SIXTH FRAMEWORK PROGRAMME

Project no.: 018480

EUROMBRA

Membrane bioreactor technology (MBR) with an EU perspective for advanced municipal wastewater treatment strategies for the 21st century.

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PP	Restricted to other programme participants (including the Commission Services)	
RE	Restricted to a group specified by the consortium (including the Commission Services)	
CO	Confidential, only for members of the consortium (including the Commission Services)	

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1. Introduction

The present report collects and comments the main outcomes of the experimental and modelling activities carried out within the Work Package 4 of the EUROMBRA project, as far as concerns nitrogen and phosphorus removal. Particularly, the report focuses on the application of mathematical models at EAWAG and UNITN and the nutrient removal efficiency observed on large pilot-scale MBRs run at UNITN under different operational conditions. Finally, inferences form the long-term full scale experience of partner EV on N and P abatement are reported and discussed. In order to make it easily accessible, the document has been structured with a brief literature review centred on the most recent developments in MBRs in terms of nutrient removal; then, all experimental and modelling studies have been reported partner by partner according to a conventional paper structure including materials and methods, discussion of main results and conclusions.

2. Literature review

The improvement of effluent quality deriving from the installation of membrane bioreactors is generally thought to be related to the its capability to produce a bacteria-free effluent suitable for wastewater reuse. Indeed, early literature on membrane bioreactors already emphasized that operational modifications taking place into the biological process (e.g. longer sludge age) allow for a more effective removal of nitrogen, because of both more stable growth of nitrifying microrganisms and large volumes for biodegradable COD removal with nitrate as electron acceptor (Kishino *et al.*, 1996; Ueda *et al.*, 1996; Cote *et al.*, 1997; Ueda *et al.*, 1999). In such sense, an extremely attractive feature of the MBR technology in the 90’s literature highlights was represented by its possible adoption for upgrading wastewater treatment plants to biological nutrient removal, although being originally designed for COD removal (Buisson *et al.*, 1998). Among early studies on P-removal in MBRs, it was commented by Adam *et al.* (2002) that biological P-removal (Bio-P) in MBRs was not studied well before on account of experience with conventional activated sludge process at high sludge ages; however, few researches showed that effective Bio-P can occur in MBRs. Further was stated that this would be accounted for by intrinsic difference of biocenosis between the two types of activated sludge. Microscopic examinations promise high potential of Bio-P removal with MBR-sludge, given the observation of large poly-P granules within the bacteria community. Investigations have shown that P-uptake during starvation conditions is greater than in exponential growth phase (Ubukata and Takii, 1998). Microorganisms of high concentrated sludge have to survive under starvation conditions; bacteria that contain poly-P survive longer as a consequence of their accumulated energy source (hydrolysis of poly-P) and thus have an important advantage in the inter-species competition (Ubukata and Takii, 1998). While comparing the physicochemical P-removal with enhanced biological phosphorous removal (EBPR), the advantages of physicochemical precipitation on EBPR, to achieve enhanced phosphorous removal, are known to be better for control and reliability; however, many drawbacks are associated with physicochemical precipitation, such as up to 25 percent sludge production increase, additional chemical consumption, salinity increase of the effluent, and potential detrimental impact on the biological nitrification, due to resulting low alkalinity and pH (Lesjean *et al.*, 2003). Monti *et al.* (2006) compared a membrane enhanced biological phosphorous removal (MEBPR) process with a conventional EBPR (CEBPR) process under challenging operating conditions. Their investigation reveals relevant differences in terms of COD, nitrogen and phosphorous removal from municipal wastewater. Following the attempts made to couple membrane technology to the EBPR process in past researches (Adam *et al.*, 2002; Lesjean *et al.*, 2003; Fleischer *et al.*, 2005) and assuming that the EBPR mechanism is stimulated by circulating the activated sludge through anaerobic and anoxic/aerobic conditions which foster the growth of phosphorous-accumulating-organisms (PAOs) in the bioreactor, the comparative study carried out by Monti *et al.* (2006) demonstrated that a membrane process can reliably sustain satisfactory EBPR performance under the operational conditions which are considered to be challenging for conventional BNR plants. The continued study by Monti *et al.* (2007) attempted to probe into the impact of HRT and SRT on the removal of COD, nitrogen and phosphorous from municipal wastewater in MEBPR process. The research showed that when sufficient carbon source was present in the influent, the MEBPR process was shown to be capable of achieving excellent Bio-P removal performance at increasingly higher hydraulic loads, presenting significant technological advance beyond conventional BNR technology. Moreover, the complete solid-liquid separation afforded by the membrane permitted the generation of treated effluent of superior quality. A recent study in support of optimized N and P removal in MBR has been presented by Abegglen *et al.* (2008) concluding that the N and P removal seems promising under optimized operational conditions. Other researches (Ahmed *et al.*, 2008) probed into the microbial community structure for Bio-P in a MBR coupled BNR process with various types of C-source, giving positive sign of Bio-P.

With the intention of extending ASM1 to incorporate biological phosphorous removal, ASM2 (Henze *et al.*, 1995) was presented; subsequently, ASM2d was proposed by Henze *et al.* (1999), in order to incorporate two anoxic processes which account for the fact that, under anoxic conditions, PAOs use stored polyhydroxyalkanoates (PHA) for both growth and phosphorus accumulation (and consequent conversion into polyP). Exploring the insight of EBPR modelling, Barker and Dold (1997) merged the activated sludge model with other concepts presented by Wentzel *et al.* (1989) along with certain extensions and modifications incorporated in. After several investigations presented in many researches, the EAWAG Bio-P module for ASM-3 (Reiger *et al.*, 2001) presented a robust model incorporating recent advances in activated sludge modelling and biological phosphorous removal process understanding. Although recent researches (e.g. Hu *et al.*, 2007) have presented modified kinetic model for biological nutrient removal activated sludge systems, it seems that EAWAG Bio-P still stands reliable and capable of grossly presenting the assimilated process understanding of activated sludge process in general.

The need for energy cost minimization in a such-widely considered energy-consuming process like MBR technology has recently lead to a growing interest towards optimized aeration in the bioprocess tank, aimed at improving nitrogen and phosphorus removal through simultaneous nitrification-denitrification (SNdN) under low dissolved oxygen concentration values or cyclic anaerobic/anoxic/aerobic operation in single-tank processes. Although early applications of these approaches in biological processes coupled with membrane solid/liquid separation have been reported since mid-90’s (Yamagiwa *et al.*, 1995 de Silva *et al.*, 1998; Yeom *et al.*, 1999), they encountered a widespread application over last 5-6 years. Wang *et al.* (2005) investigated the effect of dissolved oxygen, C/N ratio and pH on the performance of a bench scale MBR in SNdN. The research pointed out that in such a relatively ecological stable system like a membrane bioreactor, even if a certain factor slightly fluctuates, the high concentration of sludge flocs can balance the possible disadvantageous effects on simultaneous nitrification/denitrification. For instance, the activity of aerobic heterotrophs becomes stronger with a slight increase of dissolved oxygen, leading to a higher consumption of DO and organic biodegradable substrate but, at the same time, to a reduction of the anoxic volume; in this case, the denitrification would be negatively influenced but the system still shows a certain capacity for simultaneous nitrification/denitrification. Moreover, the authors suggest that such change in environmental conditions would induce a overall adaptation of heterotrophic bacteria, *Nitrosomonas*, *Nitrobacter* and denitrifiers. Lim *et al.* (2007) evaluated the effect of operational parameters (mainly the duration of aeration ON and OFF phases and the duration of suction/relaxation periods) on BOD and nitrogen removal in a 5.5 L membrane bioreactors fed with domestic wastewater. While BOD abatement was steadily higher than 97% regardless the aeration ON/OFF time distribution, an anoxic phase longer than 70 minutes was needed in order to achieve a 80% removal efficiency for nitrogen (whole cycle duration: 120 minutes). The specific denitrification rate was 2.68 gNO_3-N gVSS h^{-1} when a 40 mins / 80 mins ON/OFF distribution was adopted, with lower values of 1.92 gNO_3-N gVSS h^{-1} and 1.05 gNO_3-N gVSS h^{-1} at 50/70 and 60/60 cycles respectively. At the same time, the authors observed a gradual increase of biological phosphorus removal with increasing duration of the anoxic phase. However, the increase of soluble EPS concentration under longer anoxic conditions reflected in more rapid permeability decrease even at the considered sub-critical flux (10 L m^{-2} h^{-1}). An innovative alternating anoxic/aerobic MBR has been proposed at pilot-scale fed with synthetic sewage to enhance nitrogen and phosphorus removal (Yuan *et al.*, 2008). The process consists in a continuously aerated compartment (tank A) and an anoxic/anaerobic zone including two separate reactors (tank B and tank C) which are alternatively fed with recirculation sludge from the aerobic tank; therefore, tank B and C are cyclically operated under anaerobic and anoxic conditions for phosphorus release and denitrification respectively. The authors reported a stable COD removal (over 93%), with 67.4% and 94.1% removal efficiency for nitrogen and phosphorus. Although being a time-based process (overall cycle duration: 120 minutes), the most significant parameter to describe the process dynamics was found to be the oxidation-reduction potential (ORP). A similar approach has been proposed by Fatone *et al.* (2008) who reported the results of two long-term experiences on a large pilot-scale and a full-scale membrane bioreactor. By combining the results of the two plants, the authors point out that the ORP- and DO-based automation adopted is able to adjust the aeration tot the biotank, in a nitrogen influent loading range of 0.05-0.18 kgN m^{-3} d^{-1} and with a C/N ratio in the feedwater higher than 6. when the membrane modules are immersed in the aerobic compartment (or in a separate dedicate one), the effect of residual dissolved oxygen recycled back to the anoxic/anaerobic tank can significantly affect nutrients removal. Kim *et al.* (2007) investigated this aspect on a full-scale MBR treating 210 m^3 d^{-1}, operated at a sludge age of 30 days; the introduction of a suitable de-oxidation tank to promote the depletion of dissolved oxygen in the recycle flow was found to be crucial to achieve nitrogen and phosphorus removal efficiencies higher than 70%, even at an overall HRT of 6 hours. More recently, attempts have been carried out to predict the performance of SNdN MBRs under different operational conditions by means of conventional activated sludge models. Sarioglu *et al.* (2009) run a pilot-scale MBR treating strong municipal wastewater and observed safe nitrogen removal under simultaneous nitrification/denitrification. In a 17.5÷21.0 kgMLSS m^{-3} range of suspended solids and under a sludge age of 36 days, the SNdN was found to take place steadily, the full denitrification and the partial nitrification being the rate limiting step fro nitrogen removal. Process optimization was then performed with an extended ASM1 including the anoxic decay of heterotrophs and autotrophs as depending on the usage of a final electron acceptor; besides, switch functions were introduced for all of the growth and decay processes in order to take into consideration the diffusion limitation of substrate and oxygen. Once calibrated, the model was used to evaluate the system behaviour under three different MLSS concentrations in the range 16 - 27 kgMLSS m^{-3} and under dissolved oxygen values between 0.5 and 3 g m^{-3}. The simulations indicated that a nitrogen removal efficiency higher than 95% could be obtained at MLSS higher than 16 kgMLSS m^{-3} and at low DO values (~ 1 g m^{-3}).

Finally, the optimization of nutrient removal via either simultaneous nitrification/denitrification or intermittent aeration has boosted the interest of scientists and practitioners in the application of new configurations. Kim *et al.* (2008) tested a new MBR scheme with the membrane module placed in an elevated position from the bottom of the reactor, thus dividing the process tank into an upper zone – where aeration is aimed at membrane cleaning - and a lower zone where denitrification and Bio-P take place. According to the authors, such schema also reflects positively in the membrane fouling control, since MLSS concentration in the upper zone is lower due to sludge settling in the bottom. A novel internal air-lift submerged MBR was proposed by Li *et al.* (2008) who compared two different bench-scale setups under four different operational conditions. The air-lift scheme was able to provide satisfactory removal efficiencies of total nitrogen, which was further improved when plastic carriers were added into the down-comer zone, possibly due to a stable anoxic micro-environment in the biofilm. Moreover, the increase of oxygen consumption under high MLSS concentration resulted in dissolved-oxygen gradients along the reactor depth, which was favourable to a more effective simultaneous nitrification/denitrification. The positive effect of combined MBR/MBBR (moving bed biofilm reactor) on nitrogen removal via SNdN and reduced membrane has been also described by Yang *et al.* (2009), which compared a conventional suspended growth MBR with a biofilm MBR.

3. Experimental activity

1.1 UNITN

Results obtained from two large pilot scale MBRs are presented and discussed; further details are reported in Guglielmi *et al.* (2007), Saroj *et al.* (2008), Guglielmi and Andreottola (2009).

The first experiments took place between September 2005 and August 2006 and refers to an external immersed MBR (Plant A) equipped with a flat sheet module and with a biological process compartment solely aimed at carbon oxidation and nitrification. The second case-study covers one year operation of a different external immersed MBR equipped with a hollow fiber 3-modules cassette (Plant B), the biological process being structured according to a conventional pre-denitrification scheme. Two different automation strategies have been applied for aeration supply in the aerobic compartment, the former based on the dissolved oxygen concentration the latter driven by the effluent ammonia nitrogen concentration; a comparative evaluation is therefore presented in terms of nutrient removal, sludge production and energy consumption. Moreover, data collected at Plant B during year 2 have been used for the application of a literature model for prediction of effluent nitrogen and phosphorus under different operational conditions at steady state.

1.1.1 Materials and methods

1.1.1.1 Experimental setups

Plant A was located at the municipal wastewater treatment plant in Lavis, Trento. It consists a oxidation/nitrification chamber (~14 m^3) and a membrane chamber (~11 m^3) in which a Huber VRM® 20/36 module is immersed; assuming a per-capita influent organic loading of 110 g COD PE^{-1} d^{-1}, the plant capability was approximately 200 PE. A schematic of the plant is shown in Figure 1.

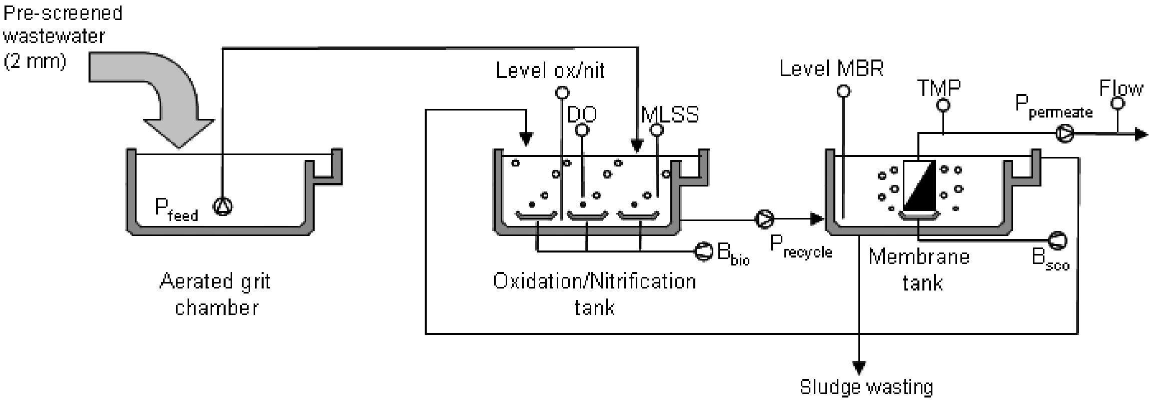


Figure 1: Flow-scheme of Plant A

Pre-screened (2 mm) wastewater is fed to the aerobic chamber by a centrifugal pump from the aerated grit chamber. A mohno-pump feeds sludge from the oxidation/nitrification chamber to the membrane tank. Another mohno-pump is utilised for permeate suctioning; sludge from the membrane tank was discharged back to the aeration chamber through an overflow discharge. Each progressing cavity pump is coupled with a frequency converter to get the desired flow-rate set by the user. Two blowers provide aeration for membrane cleaning (B_{sco} in the scheme of Figure 1) and oxygen supply to the biological process (blower B_{bio} in the scheme of Figure 1); the latter is driven by a frequency converter in order to ensure the desired set-point of dissolved oxygen concentration in the oxidation/nitrification tank. Both biological and hydraulic parameters have been continuously monitored, as permeate flow-rate, suction pressure, MLSS concentration, dissolved oxygen and temperature. All analogical inputs have been collected and stored in a data logger and the whole system was supervised by a SCADA-PLC. The Vacuum Rotating Membrane VRM 20/36 manufactured by Huber Technologies consists in a 36 membrane cartridges fit on a rotating hollow shaft (rotation speed 2 rpm) to form 6 circular elements, each with a membrane surface area of ~18 m^2. The membrane material is PES (polyetheresulphone) with a nominal pore size of 0.038 μm (ultrafiltration); the diameter of the module was ~2 m and the whole membrane surface area installed is 108 m^2. samples of influent wastewater, effluent wastewater and sludge have been analysed twice a week according to the Standard Methods (APHA, 1998). The average composition of the feedwater in the considered experimental period is shown in Table 1.

Table 1: Feedwater composition in September 2005- August 2006

Parameter	Concentration $(g\ m^{-3})$		
	average	max	min
Total COD	562	1504	175
Soluble COD	131	402	21
NH_4-N	42.3	197	11.9
TKN	65.5	237	19.8
NO_2-N	0.3	3.4	0.01
NO_3-N	1.9	9.2	0.1
TN	67.7	246.4	20.3
TP	7.5	19.1	0.9
TSS	292	1200	70

Intermittent permeate suction (10% relaxation regime) has been adopted, in order to improve the efficiency of aeration mechanical cleaning. Over the whole duration of the experimental activity, the average net permeate flow-rate has been around 1.4 $m^3\ h^{-1}$. Daily sludge wasting has been operated keeping a constant MLSS concentration of ~7700 gMLSS m^{-3} in the aeration chamber; this results in a solids retention time of ~20 days. The recirculation ratio ($Q_{recycle}/Q_{permeate}$) has been steadily kept at 5; due to the "concentration effect" operated by the membrane, the MLSS content in the membrane chamber is ~1.5 times the value in the biological process compartment. The air-sparging flow rate has been steadily kept at 0.35 $Nm^3\ m^{-2}\ h^{-1}$. No routinely chemical cleaning was carried out, rather soaking and backwashing operation were performed immediately before long-term and short-term trials for critical/sustainable flux assessment (see Guglielmi *et al.*, 2007).

Plant B is located at the same site and has been started-up on October 2006. It consists an anoxic compartment (4.7 m^3), an aerobic compartment for carbon oxidation and nitrification (8.7 m^3), a filtration tank (1.5 m^3) in which a 3-modules cassette of hollow fibre membrane is immersed (GE Zenon ZW500d®; ~100 m^2, nominal pores size: 0.04 μm). Influent wastewater is pumped to the anoxic compartment from the grit chamber after oil and sands removal and having been pre-screeened in a rotating drum fine screening (2 mm). All hydraulic parameters (suction pressure, water levels, permeate flowrate) are monitored and data collected every 30 seconds by means of a PLC-SCADA system.

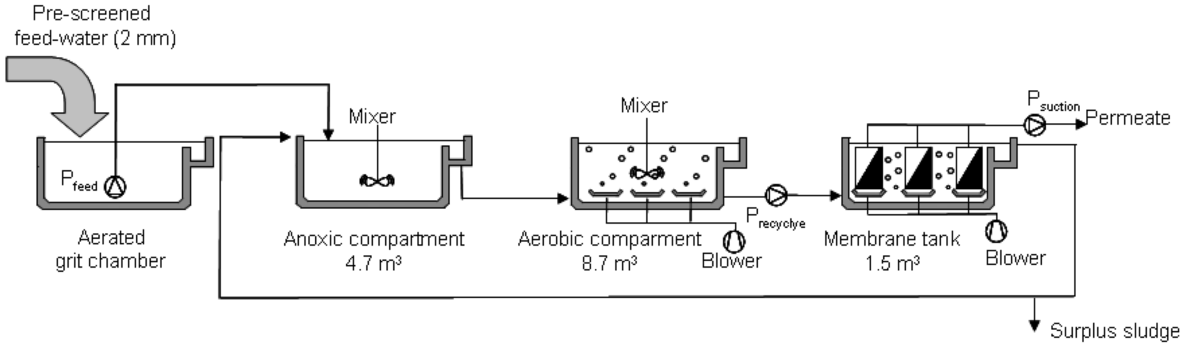


Figure 2: Flow-diagram of Plant B

On-line measurement of dissolved oxygen, mixed liquor suspended solids in aerobic and membrane tank, effluent ammonia nitrogen and effluent nitrate were also performed. A weekly calibration was carried out on the ammonia nitrogen probe, while a monthly check was used for the nitrate probe. Aeration to the biological process tank was supplied according two different schemes. The default aeration strategy consists in switching ON and OFF the blower according to the dissolved oxygen setpoint defined by the user, which was kept at 1÷2 $gO_2\ m^{-3}$; a dedicated frequency converter tune the air flowrate according to parameters of the PID cycle set by the user. This approach was used during the so-called period 1 (January-May 2008). Since June 2008 and until December 2008, a different aeration strategy was implemented in the aerobic compartment, according to which the blower is switched on when effluent ammonia reaches 3 $gN\ m^{-3}$ (high setpoint) and switched off when it drops down to 2 $g\ m^{-3}$ (low setpoint); sludge mixing was ensured by a submerged mixer installed in the oxidation tank and no control of dissolved oxygen was adopted. During both periods, influent wastewater was collected as 24-hours samples twice a week and the value of most important macro-pollutants are reported in Table 2. The fractionation of influent COD was periodically assessed on 24-hours samples stored at 4°C, according to respirometric techniques proposed by Ekama *et al.* (1986) and Ziglio *et al.* (2001), for biodegradable COD and readily biodegradable COD. The tests revealed a readily biodegradable COD fraction ranging between 15 and 22% of total COD, whereas the particulate biodegradable COD was generally in the range 45-60%; inert COD was mostly around 20-30%, the particulate fraction being 15-20% of total COD and the soluble fraction being 5-8% of total influent COD. Over the whole experimental activity, MLSS in the biotank has been kept at ~7.5÷8 $kgMLSS\ m^{-3}$, by means of daily sludge wasting; this results in a sludge age of approximately 20 days during period 1 and 23 days during period 2. Due to the recirculation ratio of about 2.5÷3.5 from the membrane compartment to the anoxic one, the concentration in the membrane tank has been typically around 10÷11 kgMLSS m^{-3}. During both period 1 and period 2 the membrane has been operated under sub-critical flux conditions, the normalized permeate flux J_{20} ranging between 12 and 20 L m^{-2} h^{-1}. Permeate suction has been operated according to a 10% relaxation regime (9 minutes suction and 1 minute pause). Aeration for membrane cleaning has been supplied continuously at a SAD_m value of 0.5 Nm^3 m^{-2} h^{-1}.

Table 2: Influent wastewater composition during period 1 and period 2 at Plant B

Parameter	Unit	Period 1 (Jan 2008 – May 2008)	Period 2 (June 2008 – Dec 2008)
COD	g m^{-3}	440.9 ± 165.7	566.2 ± 316.0
Soluble COD*	g m^{-3}	95.2 ± 51.6	140.5 ± 123.3
NH_4^+-N	g m^{-3}	32.6 ± 18.2	30.4 ± 12.0
TKN	g m^{-3}	51.9 ± 28.9	49.3 ± 17.2
NO_2^--N	g m^{-3}	0.5 ± 0.4	0.5 ± 1.1
NO_3^--N	g m^{-3}	2.4 ± 1.4	2.5 ± 4.1
PO_4^{3-}-N	g m^{-3}	2.2 ± 1.2	2.1 ± 1.5
TP	g m^{-3}	7.1 ± 4.8	8.1 ± 3.7
TSS	g m^{-3}	238.8 ± 324.1	325.9 ± 126.7
VSS/TSS	-	0.87 ± 0.05	0.82 ± 0.06
Temperature in the biotank	°C	18.1 ± 3.3	20.3 ± 5.2

1.1.1.2 Modelling

The simulation study presented here corresponds to a pilot-scale MBR previously mentioned as Plant B. The experimental data used for EAWAG Bio-P application are those collected during year 2007; influent wastewater composition and operational conditions are summarized in Table 3.

Table 3: Data used for the application of model EAWAG Bio-P on plant B

Parameter	Values
COD, g m^{-3}	610±151
NH_4 (as N), g m^{-3}	48.8±24
TKN (as N), g m^{-3}	89.2±50
TSS, g m^{-3}	313±220
Total P (as P), g m^{-3}	8±3
PO_4 (as P), g m^{-3}	3.3±2
SRT, d	~25
MLSS, kg m^{-3}	~7.5
MLVSS/MLSS	0.78

The EAWAG Bio-P module seeks the introduction of four new state variables viz. inorganic soluble phosphorous (S_{PO4}, gP m^{-3}), phosphorous accumulating organisms i.e. PAOs (X_{PAO}, gCOD m^{-3}), cell-internal storage product of PAOs (X_{PHA}, gCOD m^{-3}) and Polyphosphate (X_{PP}, gCOD m^{-3}). It is described by means of 11 process equations assimilating the whole concept of enhanced biological phosphorous removal (EBPR) in mathematical terms (Reiger *et al.*, 2001); the relevant kinetic parameters have been adopted from Reiger *et al.* (2001). The Activated Sludge Simulation program (ASIM) is a didactic program which allows implementing several biokinetic models used for the simulation of activated sludge systems. It allows the development of the biokinetic model for constant volume batch and continuous flow biological growth systems under steady state and dynamic loading conditions. ASIM offers following major options:

- Definition of biokinetic models- definition of soluble and particulate compounds, number of transformation processes, stoichiometry and kinetics of these processes
- Definition of process control strategies
- Computational solutions to find steady state solutions as well as dynamic solutions.
- Graphic support of the analysis of simulation results as well as experimental data.
- Communication of with spread sheet programs

The EAWAG Bio-P module was coded into ASIM and for the implementation of MBR configurations three compartments have been defined (corresponding to schematic in Figure

2) viz. one without oxygen (anoxic/denitrification), next one with fix oxygen set-point (aerobic/nitrification) and third one as an ideal clarifier (membrane chamber). The ASIM allows changing the oxygen set-point as desired for the simulation, and variation in operational parameters viz. SRT and recirculation ratio; the recirculation is defined with complete separation of particulate species.

1.1.2 Results

1.1.2.1 Plant A

Trends of influent and effluent COD, ammonia nitrogen, total nitrogen and total phosphorus are shown in Figure 3: Influent and effluent COD for Plant A, Figure 4, Figure 5 and Figure 6 respectively.

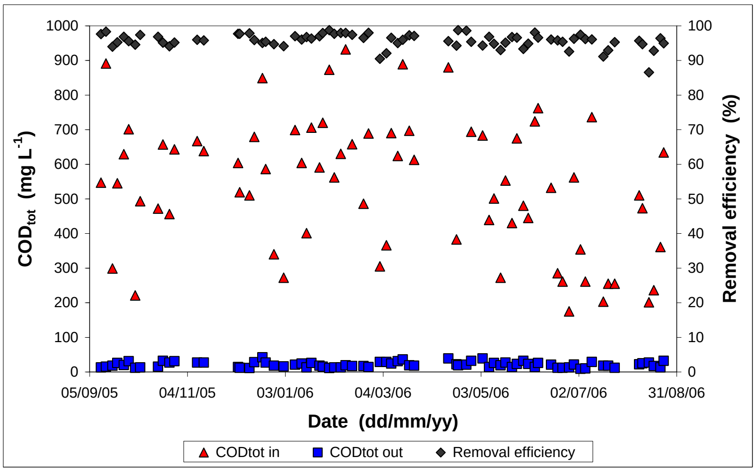


Figure 3: Influent and effluent COD for Plant A

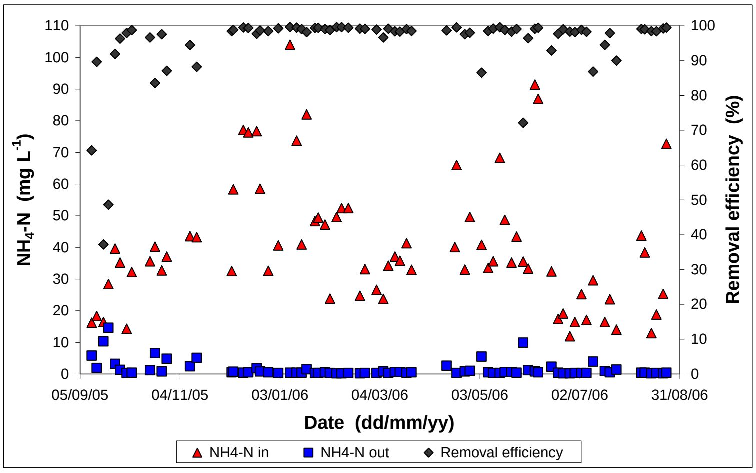


Figure 4: Influent and effluent NH4-N for Plant A

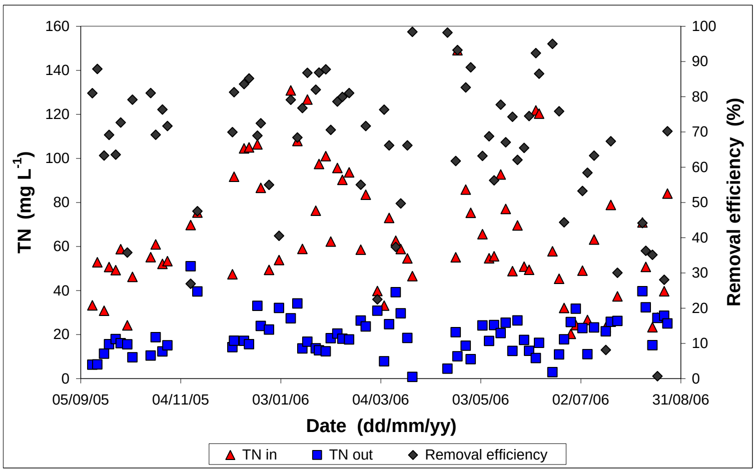


Figure 5: Influent and effluent total nitrogen for Plant A

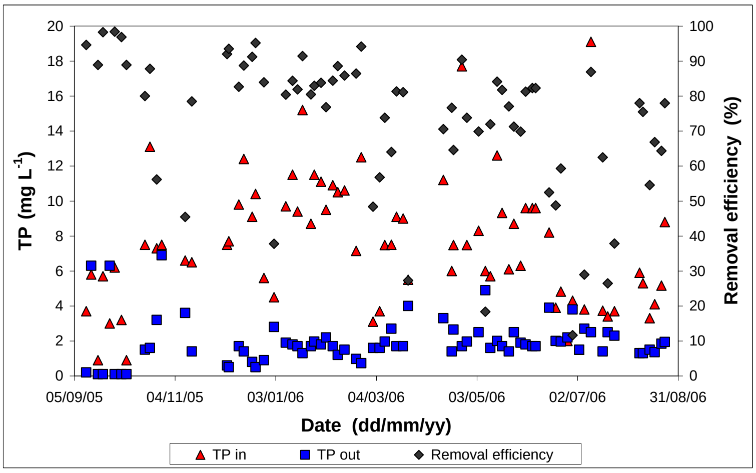


Figure 6: Influent and effluent total phosphorus for Plant A

The removal efficiencies referred to the whole experimental period are shown in Table 4; as expected, performances in terms of COD and ammonia nitrogen removal were excellent regardless the influent loadings, therefore confirming most of previous studies on MBRs mentioned in the section 2.

Table 4: Permeate characteristics for Plant A

Parameter	Concentration (g m^{-3})		
	Average ± σ	min	max
Total COD	21.1 ± 7.8	9.0	42.0
Soluble COD	15.9 ± 6.6	6.0	36.0
NH_4-N	1.5 ± 2.6	0.2	14.6
TKN	2.8 ± 3.4	0.4	20.8
NO_2-N	0.1 ± 0.2	0.0	1.4
NO_3-N	16.8 ± 10.2	0.1	48.3
TN	19.7 ± 9.3	0.8	51.0
TP	2.0 ± 1.3	0.1	6.9
TSS	1.0 ± 0.2	0.7	1.4

It was quite unexpected to see that total nitrogen removal was pretty high, although no denitrification tank was present. In order to investigate the possible degradation pathways, the mass balance equation under steady-state conditions was considered:

$$Q_{in} \cdot N_{in} - Q_{out} \cdot N_{out} - \Delta N_{den} - Q_w \cdot f_N \cdot MLVSS_w = 0 \qquad \text{Equation 1}$$

where:

Q_{in}, Q_{out} and Q_w are the mean daily influent, effluent and sludge wasting flow rates respectively (m^3 d^{-1});

N_{in} *and* N_{out} are the sum of all nitrogen forms in influent and effluent wastewater (g m^{-3});

f_N is the nitrogen content per unit of volatile suspended solids in the wasted sludge, experimentally measured (0.11 gN gVSS^{-1} on average);

$MLVSS_w$ is the volatile suspended solids concentration in the wasted mixed liquor (gVSS m^{-3}). Considering data collected during steady-state operation (~ 2 times the operational SRT), the mass balance was never closed with a ΔN_{den} value up to 52%. This was explained as a local effect of simultaneous nitrification-denitrification in the membrane tank which corresponds to almost 50% of the whole process volume; such hypothesis is also confirmed by the DO concentration values (2.0 ± 0.5 g m^{-3} in the oxidation/nitrification chamber and 0.8 ± 0.5 g m^{-3} in the membrane chamber). Therefore, the results obtained show that the biological activity taking place in the membrane tank should be considered as crucial player for the nitrogen removal process, especially in the case of MBR using flat-sheet membrane whose lower specific membrane surface area (installed membrane area/bulk volume of the module) compared to hollow fibre (HF) systems often results in a large process volume with lower oxygen transfer and more suitable conditions for nitrate reduction to gaseous nitrogen.

1.1.2.2 Plant B

The profiles of main macropollutants in feedwater and permeate (24-h samples) during period 1 and period 2 are shown in Figure 7 - Figure 16.

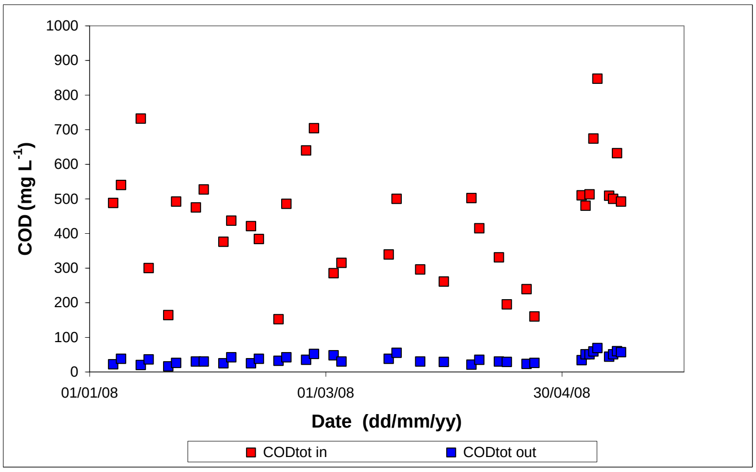


Figure 7: Influent and effluent COD for Plant B during period 1

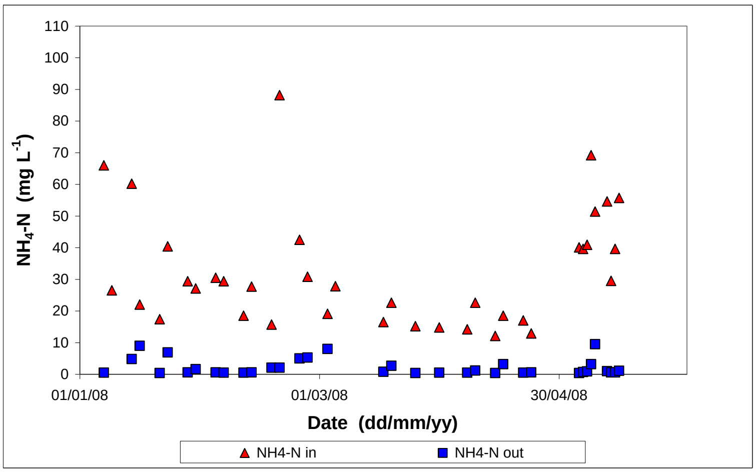


Figure 8: Influent and effluent ammonia nitrogen for Plant B during period 1

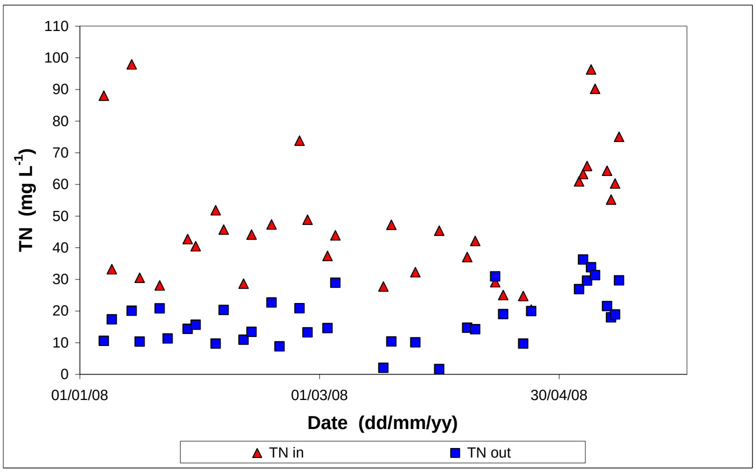


Figure 9: Influent and effluent total nitrogen for Plant B during period 1

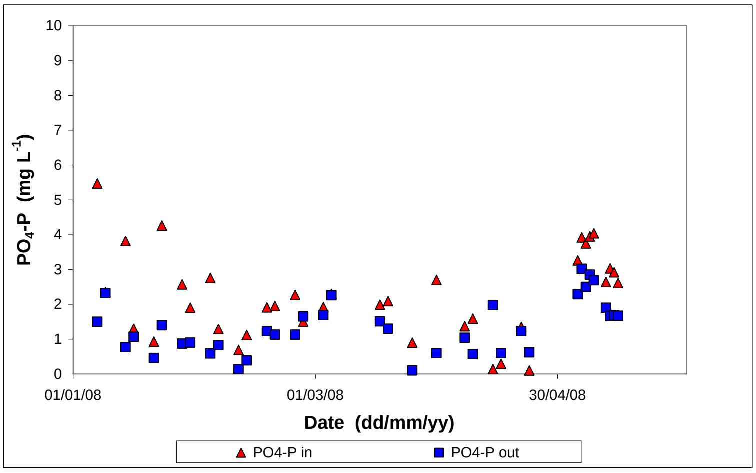


Figure 10: Influent and effluent orthophosphates for Plant B during period 1

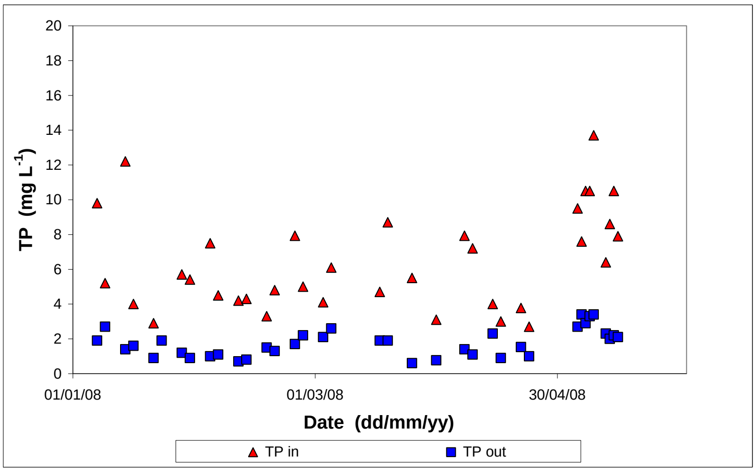


Figure 11: Influent and effluent total phosphorus for Plant B during period 1

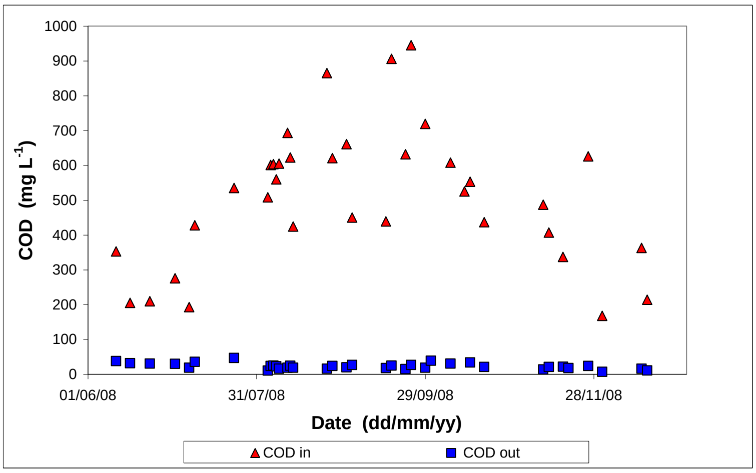


Figure 12: Influent and effluent COD for Plant B during period 2

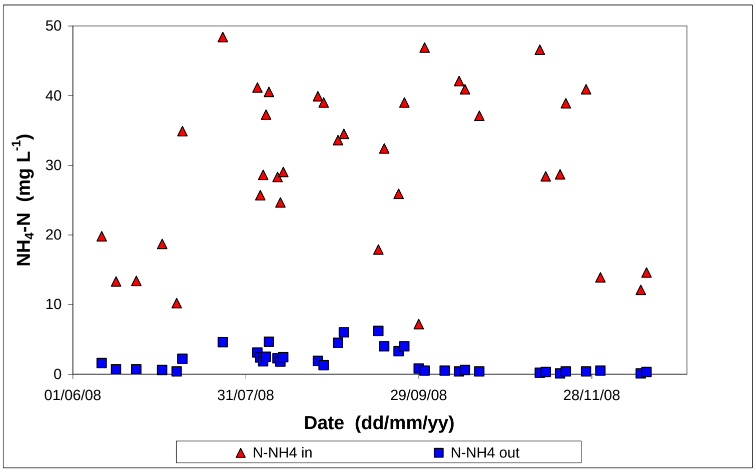


Figure 13: Influent and effluent NH_4-N for Plant B during period 2

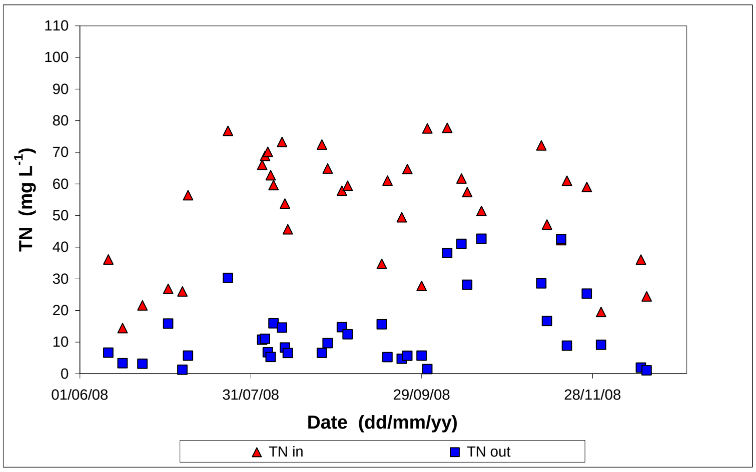


Figure 14: Influent and effluent total nitrogen for Plant B during period 2

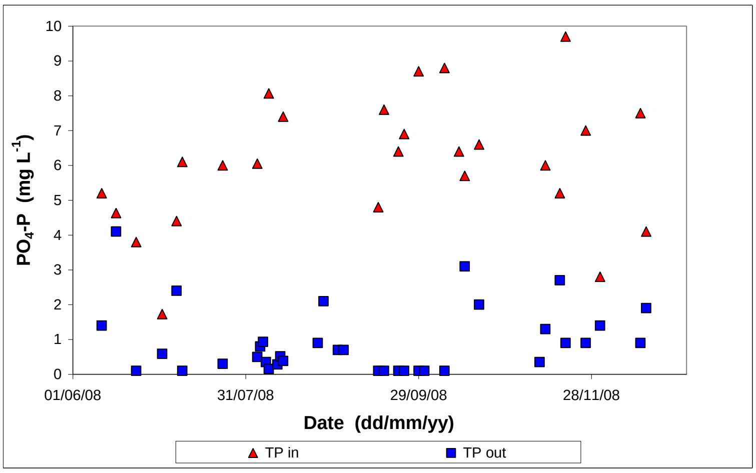


Figure 15: Influent and effluent orthophosphates for Plant B during period 2

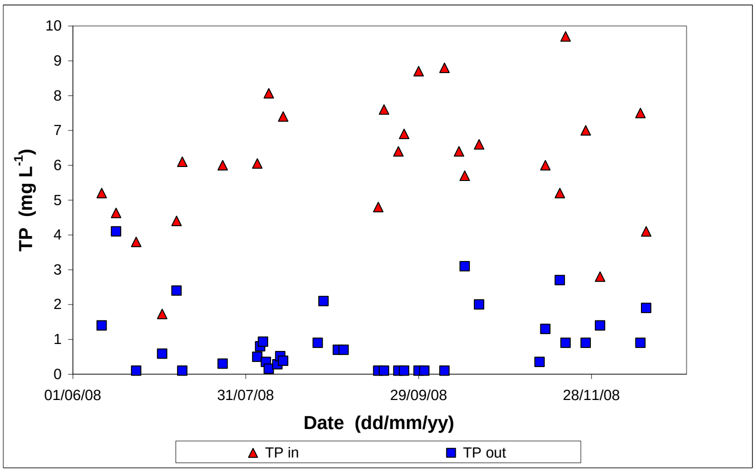


Figure 16: Influent and effluent phosphorus for Plant B during period 2

The overall comparison between the two operational strategies proposed in terms of effluent quality is reported in Table 5. A complete nitrification was steadily achieved with no nitrite accumulation in the system during both period 1 and period 2. A higher denitrification efficiency was observed during period 2, as a consequence of the longer duration of the anoxic phase, which allows for a more effective utilization of the denitrification potential of the system. This lead to a more complete utilization of the electron donor (biodegradable carbon), thus giving a higher removal efficiency of COD (91.5% for period 1 and 95.9% for period 2). Consequently, the removal efficiency of total nitrogen increased from 67.3% to 74.2%. Interestingly, a better biological phosphorus removal was achieved during period 2, mainly due to the creation of more favourable conditions for PAOs and dPAOs (denitrifying PAOs) in the anoxic compartment which cyclically go under anaerobic conditions. This is clearly highlighted by carrying out a steady state mass balance on phosphorus, according to the following equation:

$$Q_{in} \cdot TP_{in} = Q_{out} \cdot TP_{out} + Q_w \cdot MLSS_w \cdot \left(\frac{MLVSS}{MLSS} \right)_w \cdot f_P \qquad \text{Equation 2}$$

Where:

Q_{in}, Q_{out} and Q_w are the influent, effluent and surplus sludge flowrates respectively ($m^3\ d^{-1}$), TP_{in} and TP_{out} are the concentrations of total phosphorus in influent and effluent streams, and f_P is the specific phosphorus content per unit of VSS in the wasted sludge (gP $gVSS^{-1}$).

By using the experimental and operational data, an increase of 20% of the parameter f_P can be calculated over the period 2 compared to the period 1 (0.06 gP $gVSS^{-1}$ versus 0.05 gP $gVSS^{-1}$), which would indicate a higher propensity to bio-P removal for biomass under cyclic aerobic/anoxic conditions. This is in fair agreement with other experimental observations reported by Mehinold *et al.* (1999) and Hu *et al.* (2002) for conventional activated sludge processes. Moreover, microbiological studies recently reported on sludge samples collected during both periods (Silva *et al.*, 2009) indicate an increase of phosphorus accumulating microrganisms in non-EBPR membrane bioreactors assessed by means of FISH techniques.

Table 5: Permeate characteristics during period 1 and period 2

Parameter	Unit	Period 1 (Jan 2008 – May 2008)			Period 2 (June 2008 – Dec 2008)		
		Av. ± σ	Min	Max	Av. ± σ	Min	Max
COD	$g\ m^{-3}$	37.3 ± 13.1	16.0	59.0	23.4 ± 8.7	7.0	57.0
Soluble COD*	$g\ m^{-3}$	28.7 ± 10.0	13.0	56.0	19.4 ± 7.7	6.0	44.0
NH_4^+-N	$g\ m^{-3}$	1.2 ± 1.6	0.4	9.5	1.9 ± 1.7	0.1	6.2
TKN	$g\ m^{-3}$	3.3 ± 4.1	0.7	27.3	3.2 ± 2.6	0.2	14.1
NO_2^--N	$g\ m^{-3}$	0.1 ± 0.1	0.01	0.43	0.1 ± 0.2	0.01	0.6
NO_3^--N	$g\ m^{-3}$	13.5 ± 9.0	0.1	33.8	10.4 ± 2.7	0.1	42.3
PO_4^{3-}-N	$g\ m^{-3}$	1.4 ± 0.8	0.1	3.0	0.6 ± 0.7	0.1	3.0
TP	$g\ m^{-3}$	1.8 ± 0.9	0.1	3.8	0.9 ± 0.9	0.1	4.1

Typical DO patterns of dissolved oxygen, ammonia nitrogen and nitric nitrogen during period 1 and period 2 are depicted in

Figure 17. The longer duration of denitrification which was extended to the whole biological process volume resulted in a lower growth rate for heterotrophic biomass. In terms of

observed yield coefficient a 12.6% reduction was reported from the period 1 (daily sludge production: 3.97 kgMLSS d^{-1}; 0.34 kgMLSS $kgCOD_{rem}^{-1}$) to the period 2 (4.54 kgMLSS d^{-1}; 0.29 kgMLSS $kgCOD_{rem}^{-1}$). Although a higher daily production was observed, the specific excess sludge production decreased as a consequence of the more performing removal efficiency of biodegradable COD, which is also confirmed by the effluent data (see Table 5).

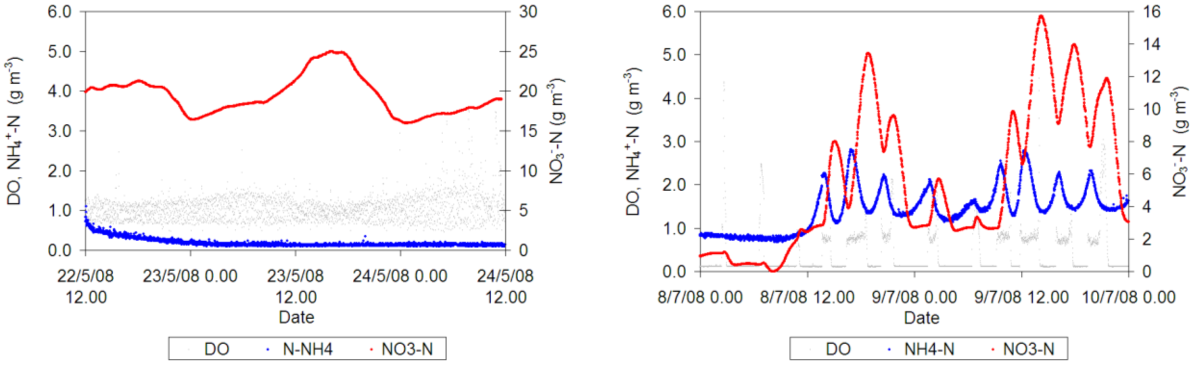


Figure 17: Daily transients of DO, NH_4-N and NO_3-N
during the DO-controlled aeration period (left hand side) and the NH_4-controlled period (right hand side)

No direct measurement of absorbed energy was available on site; however, a duration of the switch-on and switch-off periods was possible by interpreting the data collected by the on-line thermal flowmeter along the air supply pipeline of the aerobic compartment (data collection frequency: 30 seconds). Compared to the period 1, during which aeration was steadily on at variable frequency depending on the oxygen need, extended non-aeration phases were reported in period 2, resulting in a switching-off phase of about 50%, which then reflects in a similar energy saving for biological process aeration.

1.1.2.3 Modelling

The results of mathematical simulations for nitrogen and phosphorus removal at Plant B during year 2007 been reported by Saroj et al. (2008). In all the simulation trails, the 100 day simulation was assumed sufficient for reaching the stabilized conditions of the biological process. The ASIM allows simulation trials to show the 100 d steady-state values; generally, the numerical values corresponding to effluent parameter for nutrients (ammonia, phosphate) at the steady state simulations remain stable after less than 2 months of selected simulation period. The simulation results of the effects of various operational parameters on the quality of treated effluent in terms of nutrients (i.e. nutrient removal evaluation) are presented in Figure 18.

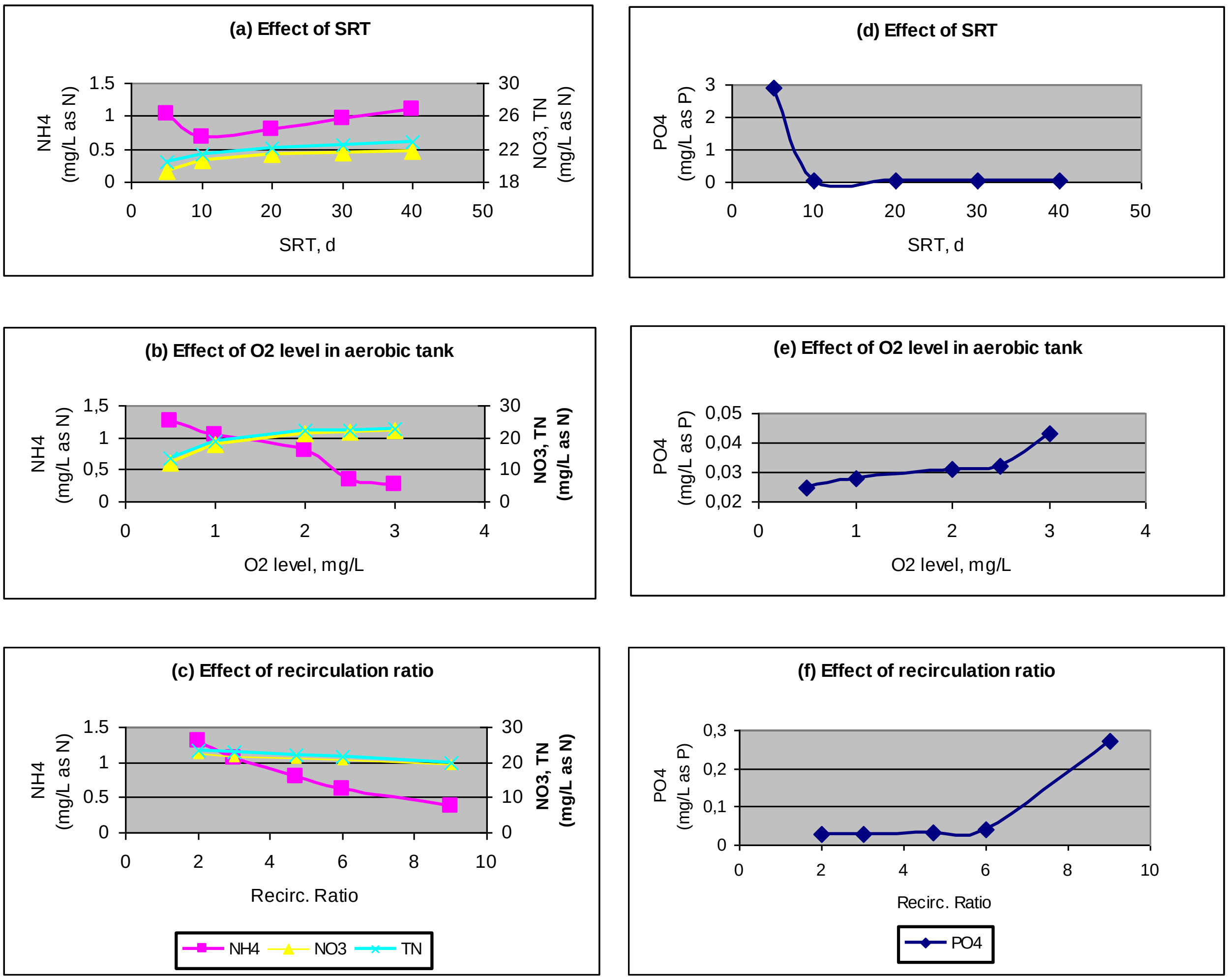


Figure 18: Simulation results of the effect of SRT (a, d), DO set-point level in aerobic/nitrification tank (b, e) and recirculation ratio (c, f) on effluent NH_4, TN and PO_4.

The simulation of the effect of SRT for PO_4 removal is strongly evident at lower SRT (poor biological PO_4 removal at SRT less than 10 days) and remains less varying at SRTs as high as 40 days. Sludge ages of more than 10 days seem good for NH_4 removal, and continues to be good till SRT of as high as 40d. Therefore SRT ranging 15-30 is sufficient considering both nitrogen (NH_4 and TN) and biological phosphorous removal. The actual SRT of 25 days seems good and an optimum selection, though, can be extended slightly longer aiming at the sludge reduction. Further, the effect of DO set-point (in aerobic/nitrification tank) is clearly evident both for NH_4 and PO_4 removal. As far as NH_4 removal is concerned, the DO of 1mg/L seems sufficient, although <1mg/L NH_4 (as N) would seem to require slightly higher DO set-point. When DO is more than 2.5 mg/L, the PO_4 removal starts getting drastically hampered. Therefore, the optimum DO set-point value seems important both for NH_4 and PO_4

removal. Remarkably, the effect of DO set-point on NH_4 and PO_4 removal trend is opposite; the reconciliation between desired extent of NH_4 and PO_4 removal would decide the optimum and appropriate DO set-point. The actual DO set-point of 2mg/L for the mentioned MBR seems a good choice reflecting a balanced compromise. Furthermore, the simulation results pertaining to the effect of recirculation on NH_4 and PO_4 removals demonstrate the similar trends as in the case of DO set-point effect. In fact, the higher recirculation ratio causes the input of higher oxygen to anoxic tank which is equivalent to having higher DO set-point. The increase in recirculation gives positive impact on NH_4 removal while reverse impact on PO_4 removal. Even if the recirculation ration of about 2-3 appears sufficient for NH_4 and TN removal, the higher ratio leads to improved NH4 removal. Again, similar to DO set-point variation effect, the recirculation ratio of more than 6 results in sudden hampering effect on PO_4 removal, and leads to only slight improvement in NH_4 removal.

1.1.3 Conclusions

Both experimental setups run at UNITN achieved steadily high nitrogen removal efficiency. In plant A, although no specific anoxic compartment was installed, a simultaneous nitrification/denitrification took place in the membrane chamber whose volume is comparable with the one of the oxidation/nitrification tank. Being membrane scouring aeration optimized for fouling prevention rather than for oxygen supply, the dissolved oxygen concentration was steadily around 1 g m^{-3}, which resulted in a limited diffusion into the floc structure and therefore favourable conditions for SNdN.

An effective process optimization for nutrient removal was investigated at Plant B. Compared with the conventional pre-denitrification scheme, the alternation of aerobic and anoxic phases in the oxidation/nitrification compartment improved the flexibility of the system, resulting in higher efficiencies for nitrogen and COD removal. Moreover, an improvement in biological phosphorus removal was recorded and a slight decrease of sludge production due to the reduction factor on heterotrophic growth kinetics was observed. Almost 50% of energy consumption due to bioprocess aeration was saved by adopting the intermittent aeration process; however, in order to further reduce energy consumption in membrane bioreactors, an integrated approach to aeration optimization should be developed considering both membrane aeration and nitrification/denitrification and mixed liquor recirculation.

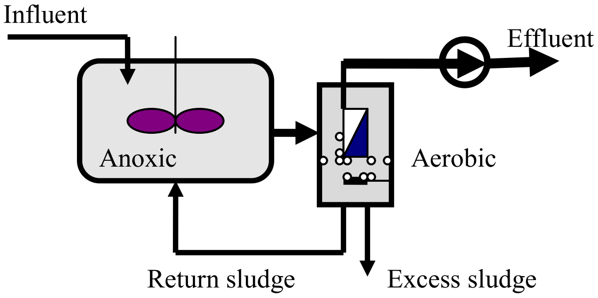
Concerning the application of mathematical models to predict the process performances, the simulation results pertaining to ASM3-BioP for mentioned MBR give interesting information on the effect of key operational parameters on treatment capacity. Such simulation results clearly show that the optimum set of operational parameters should be selected based upon the desired level of nutrient removal associated and energy expenses; the optimum set of operational parameters pertain to several preferences as they might conflict on the choice of operational parameters, e.g. the optimum recirculation ration for both NH_4 and PO_4 removal. Noticeably, the optimization here has been considered on the basis of nutrient removal only, however, in real situation, other limitations might influence. For example, even if SRT of 15 d is good for nutrient removal, the sludge production as compared with 25 days of SRT situation would be high where the nutrient removal would still be good.

1.2 EAWAG

Observations from a laboratory scale MBR at EAWAG (Dübendorf, Switzerland) fed with municipal wastewater (Dübendorf locality) were used for the study of biological phosphorous removal and nutrient removal.

1.2.1 Materials and methods

The pilot MBR was intended to treat 3.1 person equivalents (based on N load) and the total specific volume is typical for municipal plant (120 L PE^{-1}). The pilot setup consists of two main compartments viz. anoxic tank and aerobic tank. The complete mixing condition was maintained in the anoxic compartment and the continuous aeration condition was maintained in the aerobic compartment. The influent was received by anoxic tank and the membrane module was immersed in the aerobic tank (Figure 19). The aerobic tank contained 4 Zenon ZW10 units immerged. The aerobic tank received the flow from anoxic tank and a return sludge was pumped back allowing recirculation for nitrification-denitrification. The specific features of the treatment configuration were high anoxic to aerobic volume ratio and a high recirculation ratio. The whole system was continuously operated with PLC and SCADA supervision.



(a)

(b)

Figure 19: Small pilot-scale unit run at EAWAG for WP4

The operational details pertinent to MBR-IV are listed in Table 6; general (macro-pollutant removal) efficiencies of concern are enlisted together.

Table 6: Operational conditions of the experimental unit tested at EAWAG

PARAMETERS	UNIT	VALUE
$V_{aerobic}$	m^3	0.154
V_{anoxic}	m^3	0.218
Inflow	$m^3\ d^{-1}$	1.1
Return sludge	$m^3\ d^{-1}$	11.5
Excess sludge	$m^3\ d^{-1}$	0.016
$MLSS_{aerobic}$	$kg\ m^{-3}$	8.8
$MLSS_{membrane\ tank}$	$kg\ m^{-3}$	7.3
HRT	h	~8
SRT	d	~ 21
COD:N	-	~ 15
COD:P	-	~ 84
Temperature	^0C	18-19
Ammonia removal	%	97
Total-N removal	%	87
Total-P removal	%	90

For the implementation of MBR system configuration into ASIM, two compartments have been defined (corresponding to schematic in Figure 19) viz. first one without oxygen and second one with defined oxygen set-point. Two types of simulation can be carried out, one directly reaching to steady state solution and other one showing dynamic variation of any of the variables. Here, only steady state solutions giving final stabilized values have been recorded; a 100 days simulation was assumed sufficient for reaching stabilized condition with respect to biological processes; generally, the numerical values corresponding to effluent parameter for nutrients at the (ammonia, phosphate) steady state simulations remain stable after less than 2 moths of selected simulation period.

1.2.2 Results

The simulation results (Figure 20) for the effect of SRT show that unless the SRT is quite low (<10d), the effluent quality in terms of low effluent PO_4-P is promised up to SRT as high as 40d. However, very high SRT seems slowly affect the removal. The real experimental condition, which was at the SRT of 21 days, seems good selection. The nitrogen removal dependence on SRT appears similar.

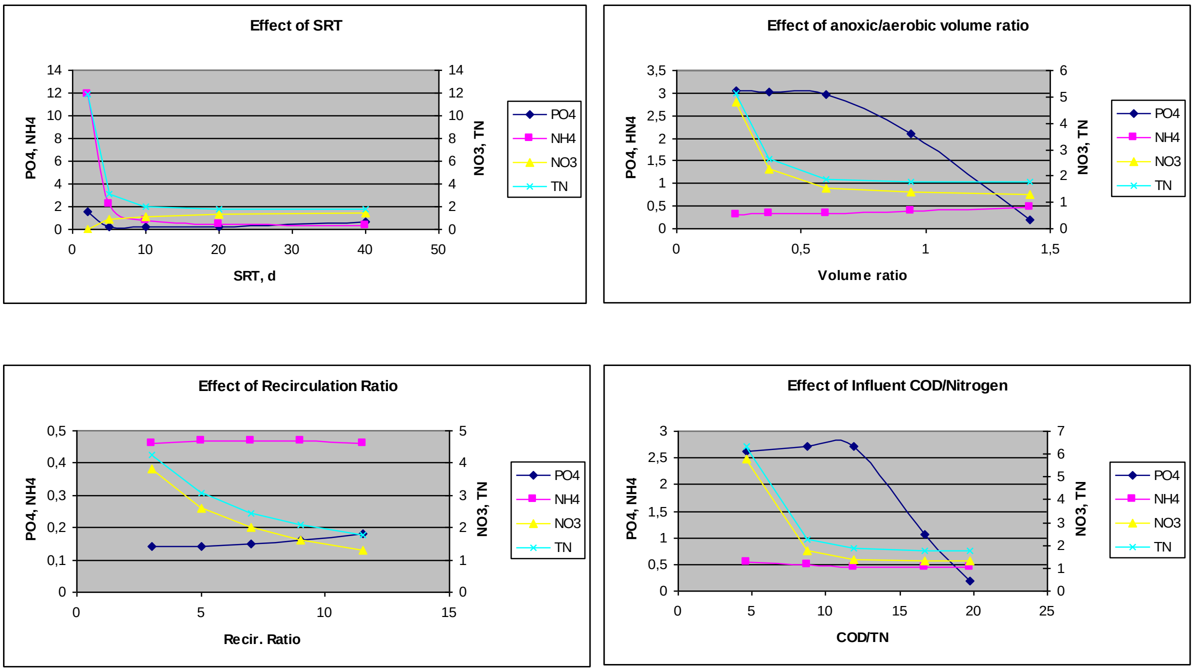


Figure 20: Simulation results of nutrient removal in MBR at EAWAG (concentration values in g m^{-3})

The effect of recirculation ratio seems not very strong for phosphorous but seems causing slight negative effect; in general, the recirculation ratio of more than 3-4 appears sufficient. The effect of COD: N ratio seems important as it influences strongly the phosphorous removal. Even though the ratio of more than around 5 appears sufficient for good ammonia or total nitrogen removal with increase in removal with increase in the ratio, phosphorous removal becomes drastically improved from around 12 to 20. Further, the effect of anoxic: aerobic volume in nitrification-denitrification system seems very important as per the simulation results. The real case of 1.5 (anoxic volume/aerobic volume) appears an excellent selection for the nutrient removal in general and phosphorous removal in particular; this observation, after detailed study has strong potential towards leading to better design of MBRs for enhanced biological phosphorous removal. In general, the simulation results comply well with the real data observed at particular values of operational parameters.

1.2.3 Conclusions

The modelling study of nutrient removal in general and biological phosphorous removal in particular provided noticeable outcomes in terms of the effect of several parameters. The unintended and good phosphorous removal in pilot MBR has been reasoned here on account of peculiar configuration in terms of unusually high anoxic/aerobic volume ratio, high recirculation ratio and an exceptionally high COD/Nutrient ratio of influent wastewater, and/or presumably, specific biology of MBRs favouring enhanced growth of pertinent micro-organisms; more focused research is required in this direction.

1.3 EV

Since 1999 the Erftverband has commissioned three full scale membrane bioreactor plants: Rödingen WWTP (1999); Nordkanal WWTP (2004) and Glessen WWTP (2008). In membrane bioreactors the hydraulic retention time can be relatively short compared to conventional wastewater treatment plants. For this reason they have been considered poor in buffering and eliminating peak loads. The actual treatment performance of the Nordkanal MBR was surveyed. Attention was paid to the nitrification step that can be especially sensitive to load variations. The results were compared with operational experiences from the two other large scale MBRs.

1.3.1 Materials and methods

All three MBR plants have been designed as contact stabilization plants with a minimum sludge retention time SRT of 25 days and employ submersed hollow fibre ultrafiltration membranes. Table 7 shows the key technical data of the three MBRs; the average daily inflow values are given for the year of 2007. All plants are connected to combined sewer systems. The average yearly inflow differs slightly from year to year depending on the amount of rainfall. The maximum daily inflow is set by the capacity of the plants pumping stations. For the calculation of the hydraulic retention time HRT the volume of the filtration tanks and the bioreactors are considered.

Table 7: Key technical data

	Rödingen, 3 000 PE	**Glessen, 9 000 PE**	**Nordkanal, 80 000 PE**
Bioreactor	400 m³ + 80 m³ (filtration tanks)	1600 m³ + 320 m³ (filtration tanks)	9300 m³
Process configuration	alternating or upstream denitrification	alternating or simultaneous denitrification	upstream denitrification
Membrane	5280 m², 2 separate filtration lines	12096 m² , 4 separate filtration lines	84480 m², 8 integrated filtration lines
Design SRT	25 d	25 d	25 d
Inflow			
Maximum	3240 m³ d^{-1}	5280 m³ d^{-1}	48000 m³ d^{-1}
Average	540 m³ d^{-1} (2007)	2250 m³ d^{-1} (since April 2008)	15270 m³ d^{-1} (2007)
HRT			
Minimum:	3.6 h	8.7 h	4.7 h
Average:	21.3 h	20.5 h	14.6 h
Effluent requirements	COD < 30 g m^{-3} NH_4-N < 4 g m^{-3} P_{tot} < 1.5 g m^{-3}	COD < 30 g m^{-3} NH_4-N < 1.5 g m^{-3} P_{tot} < 0.6 g m^{-3}	COD < 90 g m^{-3} NH_4-N < g m^{-3} P_{tot} < 1.5 g m^{-3}

Figure 21 shows the process flow diagram of the plant: the three-stage pretreatment (screen, aerated sand chamber with fat trap, sieve), the flow distribution and the four bioreactor lines. A pair of filter lines is directly inserted in each of the nitrification tanks. The filtration modules are installed on the side walls of the reactor slightly below the water line. The reactors are 4.5 meters deep and equipped with fine bubble aerators for oxygen supply. The coarse bubble aeration of the membrane filters is only 2.8 meters deep. The filters occupy about one fifth of the nitrification volume. Four horizontal agitators are installed in every basin and induce a circular current below the filtration units. Phosphorous is eliminated by simultaneous precipitation.

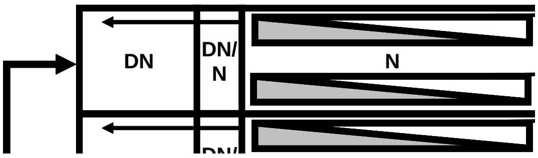


Figure 21: Process flow diagram of the Nordkanal MBR

The Nordkanal MBR employs online influent and effluent quality measurements for the values of ammonia, nitrate, phosphorous and SAC-UV. Additional values are analysed from mixed and grab samples on a regular basis. All these data have been used to asses the performance of the biological treatment processes of the Nordkanal MBR plant. The average concentrations and loads are used to calculate the actual denitrification rate of the plant.

A biokinetic model is set up for the Nordkanal MBR to see if ammonia effluent variations for a membrane bioreactor could be predicted. Therefore the Activated Sludge Model no. 3 (Gujer *et al.*, 1999) is combined with a hydraulic model consisting of a cascade of three totally mixed reactors (one virtual vessel for every existing bioreactor compartment with pre-denitrification, alternating zone, nitrification), a phase separation with complete retention of all suspended model components and a sludge recycle.

1.3.2 Results

1.3.2.1 Influent and effluent variations

The average ammonia concentrations during the period from Jan 1st 2007 to Dec 31st 2007 are 40 g m^{-3} in the influent and 0.12 g m^{-3} in the effluent. Despite these low effluent values single peaks of up to 9 mg/L occur during the winter season at low temperatures when nitrification capacity is at its minimum. The daily loading of chemical oxygen demand COD, total nitrogen TN and phosphorous P_{tot} for the respective period can be seen from Table 8. The quantile $Q_{.85}$ of the influent loading accounts for more than 90 % of the plants design capacity.

Table 8: Daily loading (quantile Q.85 and average values of sample data) Nordkanal WWTP

Daily loading	**COD**	**TN**	**Ammonia**	**Nitrate**	**P_{tot}**
Influent ($Q_{.85}$/avg.), kg d^{-1}	9000/6000	810/570	620/450	-	150/80
Effluent ($Q_{.85}$/avg.), kg d^{-1}	400/230	160/80	4/1	130/60	6/4
Sludge production, kg d^{-1}	4500	265			145
Design capacity (influent), kg d^{-1}	9600	897	-	-	123

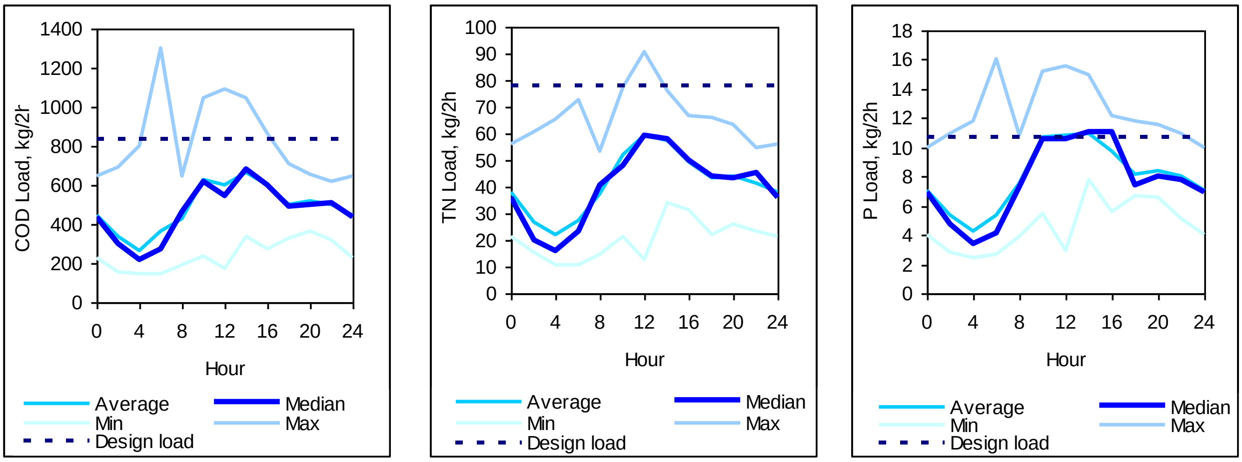


Figure 22: Actual daily influent variations of COD (left), total nitrogen (middle) and total phosphorous (right), values taken from 2-hour samples

The influent and the effluent concentrations show considerable variations especially under rain weather flow. Figure 23 gives an impression of the variability of the influent load of ammonia and the respective effluent concentrations.

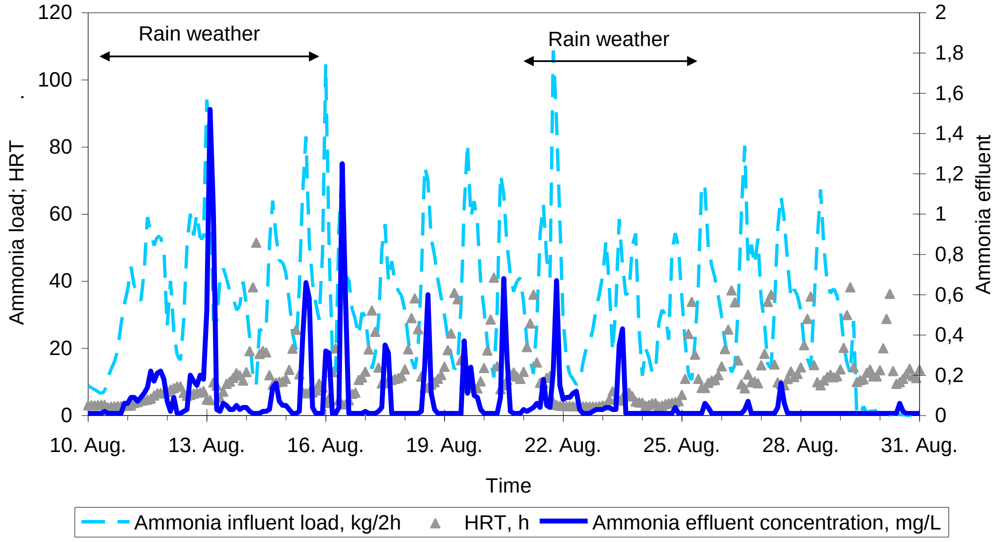


Figure 23: Influent and effluent variations of ammonia and HRT, values of online sensors, August 2007

The time variation curves of the on-line measurements in the influent and effluent reveal that the effluent ammonia peaks occur not later than four hours after the influent peak is measured. The exact time lag between these incidents depends of the inflow and the respective HRT. A more detailed investigation of selected dry weather and rain weather events shows that the time lag is significantly shorter than the average HRT as can be seen from Table 9.

Table 9: Time lag between influent and effluent ammonia peaks

Inflow	**HRT, h**	**Time lag T, h**	**Relative time lag, T/HRT**
Dry weather	> 9.0	2.0 – 4.0	< 0.30
Rain weather	4.7 – 6.0	0.5 – 1.5	0.10 – 0.25

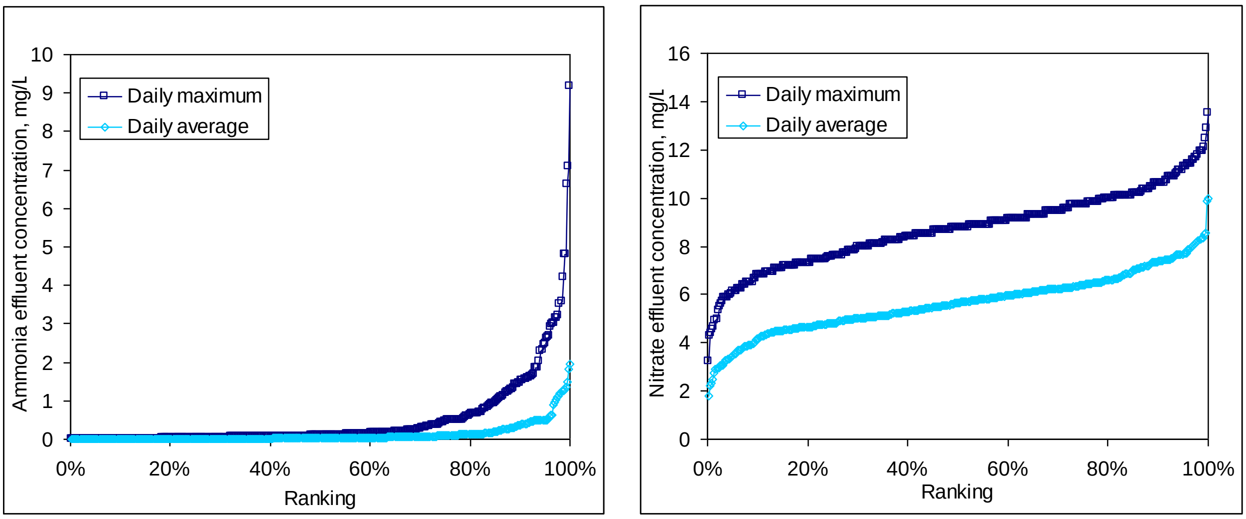


Figure 24: Daily average and peak values of the effluent concentration of ammonia (left) and nitrate (right) as measured by the on-line sensors (Jan 1st 2007 – Dec 31st 2007)

The practical experience at Nordkanal shows that the required effluent quality of 10 g m^{-3} as a peak value is maintained under all circumstances. Figure 24 shows that during 80 % of the days during the reported period the average value was lower than 0.13 g m^{-3}. The biomass content in the nitrification compartments was an average MLSS of 12.5 kg m^{-3} while volatile suspended solids VSS are at 7 kg m^{-3}. The volume of the pre-denitrification basins is at 2600 m^{3} compared to a total volume of 9300 m^{3}. The average recycle was designed to be at 400%. The average denitrification efficiency is approximately at 75 %; the denitrification rate could be calculated between 0.6 and 1.0 gN kgVSS^{-1} h^{-1}.

1.3.2.2 Process modelling

When using real influent measurement data on the model a good coincidence between real and simulated ammonia effluent peaks can be found as displayed in Figure 25. The deviations in absolute concentration values for single events are within the accuracy of the model assumptions. Also the position of nitrate effluent peaks can be predicted while the model underestimates the denitrification capacity of the real installation. A possible explanation lies in the fact that during low inflow single filter lines are shut down to adjust the filtration capacity. Measurements have shown that during these times oxygen concentrations in the tanks become unevenly distributed. Simultaneous denitrification may thus take place in the nitrification tank.

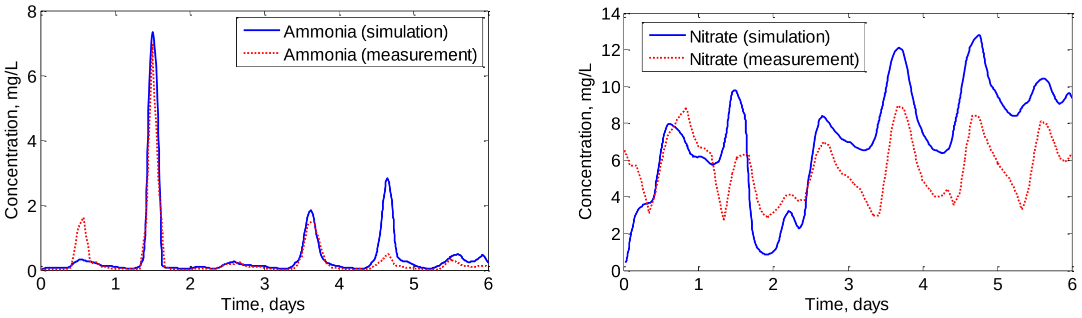


Figure 25: Example of simulation results of ammonia (left) and nitrate (right) concentrations

Similar biokinetic models have been applied at the Rödingen and Glessen MBRs. At Rödingen and Nordkanal, the model was only used to describe and investigate the observed plant behaviour after commissioning. At Glessen the model was used as a tool to design the process control strategy for the biological treatment during an earlier phase of the project (The modelling of the Glessen MBR was part of a different research project).

1.3.3 Conclusions

The nitrification and denitrification capacity at the Nordkanal MBR is sufficient to meet the required peak effluent concentrations despite a short minimum hydraulic retention of 4.7 hours. The flow characteristics of the bioreactor lines with a pre-denitrification, alternating zone an nitrification tank can be described by a cascade of three subsequent completely mixed reactors. This configuration was also used for a biokinetic model that was able to predict the position and approximate value of the effluent peaks for ammonia and nitrate. The satisfactory correlation between modelling and the operational results shows that computational plant modelling can be a useful tool when designing large scale membrane bioreactor plants.

4. References

Abegglen C., Ospelt M. and Siegrist H. (2008). Biological nutrient removal in a small-scale MBR treating household wastewater, *Water Research*, 42 (1-2), 338-346.

Adam C., Gnirss R., Lesjean B., Buisson H. and Kraume M. (2003). Enhanced biological phosphorous removal in membrane biorectors, *Water Science and Technology*, 46 (4-5), 281-286

Ahmed Z., Lim B., Cho J. Song K., Kim K. and Ahn K. (2008). Biological nitrogen and phosphorous removal and changes in microbial community structure in a membrane bioreactor: Effect of different carbon sources, *Water Research*, 42 (1-2), 198-210.

Barker P.S. and Dold P.L. (1997). General model for biological nutrient removal activated-sludge systems: model presentation, *Water Environment Research*, 69 (5), 969-984

Buisson H., Cote P., Praderie M., Paillard H. (1998), The use of immersed membranes for upgrading wastewater treatment plants, *Water Science and Technology*, 37 (9), 89-95

Côté P., Buisson H., Pound C., Arakaki G. (1997), Immersed membrane activated sludge for the reuse of municipal wastewater, *Desalination*, 113 (2-3), 189-196

de Silva D.G.V., Urbain V., Abeysinghe D.H., Rittmann B.E. (1998), Advanced analysis of membrane-bioreactor performance with aerobic-anoxic cycling, *Water Science and Technology*, 38 (4-5), 502-512

Ekama G.A., Dold P.L., Marais G.v.R. (1986), Procedures for determining influent COD fractions and the maximum specific growth-rate of heterotrophs in activated sludge systems, *Water Science and Technology*, 18 (6), 91-114

Fatone F., Battistoni P., Bolzonella D., Pavan P., Cecchi F. (2008), Long-term experience with an automatic process control for nitrogen removal in membrane bioreactors, *Desalination*, 227, 72-84

Fleischer E.J., Broderick T.A., Daigger G.T., Fonseca A.D., Holbook R.D. and Murthy S.N. (2005). Evaluation of membrane bioreactor process capabilities to meet stringent effluent nutrient discharge requirements, *Water Environment Research*, 77 (2), 162-178.

Guglielmi G., Saroj D.P., Chiarani D., Andreottola G. (2007), Sub-critical fouling in a membrane bioreactor for municipal wastewater treatment: Experimental investigation and mathematical modeling, *Water Research*, 41 (17), 3903-3914

Guglielmi G., Andreottola G. (2009), Alternate anoxic/aerobic operation for nitrogen removal in a membrane bioreactor for municipal wastewater treatment, *Proc. of the International Conference on Advances in Wastewater Treatment and Reuse*, June-July 2009, Tehran, Iran

Gujer W., Henze M., Mino T., van Loosdrecht M. (1999), Activated Sludge Model No. 3, *Water Science and Technology*, 39 (1), 183-193

Henze M., Gujer W., Mino T., Matsuo T., Wentzel M.C. and Marais G. (1995). Wastewater and biomass characterization for the activated sludge model no. 2: Biological phosphorus removal, *Water Science and Technology*, 31 (2), 13-23.

Henze M., Gujer W., Mino T., Matsuo T., Wentzel M., Marais GvR, van Loosdrecht MCM (1999), Activated Sludge Model No. 2d, *Water Science and Technology*, 39 (1), 165-182

Hu Z.R., Wentzel M.C., Ekama G.A. (2002), Anoxic growth of phosphate-accumulating organisms (PAOs), *Water Research*, 36 (19), 4927-4937

Hu Z., Wentzel M.C. and Ekama G.A. (2007). A general kinetic model for biological nutrient removal activated sludge systems- model development, *Biotechnology Bioengineering*, 98(6), 1242-1258

Lesjean B., Gnirss R., Adam C., Kraume M. and Luck F. (2003). Enhanced biological phosphorous removal process implemented in membrane bioreactors to improve phosphorous recovery and recycling, *Water Science and Technology*, 48 (1), 87-94

Li Y.Z., He Y.L., Ohandja D.G., Ji J., Li J.F., Zhou T. (2008), Simultaneous nitrification-denitrification achieved by an innovative internal-loop airlift MBR: comparative study, *Bioresource Technology*, 99, 5867-5872

Lim B.S., Choi B.C., Yu S.W., Lee C.G. (2007), Effects of operational parameters on aeration ON/OFF time in an intermittent aeration membrane bioreactor, *Desalination*, 202, 77-82

Kim J.-Y., Chang I.-S., Park H.-H., Kim C.-Y., Kim J.-B., Oh J.-H. (2008), New configuration of a membrane bioreactor for effective control of membrane fouling and nutrients removal in wastewater treatment, *Desalination*, 230, 153-161

Kim H.S., Seo I.S., Kim Y.K., Kim J.Y., Ahn H.W., Kim I.S. (2007), Full-scale study on dynamic state membrane bioreactor with modified intermittent aeration, *Desalination*, 202, 99-105

Kishino H., Ishida H., Iwabu H., Nakano I. (1996), Domestic wastewater reuse using a submerged membrane bioreactor, *Desalination*, 106 (1-3), 115-119

Meinhold J., Filipe C.D.M, Daigger G.T., Isaacs S. (1999), Characterization of the denitrifying fraction of phosphate accumulating organisms in biological phosphate removal, *Water Science and Technology*, 39 (1), 31-42

Monti A., Hall F.A., Dawson R.N., Husain H. and Kelly H.G. (2006). Towards a high-rate enhanced biological phosphorus removal process in a membrane-assisted bioreactor, *Biotechnology Bioengineering*, 94 (4), 740-752

Monti A., Hall E.R., Koch F.A., Dawson R.N., Husain H. and Kelly H.G. (2007). Towards a high-rate enhanced biological phosphorus removal process in a membrane-assisted bioreactor, *Water Environment Research*, 79 (6), 675-686.

Rieger L., Koch G., Kühni M., Gujer W. and Siegrist H. (2001). The EAWAG Bio-P module for activated sludge model no. 3, *Water Research*, 35 (16), 3887-3903

Sarioglu M., Insel G., Artan N., Orhon D. (2009), Model evaluation of simultaneous nitrification and denitrification in a membrane bioreactor operated without an anoxic reactor, *Journal of Membrane Science*, 337, 17-27

Saroj D.P., Guglielmi G., Andreottola G. (2008), Operational optimization for nutrient removal in a membrane bioreactor treating municipal wastewater, *Proc. of SIDISA08*, June 2008, Florence, Italy

Silva A.F., Carvalho G., Lousada-Ferreira M., van Nieuwenhuijzen A., Guglielmi G., Reis M.A.M., Crespo M.T.B. (2009), Presence of polyphosphate accumulating organisms in non-EBPR membrane bioreactors, *Proc. of ASPD 5, Specialised Conference on Microbial Population Dynamics in Biological Wastewater Treatment*, 24th-27th May, Aalborg, Denmark

Ubakata Y. and Takii S. (1998). Some physiological characteristics of a phosphate removing bacterium, Microlunatus Phosphovorus, and a simplified isolation and identification method for Phosphate removing bacteria, *Water Science and Technology*, 38 (1), 149-157.

Ueda T., Hata K., Kikuoka Y. (1996), Treatment of domestic sewage from rural settlements by a membrane bioreactor, *Water Science and Technology*, 34 (9), 189-196

Ueda T., Hata K. (1999), Domestic wastewater treatment by a submerged membrane bioreactor with gravitational filtration, *Water Research*, 33(12), 2888-2892

Yamagiwa K., Oohira Y., Ohkawa A. (1995), Simultaneous removal of carbonaceous and nitrogenous pollutants by a plunging liquid jet bioreactor with crossflow filtration operated under intermittent aeration, Bioresource Technology, 53 (1), 57-62

Yang S., Yang F., Fu Z., Lei R. (2009) Comparison between a moving bed membrane bioreactor and a conventional membrane bioreactor on organic carbon and nitrogen removal, *Bioresource Technology*, 100, 2369-2374

Yeom I.-T., Nah Y.-M., Ahn K.-H. (1999), Treatment of household wastewater using an intermittently aerated membrane bioreactor, *Desalination*, 124 (1-3), 193-203

Yuan L.-M., Zhang C.-Y., Zhang Y.-Q., Ding Y., Xi D.-L. (2008), Biological nutrient removal using an alternating of anoxic and anaerobic membrane bioreactor (AAAM) process, *Desalination*, 221, 566-575

Wang B., He S., Wang L., Shuo L. (2005), Simultaneous nitrification and denitrification in MBR, *Water Science and Technology*, 52 (10-11), 435-442