Original paper

Clean Techn Environ Policy 4 (2002) 171–182
DOI 10.1007/s10098-002-0157-8

Performance analysis of a denitrifying wastewater treatment plant

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Abstract The cleaning performance of an activated sludge wastewater treatment plant for biological nitrogen removal was evaluated based on the Activated Sludge Models No. 1 (ASM1) and No. 3 (ASM3). The COST predenitrification benchmark plant was considered as a case study. Several treatment plant performance indexes were defined based on performance indicators and operating cost factors. Plant operation was evaluated for three dynamic disturbance scenarios corresponding to dry, rain, and storm weather conditions. The aeration rate was seen to be the most sensitive process parameter for this case study, and the nitrate recycle flow rate ratio was the least sensitive of all parameters analyzed. Different qualitative predictions were obtained from the two models, as a result of different model structures. For all three weather scenarios, the sludge production predicted by the ASM1 model increases slightly as the oxygen transfer rate increases; in contrast, the sludge production decreases for the ASM3 model. However, the resulting difference in the sludge treatment cost does not qualitatively modify the cost performance index profile. For all control handles considered the rain weather scenario showed the highest cost performance index and the lowest global cleaning efficiency.

List of symbols

The nomenclature followed in Gujer et al. (1999) for the ASM3 model is adopted as the base nomenclature. The notation followed in Henze et al. (1987) for the ASM1 model is given in parentheses.

Common components for ASM1 and ASM3 models

S_I Inert soluble organic material (S_I), g COD m^{-3}
S_{NH4} Ammonium plus ammonia nitrogen (S_{NH}), g N m^{-3}
S_{NOX} Nitrate plus nitrite nitrogen (S_{NO}), g N m^{-3}
S_{O2} Dissolved oxygen (S_O), g COD m^{-3}
S_S Readily biodegradable organic substrates (S_S), g COD m^{-3}
X_A Nitrifying organisms ($X_{B,A}$), g COD m^{-3}
X_H Heterotrophic organisms ($X_{B,H}$), g COD m^{-3}
X_I Inert particulate organic material (X_I), g COD m^{-3}
X_S Slowly biodegradable substrates (X_S), g COD m^{-3}

Notation involved only in the ASM3 model

f_{XI} Production of X_I in endogenous respiration, g COD_{XI} (g COD_{XBM})$^{-1}$
$i_{N,BM}$ N content in biomass (X_H, X_A), g N (g COD_{XBM})$^{-1}$
$i_{N,SI}$ N content in S_I, g N (g COD_{SI})$^{-1}$
$i_{N,SS}$ N content in S_S, g N (g COD_{SS})$^{-1}$
$i_{N,XS}$ N content in X_S, g N (g COD_{XS})$^{-1}$
$i_{N,XI}$ N content in X_I, g N (g COD_{XI})$^{-1}$
$i_{SS,XI}$ SS to COD ratio for X_I, g SS (g COD_{XI})$^{-1}$
$i_{SS,XS}$ SS to COD ratio for X_S, g SS (g COD_{XS})$^{-1}$
$i_{SS,BM}$ SS to COD ratio for X_{BM}, g SS (g COD_{XBM})$^{-1}$
X_{STO} Cell internal storage product of heterotrophic organisms, g COD m^{-3}
X_{SS} Suspended solids, g COD m^{-3}

Notation involved only in the ASM1 model

f_P Fraction of biomass leading to particulate products, dimensionless
i_{XB} N content in biomass (X_H, X_A), g N (g COD_{XBM})$^{-1}$
i_{XP} N content in particulate products, g N (g COD_{XP})$^{-1}$
S_{ND} Soluble biodegradable organic nitrogen, g N m^{-3}
X_{ND} Particulate biodegradable organic nitrogen, g N m^{-3}
X_P Particulate products arising from biomass decay, g COD m^{-3}

General symbols and abbreviations

AE Aeration energy, kWh day $^{-1}$
ASM1 Activated Sludge Model No. 1
ASM3 Activated Sludge Model No. 3
BOD Biological oxygen demand, g O_2 m^{-3}

Received: 19 December 2001 / Accepted: 25 April 2002
Published online: 21 August 2002

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The financial support from the National Council for Scientific and Technical Research of Argentina (CONICET) and the European Commission (SMAC project, contract EVK1-CT-2000-00056) is gratefully acknowledged.

C Carbon

C_k Concentration of the component k, $g^{-1}\ m^{-3}$

COD Chemical oxygen demand, g O_2 m^{-3}

COST European Cooperation in the field of Scientific and Technical Research

CPI Cost performance index, Euro year $^{-1}$

EQ Effluent quality index, kg pollution units day^{-1}

IQ Influent quality index, kg pollution units day^{-1}

K_La Oxygen mass transfer rate coefficient, day^{-1}

M Sludge mass, kg

N Nitrogen

NRR Nitrate recycle flow rate ratio

OTR Oxygen transfer rate

PE Pumping energy, kWh day^{-1}

P_{sldg} Sludge production, kg day^{-1}

PU_k Polluting unit for the component k, kg day^{-1}

Q Volumetric flow rate, m^{-3} day^{-1}

SRR Sludge recycle flow rate ratio

t time, day

t_o initial time, day

t_f final time, day

TKN Total Kjeldahl nitrogen, g N m^{-3}

TSS Total suspended solids, g SS m^{-3}

α_j Yearly cost factor for the cost component j, Euro/(kg day^{-1}), Euro/(kWh day^{-1})

β_k Weighting factor for the component k, (g pollution units) $g^{-1}\ m^{-3}$

Δ Increment

η Global cleaning efficiency, dimensionless

Subscripts

e effluent

in influent

r recycle

sys system

w waste

Introduction

Wastewater streams containing nitrogenous compounds may cause serious environmental problems if they are not suitably cleaned prior to discharge into the receiving water bodies. A too high nitrogen concentration in the receiving waters can lead to eutrophication, i.e. algal outbreaks and/or fish death in rivers, lakes, and coastal areas.

Nitrogen (N) may appear in wastewater in four main forms: as organic, ammonium, nitrite, and nitrate N. However, the predominant N fractions in municipal wastewater are organic N, e.g. linked to proteins present in the wastewater, and ammonium N.

Before its discharge into the receiving waters, N can be removed from the wastewater by a combination of various biological processes that can take place under anaerobic, aerobic, and/or anoxic conditions. In the first step of the biological N removal process of activated sludge systems, the organic N fraction is converted to ammonium due to hydrolysis of proteins and other organic matter fractions containing N. Ammonium is subsequently oxidized to nitrate. The latter process, referred to as nitrification, takes place under aerobic conditions. Nitrogen can finally be removed from the wastewater by reducing the nitrate to N_2 gas, which is released to the atmosphere. This process is commonly referred to as denitrification, and requires anoxic conditions to proceed, as well as the presence of a readily biodegradable organic carbon source.

The stringent effluent limits imposed have resulted in research towards both improved design and optimized process operation of the activated sludge wastewater treatment plants (WWTPs). Indeed, in practice there exists a wide variety of hydraulic plant configurations and operation modes for the activated sludge process. For example, there are continuous, semi-continuous, and batch activated sludge plants in full-scale operation, where the continuous process type can be considered the most conventional type.

Modeling and simulation are important tools for generation and assessment of scenarios related to WWTP design and operation, aiming at minimization of the total wastewater treatment cost for a given load scenario. In that sense, a lot of basic research first focused on the understanding of the biological activated sludge mechanisms, and resulted in a number of mathematical models to describe biological N removal in activated sludge processes. The Activated Sludge Models No. 1 (ASM1) (Henze et al. 1987) and No. 3 (ASM3) (Gujer et al. 1999) are presently the most widely accepted models for description of biological N removal processes. The main differences between these two models are commented on and investigated in Mussati et al. (2002) for both steady state influent conditions and ideal disturbance scenarios.

In addition to the models, several performance indexes and criteria have been proposed during the past years to evaluate the wastewater treatment system performance. The European Cooperation in the field of Scientific and Technical Research (COST) benchmark protocol (Copp 2002) provides criteria to evaluate the effluent quality and operating costs. Vanrolleghem and Gillot (2001) included specific cost factors for aeration and pumping energy demand, waste sludge treatment, and effluent fines based on Belgian regulations. Recently, Hopkins et al. (2001) proposed a flexibility index as a tool for comparison between continuous versus batch activated sludge plant design and operation performance.

The purpose of this paper is to evaluate and compare the performance of an activated sludge predenitrification plant under realistic dynamic load conditions predicted by two activated sludge models (ASM1 and ASM3). The cleaning performance is evaluated by using economic and non-economic indicators. The ultimate aim is to determine a performance index for evaluation of the suitability of different control structures for ensuring optimal operation. An objective economic measure of the plant performance is expected by simultaneously considering operating costs for aeration, pumping, waste sludge treatment, and eventual fines imposed for remaining effluent contaminant load. The influence of the main actuators for control on the system performance is analyzed from dynamic simulation results obtained for both models. Finally, the relation between the steady state sensitivity analysis results (Mussati et al. 2002) and the system performance evaluation via dynamic simulations for the dry weather scenario is analyzed from a treatment efficiency point of view.

Methods

Performance criteria

In this paper, a model-based WWTP performance evaluation is performed by applying different economic and non-economic criteria. These criteria are the effluent quality index, the global cleaning efficiency, and the cost performance index, which are defined in the next subsections.

Effluent and influent quality indexes

The effluent quality index (EQ) summarizes simulated concentration values for all the contaminating components present in the cleaned stream into a single index. Weighting factors β_k were considered to specify the contribution of each component in the EQ (Copp 2002), which are based on the Flandes effluent quality formula for calculating fines (Vanrolleghem et al. 1996).

The effluent quality weighting factors thus allow one to quantify the relative polluting effect of each component present in the wastewater.

The EQ is calculated as follows:

$$EQ = \frac{1}{1000(t_f - t_o)} \int_{t_o}^{t_f} PU_{(t)} Q_e(t) dt \tag{1}$$

$$PU_{(t)} = PU_{TSS}(t) + PU_{COD}(t) + PU_{BOD}(t) + PU_{TKN}(t) + PU_{NOX(t)} \tag{2}$$

The polluting load PU_k (kg day^{-1}) corresponding to the component k is calculated as

$$PU_k = \beta_k C_k \tag{3}$$

where β_{TSS}= 2; β_{COD}= 1; β_{BOD}= 2; β_{TKN}= 20; β_{NOX}= 20.

For the ASM1 model:

$$\left.\begin{array}{l} C_{TSS_e} = 0.75(X_{S_e} + X_{H_e} + X_{A_e} + X_{P_e} + X_{I_e}) \\ C_{COD_e} = S_{S_e} + S_{I_e} + X_{S_e} + X_{H_e} + X_{A_e} + X_{P_e} + X_{I_e} \\ C_{BOD_e} = 0.25\left(S_{S_e} + X_{S_e} + (1 - f_P)(X_{H_e} + X_{A_e})\right) \\ C_{TKN_e} = S_{NH4_e} + S_{ND_e} + X_{ND_e} + i_{XB}(X_{A_e} + X_{H_e}) + i_{XP}(X_{P_e} + X_{I_e}) \\ C_{NOX_e} = S_{NOX_e} \\ C_{Ntot_e} = C_{TKN_e} + C_{NOX_e} \end{array}\right\} \tag{4}$$

whereas for the ASM3 model:

$$\left.\begin{array}{l} C_{TSS_e} = X_{SS_e} \\ C_{COD_e} = S_{S_e} + S_{I_e} + X_{S_e} + X_{I_e} + X_{H_e} + X_{A_e} + X_{STO_e} \\ C_{BOD_e} = 0.25(S_{S_e} + X_{S_e} + (1 - f_{XI})(X_{H_e} + X_{A_e} + X_{STO_e})) \\ C_{TKN_e} = i_{N,S_I} S_{I_e} + i_{N,S_S} S_{S_e} + i_{N,X_S} X_{S_e} + i_{N,X_I} X_{I_e} + i_{N,BM}(X_{A_e} + X_{H_e}) + S_{NH4_e} \\ C_{NOX_e} = S_{NOX_e} \\ C_{Ntot_e} = C_{TKN_e} + C_{NOX_e} \end{array}\right\} \tag{5}$$

In Eq. (1) t_f represents the end-time of the period where EQ is evaluated, and Q_e is the volumetric flow rate of the cleaned wastewater stream.

An influent quality index (IQ) can be calculated in a similar way to the EQ, but by changing the BOD coefficient in Eqs. (4) and (5) from 0.25 to 0.65 (Copp 2002).

Global cleaning efficiency

The global cleaning efficiency (η) is here defined as the reduction percentage of the wastewater contaminant load entering the treatment plant. This criterion relates the effluent to the influent quality index defined in the previous subsection, and is thus an indication of the global pollutant removal performance of the treatment plant without considering costs.

$$\eta = 100 \frac{IQ - EQ}{IQ} \tag{6}$$

Cost performance index

The cost performance index (CPI) is defined as in Vanrolleghem and Gillot (2001):

$$CPI = \alpha_{EQ} EQ + \alpha_{AE} AE + \alpha_{PE} PE + \alpha_{sldg} P_{sldg} \tag{7}$$

where EQ is the effluent quality index (Eq. 1); AE and PE represent the aeration and pumping energy consumption rates, respectively, and P_{sldg} is the sludge production rate (Eqs. 8, 9, and 10, respectively). The α coefficients are the operation cost weighting factors, which are defined in the next subsection.

- Aeration energy (kWh day^{-1})

$$AE = \frac{24}{t_f - t_o} \int_{t_o}^{t_f} \sum_{i=1}^{i=5} \left[7 \times 10^{-4} K_L a_i(t)^2 + 0.3267 K_L a_i(t)\right] dt \tag{8}$$

- Pumping energy (kWh day^{-1})

$$PE = \frac{0.04}{t_f - t_o}\int_{t_o}^{t_f}[Q_e(t) + Q_r(t) + Q_w(t)]\mathrm{d}t \tag{9}$$

- Waste sludge disposal (kg TSS day^{-1})

$$P_{sldg} = \frac{\Delta M(TSS_{sys}) + M(TSS_w)}{t_f - t_o} \tag{10}$$

$$\Delta M(TSS_{sys}) = M(TSS_{sys})_{t_f} - M(TSS_{sys})_{t_o} \tag{11}$$

$$M(TSS_{sys}) = M(TSS_{react}) + M(TSS_{settler}) \tag{12}$$

$$M(TSS_w) = \int_{t_o}^{t_f} TSS_w Q_w(t)\mathrm{d}t \tag{13}$$

The total suspended solids in the waste sludge for disposal TSS_w is calculated differently for the ASM1 and the ASM3 models. For the ASM1 model:

$$TSS_w = 0.75[X_{S_w} + X_{H_w} + X_{A_w} + X_{P_w} + X_{I_w}] \tag{14}$$

whereas for the ASM3 model:

$$TSS_w = X_{SS_w} \tag{15}$$

Effluent permit limits and operation cost weighting factors

There exist two types of economic fines to penalize the discharge of contaminants into the receiving water bodies. One type imposes fines for each polluting unit discharged. This regulation is in force in e.g. Belgium and Denmark. In the second type, fines and penalties are imposed when the contaminant levels exceed threshold values, which depend on each country or region.

The cost weighting factor α_{EQ} in the CPI (Eq. 7) takes into account the fines imposed for discharging pollutants. By modifying this factor, the regulator has a strong tool to promote further improvement of the cleaning efficiency of the WWTPs. Increasing α_{EQ} would lead to an increased interest in removing an extra pollution unit from the wastewater. The latter can make it economically feasible to invest in more advanced process control or wastewater treatment techniques.

In this paper, simulation results based on the first effluent fine type are presented. The operating cost weighting factors α suggested by Vanrolleghem and Gillot (2001) are considered (Table 1), which are based on Belgian standards.

Nitrogen removal plant configuration modeled

The COST benchmark WWTP design is used as an illustrative example (Fig. 1), which consists of a predenitrification system with two anoxic reactors, three aerated reactors, and a secondary settler. The system includes two recycle streams, one from the last aerated reactor to the first anoxic reactor, and a second one, which is sludge

Table 1. Cost weighting factors for yearly effluent fine and operating costs (Vanrolleghem and Gillot 2001)

Cost weighting factor	Value	Units
Effluent fines α_{EQ}	50	Euro/(kg day^{-1})
Sludge treat. α_{sldg}	75	Euro/(kg TSS day^{-1})
Aeration energy α_{AE}	25	Euro/(kWh day^{-1})
Pumping energy α_{PE}	25	Euro/(kWh day^{-1})

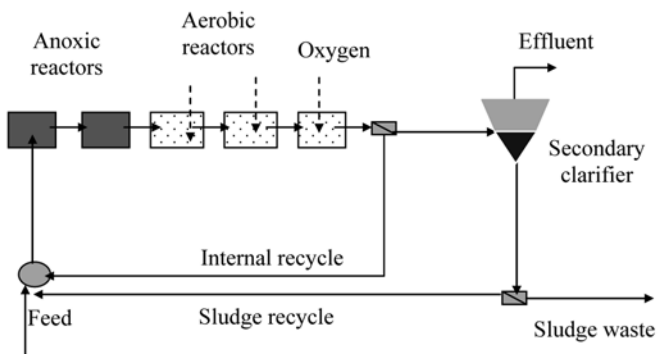

Fig. 1. Plant configuration modeled

recycle, from the secondary settler to the first anoxic reactor.

The ASM1 model (Henze et al. 1987) and the ASM3 model (Gujer et al. 1999) are chosen as the biological process models. The settling process in the secondary settler is modeled as a ten-layer settling tank using the double-exponential settling velocity model of Tákacs et al. (1991).

Model parameters

The stoichiometric and kinetic constants at 15 °C included in the COST benchmark study report (Copp 2002) were used for the ASM1 model. For the ASM3 model the parameter values were interpolated to 15 °C based on the default parameter values at 10 and 20 °C and the temperature interpolation function given by Gujer et al. (1999). The stoichiometric factor $i_{N,XS}$, which represents the N fraction in the slowly biodegradable substrate X_S, was slightly adjusted from 0.04 to 0.0426 to keep the same N fraction in the particulate matter for both models (Mussati et al. 2002).

Wastewater specifications

Compatible sets of specifications for the incoming wastewater stream are defined for the ASM1 and ASM3 models. The inlet volumetric flow rate specifications for both models characterizing the dry, rain, and storm weather scenarios and the influent component specifications for the ASM1 model given in Copp (2002) are considered. For the ASM3 model, the input specifications for the S_S, S_I, X_S, X_H, and X_I components are the same as those for the ASM1 model. The other ASM3 input concentrations are adopted to have exactly the same C and N load for both models. To achieve this, the average values for ASM1 and ASM3 dry weather specifications given in Mussati et al. (2002) were taken as a basis to generate the time-varying component specifications for ASM3 for the three disturbance scenarios based on the existing ASM1 input files.

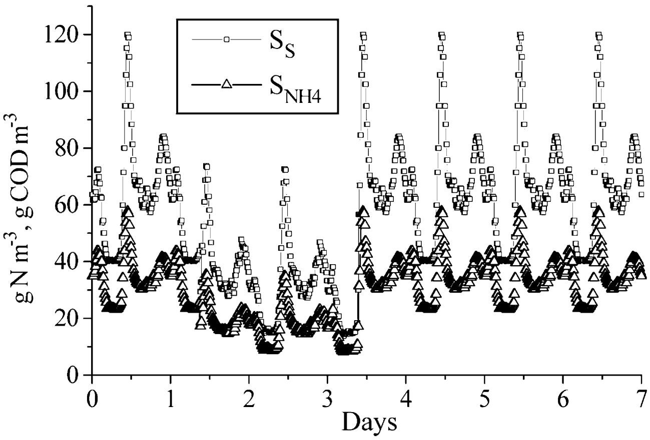

Fig. 2. Influent N-ammonia (S_{NH4}) and readily biodegradable substrate (S_S) specifications for the rain weather scenario for the ASM3 model

The dynamic S_{NH4} input specifications for the ASM3 model for the different scenarios were obtained by multiplying the ASM1 influent S_{NH4} concentrations by the $S_{NH4_{ASM3}}/S_{NH4_{ASM1}}$ ratio resulting from the dry weather average data (approx. 1.15). It was verified that this ratio is indeed almost identical for all the weather scenarios considered. The X_{SS} input concentrations are calculated based on the ASM3 stoichiometric coefficients and the component concentrations that contribute to the suspended solids fraction:

$$X_{SS} = i_{SS,X_I}X_I + i_{SS,X_S}X_S + i_{SS,BM}(X_A + X_H) + i_{SS,STO}X_{STO} \tag{16}$$

As an illustrative example, Fig. 2 shows the input specifications for S_{NH4} and S_S concentrations corresponding to the rain weather scenario for the ASM3 model.

Process analysis

The typical activated sludge process involves several biodegradation steps taking place in different tanks, rendering it a rather complex system to operate and control. Concerning N removal the following features render it difficult to operate a treatment plant:

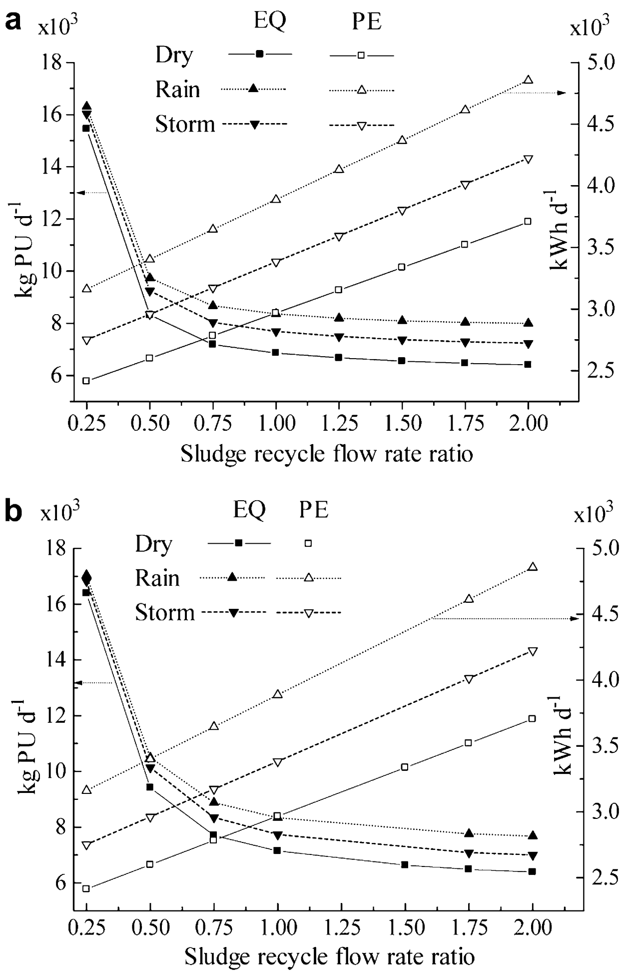

Fig. 3a, b. Effect of the sludge recycle flow rate ratio on the effluent quality index (*left-hand axis*) and pumping energy (*right-hand axis*) for the dry, rain, and storm weather scenarios. **a** ASM1; **b** ASM3

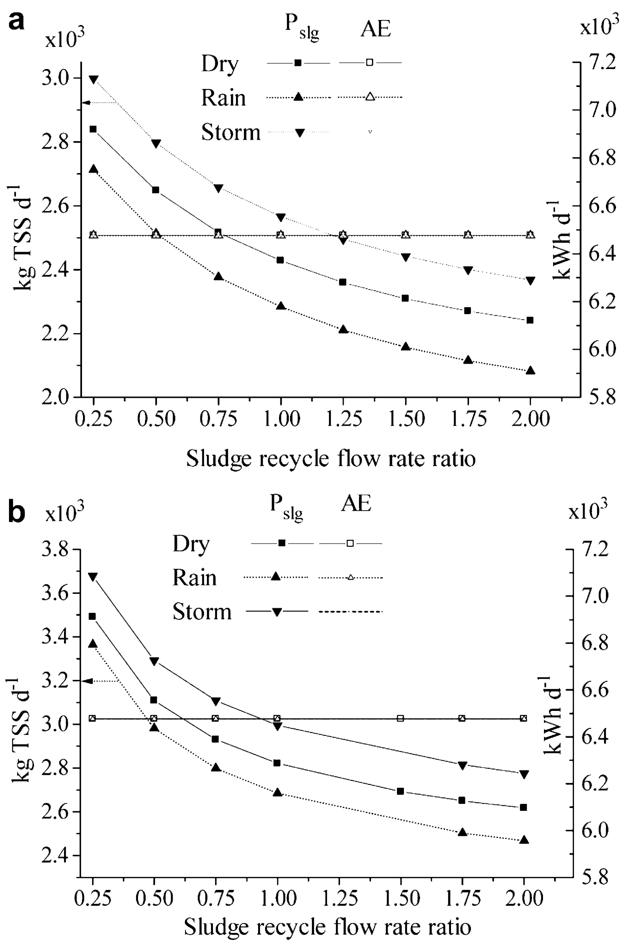

Fig. 4a, b. Effect of the sludge recycle flow rate ratio on the sludge for disposal (*left-hand axis*) and aeration energy (*right-hand axis*) for the dry, rain, and storm weather scenarios. **a** ASM1; **b** ASM3

- Autotrophic biomass (nitrifying organisms) grows slowly compared to heterotrophs. Therefore, a long sludge retention time is required to secure the nitrification process. This fact causes excessive growth of heterotrophic biomass and accumulation of inert solids.
- The wastewater composition has a significant influence on the operation and control of N removal plants. The denitrification process, which is the second step of the biological N removal, depends on the availability of readily biodegradable carbonaceous organic matter, since about 4 g of COD is consumed per g of NO_3-N denitrified (Henze et al. 1997). Therefore, a low C to N ratio in the influent wastewater stream can severely limit the N removal efficiency of the treatment plant.
- Biologically, nitrification takes place before denitrification. As the former process is accompanied by aerobic oxidation of the carbonaceous organic matter, a large fraction of the influent COD fed to the aerobic zone is oxidized and, therefore, is not available for denitrification. In a predenitrification system, which is the case studied in this work (Fig. 1), the anoxic zone for denitrification is located before the aerobic nitrification zone. This configuration allows a reduction of the denitrification effect into the clarifier. Denitrification in the settler can otherwise cause poor settling and even sludge rising. The nitrified mixed liquor is recycled from the last aerobic reactor to the anoxic zone for denitrification. This system cannot achieve 100% nitrogen removal because the last aerobic reactor discharges a fraction of the nitrified mixed liquor into the secondary settler. Thus, to be efficient these systems require a relatively high internal recycle to feed flow rate ratio, which, however, demands considerable pumping energy.

Simulation results

In this section, performance evaluation of the WWTP under study based on dynamic simulation results obtained for the ASM1 and ASM3 models are presented. The evaluation is performed by monitoring the plant performance criteria under the three disturbance scenarios (dry, rain, and storm weather conditions) for a range of control handle set points. Thus, the effluent quality index EQ and the operation cost indicators (aeration energy AE, pumping energy PE and sludge for disposal P_{sldg}), the global cleaning efficiency η and the cost performance index CPI

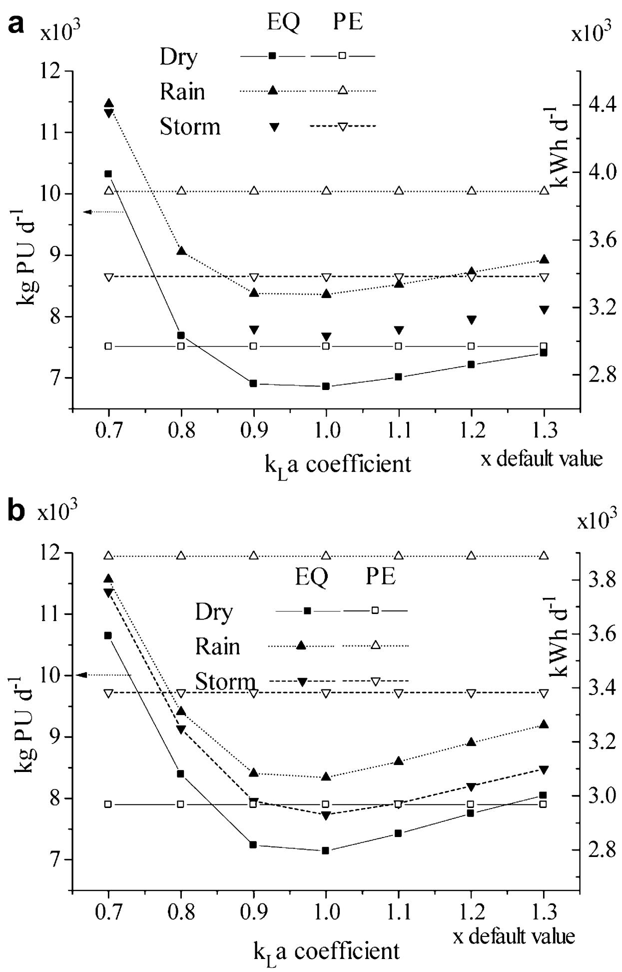

Fig. 5a, b. Effect of the oxygen transfer rate on the effluent quality index (*left-hand axis*) and pumping energy (*right-hand axis*) for the dry, rain, and storm weather scenarios. **a** ASM1; **b** ASM3

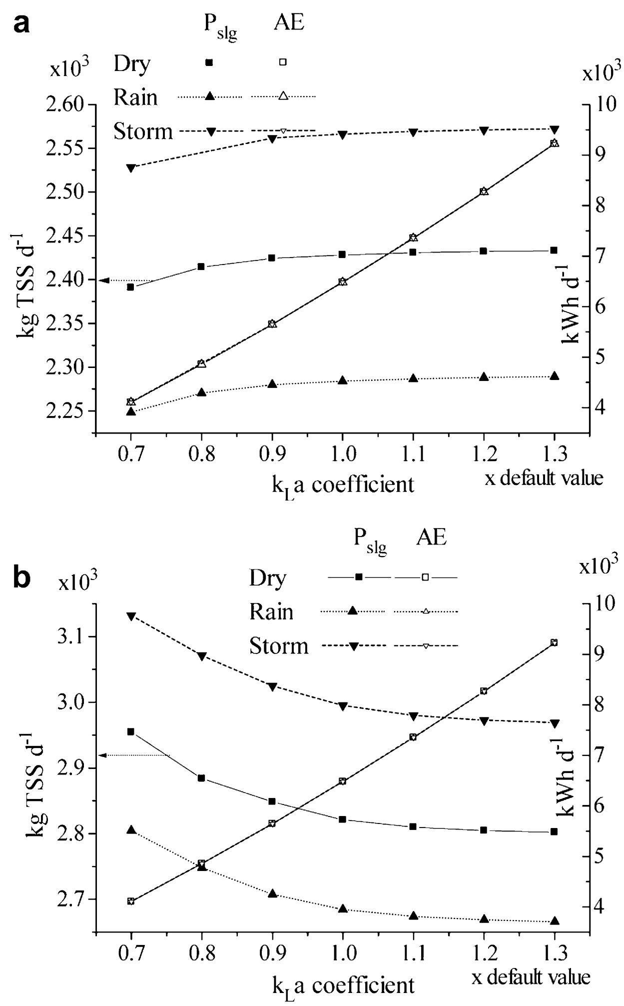

Fig. 6a, b. Effect of the oxygen transfer rate on the sludge for disposal (*left-hand axis*) and aeration energy (*right-hand axis*) for the dry, rain, and storm weather scenarios. **a** ASM1; **b** ASM3 are evaluated for different set points of the oxygen transfer rate OTR, the sludge recycle flow rate ratio SRR and the nitrate recycle flow rate ratio NRR for both models.

Effluent quality index EQ and operation indicators AE, PE, P_{sldg}

Sludge recycle flow rate ratio SRR

Figure 3 shows that the highest contaminating load is discharged into the receiving water for the rain weather scenario. It should be noted that a higher EQ indicates discharge of a higher pollutant load into the receiving water bodies, or a reduced cleaning efficiency of the WWTP. Doubling SRR does not result in a substantial change of EQ. The three influent disturbance scenarios follow the same trend. The predicted sludge production rate P_{sldg} is lowest for the rain weather scenario, and a significant increase of P_{sldg} is observed when SRR decreases (Fig. 4). The increase of P_{sldg} for decreased SRRs coincides with a detrimental effect on the effluent quality leaving the WWTP. Figure 3 also points out that SRR should not be lower than 0.75 for this particular WWTP in order to avoid severe deterioration of the effluent quality. The highest PE is obtained for the rain weather scenario. Obviously, AE is constant and identical for all the scenarios because of the assumption of a constant K_La coefficient, regardless the scenario and SRR adopted.

The above comments are essentially valid for the predictions obtained from both models. However, the ASM1 model shows a slightly flatter profile for EQ for increased SRRs. This tendency is observed for the three weather scenarios.

Oxygen transfer rate OTR

The EQ shows a minimum value around the reference K_La value for both models (Fig. 5). The P_{sldg} predicted by the ASM1 and ASM3 models shows opposing tendencies. For the ASM1 model (Fig. 6a), the amount of sludge for disposal increases asymptotically as K_La increases. This behavior is in contrast to the ASM3 model, where P_{sldg} decreases asymptotically for increasing K_La coefficient values (Fig. 6b). This fact is attributed to the difference in the ASM1 and ASM3 model assumptions with respect to the biomass decay processes. In the ASM1 model, biomass decay is independent of the dissolved oxygen concentration S_{O2} in the reactor. In the ASM3 model, the process rates of the aerobic endogenous respiration, for both

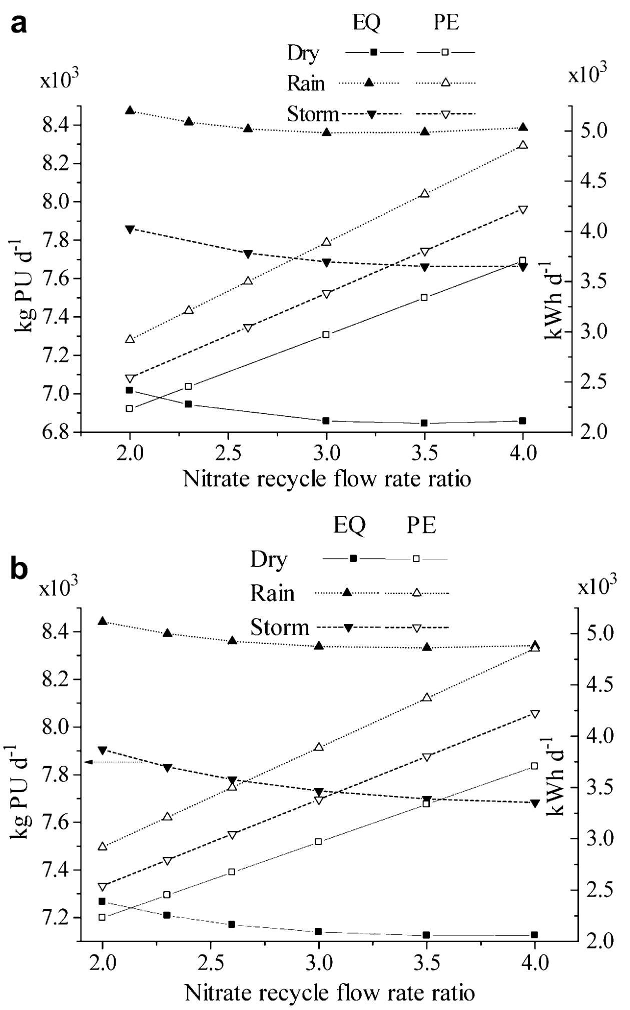

Fig. 7a, b. Effect of the nitrate recycle flow rate ratio on the effluent quality index (*left-hand axis*) and pumping energy (*right-hand axis*) for the dry, rain, and storm weather scenarios. **a** ASM1; **b** ASM3

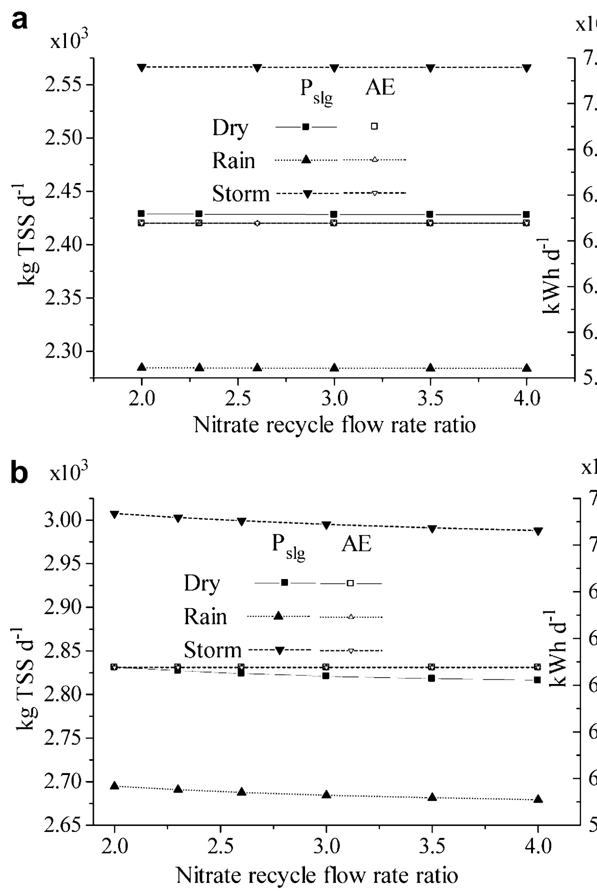

Fig. 8a, b. Effect of the nitrate recycle flow rate ratio on the sludge for disposal (*left-hand axis*) and aeration energy (*right-hand axis*) for the dry, rain, and storm weather scenarios. **a** ASM1; **b** ASM3 autotrophic and heterotrophic biomass, include kinetic factors dependent on S_{O2}. The aerobic endogenous respiration rate expression includes a Monod-type function. Thus, an increased K_La coefficient increases S_{O2}, which causes an increased decay of biomass in the aerobic reactors, and hence, a decrease of the sludge production. Similar trends are observed under the three different weather scenarios.

Obviously, AE increases when the K_La coefficient increases (Fig. 6), whereas PE remains constant for all the scenarios irrespective of the K_La coefficient values (Fig. 5).

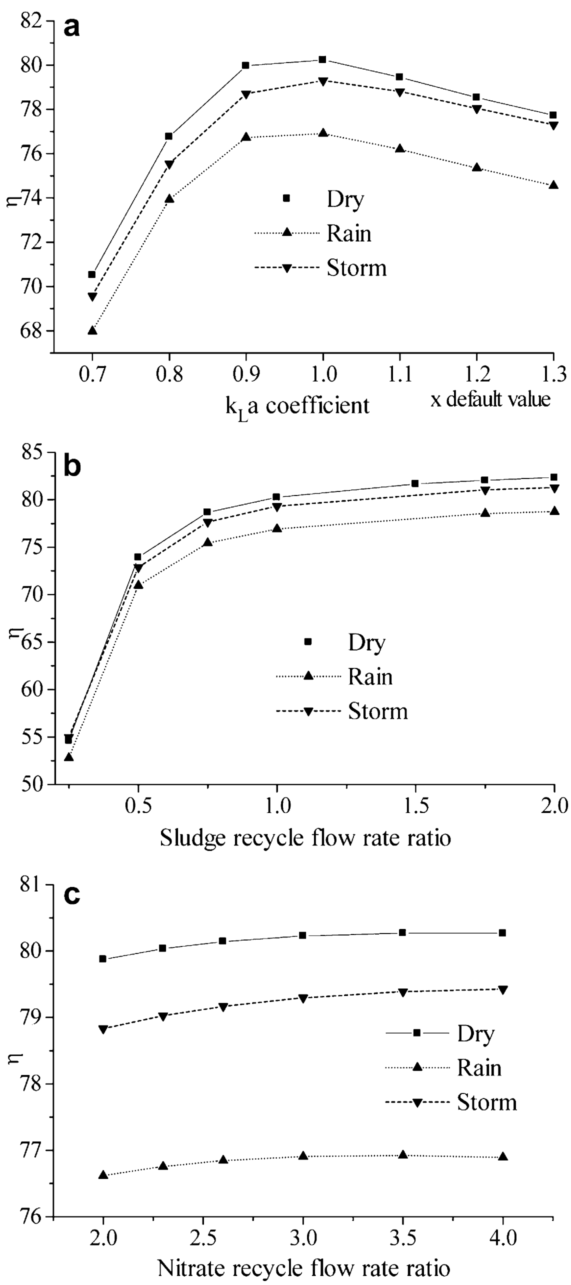

Fig. 9a–c. Global cleaning efficiency predicted by the ASM3 model for the dry, rain, and storm weather scenarios varying **a** oxygen transfer rate, **b** sludge recycle flow rate ratio, **c** nitrate recycle flow rate ratio

Nitrate recycle flow rate ratio NRR

Deterioration of the effluent quality is observed for both models as NRR decreases (Fig. 7). However, an increase of NRR above the default value of the COST benchmark (3.0) does not produce a significant improvement on the effluent quality. For the ASM1 model, P_{sldg} is essentially insensitive to this control handle along the considered set point range (Fig. 8a). However, according to the predictions based on the ASM3 model, P_{sldg} decreases slightly as NRR increases (Fig. 8b). This trend is observed for all the weather scenarios.

Global cleaning efficiency η

Similarly to EQ (Fig. 5b), Fig. 9 also shows an optimal K_La coefficient value that maximizes η. This result is valid for all the weather scenarios studied. By varying the K_La set point ±10% around the default value, η ranges from 76 to 80% for all the scenarios considered.

Unlike OTR, no optimal (minimum) SRR and NRR are observed from Fig. 9b and Fig. 9c, respectively. An essentially flat η profile results from varying NRR in the range analyzed (Fig. 9c). It can thus be concluded that NRR could easily be reduced from its default value. The main reason for this observation is a lack of carbon source in the denitrification zone. As a consequence, a variation

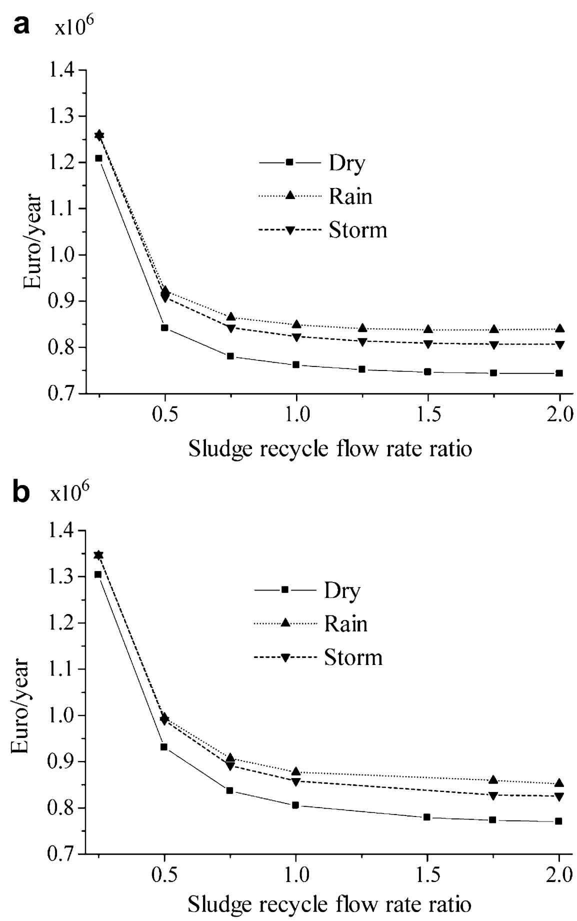

Fig. 10a, b. Cost performance index versus the sludge recycle flow rate ratio for the dry, rain, and storm weather scenarios. **a** ASM1 model; **b** ASM3 model of NRR does not have an important influence on η since the plant is rather insensitive to changes in NRR.

For SRR, low η values are obtained at ratios below 0.75 for all the scenarios. As ASM1 predicts a flatter EQ profile than ASM3 as a function of SRR (Fig. 3), ASM1 will also exhibit a flatter η profile than ASM3 for all the weather scenarios (results not shown). From a cleaning efficiency point of view, these simulation results actually indicate that it does not make sense to change the set point value of SRR from the reference value: a decrease results in a cleaning efficiency deterioration, whereas an increase does not result in a significant improvement of η.

Cost performance index CPI

Sludge recycle flow rate ratio

The CPI increases significantly when SRR is reduced below the default value (Fig. 10), mainly because a poorer effluent quality is discharged (Fig. 3) and, consequently, increased effluent fines are to be paid. A minimum CPI value is not achieved for the range of SRR studied. The increase of the pumping energy costs related to the SRR increases is not sufficiently high to provide an optimal solution (a minimum) within the range of interest. This behavior is probably connected to the way CPI is defined. A more rigorous performance criterion including other cost components (equipment maintenance, additives, capital cost, etc.) may result in identification of an optimal SRR in the range under study. In the simulation example, the effluent fine type considered in CPI even included a cost for each discharged pollution unit, whereas in many countries the effluent fines are only to be paid for the amount of pollutants discharged above the effluent permit limits. In addition, as the highest cost factor corresponds to the excess sludge treatment (Table 1), and because the sludge treatment cost dependency on SRR approximately follows the EQ profile (Figs. 3 and 5), CPI is dominated by these two cost components.

Oxygen transfer rate

Unlike the previous case for SRR, the variation of OTR manipulating the $K_L a$ coefficient reveals an optimum for CPI (Fig. 11).

The CPI exhibits for both models a minimum value at 0.9 times the COST benchmark default $K_L a$ value for all three weather scenarios. This value is different from the optimal $K_L a$ coefficient corresponding to the maximum η (Fig. 9a).

Nitrate recycle flow rate ratio

In this case, CPI is mainly influenced by the pumping cost (Fig. 7). The result is a monotonic dependence of CPI on NRR for both models (Fig. 12). The ASM3 model predicts

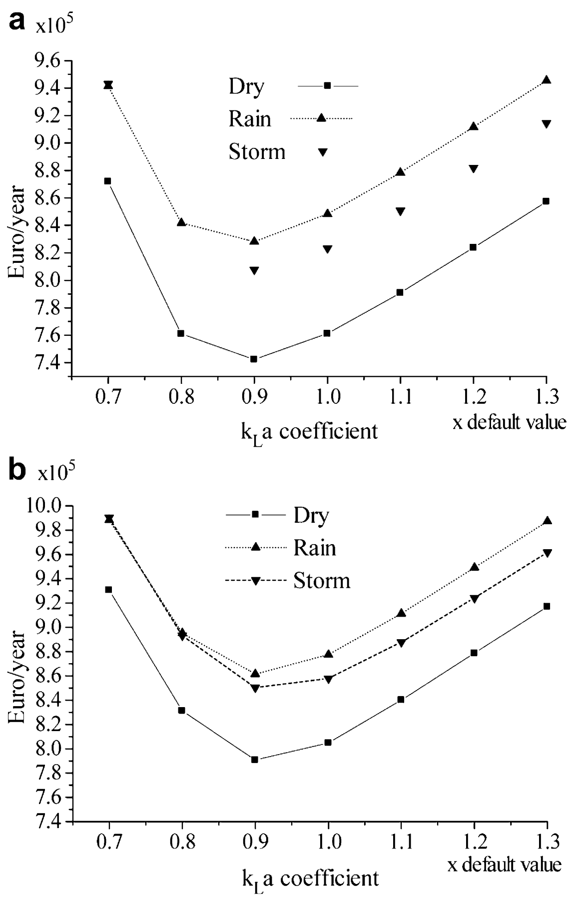

Fig. 11a, b. Cost performance index versus the oxygen transfer rate for the dry, rain, and storm weather scenarios. **a** ASM1 model; **b** ASM3 model

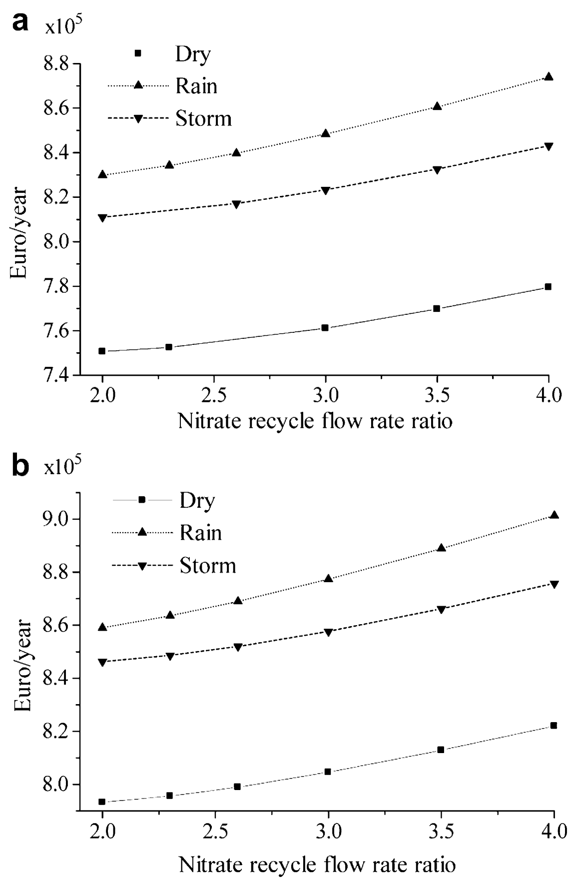

Fig. 12a, b. Cost performance index versus the nitrate recycle flow rate ratio for the dry, rain, and storm weather scenarios. **a** ASM1 model; **b** ASM3 model a higher CPI than ASM1, but the predicted dependency patterns are similar. The resulting CPI profile can again be explained by the lack of a carbon source for denitrification in the anoxic zone. The CPI indicates that NRR can be reduced. Clearly, the reduction in the pumping cost is larger than the effluent fines related to effluent quality deterioration due to this change of the nitrate recycle set point (Fig. 7).

Finally, the simulation results predicted by the ASM1 and ASM3 models are summarized and compared in Table 2.

General discussion

The CPI is probably the most appropriate criterion to evaluate the WWTP performance since it considers more parameters than any other criterion. Indeed, costs for aeration, pumping, and sludge disposal, and the cost related to effluent fines are included in this index. These cost factors are probably the most important cost factors in an activated sludge WWTP for N removal, besides the capital cost related to construction of the plant. Additional operating cost factors can always be included to obtain a more detailed evaluation of the system. The other indexes are also useful for plant performance and function evaluation. In combination with the CPI, these indexes are helpful in identifying the main factors responsible for the observed behavior of CPI when, for example, comparing a number of scenarios with different set points for a certain actuator.

A critical comment on the investigated operating point of the benchmark WWTP should be given. The relatively low C to N ratio in the inlet wastewater causes difficulties for the N removal process, since there is not sufficient carbon source available in the incoming wastewater to achieve a high denitrification efficiency, resulting in high effluent nitrate nitrogen concentrations. This fact was also reflected by the steady state sensitivity analysis to model parameters and control handles presented in Mussati et al. (2002). An extra actuator for control, i.e. a carbon source dosage system, should be the first thing to add to the plant to improve its operation, and could be investigated in future work. Recently, Carlsson and Rehnström (2001) presented a feedforward–feedback external carbon flow rate controller for the same denitrifying WWTP to improve the denitrification process efficiency.

Table 2. Comparisons between ASM1 and ASM3 predictions for the investigated benchmark WWTP

Control handle	Similarities	Differences	
		ASM1	ASM3
SRR	Rain scenario exhibits the highest EQ, PE, and CPI, and the lowest P_{sldg} and η.	ASM1 shows slightly flatter profiles for EQ, η, and CPI for increased SRRs for all scenarios.	ASM3 predicts higher EQ at low SRRs.
	For all scenarios, doubling SRR does not substantially change EQ but ratios lower than 0.75 cause severe effluent quality deterioration (high EQ) and low η. No optimal SRR values were obtained for EQ, η, and CPI in the range analyzed. No significant effects from changing the SRR setpoint from the nominal value are predicted.		ASM3 exhibits higher P_{sldg} and CPI along the whole range studied.
NRR	As NRR decreases higher EQ are predicted. Increased NRR above the default value (3.0) does not significantly improve the effluent quality.	For all scenarios, P_{sldg} is essentially insensitive in the range analyzed.	For all scenarios, P_{sldg} decreases slightly as NRR increases.
	For all scenarios, no optimal NRR values were obtained for EQ, η, and CPI in the range analyzed. Moreover, η shows a rather flat profile.		Clearly, ASM3 predicts higher P_{sldg} over the whole range studied.
	CPI is mainly influenced by pumping cost. As NRR decreases, the reduction in pumping cost is larger than the increased effluent fines. For this case, NRR can be reduced from its default value.		ASM3 predicts higher CPI, mainly at low NRRs.
OTR	For all scenarios, optimal OTR for CPI is lower compared to EQ and η criteria. Considering CPI, the aeration rate can be reduced by 10% with respect to the nominal value.	P_{sldg} increases as K_La increases.	P_{sldg} decreases as K_La increases. ASM3 predicts higher EQ and CPI over the whole range studied. η ranges from 76 to 80% for all the scenarios by varying K_La ±10% around the nominal value.

Comparison of steady state and dynamic simulation results

The steady state sensitivity analysis of the ASM1 and ASM3 models to the control handles presented in Mussati et al. (2002) showed high ASM3-to-ASM1 *relative* sensitivity values for the oxygen transfer rate and for both the waste sludge flow rate and sludge recycle flow rate ratio. The relative sensitivities were close to 1 for the nitrate recycle flow rate ratio. Inspecting the *individual* sensitivities, the oxygen transfer rate presented the highest values for most components for both models.

Here, EQ is a weighted sum of the effluent component concentrations. As such, there exists a relation between the steady state sensitivities and EQ. Despite this relation, it is rather difficult to extrapolate or generalize conclusions from the steady state sensitivity analysis to the effluent quality criterion during the different dynamic scenarios, since the dynamic sensitivities are different from the static sensitivities. However, some common patterns can still be recognized from both analysis methods.

Figure 13b shows that the EQ profiles as a function of the nitrate recycle ratio obtained for both models are flatter than the EQ profiles for the oxygen transfer rate and the sludge recycle ratio. This indicates a comparatively lower influence of the nitrate recycle ratio upon this index. This result is in agreement with the lowest individual steady state sensitivity values predicted for this control handle by the two models (Mussati et al. 2002).

For both models, when considering the nominal K_La value, at which the sensitivity analysis was performed, Fig. 13a shows that an increase of the K_La value results in a steeper slope of the EQ profile compared to the other control handles. (The nominal values for the sludge and nitrate recycle ratios at which the sensitivity analysis was performed are 1.0 and 3.0, respectively.) This result is in accordance with the larger individual sensitivity values predicted by both models for the oxygen transfer rate compared to the other control handles. The higher EQ slope for the ASM3 model compared to ASM1 is in accordance with the ASM3-to-ASM1 relative sensitivity values larger than 1 for this control handle. It can thus be concluded that the steady state analysis of sensitivity to control handles gives a good indication of the effect of the control handles on the effluent quality criterion and, hence, on the cleaning performance.

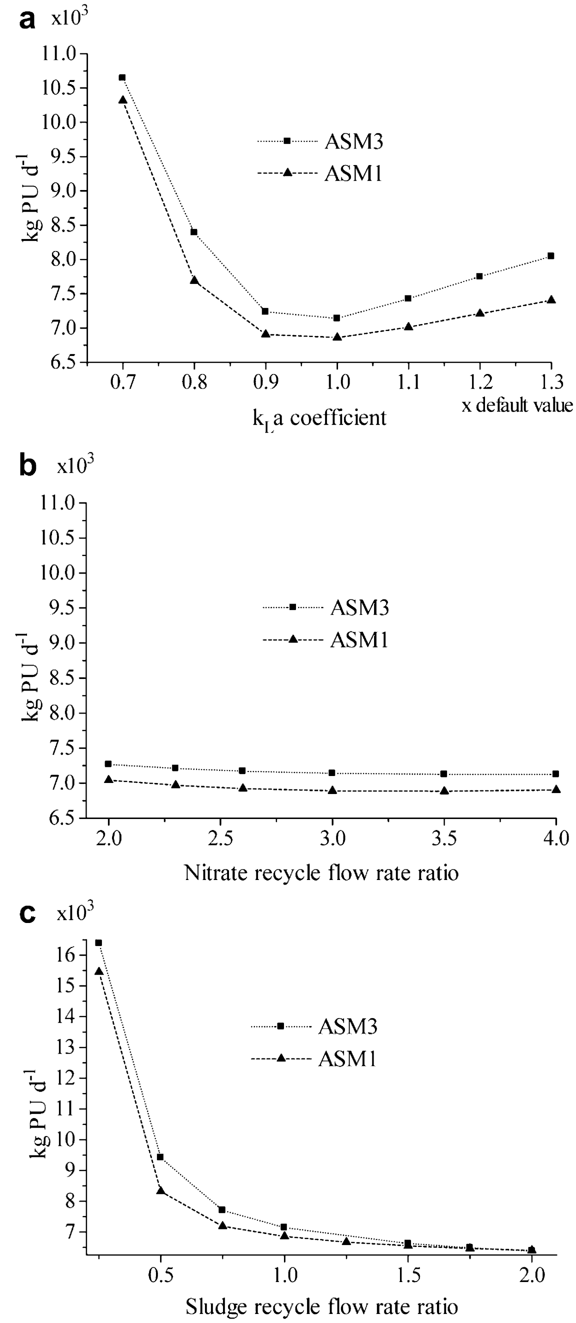

Fig. 13a–c. Comparison of the effluent quality index predicted by the ASM1 and ASM3 models for the dry weather scenario. **a** Oxygen transfer rate; **b** Nitrate recycle flow rate ratio; **c** Sludge recycle flow rate ratio

Conclusions

The effects of manipulating control handles (oxygen transfer rate, sludge recycle and nitrate recycle flow rate ratios) on the performance of a specific nitrogen removal wastewater treatment plant were analyzed under different dynamic disturbance scenarios (dry, rain, and storm weather conditions) by a selection of performance criteria.

For this case study, and for all the weather scenarios analyzed, the aeration rate is the most sensitive control handle, and the nitrate recycle flow rate ratio is the least sensitive of the three control actuators analyzed.

Different qualitative predictions were obtained from the ASM1 and ASM3 models. For all the weather scenarios, the sludge production predicted by the ASM1 model increases slightly as the oxygen transfer rate increases (Fig. 6a). In contrast, the sludge production decreases for the ASM3 model at increasing K_La coefficient values (Fig. 6b). However, the resulting difference in the sludge treatment cost is not sufficiently large to qualitatively modify the cost performance index profile.

The cost performance index is probably the most appropriate criterion to evaluate the plant performance, whereas the other investigated plant performance criteria are very useful to explain the observed behavior of the cost performance index.

For all the control handles analyzed, the rain weather scenario showed the highest cost performance index and lowest global cleaning efficiency.

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