NEPTUNE

New sustainable concepts and processes for optimization and upgrading municipal wastewater and sludge treatment

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A Specific Targeted Research Project

under the Thematic Priority 'Global Change and Ecosystems'

Work Package 1 · Technologies for WWTP upgrading

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Author(s) in alphabetic order:	Corominas Ll., Flores-Alsina X., Joss A., Lettl W., Roels J., Siegrist H., Weemaes M., Petrov P., Vanrolleghem P.A.		
Contact for queries:	Natalija Miladinovic, Hansruedi Siegrist, Adriano Joss		
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CONTENTS

LIST OF FIGURES

LIST OF TABLES

1. Introduction

Several EU member states have yet to satisfy the stringent nutrient removal requirement of the Urban Waste Water Directive (1991) for sensitive areas. An environmental engineer working in the field of wastewater treatment is faced with increasing demands on the treatment efficiency. With the new European Water Framework Directive (EWFD, 2000/60/CE) there is a shift in paradigm from emission-based wastewater treatment plant (WWTP) effluent limits to an immission-based (from the river perspective) approach. Water policies in Europe and the world have shifted towards “demand management” and therefore have to incorporate new technologies in treatment and reuse. They prime the protection of aquatic ecosystems and point towards sustainable development. As a side-effect, even small plants have to fulfill the highest effluent limits if discharging into small water bodies. If we look into North America or Australia we see a trend to extremely low effluent limits, e.g. for total nitrogen as low as 3 mg N/L for sensitive areas. Moreover, recent evidence is showing that WWTP are on the verge of a second upgrading round not only for the increase in the legislation requirements but also considering that some of the WWTP constructed in the 80’s and 90’s are becoming saturated because of population and industry growth. An important role is given to the WWTPs to reduce the urban contamination of rivers to very low levels and therefore nutrient removal in biological treatment has to be improved and optimized.

Nutrient removal in biological treatment can be improved and optimised in different ways. **First**, WWTPs can be controlled using in-situ online sensors, significantly improving removal efficiency and reducing energy and chemicals consumption. There is a broad variety of sensors available that are providing reliable results with high resolution and low maintenance. Automatic process control is an excellent solution to adapt the plant to different loadings and to free already existing but unused capacities. To guarantee Europe’s position as world leader in advanced wastewater treatment technologies it is necessary to develop, test and implement control strategies. In order to find out whether sensor installation and automatic control are valuable upgrading options different ways of assessing the value of a set of measures are possible: economic and sustainability criteria can be evaluated to select the best operating and control schemes. **Second,** the addition of Zeolite can also improve nutrient removal since it may be possible to improve sludge settling conditions and therefore increase the activated sludge concentration and therewith the solid retention time in a given reactor volume. This solution is thought as a temporary measure for an overloaded plant with bulking and foaming problems. **Third**, separate treatment of the sludge return liquors which can contain about 20-30% of the raw wastewater ammonia load of a municipal WWTP, will significantly improve the BOD to ammonia ratio in the primary effluent. These sludge return liquors can be treated in-situ using for instance the autotrophic deamonification process. Fourth, insitu sensors can also be implemented in the sewer system and plant inlet to control industrial polluters and to prevent temporary overloading and/or inhibition of the WWTP by peak loads.

In the description of the work package 1 of the Neptune project ten tasks related to the improvement and optimization of nutrient removal in biological treatment have been proposed:

1. Test protocol for on-line sensors during field application
2. Test of different in-situ sensors for typical control parameters
3. Development of new calibration methods for an existing UV spectrometer probe for organic pollutants, nitrite and nitrate in view of process control
4. Development of new methods to continuously evaluate the data quality of on-line sensors during field application and to improve sensor maintenance planning
5. Application of a spectrometer probe with delta-spectroscopy as early warning system for the detection of toxic compound or industrial wastewater in sewer or plant inlet in incombination with in-situ sensors for ammonium, pH, redox and conductivity

6. Development and implementation of innovative control strategies for nutrient removal
7. Development of a methodology to assess the value of a set of control measures based on energy consumption, sludge treatment costs and effluent quality
8. Addition of modified natural zeolite to improve full-scale plant performance
9. Up-scaling and demonstration of separate treatment of ammonia rich digester liquid with autothrophic deammonification at the WWTP Zürich (Eawag) and WWTP St.Gallen (Hunziker)

The tasks 1 and 2 can be found in the Chapter 2 of this report. In the first part the validity of the available standardized protocol to characterize sensors is checked. Then an improvement of the protocol to account for field conditions is presented. Chapter 3 presents the investigation of a spectral in-situ UV sensor to measure nitrite and nitrate concentrations (task 3). Task 4 refers to methods for checking data quality of sensors and fault-detection and the results are presented in Chapter 7. The results of task 5 are presented in chapter 10. The tasks 6 and 7 are included in chapters 4, 5 and 6 where control strategies are evaluated using a benchmark system and using calibrated models of full-scale WWTPs. Different ways of evaluating the control strategies are used, including sustainability criteria (Life Cycle Analysis) and performing multicriteria analysis. The results of the research conducted for task 8 are presented in Chapter 8 and the return liquor treatment (task 9) are summarized in Chapter 9.

2. Sensor characterisation

Until the late nineties, sensors were seen as the main obstacle in introducing reliable process control in wastewater treatment. Nowadays, reliable sensors have been developed for nearly all compounds of interest applying different measuring principles (Jeppsson et al., 2002). The number of sensors on the market in water quality is large. There are different configurations and different manufacturers for most of the measured chemical compounds. It is difficult for the end users to make an informed choice among available sensors which is important since not all sensors will give satisfying results. In order to make a selection of sensors for a given application it is necessary to have a standardized procedure to make objective comparisons.

The main methodology available is the ISO 15839 standard, entitled “Water quality – On-line sensors/analysing equipment for water – Specifications and performance tests”. Although the ISO 15839 was published in 2003 there is no critical evaluation to assess its validity to objectively characterize sensors. Therefore, the first objective of this study is to characterize sensors according to the ISO protocol to evaluate the performance of in-situ and ex-situ sensors in the water quality field and to determine if the results coming out of those tests are meaningful and useful to select the right sensor for a specific application. The second objective is to develop a protocol for standardized field conditions that will allow evaluating the effects of field disturbances on the performance of the sensors. For this study this is limited to analyzing air bubbles and turbidity effects. This new protocol can be appended to existing protocols to develop a tool that will help end-users to select the right sensors for their application. Figure 1 shows different parts of the work of this study and how they are linked to each other. A detailed description of the results can be found in Beaupré (2009).

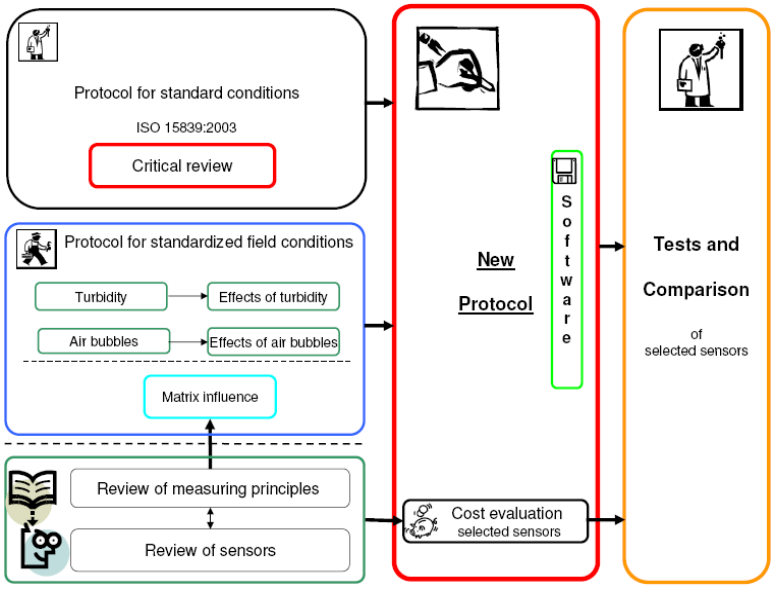

Figure 1. Approach used to develop a testing protocol for sensors in water quality management (Beaupré, 2009)

2.1. Water quality on-line sensors

The performance and reliability of many on-line sensors (e.g. nutrient sensors, flow and level meters) have improved remarkably during the last decade and can be used for monitoring but also in many different control strategies (Jeppsson et al., 2002). There are numerous sensor

systems available in the industry, and different possible classifications can be done (Vanrolleghem and Lee, 2003):

- Classification based on the functional application of the sensors:
  - Use in automatic control systems,
  - Use for monitoring purposes, i.e. providing information about the state of the plant.
- Classification based on the configuration of the monitoring systems (see Figure 2):
  - In-situ sensors: often simple, and low maintenance sensors that are used in trend monitoring and, due to their often short response time, in automatic control systems and
  - Ex-situ analyzers: often more complex maintenance-intensive analyzers that are typically found in plant effluent monitoring (because of the normally high accuracy).
- Classification based on the capacity of the sensor to be used in different locations of the wastewater treatment plant:
  - Influent
  - Effluent
  - Activated sludge reactor...
- Classification according to different measuring principles:
  - Colorimetric
  - Potentiometric
  - Ion sensitive
  - Optic

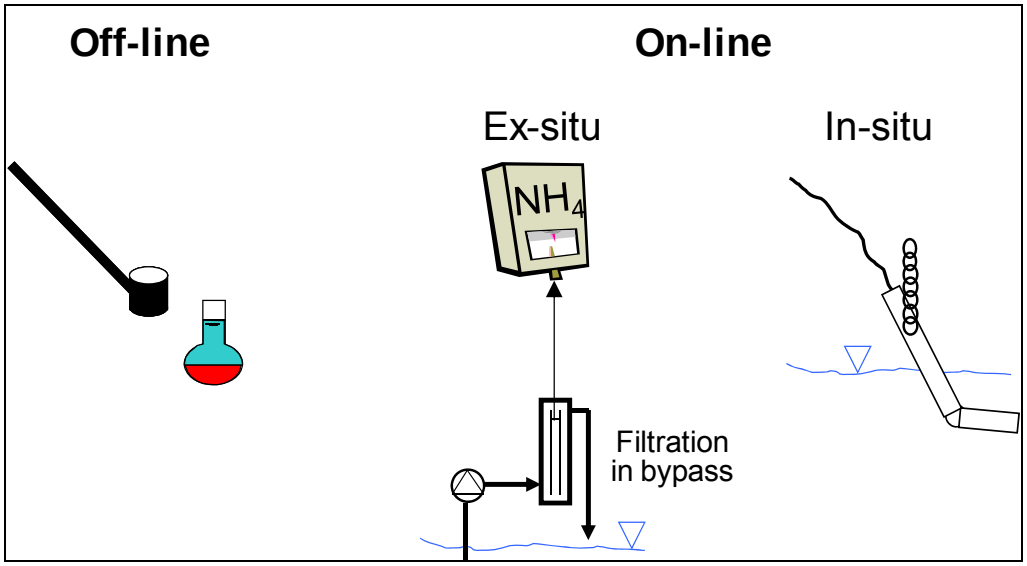

Figure 2: Definition of measuring systems based on sampling and installation

Table 1 presents a summary of different sensor technologies available, classified according to functional application, location and measuring principles.

Table 1. Objectives of control versus measuring principles

		Ex-situ					In-situ					
	Objective	**Ammonia** Colorimetric, IC, GS Electrode	**Nitrate, nitrite** Colorimetric, IC, UV	**Ortho-Phosphate** Colorimetric, IC	**P-total** Thermal chemical oxidation + colorimetric	**TOC, DOC, COD, BOD**	**Dissolved oxygen**	**Turbidity, TSS, Sludge blanket level in sec. clarifier**	**pH, redox, conductivity**	**Ammonia** Ion sensitive electrode (ISE)	**Nitrate, nitrite** UV, Ion sensitive electrode (ISE)	**Organic compounds** UV/VIS
Inf	Load monitoring Equalization of load					X		X	X	X		X
1ary inf	Load monitoring Aeration control					X		X	X	X		X
AS	Aeration control	X	X				X			X	X	
	RAS, internal recycle		X								X	
	WAS control/monitoring							X				
	P precipitation			X								
	Dosage of carbon source		X								X	X
Eff	Monitoring	X	X	X	X	X	X	X	X	X	X	X
	Control of P precipitation			X	X							

GS: Gas Sensitive, IC: Ion Chromatography, UV/VIS: Ultraviolet-visible optical measurement, TSS: Total Suspended Solids, Inf: Influent, 1ary inf: Primary influent, AS: Activated Sludge reactor, Eff: Effluent.

The demands on the measuring equipment are decreasing with ongoing treatment steps. Wheras only few sensors are robust enough to deal with the harsh conditions and the difficult water matrix in the WWTP influent, most of the available sensors can be used to monitor the effluent quality. The final choice should depend on the objective of the measurement. If the goal is to monitor whether the plant complies with the effluent limits, the most accurate sensor should be used. The best choice is then probably an ex-situ analyzer with auto-calibration. If only trends are followed or a low response time (for control) is required, the first choice is an in-situ sensor.

2.2. Standard protocol for sensor characterization

To date the standard ISO 15839 (2003) “Water quality – On-line sensors/analysing equipment for water – Specifications and performance tests” is the most complete test protocol available regarding characterization of water quality sensors. It contains two main parts: The first one determines performance characteristics under standard laboratory conditions, i.e. solutions with pure water and the measured component. The second part deals with performance characteristics in the field.

The laboratory tests are conducted with seven solutions equally distributed over the measuring range (i.e. at 5, 20, 35, 50, 65, 80, 95 %). Solutions are made with pure water and the compound measured by the sensor (e.g. ammonium, nitrate, phosphorous, etc.). For every concentration, six measurements are carried out. Depending on the characteristic to be calculated, measurements are taken on the same day separated by a blank or on different days.Table 2 shows this distribution.

Table 2. Use of measurements and constraints on scheduling (source: ISO, 2003)

Solution	Concentration (% of measuring range)	Determinant level used for	To be measured
1	5	LOD, LOQ	On the same day separated by blanks
2	20	Repeatability, LDC, bias	On the same day separated by blanks
3	35	Day-to-day repeatability	On different days
4	50	short-term drift	Equally distributed over shortest period between maintenance operations
5	65	Day-to-day repeatability	On different days
6	80	Repeatability, LDC, bias	On the same day separated by blanks
7	95	Linearity check only	On the same day separated by blanks

Measurements at 50% of the working range should be equally distributed over the time between two periods of maintenance. Since the maintenance intervals depend on the application of the probe and the sensors have not been in use elsewhere than in the lab, measurements at 50% of the working range have been carried out on six consecutive days. With the laboratory tests, the following characteristics can be determined:

Response time for positive change
Response time for negative change
Delay time for positive change
Delay time for negative change
Rise time
Fall time
Linearity
Coefficient of variation
Limit of detection
Limit of quantification
Repeatability
Lowest detectable change
Bias
Short-term drift
Day-to-day repeatability
Memory effect
Interferences
Environmental and operating conditions

The main idea of the field testing part is to expose sensors or analyzers to real conditions. Two ways are suggested: i) the measuring device could be installed directly in the field or ii) exposed to a grab sample from a real process. The problem with the first set-up is that additional uncertainties are introduced due to the variability of variables such as flow rate, concentrations, temperature, etc. The second approach includes the influence of the real water matrix on the test results. Problems to deal with are for instance biodegradation, stripping, etc. With this part of the ISO protocol, the following characteristics can be evaluated:

Response time for positive change
Response time for negative change
Delay time for positive change
Delay time for negative change
Rise time
Fall time
Bias based on (relative/absolute) differences
Long-term drift
Availability
Up-time

2.2.1. Applicability of the ISO protocol

A review has been made to evaluate the applicability of the ISO protocol by the companies that sell sensors. The review has been limited to the sensors analyzing ammonium, nitrate, nitrite, phosphate, TSS and dissolved oxygen.

To evaluate which information is provided by the manufacturers, tables have been made to classify the collected data. Table 3 presents a summary of the information provided by each manufacturer. In the left column three categories have been created: 1) Technical information, 2) accuracy and 3) cost.

Table 3. Analysis of the information provided by manufacturers

			Measured compound					
			NH_4	**NO_3**	**NO_2**	**PO_4**	**TSS**	**DO**
		Total number of sensors found	***24***	***24***	***6***	***15***	***16***	***18***
		% of sensors giving a value for the following characteristics						
technical information		filtration required	50	58	83	80	0	0
		ex/in situ	100	100	100	100	94	100
		Needs housing	21	0	0	0	0	0
		Spectral range	13	25	50	20	13	-
		Number of wavelenghts	13	13	50	20	-	-
		measuring range	71	83	50	80	81	89
		resolution	4	100	17	7	6	0
		response time	25	17	50	33	6	11
Accuracy	acording to ISO 15839:2003	Linearity	0	0	0	0	6	0
		Coeficient of variation	4	4	17	7	6	0
		Limit of detection	13	21	0	7	6	0
		Limit of quantification	0	4	0	0	0	0
		Repeteability	17	8	0	0	50	6
		lowest detectable change	0	0	0	0	0	0
		bias	0	0	0	0	0	0
		short term drift	4	0	0	7	0	11
		day-to-day repeatability	0	0	0	0	0	0
		memory effect	0	0	0	0	0	0
		interference	8	0	0	7	0	0
		T_{90}	29	33	17	20	38	72
	Other	reproducibility	25	17	0	20	0	11
		Precision	21	21	50	47	13	6
		accuracy	25	67	50	20	44	39
cost		investment cost	0	0	0	0	0	0
		spare and wear parts cost	0	0	0	0	0	6
		annual cost w/o labor	100	100	100	100	100	100
		annual cost w labor	100	100	100	100	100	100
		manual maintenance duration	8	21	17	13	6	0
		manual maintenance interval	25	38	17	33	0	0
		reagent consumption	46	50	83	60	6	0
		consumable cost	0	0	0	0	0	0
		power consumption	25	21	0	33	6	28

As an outcome of the review only few companies provide sufficient information on their sensors and none publish the full ISO test results.

2.3. Example of sensor characterization

The goal of this chapter is to characterize sensors under different conditions using the ISO 15839:2003 protocol. In the next sections the sensors tested are described and then the results of the tests are shown. The section finishes with a critical discussion of the ISO protocol.

2.3.1. Sensors tested

- Spectro::lyser™ (scan Messtechnik GmbH, Austria): The used spectro::lyser™ is a spectrometer covering the UV spectrum from 190 to 390 nm with a path length of 35 mm. The sensor is able to measure nitrate from 0 to 15 mg NO_3-N/l. The auto-cleaning is performed with pressurized air. The probe comes with a global calibration (the one used was the river calibration). On top of global calibration it is possible to add a local calibration or even specific calibration (Rieger et al., 2006). For this work, a local calibration has been done with two points, at 20 and 80 % of the working range, i.e. 3 and 12 mg NO_3-N/l. The probe is connected to a computer via a relay box; this box provides power to the probe and is processing the signal to a USB port of the computer. s::can is also providing different software versions, the one used is ana::pro V5.3e-2399. For continuous measurements, the measuring frequency has been set to the minimum, i.e. one minute. Tests with this probe have been conducted at Université Laval, Quebec, Canada. It should be noted that the probe is not covering the visible range and is therefore sensible to changing turbidity concentrations. The test was carried out to show a potential negative impact of the water matrix on the results.

- Sensor A: Sensor A is a spectrometer using the same measuring principle as the s::can probe. The difference is that the measuring cell is not directly immerged in the water: there is a small piston which is pumping water into a measuring cell. The light source and the detector are on each side of this cell. The idea is to measure settling properties and make measurements in the presence of less turbidity. The measuring range of sensor A is 0-20 mg/l of NO_3-N. Every time the piston is moving in the tube, it should clean it. The calibration used for the test is the factory calibration. Before testing, the probe was referenced in pure water also used to make the test solutions. The probe was used in manual batch mode. Tests were conducted in the company's laboratory in Germany.

- Nitratax (Hach): The measuring principle of this probe is using two specific wavelengths, instead of 256 as for the spectrometer probes. The nitrate concentration is measured at 210 nm and a turbidity correction is made with a measurement at 350 nm. The path length of the probe used is 1mm allowing the probe to measure nitrate (as NO_3-N) concentrations from 0 to 100 mg/l. The auto-cleaning of lenses is made by a wiper. No additional calibration has been made for the conducted tests (the calibration made by the technician of the company was used). The probe is connected to the SC100 controller that is connected to a central server for data logging. During continuous measurements, the measuring interval was set to the minimum setting of 15 sec. and filtering was set to zero. Tests with this probe have been conducted at EAWAG in Duebendorf, Switzerland.

- Sensor B-8mm and B-2mm: This sensor is still in development and is not yet on the market. It is also measuring nitrate with the two wavelengths principle. Two path lengths are available for this model: 2 mm with a measuring range of 0 to 50 mg/l of NO_3-N and 8 mm with a measuring range of 0 to 10 mg/l of NO_3-N. Both configurations were tested for this work. The auto-cleaning of the sensor is made by pressurized air. The calibrations used during the experiments were factory calibrations. The sensors were connected to the controller provided by the company. Signal filtering and damping were set to zero. Tests were conducted at the company's laboratory in Germany.

- Sensor C: A two wavelengths measuring principle is used for this sensor too. It can measure concentrations from zero to 50 mg/l of nitrate as NO_3-N. The auto-cleaning of lenses is ensured by pressurized air. The calibration used during experimentation was the factory calibration. The probe was connected to the controller provided by the company. The filtering was set to zero. Tests were conducted at the company's laboratory in Germany.

- SOLITAX (Hach): The SOLITAX is a turbidity sensor. It is combining two different scattering angles: 90° and 140° which are allowing the sensor to measure turbidity from 0.001 to 4000 NTU. The auto-cleaning of the lens is made by a wiper. The calibration used during the experiments was the factory calibration. The sensor was connected to a SC1000 controller from Hach that was connected to a computer to download the log files. During continuous experimentation the measurement frequency was set to the minimum, i.e. 5 seconds. Tests were conducted at Université Laval, Quebec, Canada.

2.3.2. Results of the tests

Table 4 shows the characteristics determined based on the ISO 15839:2003 protocol of different nitrate sensors. The table is built according to suggestions in the annex of the ISO 15839:2003, where the different characteristics are in rows and the different tested sensors are in columns. The closer the results are to zero the better the sensor is for the given characteristic. A graphical comparison can be found in Figure 3.

Table 4. Characteristics of nitrate sensors according to the ISO 15839:2003 testing protocol (LDC: Lowest Detectable Change, DD: Day-to-Day)

Characteristics	Units	s::can	Sensor A	Sensor B (8mm)	Sensor B (2mm)	Sensor C	Nitratax
coefficient of variation	%	6,6915	2,9327	3,4533	4,0384	**1,8493**	1,9751
limit of detection	mg NO_3-N/L	0,1250	0,0455	0,0351	0,0245	**0,0155**	0,0923
limit of quantification	mg NO_3-N/L	0,4167	0,1517	0,1169	0,0816	**0,0516**	0,3077
repeatability at 20 %	mg NO_3-N/L	0,0232	0,0232	**0,0089**	0,0450	0,0516	0,0516
repeatability at 80 %	mg NO_3-N/L	0,1959	**0,0273**	0,0494	0,0799	0,3082	0,0408
LDC change at 20 %	mg NO_3-N/L	0,0695	0,0695	**0,0268**	0,1351	0,1549	0,1549
LDC change at 80 %	mg NO_3-N/L	0,5878	**0,0820**	0,1482	0,2396	0,9247	0,1225
bias at 20 %	mg NO_3-N/L	0,7217	-0,0983	**0,0200**	0,3633	0,3333	0,4267
bias at 80 %	mg NO_3-N/L	**0,0650**	2,0567	0,3000	4,8617	4,6500	5,1767
short term drift	%/day	0,0248	**-0,0171**	0,0371	0,1103	0,2114	-0,1943
DD repeatability at 35 %	mg NO_3-N/L	0,5270	0,0407	**0,0052**	0,1036	0,3327	0,4502
DD repeatability at 65 %	mg NO_3-N/L	0,0886	0,1202	**0,0475**	0,1993	0,5282	0,8124

Overall, it can be seen that the sensors A and B (8mm) are better in terms of repeatability, bias and drift. Sensor C shows better performance for the coefficient of variation and the limits of detection and quantification. However, it can be difficult to select the right sensor because there is no single sensor that shows better performance for all characteristics. Depending on the use of the sensor (monitoring, control, etc.) and the location (WWTP, river, lake, etc.) some criteria will be more important than others. As an example, if the sensor is used for control purposes in an aeration tank of a WWTP, repeatability is an important criteria (in this case sensors A and B 8mm) because lower noise helps to do better control. In that case the drift is not a big issue because maintenance and calibration can easily be conducted by the operators. On the other hand, if the sensor is used to perform monitoring in a remote location, then the drift is an important characteristic because maintenance intervals will be long (again, sensors A and B 8mm appear best).

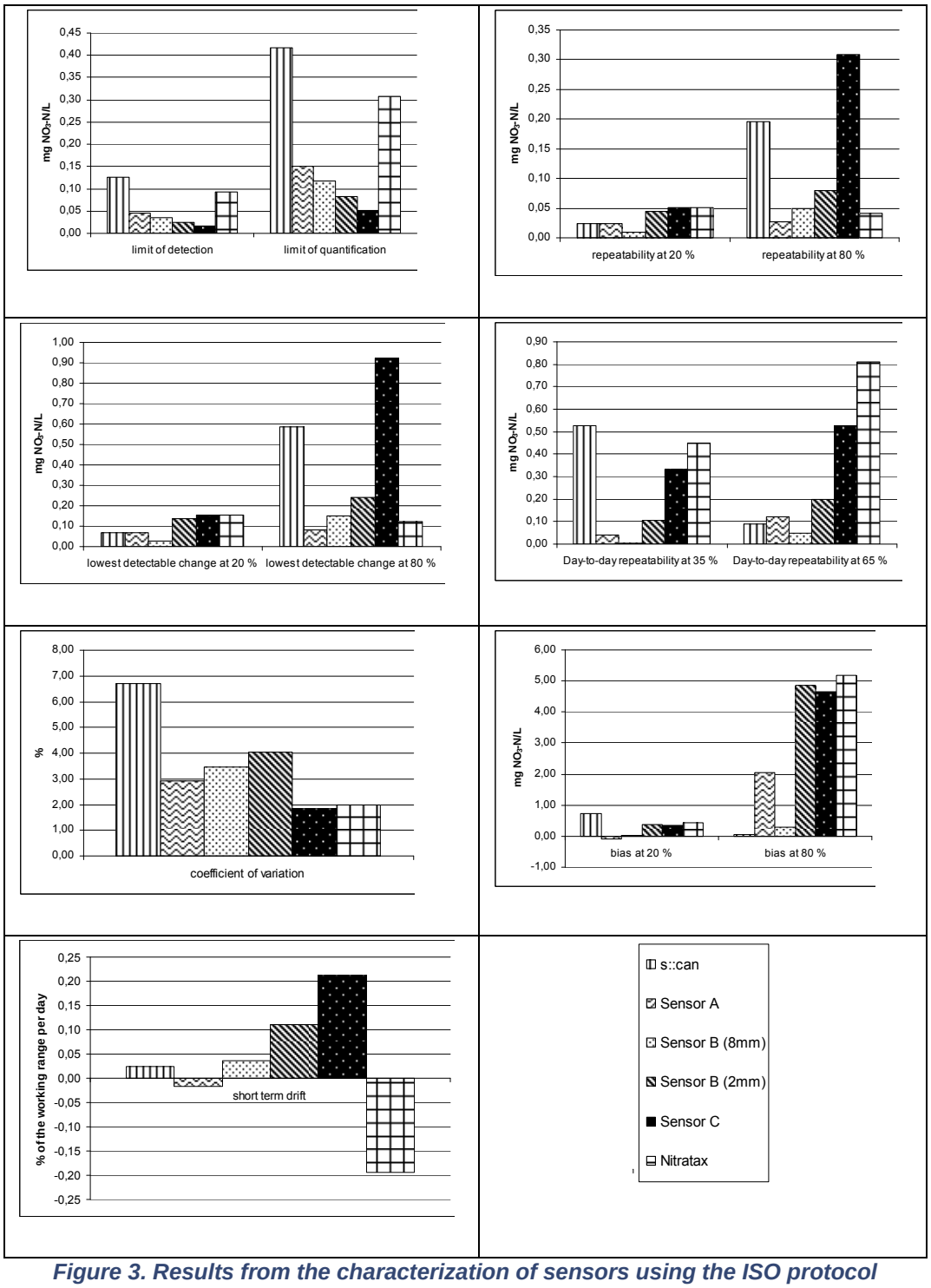

Figure 3. Results from the characterization of sensors using the ISO protocol

2.3.3. Analysis of the ISO protocol in practice

After checking the ISO protocol with different nitrate optical sensors, we have identified different improvements:

- **The comparison of sensors should be performed for the same measuring range:** According to the ISO 15839:2003 protocol the sensors are tested at their measuring range. However, some of the criteria (imits of detection and quantification, repeatability, lowest detectable change, day-to-day repeatability, bias and coefficient of variation) are influenced by the measuring range.

- **Results of drift:** They should be presented in a graphical way giving the measurement versus time.

- **Results interpretation:** The ISO 15839:2003 protocol should provide guidelines for end-users to interpret the results.

An important limitation of the ISO standard is the part on field conditions. In the ISO 15839 standard the field condition testing is not complete. This part of the protocol evaluates the performance of the whole measuring chain without making a distinction between the different segments (pump, filtering device, measuring cell, signal transmission, etc.). There is a need to have standardized field conditions to easily compare sensors without the need of doing tests in the field exactly at the same time and location. Moreover it will be easier to isolate the different segments of the measuring chain, to perform spot checking and to evaluate the effect of disturbances separately.

2.4. Test protocol for sensors to simulate field conditions in standard laboratory conditions

This section aims at presenting the extension of the ISO protocol with a protocol to characterize sensors under some field conditions. This is illustrated with experiments conducted in the laboratory. Two interferences occurring in the field have been reproduced in the lab; air bubbles and turbidity. Air bubble effects have been tested on a turbidity sensor (SOLITAX) and on nitrate sensors (Spectro::lyser™ and Nitratax) and the effect of turbidity has been tested only on the Spectro::lyser™.

2.4.1. Air bubbles

Experimental Setup

To evaluate effects of air bubbles in a standard way a laboratory setup has been built where conditions are controlled. A 170 liters plastic tank has been equipped with a nine inch diameter diaphragm diffuser (Diffuser Express) (see Figure 4). The diffuser was supplied with laboratory pressurised air and regulator with pressure gauge. The Spectro::lyser™ and SOLITAX sensors have been installed in the tank according to the manufacturer installation requirements.

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Figure 4. Tank equipped with diaphragm aerator

Experimental Procedure

First, the tank is filled with the solution at the desired concentrations of the measured variable. The experiment starts without air supplied during 30 minutes. Then the air pressure is adjusted to 0.5 psig for another 30 minutes. After this step the pressure is incremented up to 1.0 psig and 1.5 psig. After reaching the maximum, the pressure is lowered by decrements of 0.5 psig every 30 minutes until reaching zero.

The Spectro::lyser™ has been tested in solutions without nitrate (tap water) and in a solution with a nitrate concentration around 3 mg NO_3-N/L, corresponding to 20% of its measuring range. The SOLITAX analyser has been tested at 0 FNU.

Results

Figure 5 shows some of the measurements using the Spectro::lyser™ at different air pressures in a nitrate-free solution. The noise considerably increased with the air preasure (see also in Table 5 the standard deviation and the mean of the measurements for each pressure). Depending on the pressure value the standard deviation was amplified between six to nine times compared to the one without aeration in the tank. A second effect is the bias that is generated. The offset generated ranges from 0.8 to 1.5 mg NO_3-N/L depending on the pressure. For the Spectro::lyser™ there is no correlation between the air pressure and the number of NaN.

Figure 6 presents the results of the SOLITAX sensor measuring turbidity at different air pressures. As with the Spectro::lyser™ the standard deviation of turbidity values increases when the air pressure is incremented. Another thing that can be observed is that at the end, even with the aeration stopped, there is still a bias. This could be caused by the presence of remaining air bubbles in the tank after the aeration was stopped. However, the SOLITAX compared to the Spectro::lyser™ is less affected by the presence of air bubbles (bias is within the lowest detectable range), no output NaN was obtained and the amplification of noise is low.

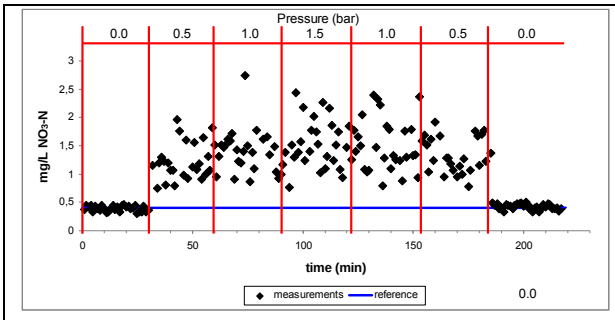

Figure 5. Nitrate concentration vs time for different air pressures for Spectro::lyser for a solution at 0 mg NO3-N/L

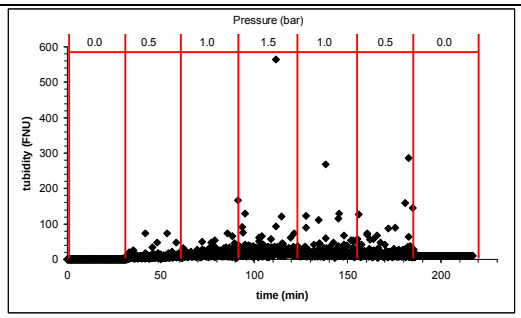

Figure 6. Turbidity vs time for different air pressures for SOLITAX for a solution at 0 FNU

Table 5. Mean and standard deviation for each pressure for Spectro::lyser for a solution at 0 mg NO_3-N/L

Pressure	mean	standard deviation	NaN
(bar)	(mg NO_3-N/L)		
0,0	0,39	0,05	0
0,5	1,22	0,32	4
1,0	1,41	0,38	5
1,5	1,54	0,42	2
1,0	1,51	0,45	3
0,5	1,29	0,35	6
0,0	0,42	0,05	0

Table 6. Mean and standard deviation for each pressure applied to the Solitax for asolution at 0 FNU

Pressure	mean	standard deviation	NaN
(bar)	(FNU)		
0,0	0,32	0,04	0
0,5	8,32	7,02	0
1,0	14,38	7,69	0
1,5	22,36	32,92	0
1,0	19,01	19,34	0
0,5	17,23	19,16	0
0,0	9,30	0,09	0

2.4.2. Turbidity

Experimental Setup

The effects of turbidity have been tested on the Spectro::lyser™. The Spectro::lyser™ is coming with a recipient that fits around the measuring cell (see Figure 7).

Figure 7. Offline measuring cell for Spectro::lyser™

Experimental Procedure

The test to evaluate the effects of turbidity on the nitrate measurements was performed as follows. First, a solution containing nitrate (approximately 3 mg NO_3-N/L) and nearly zero turbidity is introduced in the cell. A first series of measurements is made during 30 minutes. Then, the solution is spiked with the turbidity standard (AMCO Clear®) to reach a turbidity value in the cell around 10 FNU. Another 30 minutes of measurements is made with this solution.

Results

Figure 8 shows the effects of turbidity on nitrate measurements with the Spectro::lyser™.

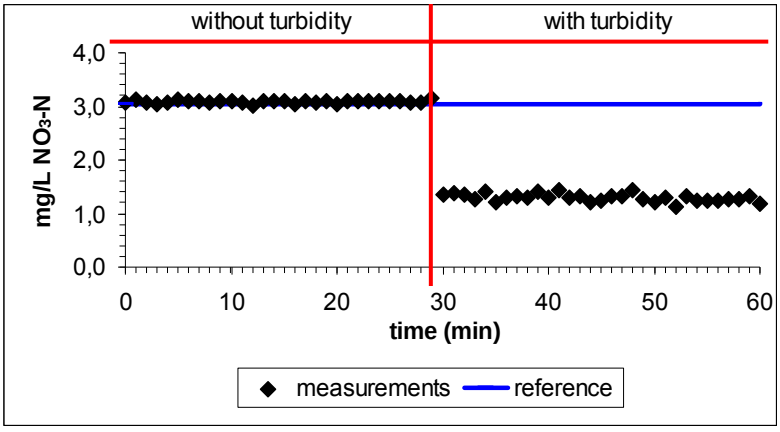

Figure 8. Effects of turbidity on nitrate measurements of the spectro::lyser (3 mg/L NO3-N and 10 FNU)

As seen in Figure 8 the addition of turbidity creates a negative shift of more than 50% (see Table 7). It would be logical to think that turbidity would generate a positive shift on an optical nitrate measurement because it is based on absorbance at a particular wavelength and particles in suspension absorb light in the entire measuring range. However, the software of the sensor compensates for the higher absorbance due to turbidity. Another effect observed in Figure 8 is that the measurement noise is increased by the presence of turbidity. Table 7 gives the standard

deviation of the measurements before and after the addition of turbidity. The measurement noise with turbidity is more than twice the one of a clear solution.

Table 7: Mean and standard deviation for each turbidity (0 and 10 FNU) with the Spectro::lyser^{tm} for a solution at 3 mg NO_3-N/L

turbidity (FNU)	mean (mg NO_3-N/L)	standard deviation
0	3,08	0,029
10	1,30	0,070

2.4.3. Conclusions

The ISO 15839:2003 protocol is the most complete protocol available to date to characterise sensors for water quality. The part of the protocol about laboratory testing has been applied to different sensors to figure out why manufactures are not providing the characteristics deduced from this protocol. In contrast to the laboratory testing part, the ISO 15839:2003 protocol accuses a lack of information concerning field testing. What is currently included in the protocol is not sufficient and it is certaintly not standardized. That is why a procedure to standardised field conditions has to be developed. A proposal for a protocol to test sensors subject to bubbles and turbidity has been presented. A procedure to evaluate effects of field conditions on measurements has been developed and the following conclusions have been achieved.

- The effects of air bubbles, i.e. noise amplification and bias, are proportional to the air pressure supplied to the air diffuser. For the Spectro::lyser™ there is no correlation between the air pressure and the number of NaN. The Solitax is less affected by the presence of air bubbles (bias is within the lowest detectable range), no output NaN was obtained and the amplification of noise is low.

- The second interference simulated was turbidity. The turbidity standard AMCO Clear® is more stable (less settling) compared to formazin. In addition, AMCO Clear® has the advantages of being non toxic, having a more uniform particles and having a longer shelf life. The same indicators of interference as for the air bubbles have been used. In this case no NaN occurred. In the case of the Spectro::lyser™, the presence of turbidity generated a negative bias and the standard deviation was increased.

It is possible to reproduce standardized field conditions in the lab as it has been demonstrated with the air bubbles and the turbidity examples. Further investigation in this direction is required to have a significant amount field conditions reproduced in the lab that can be useful to objectively compare sensors.

3. Long-term evaluation of a spectral sensor for nitrite and nitrate

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This chapter has been published as:

Rieger L., Langergraber G., Kaelin D., Siegrist H. and Vanrolleghem P.A. (2008) Long-term evaluation of a spectral sensor for nitrite and nitrate. *Wat. Sci. Tech.*, 57(10), 1563-1569.

3.1. Introduction

Nitrite is an intermediate product of the two-step process of nitrification. Since the second step of nitratation is very fast, the nitrite concentration in the effluent of a WWTP is normally very low (around 0.1 mg/l). Enrichment of nitrite in the system usually suggests that the microbiological processes are disturbed, i.e. they are inhibited due to toxic substances or to unfavourable conditions for the nitrite oxidiser. Since nitrite is a strong poison for fish, high nitrite concentrations in the effluent of WWTPs can lead to damage to organisms if the dilution of the receiving water body is too low.

Since laboratory measurements of grab samples can give only a snapshot picture and the analysis of 24h-composite samples is critical due to the unstable nitrite concentration, their meaningfulness is limited. The use of on-line analysers, which need a high sample preparation time, can lead to similar problems. In-situ sensors would therefore be advantageous in this case.

Nitrate, beside ammonia, makes up the major part of the total nitrogen concentration in the effluent of WWTPs and is therefore an important operational indicator and is also often used for legislation purposes. Measuring both parameters on-line will increase the monitoring capability and therefore enable appropriate control of the plant in order to prevent ecologically harmful discharges. Measuring them with a single probe will reduce the effort and therefore the costs.

Starting with first tests in 2003 (Rieger *et al.*, 2004) a spectral in-situ sensor was tested over a period of one and a half years in the effluent of a pilot plant in Switzerland. The sensor provides measurements of spectra between 210 and 400 nm (UV range) for every measuring cycle. The goal was to analyse i) the long-term robustness of the optical equipment and ii) the calibration stability of the underlying spectral analysis model.

The optical equipment is exposed to the difficult environmental conditions of a WWTP and therefore the problem of aging of light source and detector or changes in the optical path (e.g. due to scratches or precipitation on the optical lenses) could cause measuring errors.

The second question was to determine whether the calibration is stable over a prolonged period. It should be kept in mind that the applied method is not directly measuring nitrate or nitrite but some spectral information. Although nitrate and nitrite have known peaks in the spectra (around 200 – 250 nm, see Figure 9), the system has to deal with interferences stemming from organic matter and other disturbing ions which show an absorption in the same wavelength range or particles that block the optical measuring path. Since the interferences are of different magnitude for different water matrices, the calibration stability is not guaranteed.

3.2. Material and methods

The tested in-situ spectrometer (spectro::lyser, s::can Messtechnik GmbH, Vienna, Austria) measures the absorbance of ultraviolet light (UV from 210 to 400 nm) with a path length of 10 mm designed to enable differentiation between nitrite and nitrate at TSS concentrations up to 15 mg/l (manufacturer's specification). Physically, 256 wavelengths are measured between 210 and 400 nm (resolution ca. 0.8 nm) and these are converted to a resolution of 1 nm for calculating the concentrations. A single evaluation of the entire spectrum typically takes 15 seconds. Measuring the UV absorbance is an indirect method of determining water compounds. In principal, the sensor can be calibrated for all absorbing substances, typical applications in the water sector being measuring concentrations of organic matter, nitrate and nitrite (Figure 9).

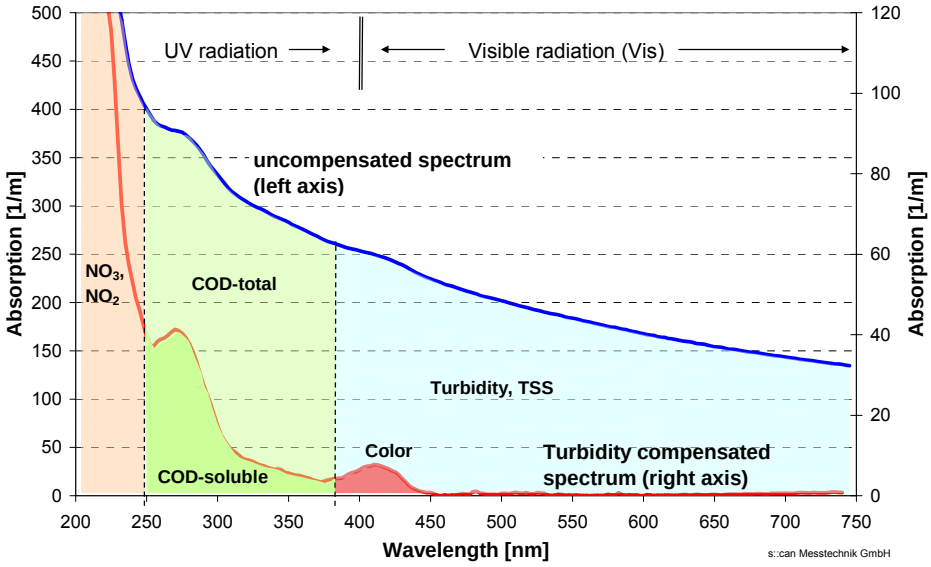


Figure 9: Absorption of different compounds within the spectrum from 200 to 750 nm (van den Broeke et al., 2006)

The spectrometer is constructed as a compact submersible sensor which enables optical spectra to be measured directly in liquid media with an accuracy approaching laboratory analysis quality. The spectrometer is equipped with an auto-cleaning system using pressurized air which has been proved to be extremely reliable (Gruber *et al.*, 2006). More information about the sensor can be found in van den Broeke *et al.* (2006) and Langergraber *et al.* (2003).

For typical waters (e.g. municipal wastewater – raw and treated, river water, drinking water etc.) the manufacturer provides a so-called *global calibration* as a default configuration of the in-situ spectrometer. To enhance the precision, a *simple local calibration* that is based on grab samples analysed in the lab has to be carried out that considers the different composition of the wastewaters to be analysed. By performing a *local PLS (Partial-Least-Square) calibration* one can improve trueness, precision and long-term stability of the results by finding a set of wavelengths better adapted to the matrix of specific wastewater (Rieger *et al.*, 2006).

3.2.1. Effluent of EAWAG pilot plant

The EAWAG pilot-scale wastewater treatment plant (Switzerland) treats the wastewater of approx. 70 p.e. (≈ 27 m^3/d inflow) and receives municipal wastewater mixed with an unknown

amount of industrial discharge. It is operated for carbon removal, full nitrification and pre-denitrification. The goal of the study was to evaluate different DO control options and especially the influence of low concentrations of dissolved oxygen in the biological stage on possible accumulation of nitrite in the effluent and measures to control it by means of changes to the aeration intensity.

The spectrometer was installed in the effluent of the secondary clarifier and used the calibration settings of a local PLS calibration that was based on a data set from another WWTP (Thunersee, Switzerland) plus lab experiments with standard addition on different matrices (Rieger *et al.*, 2004). For the reference analysis, grab samples were taken directly beside the sensor and analyzed in the EAWAG laboratory. The working range of the sensor was between 0 and 7.55 mg NO_2-N/l and 0 to 14.8 mg NO_3-N/l with median values of 0.4 mg NO_2-N/l and 5.7 mg NO_3-N/l, respectively. To test the sensor accuracy also for higher concentrations, measurements from a spiked vessel (normal effluent water matrix with addition of a high concentrated stock solution) were taken from time to time.

During the start-up phase (winter 2004/2005) an additional on-line analyser with in-situ filtration unit (TresCon NO_2, NO_x, in combination with PurCon IS, WTW, Weilheim, Germany) was installed in the effluent of the secondary clarifier of the EAWAG pilot plant to compare the results from wet-chemistry (nitrite) and on-line UV analysis (NO_x = NO_2 + NO_3) with the spectral in-situ sensor.

3.2.2. Lab analysis

The reference measurements for nitrite and nitrate were carried out in the EAWAG laboratory using the flow injection analysis method (FIA, ASIA, Ismatec AG, Glattbrugg, Switzerland) and an ion chromatograph (761 compact IC, Metrohm AG, Herisau, Switzerland) depending on the measuring range.

3.2.3. Evaluation procedure

For the evaluation of the sensor uncertainties based on comparative measurements a procedure according to Rieger *et al.* (2005) was used.

3.3. Results and discussion

3.3.1. Validation of the sensor calibration

Directly after installation at the EAWAG pilot plant, experiments were carried out to check whether the Thunersee calibration is also applicable at the new location. The sensor was put into a vessel with effluent wastewater and a number of standard additions were carried out, taking grab samples from each addition step. With this procedure it should be tested whether the sensor is able to detect higher nitrite concentrations with sufficient accuracy (as the project deals with increased nitrite concentrations). For nitrite these experiments showed excellent results, while for nitrate an offset occurred that could be eliminated by a simple local calibration, as described by Rieger *et al.* (2006).

Figure 10 and Figure 11 show the comparative measurements for nitrite and nitrate, respectively, grouped into several time periods: the December 2004 data from the validation experiments and three 6-months periods during the operation of the sensor. For nitrite (Figure 10) the highest concentrations occurred at the beginning of the validation experiments. A higher accuracy at the higher nitrite concentrations that occurred frequently during the validation

period can be observed. This indicates that the sensor is an excellent instrument to monitor nitrite effluent concentrations with high accuracy.

For nitrate (Figure 11) the validation experiments took place at higher concentrations between 5 and 10 mg NO_3-N/l. Over time the validation data also showed good agreement between lab and sensor measurements at lower concentrations.

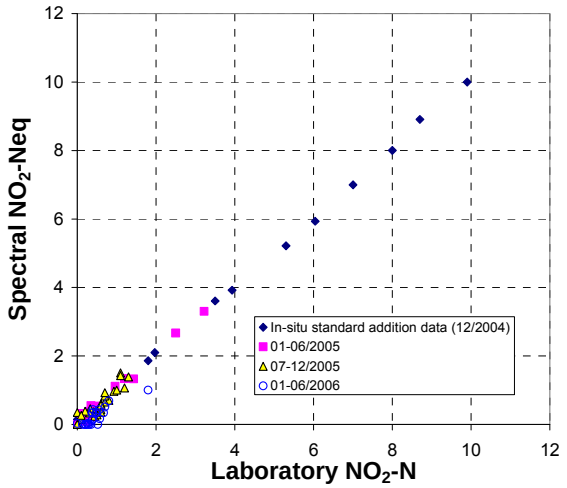


Figure 10: Validation data for nitrite grouped into different time periods.

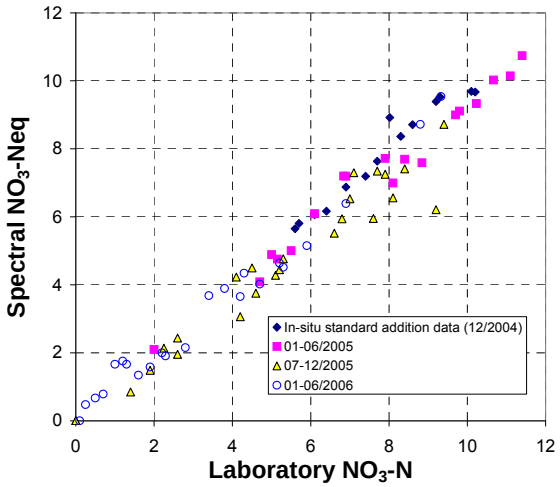


Figure 11: Validation data for nitrate grouped into different time periods.

3.3.2. Long-term evaluation

The sensor was placed in the effluent of the EAWAG pilot plant in December 2004. Reference samples for analyses in the laboratory have been taken on a regular basis, in total about 90 reference samples. The measurements lasted for a period of 1.5 years until the end of June 2006. Maintenance was limited to manual cleaning once a month.

Figure 12 shows sensor and lab data for the 3-months period April to June 2005. One can see that the EAWAG pilot plant was operated with special experimental objectives as particularly high nitrite concentrations could be observed. In general, the lab data validated the sensor data, especially for nitrite. Lab nitrate measurements too showed a good agreement with the sensor data.

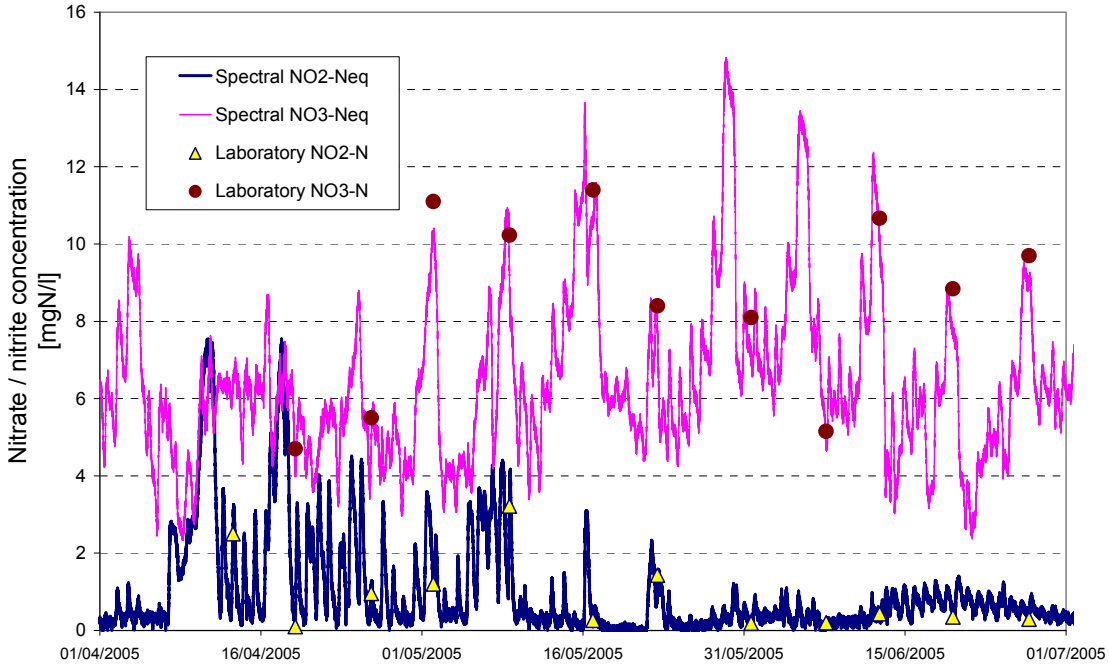


Figure 12: Comparison of sensor and lab data from April to May 2005.

Figure 13 and Figure 14 show the validation for nitrite and nitrate based on measurements in the effluent of the EAWAG pilot plant, respectively. The 95% prediction interval for nitrite was ±0.32 mg NO_2-N/l at a mean lab value of 1.15 mg NO_2-N/l. This is an extremely good result especially given the low maintenance of this in-situ sensor. Note that for concentrations above 1 mg N/l the prediction interval is even smaller (±0.23 mg NO_2-N/l at a mean lab value of 4 mg N/l, see Figure 15). The clouds of data in the lower concentration range, indicating lower data quality, are also related to the higher uncertainty of the lab measurements in this range.

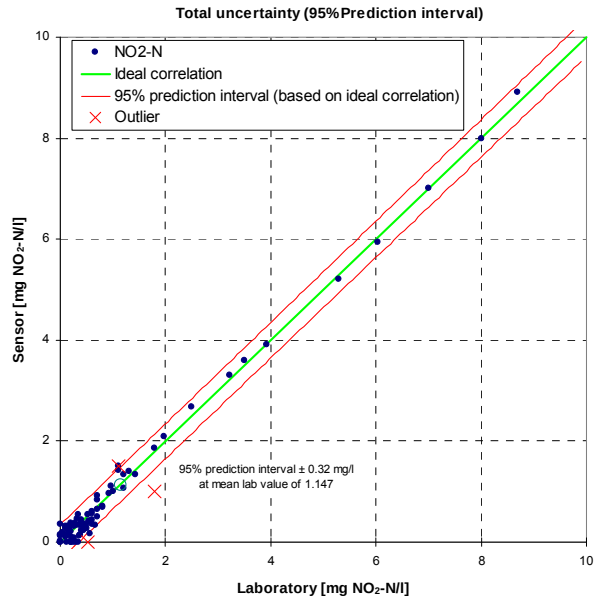


Figure 13: Validation data for nitrite based on WWTP effluent samples with a 95% prediction interval (width = ±0.32 mg/l at a mean lab value of 1.15 mg NO_2-N/l)

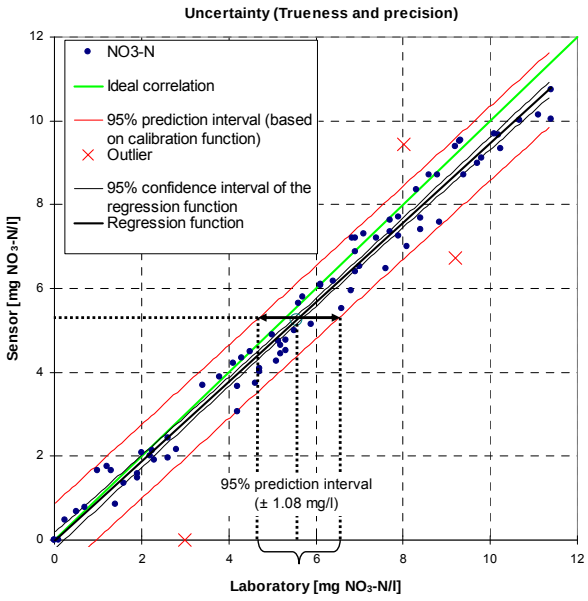


Figure 14: Validation data for nitrate based on WWTP effluent samples with a 95% prediction interval (width = ±1.08 mg/l at a mean lab value of 5.55 mg NO_3-N/l)

For nitrate the 95% prediction interval was ±1.08 mg NO_3-N/l at a mean lab value of 5.55 mg NO_3-N/l. The broader prediction interval compared to the nitrite measurements cannot be sufficiently explained. The same device and the same wavelength ranges are used for nitrite and nitrate and therefore problems with particles or other clogging effects can be excluded. Other calibrations based on different wavelengths did not provide better results. Since also the TresCon on-line UV analyser shows comparable prediction intervals of ±1.47 mg/l (see below, Figure 18) two hypotheses can be drawn: i) the low precision is caused by the measuring principle and by unknown interfering compounds in the water matrix or ii) the lab measurements are not accurate enough for the water matrix under evaluation. The lab equipment was carefully tested by carrying out standard additions on pure water as well as on different wastewater samples, but a changing matrix could still have caused uncertainties of the measurements.

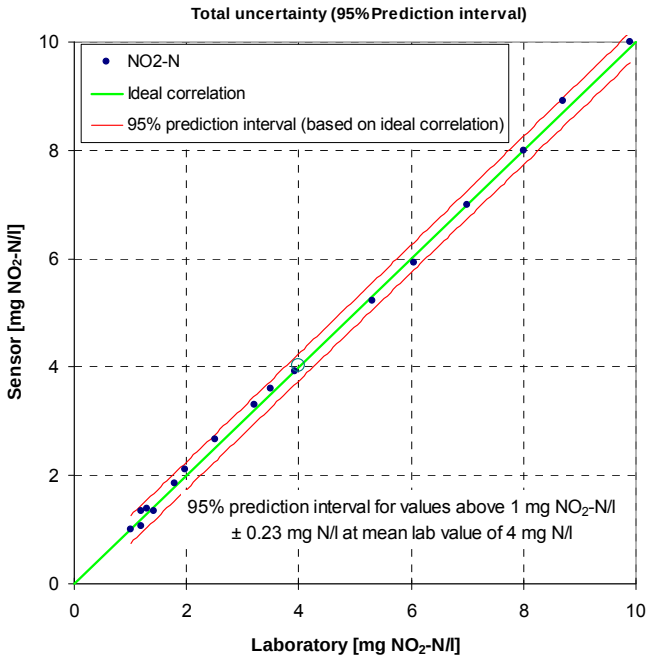


Figure 15: 95% prediction interval for nitrite for values above 1 mg NO_2-N/l (width = ±0.23 mg/l at a mean lab value of 4 mg NO_2-N/l)

Figure 16 shows the residuals (sensor minus lab value) for nitrite and nitrate over time. Starting in March 2006 a drift is visible for nitrite as well as nitrate. An evaluation of the sensor manufacturer revealed problems with the light source and therefore the probe had to be sent back to the supplier.

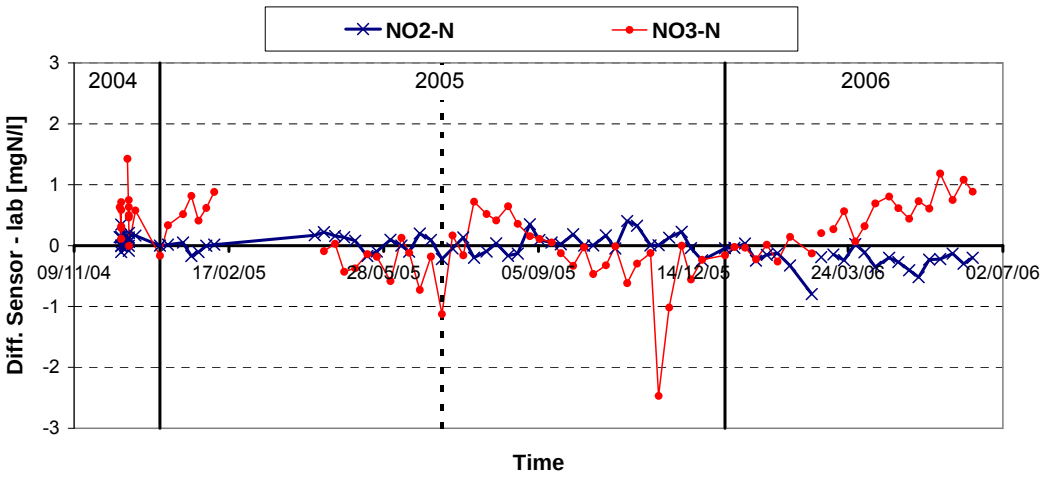


Figure 16: Residuals (sensor minus lab value) for nitrite and nitrate over time.

3.3.3. Comparison with on-line analyser

In winter 2004/2005 the spectral in-situ sensor was also tested against an on-line system using wet-chemistry (NO_2) and UV (NO_x), respectively, both after a filtration unit. Figure 17 and Figure 18 show the results from comparative measurements (taking the response time of the on-

line system into account). Both nitrite (95% prediction interval ±0.51 mg N/l at mean lab value of 0.45 mg N/l) and nitrate (±1.47 mg N/l at 4.3 mg N/l) show worse results than the results from the spectral sensor. The reason for the bad results of the wet-chemistry analyser could be related to the filtration unit, which causes a delay of approx. 10 min and, in addition, mixing of the sample in the fast loop tubes and other volumes of the sampling system. The result is that peaks are damped out by the system (see Figure 12).

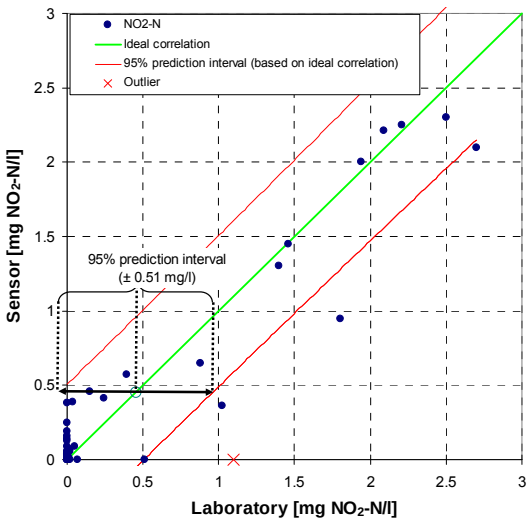


Figure 17: Wet-chemistry on-line analyser: Validation data for nitrite based on WWTP effluent samples with a 95% prediction interval (width = ±0.51 mg/l at a mean lab value of 0.45 mg NO_2-N/l)

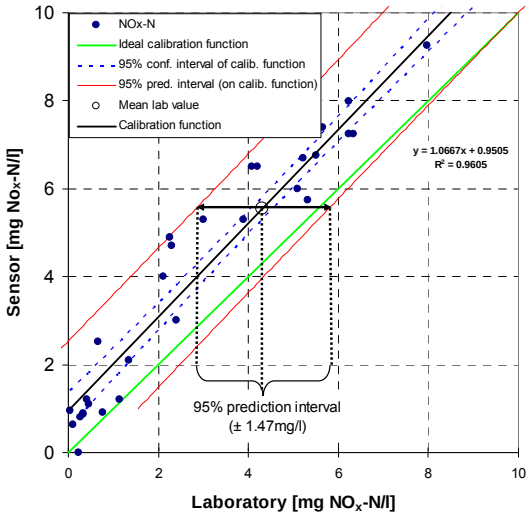


Figure 18: UV on-line analyser: Validation data for nitrate + nitrite (NOx-N) based on WWTP effluent samples with a 95% prediction interval (width = ±1.47 mg/l at a mean lab value of 4.3 mg NO_3-N/l)

The on-line UV analyser for NO_x has a limited number of wavelengths and therefore fewer options for individual calibration in comparison to the in-situ spectral probe. The off-set of about 1 mg/l is clearly a calibration problem. Since a first calibration with five standards (based on pure water) during start-up showed excellent results, the source of the problem is presumably related to the wastewater matrix with a strong industrial influence.

After three months of operation, it was decided to remove the on-line analyser and only rely on the optical probe due to its better performance and lower maintenance requirements.

3.4. CONCLUSIONS

A spectral in-situ sensor measures 256 wavelengths per measurement but only few wavelengths are used to correlate the spectral information to the nitrite respective nitrate concentrations in the liquid to analyse.

The tested spectral in-situ sensor was calibrated based on data from another WWTP and additional lab experiments (standard addition on wastewater matrix from the same WWTP). PLS regression was used to develop and calibrate the underlying multivariate model.

It could be demonstrated that the sensor was able to accurately predict the nitrite and nitrate concentration in the effluent of the EAWAG pilot-scale plant with a precision of ±0.32 mg N/l (95 % prediction interval at mean lab value of 1.15 mg N/l) and ±1.08 mg N/l (at 5.55 mg N/l) for nitrite and nitrate, respectively. This proofs that the calibration developed for another WWTP is also applicable to other wastewater matrices.

Comparative measurements were carried out over a period of 1.5 years and the results showed constant accuracy except for the last three months where a drift occurred for nitrite as well as nitrate. The drift could be related to a problem with the light source, which was replaced by the manufacturer.

As a summary it can be stated that the long-term stability is excellent, especially since the sensor requires only a minimum of maintenance. A second on-line analyse with a filtration unit was tested but removed due to the better results of the in-situ sensor and the high demand of the on-line system in terms of consumption of chemicals and maintenance.

The sensor showed good results for nitrite in the low concentration range and even better results for higher concentrations (up to 10 mg N/l). This allows to use the sensor for alarm systems as well as for control concepts at WWTPs.

4. Control in WWTPs

4.1. Introduction

On-line control using sensors started in the 1970s when the sensors reached a level of precision suitable for control. The main focus at the beginning was on control of aeration (e.g. Olsson and Andrews, 1981) and maintaining the biomass in the system. From then, measuring technologies have become more reliable and economic, actuators offer more flexibility, computation power has increased significantly and more knowledge on the unit processes is available. The implementation of Instrumentation, Control and Automation (ICA) in wastewater treatment plants has been demonstrated to be useful to optimize process performance by adapting the process operation according to the environmental conditions and the process requirements. Therefore, different control strategies have been developed that aim at different operating objectives: 1) Keeping the plant running, 2) Improving effluent quality, 3) Reducing operation costs and 4) Decreasing carbon footprint.

The three elements that describe a control strategy are the ***manipulated variable,*** the ***measured variable*** (more information can be found in the sensors chapter) and the ***control algorithm***. In the following sections a description of manipulated variables is presented followed by the concepts behind the control algorithms. Finally, a review on the most used control strategies is presented. Information about measured variables can be found in the chapter entitled “best available technology for sensors”.

4.2. Manipulated variables

Before going into detail with controller description, an analysis of the manipulated variables is presented. In a wastewater treatment plant the actuators regulating the process are grouped in different types: the hydraulic-related variables (internal recirculation, external recirculation, sludge wastage, rejected water recycle, by-passing pumps...), aeration and chemical addition variables. Figure 19 presents a scheme of a typical wastewater treatment plant consisting of a primary sedimentation tank, aeration tank and secondary clarifier with its actuators and the main effects they have on the process when manipulating them. In Figure 19 it can be seen that there is a limited number of actuators and that a single actuator might have an effect on different processes and relevant variables (nitrification, denitrification, phosphorus removal efficiencies, sludge and hydraulic retention times and costs). This is the reason why one should study the control of a plant from a plant-wide perspective.

The range of manipulation of these variables is limited by physical, process or economic constraints. Some of the key manipulated variables and their limits are described below.

Manipulation of wastage flow rate: Wastage flow rate is used to control the Sludge Retention Time (SRT) of the system (directly related to the MLSS concentration in the reactors). The minimum and maximum boundaries for selecting the SRT are:

- Minimum boundary: The minimum SRT should be high enough to allow nitrifier organisms to grow in the system. In winter the solids concentration in the reactors can be increased to make sure that the nitrifiers are not washed out.

- Maximum boundary: The maximum MLSS concentration value is selected according to the maximum capacity of the secondary clarifier. One has to take into account that at a higher MLSS concentration a) the sludge production decreases (which means less costs associated to sludge treatment) and b) the oxygen demand increases due to higher endogenous respiration.

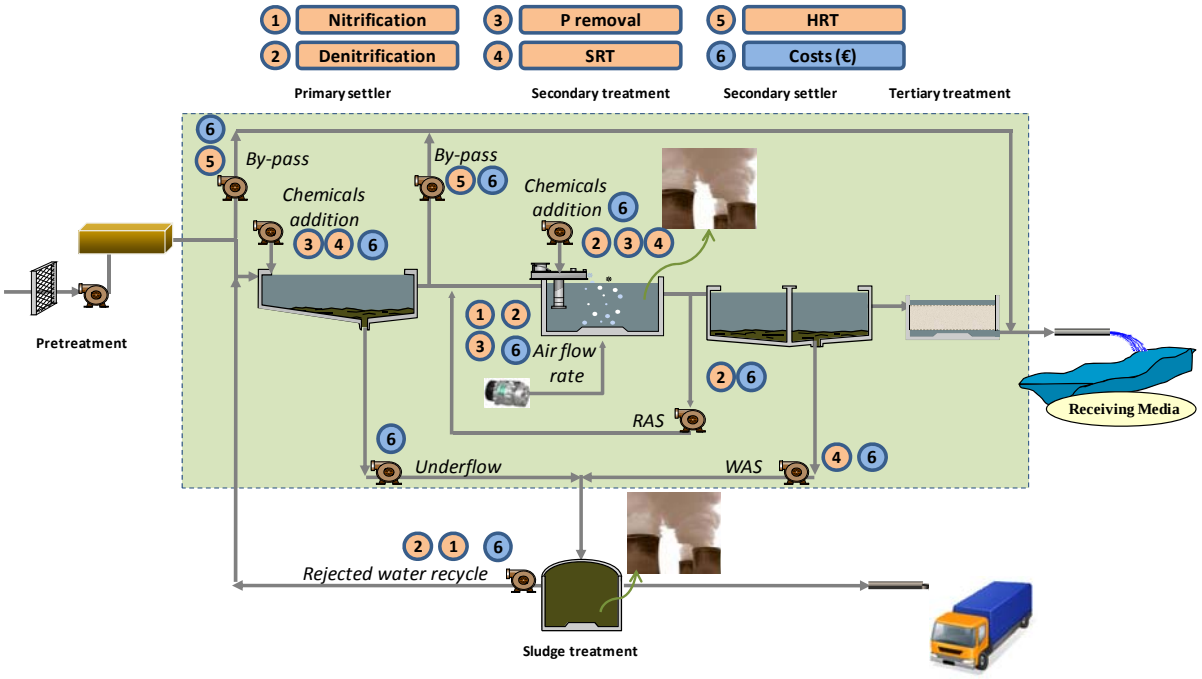


Figure 19. Schematic outline of control options and main effects (numbers in circles) in a wastewater treatment plant. In green the water line.

Manipulation of internal recirculation of nitrates: In predenitrification activated sludge systems the internal recirculation is used to bring the nitrates generated in the aerobic reactors (nitrification) to the anoxic tanks where the denitrification occurs. The minimum and maximum boundaries are determined by the following:

- Minimum boundary: A minimum flow rate is required to transfer the required nitrate to the anoxic reactors to achieve good nitrate concentrations to denitrify (normally values between 1 and 2 mg NO_3-N/L are pursued).
- Maximum boundary: This is determined by the maximum capacity of the pump. Moreover, one has to take into account that a too high internal recirculation rate might bring too much oxygen into the anoxic reactor that will reduce the denitrification rate.

It is important to state that the internal recirculation is not sensitive if organic matter limits the denitrification (Yuan *et al.*, 2002).

Manipulation of aeration: The airflow rate supplied to the reactors is the most studied manipulated variable. As presented in Figure 19 it influences all biological processes and it causes a significant part of the operation costs. The minimum boundary is selected according to the oxygen requirements of the process and the maximum boundary is related to the costs.

Chemicals addition: The most common chemical compounds added in the WWTPs are 1) the external carbon sources for systems with limiting biodegradable substrate with the aim of improving denitrification and biological phosphorus removal and 2) the iron/alum salts for phosphate precipitation. The dosing location is crucial to maximize the benefits of adding the chemical. In the case of external carbon sources the addition is proportional to the nitrate concentration at the end of the anoxic reactor and is conducted at the beginning of the anoxic zone. Iron/alum salts are added proportionally to the concentration of phosphorus in the reactors. For the salts three possibilities are available: pre/post and simultaneous precipitation.

For chemicals addition the maximum boundary for chemicals addition is imposed by the costs and the increase in the solids concentration of the reactor that could overload the secondary settler. The minimum boundary depends on the desired effluent quality.

Manipulation of digester reject water recirculation: Reject water coming from the digesters (that normally passes through a buffer tank) and recycled at the beginning of the WWTP can represent up to 20-25% of the total nitrogen incoming load (Metcalf & Eddy, 2003). Therefore, a common practice is to manipulate the flow coming out from the buffer tank to make sure that this water with high nitrogen concentration does not overload the system.

4.3. Control algorithms

The control algorithms translate the information of the sensors (inputs) into actions for the actuators (outputs). In order to describe a control loop the following terms are defined:

- Disturbances: variables that change and affect the process but over which we have no direct control.
- Manipulated variables: those variables affected by the actuators and which have an effect on the process.
- Measurements: signals that we get from the sensors.

Depending on the type of physical and data flows two different control loops are considered, feedforward and feedback (see Figure 20). In feedforward control loops an action can be initiated before the disturbance causes adverse effects on the system. The difficulty with feedforward control is that the effect of the disturbances on the system must be accurately predicted with a model. In a feedback control loop, the measured variable (S) is compared to the reference value that is selected and the error is used to calculate the actuator (A) action that will drive this error to zero.

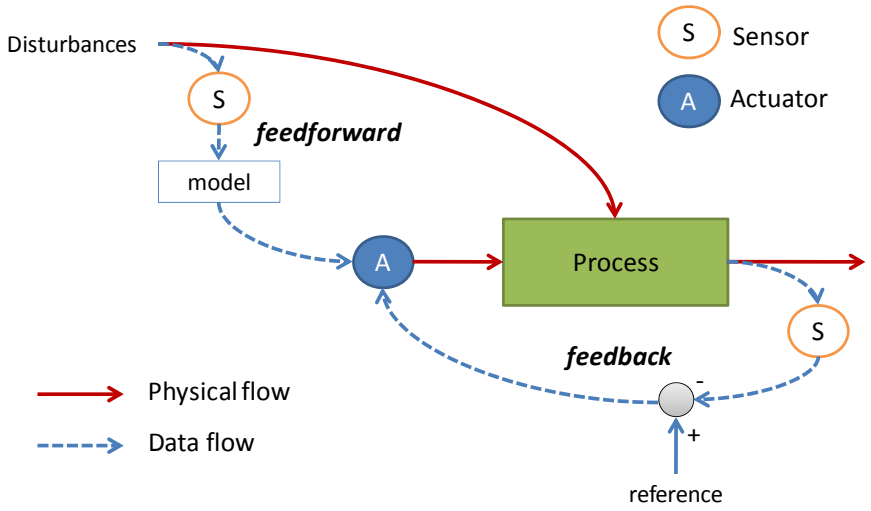


Figure 20. Feedback and feedforward control

As an example the control of aeration is presented. In this case, aeration is controlled with the objective to reach good effluent quality (especially regarding nitrogen concentrations) and save energy. Aeration is used to regulate the nitrification and denitrification processes. For this case, different possibilities are shown in Figure 21. The first possibility is to maintain the dissolved

oxygen concentration (O_2) at a setpoint value (defined from experience or optimization studies) using a feedback PI controller. The second possibility is to add a PI controller to mantain the ammonium concentration at a setpoint value. Therefore, ammonium is measured and a master controller$_{teht}$ determines the O_2 setpoint of the slave PI O_2 control.

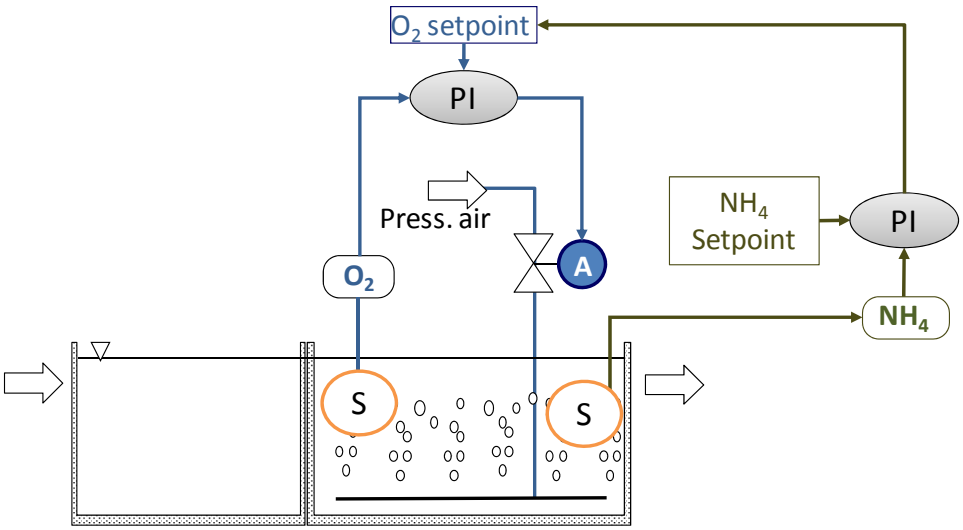


Figure 21. Aeration control (blue: DO control; green: NH_4^+ feedback control; red: NH_4^+ feedforward control)

Many control algorithms have been proposed in the literature ranging from simple classical control (P, PI, PID), to model-based control (such as Model Predictive Control), adaptive control,, rule-based control, neural network and fuzzy control.

4.4. Description of Control Strategies

Many control strategies have been developed for WWTP in the last decades. These control strategies have been tested directly at full-scale and in other cases a model has been used prior to the implementation. In this chapter some indications are given about references regarding control strategies and the most relevant ones are described. As mentioned before the description of a controller is based on the objective, the measured, the manipulated variable and the type of control algorithm used.

Control strategies are found in different sources:

- **Publications**: A summary of control strategies can be found in Olsson *et al.* (2005), Copp (2002), etc. There are also a large number of scientific papers describing and evaluating control strategies. However, many control strategies applied at full-scale remain unpublished and it is difficult to evaluate the degree of implementation and success.

- **Conferences:** The Instrumentation, Control and Automation conferences organized by IWA are the platforms where new control strategies and case studies are presented. The conferences organized by WEF (e.g. WEFTEC) are other platforms where new developments are discussed.

- **Project deliverables**: IWA Task Group on Benchmarking control strategies, Smart control of wastewater systems (SMAC, Thornberg and Gernaey, 2004), Getting Systems

Engineering into Regional Wastewater Treatment Strategies (WWTSYSENG), TELEMAC...

The strategies that have been developed and are normally used in continuous activated sludge systems can be grouped according to the objectives they are designed for. In Table 8, a summary of control strategies is presented, organized along the following objectives:

- **Nitrification**: This process is optimized by controlling aeration, SRT and ammonia loading. Originally, nitrification was controlled by establishing a DO setpoint (normally 2 mg/L). Latter, a cascade controller was added defining a dynamic DO setpoint depending on the effluent ammonia concentration.

- **Denitrification:** This process can be optimized by adapting the internal recycle rate and/or external carbon addition. Normally a nitrate setpoint between 1 and 2 mg/L is set at the end of the anoxic zone.

- **Phosphorus removal:** The practice is to add chemicals, either external carbon source (Bio-P removal) or salts of alum or iron (Chemical P-removal).

- **SRT:** All biological processes can be improved by controlling SRT. This is essential to avoid organisms wash-out.

- **HRT**: Controlling the HRT is a protection against wash-outs during rain events.

Table 8 does not pretend to be a full review of control strategies but a compendium of the strategies that have been traditionally used in the wastewater treatment field with references where detailed information can be found.

Table 8. Summary of WWTP control strategies

	Measured variable/s	Manipulated variable	Control algorithm	Reference
Nitrification	DO (Aerobic)	Flow rate (aeration)	On/Off	Olsson *et al.*, 1981
	DO (Aerobic)	Flow rate (aeration)	Rule based (fuzzy)	Ferrer *et al.*, 1998
	DO (Aerobic)	Flow rate (aeration)	P, PI, PID	Copp 2002
	NH_4^+ (Aerobic)	DO setpoint	Cascaded On/Off	Olsson *et al.*, 2005
	NH_4^+ (Aerobic)	DO setpoint	Cascaded rule based (fuzzy)	Kalker *et al.*, 1999
	NH_4^+ (Aerobic)	DO setpoint	Cascaded feedback	Vrecko *et al.*, 2006
	NH_4^+ (Influent)	DO setpoint	Cascaded feed forward	Vrecko *et al.* 2003
	OUR (Aerobic)	DO setpoint	Cascaded On/Off	Sumacz-Gorska *et al.* 1998; Vanrolleghem and Gillot 2002
Denitrification	NO_3^- (Anoxic)	Flow carbon dosing	Feed forward + PI feedback	Yuan *et al.*, 1997; Samuelsson and Carlsson 2001
	NO_3^- (Anoxic)	Flow internal recycle	PI	Londong 1992;
	NO_3^- (Aerobic), NO_3^- (Anoxic)	Carbon dosing and internal recycle	PID	Yuan and Keller, 2003
	OUR (Anoxic)	DO setpoint	Cascaded On/Off	Klapwijk *et al.*, 1998;
P removal	Flow (Influent)	Flow precipitant dosing	Flow proportional feed forward	Olsson *et al.*, 2005
	Flow /PO_4^{3-} (Influent)	Flow precipitant dosing	Load proportional feed forward	Olsson *et al.*, 2005
	PO_4^{3-} (Effluent)	Flow precipitant dosing	PI feedback	Ingildsen, 2002
	PO_4^{3-}, NH_4^+, NO_3^- (Aerobic)	Flow precipitant dosing	On-Off	Devisscher et al, 2002
Sludge	Temperature/TSS	Flow wastage	Cascaded feed forward	Olsson *et al.*, 2005
	Flow daily average / TSS	Flow wastage	Cascaded feed forward	Olsson *et al.*, 2005
	Moving average of NH_4^+/TSS	Flow wastage	Cascaded feed forward	Olsson *et al.*, 2005
	Flow (Influent)	Flow external recycle	P	Olsson *et al.*, 2005
	SBH (Settler)	Flow external recycle	Cascaded feed forward	Olsson *et al.*, 2005

4.5. Evaluation of control strategies

Simulations provide a cost-effective way of testing and evaluating control strategies prior to implementing them. In order to be able to compare different control strategies a platform for simulation purposes was developed by the IWA Task Group on Respirometry in Control of Activated Sludge Processes (Spanjers et al, 1998) together with the different Working Groups of COST Actions 682 and 624 as summarized in Copp (2002). It is called Benchmark Simulation Model nº1 (BSM1). It consists of a comprehensive simulation model of the plant, plant layout, influent files, controllers, sensors, procedure for performing the simulations, and includes several evaluation criteria for plant performance. Currently, the IWA Task Group on Benchmarking of Control Strategies for WWTPs is developing an upgrade of the BSM1. First of all BSM1_LT (Rosen *et al.*, 2004) attempts to extend the applicability for evaluation of process monitoring methods by providing a long term influent model/file (Gernaey *et al.*, 2006a), models for faults on sensors and actuators (Rosen *et al.*, 2008) and modifying biokinetic model to include inhibition and toxicity. The BSM2 has been developed within the scope of plant-wide control by also including the primary clarifier, thickener unit, anaerobic digester and a dewatering unit (Jeppsson *et al.*, 2006). Moreover, the IWA/COST simulation benchmark that was extended to include expert reasoning for system performance evaluation is presented in Comas *et al.* (2005), and enables settling problems of biological origin to be detected. More than 100 scientific papers have been presented, according to Jeppsson and Vanrolleghem (2009), using the benchmark or part of it.

Beyond the virtual reality benchmark applications, real cases using model approaches to evaluate possible control strategies prior to implementing them can be found in literature. For instance the control of the DO in the aerobic reactor of a plant can be tested (Demey *et al.*, 2001). Regarding modeling and control strategy testing for nitrogen and phosphorus removal in a WWTP, Ingildsen (2002) extends the control scheme for N removal used by STAR® (Oennerth *et al.*, 1996) with biological and chemical phosphorus removal. In Ayesa *et al.* (2006) a model is used for testing supervisory control strategies before the implementation on a WWTP. Another application is related to calculating the costs of controlling the system. Devisscher *et al.* (2005) presents a methodology for estimating the costs and benefits of advanced control of WWTPs.

The work from IWA Task Group on Respirometry in Control of Activated Sludge Processes is the first attempt to provide standardized indices to evaluate process and control performances. The scientific community has realised that the control implementation leads to trade-off situations that are difficult to solve (e.g. improving effluent quality is achieved thanks to increasing operating costs). Different methods are now being applied (e.g. multicriteria analysis) in order to analyse the results from a multicriteria point of view (Flores et al, 2007). Also, uncertainty studies have been conducted in order to evaluate how different controllers deal with uncertainties in the influent characteristics (Flores-Alsina *et al.*, 2008) and the robustness of the systems under control (Vanrolleghem and Gillot, 2002). The final aim of the research work is to provide tools to the plant manager so that a proper decision can be taken on the operation of the plant for different environmental conditions and also to account for uncertainties. The objectives of the plant operation might change over time and control is a good tool to provide the required flexibility.

From the references provided in this section it is deduced that the implementation of control improves effluent quality and reduces costs. However, most of these references have only dealt with nitrogen removal. The work on control presented in this report focuses on evaluating different control options for nutrient removal using models (Chapter 5 and 6). An innovative aspect for control strategies that is also addressed is the use of fault-detection to assess the quality of the data coming from the sensors which are used for control (Chapter 7).

5. Evaluation of control strategies using a benchmark platform

5.1. Neptune benchmark wastewater treatment plant

The motivation of this section brings us to develop the “Neptune Simulation Benchmark” to objectively compare different control strategies for nutrient removal. Each simulation will be evaluated under the same conditions to ensure unbiased comparisons. In the section, it is described how the Neptune benchmark is developed, as simulation platform including plant layout, simulation models and model parameters, a detailed description of the disturbances to be applied during testing, a simulation procedure and finally a set of evaluation criteria for testing the relative effectiveness of the simulated control strategies. The different elements of the “*Neptune Simulation Benchmark*” are depicted in Figure 22.

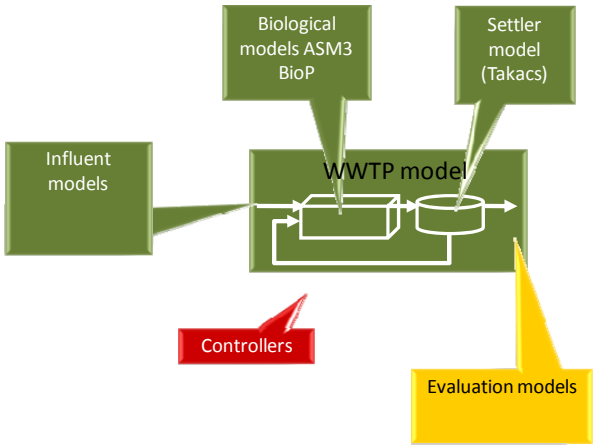


Figure 22. Elements of the “Neptune simulation Benchmark” Plant layout

The “*Neptune simulation benchmark*” plant layout is comprised of seven reactors in series with a 10 layer secondary settler. Figure 23 shows a schematic representation of the layout.

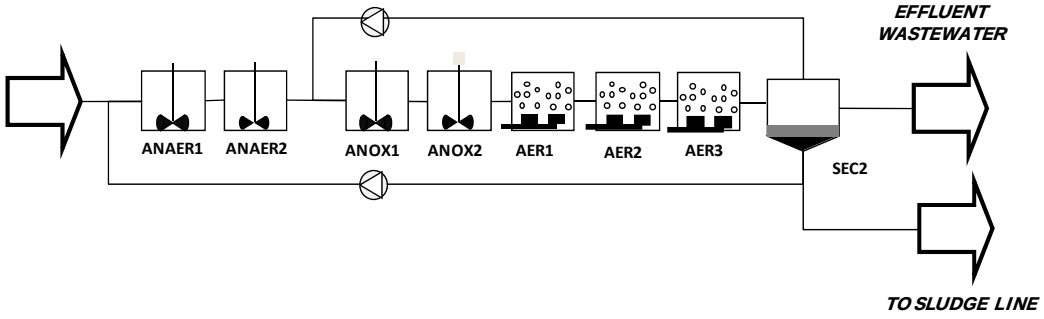


Figure 23. Schematic representation of the “Neptune simulation Benchmark” configuration showing tanks 1 (ANAER1),2 (ANAER2),3 (ANOX1) & 4 (ANOX2)mixed and un-aerated and tanks 5 (AER1),6 (AER2) and 7 (AER3) aerated.

The selected configuration is the A2O and represents the basic process configuration for organic carbon, nitrogen and phosphorus removal. The plant consists of an anaerobic zone followed by an anoxic and aerobic zone. The aerobic and the anoxic zone by means of an internal recycle. In the anaerobic section (without oxygen and nitrates) the anaerobic P release is promoted and

provides the phosphorus accumulating organisms (PAO) with competitive advantage over other bacteria. The, PAO organisms grow using intracellular storage products as a substrate during the aerobic and anoxic phase, with oxygen or nitrate as electron acceptors and consuming nitrogen and phosphorus as nutrients. Additionally, in the aerobic zone the remaining organic matter and the ammonia (ammonia is the form of nitrogen most commonly found in the influent wastewater) are oxidized to carbon dioxide and nitrate. The nitrate transported by the internal recirculation is reduced to nitrogen gas in the anoxic section. This reduction requires an electron donor, which is supplied in the form of influent wastewater or an external carbon source.

The design was carried out following the Metcalf & Eddy guidelines (Metcalf & Eddy 2003). Firstly, it is designed the aerobic zone for organic carbon removal and nitrification. Secondly, the anoxic tank and the internal recycle system. Finally, the anaerobic zone and the secondary settler are dimensioned. The initial assumptions used to perfom the design are summarized in Table 9. The aerobic zone volume is sized on the basis of the net specific growth rate of nitrifying organisms, the desired mixed liquor suspended solids concentration and the total mass of solids that has to be removed to maintain the required sludge residence time. Next, the necessary internal recycle flow-rate is calculated through a mass balance which includes the nitrate produced in the aerobic zone, the nitrate in the return activated sludge and the desired nitrate in the effluent. The anoxic volume is designed by comparing the nitrate produced in the aerobic zone and the nitrate which potentially can be removed for a given hydraulic retention time. The anaerobic volume is approximated assuming a hydraulic residence time of 2 hours. Finally, the secondary settler area is sized assuming from the solids loading rate.

Table 9. Initial design assumptions for the Neptune benchmark layout

Initial design conditions	Value	Units
Effluent N ammonium	1	g m^3
Effluent N nitrate	8	g m^3
Reactor dissolved oxygen	1	g m^3
Safety factor	1.5	-
Design reactor MLSS	3500	g m^3

The physical characteristics of the biological reactor and the settler are summarized in Table 10. The selected operational variables are listed in Table 11.

Table 10. Physical characteristics of the "Neptune Simulation Benchmark"

	Physical Configuration	Units
Volume - ANAER1 + ANAER2	1000 + 1000	m^3
Volume - ANOX1 + ANOX2	1500 + 1500	m^3
Volume - AER1 + AER2 + AER3	3000 + 3000 + 3000	m^3
Depth – SEC2	4	m
Area – SEC2	1500	m^2
Volume – SEC2	6000	m^3

Table 11. Default operational characteristics of the "Neptune Simulation Benchmark"

	Operational conditions	Units
External recirculation flow-rate (Q_r)	23450	m^3 $days^{-1}$
Internal recycle flow-rate (Q_{intr})	105525	m^3 $days^{-1}$
Wastage flow-rate (Q_w)	400	m^3 $days^{-1}$
K_La 1 & 2 – ANAER1 & ANAER2	0,0	$days^{-1}$
K_La 3 & 4 – ANOX1 & ANOX2	0,0	day^{-1}
K_La 4,5 & 6 – AER1, AER2 & AER3	120, 120 / 80	day^{-1}

5.1.1. Influent profile generation

Plant performance evaluation is based on one year´s simulated influent data generated according to the principles outlined in Gernaey *et al.* (2006a). The model has been adapted to include phosphorus using the ASM3 Bio P model (Rieger *et al.*, 2001).

The general structure of the proposed approach to generate the influent flow rate is illustrated in Figure 24. The household model contributes to the influent flow rate dynamics with diurnal influent variations, a weekend effect and a seasonal effect. The industry contribution is generated similarly to the households. The main difference consists in a flow peak on Friday afternoon assumed to the cleaning of the installations at the end of the working week. Further, there is an additional contribution of the infiltration and rain. A fraction of (aH) the flow rate resulting from the rainfall is assumed to originate from run-off from impervious surfaces, and is transported directly from the sewers. Rainfall on pervious surfaces, represented by a fraction (1-aH) of the flow rate resulting from rainfall, is assumed to influence the groundwater level, and thus also the contribution of the infiltration to the influent flow rate. A seasonal effect corrector is combined with the rainfall assumed to fall on pervious surfaces and the sum of both is passed through a soil model. Finally, the infiltration flow rate and the output of the soil model is combined with the other flow rate contributions, and the resulting flow rate is finally passed through a sewer model.

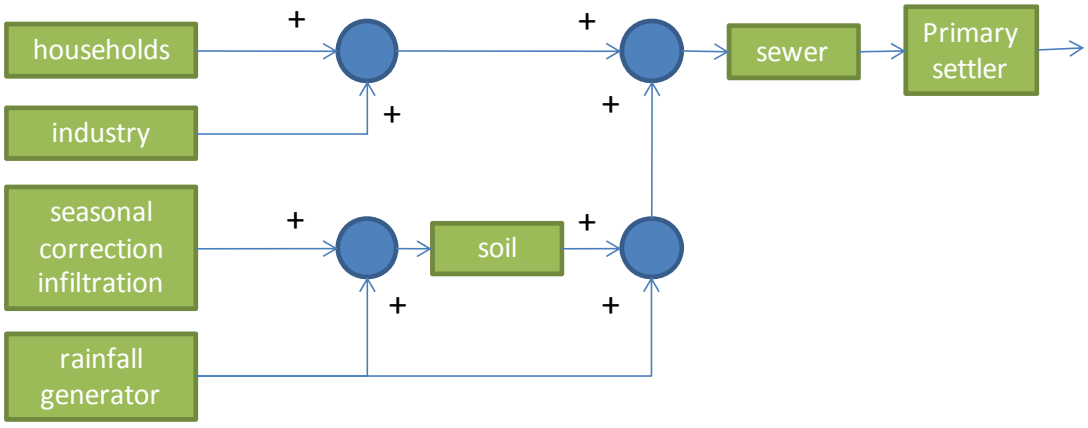


Figure 24. Schematic representation of the influent flow rate model

Figure 25a,b,c and d shows a 10 day snapshot of the yearly generated dynamic profile highlighting the effect of the different elements involved in the influent flow rate generator. In Figure 25a can be seen the contribution of both rainfall and infiltration on the total flow rate *i.e* households and industry. Figure 25b shows the expected flush of the sewer particulate material after a rain/storm event. The re-suspension of these particles is reflected in the data through a significant increase of the TSS. Finally in Figure 25c and b the smoothing effect of the sewer system on both the influent wastewater flow rate and the pollution peaks can be observed.

The generated influent file contains 609 days with samples taken every 15 minutes. In the resulting influent profile, typical phenomena observed in a WWTP such as: 1) diurnal variation, 2) a lower average flow rate and pollutant concentrations during weekends compared to week days, in an attempt to simulate a WWTP that receives mixed municipal – industrial wastewater, 3) seasonal phenomena reflecting the typical effects of sewer systems and urban drainage, i.e. increased infiltration in winter due to higher infiltration levels, 4) holiday periods during which a lower average wastewater flow rate is maintained for an overall period of several weeks. Figure 25 shows some of the above-mentioned phenomena broken down in terms of both flow rate and organic load in different time scales of one day (Figure 16 a and b), 20 days (Figure 26c and d) and one year (Figure16e and f). With respect to the one-year temporal series, an exponential 3-day filter has been used to clarify its evolution and avoid noisy representations. The average wastewater flow rate to be treated is of 20000 m^3day^{-1} (see the complete profile in Figure 25e), with an organic and nitrogen load of 12200 kg COD day^{-1} and 1140 kg N day^{-1} respectively. The influent model parameters have been modified to have a low biodegreabale influent with a high content of nitrogen and phosphorus. The averaged values have been used for the design of the wastewater treatment plant.

5.1.2. Process Models

Simulations are performed using the WEST modelling environment (MOSTforWATER, 2007). A modified version of the EAWAG – Activated Sludge Model No 3 Bio P including chemical phosphorus removal was chosen as (bio) chemical model (Rieger *et al.*, 2001). The modified EAWAG-ASM3 bioP model has 19 state variables and describes (bio) chemical phosphorus removal with simultaneous nitrification and denitrification in activated sludge systems by means of a set of non linear differential equations. The double exponential velocity function of Takacs *et al.*, (1991), based on the solids flux concept was selected as a fair representation of the settling process, using a 10 layer discretization. The kinetic parameters are adjusted according to the influent temperature using the Arrhenius equation. The default parameters for the activated sludge and the settling model can be found in Rieger *et al.*, (2001) and Copp. (2002), except for phosphorus precipitation kinetics, which were adjusted (Gernaey *et al.*, 2002). It is important to highlight that the settling characteristics are assumed to be constant along the case study although the authors are aware that the floc characteristics may change in system with chemical precipitation.

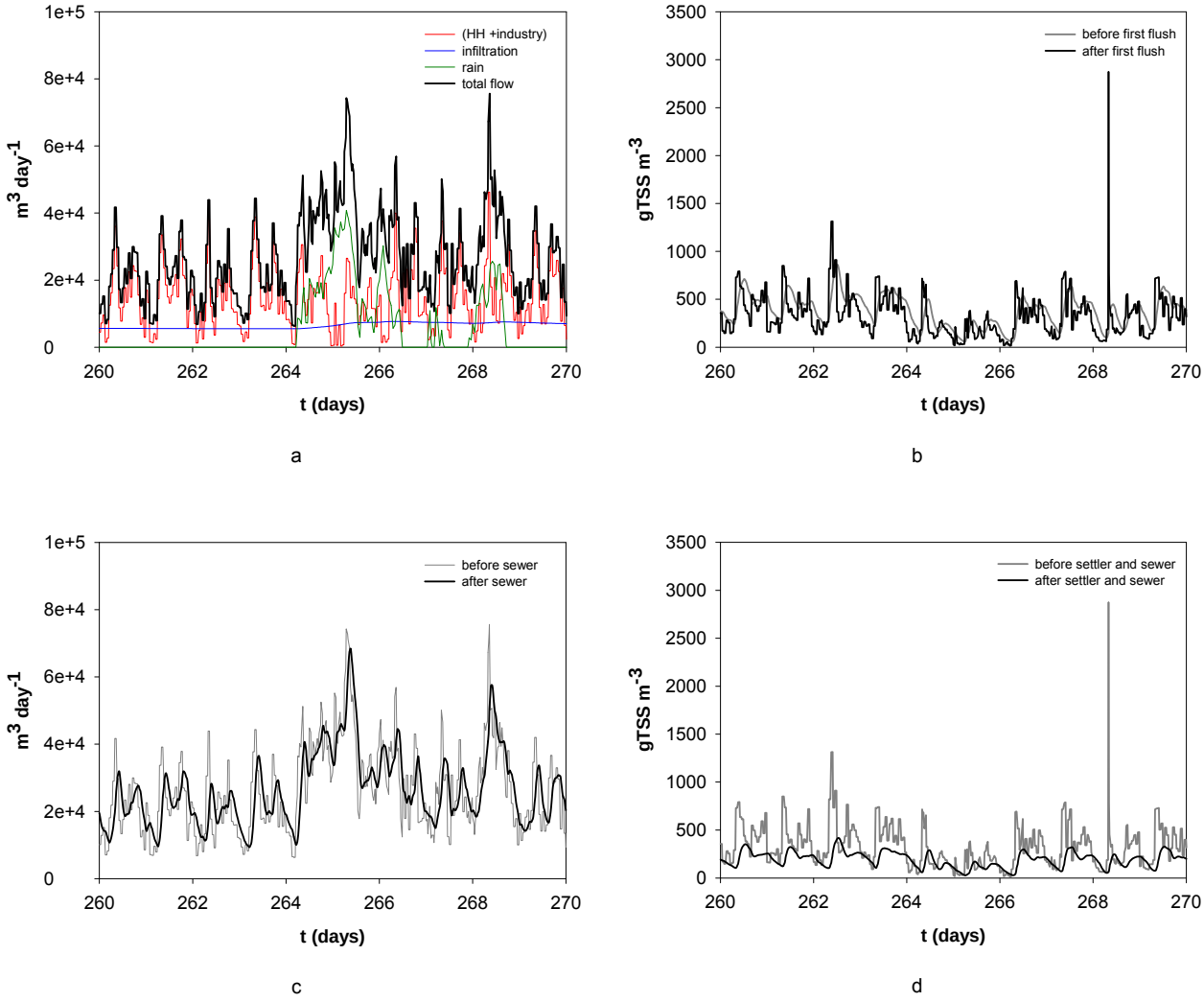


Figure 25. Snapshof of the generated yearly showing the effects of the blocks comprising the Neptune influent generator a) household +industry, rain and infiltration, ii) first flus with sudden increase of TSS, iii) smoothing effect of the sewer system and iv)TSS removal efficiency of the primary settler

5.1.3. Evaluation criteria

A set of evaluation criteria X = X_1,....,X_{24} is used to compare the different control strategies implemented in the "*Neptune Simulation Benchmark*". The effluent quality index (EQI) (Copp 2002) and the risk of suffering microbiology-related TSS separation problems (Comas *et al.*, 2008) are calculated similarly to the BSM1, but adapting them to the (bio)chemical P removal. Also, the original operational cost index (OCI), suggested by Vanrollghem and Gillot (2002) have been modified in order to include chemical addition cost. A description of other criteria such as time plant in violation (T_{viol}) can be found in Copp (2002). See more details in Table 12.

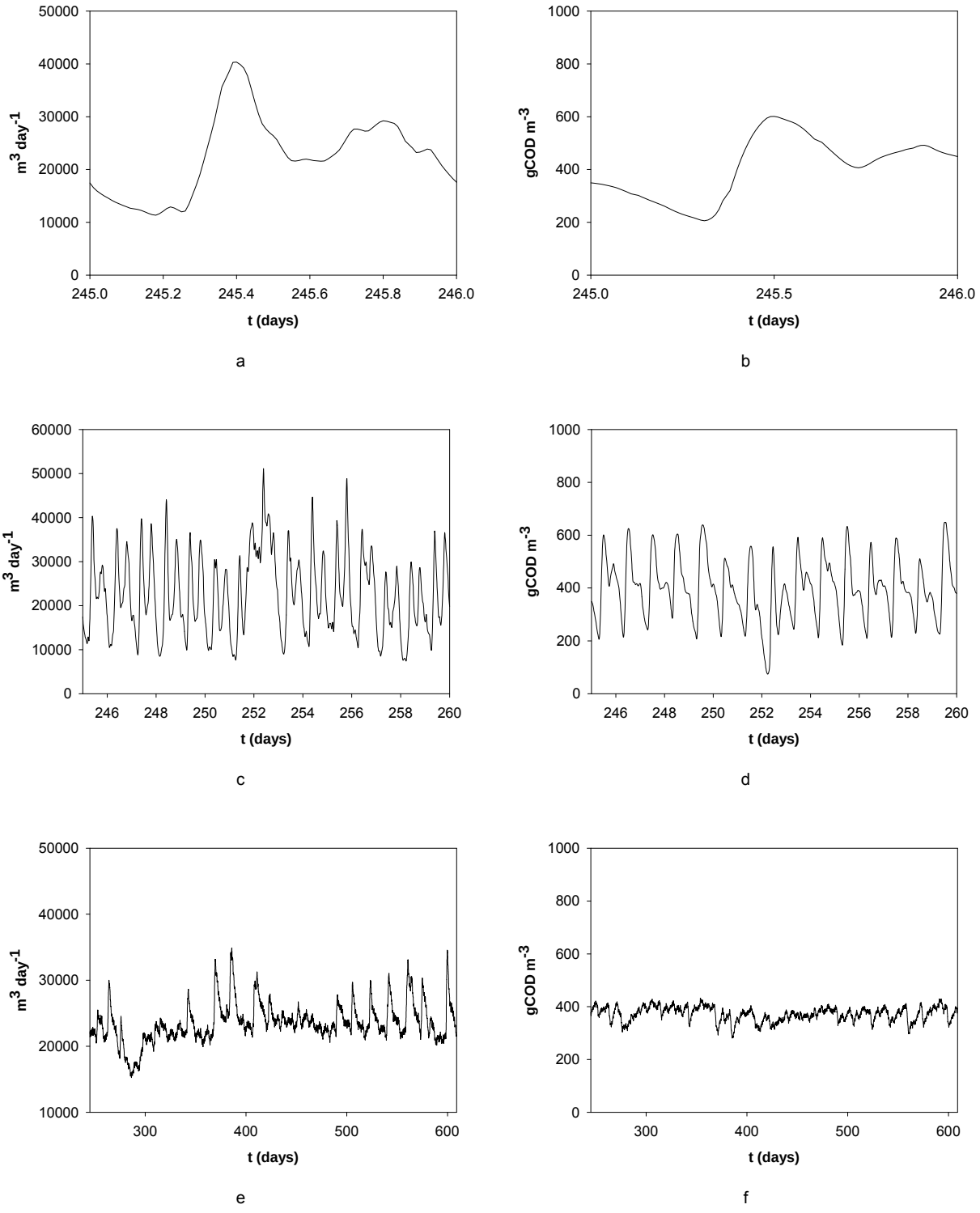


Figure 26. Influent wastewater flow (a,c,e) and components (b,d,f) for one day (a,b), for one month (c,d) and for one year (smoothed data) (e,f)

Table 12 List of evaluation criteria, units and references used in this case study

X	Evaluation criteria	units	reference
X_1	Total Kjeldahl Nitrogen (TKN)	g N·m^{-3}	Copp (2002)
X_2	Total Nitrogen (TN)	g N·m^{-3}	
X_3	Total Phosphate (SPO4)	g P·m^{-3}	
X_4	Total Phosphorus concentration (TP)	g P·m^{-3}	
X_5	Chemical Oxygen Demand (COD)	g COD·m^{-3}	
X_6	Biochemical Oxygen Demand (BOD_5)	g COD·m^{-3}	
X_7	Total Suspended Solids (TSS)	g TSS·m^{-3}	
X_8	Effluent Quality Index (EQI)	kg poll·day^{-1}	
X_9	Sludge Production (P_{sludge})	kg TSS·day^{-1}	Vrecko *et al.*, (2006
X_{10}	Aeration Energy (AE)	kWh·day^{-1}	
X_{11}	Pumping Energy (PE)	kWh·day^{-1}	
X_{12}	Metal Salt Addition (MS)	Kg metal·day^{-}	
X_{13}	External Carbon Source (CS)	kg COD·day^{-}	
X_{14}	Mixing Energy (ME)	kWh·day^{-1}	
X_{15}	OCI	-	
X_{16}	Nviolation (L = 18 g m^{-3})	**%**	Copp (2002)
X_{17}	CODviolation (L = 100 g m^{-3})	**%**	
X_{18}	SNHviolation L = 4 g m^{-3})	%	
X_{19}	TSSviolation (L = 30 g m^{-3})	**%**	
X_{20}	BOD5violation (L = 20 g m^{-3})	**%**	
X_{21}	Pviolation (L = 2 g m^{-3})	%	
X_{22}	N deficiency bulking	%	Comas *et al.*, (2008
X_{23}	DO deficiency bulking	%	
X_{24}	Low FMbulking	%	

5.1.4. Implemented control strategies

Several control strategies have been implemented and were compared to a default open loop base case (A_1). The settings of the open loop case considered in this study were summarized in Table 11. Next, sixteen control strategies [A = (A_2,...,A_{17})], summarized in Table 13 and Table 14, were applied to the activated sludge section. The simulation results (open loop case + 16 control strategies) are the starting point for the work presented in this paper. All the simulations (609) days were preceded by steady state simulations (200 days). Only the data generated during the last 364 days of simulation were used for plant performance evaluation.

Figure 27 represents the behaviour of the different controllers. In Figure 27a there is a snapshot of the DO profile in the controlled aerated reactor (AER3) with and without a controller. As can be seen from the dotted line representing the evolution of DO in that reactor, aeration intensity (KLa = constant) is not adequate during daytime and is excessive at night. With the DO controller (A_2) the oxygen is nicely maintenained on the set-point, but the aeration is varying a lot (results not shown). Figure 27b shows the evolution of nitrate nitrogen with and without controller in ANOX2, with manipulation of the internal recycle (Q_{intr}). Figure 27c shows the ammonium nitrogen in the third aerobic reactor (AER3) with and without a controller. Without the ammonium controller, the aeration set point (DO = 2 g(-COD) m^{-3}) is not adequate and needs to be changed according to the nitrogen load. Figure 27d shows the change in the MLSS set point when the temperature changes in order to keep the biomass active during winter periods. In Figure 27e illustrates the evolution of the effluent P when the metal salt is added. Finally, the control idea of Sumarcz-Gorska, which suggested stopping aeration as soon the respiration rate (OUR) drops below a certain threshold, was the inspiration behind this controller: if the respiration rate is sufficiently low, aeration is switched off in the three aerated reactors (AER1-2 & 3) and denitrification can take place as shown in Figure 27f.

Table 13. Control strategies evaluated in this case study.

Characteristics	3 DO controller	Ammonium controller	nitrate controller (Q_{intr})	nitrate controller (Q_{carb})	MLSS controller	Surmacz controller	Phosphorus controller
Reference	Vanrolleghem and Gillot, 2002	Vrecko *et al.*, 2006	Copp, 2002	Vrecko *et al.*, 2006	Vrecko *et al.*, 2006	Vanrolleghem and Gillot, 2002	Gernaey *et al.*, 2002
Measured variable(s)	S_O in AER, 1,2 & 3	S_{NH} in AER3	S_{NO} in ANOX2	S_{NO} in ANOX2	TSS in AER3	OUR in AER1	S_{PO4} in AER3
Controlled Variable(s)	S_O in AER 1,2 & 3	S_O set-point in AER1, 2 & 3	S_{NO} in ANOX2	S_{NO} in ANOX2	TSS in AER3	S_o set-point in AER1 & 2	S_{PO4} in AER3
Set point/critical value	2, 2 & 2 g (-COD)·m^{-3}	1 g N·m^{-3}	1 g N·m^{-3}	1 g N·m^{-3}	3500 g TSS·m^{-3} (if T< 15°C) 4000 g TSS·m^{-3} (if T> 15°C)	650 g COD·m^{-3} ·d^{-1}	1 g P·m^{-3}
Manipulated variable	K_La	S_O set point in 3 DO strategy	Q_{intr}	Q_{carb}	Q_w	S_O set point in 3 DO strategy	Q_m
Control algorithm	PI	Cascaded PI	PI	PI	Cascaded PI	ON/OFF cascaded PI	PI

Table 14. Combination of controllers in the different evaluated alternatives

Implemented in	3 DO controller	Ammonium	nitrate	nitrate	MLSS controller	Surmacz	Phosphorus
Alternative A_1							
Alternative A_2	X						
Alternative A_3	X		X				
Alternative A_4	X	X	X				
Alternative A_5	X	X	X		X		
Alternative A_6	X			X			
Alternative A_7	X	X		X			
Alternative A_8	X	X		X	X		
Alternative A_9	X						X
Alternative A_{10}	X	X					X
Alternative A_{11}	X	X			X		X
Alternative A_{12}	X		X			X	
Alternative A_{13}	X		X		X	X	
Alternative A_{14}	X			X		X	
Alternative A_{15}	X			X	X	X	
Alternative A_{16}	X					X	X
Alternative A_{17}	x				X	X	X

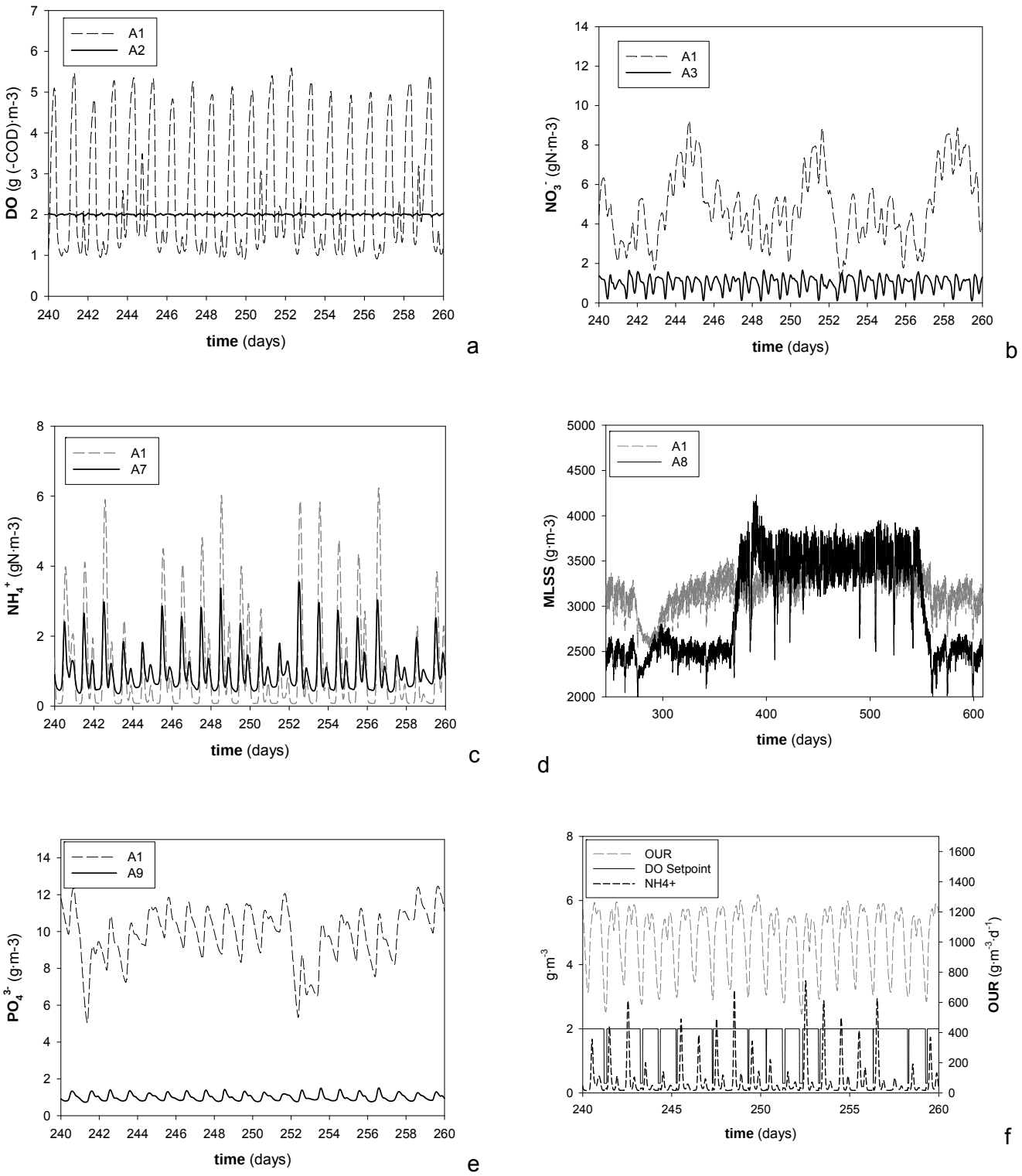


Figure 27. Behaviour of the different controllers studied: a) DO controller, b*) Nitrate controller by means of internal recycle, c) Ammonia cascade controller, d) MLSS controller, e) Phosphate controller with chemical addition, f) Surmacz controller*

5.2. Results

The behaviour of the proposed controllers is evaluated in simulation using the mentioned set of evaluation criteria. The results comprise a huge and complex multi-criteria matrix composed of 24 rows (evaluation criteria) and 17 columns (control alternatives) combining data of different nature e.g. environmental, economical and technical criteria (see a shanshopt of the evaluation criteria in Table 15 with the effluent criteria and the operational cost index). From the results summarized in this table it can be seen that it is difficult to drawn meaningful conclusions just with the visual inpection of the results. Also, it has to be added that this table is just a "small" portion of a bigger table with additional 15 criteria.

Table 15 Snapshot of some of the evaluation criteria for the 17 evaluation control strategies

	X_1	X_2	X_3	X_4	X_5	X_6	X_7	X_8	X_{15}
A_1	3.4	13.19	9.47	9.27	55.07	1.58	16.02	14040	12754
A_2	3.18	12.99	9.48	9.29	55.09	1.59	16.03	13906	12465
A_3	2.87	12.71	9.09	8.89	54.98	1.6	16.12	13432	12302
A_4	2.68	11.48	8.15	7.95	54.93	1.67	16.4	12590	12385
A_5	2.77	11.26	8.02	7.82	55.44	1.68	16.99	12579	12260
A_6	3.37	9.25	5.73	5.49	58.56	2.27	20.33	11098	20908
A_7	3.39	8.66	5.31	5.06	59.11	2.4	21.07	10777	19703
A_8	4.05	9.44	5.84	5.64	55.07	1.93	16.96	11060	18681
A_9	3.18	12.88	1.2	1.01	54.2	1.51	16.6	8213.9	19137
A_{10}	3.12	11.94	1.19	1.01	54.24	1.57	16.75	7984.3	18664
A_{11}	3.36	11.87	1.19	1	54.42	1.58	16.98	8104.6	18536
A_{12}	2.87	12.42	8.88	8.68	54.97	1.62	16.19	13235	12714
A_{13}	2.89	12.61	8.86	8.66	55.77	1.58	16.99	13402	12489
A_{14}	3.35	9.27	5.77	5.53	58.49	2.26	20.27	11113	22793
A_{15}	4.64	10.15	5.58	5.38	55.1	1.93	16.95	11312	22376
A_{16}	3.29	12.37	1.19	1	54.22	1.54	16.69	8150.4	19269
A_{17}	3.86	12.74	1.18	0.99	54.45	1.53	16.98	8526.1	19128

In order to overcome the limitations of evaluating multiple alternatives and multiple criteria Gernaey *et al.* (2007) suggested an approach based on a pairwise comparision. Thus, Figure 28 shows the correlation between the effluent quality index (X_8) and the operating cost index (X_{15}). At this time, this plot differentiates strategies without chemicals addition (A_1, A_2, A_3, A_4, A_5, A_{12} and A_{13}) with an external carbon source (A_6, A_7, A_8, A_{12} and A_{13}) and finally with metal precipitation (A_9, A_{10}, A_{11}, A_{16} and A_{17}). Figure 28b correlates the bulking due to low DO concentration and low F/M ratio and basically separates strategies with external carbon source (A_6, A_7, A_8, A_{14} and A_{15}) from the rest of the controllers. This approach gives a quick but only partial overview of controller performance. First and foremost the relationships between the control strategies discovered in each plot are based only on a single pair of criteria. Secondly, this approach is not capable of finding the main features amongst multiple criteria. Finally, it is not possible to know if the criteria used to find a relationship are really discriminant or not with respect to the rest of the criteria. Therefore, other tools are necessary to carry out further complex evaluations to deal with both complexity and ambiguity amongst those indices during multi-criteria evaluation. Thus, the evaluation matrix (data for 24 evaluation criteria collected for 17 control strategies) is subjected to the multivariate statistical techniques as suggested in Flores *et al.* (2007) to explore the behaviour of the control strategies tested.

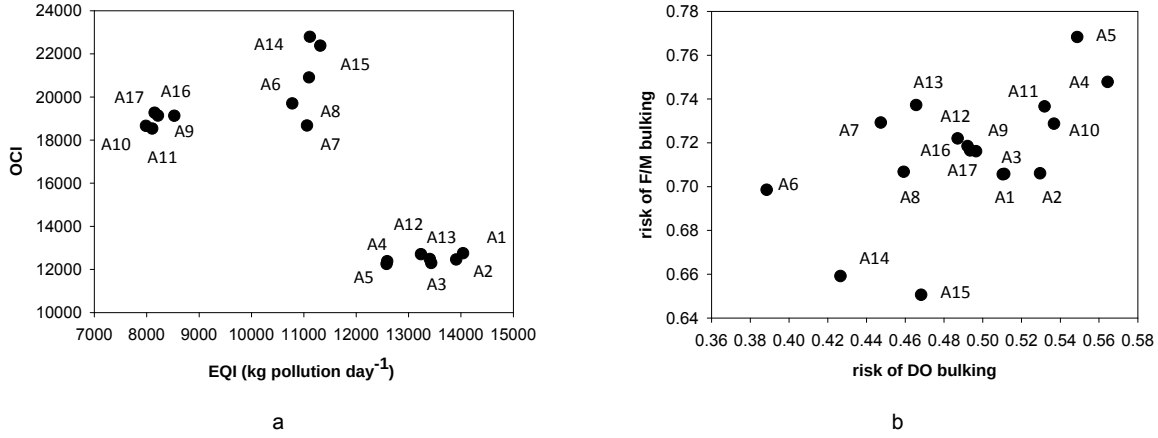


Figure 28. Representation of two pairs of evaluation criteria for all the evaluation control strategies

5.2.1. Multivariate statistical techniques used in this case study

Cluster analysis (CA) is an unsupervised pattern recognition technique that uncovers intrinsic structure or underlying behaviour of a data set without making a priori assumptions. Classification of the objects or a system into categories or clusters is based on the nearness or similarity of data points; see, for example, Hair *et al.* (1998). In this report hierarchical clustering is performed on the data set – after scaling the variables between 0 and 1 – by means of Ward's method, using the Euclidian distance as a measure of similarity. In equation 1 this distance is represented for n criteria, evaluated from a point [X ={X_1,..X_i,..X_n}] to a point [Y ={Y_1,..Y_i,..Y_n}].

$$d_n = \left(\sum_{i=1}^{n} \left| X_i - Y_i \right|^2 \right)^{1/2} \qquad \text{(Eq 1)}$$

Principal component Analysis (PCA) extracts the eigenvalues and eigenvectors from the covariance matrix of the autoscaled variables [X ={X_1,..X_i,..X_n}]. The set of [PC ={PC_1,..PC_i,..PC_n}] principal components (PCs) are the uncorrelated (orthogonal) variables obtained by multiplying the original correlated variables with the eigenvectors. Each eigenvector consists of a vector of coefficients (loadings) [a ={a_1,..a_i,..a_n}] as shown in equation 2. PCA allows the dimensionality of the original data set to be reduced with a minimum loss of information. Factor analysis (FA) further reduces the contribution of less significant variables obtained from PCA and results in the new groups of variables known as varifactors (VF) extracted through rotating the axis defined by PCA (Hair *et al.* 1998).

$$PC_j = a_{1,j} X_1 + a_{2,j} X_1 + ... + a_{i,j} X_i + ... + a_{n,j} X_n = \sum_{i=1}^{n} a_{i,j} X_i \qquad \text{(Eq 2)}$$

Discriminant Analysis (DA) is used to determine the variables which allow discrimination between two or more naturally occurring groups (Johnson and Wichern, 2002). It operates on raw data and the technique constructs a discriminant function [D ={D_1,..D_i,..D_z}] for each group (see equation 3) where j is the number of the function, C_j is the constant inherent to each function, k is the number of criteria used to classify a set of data into a given group, and b_i is the weight coefficient assigned by DA to a given performance evaluation criteria (X_i).

$$D_j = C_j + b_{1,j}X_1 + b_{2,j}X_2 + ... + b_{i,j}X_i + ... + b_{1,j}X_k = C_j + \sum_{k=1}^{k} b_{i,j}X_j \quad \text{(Eq 3)}$$

5.2.2. Cluster Analysis

Cluster analysis rendered a dendogram where all the implemented control strategies are grouped into three main statistically significant clusters (cluster 3.1, 3.2 and 3.3). The first (strategies A_1, A_2, A_3, A_4, A_5, A_{12} and A_{13}), the second (A_6, A_7, A_8, A_{14} and A_{15}) and the third (A_9, A_{10}, A_{11}, A_{16} and A_{17}) cluster corresponds to strategies with and without additional of external chemical products. If the clusters are further classified, five groups of control strategies can be found (cluster 5.1, 5.2, 5.3, 5.4 and 5.5). Thus, the second cluster (cluster 3.2) is subdivided into sub-group cluster 5.2 and 5.3 containing strategies A_9, A_{10}, and A_{16} and A_{11} and A_{17} respectively The second cluster is sub-divided into a subgroup of strategies A_6, A_7, and A_{14} (cluster 5.4) and A_8 and A_{15} (cluster 5.5) the clustering indicates that there are five different types of control strategies, where the presence and the absence of external chemicals and /or a MLSS controller are key elements creating the differences between the clusters (see Figure 29).

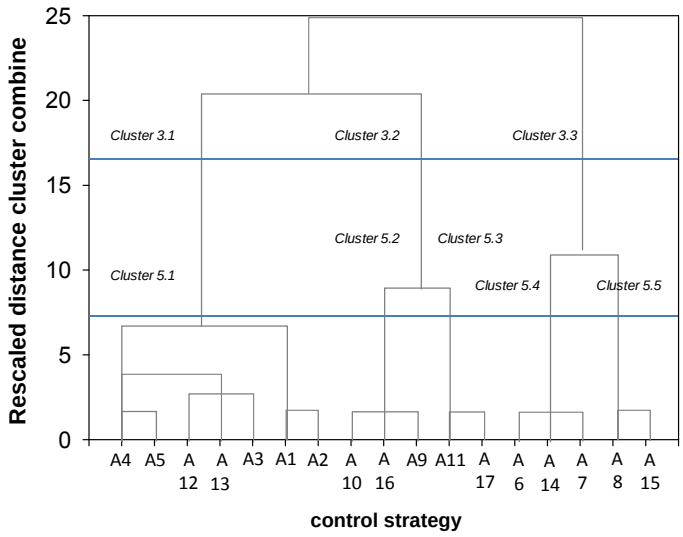


Figure 29 Dendogram showing clustering of the implemented control strategies in the Neptune

5.2.3. Principal component/Factor analysis

PCA/FA is applied to the autoscaled simulation output to compare the evaluation criteria between the implemented control strategies and to identify the most influential factors. PCA of the entire data set evolved four PCs with eigenvalues > 1. A varimax rotation if the PCs to four different VFs explained about 93 % of the total variance. The values of the PCs are further cleaned up with this technique and in VFs the original variables are contributed more clearly (see Table 16). The factor loadings are classified as “strong”, “moderate” and “weak” corresponding to the absolute loadings of >0.70, 0.70-0.50 and < 0.25 (**Liu *et al.*, 2003**).

VF1, which explains 42.18 % of the total variance, has strong (in bold) positive loading on X_5, X_6, X_7, X_9, X_{10}, X_{13}, X_{17}, X_{19}, X_{20} and strong negative loading for X_2, X_{16} and X_{23}. This VF describes the effect of the external carbon source addition. It is important to emphasize that a periodic addition of an external carbon source (X_{13}) implies the subsequent increase of the sludge production (X_9) and aeration energy (X_{10}). In addition, there is a decrease of the effluent total nitrogen (X_2 and X_{16}) as a direct consequence of the lower effluent nitrate

concentrations). Nevertheless, as a trade-off of the external carbon source addition, there is a decrease of the overall organic matter pollution removal efficiency (X_5, X_6, X_7, X_{17}, X_{19} and X_{20}) a higher demand of oxygen in the aerobic zone i.e. higher aeration energy (X_{10}). Thus, it is possible to promote operating conditions that potentially could lead to low DO bulking (X_{23}). VF2 which explains 24.9 % of the total variation is positively correlated with X_3, X_4, X_8, X_{21} and negatively with X_{12}. This VF highlights that with the addition of a metal salt (X_{12}) it is possible to achieve very low concentrations of phosphorus in the effluent (X_3, X_4 and X_{21}) improving the overall wastewater treatment removal efficiency (X_8). Criteria X_1, X_{17} and X_{24} present strong loading in VF3 (19.7% of the total variance) and indicates low nitrification efficiency. Finally VF4, which explains 5.6 % of the total variance, has strong positive loading with mixing energy. The criteria X_{22} (bulking due to influent C and N desequilibriums) is not included in the analysis because exhibits a constant value (i.e. variance zero). This is because the influent is allways the same from all the controllers.

Table 16. Loading of the evaluation criteria on the four first rotated PC for the complete data set.

	VF1	VF2	VF3	VF4
X_1	0.03	-0.18	**0.91**	0.15
X_2	**-0.81**	-0.02	-0.30	-0.33
X_3	-0.02	**0.99**	-0.12	0.04
X_4	-0.03	**0.99**	-0.12	0.04
X_5	**0.96**	0.24	0.00	0.06
X_6	**0.93**	0.14	0.26	0.21
X_7	**0.98**	-0.10	0.04	0.07
X_8	-0.06	**0.99**	-0.08	0.01
X_9	**0.70**	-0.24	*0.61*	0.24
X_{10}	**0.71**	0.19	*0.61*	-0.07
X_{11}	0.24	*-0.58*	0.55	0.18
X_{12}	-0.30	**-0.93**	-0.09	-0.16
X_{13}	**0.75**	0.09	*0.61*	0.14
X_{14}	-0.16	-0.22	0.02	**-0.88**
X_{15}	*0.55*	*-0.66*	*0.64*	0.01
X_{16}	**-0.86**	0.12	-0.09	0.13
X_{17}	**0.98**	0.00	-0.02	0.07
X_{18}	-0.16	-0.12	**0.95**	-0.11
X_{19}	**0.98**	0.00	0.00	0.07
X_{20}	**0.98**	0.01	0.04	0.08
X_{21}	0.19	**0.97**	0.02	0.13
X_{23}	**-0.77**	-0.10	-0.41	0.34
X_{24}	-0.26	-0.04	**-0.84**	0.14

It is important to highlight the role of some moderate factor loadings (0.70 – 0.5) have in the created factorial model. For example X_9 and X_{10} have a moderate role in VF3. The correlation between (X_9) and (X_1 and X_{18}) is mainly due to the improvement of the nitrification process when the airflow increases. Also, the increase the sludge production (X_{10}) consequently decreases the F/M ratio and finally increases the risk of bulking due to low F/M. Another example is the influence of the operating cost index (X_{15}), which is relatively high in VF1, VF2 and VF3. Thus, the addition of either an external carbon source (VF1) or a metal salt (VF2) and higher aeration energy (VF3) increase the global operating cost.

Once the principal components are identified and labelled, the scores obtained by the implemented control strategies, can be calculated as a linear combination of the original variables. The representation of the scores is depicted in Figure 30.

As expected, the results of PCA/FA are in good agreement with CA. Control strategies with external carbon source (cluster 5.4 and 5.5) present high scores in VF1 and are

characterized by high operating costs and low effluent nitrate concentrations in the effluent (more information about the abslute N values can be found in Table 15). Cluster 5.3 present high scores in VF2 associated to the addition of a metal salt and low effluent phosphorus concentrations (see Table 15 for effluent P values). This fact is attributed to the low soluble organic matter coming with the influent that makes a complete biological nitrogen removal really difficult without the addition of chemicals. Thus, in order to achieve low concentrations of nitrate and phosphates in the effluent it is necessary to add either external carbon source or a metal salt. Strategies A_4, A_5 (with ammonia controller and without chemical addition) presents the highest nitrification efficiencies. Low scores in VF4 are obtained by those strategies with an OUR controller ($A_{12,}$ A_{13}, A_{14}, A_{15}, A_{16} and A_{17}) mainly due to higher mixing energy consumption due to the activation/deactivation of the aeration system in the aerobic zone.

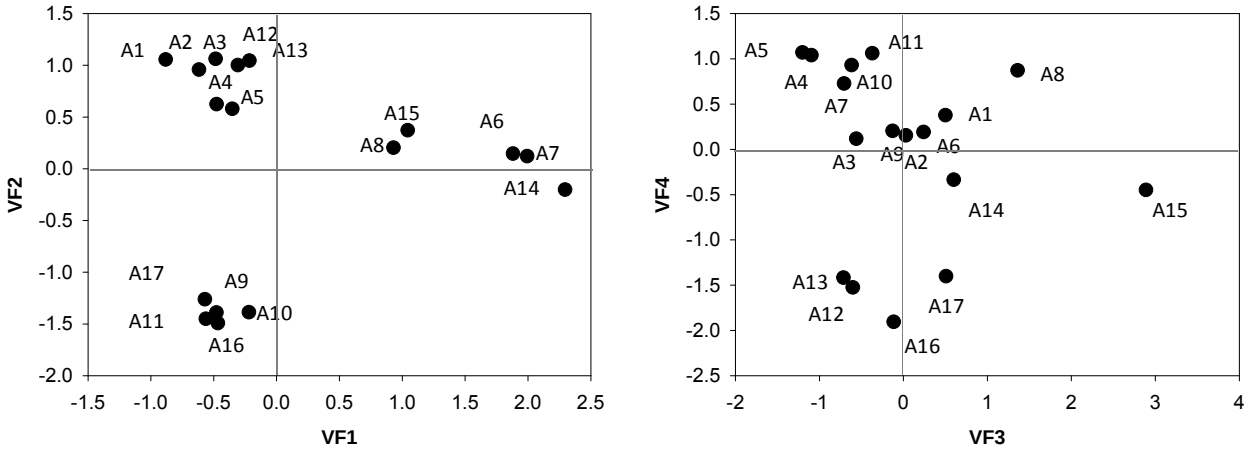


Figure 30. Principal component scores for the implemented control strategies for the principal component 1 and 2 (a) and for principal component 3 and 4 (b)

5.2.4. Discriminant Analysis

Finally discriminant analysis (DA) is performed aiming at dividing the original data set into three groups (control strategies with and without external chemical addition), identified by CA. The control strategy is the grouping variable, while all the evaluation criteria are the independent variables. DA is performed using all the evaluation criteria exept X_{22} (again because its nul variance) and it has rendered classification matrixes (CM) assigning 100 % of the cases correctly. The stepwise DA shows that criteria X_9, X_{10}, X_{17} and X_{21} are the discriminant parameters. The correct grouping pattern of DA coincides with the clusters obtained in CA. Both CA and DA predict important differences, operational costs and plant performance due to the impact of the addition of chemicals. The discriminant functions are listed in Table 17.

Table 17. Classification functions for discriminant analysis of the implemented WWTP control strategies

	Description	$b_{1,j}$	$b_{2,j}$
X_9	Sludge production (P_{sludge})	1.32	1.37
X_{10}	Aeration energy (AE)	1.29	0.93
X_{17}	CODviolation	1.41	0.40
X_{21}	Pviolation	1.83	-0.08

Figure 31, represents the scores of each control strategy to a determined discriminant function (Dj). Thus, D1, with the highest discriminant ability (Figure 31) separates cluster 3.3

of cluster 3.1 and 3.2. This is mainly due to the effect of the metal salt addition in the overall plant performance. The addition of the external carbon source explains the separation of cluster 3.1 and 3.2 as also shown in Figure 31.

According to DA the external carbon source and the addition of a metal salt show a clear effect on the overall process performance. As it was already observed during the PCA/FA analysis, the immediate effect of the metal salt is the dramatic reduction of the effluent phosphorus concentration (X_{17}). On the other hand, the addition of carbon as electron donor to enhance denitrification increased the sludge production (X_{10}), the aeration energy (X_{17}) and reduced the overall organic matter pollution removal efficiency (X_{17})

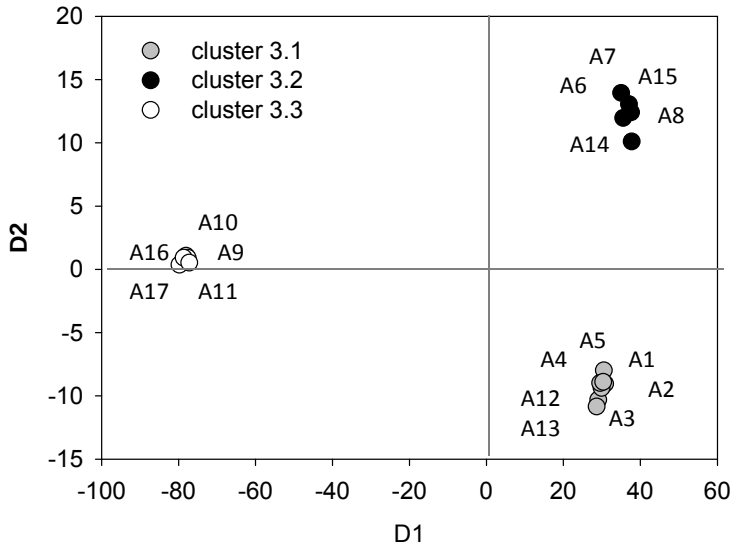


Figure 31. Classification functions for the discriminant analysis of the implemented control strategies

5.3. Discussion of the results

The Neptune Simulation Benchmark suggested in this report has proven to be a useful tool to evaluate control strategies strategies for nutrient removal. The development of the "*Neptune Simulation Benchmark*" allowed simulating different control schemes under the same conditions to ensure unbiased comparisions. The "Neptune Simulation Benchmark" supposes an extension from its predecessor the "*Benchmark Simulation Model No 1*" and brings a series of advantatges:

i) The plant configuration has been changed and it is properly designed according to accepted guidelines

ii) The influent file has been engineered to have low biodegradable fraction and a high nutrient load in order to test plant performance in extreme situations

iii) The control strategies are evaluated in a long term basis, being possible to see daily, monthly and seasonal variation

iv) The biological model includes (bio)chemical nutrient removal

v) A new set of indices has been adapted/proposed taking into account both nitrogen and phosphorus removal

Next, the behavior of the different controllers has been evaluated measuring their degree of satisfaction for different objectives using the *"Neptune Simulation Benchmark*". Nevertheless, the evaluation of the different alternatives was complex due to the fact that several factors, e.g. economic, environmental, technical, and legal, had to be taken into account simultaneously. The result was a hugely complex evaluation matrix consisting of a large number of criteria which was difficult to interpret, thus making it difficult to draw meaningful conclusions. The authors used a set of multivariate statistical methods suggested by Flores *et al.* (2007) in order to facilitate the analysis of the obtained multi-criteria matrixes.

The results of the multivariate analysis generated several points of discussion.

i) Cluster analysis (CA) proved to be a useful tool offering reliable classification of groups of control strategies according to their behaviour. CA performed this function well, rendering five groups of similar control strategies and identifying similar patterns in the control strategies with and without chemicals addition and/or TSS controller.

ii) Principal component analysis/factor analysis (PCA/FA) showed the main correlations between the evaluation criteria and the control strategies influencing those criteria. The five PCs identified were responsible for 93 % of the total variability (compared to 24 original variables). As a result, various synergies were identified e.g. carbon and metal addition with higher nitrogen and phosphorus removal. Tradeoffs were also identified e.g. chemical addition against higher operating costs, carbon addition against worse organic matter pollution removal. In addition, with the results of the factorial scores, it proved possible to identify the similarities between the implemented control strategies and the PCs extracted in the first part of the analysis. For example alternatives with an ammonia controller were located in the VF3 that correlated nitrification efficiency.

iii) Finally, discriminant analysis (DA) showed that only 4 parameters were useful for discriminating within the classes obtained by CA. Two discriminant functions were obtained, allowing 100% correct assignation and resulting in considerable data reduction. The representation of the discriminant scores allowed the important features amongst the discriminant variables and the group of classified control strategies to be found.

Overall, this analysis shows a straightforward way of characterizing alternatives: For example, in case of being necessary an environmentally friendly alternative, one would go for one of the control strategies within cluster 5.2, 5.3, 5.4 and 5.5, i.e. more expensive to operate but with lower eutrophication potential due to a reduced effluent N and P. On the other hand, if there are some budgetary limitations, the alternatives with the better chance of being successful are included in cluster 5.1. This method provides to process engineers, plant operators and decision makers more knowledge than current evaluation methods, highlighting pros and cons of each decision and enhancing the understanding of the whole evaluation process

Some of the conclusions that arise concerning the control behaviour have to be taken with care and it is dangerous to make universal assumptions. For example, in some cases, it was found that the implementation of some controllers, did not come up with substantial either cost reduction or effluent quality improvement to make the investment worthwile e.g. OUR and ammonium controllers. The controllers presented in this paper are selected and combined arbitrary and not optimized i.e. the values were taken from literature. Hence, the simulations are result of the complex interactions amongst them. For this reason, they do not necessarily reflect their sole and true behaviour. Rather, the analysis presented hereby is intended to be valuable research tool to coordinate the discussion and plan future research activities in order to identify the performance of some control strategies at the time to handle nutrient removal strategies. The authors will extend the analysis with additional simulations

modifying the setpoints of the proposed controllers in order to evaluate the implication of, for example, higher or lower oxygen, MLSS, ammonia or nitrate in the bioreactors.

It is important to point out that the results of this base case analysis depend strongly on the model selection prior to performing the simulations. Even though ASM3 + Bio+ Chem P is accepted as one of the models that is probably best describing nutrient removal in plant treating municipal wastewater when modelling activated sludge plants, there is often disagreement on the best model to apply for a given case. The representation of biomass decay (Siegrist *et al.*, 1999), the modelling of nitrogen removal (Henze *et al.*, 2000) and the oversimplification of the settling models (i.e. non-reactive in most cases, despite the fact that a significant amount of biomass is often stored at the bottom of the secondary clarifier, e.g. Gernaey *et al.*, 2006b) are key issues that are still under discussion. For this reason, the use of different model descriptions could result in different results and also conclusions.

Another interesting point is the impact of the initial list of evaluation criteria. The results of the multivariate analysis showed that redundant information is included during the analysis, and only few of the initial 24 set of criteria present a clear variation from one alternative to another. Nevertheless, the reader should be aware that that it is impossible to know a priori which would be the main correlation between evaluation criteria and the alternatives generated. Each PCA model is really case-specific and some changes might happen from one study to another when different control strategies and evaluation criteria involved. For this reason, the authors advocate for the use of techniques such PCA/FA to improve the accessibility to the information needed for effective evaluation of control strategies. As a side effect, there is also a reduction in the cognitive load on the decision maker, yielding more knowledge than current evaluation methods and enhancing understanding of the whole evaluation process.

5.4. Evaluation of control strategies using life cycle analysis

Beyond the multicriteria analysis for evaluating best control options in a multidimensional way, Life Cycle Analysis (LCA) can also be used to evaluate control strategies strictly from an environmental point of view. This subject is treated in this separate section since LCA is a weighted sum of some of the criteria used in the previous analysis and the multidimensional analysis would mask the LCA outcomes. LCA criteria can be included into the newly developed simulation benchmark for evaluating control strategies. An LCA allows defining WWTP optimization goals taking into account potential impacts of treatment operation on e.g. eutrophication and climate change over the lifespan of a plant.

In a first step within the LCA framework, the parameters and resource consumptions are selected as a measure of the potential impact of control strategy inclusion. Overall, energy consumption, effluent nutrient concentrations, and chemicals used can be included as evaluation criteria. Savings in construction resources can also be considered since control implementation allows increasing the treatment capacity without the need of extending the plant. The potential impact of the analyzed control strategies is evaluated in terms of greenhouse and eutrophication effects. The results obtained from the different control strategies discussed in the previous section are evaluated using LCA indices. Following, an explanation on the LCA calculation and a discussion of the results is presented.

The variables included in the LCA, that define the boundaries of the studied system are presented in Table 18. Neither the N_2O produced nor the acute effect of ammonium toxicity are included in the calculations. The functional unit of the study is $1m^3$ of treated wastewater.

Table 18. Variables and associated impact

Variables	Impact factors
Nitrogen	37.23 mPET*year / kg N

Phosphorus	269.2 mPET*year / kg P
Electricity consumption	0.12324 mPET*year / kWh
Sludge production	0.1 mPET*year / kg 37% DM sludge
Infrastructure	0.127 mPET*year / m³ influent treated
$FeCl_3$ 40% dosing	2.611 mPET*year / kg
Sodium acetate dosing	0.7781 mPET*year / kg NaOAc

The impact potentials are expressed in person-equivalents per year (mPET*year). This unit comes from the normalization of the impact potentials. In the method used in NEPTUNE this reference information is represented by the total impact potential in the reference region divided by the number of persons in the region. The normalised impact potential (NIP) is then calculated by taking the impact potential of a given impact category (e.g. ETWC) and dividing it by the corresponding normalization reference:

$$NIP_{impact\ category\ A} = IP_{impact\ category\ A} / NR_{impact\ category\ A}$$

The values obtained for each impact cathegory have been obtained from the Neptune databases. More information can be found in the deliverables provided in the WP4.

The use of the LCA to evaluate the control strategies is based on multiplying the impact factors and the variables obtained in the simulations and dividing by the volume of treated wastewater, which is the functional unit (e.g. if 10 kWh/d are consumed the impact potential from electricity consumption becomes: NIP=10*0.12324mPET·year/d = 1.2324 mPET·year/d. If 10 m^3/day of wastewater are treated then the $NIP_{electricity}$ = 0.12324mPET·year/m^3).

For each simulation the avoided and the induced impacts are compared to a reference case (in this study “no treatment” is the reference case). The avoided impact is obtained from the difference between the effluent nutrient impact for each of the scenarios and the effluent nutrient impact of the “no treatment” situation. The induced impact is calculated from the electricity consumed, the sludge produced, the infrastructure and the chemicals added for each of the scenarios.

The induced and avoided impacts for some of the evaluated control strategies are presented in Figure 32. The reference case “no treatment” would generate an impact of 4.61 mPET·year/day. As can be observed, when having no control (A1) the induced impact is mainly due to construction of the WWTP and some impact is avoided. The implementation of controllers for DO, internal recycle, ammonia and TSS (A2, A3, A4 and A5) leads to an increase of the avoided impact with minimal change in the induced impact. **The most environmental friendly strategies are the ones that include metal and carbon addition (from A6 to A10). These strategies allow reducing significantly the concentration of ammonia and phosphorus in the effluent, which have a high impact value.** Therefore, the avoided impact is much higher and the induced impact increases due to chemical use. Evaluation of control using LCA criteria will always lead to better results for strategies that improve nutrient removal. **As can be seen in Figure 32 energy consumption does not have a significant impact on the induced impacts. Therefore, control strategies aimed at reducing energy consumption will not be sensitive to this LCA approach.** It is worth to mention that the analysis has been conducted with no weights for the different impact categories. At this moment there is no agreement on the weights and further discussion is needed between decision-makers and researchers to define the relative importance of the different impact cathegories.

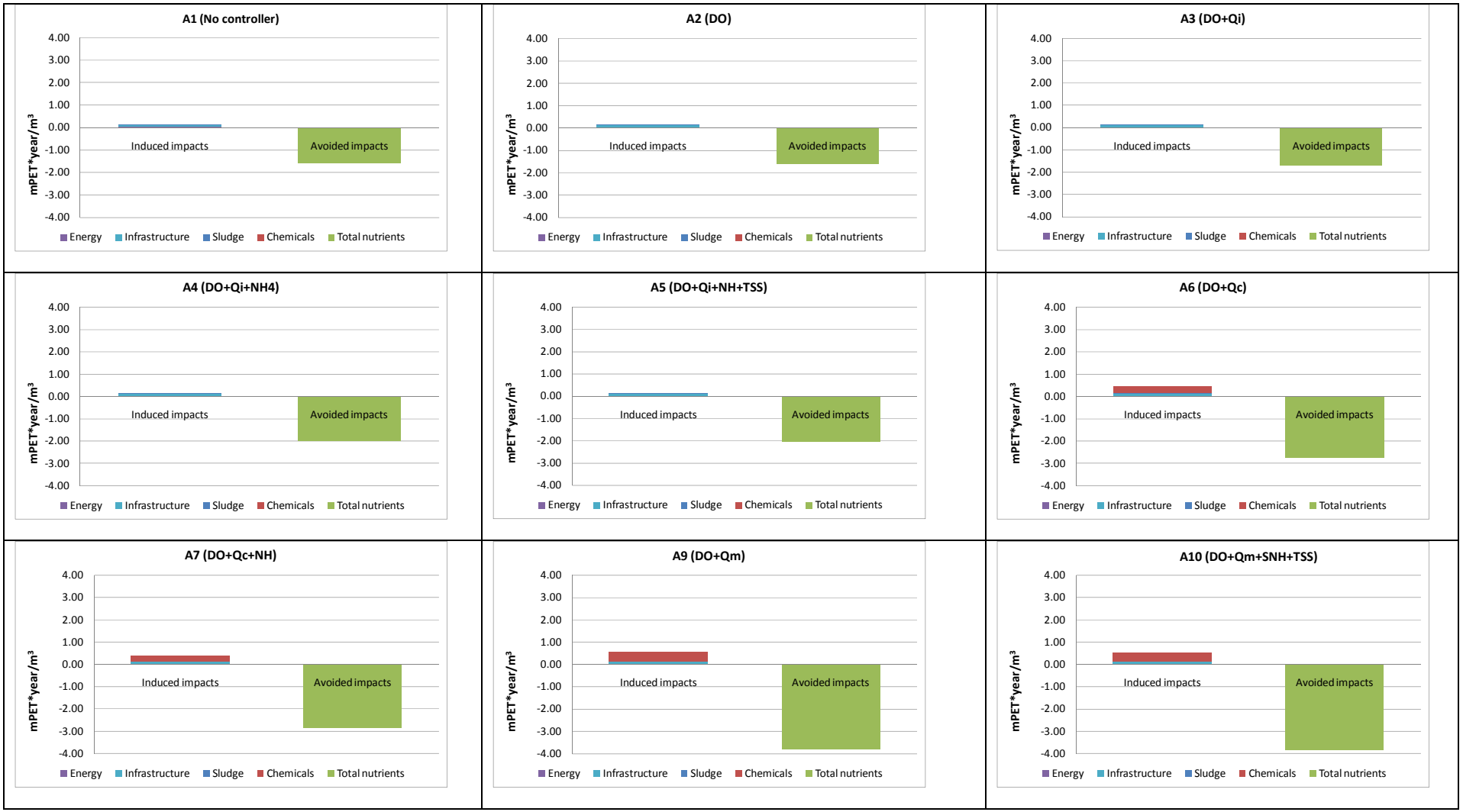


Figure 32. Induced and avoided impacts for some of the evaluated control strategies

6. Control of full-scale nutrient removal plants with in-situ online sensors

6.1. Introduction

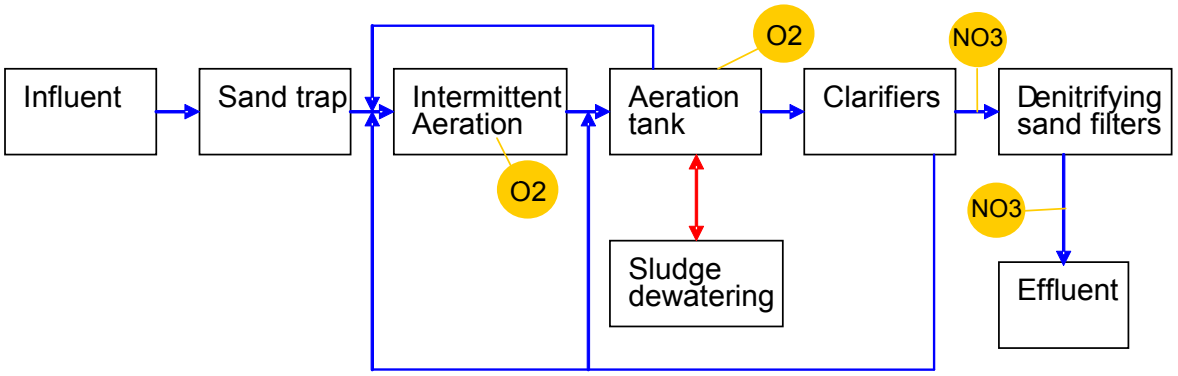
Online control for nutrient removal is standard practice in municipal wastewater treatment facilities. Until recently, the goal of online control at Aquafin was to reach the effluent consent at the lowest cost. Currently there is no stimulus to produce a cleaner effluent than strictly necessary since Aquafin doesn't pay a levy for the residual pollution. This report evaluates a new methodology to assess and tune online controllers. The new methodology tries to reduce the impact of wastewater treatment following a life cycle analysis approach.

For this purpose dynamic, calibrated, ASM2d models were made of 3 full-scale WWTP's in Flanders on which the two methodologies were compared.

6.2. Outline of the 3 plants

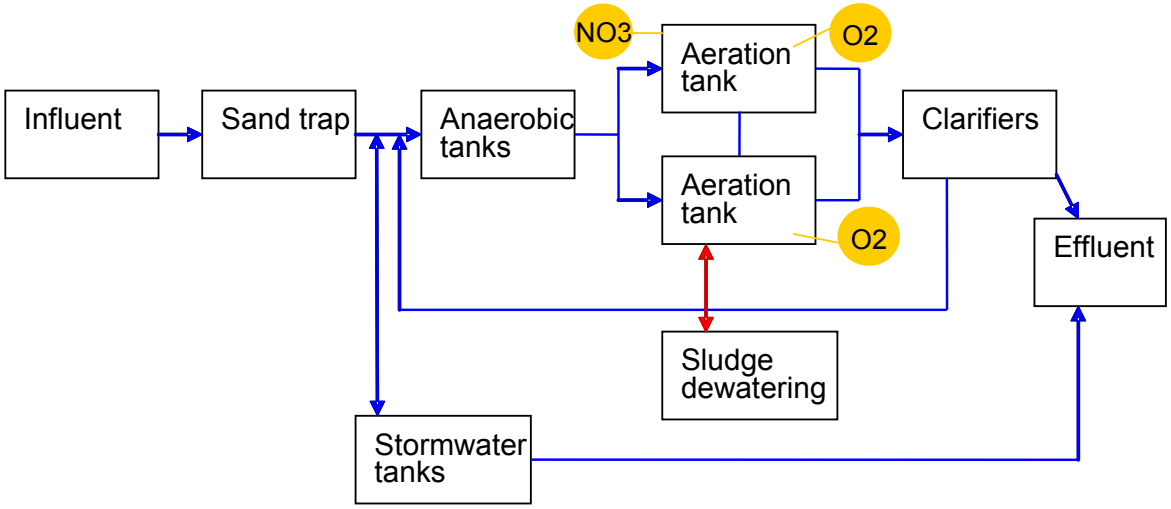
6.2.1. Plant 1

Plant 1 treats the wastewater of 27.000 people (3Q14). The water is pumped up with screw pumps and flows through an aerated sand trap followed by a tank with intermittent aeration before entering the aeration tank (continuous aeration). After having passed through the clarifiers, sodium acetate is added to remove nitrate in the continuous denitrifying sand filters. Mixed liquor from the continuous aeration tank is pumped back to the intermittent aeration tank. Currently both aeration tanks have oxygen sensors that control the aerators (PID control). Sodium acetate dosage to the sandfilters is controlled dynamically using flow, nitrate-influent and nitrate-effluent measurements.



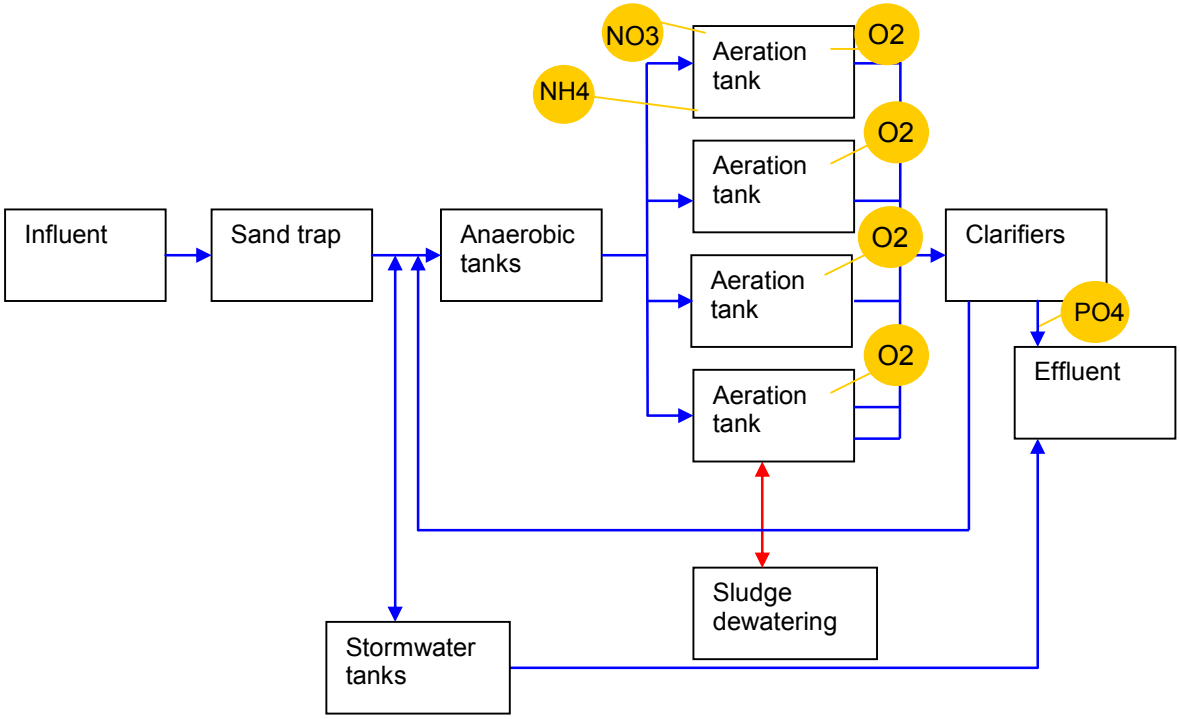
6.2.2. Plant 2

Plant 2 treats 6Q14 of 100.000 people biologically (stormwater tanks are not used anymore). The water is pumped up with screw pumps and flows through an aerated sand trap followed by a series of anaerobic tanks before entering the intermittent aeration tanks. A nitrate sensor controls the length of the aerated phase. The oxygen sensors control the aerators during an aerated phase.



6.2.3. Plant 3

Plant 3 treats 6.8Q14 of 270.000 people. 5Q14 is treated biologically. The water is pumped up with screw pumps and flows through an aerated sand trap followed by a series of anaerobic tanks before entering the 4 parallel aeration tanks. Aeration is intermittent and the length of the aerated phase is controlled by a rule based control algorithm that uses the online ammonium and nitrate signals. The oxygen sensors control the blowers during the aerated phase. Metal salt is dosed in the aeration tanks proportionally to the phosphate concentration in the effluent.



6.3. Calibrated model of the 3 plants

6.3.1. General information

To calibrate a dynamic WWTP model the influent composition is essential. This is problematic since only once a week (for the bigger plants, the sampling frequency is even lower at smaller plants) the influent is sampled and BOD, COD, TN, TP and SS are analysed. The influent composition of the days on which no sampling took place needs to be predicted. An in-house developed grey box model is used for that purpose. The model takes into account sedimentation and resuspension in the sewer system and automatically produces an influent file.

For the calibration itself we worked with the standard parameters for the biological models (Henze *et al.* IWA report No 9). The only parameter which was altered was the saturation coefficient for ammonium of the nitrifiers. Tuning of the models was done solely by changing the influent fractionation parameters (Stowa protocol) and the specific aeration energy (kg O_2/kWh).

A power logger was used to measure power consumption of the most important energy consumers responsible for about 90% of the total consumption. This information guaranteed a limited offset between real power consumption and modelled power consumption. It also allowed to correct existing online power measurements that are based on measurements of one phase only, disregarding the power factor.

Calibration results for the 3 plants are presented below in a number of graphs. The first 3 graphs show ammonium, nitrate and phosphate concentrations. The black dots represent the real measurements, the blue line is the modelled effluent quality. A pie chart is also included that shows the energy requirement of the different processes. Finally the modelled monthly energy consumption is plotted versus the real one in a bar graph.

6.3.2. Plant 1

Agreement between model and reality was quite good for ammonium and nitrate. However, there was a strong offset between modelled and measured phosphate effluent. This offset was investigated in detail but the problem could not be pinpointed. Phosphate was removed chemically in the plant and real iron dosage was imposed to the model. Usually when there is chemical P removal models and reality correspond quite well. The following hypotheses were put forward:

- Although an anaerobic tank is not present phosphate accumulation organisms are present (this was also observed in an AQF MBR without anaerobic tank).

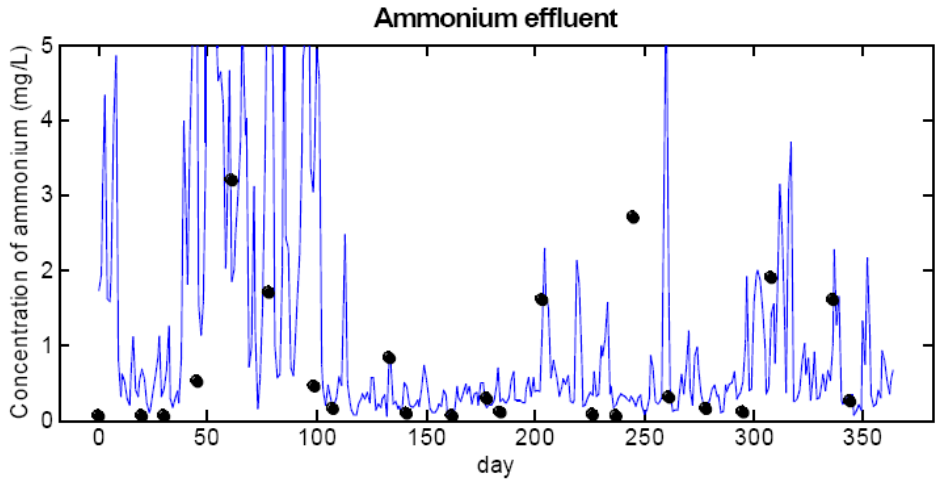
- We know the groundwater in the area contains a lot of iron. This iron is bound to reducing substances (e.g. S^{2-}) and is released, becomes available, in the aeration tank and then reacts with phosphate.

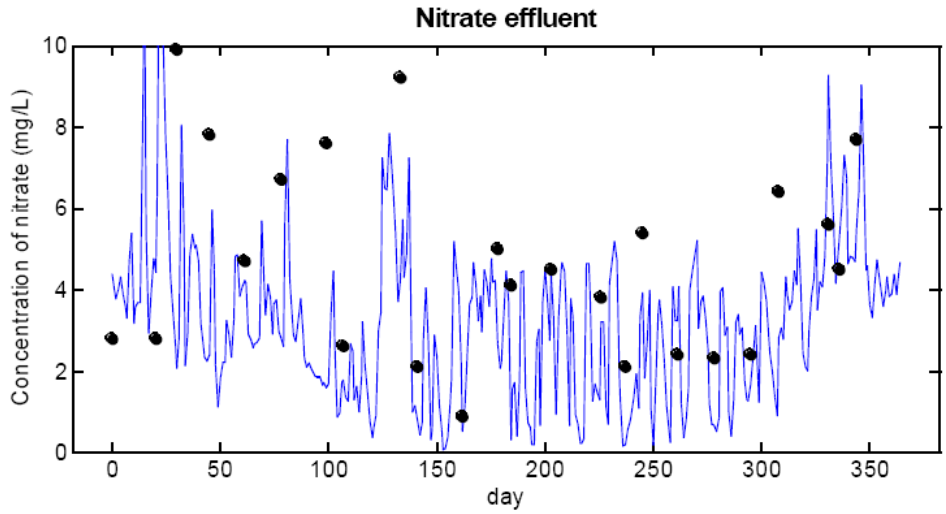
Further investigation is necessary to check both hypotheses.

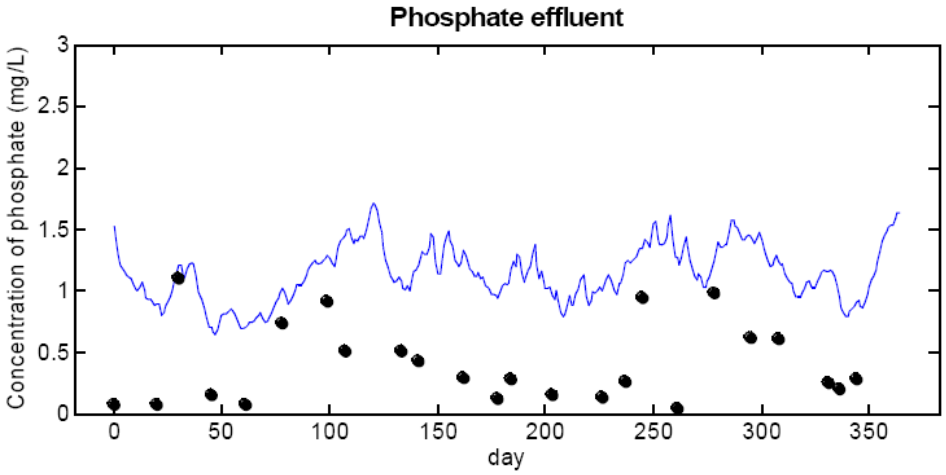
Both the energy needed for heating the buildings and for the sludge line were surprisingly high. Installed electrical heating power was 80 kW(!) and the sludge dewatering hall was not insulated. Thickened sludge from other installations in the neighbourhood is dewatered in plant 1 and this explains the high power consumption for the sludge line. The sludge line power consumption encompasses the power for the mixers in the sludge buffers, the centrifuges and the dewatered sludge pumps.

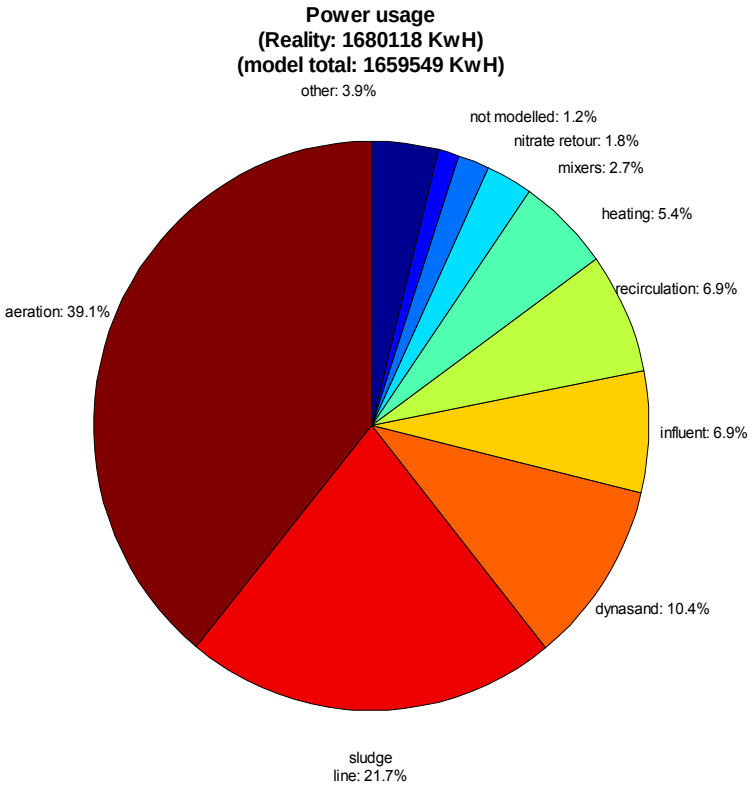
The deviation of the modelled sludge production, chemical dosage and power consumption from the real ones is demonstrated in the table below.

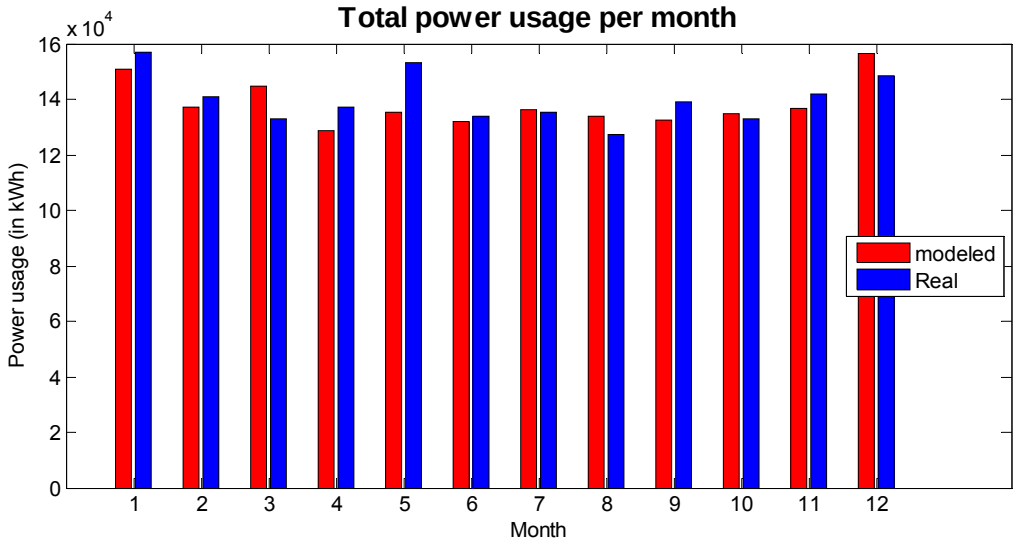
	Modelled amount	Deviation (model/reality)
Waste sludge (ton DS)	452	6%
Carbon source (m^3) (80 g COD/l)	773	-8%
Iron dosage (m^3) ($FeClSO_4$ 40%)	109	0 %
Power used (kWh)	1659549	-1%


Ammonium effluent
Concentration of ammonium (mg/L)
day
0
1
2
3
4
5
0
50
100
150
200
250
300
350


Nitrate effluent
Concentration of nitrate (mg/L)
day
0
2
4
6
8
10
0
50
100
150
200
250
300
350


Phosphate effluent
Concentration of phosphate (mg/L)
day
0
0.5
1
1.5
2
2.5
3
0
50
100
150
200
250
300
350





6.3.3. Plant 2

Agreement between model and reality was quite good for ammonium, nitrate and phosphate. An acceptable overestimation for ammonia and an underestimation for phosphate is noted.

The power measurements in combination with the model demonstrated a remarkably low SAE of 0.6 kg O_2/kWh for the surface aerators (Aquaturbo). As a consequence aeration has an unusually large share in the total energy consumption. The low SAE was cross checked with the aeration tests done at the start-up of the plant. Already at start-up the requested

SAE of 1.2 kg O2/h was not achieved. The SAE was also calculated with a static aeration tank design model. The static model predicted an SAE of 0.7 kg O_2/kwh, confirming the outcome of the dynamic model.

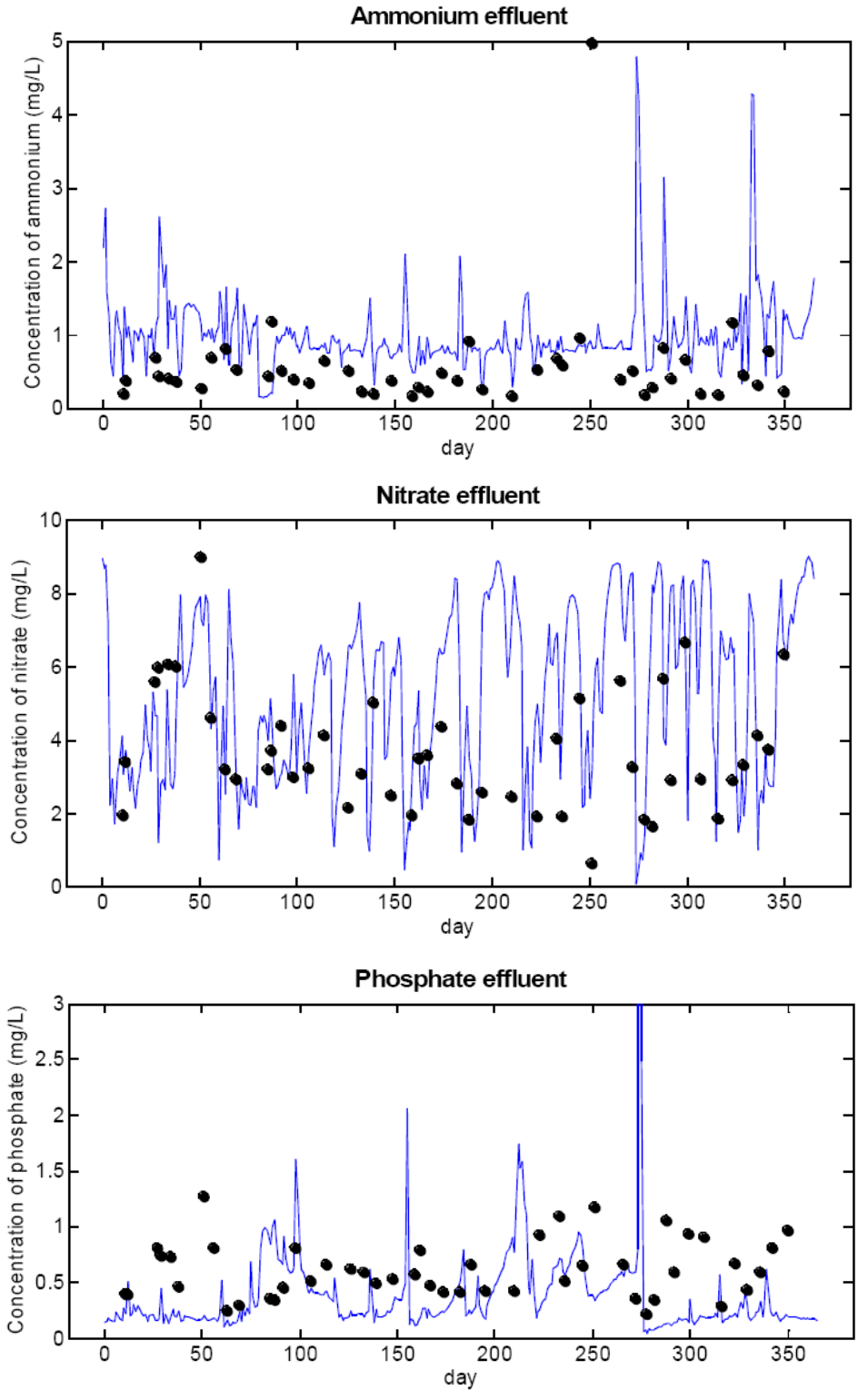
After internal discussion it was decided that the aeration system of plant 2 needs to be replaced ASAP by bubble aeration. This would halve the total power consumption. The payback time for the replacement is less than 3 years.

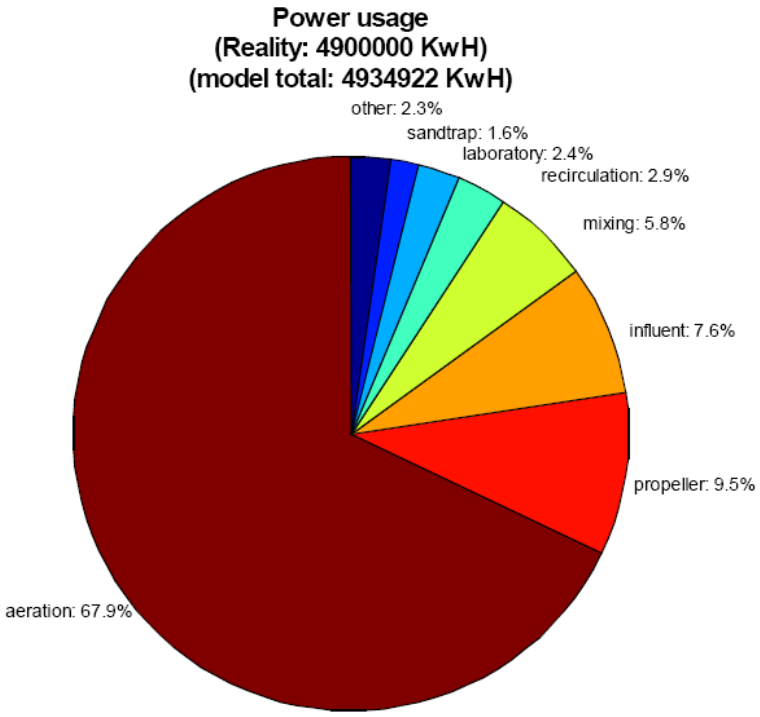
In the scenario analysis part where we assess and tune different controllers we have considered plant 2 with the current controllers and settings but with a new and more efficient aeration system as the base scenario to prevent overestimation of cost savings.

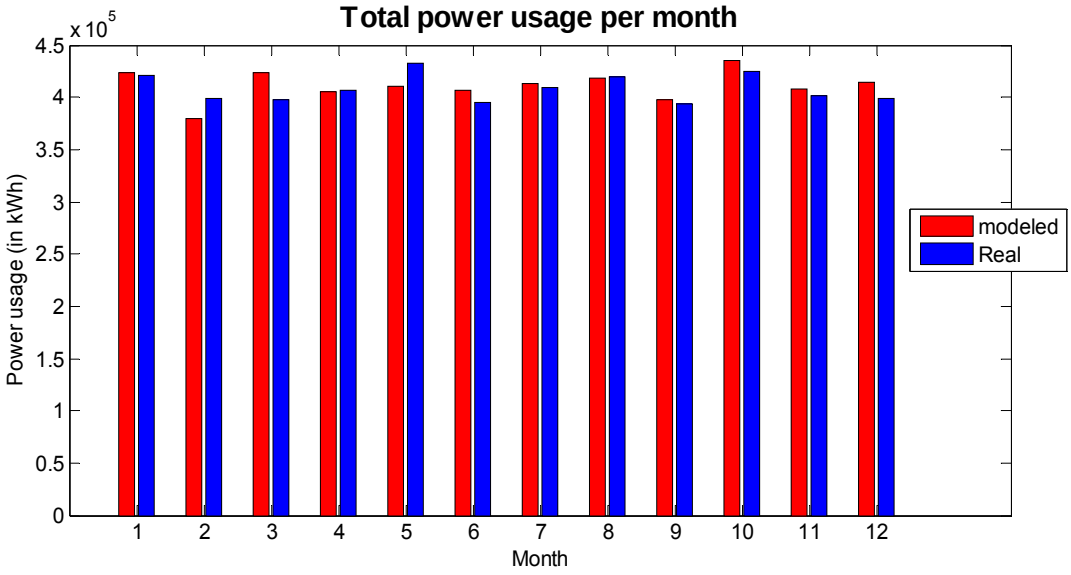
Energy consumption of the sludge dewatering system (other 2%) was low because a belt press was used (low energy requirement) and because no external sludge was dewatered at plant 2.

The deviation of the modelled sludge production and power consumption from the real ones is demonstrated in the table below.

	Modelled amount	Deviation (model/reality)
Waste sludge (ton DS)	2525	0%
Carbon source (m^3) (80 g COD/l)	0	0%
Iron dosage (m^3) (FeClSO4 40%)	0	0%
Power used (kWh)	4933799	1%


Ammonium effluent
Concentration of ammonium (mg/L)
day
Nitrate effluent
Concentration of nitrate (mg/L)
day
Phosphate effluent
Concentration of phosphate (mg/L)
day


Power usage
(Reality: 4900000 KwH)
(model total: 4934922 KwH)
other: 2.3%
sandtrap: 1.6%
laboratory: 2.4%
recirculation: 2.9%
mixing: 5.8%
influent: 7.6%
propeller: 9.5%
aeration: 67.9%


Total power usage per month
x 10^5
Power usage (in kWh)
4.5
4
3.5
3
2.5
2
1.5
1
0.5
0
1
2
3
4
5
6
7
8
9
10
11
12
Month
modeled
Real

6.3.4. Plant 3

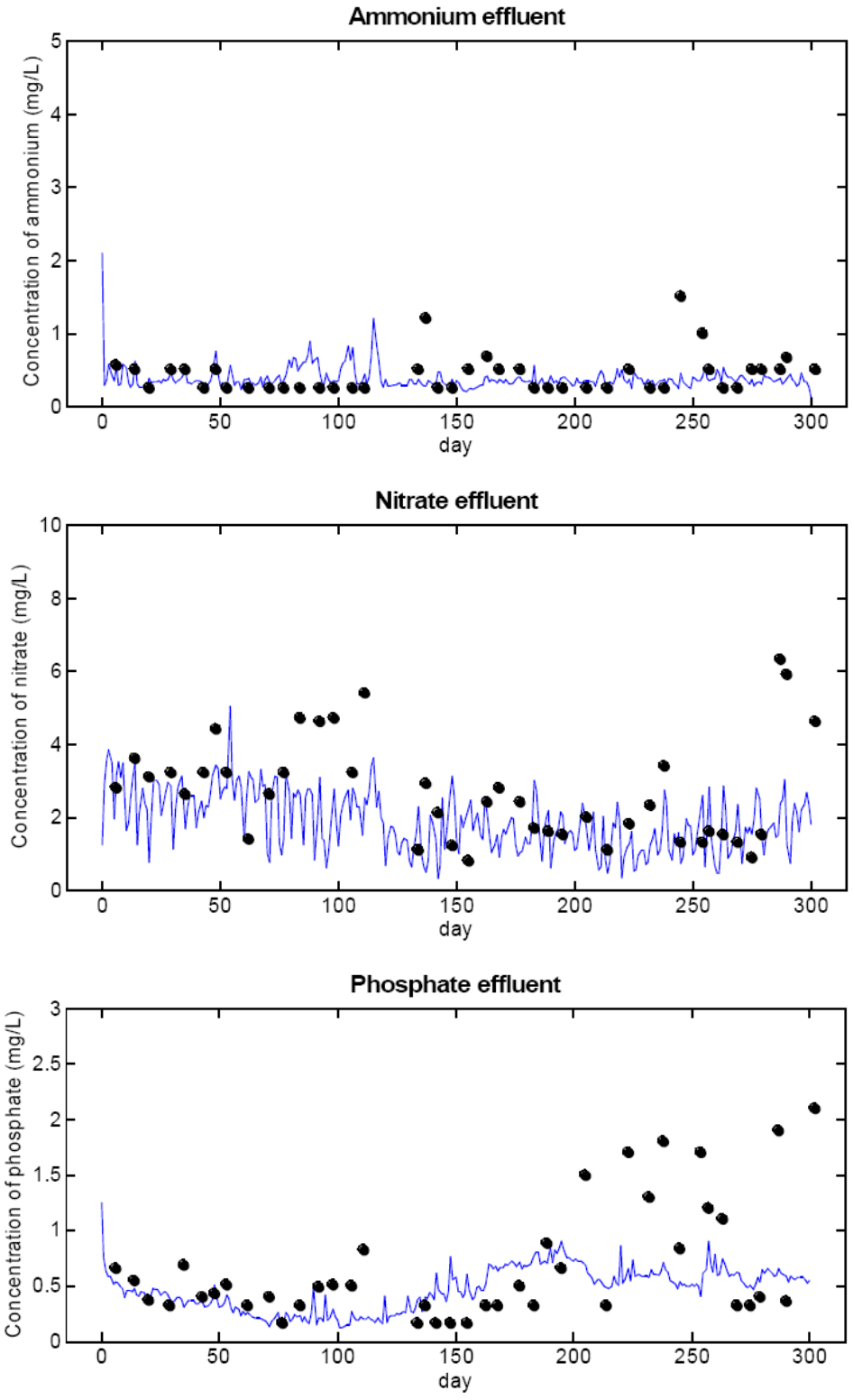
Agreement between model and reality was on average quite good for ammonium and nitrate. However, the modelled effluent concentrations were less dynamic than reality. Initially the correlation for phosphate is very good but near the end of the modelled year worsening of bio-P performance was not observed to the same extent in the model.

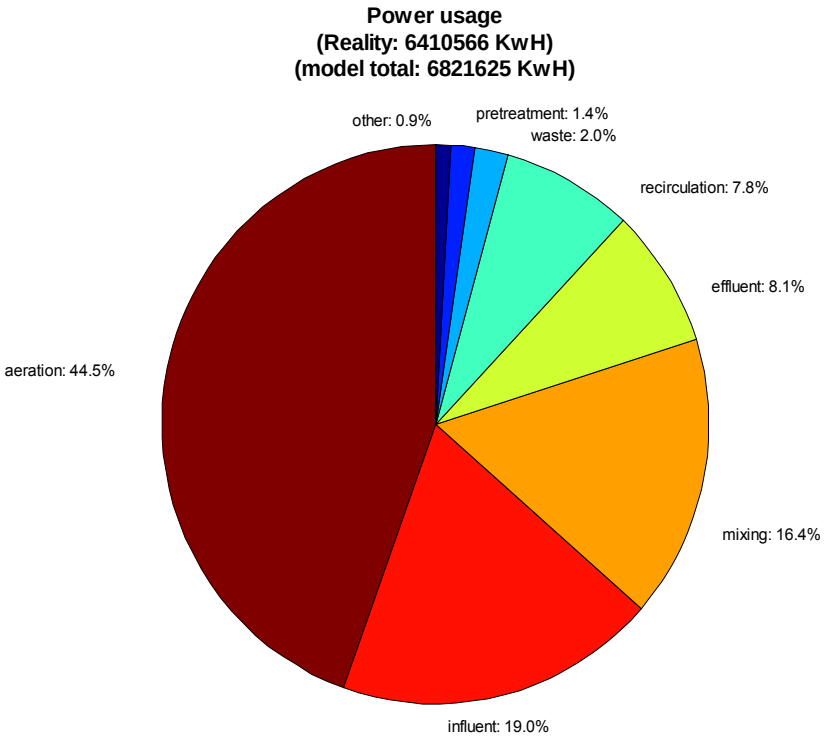
Dewatering and drying of sludge at plant 3 is done by a third party and therefore energy consumption of the sludge line was not modelled. We did model the energy consumption of the waste sludge pump though. Plant 3 is located close to the sea and at high tide effluent needs to be pumped up. The energy requirement for the effluent pumps mounts up to 8.1 % of the total.

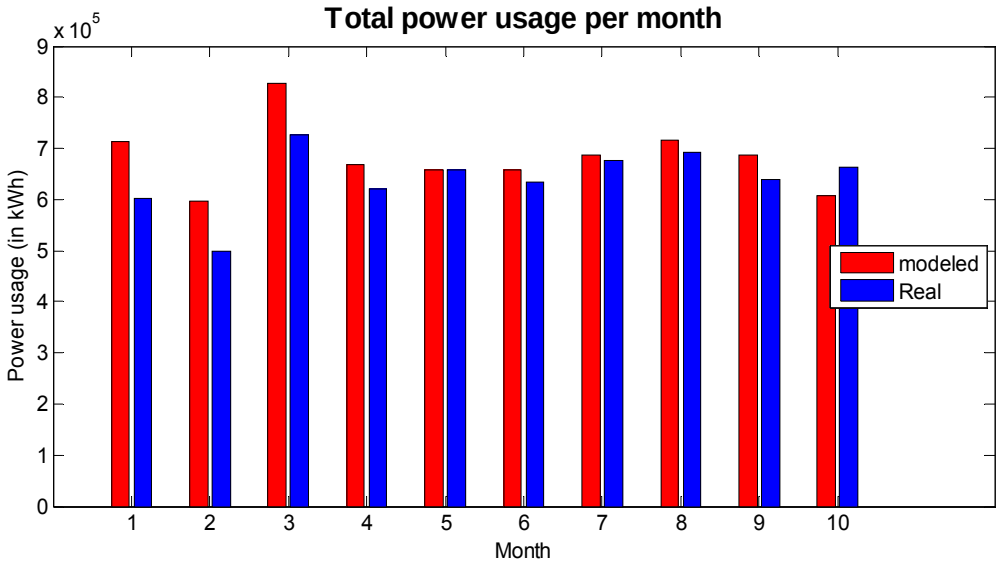
For the first 3 months there is a large deviation between modelled monthly energy consumption and reality. During the rest of the year (only 300 days were modelled) model and reality are in close agreement. We attribute the deviation of the first 3 months to the temperature dependency of the efficiency of the volumetric blowers which was not modelled.

The deviation of the modelled sludge production, iron dosage and power consumption from the real ones is demonstrated in the table below.

	Modelled amount	Deviation (model/reality)
Waste sludge (ton DS)	3929	4%
Carbon source (m^3) (80 g COD/l)	0	0%
Iron dosage (m^3) ($FeClSO_4$ 40%)	225	-13 %
Power used (kWh)	6821625	6%


Ammonium effluent
Concentration of ammonium (mg/L)
day
Nitrate effluent
Concentration of nitrate (mg/L)
day
Phosphate effluent
Concentration of phosphate (mg/L)
day





6.4. Comparison of the two optimisation strategies

The first optimisation strategy, reaching the effluent consent at lowest cost, doesn't need any explanation. The second one, achieving the lowest impact, however, does.

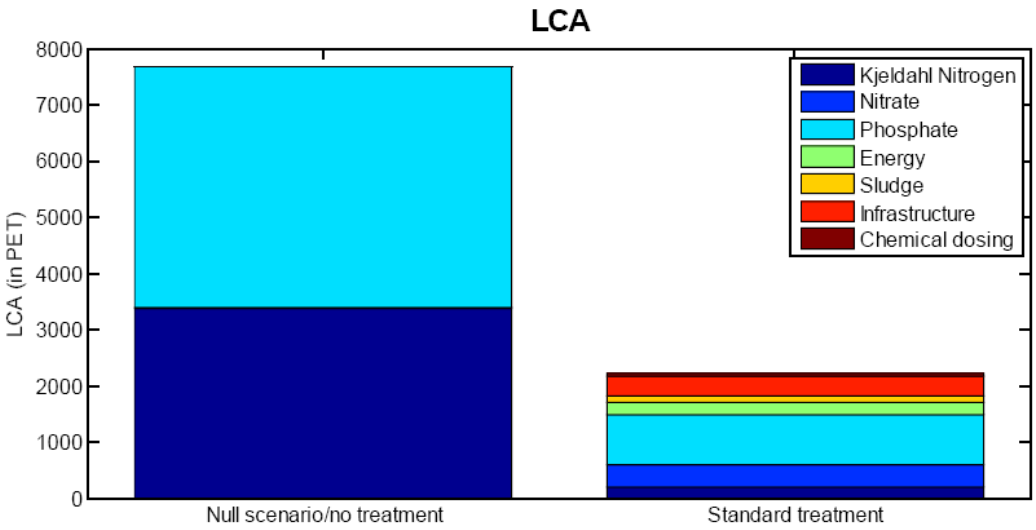
Impact is expressed as mPET: milli people equivalents targeted. 1 PE represents the environmental impact of 1 hypothetical person in a defined country and year. Impact is composed of a number of impact categories such as global warming, eutrophication, acidification, ozone depletion, ecotoxicity, human toxicity, …..

E.g. the EDIP97 methodology normalises the global warming impact of 1 PE to 8700 kg CO_2-equivalents per year.

Impacts for most inputs and outputs of a WWTP were provided to us by Henrik Fred Larsen, the LCA specialist of the NEPTUNE consortium. The values we used are listed below.

Parameter	Impact
Nitrogen	37.23 mPET / kg N
Phosphorus	269.2 mPET / kg P
Electricity consumption	0.12324 mPET / kWh
Sludge production	0.1 mPET / kg 37% DM sludge
Infrastructure	0.127 mPET / m³ influent treated
$FeCl_3$ 40% dosing	2.611 mPET / kg
Sodium acetate dosing	0.7781 mPET / kg NaOAc

In the graph below the impact of wastewater discharge with and without treatment is shown for plant 1. The low impact of energy and the rather high impact of effluent phosphate is noted.



6.4.1. Plant 1

Plant 1 is equipped with an advanced controller for dosing sodium acetate. Aeration in the aeration tank is continuous and the length of the aerated phase in the intermittent tank is controlled manually. During aeration a PID controller is responsible for maintaining the oxygen concentration at a predefined set point.

The following controllers/settings/changes were tested:

- Change from continuous aeration in the aeration tank and manual control in the intermittent tank to online control of aeration length using an extra NH_4 sensor.
- A strongly fluctuating MLSS concentration was noted. Existing manual control of the waste pump was replaced by automatic control based on a turbidity sensor. This action was intended to reduce the risk of solids washout.

- Pumping capacity from the aeration tank to the intermittent tank was doubled
- Oxygen setpoint was changed in the range 0.5 – 3 mg/L
- Install mixers in the aeration tank (without mixers a pulsed aeration is necessary during denitrification in the second aeration tank to avoid sludge settling)
- Change the settings of the sodium acetate controller (denitrifying sandfilters)

In total roughly 2000 simulations were run and the results of the best scenario using the classic approach and the best one using the lowest impact approach are presented below and compared with the current situation. Both 'best' scenario's used automatic aeration control, automatic waste pump control and doubled pumping capacity of the internal recycle pumps.

From the table below one can deduce that online control of the length of the aerated phase allows more denitrification in the aeration tanks leading to impressive reductions in sodium acetate consumption. The classic scenario obviously leads to the highest cost reduction but also to a lower impact. The lowest impact scenario tends to result in an improvement of the effluent quality.

If one moves from the classic to the lowest impact optimisation strategy an extra impact reduction of 1 % will cost about 1 % extra. Or, at that point, the marginal cost of impact reduction with 1 % is 1 %.

	Current situation	Meet the effluent consent and lower costs (%)	Lowest impact (%)
Ntot removal %	82.7	- 0.8	+ 2.5
Ptot removal %	78	0	0
Power used (kWh)	1.659.549	-5.6 %	- 2.3 %
Waste sludge (ton DS)	452	1.6 %	2.2 %
Carbon source (m^3) (80 g COD/l)	773	- 47 %	- 35 %
Iron dosage (m^3) (FeClSO4 40%)	109	0 %	0 %
Costs*	/	- 15 %	- 10 %
Environmental impact / LCA	2230	- 3 %	- 7.2 %

* Sum of operational costs for electricity consumption, sludge disposal and chemical dosing

6.4.2. Plant 2

The length of the aerated phase in the two intermittent tanks is controlled based on an algorithm which uses the nitrate and the oxygen measurements. No iron nor carbon source was dosed. We verified the effect of different settings for the current aeration controller and we also tested new controllers that use nitrate, oxygen and ammonium measurements. We repeat here what was already discussed in the calibration section: a 50 % energy reduction can be achieved in this plant if a more efficient aeration system is installed (ROI less than 3

years). The energy consumption of teh current system was already evaluated for this new situation so as to not overestimate the benefits of control.

In total roughly 2000 simulations were run and the results of the best scenario using the classic approach and the best one using the lowest impact approach are presented below and compared with the current situation. Both 'best' scenario's used automatic aeration control based on a new control strategy (inputs: oxygen, nitrate and ammonium). Only limited cost savings were obtained using the classic optimisation strategy (meet consent at lowest costs). We attribute this to the fact that there was already a controller in place. The new aeration algorithm which also relies on the ammonium measurements proved to be much more efficient in removing nitrogen and phosphorus.

Looking at the 'lowest impact' scenario it needs to be stated that improvement in effluent quality is impressive taking into account the fact that a controller was already in place.

	Current situation	Meet the effluent consent and lower costs	Lowest impact
Ntot removal %	72	+ 4	+ 8
Ptot removal %	87	+ 4	+ 5
Power used (kWh)	2476800	- 7 %	- 2 %
Waste sludge (ton DS)	2525	0 %	0 %
Carbon source (m^3) (80 g COD/l)	0	0 %	0 %
Iron dosage (m^3) (FeClSO4 40%)	0	0 %	0 %
Costs*	/	- 2 %	0 %
Environmental impact / LCA (mPET)	7560	-13 %	- 22 %

* Sum of operational costs for electricity consumption, sludge disposal and chemical dosing

6.4.3. Plant 3

In plant 3 an advanced aeration controller using nitrate, ammonium and oxygen measurements is already in place. Metal salt dosing is governed by a proportional controller using online effluent phosphate measurements. These two controllers represent the state of the art of online control at Aquafin so we presumed that the margin of improvement for this plant would be limited.

The following controllers/settings/changes were tested

- Number of aeration tanks in operation (currently only 3 out of four aeration tanks are used) and MLSS concentration in the aeration tanks
- Settings of the aeration algorithm

- Settings of the proportional metal salt controller
- A number of alternative controllers that tackle phosphate release in the aeration tanks during peak BOD loading
- Position of the phosphate sensor: effluent versus aeration tank
- Flow limitation to the anaerobic tank during peak flows

It was observed that a longer SRT leads to better nitrogen removal. For phosphorus removal an optimum was found at 23 days. Currently phosphate release in the aeration tanks during peak BOD loading is prevented by always applying an oxygen setpoint of 4 mg/L. An alternative aeration controller that switches from denitrification to aeration when nitrate is below 1.2 mg NO_3-N/L allowed to operate continuously at an oxygen setpoint of 2 mg/L thus saving a lot of energy. However, even with this new controller, the lowest impact was obtained using the 4 mg O_2/L setpoint. Placing the phosphate sensor in the aeration tank led to a significantly lower metal salt consumption.

The classic optimisation strategy leads to a cost reduction and an impact reduction of 7%. Aiming towards the lowest impact results in a cost reduction of 2% and an impact reduction of 11% resulting in a marginal cost of impact reduction with 1% of 1%.

	Current situation	Meet the effluent consent and lower costs	Lowest impact
Ntot removal %	87	+3	+ 4
Ptot removal %	89	0	+ 1
Power used (kWh)	6.821.625	- 16 %	0 %
Waste sludge (ton DS)	3848	- 1 %	- 1 %
Carbon source (m^3) (80 g COD/l)	0	0	0
Iron dosage (m^3) (FeClSO4 40%)	255	- 79 %	- 76 %
Costs*	/	- 7 %	- 2 %
Environmental impact / LCA (mPET)	12418	- 7 %	- 11 %

* Sum of operational costs for electricity consumption, sludge disposal and chemical dosing

6.5. Conclusion

Dynamic, calibrated, Asm2d models were made of three full scale WWTP's in Flanders. We tried to fit real energy consumption, sludge production, chemical demand, effluent nitrogen and phosphate concentrations with modelled ones. The worst fit was obtained for the effluent phosphate concentration. Extra attention for a correct prediction of effluent phosphate concentration is necessary since phosphate in the effluent represents a large environmental impact (up to 50%).

The three plants were already equipped with sensors and an online control system. The goal of this exercise was to see to what extent extra sensors, different settings or different rule-based control algorithms could lead to better results. Two optimisation strategies were compared; meeting the effluent consent and reducing costs versus reducing the environmental impact.

The 'classic' optimisation strategy, meeting the consent and lowering the costs, led to a cost reduction of 2 – 15 % and an impact reduction of 3 – 13 %. The new optimisation strategy, lowering the environmental impact of the plant, resulted in a cost reduction of 0 – 10 % and an impact reduction of 7 – 22 %.

Optimisation towards lower environmental impact leads to a cleaner effluent than the legally imposed quality. It also favours biological over chemical phosphorus removal. Effluent ammonia concentrations are generally lower using the new optimisation goal. All in all we were very pleased with the balanced optimisation that results from this new lowest environmental impact approach. We also think this new goal is very compatible with the way operators tend to manually control the plants. When applied on a larger scale the 'cost per mPET reduction' could be compared in different plants allowing better prioritisation of investments.

It is advisable to standardize the impacts of inputs and outputs of a WWTP in a European context. We should decide on which impact categories to include and how to weigh these impact categories. Moreover central guidance is necessary with regards to the database that should be used to determine the environmental impact of inputs and outputs.

It has already been proven numerous times that online control reduces operational costs and increases treatment efficiency. This report demonstrates that a plant that is already (partly) controlled online can perform even better if the correct controller settings are applied. These correct settings vary from plant to plant even when layouts are similar since every plant has its own characteristic influent composition. Custom made controllers are necessary to achieve the best performance.

7. Evaluation of fault-detection strategies performance in wastewater treatment processes

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This chapter has been published as:

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7.1. Introduction

The use of on-line sensors for use in control and automation for optimized operation of Wastewater Treatment Plants (WWTPs) has gained popularity in recent years. As a result, large quantities of data are provided, which makes manual expert-based quality evaluation of these data impossible. Therefore, the development of strategies and methods that allow detecting faults (process monitoring) and identifying their root cause (fault diagnosis) are urgently needed. These methods can be used to improve data quality and to account for the effects of faults on active controls and overall plant performance (fault-tolerant control).

Fault detection is not a new topic in research. Today, many methods are available and applied in diverse fields (Venkatasubramanian *et al.*, 2003a,b,c). In the field of wastewater treatment process-history based methods have been investigated most frequently. For instance, fuzzy logic is used in Genovesi *et al.* (1999) for monitoring anaerobic digestion. Applied Statistical Process Control (SPC) methods range from univariate methods like control charts (Schraa *et al.*, 2006) to multivariate methods, e.g. based on Principal Component Analysis (PCA) (Rosen and Lennox, 2001; Lee and Vanrolleghem, 2003; Yoo *et al.,* 2006; Villez *et al.,* 2008). Adaptive methods, in which some of its parameters can change over time, have also been proposed to account for changing process behaviour (Lennox and Rosen, 2002; Lee *et al.,* 2005; Aguado and Rosen, 2008).

Many other methods remain untested though and only minimal practical guidelines exist for the selection and application of methods. In order to objectively compare available methods, a simulation platform (BSM1_LT) has been developed (Rosen *et al.*, 2004), which is based on the work of the International Water Association (IWA) Task Group on Benchmarking Control Strategies (BSM, 2009) and includes models that describe typical faults in WWTPs. One task that remains unaddressed within the BSM1_LT platform is the development of objective criteria for comparison of monitoring algorithms.

Therefore, the goal of this paper is to present an objective index for monitoring performance that allows screening different methods. The usefulness of the index is illustrated by the evaluation of different fault detection methods found in literature. Effects of different control system configurations on the monitoring performance of these methods are evaluated as well.

7.2. Materials and methods

7.2.1. Simulated system

The simulation platform used is the BSM1_LT (Rosen *et al.*, 2004). This platform includes model, process configuration (pre-denitrification plant with 5 activated sludge tanks in series, 2 anoxic and 3 aerobic), control systems, benchmarking procedures and evaluation criteria (for process and controller performance). It comprises a one year evaluation period and includes temperature dependence of kinetic parameters. Given the focus on sensors and actuators, realistic models for both sensors and actuators are included (Rieger *et al.*, 2003; Rieger *et al.*, 2006), as well as descriptions for equipment and process faults (Rosen *et al.*, 2008). Thus, emulating often-encountered problems in real WWTP data, the proposed monitoring strategies and algorithms can be benchmarked within a realistic environment.

The simulation protocol for BSM1_LT is as follows: First, the model is run to steady state for 200 days using a constant influent, without any fault. Afterwards, dynamic simulation is conducted using dynamic influent data with sensor and fault models active. The dynamic influent file includes influent flows and concentrations for a period of 609 days at 15 minute interval. The first 245 days are used for training the monitoring methods. From day 245 to 609 (i.e. 364 days or one year) the validation period is run and used to evaluate the monitoring algorithm performance.

7.2.2. Control strategies

The system is simulated for four scenarios with different degrees of control complexity:
Case 1: The measured variable is the Dissolved Oxygen (DO) (modeled as the class A sensor defined in Rieger *et al.*, 2003) in the 2nd aerobic reactor and the manipulated variable is the airflow supplied to the three aerobic reactors. Faults in the DO sensor are included in this case.
Case 2: A cascade control is introduced by means of an ammonia sensor (modeled as class B0) in the 3rd aerobic reactor and an On-Off controller that manipulates the setpoint of the DO controller (0 or 2mg·L^{-1}). Different situations are considered in relation to the faults introduced:

Case 2.1. No faults are considered neither on the DO nor on the ammonia sensor.
Case 2.2. Faults are only considered on the DO sensor.
Case 2.3. Faults are considered on both the DO and the ammonia sensors.

7.2.3. Fault scenarios

The illustrative example focuses on sensor faults and not on process faults. The sensor models from Rieger *et al.* (2003) were extended to include seven sensor states as defined in Rosen *et al.* (2008): 1) fully functional, 2) drift, 3) shift, 4) fixed value, 5) complete failure, 6) wrong gain and 7) calibration. The occurrence of the faults is modeled using the Markov Chains method. This model is run separately for each sensor and the outcomes are stored in files that are used as input for the simulations (Figure 33 top presents the sequence used for the DO sensor). For this study the sensors used belong to the class “bad”, which means that on average, the failure rates are once every two weeks. In the case of the DO sensor it was fully functional for 66.5% of the total time, while the remainder was split into drift problems (15%), shift (6.5%), fixed value (2.4%), complete failure (4.6%), wrong gain (4.5%) and calibration (0.5%).

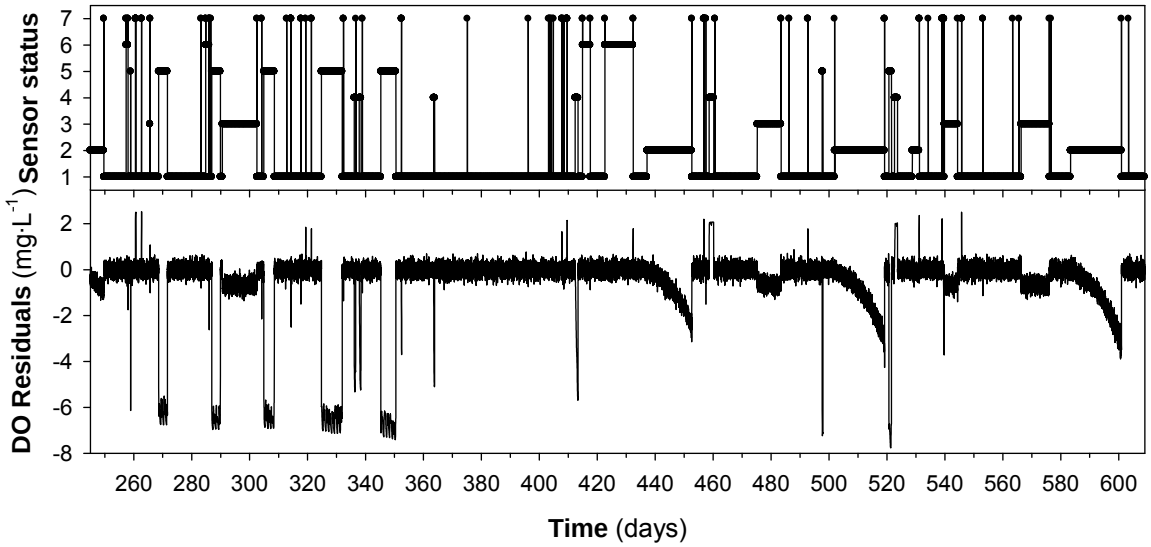

Figure 33. DO sensor status and DO residuals (faulty value minus true value) during evaluation period

The phenomenology of the faults was modeled as described in Rosen *et al.* (2008) but including some modifications. In Figure 34 the structure of the sensor model including faults is presented.

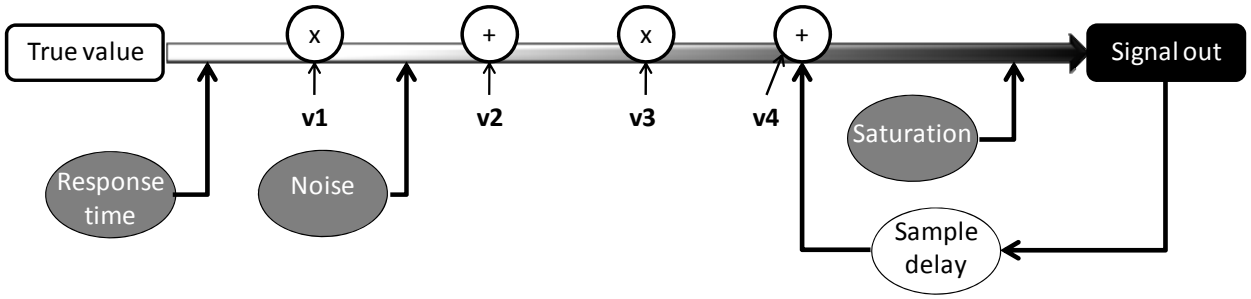

Figure 34. A sensor model including faults (class A)

It can be seen that each fault is described by a four element vector. Each of these elements adds or multiplies the true value by the vector elements (v1, v2, v3 and v4) presented in Table 19. Parameters used were: fg = 2.0 (doubling of the slope of calibration curve), fr = 0.25·(7 days)$^{-1}$ (drift speed), and c0 (the calibration point) = 2 mg·L^{-1} for the DO sensor and 10 mg·L^{-1} for the ammonia sensor. The residuals (difference between the output from the model and the true value) from the DO sensor are plotted in Figure 33 (bottom).

Table 19. Vectors used to describe faults phenomenology

Sensor status	v1	v2	v3	v4
1. Fully functional	1	0	1	0
2. Excessive drift	1-(t-t0)*fr	0	1	0
3. Shift	fb	0	1	0
4. Fixed value	0	0	0	Previous (signal out)
5. Complete failure	0	0	1	0
6. Wrong gain	fg	(1-fg)*c0	1	0
7. Calibration	0	0	0	Previous (signal out)

7.2.4. Fault-detection methods

Different fault-detection methods were tested and compared (univariate methods –control charts, and multivariate methods – PCA and unfolded PCA). In order to ensure objective comparison the methods were calibrated and tuned with the same "fault-free" data set. The theoretical confidence interval corresponding to α=0.27% was used for all methods, corresponding to the 3-σ two-sided confidence interval for normally distributed data.

Univariate Control charts. A control chart is a graphical display of a quality statistic that is measured or computed from a univariate time series. A center line represents the average value and boundaries are set to the quality statistic (normally an Upper Control Limit and a Lower Control Limit). The Shewhart control chart (method A from now on), the Exponentially Weighted Moving Average (EWMA) control chart (B) and the Shewhart control chart on the residuals of EWMA (C and D) were implemented as defined in Montgomery (2009).

Principal Component Analysis (PCA). PCA is a technique that involves a mathematical procedure that transforms a number of possibly correlated variables into a smaller number of uncorrelated variables called principal components (PCs) (Johnson and Wichern, 2002). The first PCA-based model used in this study is standard PCA, in which a multivariate observation consists on a set of measurements taken at a particular time instant. The following measurements were included: DO, ammonia, airflow, DO setpoint and temperature in the 5th reactor and nitrate in the 2nd reactor. The dimension was reduced to 1 or 2 PCs (the dimensionality which was giving better monitoring performance was selected).

The second PCA-based model is the so called Unfolding PCA model (UPCA, Aguado *et al.*, 2007), formerly known as Multiway PCA (Nomikos and MacGregor, 1994). In this case, by means of so-called unfolding, a single multivariate observation consists of the data obtained in one day (e.g. Aguado *et al.*, 2007). This means that a 576-variate data set is analyzed (6 variables x 96 measurements in a day). UPCA-models were considered with number of PCs ranging from 1 to 90. Statistical confidence limits for the corresponding Hotelling's T^2 and Q statistics are calculated as in Nomikos and McGregor (1994).

7.2.5. Monitoring Evaluation index

A monitoring evaluation index is proposed to compare the performance of the evaluated monitoring techniques. This index accounts for a) accuracy, i.e. the rate by which observations are correctly assigned to the normal or abnormal class, b) detection speed and adaptation and c) number of detected fault events. A point awarding system is used as a measure for accuracy. From the true fault sequence and for each fault class (c, c=1: normal, c=2: abnormal), an exponentially decreasing function is computed, making the maximum of 100 points available at start of a fault event (t_{start}) and exponentially decreasing the number of points towards the end of the fault event (t_{end}) by means of equation 1. The points available at a given time are awarded only when the monitoring technique classifies the observations to the right class (normal or abnormal).

$$\mathbf{W}(t,c) = 100 \cdot \exp(-(t-t_{start} / t_{end}-t_{start})) \quad t_{start} \le t \le t_{end} \qquad (1)$$

The true sequence of faults (input for the simulation) is compared to the sequence obtained from the monitoring methods and the following ratios (r_i) are calculated:

- $\boldsymbol{r_{1,c}}$: ratio between the *points* obtained and the total available points of that class.
- $\boldsymbol{r_{2,c}}$: ratio between the number of *observations* correctly assigned to a given class to the total number of *observations* that truly belong to that class.
- $\boldsymbol{r_{3,c}}$: class-specific misclassification rate, i.e. the ratio between the number of *observations* in a given class that are misclassified and the number of *observations* in that class.

- $r_{4,c}$: ratio between the number of *fault events* correctly assigned to a given class to the total number of *fault events* that truly belong to that class.

A global index (G) can be calculated from the previous ratios as shown in equation 2.

$$G = \alpha\left(\sum_{c=1}^{C} r_{1,c} - \sum_{c=1}^{C} r_{3,c}\right) + \beta\frac{1}{C-1}\sum_{c=2}^{C}\left(\frac{r_{1,c}}{r_{2,c}}\right) + \gamma\sum_{c=2}^{C} r_{4,c} \qquad (2)$$

where α, β, γ are weighting factors, C is the number of defined classes (c=1 represents the "normal behaviour" category, c=2 to c=C correspond to each type of fault). The first term represents accuracy, the second term accounts for speed and adaptation and the last term indicates the number of faults detected. Having limited information regarding benefits and costs associated with these terms, α, β, γ were arbitrarily set to the same value of 1/3.

7.3. Results

7.3.1. Process and controller performance

The impact of the faults on the process performance was evaluated by comparing the results from the scenario without faults (2.1) to the scenarios including faults (2.2 and 2.3) (see Table 20). When including faults in the DO and ammonia sensors around 32 % more aeration energy was used compared to the non-faulty scenario. Effluent quality remained largely unaffected (similar values for E.Q.) and nitrification efficiency was even increased thanks to the increased aeration in case 2.2.

Table 20. Process performance results (indices taken from BSM, 2009)

Variable	Unit	Case 2.1	Case 2.2	Case 2.3
Effluent quality (E.Q.)	kg poll.units·day^{-1}	9840	9662	9819
Average aeration energy	kWh·day^{-1}	2583	3410	3452

7.3.2. Monitoring performance

The results obtained from the monitoring exercises were fed into the evaluation system without first passing through a fault diagnosis system. Therefore, the six faulty observations were considered to belong to the same unique class of faulty observations (class 2). The fault-detection methods described above were tested to monitor the DO sensor for the different cases studied (case 1, case 2.2 and case 2.3).

The performance of the algorithms is assessed by the global index values and the individual contributions of the terms included in the index (accuracy, detection speed and events detected) (see Figure 35). The maximum G value that can be obtained from an optimal monitoring algorithm is 1.33 (1/3·2+1/3·1+1/3·1). It can be seen that all methods were far from performing optimally. For case 1 (only DO control) higher G values were obtained reaching the maximum value of 0.95 with the control chart on the residuals of the EWMA model (method C). This method presented high detection speed (0.79) with very low accuracy (1e-2). When using the same control chart for the airflow rate (D) higher accuracy was obtained (0.12) at the expense of decreased detection speed (0.56). The UPCA (method F) obtained the maximum value for events detected (0.33) but mainly because it was signalling alarm 62% of the time. The PCA (method E) and the static univariate methods (A and B) were more accurate than the others but overall, they performed worse (G values around 0.65).

For cases 2.2 (cascade control with only faults in the DO sensor) and 2.3 (cascade control with faults in the DO and ammonia sensors), lower G values were obtained compared to case 1. The decrease in the G value was significant for methods C and D with a reduction between 50% and 60%. It is worth to mention that for all the methods a decrease of accuracy was observed with increased controller complexity.

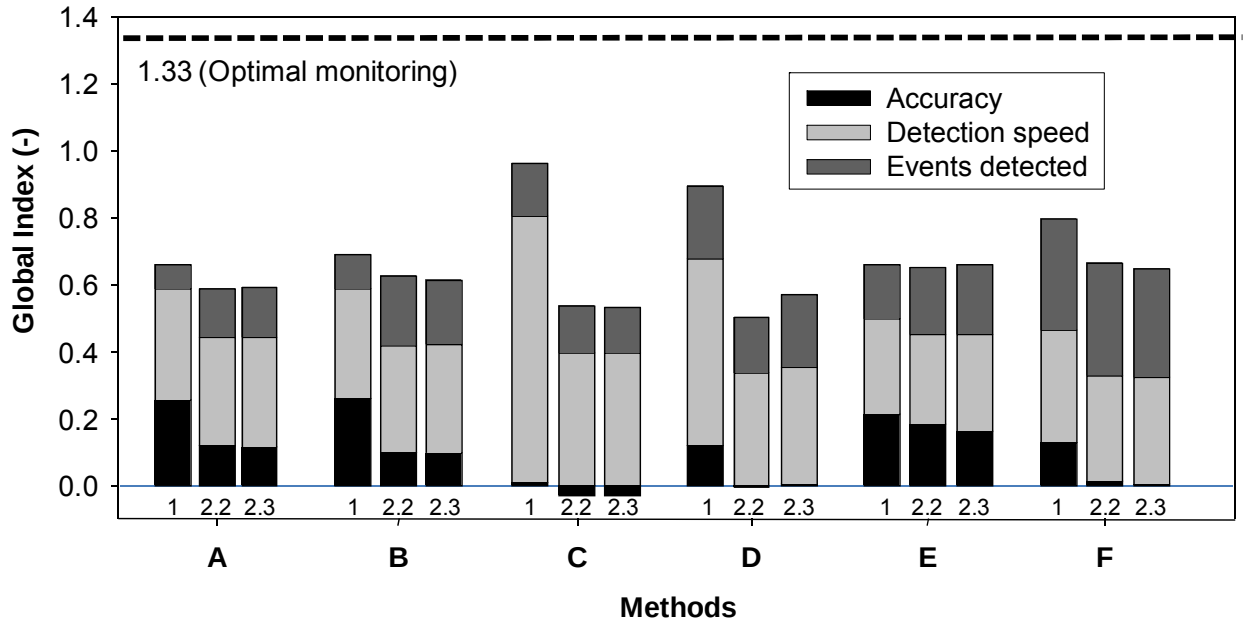

Figure 35. Global index for methods tested and for the three studied cases (1, 2.2 and 2.3). A: Shewhart on DO; B: EWMA on DO; C: Shewhart on EWMA residuals on DO; D: Shewhart on EWMA residuals on Airflow; E: PCA; F: UPCA

7.4. Discussion

Fault-detection methods

A good monitoring method should be accurate (correctness in classification) and fast in detecting the faults (speed). Low performance was observed for all fault-detection methods tested in this study. The reason is that these methods cannot cope with certain real-life aspects of the BSM1_LT platform, such as changing and non-linear process behaviour. Future research should be aimed at the evaluation of methods that account for such characteristics.

When going into more detail, the methods properly detected high changes in the process mean (e.g. caused by complete failure or calibration) but were not able to detect low changes in the process mean (e.g. caused by drift and shift) (results not shown). This has a significant impact on the final results since drift and shift events occur in total 21% of the time while the rest of faults only occur 12% of the time. Better fault-detection performance may be achieved by using in parallel several methods aimed at detecting different fault types.

From the results obtained in this study, it is apparent that the control strategy has considerable effects on the results of the monitoring methods. In case 2 (using a cascade configuration), the controller on the ammonia switches the DO control 'On' and 'Off' inducing sharp fluctuations in the DO variable. The methods tested were not able to account for these fluctuations. Given the On-Off nature of the outer loop control action, the development of separate control charts for the 'On' and 'Off' situation is likely to improve results.

Further work will be conducted to improve the methods used in this work. For instance, investigation can be conducted on the variables selection, especially for multivariate methods

(PCA and UPCA). Moreover, already existing methods will be included in the analysis such as the method presented in Aguado and Rosen (2008), where adaptive and multivariate features are combined. Overall, an effort has to be made to bring methods in the water treatment field that can account for daily, weekly and seasonal patterns in the data and that are able to use redundant information.

Benchmark platform
The use of the BSM1_LT platform is a cost-effective way of investigating the effects of sensor faults on the process performance. In this study, including faults resulted in an increase of the energy used mainly due to the simulated downward drifts. The selection of different fault model parameters might lead to different operating performances.

The BSM1_LT platform allows testing the fault-detection algorithms under standardized realistic environment conditions. The measurements obtained from the simulations are more realistic what allows testing sophisticated methods that can be brought into practice.

Another important contribution of the BSM1_LT is the monitoring evaluation index that proved to be a valid tool to pre-screen methods. However, the index still needs further investigation to improve its sensitivity and to account for new properties of methods that will be implemented/developed in the future (e.g. adaptive methods). Some work has to be further developed on the weighting. The α, β and γ values were arbitrarily chosen for the applied global index. These parameters can highlight different aspects of monitoring performance such as accuracy, the misclassification rate and speed. It is to be expected that the relative performance of monitoring methods is sensitive to the choice of values made. For practical relevance, these parameters should reflect the benefits and costs associated with implementing monitoring strategies. Another issue is the normalization of the different parts of the index that might be necessary to account for faults that have major impact on the performance of the system or for different fault type occurrences. Better knowledge on the methods and further applications will allow better tuning this index.

As a last note, one may consider that in practice an alarm in an on-line application eventually leads to a control action aimed at compensating the problem, either by operator-based or automatic adjustment of plant operation (i.e. fault tolerant control). In both cases, the monitoring system is expected to lead to modified plant performance. Economic evaluation of the resulting closed loop plant performance may then be easier and more relevant to practice than the evaluation of the monitoring system on its own.

7.5. Conclusions

The first case study using BSM1_LT for evaluating monitoring performance in wastewater treatment systems (focusing on sensor faults) has been presented. An evaluation index for monitoring performance has been developed as a combined effort of the IWA Task Group on Benchmarking Control Strategies and experts on monitoring methods. The results obtained have proved that the index presented is a valid tool to pre-screen fault-detection methods and to pinpoint their limitations.

From the results obtained it can be concluded that all methods tested present poor fault-detection performance. Changing process behaviour and accounting for the multivariate nature of the data are features that should be combined in the methods. When bringing monitoring into practice, one has to take into account that control complexity affects monitoring performance (e.g. decreasing accuracy). Further research will be conducted to investigate the effect of variable selection and redundancy on the methods performance and the assessment of costs and benefits of positive alarm detection, false alarms, speed and accuracy.

The rather elaborate set of considerations with respect to this study is expected to encourage engineers and researchers to develop and test their methods using the BSM1_LT platform and the evaluation criteria.

8. Zeolite addition for improvement of WWTP performance

8.1. Introduction to Zeolite addition and goal of the experiments

In plants of insufficient capacity experiencing seasonal loads it is difficult to achieve consistent nitrification because the necessary sludge age and mixed liquor suspended solids cannot be maintained within limits that provide for good process performance and don't exceed the safe solids loading to the final clarifier.

Zeolites have some capacity for adsorbing ammonia and their filtering abilities offer a versatile and environmentally friendly option to capture contaminants found in water systems.

Zeolites have a natural negative charge which gives them the capacity to adsorb cations and some organic contaminants and undesirable odours. The density and the characteristic of zeolite to be incorporated into the sludge allow further significant improvement of the sludge settleability.

With the contribution of the NEPTUNE project, full-scale trials were conducted in the period 2007 – 2009. The experiments consisted of addition of polymer-modified zeolite (SZEDIMENTIN MW) into the biological step of seasonally loaded wastewater treatment plants in Bulgaria. A parallel process without addition of zeolite was run to allow monitoring of the process improvements.

The special effect of SZEDIMENTIN MW is that it causes rapid settling of sludge in the sedimentation tank, the supernatant after sludge sedimentation becomes of a very good quality and bulking of sludge is prevented. The trials aimed to improve the reduction of pollutants and the effluent quality, and to increase the plant capacity without construction of additional structures.

8.2. Selection of Zeolite and dose

8.2.1. Selection of type of zeolite

Upon investigation of the available zeolites, two types of zeolite were selected for laboratory trials:

- Natural ZEOLITE-BG (as mined), and
- Modified SZEDIMENTIN – MW (heat-activated, treated with a polyelectrolyte, containing lyophilised bacteria).

Sedimentation of 30 minutes was conducted in a 1000 ml cylinder of 0.333 m height, to determine the effect of the zeolites on the sludge settleability.

The concentration of the activated sludge was 4200 mg/l and the sludge volume index (SVI) was 114 cm^3/g.

It was monitored that at low concentrations up to 20 mg/l, the SVI was similar for both zeolites. At concentrations of 30 mg/l and higher, SZEDIMENTIN – MW produced a significantly reduced sludge volume (Figure 36). Also, SZEDIMENTIN-MW resulted in visually reduced turbidity. Therefore, the modified SZEDIMENTIN – MW was selected for all further experiments. The composition of the selected zeolite is shown in Table 21.

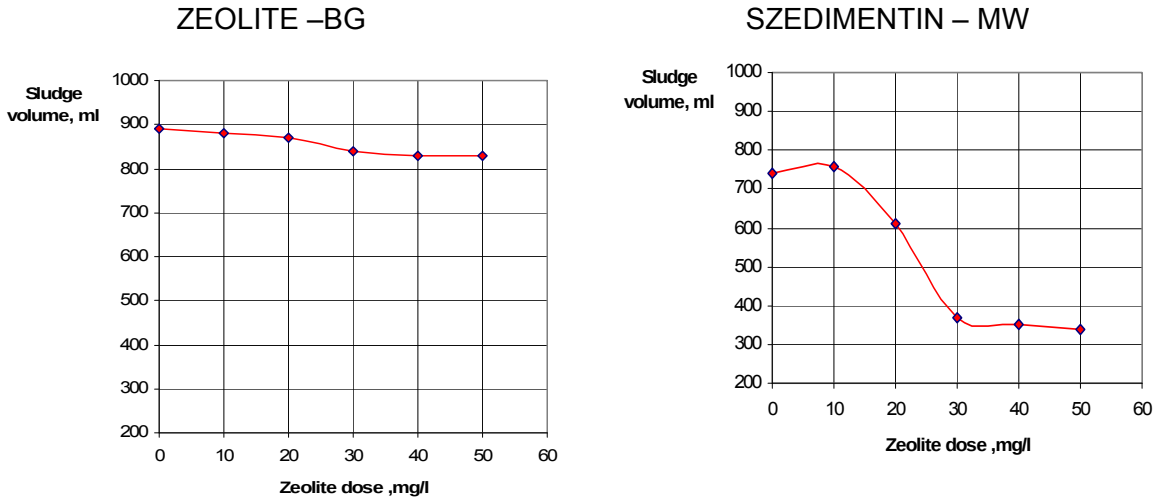

Figure 36. Sludge volume comparison for Zeolite- BG and SZEDIMENTIN-MW after 30 min settling

Table 21. Composition of SZEDIMENTIN-MW

Mineralogical composition	%	Chemical composition	%
Clinoptilolite	25 - 35	SiO_2	74,74
Montmorillonite	25 - 35	Al_2O_3	12,68
Surface-aktiver volcanic glass	25 - 30	Fe_2O_3	1,26
Feldspars	3 - 8	FeO	0,05
Kaolinite	10 - 15	CaO	0,95
Oxides of iron and manganese	3	MgO	0,88
		Na_2O	0,36
		K_2O	1,83
		TiO_2	0,10
		Ignition loss	7,15

8.3. Selection of dose

Following the selection of zeolite, the dose at which maximum efficiency is achieved was investigated. The dose was increased up to 50 mg/l in increments of 10 mg/l and two parameters were evaluated, as described below.

8.3.1. Effect of increasing dose of SZEDIMENTIN - MW on the sludge volume

At low concentrations of up to 20 mg/l the addition of zeolite had no significant effect. A dose of 30 mg/l reduces the sludge volume below 500ml in 15 min. However, a further increase of the dose has less effect (Figure 37).

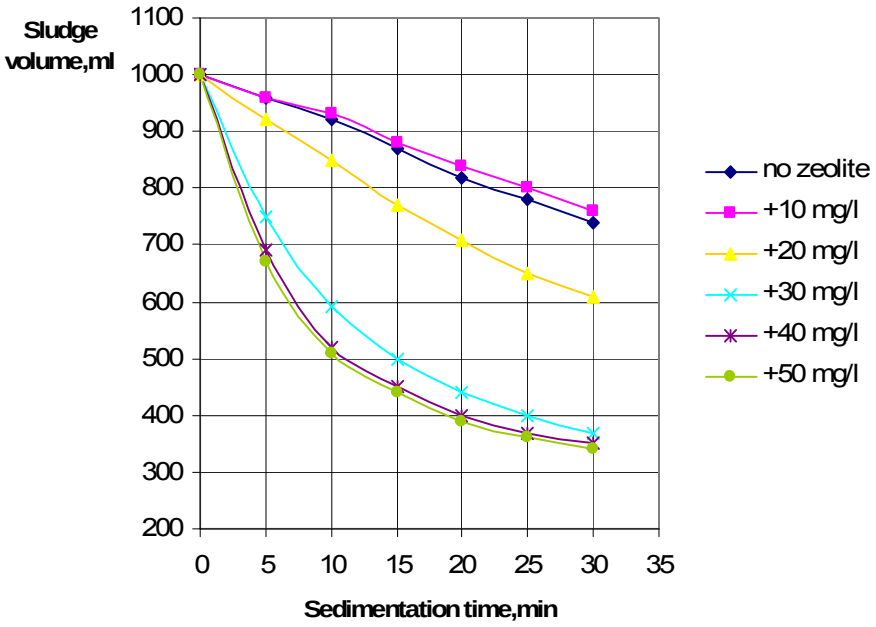

Figure 37. Effect of increasing the dose of SZEDIMENTIN - MW on the sludge volume

8.3.2. Effect of increasing dose of SZEDIMENTIN - MW on the sludge settling velocity

At low doses or no zeolite, the settling velocity was steady and low. Concentrations above 30 mg/l resulted in very rapid settling in the first 15 min, after which the speed dropped to more steady levels (Figure 36).

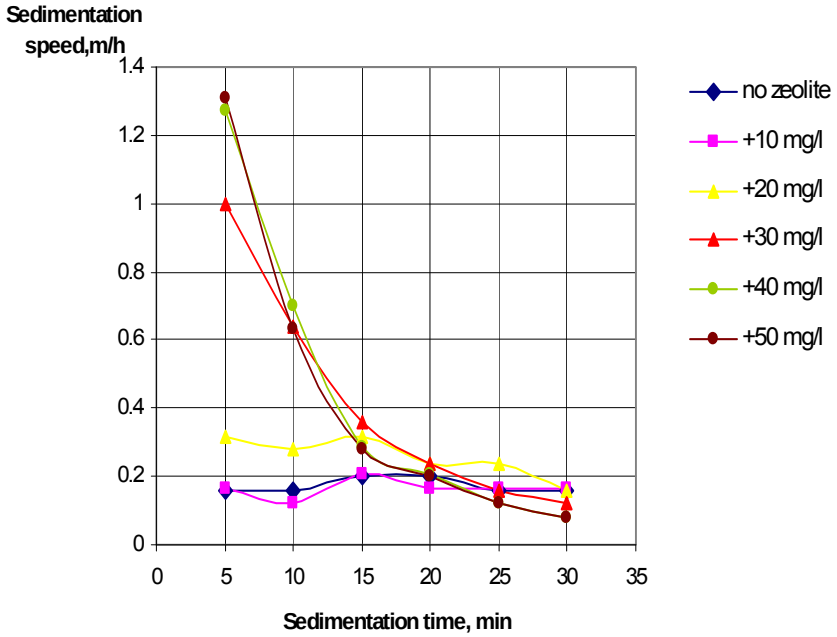

Figure 38. Effect of increasing the dose of SZEDIMENTIN - MW on the sludge settling velocity

8.4. Dosing

8.4.1. Equipment

An automatic tenzometric dosing system was used to dose the zeolite to the activated sludge bioreactors. Figure 39 shows the schematic of the system and its key elements:

- Storage container for zeolite.
- Water inlet slots to allow for suspension of the zeolite; the water serves as carrier and provides for mixing with the wastewater. This part has to remain submerged below water level.
- Suspension outlet slots to carry away the mixture of zeolite and water and distribute it evenly to the wastewater.

Figure 40 shows the installation of the system in the bioreactor during the trials.

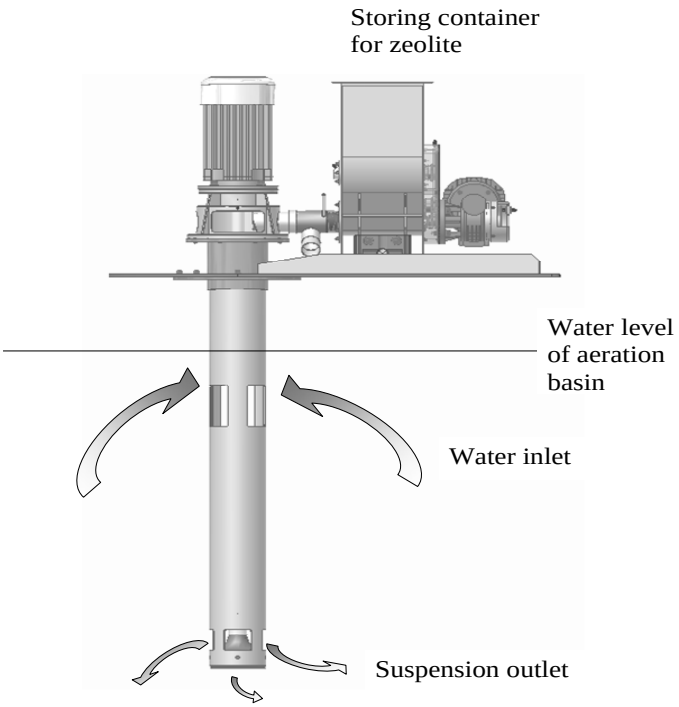

Figure 39. Zeolite dosing by automatic tenzometric dosing system

Figure 40. Automatic tenzometric dosing system installed at Zlatni Pqsyci WWTP

8.5. Dosing points

8.5.1. Location

To determine the application point in the bioreactor that would result in better effluent quality and optimal utilization of the zeolite, experiments were conducted in three parallel streams of the bioreactor. Zeolite was dosed at 50 kg/d to each stream in various locations. Consequently the water was settled in a secondary settler and sludge was returned to the process.

In the first stream (Stream I) no zeolite was added and the effluent water quality was used as a basis for comparison with the quality of the effluent from the other two streams. In the second stream (Stream II) zeolite was added at the end of the bioreactor before secondary settling, and in the third stream (Stream III) dosing appeared at the start of the biological treatment (Figure 41).

8.5.1. Sampling and analyses of the results

24-h composite samples were collected weekly at the outlet of the secondary settler and analyzed for BOD_5, COD, $N\text{-}NH_4^+$, Suspended solids, Total Nitrogen and Total Phosphorus, and the Sludge Index was measured.

The measured concentrations were used to calculate the treatment efficiencies for Stream II and III. The averaged concentrations and treatment efficiencies for some parameters are presented in Figure 42.

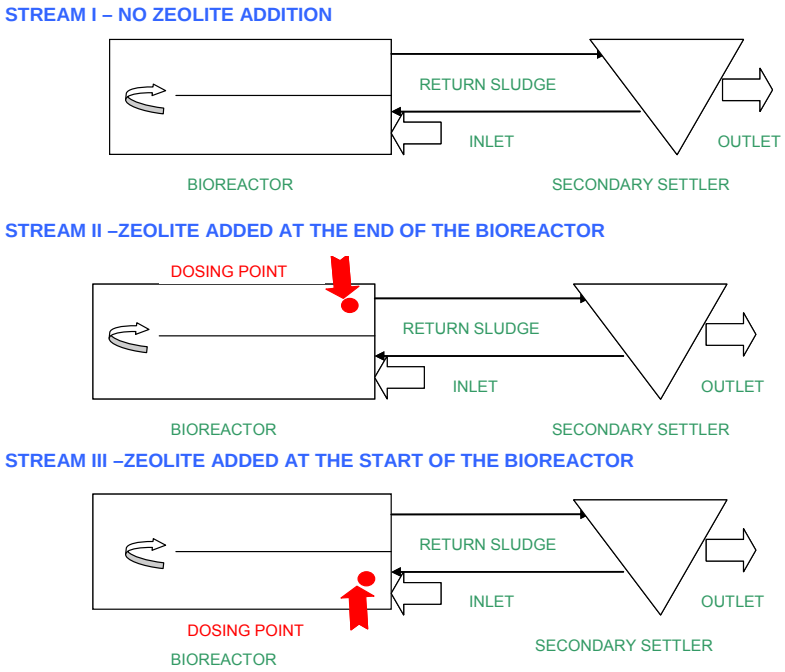

Figure 41. Flow diagram of the secondary treatment and zeolite dosing points

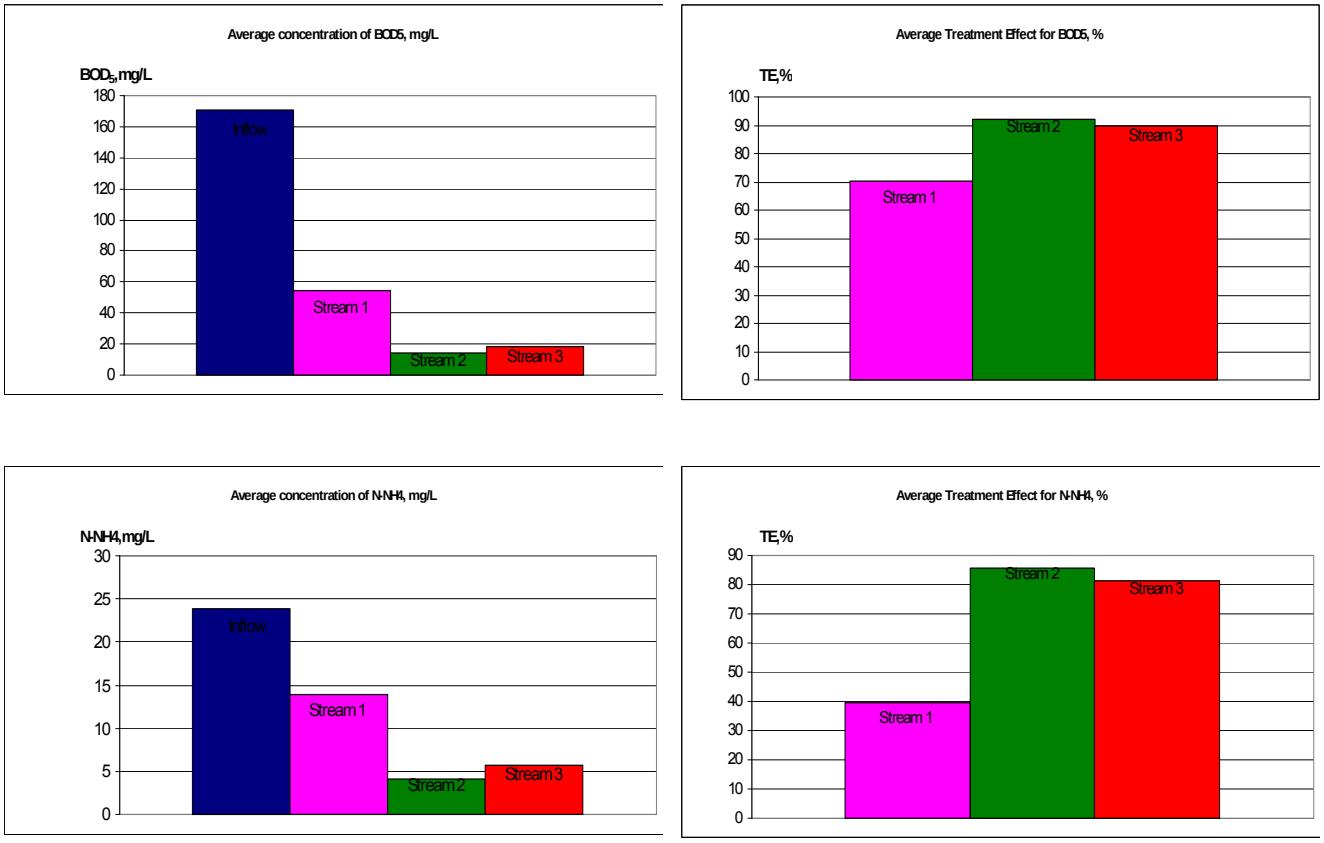

Figure 42. Averaged concentrations (left) and Treatment efficiencies (right) for Stream II and III for BOD_5 and ammonium nitrogen

For BOD, while dosing at the end of the stream provided slightly better results, the location of the dosing point did not influence the final treatment efficiency significantly.

For ammonium nitrogen, better results were achieved when dosing at the end of the reactor, similar to BOD. The treatment efficiency (%) for ammonium nitrogen stayed constantly high and very good results were achieved.

Little treatment effect was monitored for total nitrogen - 22% reduction in concentration, if added at the end, and limited effect of 7%, if added at the beginning of the process. For total phosphorus, similar reduction in concentration was registered for both dosing locations, averaging at 25%.

For suspended solids, an adverse effect occurred when zeolite was added at the end of the process or no effect was found when it was added at the start of the process. Most probable reason for that is the observed sludge bulking - irregularity in overloaded real-life systems - that masked the effect of zeolite addition.

8.5.2. Conclusion and selection of dosing point

In summary, better control over bulking of sludge in the secondary settlers at peak hydraulic loads was achieved when dosing occurred at the end of the bioreactor, immediately before the settler.

A constant concentration of activated sludge was maintained at minimum hydraulic and organic loads in winter. This was a notable improvement from previous low load periods, when the sludge concentration in the bioreactor was reduced to zero for a few months.

When the trial was carried out, a preliminary dosing period to improve sludge settleability was not conducted. Therefore, the sludge bulking observed is the most probable reason for the lack of reduction of suspended solids concentration. However, the rest of the parameters showed better values for the dosing location at the end of the bioreactor and . therefore, this point was selected for all further experiments.

8.6. Full-scale experiments

Experiments were conducted in summer 2008 at the Zlatni Pqsyci WWTP. While positive results such as improved removal efficiency for BOD, COD, ammonium nitrogen and SVI were achieved, there was the restriction that this plant has a typical seasonal loading, with over-loading in summer for three to four months, and an under-loading for the rest of the year. As a result, the experiments were restricted to the summer season when the water temperature is high (27 degrees C). To be able to continue the experimental work in winter and investigate the treatment efficiency of the zeolite at low water temperature (8 – 10 degrees C), a plant with all-year-round overloading was selected and the winter 2009 experiments were conducted at Dylgopol WWTP. Then, in summer 2009 additional experiments were carried out at Zlatni Pqsyci WWTP, to obtain more data at the same experimental conditions.

The similarities and differences between the three dosing campaigns are outlined below.

8.6.1. Common aspects between the 2008 and 2009 campaigns

Baseline comparison

At both plants, the results from the controlled streams with addition of zeolite were compared against the wastewater quality without addition of zeolite to allow evaluation of the pollutant removal and treatment efficiencies.

Sludge build-up

At both plants, before the start of the sampling and analyses, a 14-day period was allowed to build up the zeolite concentration in the sludge and permit sludge quality transformation.

Sampling details

24-h composite samples were collected from each stream.

Parameters controlled

All samples were analyzed for BOD, COD, $N\text{-}NH_4^+$, Suspended solids, Total Nitrogen and Total Phosphorus, and the Sludge Index was measured.

8.6.2. Zlatni Pqsyci WWTP - 2008

8.6.2.1. Description of the plant

Zlatni Pqsyci WWTP is a conventional activated sludge plant, which provides secondary treatment of an average daily flow of 3000 m^3/d in summer. The plant is located in a touristic region and has a typical seasonal loading, with over-loading in summer and under-loading for the rest of the year. The main plant treatment process consists of coarse screens and grit chamber, and a bioreactor that consists of three parallel streams, each built of a primary sedimentation tank, bioreactor, secondary sedimentation tank and chlorine contact tank. Return activated sludge is fed from the secondary clarifiers to the inlet of the bioreactor. The aeration is fine bubble aeration supplied from blowers. The treated effluent is discharged via a deep-water discharge pipe into the Black Sea.

The process flow diagram of the secondary treatment process was presented in Figure 40.

8.6.2.2. Details on the full-scale 2008 measuring campaign

The full-scale experiments were carried out from 09/06/2008 to 15/09/2008 and had a dual purpose:

- To optimize and determine the most effective dose of zeolite by conducting experiments at various concentrations as follows:

  Stream 3: Dose of 31.10 kg/d (0.005 kg/m^3), to maintain 7.5 % zeolite in the activated sludge of one stream.

  Stream 2: Dose 20.75 kg/d (0.003 kg/m^3), to maintain 5.0 % zeolite in the activated sludge of one stream.

  Stream 1: 0 % concentration of zeolite in the activated sludge. This stream is used for reference.

In preparation of the bioreactor for dosing, two tenzometric dosing systems were installed at the end of the bioreactor in stream two and three. Each system was set to dose the quantities of zeolite discussed above.

The actual zeolite dosing started in May and by the end of the month the effect of zeolite dosing was in place. This allowed two to three weeks for sludge formation in the bioreactor, before sampling for nutrients and micro-pollutants began. Then the dosing continued and 24-hour composite samples were collected daily. All samples were analyzed for nutrient removal.

- The second purpose of the experiments was to conduct sampling and analysis of the wastewater, in cooperation with INCDTIM Romania, for the effect of zeolites on removal of micro-pollutants.

  Siluet B collected samples and INCDTIM performed solid-phase extraction (SPE) on site. The goal was to determine the reduction of micro-pollutants (alkyl phenols/ ethoxyl and flame retardants) in the wastewater. The SPE samples were analyzed in the existing INCDTIM laboratory in Romania.

Two weeks of effluent sampling produced 42 samples in total (2 weeks x 7 days x 3 samples one in each stream).

8.6.3. Dylgopol WWTP - 2009

8.6.3.1. Description of the plant

Dylgopol WWTP is located outside the town of Dylgopol, 70 km west of Varna. The permanent population is 5500 people and the sewerage system is developed only to 50%.

The treatment plant has been in operation since 1982. However, the construction was staged in two lots of equal capacities and only the first stage was built. The current capacity of the plant is 630 m^3/d with the possibility to upgrade it to full design capacity of 1260 m^3/d through construction of stage two.

The treatment process is performed in an Imhoff-tank, followed by an aerated bioreactor and secondary settlers. The plant is designed for organics removal and nitrification, but manages to achieve nitrification only in summer. The process does not include denitrification or phosphorus removal. Characteristic for the current plant operation is a periodic increase in SVI leading to sludge bulking in the secondary settler resulting in effluent quality deterioration.

The Process flow diagram of the Dylgopol WWTP, showing the temporary zeolite application point is shown on Figure 43 and pictures of the plant are included in Figure 44.

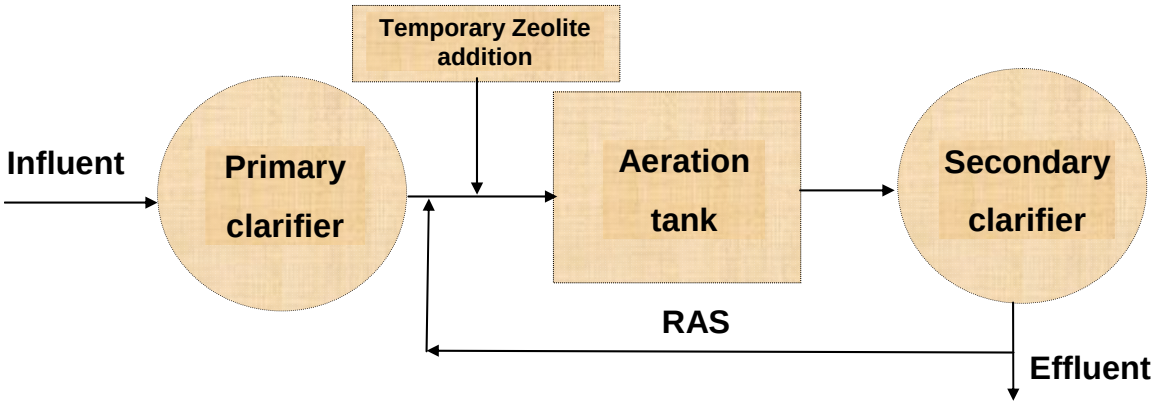

Figure 43. Process flow diagram of Dylgopol WWTP

Figure 44. Dylgopol WWTP

8.6.3.2. Details on 2009 winter measuring campaign

The goal of the experiment was to observe the improvement in the treatment capacity of the plant at low water temperatures in winter (8-10 degree C) as a result of addition of zeolite. Expected improvements were an increase of the MLSS in the bioreactor; reduction of the SVI and, as a result of that, improved sludge settling and nitrification thanks to the increase in sludge age. Overall improved water quality was expected, most pronounced for ammonium nitrogen.

Ten grams zeolite/(m^3 wastewater) were added before the bioreactor at the point of return of activated sludge. The experiments were staged as follows:

- No zeolite addition:

In the period 12 Jan - 25 Jan 09, the plant parameters were measured and monitored in order to build-up a base line data for comparison.

- With zeolite addition:

  In the period 26 Jan - 8 Feb 09, which was equal to 1.5 SRT, zeolite was dosed without process sampling. This aimed to allow sludge build-up and activated sludge quality transformation.

- With zeolite addition: 9 Feb - 16 Mar 09;

  The dosing continued and sampling and analyses were undertaken. The change in plant process parameters was measured and monitored.

8.6.4. Zlatni Pqsyci WWTP – 2009

8.6.4.1. Description of the plant

Detailed description of the plant is presented in Section 8.6.2.1 and the process flow diagram in Figure 41.

8.6.4.2. Details on the full-scale 2009 summer measuring campaign

The full-scale trial was carried out in the period 1-26 June 2009 and had a purpose to collect more data at conditions repeating those in 2008 to build up the database and confidence in the results.

Similarly to 2008, the experiment was carried out in two stages. Zeolite was dosed for two weeks to allow sludge build-up and activated sludge transformation. The process parameters were not measured in that period. Then, the dosing started and continued for two weeks and changes in plant process parameters were measured for that period and two weeks after the dosing have ceased. The effluent quality was compared to that in the parallel streams where zeolite was not dosed.

The effluent was sampled after the secondary settler. Daily, 24-h mixed samples were collected and analyzed for nutrients removal.

The difference with the trial in 2008 was that zeolite was dosed in a single stream of the bioreactor before the secondary settler, at a single dose of 10 g/ m3 wastewater, which based on the results in 2008, was determined as an optimal dose. One tenzometric dosing system was installed at the end of the bioreactor in stream three only.

8.7. Results

8.7.1. Plant performance

8.7.1.1. Efficiencies for Nutrient removal

- **Zlatni Pqsyci WWTP (2008)**

The results from the dosing and sampling campaign in 2008 are shown in the figures below for each key water characterization parameter. They are also compared to the results from the experiment for selection of the dosing point carried out in 2007 (results are marked as Stream II in the following graphs). It is important to note that the 2007 results are shown for general comparison and only average (and not discrete) values can be compared as the dates, number of samples as well as the wastewater quality at the inlet differed. Stream II represents dosing of 50 kg/d zeolite at the end of the bioreactor before the settler.

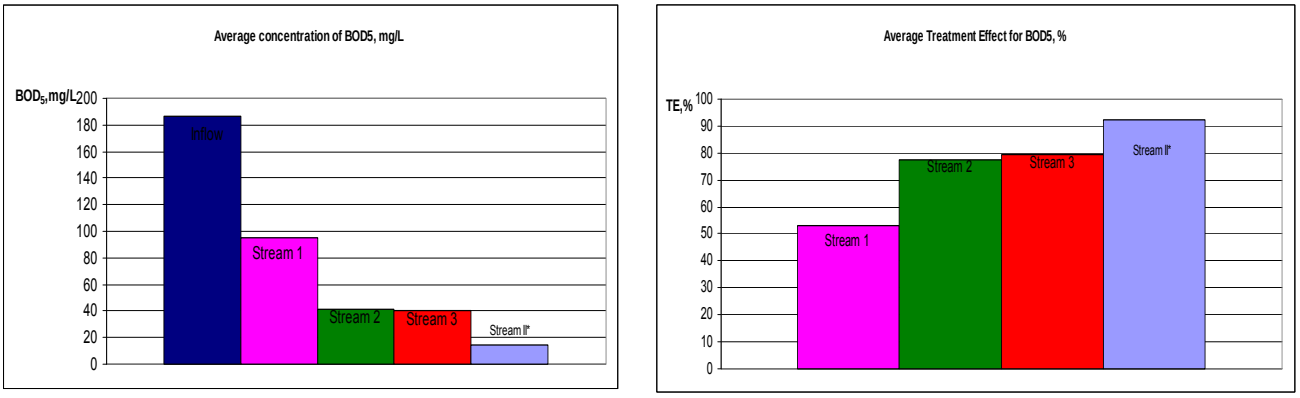

Figure 45. Averaged concentration (left) and averaged treatment efficiency (right) for BOD

As Figure 45 shows, the addition of zeolite had a noticeable effect on the removal of BOD with a reduction of concentration nearly 60% even at the lower dose. At the same time, an increase of the dose had an insignificant effect.

Compared to the previous results of 14 mgBOD/L on average, achieved with a dose of 50 kg/d, the concentrations now were higher, averaging at 41 mg/L for stream 2 and 3.

The treatment effect rose slightly from 77.7 to 79.7%, but remained lower than the treatment effect of 92% at the dose of 50 kg/d of the 2007 experiment.

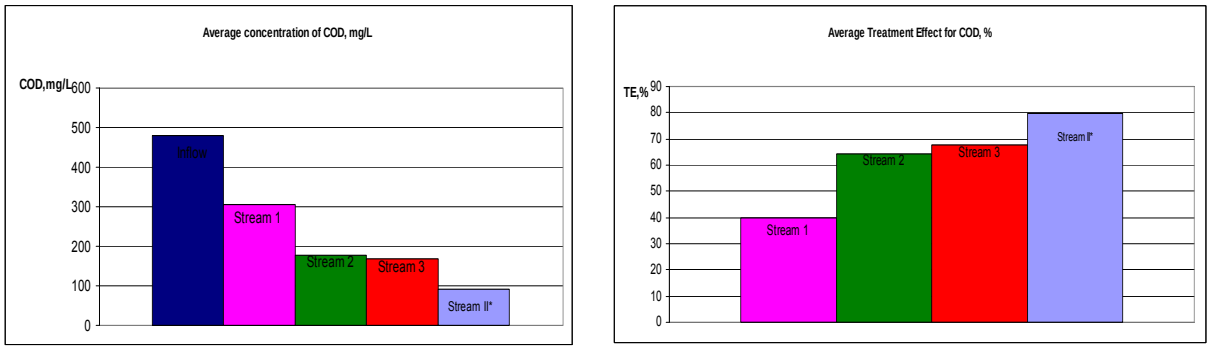

Figure 46. Averaged concentration (left) and averaged treatment efficiency (right) for COD

Figure 46 shows that the average concentration of COD was reduced significantly by 42 and 45 % for the lower and higher dose of zeolite respectively. Similar to the BOD_5, the increase of the dose had no significant effect.

Compared to the 92 mg COD/L on average, achieved with a dose of 50 kg/d in 2007, the concentration now was almost doubled with 176 and 170 mg COD/L for the two doses.

The treatment effect rose slightly from 64.2 to 67.6 %, but remained lower than the treatment effect of 79.5% at dose of 50 kg zeolite/d.

Clearly, for COD removal the most efficient dose was 31 kg/d.

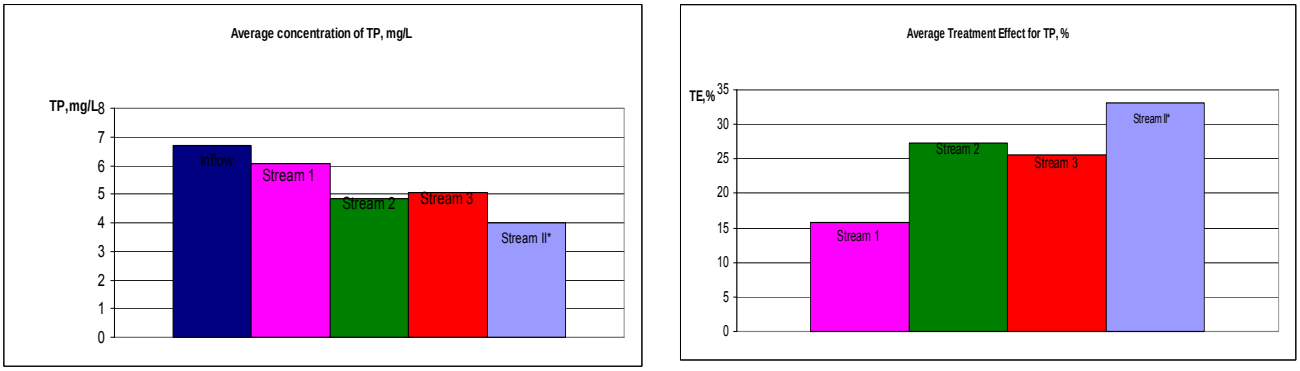

Figure 47. Averaged concentration and averaged treatment efficiency for Total Phosphorus

As shown on Figure 47, the average phosphorus discharge concentration of about 5 mg/L remained slightly higher than the 4 mg/L achieved with 50 kg/d dose in 2007. The percent reduction in concentration was lower at both 21 and 31 kg/d compared to the 50 kg/d dose (20.2 and 16.3 vs 27.4%).

At 31 kg/d, the average treatment effect was 25.6 %, which was lower than the 33% achieved at 50 kg/d.

It can be seen that the efficiency of total phosphorus removal increased by several per cent with increase of the dose to 50 kg zeolite/d, but there was not an improvement for the lower step in concentration from 21 to 31 kg/d.

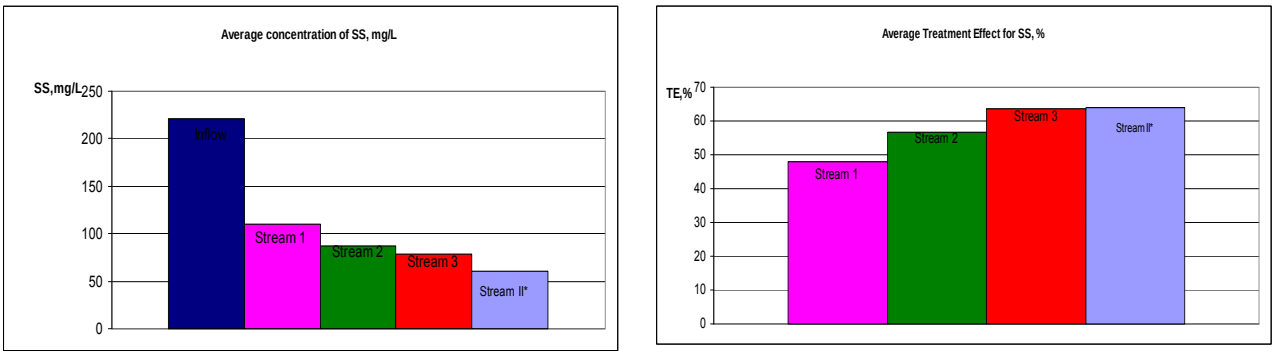

Figure 48. Averaged concentration (left) and averaged treatment efficiency (right) for Suspended Solids

Figure 48 shows that the effect on suspended solids was less pronounced compared to the for BOD5 and COD. The percent removal increased from 20.4 to 28.9% (87.5 and 78.2 mg SS/L) for stream 2 and 3 respectively and exceeded that achieved in the previous experiments with 50 kg/d dose.

While the treatment effect leveled out at 64% for 31 and 50 kg/d zeolite (63.7 and 64.0% respectively), the increase in the treatment efficiency was improved to 33% at 31 kg/d from only 5% at 50 kg/d.

Some of the data showed no improvement following zeolite dosing. This may be due to sludge bulking that masked the treatment effect. However, this is an irregularity typically observed in real-life systems. For stream 1 without addition of zeolite, bulking of sludge was observed resulting in a negative treatment effect.

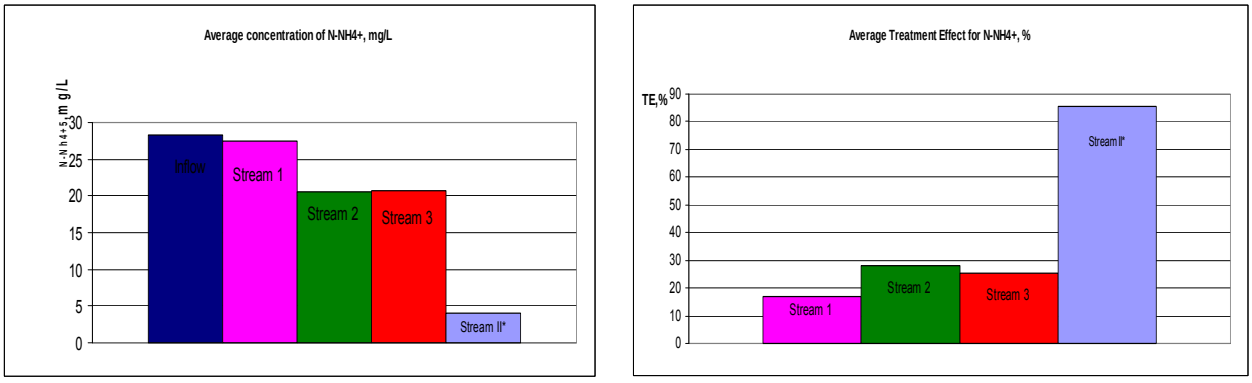

Figure 49. Averaged concentration and averaged treatment efficiency for N-NH4

As seen on Figure 49, the average ammonium concentrations for stream 2 and 3 leveled out at 20.6 mg/L.

The reduction was not significant with an average removal of 25%. This was markedly lower than the 70% reduction in concentration achieved at 50 kg/d.

The increase in treatment efficiency was 66.6 and 50.9% for stream 2 and 3, compared to more than 100% increase with 50 kg/d.

Clearly, the higher dose of 50 kg/d has achieved better removal of ammonium nitrogen.

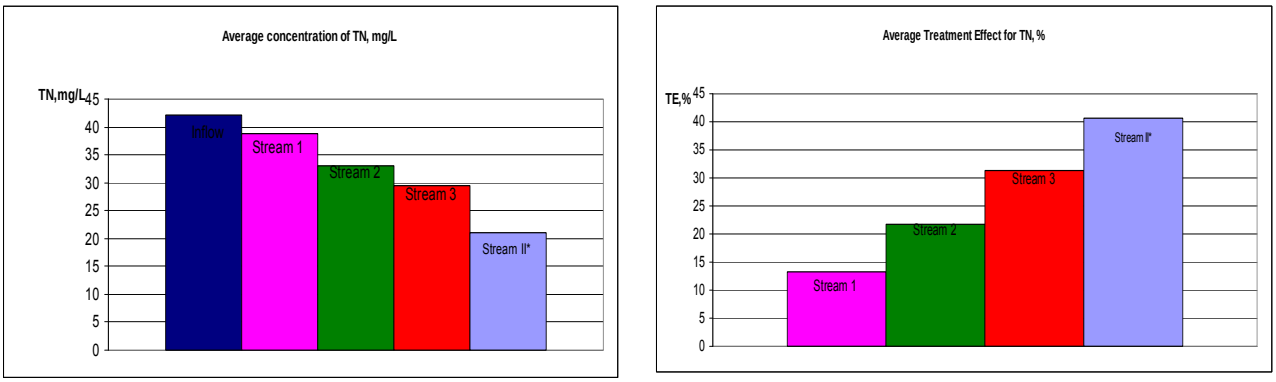

Figure 50. Averaged concentration (left) and averaged treatment efficiency (right) for Total Nitrogen

Similar to suspended solids, Figure 50 shows that the effect of zeolite addition was not well pronounced. The percent of removal was only increased by 15 up to 24.3 % for stream 2 and 3.This was almost identical with the 22.3 % removal at 50 kg/d dose.

The average treatment effect was the lowest at 21.7% for dose 21 kg/d, increasing gradually to 31.3%, for 31 kg/ d and 40.5% at 50 kg/d dose.

This was an indication that better nitrogen removal can be achieved with increase of the dose. However, the increase of efficiency was more proponounced for the set of lower doses where change of 10 kg/d resulted in nearly 10% better efficiency, while further raise of the dose by 19 kg/d, from 31 to 50, boosted the efficiency by only 9%.

As a result of the addtion of zeolite and improved removal of BOD_5, the sludge production increased and for the 31 kg/day dose it was 60%. This was calculated as a ratio of the net increase of sludge production to the sludge production without zeolite, not accounting for the mass of zeolite.

- **Dylgopol WWTP (2009)**

The results from the dosing and sampling campaign in 2009 are summarized in Table 22

Table 22. Results from the dosing and sampling campaign in 2009

No.	Parameters	No zeolite 12 Jan - 25 Jan 09			With zeolite 9 Feb - 16 Mar 09		
		min	average	max	min	average	max
1	Influent, m3/day (averaged for the period)		678			635	
2	MLSS, g/L	1.5	1.6	1.8	3.2	3.3	3.5
3	Temperature in the bioreactor, degree C		10			11	
4	Bioreactor volume, m^3		180			180	
5	Mass of activated sludge, kg	-	288	-	-	594	-
6	BOD5 inlet, mg/L	-	159	-	-	145	-
7	Ammonium nitrogen inlet, mg/L	21	24	27	20	22.5	25
8	Suspended solids inlet, mg/L	-	164	-	-	155	-
9	F/ M ratio, kg BOD5/ kg dry solids.day	-	0.37		-	0.15	-
10	Sludge retention time, days	-	8.5		-	20.5	
11	Sludge volume index, cm^3/g	145	152.5	160	82	86	90
12	BOD5 outlet, mg/L	-	20.6	-	-	9.0	-
13	Ammonium nitrogen outlet, mg/L	12	15	18	2.5	4.9	7.2
14	Suspended solids outlet, mg/L	-	41	-	-	24.8	-

The effluent quality improvement results are depicted in Figure 51. It should be noted that the comparison is only indicative as the inlet quality for period 12 Jan - 25 Jan 09 slightly differed from that for the period 9 Feb - 16 Mar 09.

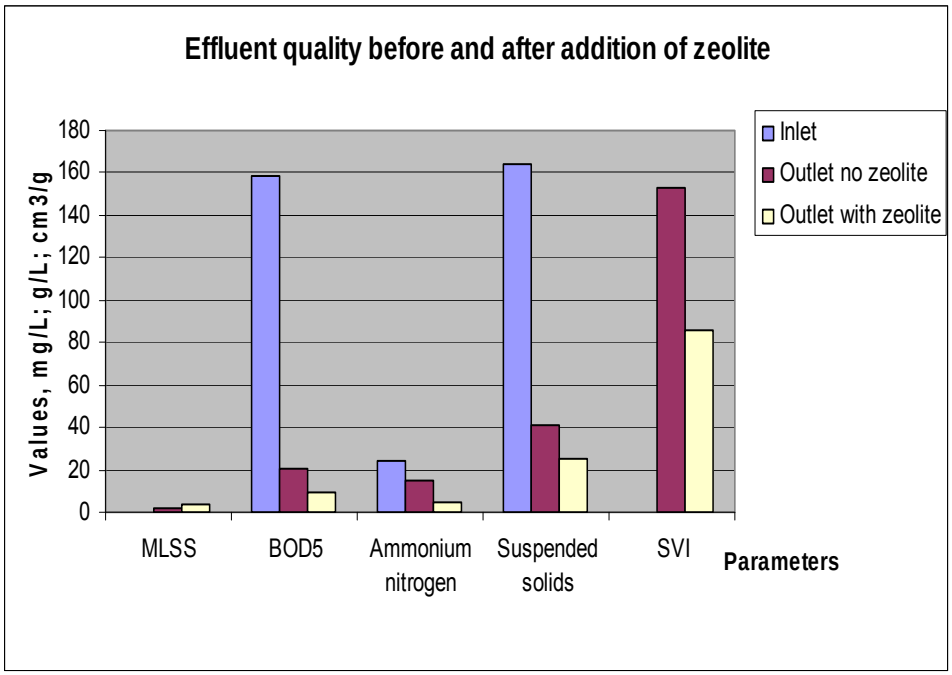

Figure 51. Effluent quality before and after the addition of zeolite

On comparison of the effluent quality before and after the addition of zeolite, the treatment efficiency as a result of the zeolite addition has increased by 108.6% for ammonium nitrogen, by 7.8% for BOD_5 and by 12.0% for Suspended solids, as shown on Figure 52.

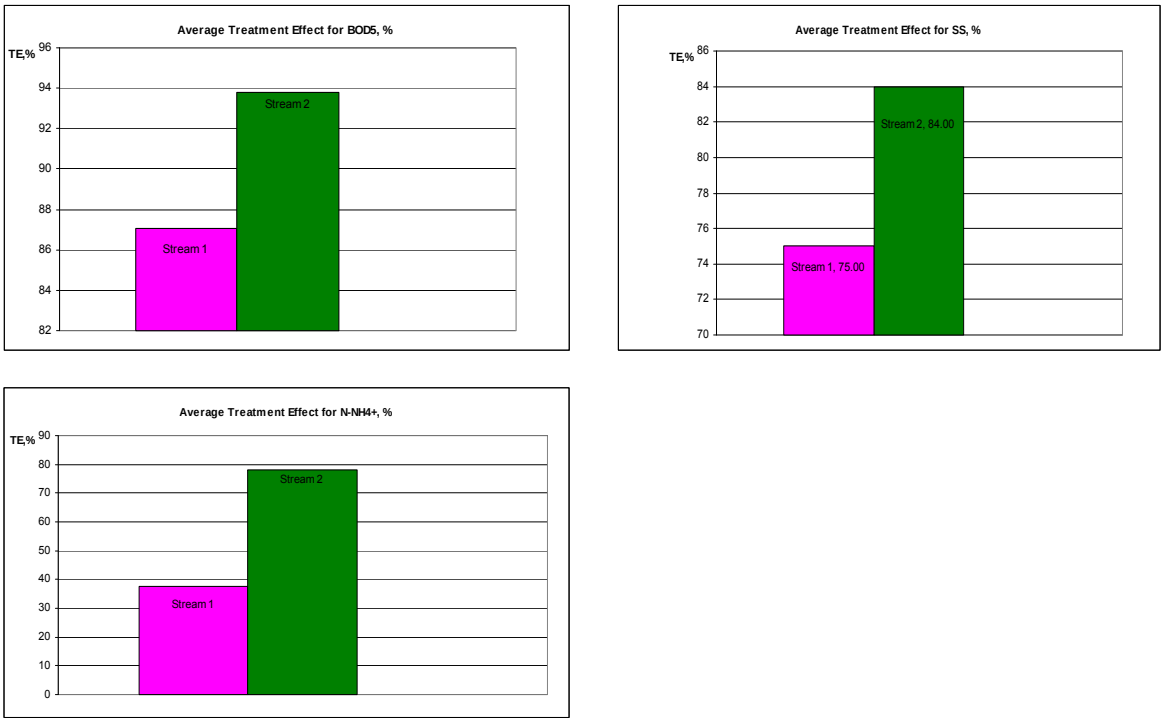

Figure 52. Average treatment efficiencies for BOD_5 (top left), Suspended solids (top right) and Ammonium nitrogen (bottom) before and after addition of zeolite

In summary, the analyses of the results show the following has been achieved:

- Improved settling of the activated sludge in the secondary settler; decrease of the SVI from 150 to 85 cm^3/g on average;

- Increase of the activated sludge concentration in the bioreactor from 1.6 to 3.3 g/L on average without sludge bulking;

- Increase of the sludge age from 8.5 to 20.5 days, and as a result – improved nitrification even at the low temperature of 8-10 degree C;

As a result of the addtion of zeolite and improved removal of BOD_5, the sludge production increased by 8.4%. This was calculated as a ratio of the net increase of sludge production to the sludge production without zeolite, not accounting for the mass of zeolite added. This value was significantly lower than the increase of sludge production for Zlatni Pqsyci WWTP because of the lower incoming flows and inlet concentrations of BOD_5, and comparatively better treatment efficiency even without zeolite.

- **Zlatni Pqsyci WWTP (2009)**

The results from the dosing and sampling campaign in 2009 are shown in the figures below for each key water characterization parameter. Stream 1 represents the effluent results without addition of zeolite and Stream 2 those with 10 mg/L zeolite dosed.

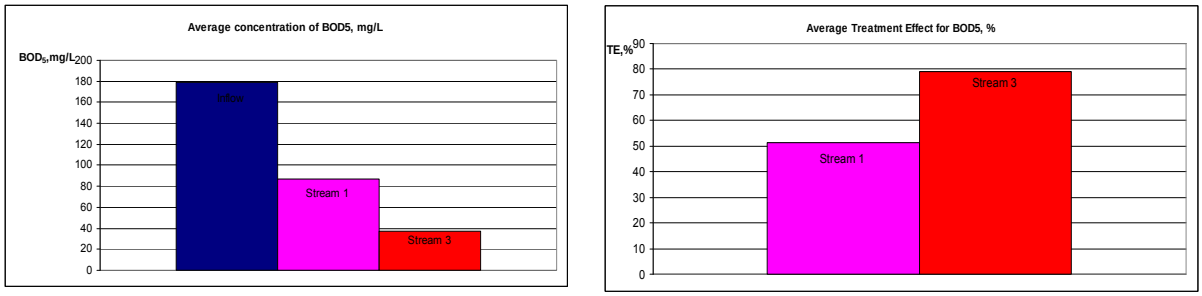

Figure 53. Averaged concentration (left) and averaged treatment efficiency (right) for BOD_5

As Figure 53 shows, the addition of zeolite had a noticeable effect on the removal of BOD_5 with a reduction of concentration from 180 mg/L at the inlet to 37.5 mg/L in the final effluent, which compared to the 87 mg/L in the stream without zeolite, resulted in 57% increase of removal . The treatment effect rose from 51 to 79%, equating to nearly 54% increase.

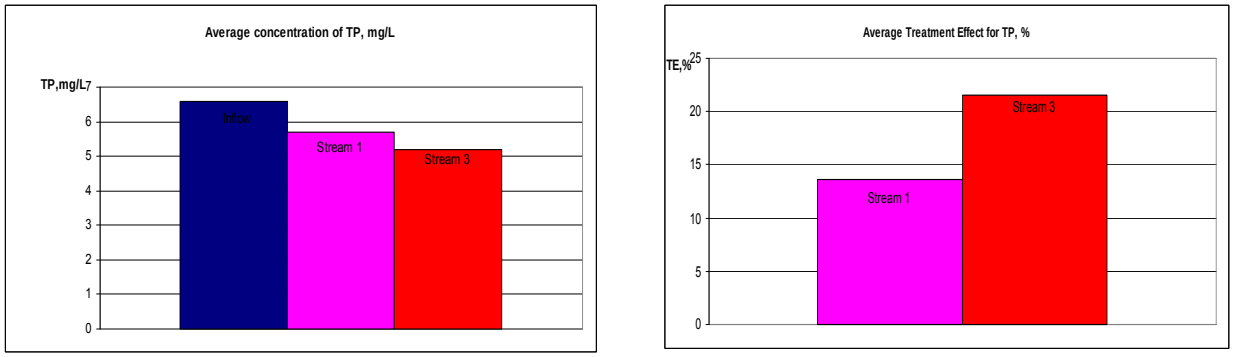

Figure 54. Averaged concentration and averaged treatment efficiency for Total Phosphorus

The average phosphorus discharge concentration was reduced from 5.7 mg/L without zeolite to 5.18 mg/L with zeolite, resulting in 9% increase of removal. The average treatment effect rose from 13.6 to 21.5 %, showing 58% increase.

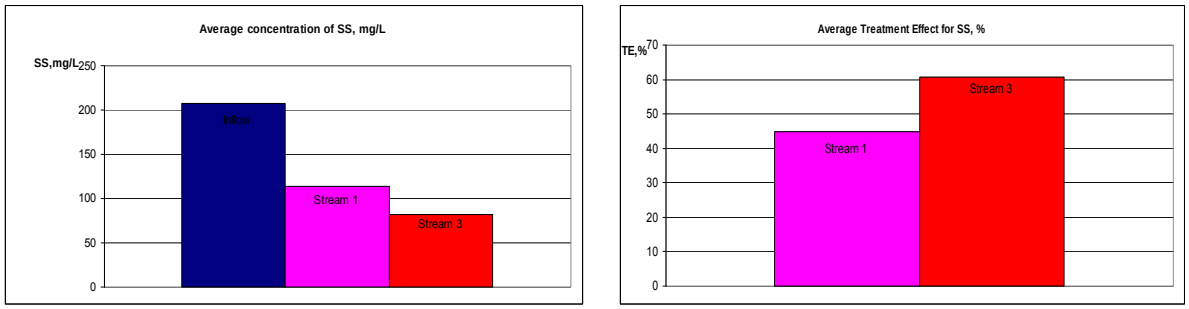

Figure 55. Averaged concentration (left) and averaged treatment efficiency (right) for Suspended Solids

Figure 55shows that the concentration of suspended solids was reduced from 207 mg/L at the inlet to 81.5 mg/L at the outlet with addition of zeolite. Compared to the removal without zeolite, the increase of removal was 28.5%. The treatment efficiency rose from 45 to 60.6%.

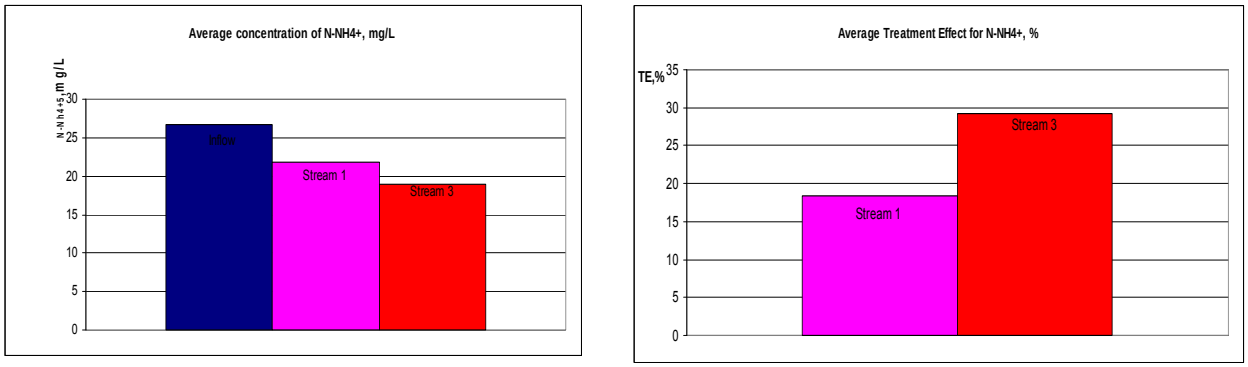

Figure 56. Averaged concentration and averaged treatment efficiency for N-NH4

As seen on Figure 56, the average ammonium concentrations was reduced from 26.7 mg/L at the inlet to 18.9 mg/L at the outlet with addition of zeolite. Compared to the removal without zeolite, the increase of removal was 13.3%. The treatment efficiency rose from 18.3 to 29.2%, a 59% increase.

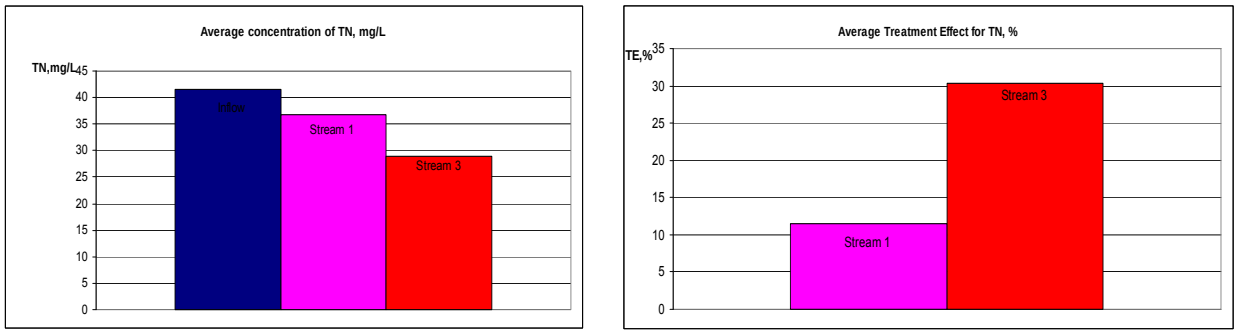

Figure 57. Averaged concentration (left) and averaged treatment efficiency (right) for Total Nitrogen

As seen on Figure 57, the average total nitrogen was reduced from 41.6 mg/L at the inlet to 29.0 mg/L at the outlet with addition of zeolite. Compared to the removal without zeolite, the increase of removal was 21.2%. The treatment efficiency rose from 11.5 to 30.3%, a 162% increase.

The above results were evaluated against the data of the experiments in 2008 and a comparison is presented in Table 23 below.

Treatment efficiency (removal) and increase of treatment efficiency achieved at Zlatni pqsyci WWTP			
	Year	2008 (summer)	2009 (summer)
	Dose, mg zeolite /L wastewater	10.3	10.0
	Dose, kg zeolite/day	31	30.0
Suspended solids	Treatment efficiency, %	63.7	60.6
	Increase of treatment efficiency, %	33	34.9
Ammonium nitrogen	Treatment efficiency, %	25.5	29.2
	Increase of treatment efficiency, %	50.9	59.2
BOD_5	Treatment efficiency, %	79.7	79.05
	Increase of treatment efficiency, %	49.8	53.8
TP	Treatment efficiency, %	25.6	21.5
	Increase of treatment efficiency, %	61.4	57.8
TN	Treatment efficiency, %	31.3	30.3
	Increase of treatment efficiency, %	134.5	162.5

Table 23. *Comparison of the treatment efficiency and increase of treatment efficiency in 2008 and 2009*

- The treatment efficiency (% removal) of suspended solids was nearly 61% and was slightly lower than the 63.7% in 2008. The increase of treatment efficiency was 35%, only marginally higher than the 33% in 2008.
- The treatment efficiency (% removal) of ammonium nitrogen was 29% showing improvement from the 25% in 2008. The increase in treatment efficiency was 59%, compared to 51% in 2008.
- The treatment efficiency (% removal) of BOD_5 leveled out with that in 2008 at 79%. However, the increase of treatment efficiency was higher nearly 54% - 4% higher than in 2008.
- The treatment efficiency (% removal) and the increase of treatment efficiency of total phosphorus remained lower than those in 2008, at 21.5% and 57.8%.
- The treatment efficiency (% removal) of total nitrogen was nearly leveled out with that in 2008 at 30%, with only one per cent difference. However, the increase of treatment efficiency was higher at 162% compared to 135% in 2008.

As a result of the addtion of zeolite and improved removal of BOD_5, the sludge production increased by 54%. This was calculated as a ratio of the net increase of sludge production to the sludge production without zeolite, not accounting for the mass of zeolite added. This value was similar to the 60% value in 2008.

8.7.1.2. Efficiencies for micropollutants removal – Zlatni Pqsyci WWTP

Samples were collected from the inlet to the bioreactor, from the outlet reference stream without zeolite addition, and from the outlets of the streams where 21 kg/d and 31 kg/d zeolite was dosed per stream.

The test results are summarized in Table 24 below. Compounds 1 to 11 were analyzed by NCDITIM and 12-19 by IRSA. Some of the results were found to be below the limit of quantitation (LOQ).

Table 24. Concentration of micropollutants, ng/L

No.	Compound	Limit of Quantification (LOQ)	7-8 July 08 (day 1)				8-9 July 08 (day 2)				9-10 July 08 (day 3)			
			Point 0	Line 1	Line 2	Line 3	Point 0	Line 1	Line 2	Line 3	Point 0	Line 1	Line 2	Line 3
1	Buthylhidroxianisole (BHA)	30	824	717	825	413	1007	866	945	475	805	749	709	714
2	Buthylhidroxitoluene (BHT)	30	560	343	220	88	306	269	215	137	264	237	196	206
3	Prometon	7	*	*	*	*	*	*	*	*	*	*	*	*
4	Atrazin	40	*	*	*	*	*	*	*	*	*	*	*	*
5	Propazin	13	*	*	*	*	*	*	*	*	*	*	*	*
6	Ametrin	30	*	*	*	*	*	*	*	*	*	*	*	*
7	Diuron	20	*	*	*	*	*	*	*	*	*	*	*	*
8	Isoproturon	7	*	*	*	*	*	*	*	*	*	*	*	*
9	Tonalide	15	251	349	150	159	166	251	199	154	213	199	182	144
10	Triclosan	7	762	264	476	215	490	459	505	288	496	472	531	254
11	Carbamazepine	10	72	77	77	112	105	114	105	113	60	37	44	58
12	4-n-Octylphenol (4-OP)	0.2	*	*	*	*	*	*	*	*	*	*	*	*
13	4-ter-Octylphenol (t-OP)	2	13	7	7	13	24	11	19	24	45	10	8	*
14	4-ter-Nonylphenol technical (NP techical)	5	196	194	189	191	173	167	162	172	225	134	143	182
15	4-Nonylphenol (4n-NP)	2	*	*	*	*	*	*	*	*	*	*	*	*
16	4-Octylphenol mono ethoxylate (OP1EO)	5	22	8	11	7	42	26	7	11	18	13	11	15
17	4-Octylphenol di ethoxylate (OP2EO)	0.2	8	2.2	9	10	19	20	2.5	9	19	12	8	7
18	4-Nonylphenol mono ethoxylate (NP1EO)	5	287	133	146	159	222	157	138	169	111	66	61	78
19	4-Nonylphenol di ethoxylate (NP2EO)	0.2	87	84	66	85	86	75	79	88	115	89	102	65

Note 1: Point 0 - inlet; Line 1 - without zeolite; Line 2 - 21 kg/ d per stream zeolite dose; Line 3 - 31 kg/ d per stream zeolite dose.
Note 2: Compounds 1 to 11 analysed by INCDTIM, compounds 12 to 19 analysed by IRSA.
Note 3: Antioxidants (1,2), triazines (3-6), phenyl urea herbicides (7,8), fragrances (9), antimicrobial agents (10) and antiepileptic drugs (11) in ng/l. The symbol * is for concentration under LOQ.

The key results achieved are described below:

- Antioxidants (items 1,2) showed stable reduction. The treatment effect increased significantly with an increase of the dose of zeolite. However, this effect was lost on the 3rd day.

- The fragrances (item 9) showed noticeable reduction for all three days.

- Triclosan (item10) demonstrated a distinctive improvement (50% reduction) of the treatment efficiency with the higher dose.

- Steady reduction was observed for item 13 for one day only.

- A large part of the micropollutants was found in concentrations below the LOQ, even in the influent wastewater.

In summary, although some micropollutants (antioxidants and fragrances) showed a trend of reduction when zeolite was dosed to the water and the dose was increased, the results showed some inconsistencies.

In addition, the limited number of samples did not allow to build up data database that is sufficient for statistical analyses, and hence observation of definite trends.

In conclusion, the results of the experiment did not present enough evidence that the addition of zeolite has a positive effect on the removal of micropollutants. The results were inconclusive and further investigation would be required to confirm the effect of zeolites on the removal of micropollutants.

8.7.2. Costs

The trials conducted under NEPTUNE have demonstrated the potential of this technology to increase the capacity and performance of existing plants with minimal capital expenditures.

An additional cost of 0.021 €/ m3 was incurred for a dose of 10.3 mg zeolite/l wastewater. This was for average daily inflow of 9 000 m^3. The costs increased with reduction of the plant size and reached 0.025 €/m^3 for a plant of 630 m3/d inflow. No additional personnel cost is expected once the system is set and operational, assuming the system can be monitored and maintained by the regular plant staff.

An example of the costs associated with zeolite addition is presented in Table 25.

Table 25. Capital and operational costs for a zeolite dosing system for 9000 m3/d WWTP

Cost elements	Unit	Number of units	Unit price, €
Tensometric dosing unit	#	3	5000
Personnel	Hours/ week	3.0	Varies
Energy consumption	kW	0.2	Varies
Zeolite	€/ m3	0.021	-

In comparison with the alternative of construction of new infrastructure, the costs for zeolite addition are negligible. Still, a detailed economic analysis is needed for each specific application to evaluate the total impact of zeolite addition on the overall wastewater treatment process, especially if the excess sludge cannot be used in agriculture, but has to be incinerated.

A summary of the dose and the associated cost of zeolite addition are presented in Figure 58. The costs are for the zeolite only and are subject of market price.

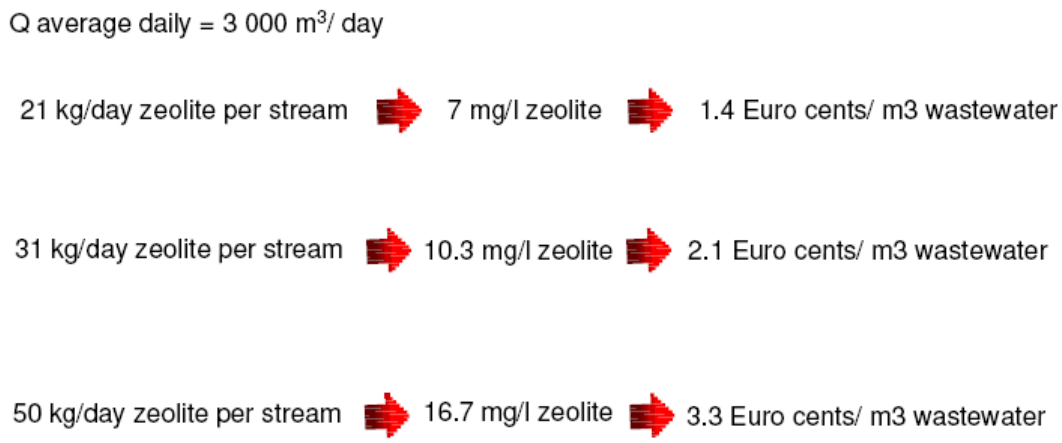

Figure 58. Cost of zeolite addition for 9'000 m3/d WWTP

8.8. Conclusions

The experiments conducted under the NEPTUNE project confirmed the capacity of zeolites in general and SZEDIMENTIN-MW in particular, to remove wastewater nutrients and showed a significant effect in the elimination of ammonium nitrogen and on SVI improvement. The nitrification was enhanced and rapid settling of sludge in the secondary settling tanks was achieved preventing bulking of sludge and solids carry-over in the final effluent. The quality of the final effluent was improved to levels, which were difficult to achieve otherwise during high load periods.

It was proven that the effect of zeolite on elimination of ammonium nitrogen was equally good at high summer and at low winter temperatures.

The experiments confirmed that zeolites are an inexpensive way to increase the capacity of seasonally overloaded plants without construction of new infrastructure. Excellent results were achieved with a dose as low as 0.005 kg/m^3.

However, although the dosing installation was simple to install and run, it was not fully automated and required manual handling for loading of the zeolite.

A draw back of the process was that additional sludge was generated, increasing the requirements for processing and disposal.

Being an inexpensive and environmentally friendly technology, the limits of application of zeolite addition should be fully explored. Further investigation of the applicability of zeolite to wastewaters of various nature and quality would give a more profound understanding of the extent of application of this technology.

9. Treatment of digester liquid

9.1. Scope of digester liquid treatment

Equipping the anaerobic sludge digester with a post-treatment unit for nitrogen removal opens a new window for plant optimization in terms of nutrient removal as well as overall energy requirement. As discussed below this concerns mainly the following aspects:

- reduction of effluent nitrogen load
- reducing plant wide energy consumption by disposing biowaste into the digester
- reducing greenhouse gas emission
- cost reduction

9.1.1. Reduction of effluent nitrogen load

As illustrated by the comparison in Figure 59, removing the nitrogen load from the digester supernatant avoids the recycle of the ammonia nitrogen from the digeser to the influent. In the case of unchanged denitrification, this directly translates to a corresponding reduction of the effluent N-load: this is often the case since denitrification is mostly limited either by substrate, nitrate recycle or availability of denitrification volume. Since the separate treatment with combined nitritation/anammox is significantly more efficient in terms of resource requirement compared to nitrification/denitrification in the activated sludge tank (section 9.3), separate treatment may represent an optimal choice for reducing effluent nitrogen load, for WWTP equipped with a digester.

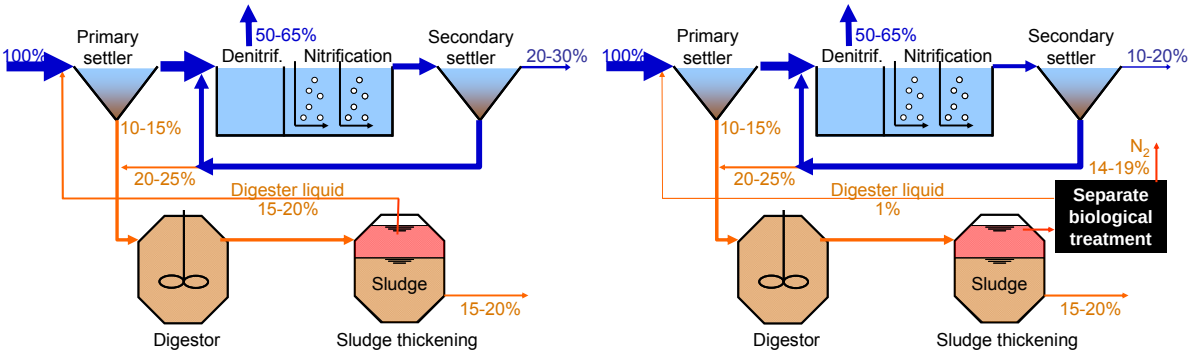

Figure 59 Nitrogen mass flux in a municipal WWTP achieving 50-65% denitrification in the activated sludge tank with (left) and without (right) digester liquid treatment. Percents indicate typical nitrogen loads normalized to the influent load.

9.1.2. Plant wide energy consumption

Removing the nitrogen load from the digester liquid allows accepting more biowaste for disposal via anaerobic digestion. As discussed by Siegrist and co-worker (2008), this allows increasing the biogas production and by using the biogas for power generation to cover the entirely or partly the energy requirement of the WWTP: applying this concept, the WWTP Zürich-Werdhölzli (600'000 population equivalents; digestor liquid treated in two reactors for combined nitritation/anammox of 1400 m^3 each) is currently covering 90% of its energy requirement achieving a net energy demand of only 30 Wh per m^3 of wastewater treated (Burger et al., 2009). Similar results have been reported from the WWTP Strass (Wett, 2008).

Optimizing plant design with an efficient N-removal after digestion may make WWTP with big primary clarifiers preferable (e.g. with hydraulic retention time in the primary clarifier of about 2 h instead of only 1 h): thanks to the 15% to 20% reduced influent N-load (Figure 59) less organics from the influent is required to achieve the required denitrification in the water line. With increased settling time more of this organics (and the corresponding part of the N-load) may therefore be diverted to the digester for biogas production.

A further aspect relevant for energy consumption is that nitrogen removal in separate treatment is significantly more efficient than conventional nitrification/denitrification (Joss et al., 2009). In the following figures typical for combined nitritation/anammox are compared to conventional nitrification/denitrification:

- Electricity consumption for aeration of 1.0 instead of 2.4 kWh electricity per kg of N removed, since only half of the ammonia has to be oxidized and only to N^{+III} (nitrite) instead of N^{+V} (nitrate)
- No organic substrate is required for N-removal: 2.2 kg methanol per kgN used in conventional separate treatment corresponds to 12 $kWh \cdot kgN_{removed}$. Alternatively incoming organic substrate may be used instead of methanol for denitrification without significant change in terms of the net energy balance, since this organic substrate could be conveyed to the digester for biogas production if not used for denitrification.

9.1.3. Greenhouse gas emission

The greenhouse gas emission of nitrogen removal is dominated by a) the CO_2-equivalents emitted due to the overall energy requirement of the process and b) direct emissions of nitrous oxide (N_2O). To comply as a favorable treatment option, separate digester liquid treatment must be better than the reference, which in most cases is a plant without separate digester liquid treatment achieving nitrification/denitrification in the activated sludge tank.

Since N_2O is 310 times more climate relevant compared to CO_2 on a weight basis, if >1% of the influent N-load (untreated wastewater) is emitted as N_2O, this emission is more relevant for the climate, than the 0.3 to 0.5 kWh electrical energy typically consumed to treat 1 m^3 of wastewater. According to new studies, there is quite some uncertainty on the N_2O emission to be expected from activated sludge treatment, since the emission has been shown to increase beyond the threshold of relevance under not yet clearly identified operating conditions. An average emission of 0.1% of the removed N-load is regarded here as representative for a good activated sludge performance (von Schultess et al., 1995).

Joss and co-worker (2009) measured the N_2O emission of combined nitritation/anammox and came to the conclusion, that even under favorable operating conditions of the activated sludge system the energy saving of combined nitritation/anammox is clearly weight more than the N_2O emission monitored on a full scale installation (Table 26). This leads to the conclusion that separate sludge liquid treatment with nitritation/anammox allows minimizing the greenhouse gas emission.

Table 26: Emissions of CO_2 equivalents from nitritation-anammox (continuous aeration) and a conventional nitrification/denitrification process at the WWTP Zurich. From Joss et al., 2009.

		Conventional nitr./denitr. in the water line	Nitritation-anammox in supernatant of digested sludge
O_2 consumption	kg O_2 $kg^{-1}_{N\ eliminated}$	4.3 [1)]	1.9 [2)]
Aeration energy [3)]	kWh $kg^{-1}_{N\ eliminated}$	2.4	1.0
Aeration (CO_2 equivalent) [4)]	kg CO_2 $kg^{-1}_{N\ eliminated}$	**1.4**	**0.6**
Carbon source	kg_{MeOH} $kg^{-1}_{N\ eliminated}$	2.2	-
Carbon source (CO_2 equ) [5)]	kg CO_2 $kg^{-1}_{N\ eliminated}$	**3.1**	**-**
N_2O production	gN_2O $kg^{-1}_{N\ eliminated}$	3.1 [6)]	6.3 [7)]
N_2O production (CO_2 equ) [8)]	kg CO_2 $kg^{-1}_{N\ eliminated}$	**1.0**	**1.9**
Total CO_2 equivalents	**kg CO_2 $kg^{-1}_{N\ eliminated}$**	**5.5**	**2.5**

Assumptions: [1)]NH_4^+ oxidized to 95% to NO_3^- and incorporated into the biomass to 5%; [2)]NH_4^+ oxidized to 90% to N_2 and to 10% to NO_3^-; [3)]0.55 kWh $kg^{-1}O_2$, (α factor = 0.6; Wagner and Pöpel, 1998); [4)]0.61 kg CO_2 $kWh^{-1}_{electrical}$ (EPA, 2000); [5)] 1.4 kg CO_2 kg^{-1}MeOH; [6)] 0.1% of the removed N load emitted as N_2O (von Schultess et al., 1995); [7)] 0.4% of the N load emitted as N_2O, (Joss et al., 2009); [8)] 310 kg CO_2 $kg^{-1}N_2O$ (IPCC, 2001).

9.1.4. Cost reduction

For the evaluation of costs and cost savings the following aspects need to be considered:

- Investment costs: combined nitritation/anammox achieves a net N-removal rate of > 500 $gN \cdot m^{-3} \cdot d^{-1}$, resulting in a hydraulic retention time for typical digester liquid of about 1.5 d; accordingly the reactor volume does not represent a dominant cost issue; a detailled description of the reactor set up is given in Joss et al., (2009)
- Personnel costs: mainly supervision and maintenance of sensors, pumps and aerators
- Energy costs may be estimated based on the local situation and the figures given in Table 26
- Costs for co-substrate: the generally rising costs for energy in the last decade go in parallel with increased value of co-substrates (e.g. methanol or acetate); typically required quantities are given in Table 26

A general cost calculation cannot be given here, since it is too much dependent on the local situation (availability of personnel and infrastructure, price for electricity). Nevertheless reducing costs for achieving a set nitrogen removal or energy consumption goal represents a major motivator to implement separated digester liquid treatment.

9.2. Combined nitritation / anammox

Combined nitritation / anammox with a sequencing batch reactor is discussed here in detail, because it is deemed as one of the most efficient solutions for digester liquid treatment and to present the advances made within the project Neptune. More details are found in Joss et al. (2009).

9.2.1. Reactor setup

As illustrated in Figure 60 the sequencing batch reactor is composed by the following components:

- reactor: completely stirred batch reactor dimensioned for 1 to 2 days hydraulic retention time
- stirrer
- aeration unit for fine bubble aeration dimensioned for supply air at a rate of maximally 0.15 $kgO_2 \cdot m^{-3}_{reactor} \cdot h^{-1}$ (for a maximal activity of 1 $kgN \cdot m^{-3}_{reactor} \cdot d^{-1}$ and continuous aeration); for start-up the minimal air supply should be as small as possible (i.e. switching to intermittent aeration)

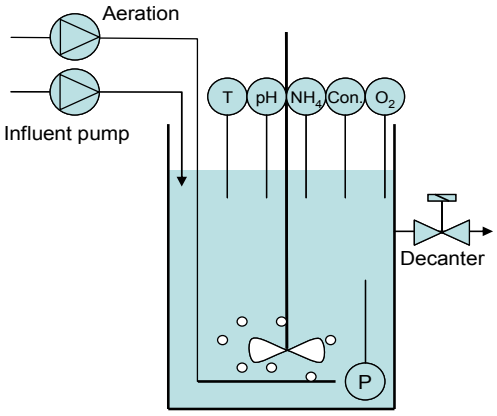

Figure 60: Schematic diagram of the reactor setup; sensors: temperature (T), pH, ammonia (NH4), conductivity (Con.), oxygen (O_2) and fill level (P).

- influent pump: 1 to 2 hours feeding during each batch of ca. 8 hours duration
- decantation unit
- online sensors for temperature, pH, ammonia, conductivity, soluble oxygen and filling level
- sprinkler unit for foam control

9.2.2. Process control

The batch process is devided into the following phases:

- filling of fresh supernatant; the exchange volume per batch is normally in the range of 5% to 30%
- aeration: continuous or intermittent (for regular operation resp. start-up); the areation phase is terminated when the target NH_4^+ concentration is reached; in case of intermittent aeration, the stirring unit is to be switched on when no air is supplied to the reactor
- stirring: for removal of residual NO_2^- prior to sedimentation and control of NO_2^- accumulation during the aeration phase
- sedimentation: under regular operation the sludge features good settling performance making further devices for sludge retention useless (e.g. cyclone); only during the startup of one reactor, a flucculant was added during few weeks, when the sludge featured bad settling characteristics

- decantation of treated effluent
- pause before starting the next batch for adapting to the digester liquid volume to be treated

The conductivity signal is used as redundant control for the NH_4^+ sensor: the two signals are parallel under normal operating conditions, since ammonia and bicarbonate ions are the dominant ions removed during the process. The air supply is controlled via flow rate of the blower, while the oxygen sensor is used only as a control, to avoid the O_2 concentration trespassing 0.5 to 1 $mgO_2 \cdot L^{-1}$ (the blower is automatically shut down in this case). pH is only monitored (not used for reactor control). The temperature range from 20 to 35 °C has been tested successfully for operation: with the supernatant coming from mesophilic digestion, the temperature range can mostly be kept without active heating. According to current full scale experience, foam is encountered at times, making a sprinkler unit required. Excess sludge is withdrawn by decanting while stirring.

9.2.3. Process stability

Based on the experience of five full scale reactors we directly followed since start-up in 2008 and several reactor years of piloting, the stability of the process as described in section 9.2 can be rated as generally very high.

According to current full scale operation experience, problems with process stability have been encountered only in the following situations (Joss et al., in preparation):

- Sudden activity loss by the ammonia oxidizers, presumibly due to a yet unidentified toxic effect; the activity loss leads to a decreased O_2 consumption. If the air supply is adapted to the activity, the accumulation of O_2 in the reactor may lead to reversible inhibition of the anammox biomass, resulting in NO_2^- accumulation. If the air supply to the reactor is adjusted to the consumption activity by reducing the output of the blower, the only consequence to be faced by the operator is a transiently reduced throughput of digester liquid, while the inhibitory effect persists. The effect is currently being studied.
- Nitrite oxidizers growing into the system scavenge nitrite from the anammox biomass, producing nitrate, which accumulates in the effluent since denitrification is mostly limited by the availability of organic substrate. The establishment of a relevant population of nitrite oxidizers occurs if excess air is supplied to the system, leading to inhibition of the anammox activity and accumulation of NO_2^-. Currently no strategy for quick removal of nitrite oxidizers is available once a significant population has established in the reactor. The operation strategy is therefore to avoid supplying air in excess, operating the reactor always slightly below the maximal performance

9.2.4. Performance

With up to 1 $kgNremoved \cdot m^{-3} \cdot d^{-1}$ combined nitritation / anammox is competitive with process alternatives (section 9.3). The main advantages of the combined system in a single batch reactor are:

- simplicity of the process (i.e. small number of units requiring supervision and maintenance)
- rubostness and understandable strategy for process monitoring and control

- ease of start-up, which is comparable to other suspended sludge systems, as long as sufficient sludge is available (with a multitude of full scale projects currently in the pipeline, this is supposed to be the case in few years only)

9.3. Treatment alternatives

Since the project Neptune primarily focused on combined nitritation / anammox, the reader may refer to literature for details on process alternatives, while here only the most important differences are pointed out.

The following processes may be used for the same scope as combined nitritation / anammox (Wyffels et al., 2004; Abma et al., 2007; Abma et al., 2007b; Gustavsson et al., 2008; van Kempen et al., 2001; van der Star et al., 2007):

- nitrification / denitrification in the activated sludge tank (water line): the advantage is that here no additional reactor is required; this solution is expected to be significantly less efficient in terms of energy requirement and greenhouse gas emission (section 9.1)

- nitrification / denitrification in a separated reactor treating digester liquid: this solution is expected to be significantly less efficient in terms of energy requirement and greenhouse gas emission (section 9.1)

- two step process segregating nitritation and anammox in separate reactors: very high performance is achieved in the anammox reactor, while the nitritation reactor achives rates similar to the combined system; the two reactor system requires additional units and control.

- nitritation / anammox achieved by attached growth reactors (biofilm reactors): start-up is expected to be more time consuming than for suspended sludge reactors, assuming sufficient inocculum available; it is not yet clear, which strategy is effective to avoid growth of nitrite oxidizers

10. Identification of toxic chemicals and industrial waste-water with in-situ sensor in the sewer or WWTP inlet

The load and quality of industrial wastwater is normally approved by the local water authority based on the water protection guidelines to prevent negativ effects on the sewer system and the central municipal wastewater treatment plant. Nevertheless, if highly polluted industrial wastewater or toxic chemicals are entering the sewer system as short peak load events aeration and nitrification of the municipal WWTP could be strongly overloaded or inhibited.

To detect such events the normal 24h composite sampling of COD and ammonia at the WWTP inlet is not sufficient. Peak loads are passing the plant inlet within a short period in the range of minutes to hours and might change the wastewater matrix that can only be detected by permanently monitoring (in short intervals of minutes) the wastewater with a combination of different in-situ sensors (UV/Vis spectra, ammonia, pH, conductivty, redox,...).

Within the wastewater treatment plant installation of on-line sensors directly in the channel can easily be done. Details about the identification of industrial waste water using the s::can UV-Vis spectro::lyser are explained in section 10.1. The direct installation of the sensors in the sewer system can cause a lot of problems due to low sewage depth, sand, gravel, papers, hairs and plastic pieces. Therefore a specific sensor setup was used, which is explained in section 10.2.

10.1. In situ wastewater monitoring with spectrometer probe

A UV-Vis spectrometer probe with 1mm optical pathlegth was installed in the inlet channel of two different waste water treatment plants in Belgium to monitor the influent for a longer period. Both treatment plants are highly influenced by waste water from industries.

10.1.1. Monitored absorbance spectra

At the WWTP of Aartselaar absorbance spectra were monitored from May 2007 until June 2008 and at the WWTP of Ronse the spectro::lyser was installed from August until November 2009 to monitor the wastewater. To have a first impression of the highly variation of different waste water composition several absorbance spectra (“fingerprints”) of the WWTP Ronse are displayed in Figure 61. Horizontal the wavelength in nm and vertical the corresponding absorbance (extinction) in Abs/m is displayed.

The different shape of the fingerprints can either be caused by influent from different industries or by different production cycles of one industry.

The waste water of the WWTP Ronse can be characterised by a huge variance of waste water matrices. Different fingerprints are not only detected between weekend and working days but also within a few hours of only one day of monitoring. The monitored fingerprints during the night from Friday 28th to Saturday 29th of August are displayed as a 3D diagram in Figure 62 and gives an impression how quickly different types of wastewater will enter the treatment plant and can be identified.

One typical matrix shows a significant absorption peak in the UV range at around 280nm and seems to appear periodically but only for short periods. The other typical matrix can be identified by a significant increase of absorbance in the visible range (400 to 600nm). Finally, at the end of the analysed time frame, the typical fingerprint of the weekend with rather low absorbance both in the UV range and in the visible range was monitored.

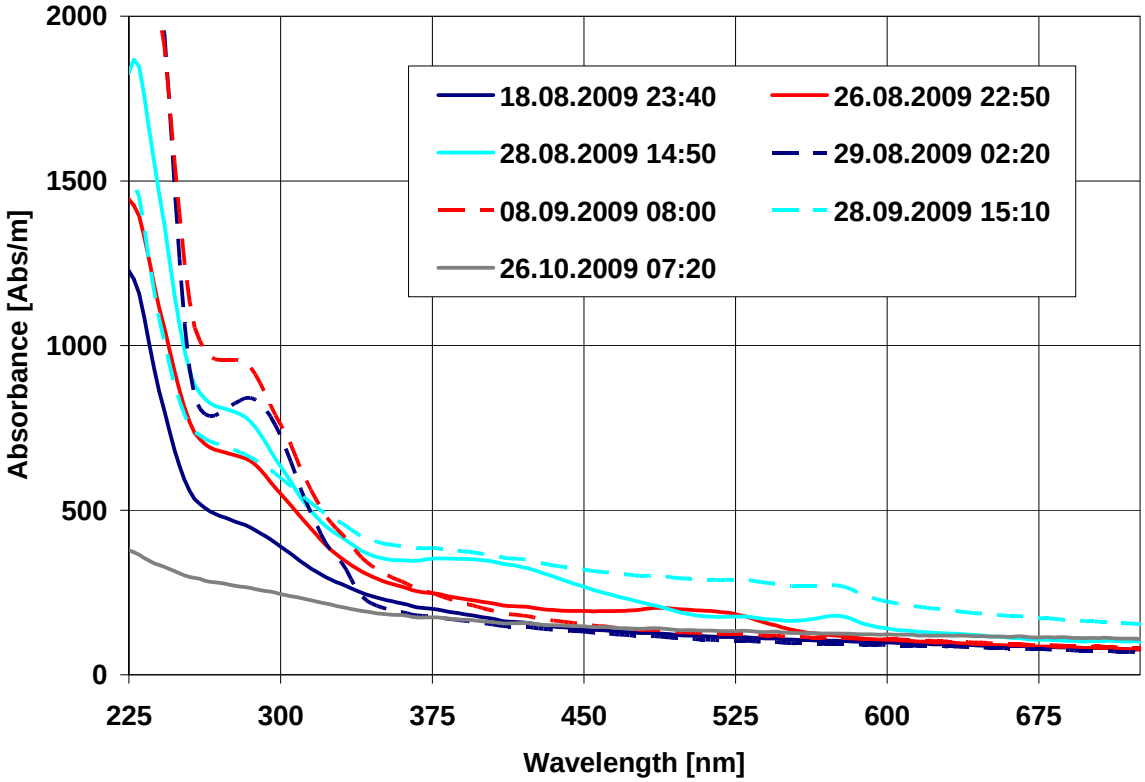

Figure 61: Fingerprints of waste water monitored at WWTP Ronse with 1mm spectro::lyser.

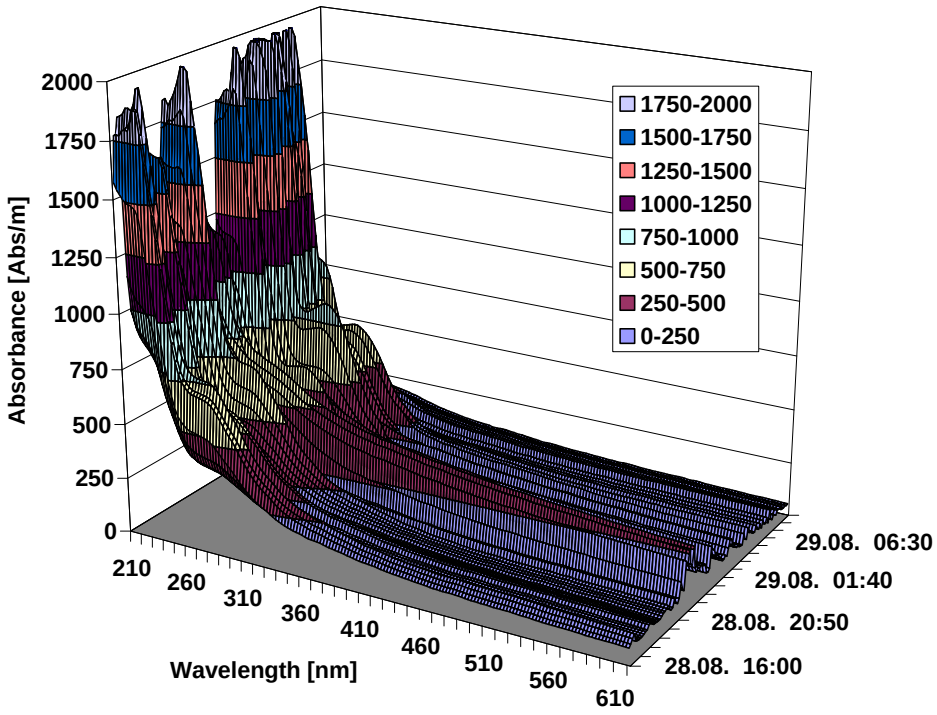

Figure 62: 3D display of fingerprint variation monitored on-line at WWTP Ronse.

10.1.2. Analysing the amount of industrial influent

To identify different types of waste water, obviously caused by industrial influent, the ana::larm software can be used. This specific software, developed by s::can, uses the measured absorbance spectra (fingerprints) over a longer period to learn the shape and features of the “normal” spectrum of the water body in a defined range. Absorbance spectra or 1st and 2nd order derivatives of UV/Vis spectra are used to identify abnormalities from the “normal” spectral pattern.

One option is to use only the fingerprints monitored during weekends to train the ana::larm software for the “normal” shape of the water matrix. This will result in a lot of alarm events during the working days.

The second option, also used in this case, is to use all monitored fingerprints for training and define a specific amount of data to be treated as an alarm trigger. The time series below shows the already calibrated results of the ana::larm software. The different alarm parameters have been individually trained to the specific application Figure 63.

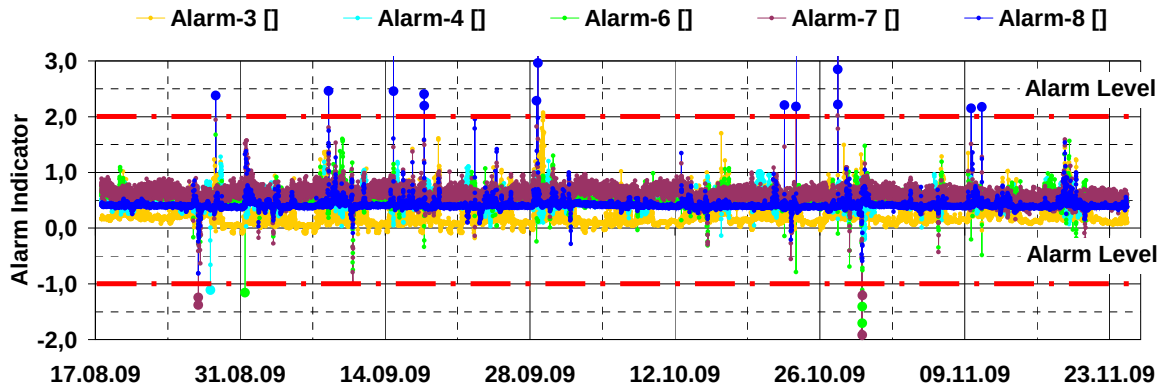

Figure 63: Individual calibrated alarm parameters for identification of non typical waste water.

By default all alarm parameters are calibrated between 0 and 1. A warning will be indicated if the values are less than -0,1 or higher than 1,1. Readings of the alarm parameters less than -1,0 and higher than 2,0 will cause an alarm.

Several events (marked with a bold dot in Figure 63) are detected by the ana::larm software within the monitoring period. One of this events, the alarm detection on 14th of September is analysed more in detail below (Figure 64).

In the late evening of Monday the 14th of September a significant peak in the visible range of the absorbance spectra causes this alarm. This abnormality, obviously caused by coloured wastewater, was only detected by one measurement. Thus this untypical wastewater composition occurred for a shorter period than 10 minutes. This indicates, that grab samples, even when sampling each hour, will never catch up such short period events.

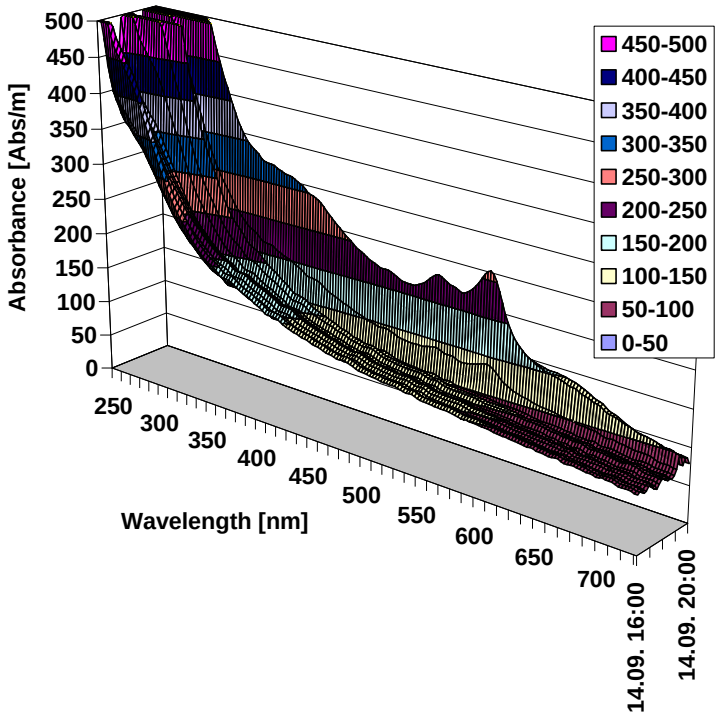

Figure 64: Monitored fingerprints during alarm event caused by non typical waste water.

10.2. Wastewater monitoring station

10.2.1. In-situ sensor setup

With the the in-situ sensor setup “uni-pass” (Figure 65, Hofer, 2009) that pumps sewage from the sewer into a plastic channel, equipped with different in-situ sensors and an automatic cleaning system, the wastewater quality of different sewers (serving more than 5000 PE, minimum flow required) and the WWTP inlet were reliably monitored over several weeks.

With the parallel monitoring of different parameters (UV spectra, NH_4 and conductivity) specific industrial wastewater can be detected and the polluter localized in the sewer system.

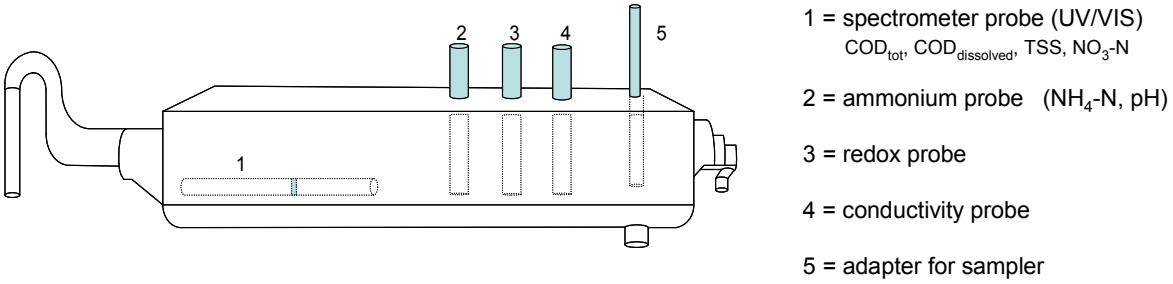

Figure 65: Uni pass system to measure sewage from sewer or plant inlet with a combination of different in-situ sensors and the possibility to take peak event sample.

10.2.2. Peak load events from industry

At the treatment plant Limmattal (120'000 PE) three different industrial peak events could be detected and localized in the sewer system (Hofer, 2009):

a) short (< 1hour) ammonium and COD peaks,

b) long (hours) conductivity, nitrate and ammonia peaks and

c) short high COD peaks (Figure 66, top).

During all three events the UV spectra was significantly changed between 220 and 350 nm wavelength (Figure 66, bottem).

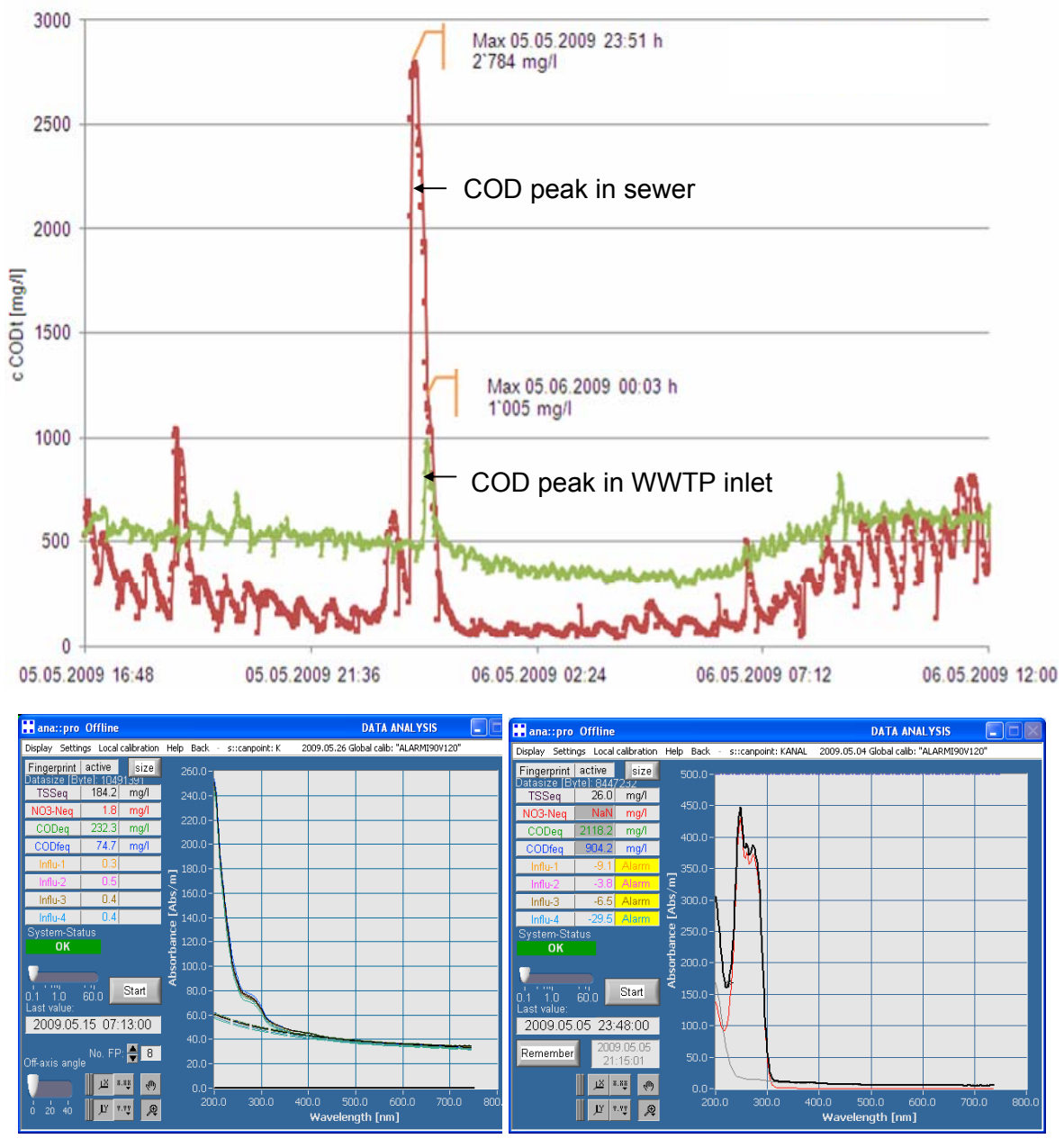

Figure 66: Localization of high COD peak in plant inlet and the sewer system (top). Comparison of UV spectra of typical municipal wastewater (bottem left) and the modification during above industrial COD peak (bottem right).

10.3. Conclusions

The installation of an UV-Vis spectrometer probe with 1mm optical pathlegth in the inlet channel of waste water treatment plants allowed to characterise a huge variance of waste water matrices within a few hours of only one day of monitoring.

To identify different types of wastewater, obviously caused by industrial influent, the ana::larm software, developed by the partner s::can, can be used. It analysis the measured absorbance spectra (fingerprints) over a longer period and identifies abnormalities from the “normal” spectral pattern. Because such untypical wastewater compositions are occurring for a shorter period than 10 minutes, grab samples, even when sampling each hour, will never catch up such short events.

The developed uni-pass (by-pass) system equipped with s::can, ammonium, conductivity and redox sensors is capable for quality measurements of wastewater in WWTP inlet and different position in the sewer that serve at least 5000 PE (minimal flow required). With the combination of different measurement parameters specific industrial polluters could be determined and localized in the catchment. The online monitoring with this system can be an excellent basis to introduce measures (temporary storage of industrial wastewater at the plant or in industry, introducing (peak) pollution fees for industry that induce load equalization and reduction) preventing temporary overloading and inhibition of the WWTP biology.

11. Overal Conclusions

It has been demonstrated that nutrient removal can be improved in biological treatment in different ways. The following conclusions have been achieved from the research conducted:

- Different in-situ sensors have been characterized according to the ISO 15839:2003 protocol. This study allowed identifying drawbacks and improvements to the protocol, e.g. the comparison of sensors should be performed for the same measuring range.
- After identifying the limitations for characterizing sensors under field conditions a suggestion is presented in this project for air bubbles and turbidity.
- A successful calibration method based on PLS regression was used for an existing UV spectrometer sensor. With that calibration the sensor was able to accurately predict the nitrite and nitrate concentrations for a long-term period.
- Univariate and multivariate methods were implemented to continuously evaluate the data quality of on-line sensors. An evaluation index was also developed in order to compare the performance of the methods to detect the faults. As it was demonstrated these methods were not able to detect the faults of sensors involved in control loops with sufficiently accuracy and speed and therefore further research is needed.
- Different control strategies were implemented and evaluated to improve nutrient removal. The use of control can reduce environmental impact and energy consumption. The selection of correct control settings is essential to maximize the benefits of control.
- Different sets of control strategies have been evaluated using two types of approaches. In the first approach different multivariate techniques are used to identify groups of strategies that perform similarly and to identify which are the relevant criteria. In the second approach Life Cycle Analysis has been used to measure the environmental impact of control implementation. This has been demonstrated for both a Neptune benchmark plant and for different full-scale wastewater treatment plants.
- Addition of modified natural zeolite can improve full-scale plant performance, especially to improve nitrogen removal and SVI, for both summer and winter periods.
- Separate treatment of digester liquid with combined nitritation / anammox in a sequincing batch reactor is regarded as a robust and cost effective solution for improving nitrogen removal of WWTP; it allows increasing the amount of biowaste disposed into the anaerobic digestor. Full scale installations confirm that by using the produced biogas for power generation, the entire energy requirement for wastewater treatment can be covered.
- An in-situ sensor setup, permanently feeded with sewage and equipped with spectrometer probe with delta-spectroscopy as well as sensors for ammonium, pH, redox and conductivity, was successful tested in the sewer system and inlet of a plant with 120’000 population equivalents as early warning system for the detection of industrial wastewater inhibiting or overloading biological treatment. Together with the

industrial register of the community specific industrial polluters could be identified and measures suggested improving WWTP operation, e.g. temporary storage of highly loaded wastewater and implementation of an ammonium and COD pollution fee based on average and peak loads to reduce overloading of the WWTP.

12. Acknowledgment

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