicting:

- The oxygen concentration trend. Looking at this graph the different experimental phases appear clear: (i) the continuous aeration phase, necessary to remove the existing organic substrate in the sample—mainly

produced by hydrolysis occurring during the transport stage (Keskitalo et al., 2010)—and to reach constant temperature and oxygen concentration; (ii) the first alternate aeration phase, needed to ensure the achievement of the endogenous phase. The pattern of dissolved oxygen concentration starts to become alternate, as a consequence of oxygen concentration that increases during the aeration phase and decreases as soon as the aeration is switched off; (iii) the addition of acetate, which leads to a sudden decrease of oxygen concentration, due to maximum growth rate of heterotrophic bacteria (no limitation to the growth rate occurs in this phase); (iv) the gradual increase of the oxygen concentration, caused by lack of external substrate and consequent reduced heterotrophic growth rate (limited by the substrate scarcity); (v) the endogenous respiration reached again, but with a slight higher value, which may possibly arise from storage phenomena.

- The OUR respirogram shows the OUR trend throughout the experiment and reflects all the five phases described above.

Table 3.3 summarizes the operational conditions and the main result values: the maximum oxygen uptake rate measured (which is the average of the maximum values); the average endogenous respiration rate (which is averaged over around five values before the acetate addition); the maximum exogenous respiration rate (which is calculated from the difference between the previous two values); the SOUR maximum uptake rate measured.

Comparison of the 1^{st} and 2^{nd} experiment:

- Both test performed show similar behaviour with only slight differences. The two test in series carried out in the 1^{st} experiment reach the same maximum oxygen uptake rate and endogenous respiration rate values.

- Regarding the maximum oxygen uptake rate values shown in the respirograms, it is visible how in the 1^{st} experiment it is made up of only one value, whereas in the 2^{nd} experiment it consists of five values, building the so-called plateau. This is the result of two different operational conditions: in the 2^{nd} experiment the acetate amount added was doubled and the time interval of aeration and non-aeration reduced from 5 to 3 minutes. These changes were made on purpose to increase the reliability of the maximum oxygen uptake rate calculated in the 1^{st} experiment: the increased acetate availability kept the specific growth rate at its maximum value for a longer period of time; furthermore, the reduced aeration-non aeration cycle allowed the recording of more oxygen descending phases, leading to the formation of the plateau.

- In the 2^{nd} experiment, the respirogram recorded an anomalous behaviour in the descending phase, after the acetate addition. This may arise from malfunctioning of the aeration system or the oxygen-recording sensor.

- On the descending part of the trajectory, all respirograms exhibit the typical shape caused by aerobic storage of readily biodegradable sub-

Table 3.3: Characteristic values for the two experimental trials and operating condition.

		1^{st} Experiment		2^{nd} Experiment	
		A	B	A	B
VSS	mg l^{-1}	2293	2247	2860	2900
Max OUR	$mgO_2(l\ h)^{-1}$	68.5	62.5	66.6	58.1
Endogenous respiration	$mgO_2(l\ h)^{-1}$	16.9	15.4	13.7	11.9
Exogenous respiration	$mgO_2(l\ h)^{-1}$	51.6	47.1	52.9	46.2
Max SOUR	$mgO_2(gVSS\ h)^{-1}$	29.9	27.8	23.3	20
Endogenous respiration	$mgO_2(gVSS\ h)^{-1}$	7.4	6.9	4.8	4.1
Exogenous respiration	$mgO_2(gVSS\ h)^{-1}$	22.5	21	18.5	15.9
Aeration on-off cycle	min	10	10	6	6
Activated sludge volume	l	1.5	1.5	1.5	1.5
Temperature	°C	15	15	15	15
Acetate solution addition	ml	6	6	12	12
Acetate COD addition	mg l^{-1}	80	80	160	160
Recorded maximum O2	$mgO_2(l\ h)^{-1}$	9.5	9.5	9	9

strate (Hoque et al., 2008 and Guisasola et al., 2005). Two endogenous respiration rates are present in each test: at the beginning of the test, before the acetate addition, and at the end of the test, after all the external substrate is completely degraded. This second endogenous respiration rate is slightly higher than the first, possibly justified by the storage phenomena, which involves the consumption of oxygen and consequently the oxygen respiration rate increase (Petersen, 2000). In the Activated Sludge Model 3, the storage phenomena is included, assuming that all S_s first becomes stored material and later becomes available for cell growth. This may not reflect reality, but currently no model is available to predict the separation of S_s into direct growth and storage (Petersen, 2000).

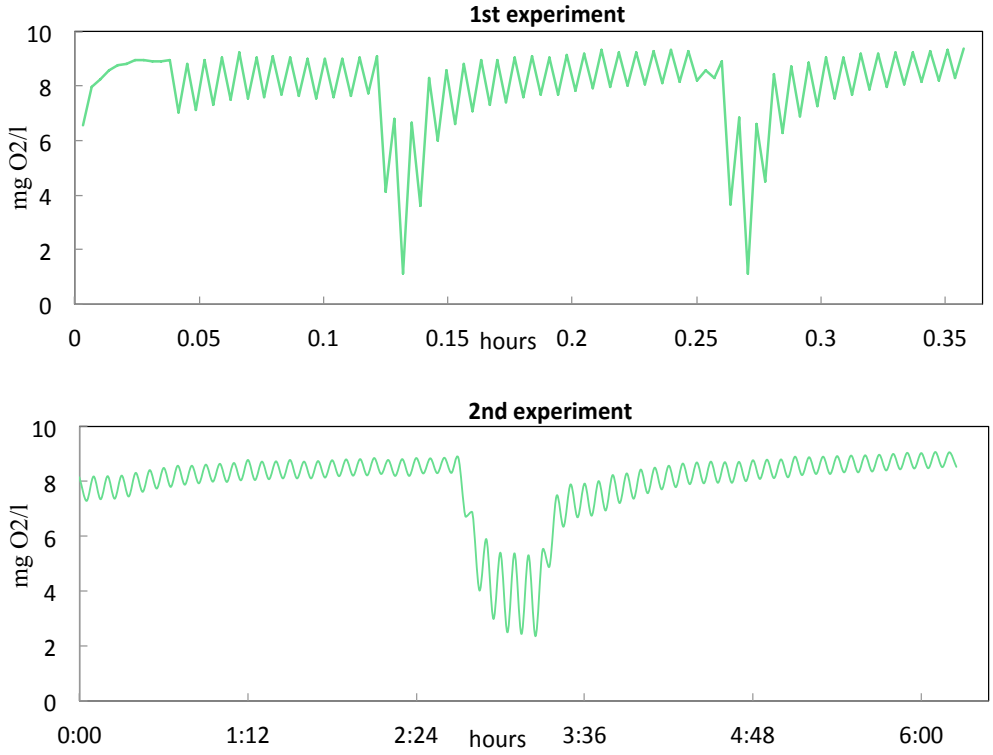

Figure 3.8: Oxygen concentration throughout the OUR experiments, for both trials.

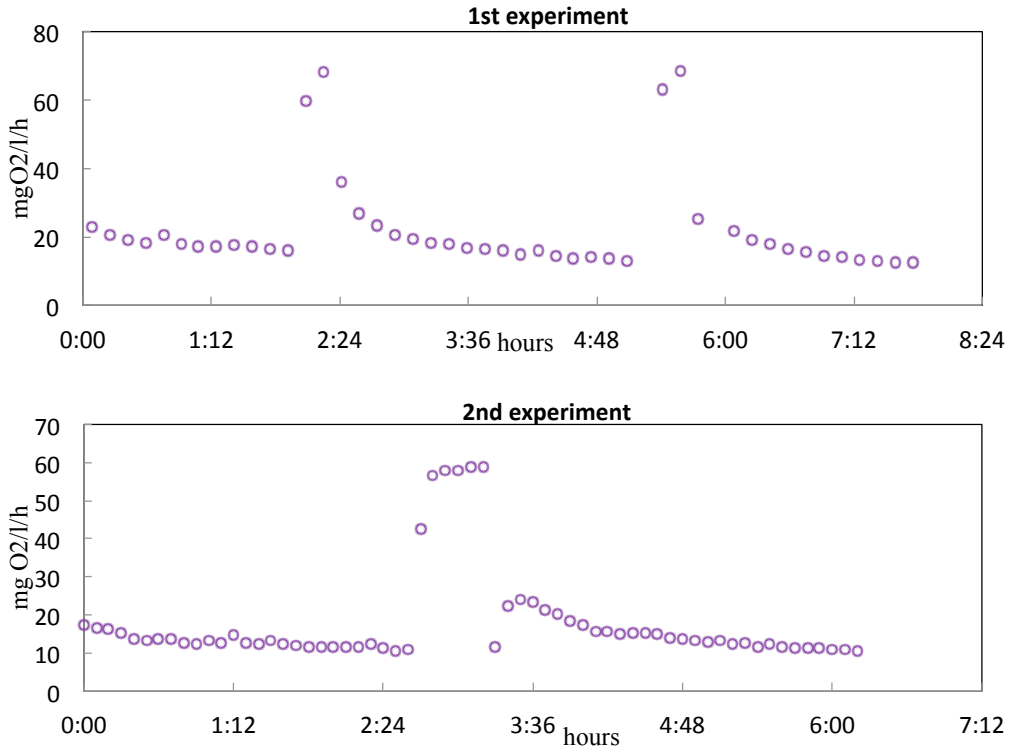

Figure 3.9: Respirometers: Oxygen uptake rate values throughout the OUR experiments, for both trials.

3.1.3 Sub-model calibration and validation

As explained in Section 2.6.2, a Matlab sub-model was created in order to simulate the two respirometry experiments and define the starting values for three kinetic parameters of the heterotrophic metabolism required by the Activated Sludge Model 1.

Some sets of starting values for these three parameters were defined by the sub-model calibration performed on the 2^{nd} experimental trial. More than one set of values were able to simulate the test. In particular, two of them (reported in Table 3.4) were considered equally valid and were used for the sub-model validation and chosen for further calibration in the full-scale model.

To assess the predictive power of the sub-model, the Nash-Sutcliffe model efficiency coefficient is used—between the observed and the calibrated / validated values for the OUR. Figures 3.10 and 3.11 show the result of the sub-model calibration and validation. Each Figure illustrates:

- heterotrophic biomass growth during the experiment; its initial value choice is described in Section 2.6.2; the behaviour it has was expected: it starts to increase as soon as the acetate is added.
- the substrate concentration; it is added suddenly (indeed a peak is present) and than is gradually removed, reflecting biomass growth and oxygen consumption increase.
- the oxygen concentration.
- the OUR respirogram.

The calibrated values appear to have a slightly smaller Nash efficiency coefficient; the reason is that the respirogram of the 2^{nd} experiment presents an anomalous behaviour when reaching the second endogenous respiration rate, which was neglected during model calibration.

Table 3.4: Set 1 and set 2 in the 1^{st} and 2^{st} experiment. Nash efficiency is calculated for each of them.

	1^{st} experiment		2^{st} experiment	
	set 1	set 2	set1	set2
μ_{maxH}	2.96	4.40	2.96	4.4
b_h	0.259	0.37	0.259	0.37
K_s	22	22	22	22
X1	1400	950	1100	760
Nash efficiency	0.794	0.796	0.433	0.416

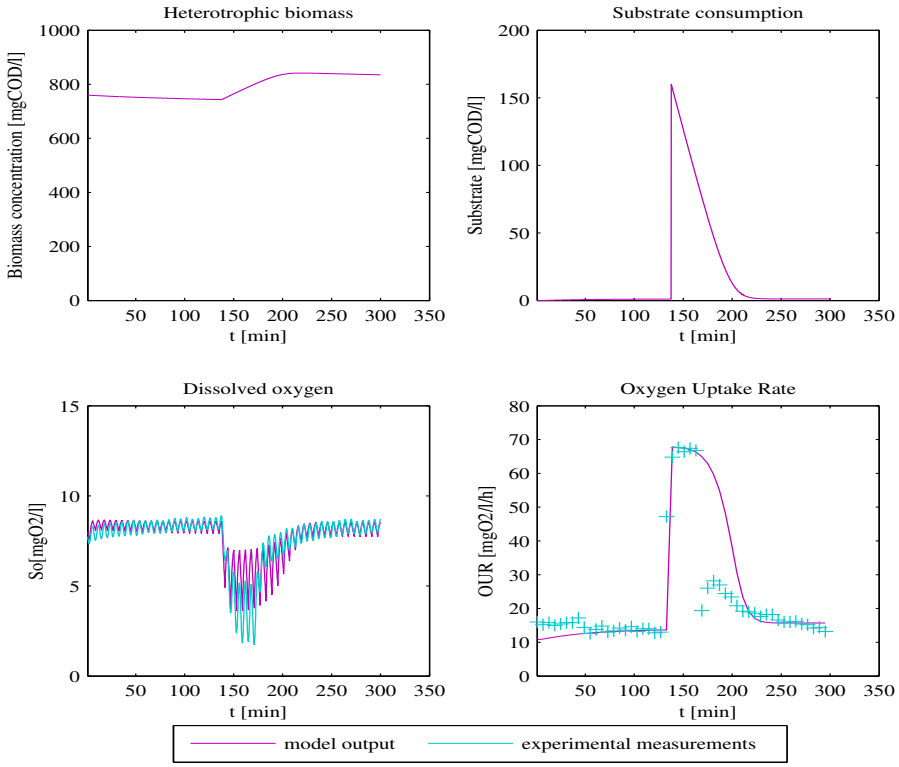

Figure 3.10: Result of the sub-model calibration conducted on the 2^{nd} experiment. The four graphs represent: i)the biomass concentration; ii) the biodegradable substrate concentration; iii) the oxygen concentration; iv) the respirogram with the Oxygen Uptake Rate values calculated.

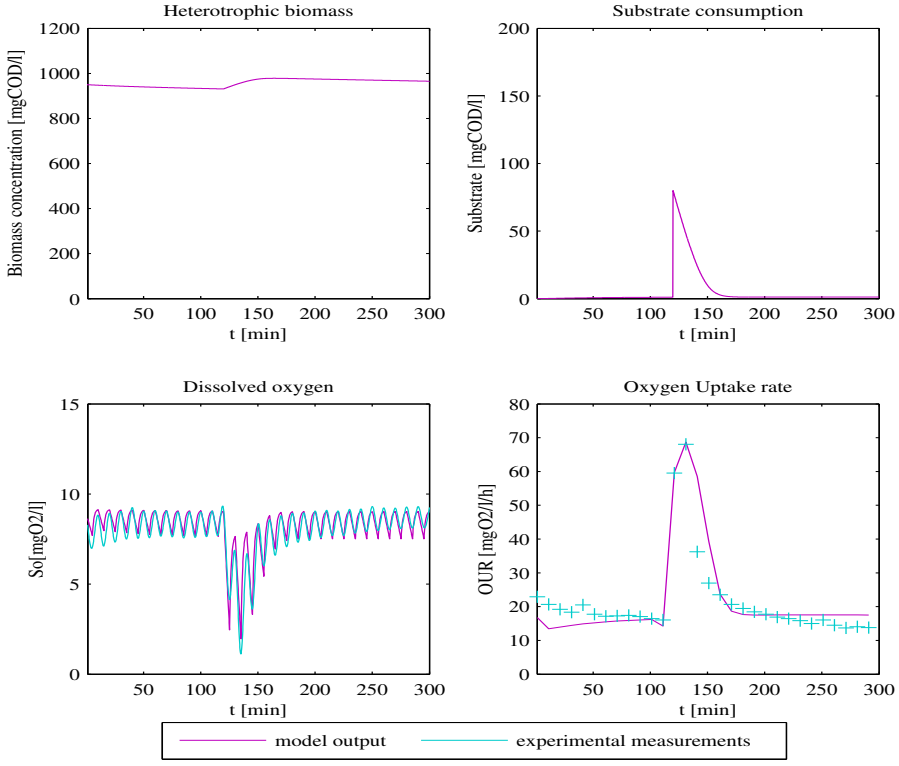

Figure 3.11: Result of the sub-model validation conducted on the 1^{st} experiment. The four graphs represent: i)the biomass concentration; ii) the biodegradable substrate concentration; iii) the oxygen concentration; iv) the respirogram with the Oxygen Uptake Rate values calculated.

3.1.4 Settling Column Test

The result of the test is pictured in Figure 3.12. It was performed on mixed liquor samples for four different sludge solids concentrations: 2.6, 2.1, 1.6, and 0.8 mg TSS d^{-1}. The first is the mixed liquor itself; the others are obtained through proper dilution with effluent treated wastewater.

The zone settling velocity is calculated for each of the four curves on the straight part of the trajectory (see Table 3.5). The four experimental values were then plotted in a suitable semi log (natural) diagram and the best-fit straight line drawn (see Fig. 3.13). The gradient of this straight line is the constant k and the linear coefficient is the log of v_o. The obtained maximum theoretical Vesilind velocity is 126 m d^{-1}. This is the starting value inserted in the full-scale model for further calibration. The final v_o (after calibration) is 135 m d^{-1}, not very far from the experimental value obtained.

The settling properties of the Sjölunda activated sludge are good, in spite of the very slow sludge age.

Table 3.5: Activated sludge concentration, zone settling velocity computed, and its natural logaritm, for each sample tested.

X gTSS l^{-1}	ZSV m h^{-1}	ln ZSV
0.76	1.945	0.665
1.56	0.866	-0.144
2.08	0.400	-0.915
2.57	0.268	-1.315

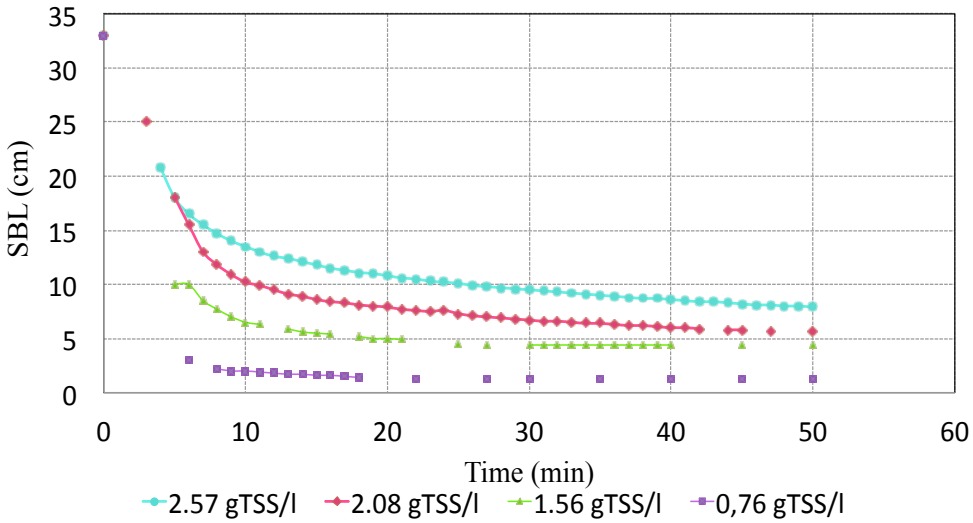

Figure 3.12: Sludge blanket level in function of time

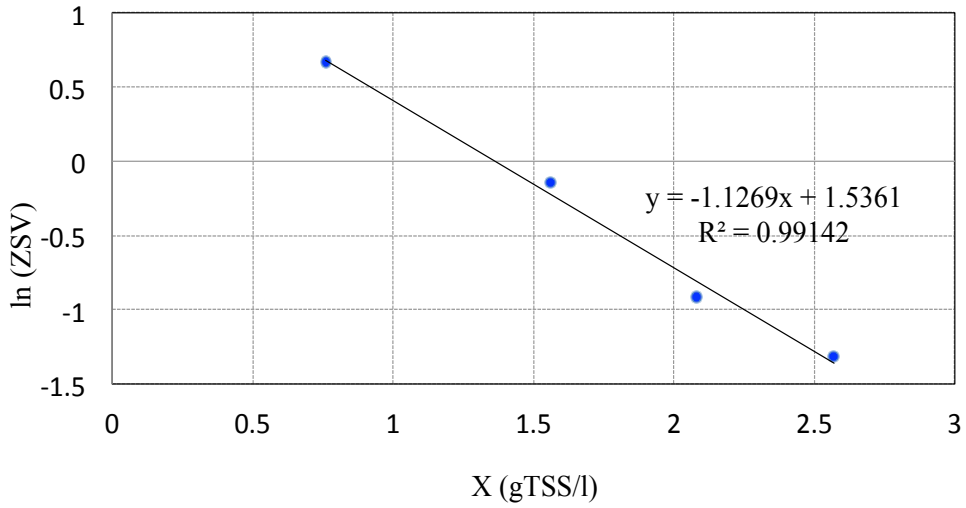

Figure 3.13: Natural logarithm of ZSV in function of the solids concentration

3.2 Results of the model calibration

3.2.1 Input files data

In this Section, the results of the calibration of the influent state variables values are offered. Table 3.6 reports the values composing the constinfluent file for the steady state simulation, characterizing the average influent wastewater. The two rows illustrate: (i) the fractions (referred to as 1^{st} Attempt) deriving from the application of the theoretical relationships to the measuring campaign analyses averaged over the day (described in Section 2.5.1.3); (ii) the fractions (referred to as Calibrated) deriving from the full-scale final model calibration starting from the previous ones (described in Section 2.6.4).

Table 3.6: Constinfluent file. The second set is the final calibrated one.

	S_i	S_s	X_i	X_s	X_{bh}	X_{ba}	X_p	S_o
1^{st} Attempt	47.7	165.4	166.2	77	0	0	0	0
Calibrated	47.7	165.4	80	162.6	0	0.5	0	0

	S_{no}	S_{nh}	S_{nd}	X_{nd}	S_{alk}	TSS	Q_i
1^{st} Attempt	2.29	41.7	7.4	11.2	6.4	153.2	13448.8
Calibrated	2.299	41.7	7.4	11.2	6.4	153.2	13448.8

Table 3.7: Percentage distribution of COD and N components of the final set of variables.

	S_i	S_s	X_i	X_s	X_{bh}	X_{ba}	X_p	S_o
% of CODtot	10	36	18	36	0	0 .1	0	

	S_{no}	S_{nh}	S_{nd}	X_{nd}	S_{alk}	TSS	Q_i
% of Ntot	4	67	12	18			

The creation of the dynamic input file started from the constinfluent calibrated file. In order to keep the COD fractionation, determined through calibration, the percentage of each fraction, relative to the total COD, is applied in the dynamic file to the total amount of COD and N of each time-step. Only the soluble non-biodegradable fraction, S_i, was kept equal to the same value, 47.7 mg l^{-1}, throughout the file.

3.2.2 Parameters

Table 3.8 summarizes the values of all the final parameters employed in the BSM1. Out of all the calibrated parameters, the only value outside the ranges proposed by Petersen (2000) is the half saturation constant, K_s. Indeed, the K_s value obtained from the OUR experiment and from further full-scale model

calibration is 40 mg l^{-1}, which is higher than the default value. The reason for this choice is explained in the following lines. This plant is a high-loaded activated sludge treatment plant (see Section 2.4) and it is characterised by low removal efficiency; an average value for BOD removal is around 85%. The Activated Sludge Model 1 was not created to simulate this kind of operational conditions (Petersen, 2000) and therefore using the default K_s value it is not able to meet the measured effluent BOD concentration. The expedient found was to increase the K_s value, so that it could act as a limiting factor to the BOD removal rate and reduces its removal efficiency. For this case study, it could be questioned whether it was necessary to determine some kinetic parameters in lab-scale experiments, since the resulting calibrated parameters are not far from the ASM1 default parameter set—except for the half saturation coefficient, K_s—by Henze et al., 1987. Still, the lab-scale results gave extra confirmation on the parameter set of the calibrated model, thereby increasing the quality and confidence of the model calibration.

3.2.3 Steady-state simulation

In this Section, the results of the steady-state calibration are presented. In Table 3.9, the reference and the model output values of the process operational conditions are reported. The reference values derive from the measuring campaign analyses averaged over the day. In Table 3.10, the concentrations of the main pollutants are given, for both the measured and the simulated values.

Out of all the compounds selected to focus the calibration on, the BOD is the one that presented most difficulties, due to its high value in the effluent; whereas, COD and TSS in the effluent, together with TSS in the reactor show a satisfactory fit. Effluent ammonium and total nitrogen are described quite well.

3.2.4 Dynamic simulation

In Table 3.11, the removal efficiency of the model simulation for each pollutant parameter are compared to those measured on the day of the measuring campaign.

Table 3.8: Final parameter set after model-calibration.

Parameter	Source[a]	Value
Stoichiometric parameters		
Y_a [g cell COD formed.(g N oxidized)$^{-1}$]	BSM1	0.24
Y_h [g cell COD formed.(g COD oxidized)$^{-1}$]	BSM1	0.67
f_p [-]	BSM1	0.08
i_{xb} [g N.(g COD)$^{-1}$ in biomass]	BSM1	0.08
i_{xp} [g N.(g COD)$^{-1}$ in particulate products]	BSM1	0.06
Kinetic parameters		
μ_h [d^{-1}]	OUR	4.4
K_S [g COD.m^{-3}]	OUR	40
$K_{O,H}$ [g (-COD).m^{-3}]	BSM1	0.2
K_{NO} [g NO_3-N.m^{-3}]	BSM1	0.5
b_h [d^{-1}]	OUR	0.37
η_g [-]	BSM1	0.8
η_h [-]	BSM1	0.8
k_h [g slowly biodegradable COD.(g cell COD.d)$^{-1}$]	BSM1	3
K_X [g slowly biodegradable COD.(g cell COD)$^{-1}$]	BSM1	0.1
μ_A [d-1]	BSM1	0.5
K_{NH} [g NH3-N].m^{-3}	BSM1	1
b_a [d-1]	BSM1	0.05
$K_{O,A}$ [g (-COD).m^{-3}]	BSM1	0.4
k_a [m3 (g COD.d)$^{-1}$]	BSM1	0.05
Settler parameters		
v'_o, maximum settling velocity [m.d^{-1}]	Takács	112.1
v_o, maximum Vesilind settling velocity [m.d^{-1}]	Z.S.V.	135
rh, Hindered zone settling parameter [m^3.(g SS)$^{-1}$]	Takács	0.000293
rp, Flocculant zone settling parameter [m^3.(g SS)$^{-1}$]	Takács	0.0027
fns, Non-settleable fraction [-]	Takács	0.00259

[a] The parameters values were selected according to 4 sources: i) BSM1, it refers to the Benchmark Simulation Model 1 default values; ii) OUR, it refers to those parameter that have been indirectly estimated with the OUR Test, and subsequently calibrated through a sub-model; iii) Takács, it refers to the article by Takács et al. (1991), where settler parameters values for high-loaded plant are suggested; iv) Z.S.V., it refers to the Zone Settling Velocity Test, through which was obtained a first estimation of Vo.

Table 3.9: Average operational conditions deriving from both the measured and the model steady-state output data.

Parameter	Units	Values Measured	Values Simulated
Influent flow rate	m^3d^{-1}	13449	13449
Temperature	°C	14	14
Mixed liquor solids concentration	mg SS l^{-1}	2200	2140
Sludge solids concentration	mgTSS l^{-1}	9000-10000	9887
Sludge age	d	1.3-1.4	1.3
Sludge load (anox+aerobic)	kgBOD(kgSS d)$^{-1}$	0.72	0.81
Sludge load (aerobic only)	kgBOD(kgSS d)$^{-1}$	0.96	1.09
D.O. 1^{st} reactor	mg l^{-1}	0.5	0.5
D.O. 2^{nd} reactor	mg l^{-1}	0.8	0.8
D.O. 3^{rd} reactor	mg l^{-1}	1.7	1.7

Table 3.10: Comparison between pollutant concentrations derived from measurements and model steady-state outputs (expressed in mg l^{-1}, except for the alkalinity expressed in mol HCO_3 h^{-1}).

	Focus of the steady-state calibration				Other effluent variables			
	CODeff.	BODeff.	TSSeff.	TSS reactor	NH_4-N	NO_3-N	Ntot	Alk.
Meas[a].	157	36	66	2200	38	0.028	46	6.4
Model[b]	156	27	64	2140	38	0.05	48	6.3
S.D.[c]	0.99	0.75	0.97	0.97	1	0.56	0.96	0.98

[a] Effluent quality measured on the measuring campaign day (average values)
[b] Model outputs after steady-state calibration
[c] Standard deviation to evaluate the accuracy of the model

Table 3.11: Influent, effluent, removed loads (kg d^{-1}) measured and simulated in steady-state condition. In the last row the model description of the removed loads is given

Parameters	CODtot	BODtot	TSS	NH4-N	NO3-N	Ntot
Measurements						
Influent load	6135	2607	2063	561	31	842
Effluent load	2112	485	899	515	0.4	620
Load removed	4023	2122	1163	46	31	223
Percentage removed %	0.66	0.81	0.56	0.08	0.99	0.26
Steady-state simulation outputs						
Influent load	6136	2872	2060	562	32	841
Effluent load	2165	385	887	508	0.7	644
Load removed	3972	2486	1174	54	31	197
Percentage removed %	0.65	0.87	0.57	0.1	0.98	0.23
S.D.	0.99	0.85	0.99	0.86	0.99	0.89

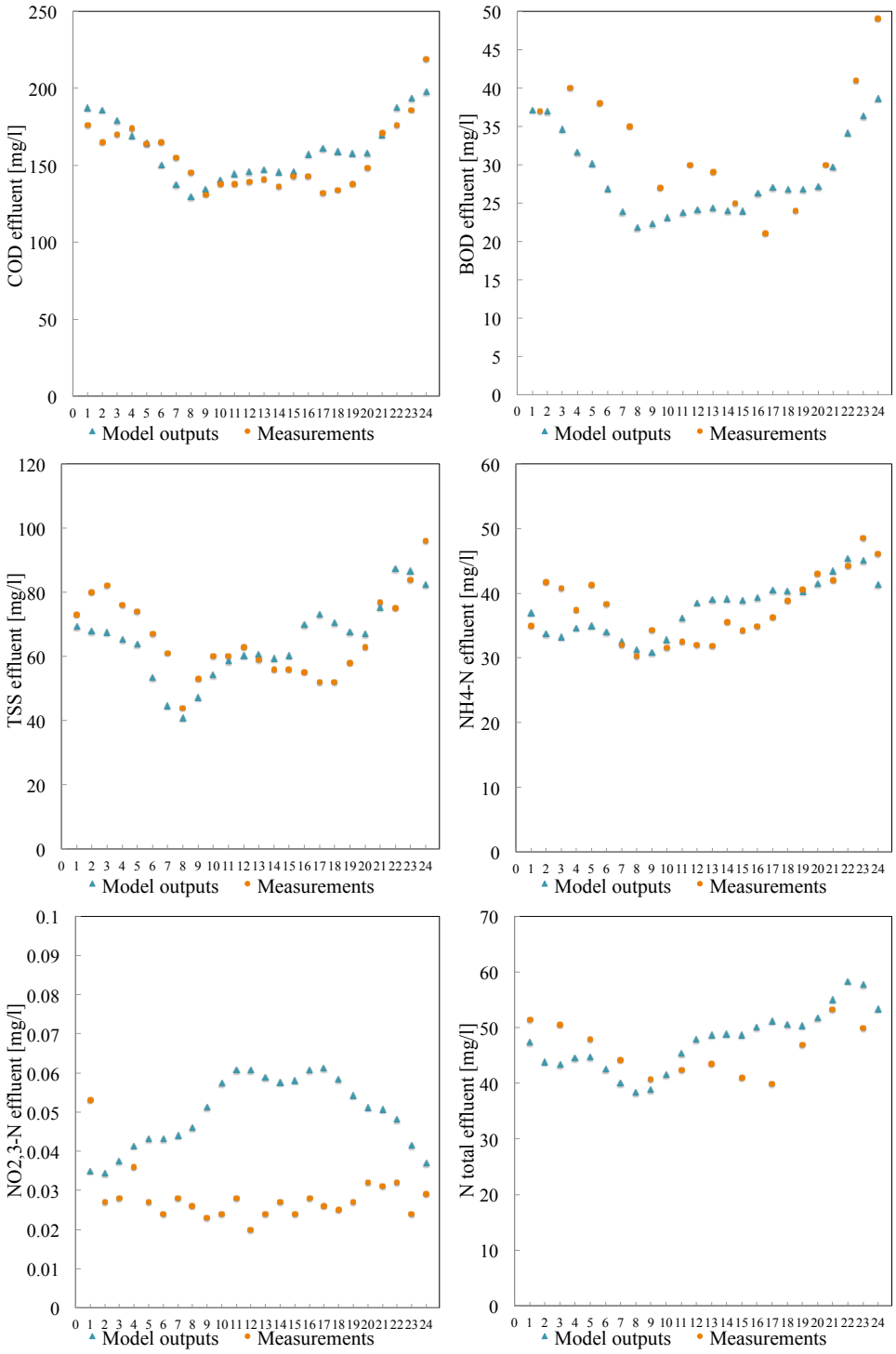

Figure 3.14: Comparison between dynamic model outputs and measurements of effluent wastewater quality performed on the measuring campaign day

3.3 Model validation

In this Section the results of the steady-state validation are illustrated. Table 3.12 shows the model input file created starting from the influent quality data available as laboratory routine work of Autumn 2012. The operational conditions of the plant imposed for the validation, such as TSS in the reactor, oxygen concentration set-points, recycle and wastage sludge flows, are average values registered in the same time period (described in Table 3.13).

As shown in Table 3.14, the model outputs do not reflect the measured data accurately. This may be explained by observing the big differences in pollutants removal efficiencies of data registered on the measuring campaign day (data used for the calibration) and those of the Autumn 2012. For example, on the day of the measuring campaign, the TSS removal efficiency was lower than average values of Autumn 2012 and this efficiency was imposed to the model through calibration; this is reflected in the model validation outputs, which down-estimate the TSS removal efficiency (TSS simulated is 59, whereas the measured is 35 mg l^{-1}).

Table 3.12: Constinfluent file for the model validation

S_i	S_s	X_i	X_s	X_{bh}	X_{ba}	X_p	S_o
47.7	152.3	73.7	149.8	0	0.5	0	0

S_{no}	S_{nh}	S_{nd}	X_{nd}	S_{alk}	TSS	Q_i
2.4	33	5	7.6	6.4	141.1	11610

Table 3.13: Average operational conditions deriving from both the measured and the model validation output data.

		Measured	Simulated
Influent flow rate	$m^3\ d^{-1}$	11610	11610
Temperature	°C	14	14
Mixed liquor solids concentration	mg SS l^{-1}	2600	2700
Sludge solids concentration	mgTSS l^{-1}	8100	7759
Sludge age	d	2.27	2.13
Sludge load (aerobic only)	kgBOD(kgSS d)$^{-1}$	0.51	1.09
D.O. 1^{st} reactor	mg l^{-1}	0.2	0.2
D.O. 2^{nd} reactor	mg l^{-1}	0.7	0.7
D.O. 3^{rd} reactor	mg l^{-1}	2	2

Table 3.14: Average operational conditions deriving from both the measured and the model validation output data.

	CODeff.	BODeff.	TSSeff.	TSS reac	NH_4-N	NO_3-N	Ntot
Measurements	185	23	35	2600	31	1.1	27
Simulations	138.19	21	59	2700	26	0.09	35

3.4 Results of the model implementation

3.4.1 Optimization solution I: Improvement of biogas production through anaerobic digestion of the biological sludge

To improve biogas production, effort was given to increase the load of biodegradable substrate extracted from the secondary settler and sent to the anaerobic digestion plant. The sludge age was changed through wastage sludge extraction manipulation, and the effect it has upon the quality of the wastage sludge, and of the effluent wastewater, was monitored. The sludge age range explored was between 15 and 60 hours. Higher values lead to solids loss in the effluent for bad settling performance. The model was run dynamically several times, each time decreasing the wastage sludge extracted to increase the sludge age.

The dynamic simulations offered dynamic trajectories of the monitored compounds. These data averaged over the day were employed to compare different initial conditions, to evaluate the average plant behaviour and perform the energy balance; after having established that the tested configuration has a positive economic impact, the dynamic answer of the plant process has to be considered. Indeed, although from an average point of view, the model suggests good plant operation and economic profit, the existing diurnal variations could affect the plant performance during the day negatively, something that could only be detected by looking to the dynamic simulation results.

The idea of simulating sludge age changes had the main aim of investigating a way to increase biodegradable particulate substrate loaded to the anaerobic digestion. The case with the highest sum of X_s and X_{bh} loads in the sludge was selected as the most interesting from an energy viewpoint. As shown in Fig., this condition is obtained for a sludge age of 20 h and a wastage sludge of 320 m^3 d^{-1} (the reference case, reflecting the current operating condition, has a sludge age of 32 h and a wastage sludge of 190 m^3 d^{-1}). The red point in Fig 3.15 represents the reference case, whereas the blue point is the case of maximum X_{bh}+X_s load.

Exploring this specific case of maximum X_{bh}+X_s load, several considerations may be made by looking at the other plant performances and comparing them with the reference case:

- the biomass in the reactor decreases: the total suspended solids concentration in the reactor decreases from 2100 to 1405 mg TSS l^{-1};
- this lower biomass requires less oxygen supply, with consequent aeration energy reduction;
- COD concentration in the effluent increases from 160 to 173 mg COD l^{-1}: the soluble biodegradable COD, S_s, is the main fraction to raise, from 26 to 47 mg l^{-1};

This higher effluent COD requires further aerobic treatment if this sludge age change has to be applied in the full-scale plant; to evaluate the cost of degrading this increased COD, the oxygen concentration in the aerobic

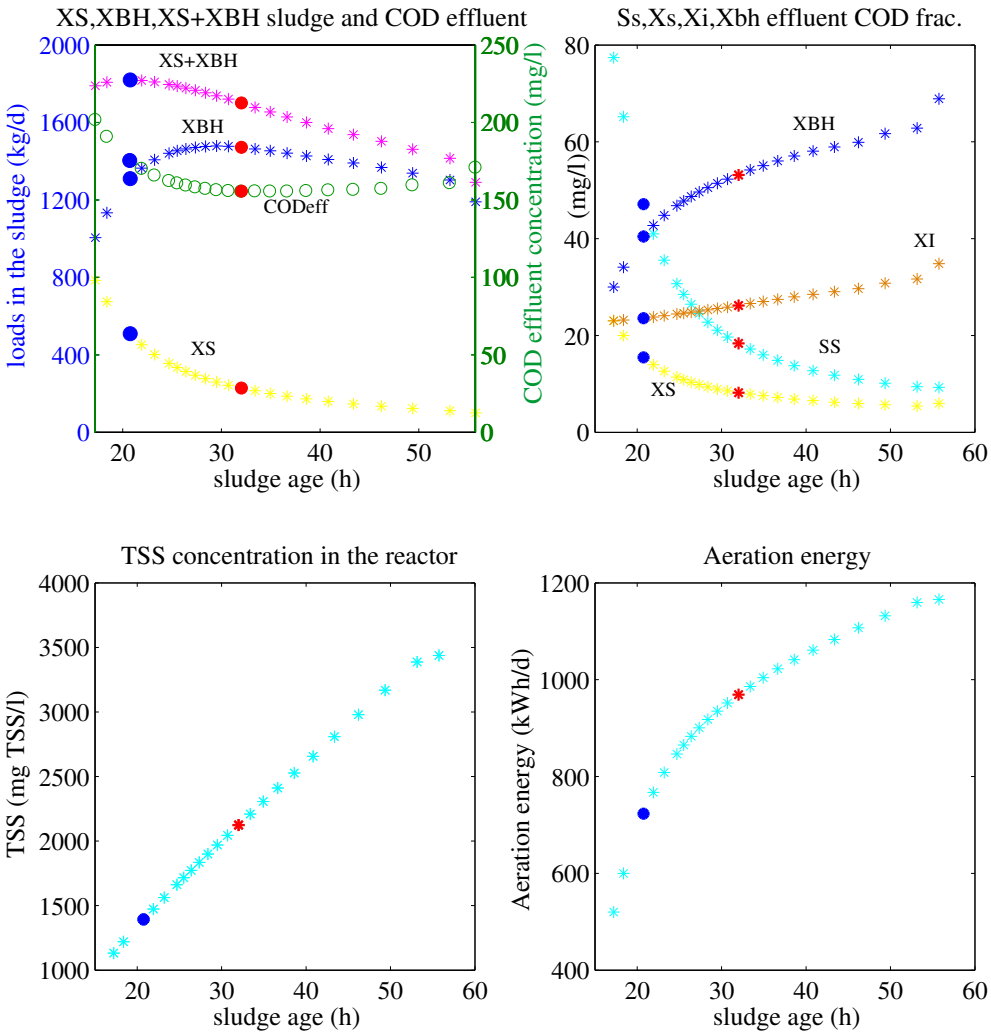

Figure 3.15: Daily average values over sludge age:i) biodegradable particulate load in the wastage sludge and COD effluent concentration; ii) COD fractionation in the effluent; iii) Biomass in the reactor; iv) aeration energy consumption. Red dot represents the reference case, whereas the blue dot represents the case of maximum X_{bh}+X_s load.

reactors was manipulated, until the effluent wastewater quality reached the reference case. To achieve this goal, the new oxygen concentration set-points were adjusted to 0.8 mg l^{-1} (from 0.5 mg l^{-1}) and to 2 mg l^{-1}(from 1.7 mg l^{-1}), in the second and third aerobic reactor respectively; the first aerobic reactor set-point was not changed.

The result obtained was:

- COD and BOD in the effluent decreased and became comparable to the reference case, 160 and 28 mg l^{-1} respectively;

- the biomass in the reactor increased slightly, for to the higher oxygen availability;
- the X_s concentration in the sludge decreases to a value similar to the reference case, due to heterotrophic growth enhanced by the higher oxygen and biomass concentration. The lower X_s concentration in the sludge decreases the carbon biodegradable load sent to the anaerobic digestion; but still appears to be higher than the reference case (since the wastage flow extracted is higher) and consequently convenient from an energy viewpoint.
- X_{bh} concentration in the sludge increases for the higher oxygen availability; higher loads are sent to the digestion.

The energy balance was carried out by comparing the reference case with the case of maximum X_s+X_{bh}. The energy balance was made starting from the reference condition and evaluating the additional biogas production, the aeration energy saved running the model in these operating conditions, and the aeration energy required for degrading the increased effluent COD.

The reference condition is characterized from the following values:

- Qwaste: 190 m^3 d^{-1}
- sludge age: around 32 h
- CODeff : 163mg l^{-1}
- X_s+X_{bh} load in the sludge: 1653 kg d^{-1}
- Aeration energy consumed: 940 kWh d^{-1}
- Energy produced from biogas: 3707 kWh d^{-1}

The case of maximum X_s+X_{bh} load sent to the anaerobic digestion, prior to increasing the oxygen concentration in the aerobic reactors is characterized as follows:

- Qwaste: 320 m^3 d^{-1}
- sludge age: 20 h
- CODeff: 175 mg l^{-1}
- X_s+X_{bh} load in the sludge: 1819 kg d^{-1}
- Aeration energy consumed: 723 kWh d^{-1}.
- Energy produced from biogas: 4079 kWh d^{-1}

Afterwards, the oxygen concentration set-points were increased to reduce the COD in the effluent, as explained before, and the new operating conditions become:

- Qwaste: 320 m^3 d^{-1}

- sludge age: around 20 h
- CODeff : 163mg l^{-1}
- X_s+X_{bh} load in the sludge: 1847 kg d^{-1}, higher than with less O2
- Aeration energy consumed: 880 kWh d^{-1}, higher than with less O2
- Energy produced from biogas: 4140 kWh d^{-1}, higher than with less O2

The final energy balance was computed between the reference case and these last operating conditions, offering the following results:

- increased energy produced from biogas= 4140-3707= 433 kWh d^{-1}
- decreased aeration energy (including the energy for degrading the additional COD)= 940-880=60 kWh d^{-1}
- the balance is 433+60= 493 kWh d^{-1}

Extending the optimization solution to the other three equal lines, the daily energy saving becomes 1972 kWh d^{-1}.

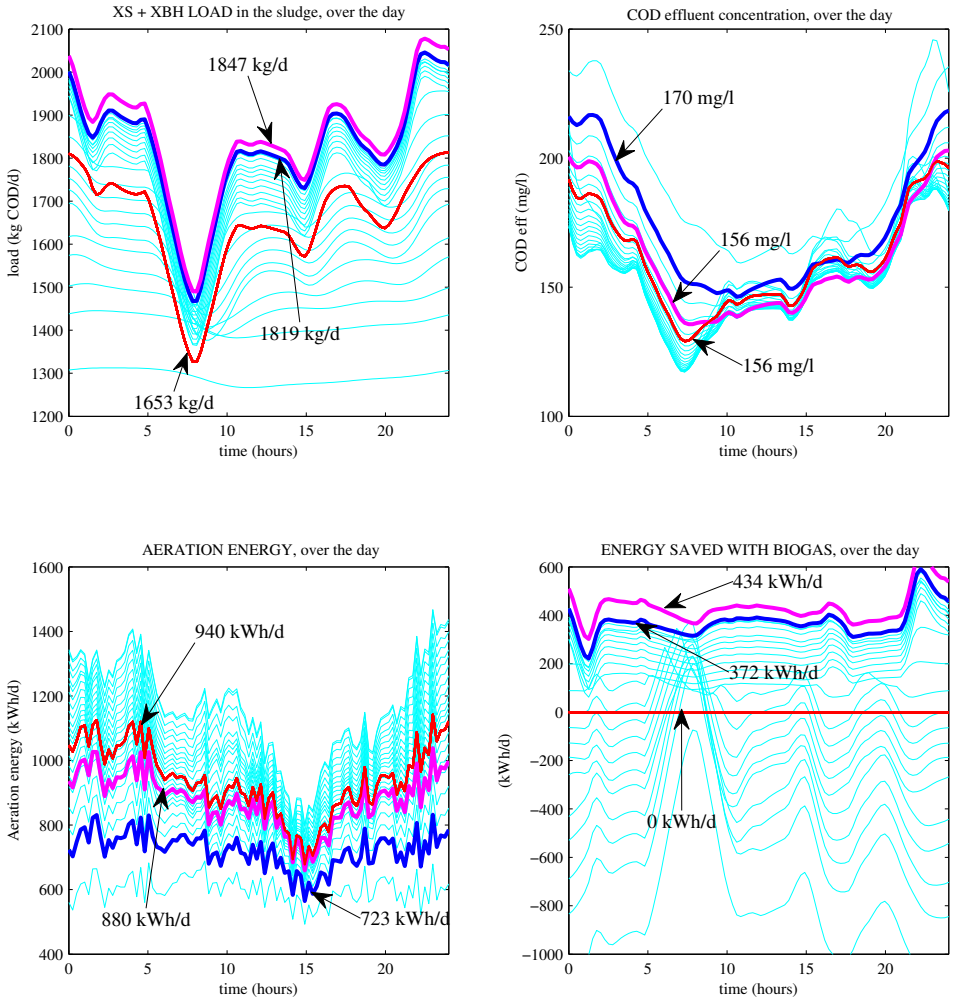

Figure 3.16: Trends over the day from sludge age variation: i)sum of loads of Xs+Xbh in the sludge; ii) COD concentration in the effluent; iii)aeration energy consumed; iv)additional energy production respect to the reference case. The red line represents the reference case, the blue line the case of maximum $X_{bh}+X_s$ load, and the magenta line the optimized case.

3.4.2 Optimization solution II: Enhancement of pre-denitrfication

This Section shows final results emerging from the second optimization solution investigation, which aims to enhance the pre-denitrification, by recycling part of the nitrifying trickling filter nitrate-rich effluent (referred to as Qr_{NO_3}). Graphs with effluent quality and Table with nitrate load removal efficiency and cost balances are presented for all the three scenarios explored. Each graph includes the reference curve (coloured in red) depicting the current plant performances and used as comparison tool. Further graphs are reported in Annex I, to understand the plant performance better. The first

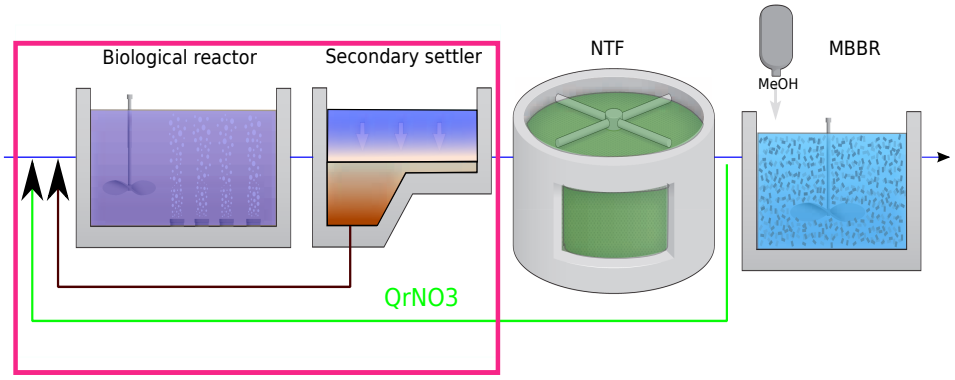

Figure 3.17: Plant configuration with considered for Qr_{NO_3} recycle.

result to come out was the limited capacity of the secondary settler. Indeed, a deteriorated effluent was obtained even for small values of Qr_{NO_3} (2000 m^3d^{-1}): all COD fractions in the effluent increased and pollutants removal efficiency decreased considerably. These model outputs were interpreted as being the result of hydraulic overloading of the settler. Indeed, the hydraulic loading rate was already high, around 1.2 $m^3m^{-2}h^{-1}$, and became higher as the recycle stream was added. The same effect was observed for all three scenarios tested.

These operating conditions seemed to prevent the thorough exploitation of the pre-denitrification potential. The solution to overcome this limitation consists of increasing the settler capacity by diverting part of the mixed liquor coming from the biological reactor to a second settler currently not in use, which has half the volume of the starting one. In this new plant configuration, all three scenarios were explored again, for increasing recycling flow rates. The new results confirm that the settler was the limiting factor: the wastewater effluent is no longer deteriorated allowing a higher amount of Qr_{NO_3} to be recycled and the pre-denitrification potential to be completely exploited. All Graphs and Tables included in this Section refer to this new plant configuration.

Each scenario impacted differently on the plant performance in terms of pollutants removal efficiency, effluent quality and energy saved, generating three maximum allowable recycling flows that assure effluent quality, one for each scenario. Once the maximum allowable flows were detected, the eco-

nomic factor became relevant in the selection of the most promising scenario to be tested in the real full-scale plant. The following lines illustrate the quality results of the three scenarios separately and afterwards they are compared by means of the economic balance.

Wastewater quality performances

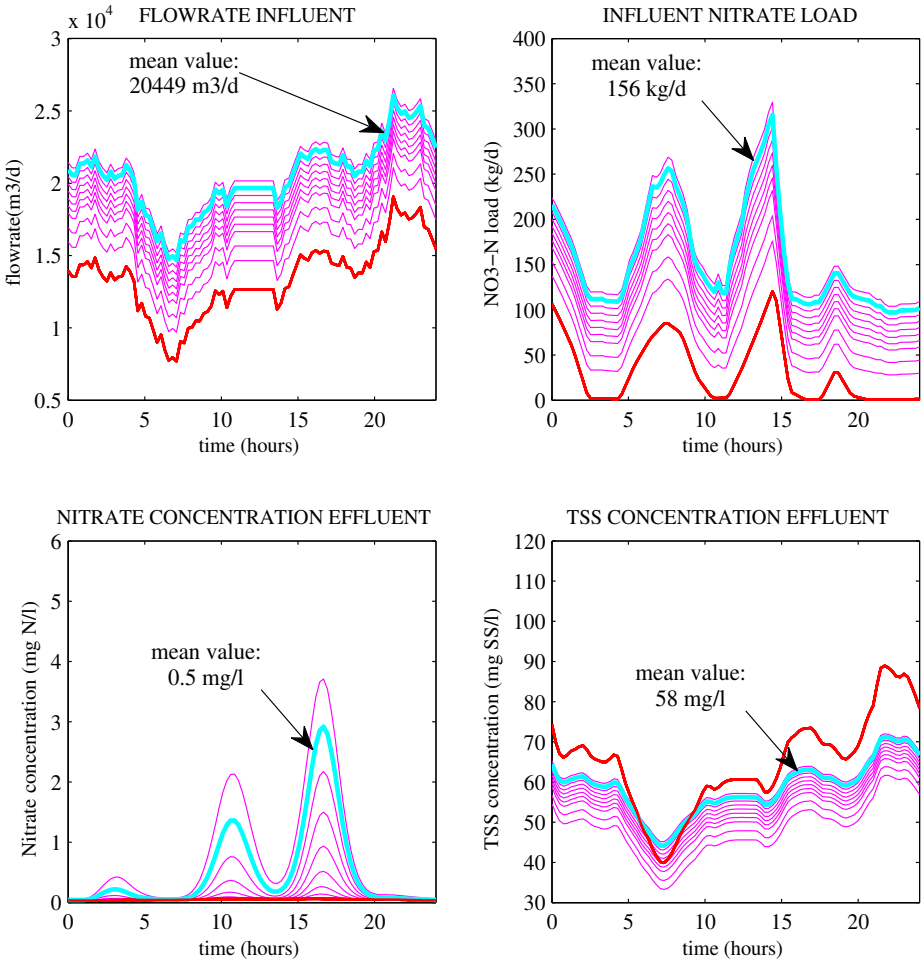

Figure 3.18: Constant recycle scenario: i) influent flow rate to the activated sludge unit ($Q_{mainstream}$+ Qr_{NO_3}); ii) nitrate load influent in the activated sludge unit; iii) effluent nitrate concentration; iv) effluent TSS concentration. The red line is the reference case. The blue line is the maximum allowable.

Constant recycle scenario

This scenario explored the effect of recycling Qr_{NO_3} at a constant amount over the time; each dynamic simulation tested an increasing recycling flow rate and each time the effluent quality was monitored. When the stream reached 7000 $m^3 d^{-1}$, effluent nitrate concentration was around 0.5 mg l^{-1}, but two peaks reached 3 mg l^{-1}. In order to not further exceed nitrate concentration,

this was assumed to be the maximum allowable Qr_{NO_3} recycle.

Evaluating the 7000 $m^3 d^{-1}$ case:

- The pre-denitrification potential appeared to act as the limiting factor: higher amount of flow would deteriorate the effluent in terms of nitrate concentration. The effluent peaks reflect the influent nitrate load peaks (plus the time delay equal to the hydraulic retention time).
- Total solids concentration in the effluent is generally lower than the reference case; therefore, the settler hydraulic capacity is not of concern.
- COD in the effluent assumed a similar concentration to the reference case as well.
- The maximum nitrate load treated in this scenario was 145 $\mathrm{kgNO_3\ d^{-1}}$. Considering that the current nitrate load removed is around 30 $\mathrm{kgNO_3\ d^{-1}}$—calculated from the measuring campaign data—this scenario increments the load treated 4.7 times.

Flow equalisation scenario

In this scenario Qr_{NO_3} is not pumped at a constant value, as in the case before; it is pumped according to the mainstream flow, in order to obtain a constant flow rate entering the activated sludge unit (see Fig 3.19).

Again, the pre-denitrification potential results to be the limiting factor, defining the maximum allowable recycle flow rate at 5500 $m^3 d^{-1}$.

Evaluating the 5500 $m^3 d^{-1}$ case:

- The nitrate effluent concentration is higher than the previous scenario. This fact could be explained by the highly variable nitrate load entering the activated sludge unit. Indeed, even though the influent flow rate was equalised through the Qr_{NO_3} recycle, only its quantity and not its quality was made constant. The periods of the day of more intense Qr_{NO_3} recycle coincide with the highest nitrate concentration in the NTF effluent, leading to very high nitrate peak loads in the activated sludge unit. During these peaks, the denitrification potential is reached and exceeded, therefore the nitrate leftovers are found in the effluent. Compared to the previous scenario both the mean and the peaks of nitrate concentration are higher. These reasons forced a restriction of the maximum allowable recycle flow rate to 5500 $m^3 d^{-1}$.
- Very good performance concerning the total solids effluent was obtained; TSS concentration became uniform over the day and the removal efficiency appeared high. The secondary settler was significantly improved by the constant influent flow, as is clearly visible in Fig 3.19.
- The maximum nitrate load treated in this scenario was 124 $\mathrm{kgNO_3\ d^{-1}}$, lower than the previous scenario.

Nitrate load equalisation scenario

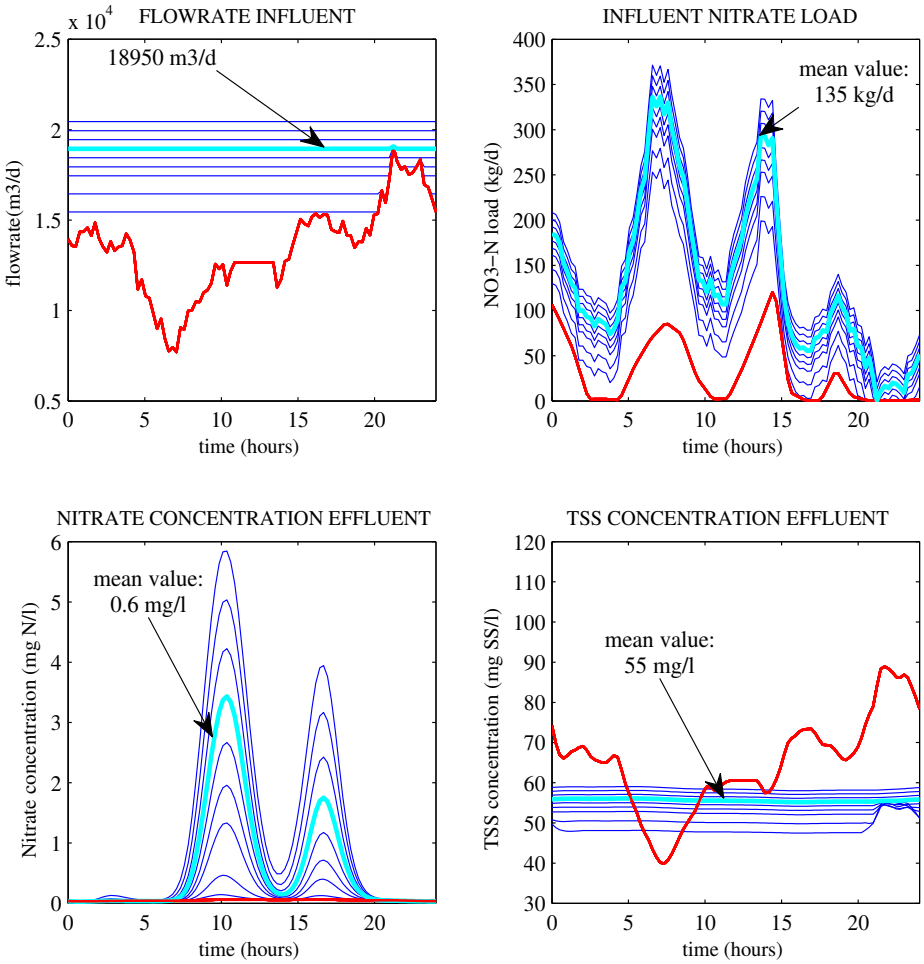

Figure 3.19: Flow equalization scenario: i) influent flow rate to the activated sludge unit ($Q_{mainstream}$+ Qr_{NO_3}); ii) nitrate load influent in the activated sludge unit; iii) effluent nitrate concentration; iv) effluent TSS concentration. The red line is the reference case. The blue line is the maximum allowable.

In this scenario the influent nitrate/nitrite load to the activated sludge was equalised. Qr_{NO_3} is pumped according to the SBR effluent, rich in nitrate, in order to obtain a constant load of nitrate entering the activated sludge section. Unlike the other two cases, the denitrification potential did not act as the limiting factor; instead, the secondary settler represents the critical point. TSS and COD concentration in the effluent are higher than for the previous scenarios and the loads removed lower. The maximum allowable recycle flow rate was 7500 m^3d^{-1}, according to effluent TSS and COD.

Evaluating the 7500 m^3d^{-1} case:

- The nitrate load in the activated sludge influent was made constant, leading to very low nitrate effluent and better performance of the pre-

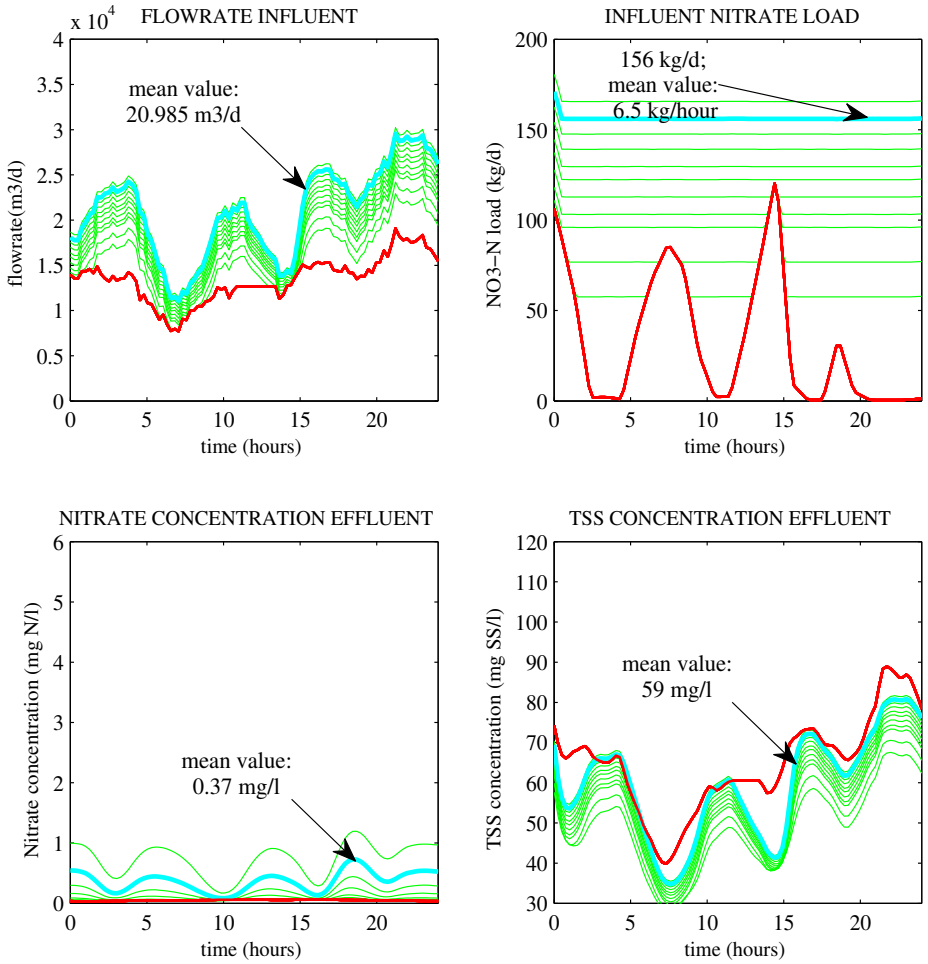

Figure 3.20: Nitrate load equalization scenario: i) influent flow rate to the activated sludge unit ($Q_{mainstream}$+ Qr_{NO_3}); ii) nitrate load influent in the activated sludge unit; iii) effluent nitrate concentration; iv) effluent TSS concentration. The red line is the reference case. The blue line is the maximum allowable.

denitrification reactor, resulting in uniform nitrate effluent and highest nitrate load removal.

- TSS and COD loads removed were lower compared to the other scenarios, but the quality was still around the reference case. Indeed, to achieve constant nitrate load influent flow, their amount became highly variable, with consequent bad performance of secondary settler.

- The maximum nitrate load treated in this scenario was 148 $kgNO_3\ d^{-1}$, the highest removal efficiency compared to the other scenarios.

Economic Savings potential The comparison balance was computed in economic terms considering the savings and the expenses over the day, for

Table 3.15: Comparison of nitrate loads in influent, effluent, and removed for the three scenarios and the reference case. In the last column removal efficiencies are reported.

	Influent kg d^{-1}	Effluent kg d^{-1}	Removed kg d^{-1}	Removal efficiency %
Reference situation[a]	31.5	0.7	30.9	97.9
Constant Recycle Scenario[b]	156.1	10.7	145.4	93.1
Flow Equalization Scenario[c]	135.6	11.7	123.9	91.3
Nitrate Load Equalization[d]	156.2	7.8	148.5	95

[a] Qr_{NO_3}=0 m^3 d^{-1}
[b] Qr_{NO_3}=7000 m^3 d^{-1}
[c] Qr_{NO_3}=5500 m^3 d^{-1}
[d] Qr_{NO_3}=7500 m^3 d^{-1}

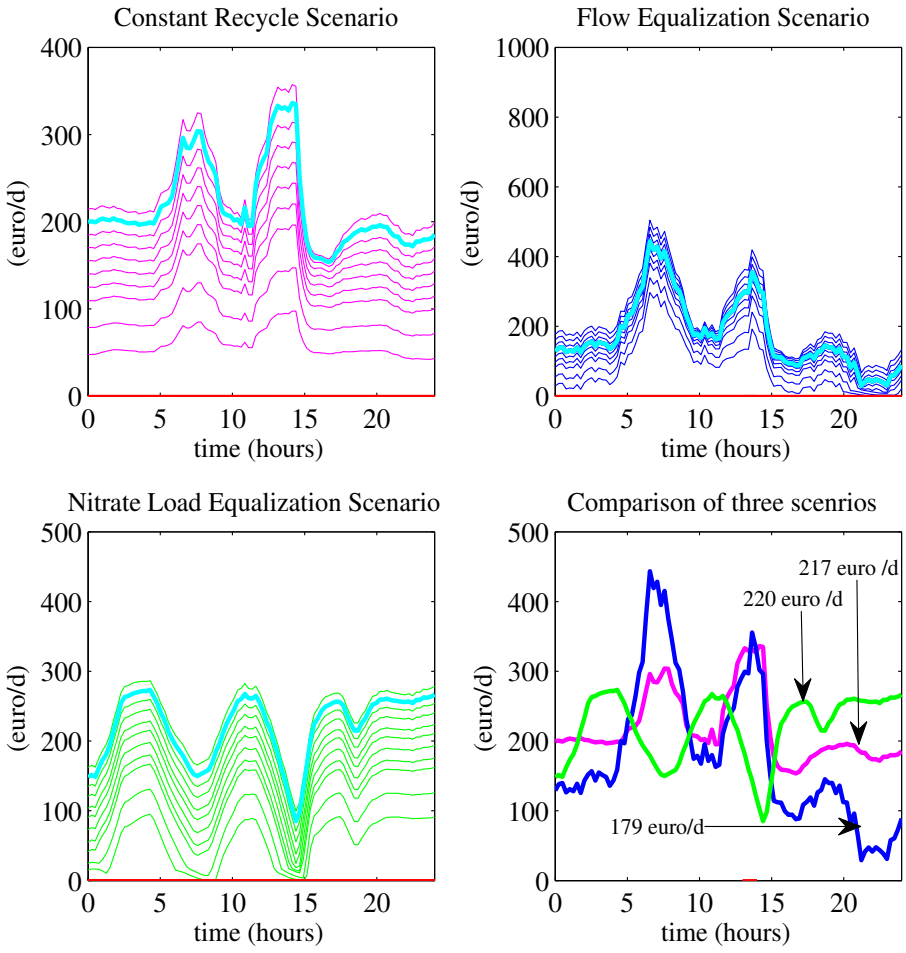

Figure 3.21: Economic balances for the three scenarios.

Table 3.16: Contributors of the economic balance, expressed in terms of euro

Contributors	Reference case	Constant recycle	Flow equalization	Nitrate load equalization
Qr_{NO_3} [m^3 d^{-1}]	0	7000	5500	7500
Methanol[€ d^{-1}] 1	0	+215	+178	+217
Aeration energy [€ d^{-1}]	0	+45	+34	+49
Pump energy [€ d^{-1}]	0	- 43	- 34	- 46
Balance				
1 line in 1 day [€ d^{-1}]	0	+217	+179	+220
4 lines in 1 day [€ d^{-1}]	0	+868	+717	+880
4 lines in 1 year [€ $year^{-1}$]	0	+316,966	+261,574	+321,142

each time step. Figure 3.21 illustrates how the increasing economic savings reflect the increasing Qr_{NO_3} recycled for each scenario (graphs A, B, and C). The last graph of the same Figure compares the saving curves of the three scenarios' maximum flow rates.

The mean daily savings for the three curves are reported in Table 3.16; even though the three scenarios have different saving potentials, an average value to refer to could be 200€ d^{-1}. Extending the daily saving values to the whole year and considering applying the same optimization solution to the other three equal parallel lines, we obtain an economic saving of between 261,574 and 321,142 € $year^{-1}$, according to the scenario analysed (summarised in Fig 3.21).

It is interesting to note that the *Flow equalisation scenario*, which produces less economic saving, has the highest specific saving per cubic meter of Qr_{NO_3} recycled. The *Nitrate load equalisation scenario*, has the lowest specific saving per cubic meter of Qr_{NO_3}, but nevertheless permits the highest economic saving since a higher flow rate (and therefore nitrate load) may be recycled.

The values reported in Table 3.21 are the result of the thorough economic balance; methanol cost saved has the biggest impact on the economic balance; the aeration energy cost saved is almost balanced by the cost for pumping the recycle flow to the head of the biological tank.

4 Conclusions

This study confirms the actual possibility of optimizing the energy balance efficiency of the high-loaded activated sludge system under investigation, by acting on its operating conditions. Energy efficiency was not the only aspect taken into account, but economic savings and the pollutants removal performances were combined to create one single purpose of improvement.

Two optimization strategies were explored and evaluated upon all three of these major factors, and the most promising solutions were selected and proposed for full-scale plant implementation and testing. The initial investment costs involved are limited, which is interesting from a plant management point of view. At the end of the whole comparison procedure designed to detect the best optimization strategies—in which the effluent quality plays the major role—the final comparison was made in energy and economic terms.

In this Section, the main conclusions regarding the energy efficiency strategies are followed by the model calibration and experimental work findings. It closes with a list of the study constraints, and the presentation of further research objectives.

The first optimization solution explored is the enhancement of biogas production. It was demonstrated how by starting from the current plant management conditions, there is still place for improving biogas production without altering the treatment efficiency. The operating condition that maximize the biogas production while assuring effluent wastewater quality were selected as the optimal one. It involves decreasing sludge age from 32 hours to 20 hours and at the same time increasing the oxygen supplied to the aerobic reactors. The wastage sludge extraction is increased from 190 to 320 m^3 d^{-1} and at the same time the oxygen concentration set-points are raised to 0.8 and 2 mgO_2 l^{-1} in the second and third aerobic reactor respectively (starting from 0.5 and 1.7 mgO_2 l^{-1}).

From the energy balance computed on the optimal condition, an increased energy of 1970 kWh d^{-1} emerged to be potentially produced. The lower sludge age is not likely to generate settling difficulties since the solids loading rate into the secondary settler is decreased from 2.6 to 1.7 kg TSS$(m^2\ h)^{-1}$.

The second optimization solution explored the possibility of improving pre-denitrification potential. It was achieved by recycling the Nitrifying Trickling Filter rich-nitrate effluent to the head of the biological tank. The way the recycled stream (Qr_{NO_3}) is pumped over the day is the key aspect that influences the whole plant performance. All three scenarios tested seem to generate an overall saving potential while preserving the effluent wastewater

quality.

Two scenarios out of three—the *Constant recycle scenario* and the *Nitrate load equalization scenario*—appear to better accomplish the aim of increasing the nitrate load treated; the higher the load of nitrate removed in pre-denitrification the higher the methanol saved and thus the higher the economical savings.

The *Constant recycle scenario* was found to be the most promising one. From an economic viewpoint it results similar to the *Nitrate load equalization scenario*, since both strongly improve the pre-denitrification potential, increasing the nitrate load treated by 4.7 times. On the other hand, it is easier to manage, being the recycle flow constant, and therefore this scenario is suggested for full-scale plant testing.

The *Flow equalization scenario* appears to not optimize the pre - denitrification potential, but shows strong improvement of the settler capacity and produces an effluent quality characterized by low and constant TSS concentration; focusing on this aspect, the implementation of this scenario in the full-scale plant may be employed as a convenient way of running the activated sludge system assuring an improved settling capacity.

The economic balance carried out on all three scenarios for each maximum potentially recyclable flow appears to create an average saving potential of around 800 € d^{-1}. The main contributor is the methanol saved in post - denitrification, whereas the aeration energy cost saved is balanced by the recycling pumping cost.

The first aspect addressed was the determination of the factor that limits the amount of possible recycle flow, among the pre - denitrification and the settling capacity. It was demonstrated how the already high hydraulic loading rate 1.2 $m^3(m^2\ h)^{-1}$ on the secondary settler prevents an additional stream inclusion.

The idea of extending the secondary settler volume to a currently unused settler compartment seems to be the only way of completely exploiting the pre-denitrification potential. The three scenarios tested in this new plant configuration appeared to be limited alternatively by the settling and the pre-denitrification potential; indeed, if the maximum recyclable flow is exceeded, effluent TSS or nitrate are increased respectively .

The model implemented for investigating the activated sludge under study was the Benchmark Simulation Model 1, which was able to simulate the plant in point in high-loaded condition, although it was considered one of the model constraints (Petersen, 2000).

The model calibration procedure was extensively outlined throughout the thesis; it was clearly illustrated how the additional information obtained from the experimental work helped decide on realistic model parameters and allowed a reliable calibration result. Identifiability problems did not occur thanks to the proper definition of realistic parameter ranges and to the help of the experiments performed on wastewater and activated sludge samples.

After steady-state calibration, the dynamic simulation outputs appear to well meet the dynamic trajectories measured during the monitoring campaign (especially for effluent COD and TSS). For this purpose, the monitoring cam-

paign carried out turned out to be essential. The results it offered—dynamic trends over the day of the main pollutants both in influent and effluent—were used to characterize the influent wastewater, to create the dynamic input file required by the model, to compare the effluent output of the model, and to calculate the pollutants removal efficiencies. Some measuring campaign results appeared to be out of range (especially for the effluent TSS concentration) when compared with the average routine laboratory data. The potential reasons are discussed in the previous chapter.

Respirometry tests carried out offered useful information about the evolution of heterotrophic bacteria activity over time. The test was performed twice; the first experimental trial helped to improve the final design, where the typical plateau curve was obtained, allowing a better definition of the maximum growth rate coefficient.
The main heterotrophic kinetic parameters were acquired through the test results and the sub-model calibration and validation: the maximum specific growth rate, μ_{maxH}; the decay rate coefficient, b_h; and the half-saturation coefficient, K_s.

The Zone Settling Column test was useful for investigating the plant settling performance and for experimentally determining the value of a settling parameter required by the Takács settling velocity model. The tests results showed good settling properties, even though the low sludge age could suggest settling difficulties. A possible reason is the low solids loading rate into the secondary settler.

Limitations

In the course of the work presented in this thesis, several types of problems and questions that deserve future attention have been encountered. In relation to the results that have been presented, a number of issues and extensions that need to be focused upon are defined below.

Some on-line sensors accuracies are not known and for some quantities, like the wastage sludge and recycle sludge flows, on-line sensors are the only way employed for recording them.

The Zone Settling Column test, was performed during a rainy day when the settling quality is generally different from a dry day, which is the weather condition assumed in the modelling. In addition, the experiment equipment was not perfectly in line with the one described in literature: the column employed had a smaller diameter and the activated sludge poured inside was not continually stirred.

Concerning the knowledge of the activated sludge quality, it was not accurately defined on the day of the measuring campaign (for practical problems), leading to some uncertainties during model calibration.

In available literature few papers describe the operating conditions and process performances of a high-loaded activated sludge plant. Understanding whether some of the parameters typically assume different values under these operating conditions would have been useful to better characterize the model.

With respect to the investigation for enhancing pre-denitrification potential, a model for the Nitrifying Trickling Filter was not employed, since it would have required a further extensive study; this could become the central

focus of a new research.

The presence of more on-line sensors capable of recording pollutants concentrations over the day would have helped during modelling, since the biological processes occurring in the activated sludge treatment plant are highly non-linear and time varying.

The pumps operation considered in the optimization solution II was assumed to be ideal being this a preliminary study.

Additional information required and further research proposed

Some considerations are required to outline how this study could be improved and extended.

The hydraulic behaviour of the several reactors could be explored through a tracer study. It would provide information about the degree of mixing of the aerated tanks and the overall hydraulic retention time (HRT). Indeed, most activated sludge reactors are not totally mixed and the entire volumes of the reactors are not used.

The three activated sludge lines parallel to the unit under investigation should be dynamically monitored in order to assure whether they are actually equal.

Further monitoring campaigns would provide proper data for the model validation.

Further investigation could be made in order to understand the potential of combining the two optimization solutions together.

This study could be extended by computing the carbon-footprint of the current plant operating conditions and comparing it with those obtained from the implementation of the best optimization strategies detected.

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