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Including greenhouse gas emissions during benchmarking of wastewater treatment plant control strategies

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ABSTRACT

The main objective of this paper is to demonstrate how greenhouse gas (GHG) emissions can be quantified during the evaluation of control strategies in wastewater treatment plants (WWTP). A modified version of the IWA Benchmark Simulation Model No 2 (BSM2G) is hereby used as a simulation case study. Thus, the traditional effluent quality index (EQI), operational cost index (OCI) and time in violation (TIV) used to evaluate control strategies in WWTP are complemented with a new dimension dealing with GHG emissions. The proposed approach is based on a set of comprehensive models that estimate all potential on-site and off-site sources of GHG emissions. The case study investigates the overall performance of several control strategies and demonstrates that substantial reductions in effluent pollution, operating costs and GHG emissions can be achieved when automatic control is implemented. Furthermore, the study is complemented with a scenario analysis that examines the role of i) the dissolved oxygen (DO) set-point, ii) the sludge retention time (SRT) and iii) the organic carbon/nitrogen ratio (COD/N) as promoters of GHG emissions. The results of this study show the potential mechanisms that promote the formation of CO_2, CH_4 and N_2O when different operational strategies are implemented, the existing synergies and trade-offs amongst the EQI, the OCI and TIV criteria and finally the need to reach a compromise solution to achieve an optimal plant performance.

1. Introduction

The increasing demands on effluent quality at lower operational costs have promoted the development of new technologies and the implementation of control concepts to improve the overall performance of wastewater treatment plants (WWTP). Full-scale applications have shown the feasibility of automatic control in aeration systems, chemical dosage and recycle flows (Oennerth et al., 1996; Ingildsen et al., 2002; Devisscher et al., 2002; Olsson et al., 2005). Dynamic simulation studies have also been used to compare the performance of different control strategies (Zhao et al., 1995; Spanjers et al., 1998; Corominas et al., 2006; Stare et al., 2007; Flores-Alsina et al., 2009; Machado et al., 2009) or to evaluate them before full-scale implementation (Ayesa et al., 2006). Plant-wide operation has also been introduced to take into account the interactions between the processes (Gujer and Erni, 1978; Lessard and Beck, 1993; Jeppsson et al., 2007). However, the increasing interest for greenhouse gas (GHG) emissions from wastewater treatment leads to re-think the traditional engineering approaches by adding this new dimension. Therefore, new tools are needed to estimate the GHG emissions and evaluate different operation schemes that prevent or minimize their generation in WWTP.

During the last years, the scientific community has developed some mathematical tools to estimate/evaluate the

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Nomenclature

A	alternative
ADM	anaerobic digestion model
AE	aeration energy ($kWh \cdot day^{-1}$)
AER	aerobic section
AOB	ammonium oxidizing bacteria
ANOX	anoxic section
ASM	activated sludge model
BOD	biochemical oxygen demand ($g\ m^{-3}$)
BSM2	Benchmark Simulation Model No 2
CH_4	methane ($kg\ CH_4 \cdot day^{-1}$)
CO_2	carbon dioxide ($kg\ CO_2 \cdot day^{-1}$)
CO_{2e}	equivalent carbon dioxide ($kg\ CO_{2e} \cdot day^{-1}$)
COD	chemical oxygen demand ($g\ m^{-3}$)
DO	dissolved oxygen concentration ($g\ m^{-3}$)
EC	consumption of external carbon source ($kgCOD \cdot day^{-1}$)
EQI	effluent quality index ($kg\ pollution \cdot day^{-1}$)
GHG	greenhouse gas
HE^{net}	net heating energy ($kWh \cdot day^{-1}$)
K_{FNA}	inhibition constant for free nitrous acid ($g\ N\ m^{-3}$)
K_La	volumetric oxygen transfer coefficient (day^{-1})
ME	mixing energy ($kWh \cdot day^{-1}$)
MP	methane production ($kgCH_4 \cdot day^{-1}$)
N	nitrogen
NH_4^+	ammonium ($g\ N\ m^{-3}$)
NO	nitric oxide ($g\ N\ m^{-3}$)
N_2O	nitrous oxide ($g\ N\ m^{-3}$)
NOB	nitrite oxidizing bacteria
NO_2^-	nitrite ($g\ N\ m^{-3}$)
NO_3^-	nitrate ($g\ N\ m^{-3}$)
PE	pumping energy ($kWh \cdot day^{-1}$)
PI	proportional integral controller
Q_e	effluent flow ($m^3 \cdot day^{-1}$)
Q_{carb}	external carbon source flow rate ($m^3 \cdot day^{-1}$)
Q_{intr}	internal recycle flow rate ($m^3 \cdot day^{-1}$)
Q_r	external recirculation flow rate ($m^3 \cdot day^{-1}$)
Q_W	waste flow rate ($m^3 \cdot day^{-1}$)
SP	sludge production ($kgTSS \cdot day^{-1}$)
SRT	sludge retention time (days)
TIV	time in violation (%)
TKN	total Kjeldahl nitrogen ($g\ m^{-3}$)
TN	total nitrogen ($g\ m^{-3}$)
TSS	total suspended solids ($g\ m^{-3}$)
WWTP	wastewater treatment plant

generation of GHG in WWTP. However, these methods appear to be unsuitable to evaluate WWTP control strategies for several reasons. First of all, the current approaches are based on steady state calculations, i.e. empirical approaches (e.g. IPCC, 2006; LGO, 2008; NGER, 2008) or comprehensive models (Bridle et al., 2008; Bani Shahabadi et al., 2009; Pagilla et al., 2009), without taking into account the wastewater treatment dynamics. Thus, it is not possible to consider how changes in the influent load (daily, weekly, seasonal), temperature (winter/summer) and operating conditions (DO, SRT, COD/N ratios,) influence the production/emission of GHG. Second of all, some of these estimations are focused on particular wastewater sections/compounds/technologies and do not consider the whole. For example, Cakir and Stenstrom (2005), Keller and Hartley (2003) and Monteith et al. (2005) study the contribution of aerobic degradation of carbonaceous biochemical oxygen demand (CBOD) to GHG emissions. Other investigations such as those by von Schulthess and Gujer (1996); Hiatt and Grady (2008a,b), and Foley et al. (2010) quantify the N_2O emissions in aerobic-anoxic activated sludge plants. Batstone et al. (2002) and Greenfield and Batstone (2003) evaluated methane (CH_4) and carbon dioxide (CO_2) emissions under anaerobic conditions. Nevertheless, with the aforementioned approaches it is difficult to have the overall carbon footprint in terms of CH_4, CO_2 and N_2O when the plant is running under a certain operational mode. Third of all, none of these approaches include multi-criteria evaluation combining GHG emissions information with rigorous quantification about the effluent quality and operational costs, enabling to quantify and compare their overall sustainability.

In order to overcome these limitations and to include the GHG emissions during the evaluation of WWTP control strategies, the authors suggest combining pseudo-empirical equations and mechanistic models. This approach explores influent, effluent and operational variables at each simulation step. Then operational procedures or control strategies that may cause favorable conditions for GHG emissions can be quantified, providing an additional dimension to the traditional effluent quality, economical and legal criteria.

The main objective of this paper is therefore to demonstrate how GHG can be quantified during the evaluation of control strategies. This paper details the rationale of how the different sources of CO_2, CH_4 and N_2O can be taken into account dynamically within the evaluation procedure. The performance of this approach is then evaluated alongside a number of simulated scenarios where a modified version of the IWA Benchmark Simulation Model (BSM2G) is studied under open-loop and closed-loop regime.

2. Methods

2.1. *Wastewater treatment plant under study and evaluation criteria*

The WWTP under study has the same layout as the IWA BSM2 proposed by Nopens et al. (2010) (see a schematic representation in Fig. 1). The activated sludge unit is a modified Ludzack-Ettinger configuration consisting of five tanks in series. Tanks 1 (ANOX1) and 2 (ANOX2) are anoxic with a total volume of 3000 m^3, while tanks 3 (AER1), 4 (AER2) and 5 (AER3) are aerobic with a total volume of 9000 m^3. The circular secondary settler (SEC2) has a surface area of 1500 m^2 with a total volume of 6000 m^3. The BSM2 plant also contains a primary clarifier (PRIM), a sludge thickener (THK), an anaerobic digester (AD), a storage tank (ST) and a dewatering unit (DW). The primary clarifier is modeled in accordance with Otterpohl and Freund (1992) and Otterpohl et al. (1994). A modified version of the ASM1 (Henze et al., 2000), based on the

ASMN suggested by (Hiatt and Grady, 2008a) is chosen as the biological process model. This model incorporates two nitrifying populations – ammonia oxidizing bacteria (AOB) and nitrite oxidizing bacteria (NOB) – using free ammonia and free nitrous acid, respectively as their substrates. The model incorporates also four step denitrification, (sequential reduction of nitrate to nitrogen gas via nitrite, nitric oxide, and nitrous oxide), using individual reaction specific parameters. The parameter values suggested in Hiatt and Grady (2008a) were used, except for the K_{FNA} (inhibition constant for free nitrous acid) that was reduced from 1 10^{-4} (used for high nitrogen loads) to 1 10^{-6} g m^{-3} (used for low nitrogen loads) to promote NOB growth (Snip, 2010; Corominas et al., 2010). To account for seasonal variability, liquid-gas saturation constants, kinetic parameters, transfer coefficients and equilibrium reactions are temperature dependent. Stripping equations for the gases were implemented as in Foley et al. (2010). The double exponential settling velocity function of Takács et al. (1991) is used to model the secondary settling process through a one-dimensional model consisting of ten layers. Regarding the thickener and dewatering units, these are modeled as ideal, continuous processes with no biological activity, and a constant percentage of TSS in the concentrated sludge flows leaving the thickening and dewatering units. The widely recognized Anaerobic Digestion Model No. 1 (ADM1) (Batstone et al., 2002) is the dynamic anaerobic digestion model implemented. The interfaces presented in Nopens et al. (2009) have been modified (see Corominas et al., 2010) to link the biological model and ADM1, by considering COD and N balances for all oxidized nitrogen compounds. Finally, the influent wastewater composition follows the principles outlined in Gernaey et al. (2006). Further information about the BSM2 layout and the description of the process models can be found in Jeppsson et al. (2007).

A set of evaluation criteria is used to compare the simulation results in the BSM2. The overall pollution removal of the plant is obtained by calculating the effluent quality index (EQI), which is expressed in units of kg pollution·day^{-1}. Compared to Nopens et al., (2010), the EQI was modified to include the different oxidized nitrogen forms ($NO_x = NO_3^- + NO_2^- + NO + N_2O$) on the receiving water (see Eq. 1):

$$EQI = \frac{1}{(t_2 - t_1)\cdot 1000}\int_{t_1=245\,days}^{t_2=609\,days} (B_{SS}\cdot TSS_e(t) + B_{COD}\cdot COD_e(t) + B_{TKN}\cdot TKN_e(t) + B_{NOX}\cdot NO_{X,e}(t) + B_{BOD5}\cdot BOD_e(t))Q_e(t)\cdot dt \quad (1)$$

where TSS_e, COD_e, TKN_e, $S_{NOx,e}$ and BOD_e represent, respectively, the total suspended solids, the chemical oxygen demand, the total Kjeldahl nitrogen, the oxidized nitrogen concentration and the biochemical oxygen demand in the effluent. Q_e is the effluent flow rate and t time. The weights for the different pollutants are: $B_{SS} = 2$, $B_{COD} = 1$, $B_{TKN} = 30$, $B_{NOx} = 10$ and $B_{BOD5} = 2$.

The operational cost index (OCI) is calculated as the weighted sum of the aeration energy (AE), the pumping energy (PE), the consumption of external carbon source (EC), the sludge production (SP), mixing energy (ME), heating energy (HE) and methane production (MP) as shown in Eq. 2 (Nopens et al., 2010).

$$OCI = AE + PE + 3\cdot EC + 3SP + ME - 6MP + \max(0, HE^{net}) \quad (2)$$

The final part of the evaluation procedure involves the calculation of the percentage of time (%) that the plant is in violation for five different pollutants: NH_4^+ (4 g N m^{-3}), TN (18 g N m^{-3}), COD (100 g COD m^{-3}), BOD (10 g COD m^{-3}) and TSS (30 g TSS m^{-3}). More details about the "time plant in violation" (TIV) criterion can be found in Copp (2002).

2.2. Estimation of the greenhouse gas (ghg) emissions

The comprehensive approach suggested by Monteith et al. (2005) and extended in Bridle et al. (2008) is used to estimate all potential GHG emissions from the studied WWTP that cannot be obtained from the explicit results of the modified BSM2. A comprehensive description of the methodology can be found in Snip (2010) and Corominas et al. (2010).

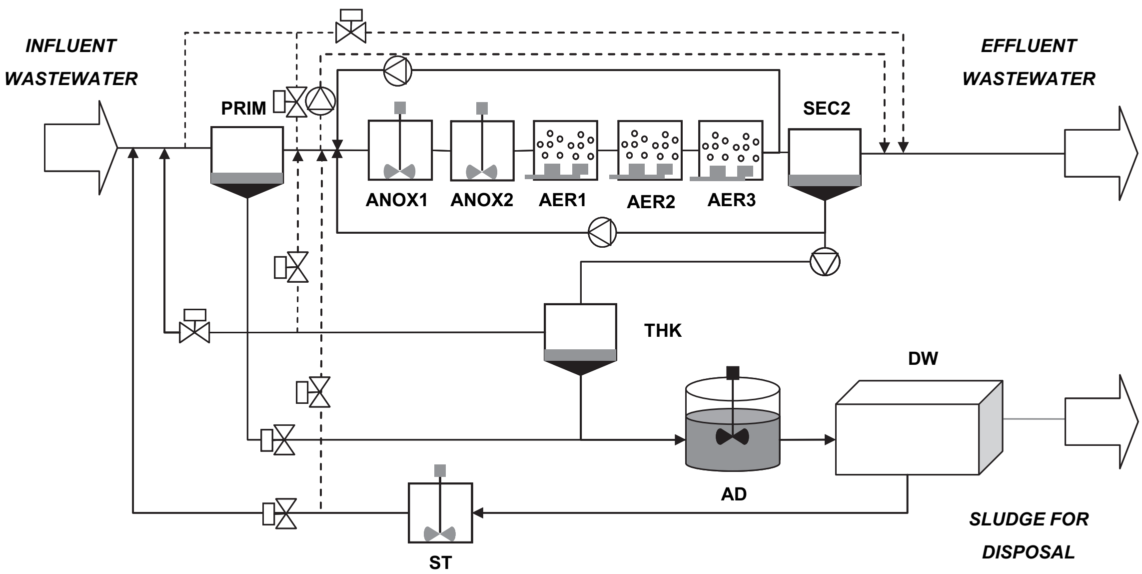

Fig. 1 – Flow scheme of the treatment plant under study.

The overall GHG evaluation comprises the estimation of the following GHG emissions: i) direct secondary treatment, ii) sludge processing, iii) net power, iv) embedded GHG emissions from chemical use and finally v) sludge disposal and reuse. In order to deal with the different nature of the generated GHG (CO_2, CH_4 and N_2O), they are converted in units of CO_2 equivalent (CO_{2e}). It is important to highlight that the presented procedure only takes into account GHG emissions produced within the WWTP limits.

- *Direct secondary treatment emissions*. The emission from the activated sludge section includes the CO_2 generated from biomass respiration and BOD oxidation, the N_2O generated from nitrogen removal and the CO_2 credit from nitrification. The first two processes are estimated following the methodology proposed by Monteith et al. (2005). N_2O emissions are given by the modified ASMN model. It is important to highlight that N_2O production is only considered during heterotrophic denitrification. Other important pathways like N_2O production during nitrification are not considered in this study (see discussion). Finally, the credit from nitrification is calculated with the factor 0.31 kg of CO_2consumed (kg N nitrified)$^{-1}$ (Tchobanoglous et al., 2003).
- *Sludge processing*. The emissions of GHG during sludge treatment are mainly generated in the anaerobic digester. Direct biogas CO_2 and CH_4 emissions are quantified using ADM1. In this case it is assumed that the biogas is fed directly into a gas-fired combustion turbine converting the CH_4 into CO_2 and generating electricity and heat (in turn used to heat the anaerobic digester). The CO_2 generated during anaerobic digestion and the CO_2 produced in the combustion are released to the atmosphere.
- *Net power GHG*. The total energy consumption is quantified using the OCI defined in Eq 2. The credit refers to the electricity generated by the turbine and it is calculated by using a factor for the energy content of the methane gas (50,014 MJ (kg CH_4)$^{-1}$) and assuming a 43% efficiency for electricity generation, which is a reasonable value according to Saravanamuttoo et al. (2009). The net power is the difference between the total energy consumption and the credit.
- *Chemicals*. The embedded GHG emissions associated with chemicals used at the WWTP have been limited to the external carbon source. These emissions are estimated by using the emission factor of 1.54 g $CO_{2e}\cdot$g methanol^{-1} (Dong and Steinberg, 1997).
- *Sludge disposal and reuse*. CO_2 emissions associated with trucking of bio-solids are quantified by multiplying the truck movements by the distance to the reuse (150 km to agriculture, 20 km to compost and 144 km to forestry). The CO_2 emissions by mineralization are calculated based on the sludge mass times the carbon concentration times the factor of CO_2 to carbon. It is assumed that 38% of sludge goes to agriculture, 45% to a compost site and 17% to forestry (Bridle et al., 2008).

2.3. *Implemented control strategies*

A default open-loop control strategy (A_0), referred to as BSM2G open-loop control, is the reference case and has the following characteristics: Q_{intr} = 61,944 $m^3\cdot day^{-1}$ (internal recycle flow rate), $Q_W = 400\ m^3\cdot day^{-1}$ (waste flow rate), $Q_r = 20{,}648\ m^3\cdot day^{-1}$ (external recirculation flow rate), Q_{carb} = 5 $m^3\cdot day^{-1}$ (external carbon source addition rate) and $k_La1 = k_La2 = 2\ day^{-1}$ and $k_La3 = k_La4 = k_La5 = 140\ day^{-1}$ (aeration intensity, represented as the volumetric oxygen transfer coefficient) respectively. A low K_La is expected in the anoxic zone (and consequently some stripping) as a side-effect of mixing (von Schulthess and Gujer, 1996). Next, three different control strategies (A_1, A_2 and A_3) are implemented and compared to the base case. The first strategy (A_1) is based on a simple PI loop controlling the dissolved oxygen (DO) in the 2nd aerobic tank (AER2) through manipulation of the aeration flow in AER1,2 and 3 (K_La3, 4 and 5) (set-point = 2 g O_2 m^{-3}). K_La5 is set to half the value of K_La3 and K_La4. In A_2, a controller of the nitrate (NO_3^-) in the 2nd anoxic tank (ANOX2) manipulating the internal recycle flow rate (Q_{intr}) is added to A_1 (set-point = 1 g N m^{-3}). Finally, A_3 completes A_2 with a cascade PI ammonium (NH_4^+) controller that manipulates the DO set-point (set-point = 2 g N m^{-3}). Additionally, in A_1, A_2 and A_3 two waste flow rates in SEC2 are imposed depending of the time of the year in order to keep the biomass in the system during the winter period, i.e. starting/end date, (Q_W = 300 m^3 day^{-1}). For the rest of the simulation time the waste flow is set to 450 m^3 day^{-1}.

The DO sensor is assumed to be close to ideal with a response time of 1 min in order to prevent unrealistic control applications. The NO_3^- and the NH_4^+ sensors have a time delay of 10 min, with zero mean white noise (standard deviation of 0.5 gN m^3) (Rieger et al., 2003). Finally, the aeration system (K_La), is defined with significant dynamics assuming a response time of 4 min. As discussed in Corominas et al. (2010) all dynamic simulations (609 days) are preceded by a steady state simulation (200 days) but only the data generated during the final 364 days (t) of dynamic simulation are used for plant performance evaluation.

3. Case study control evaluation

The results obtained for the open (A_0) and closed-loop (A_1, A_2 and A_3) simulations have been evaluated with respect to effluent quality, operating costs, legal criteria and GHGs production (see Table 1 and Table 2). Compared to alternative A_0, the DO controller implemented in alternatives A_1, A_2 and A_3 substantially reduces the aeration energy (AE) and the OCI by up to 6%. In the same order of magnitude (up to 7%) EQI is also improved in all closed-loop strategies. This is mainly due to a more efficient use of the aeration system adapting the airflow rate to the oxygen demand for organic matter and nitrogen removal. The implementation of a NO_3^- controller in alternatives A_2 and A_3 slightly improved denitrification efficiency and reduced both the effluent total nitrogen concentration (TN) and the percentage of violation in terms of TN (TIV_TN). Since the internal recycle (see the increase of PE in Table 1) between AER3 and ANOX1 is strongly influenced by the influent load and process performance, the NO_3^- controller ensures an optimal use of the incoming organic matter for denitrification. Finally, the effect of the NH_4^+ controller implemented in A_3 must be mentioned. The variation of the DO set-point in AER2 according to the nitrification needs substantially reduces the ammonium peaks (see TIV_NH_4^+ in

Table 1 – Effluent quality, operational cost and legal criteria for the different control strategies.

	A_0	A_1	A_2	A_3	Units
Total Kjeldahl Nitrogen (TKN)	3.83	3.97	4.05	3.99	$g N m^{-3}$
Total Nitrogen (TN)	15.04	13.62	13.12	12.39	$g N m^{-3}$
Chemical oxygen demand (COD)	49.19	41.73	49.71	49.76	$g COD m^{-3}$
Biochemical oxygen demand (BOD_5)	3.14	3.16	3.15	3.17	$g COD m^{-3}$
Total suspended solids (TSS)	17.51	15.31	15.31	15.31	$g TSS m^{-3}$
Effluent quality index (EQI)	**6448**	**6239**	**6172**	**5995**	**$kg poll \cdot day^{-1}$**
Sludge production (SP)	2703	2674	2673	2674	$kg TSS \cdot day^{-1}$
Aeration energy (AE)	5627	4843	4821	5048	$kWh \cdot day^{-1}$
Pumping energy (PE)	447	442	467	496	$kWh \cdot day^{-1}$
External Carbon addition (EC)	2000	2000	2000	2000	$kg COD \cdot day^{-1}$
Mixing energy (ME)	768	768	768	771	$kWh \cdot day^{-1}$
Net Heating energy (HE^{net})	4289	4247	4248	4247	$kWh \cdot day^{-1}$
Methane production (MP)	1141	1126	1125	1126	$kg CH_4 \cdot day^{-1}$
Operational cost index (OCI)	**14,107**	**13,324**	**13,323**	**13,580**	–
Time in violation for TN (TIV_TN)	5.72	2.17	1.09	1.35	%
Time in violation for COD (TIV_COD)	0.06	0.07	0.06	0.06	%
Time in violation for ammonium (TIV_NH_4^+)	18.94	19.44	20.83	5.40	%
Time in violation for TSS (TIV_TSS)	0.27	0.30	0.30	0.30	%
Time in violation for BOD_5 (TIV_BOD_5)	0.23	0.23	0.23	0.23	%

Table 1) at the expense of higher aeration energy (AE). No substantial differences are observed in the rest of effluent (COD, BOD_5 and TSS) and economic (SP, EC, HE^{net} and MP) criteria.

The plant under control can reduce the GHG emissions with up to 9.6% (from 1.142 to 1.032 kg CO_{2e} m^{-3} treated wastewater). The main differences are found in the direct secondary emissions and in the power consumption (see Table 2). A significant reduction of emitted N_2O is observed in the control strategies with a DO controller (from 16.35 kg $N_2O \cdot day^{-1}$ for A_0 to 10.09, 10.92 and 13.96 kg $N_2O \cdot day^{-1}$ for A_1, A_2 and A_3) since the DO controller prevents the system from nitrite accumulation (see a detailed discussion in the following section). In addition, it is important to notice the differences between alternatives A_2 (NO_3^- controller) and A_3 (NH_4^+ controller) in terms of N_2O emissions. The performance of the NO_3^- controller is heavily affected by temperature. During winter time, when nitrification is lower, it is necessary to increase the internal recycle in order to maintain the desired set-point in the anoxic section because there is a lower production of nitrates in the aerobic zone. As a consequence, it also increases the quantity of oxygen transported from AER3 to ANOX1 worsening the overall denitrification efficiency. As N_2O is an intermediate in the denitrification process, incomplete denitrification leads to N_2O emissions. In the case of the cascade NH_4^+ controller (A_3) the increase of N_2O is due to a couple of reasons. Firstly, the sudden increase of aeration (K_La) in the aerated zones to smoothen the ammonium peaks, increases the quantity of DO returning into the anoxic zone. As in the previous case (NO_3^- controller), this situation leads to problems in the anoxic zone (incomplete denitrification). Secondly, long periods with low aeration intensity (see the concomitant increase of the mixing and pumping energy in Table 1) lead to increased nitrite accumulation due to DO

Table 2 – GHG emissions for the different control strategies.

	A_0	A_1	A_2	A_3	Units
Biomass respiration	0.180	0.191	0.191	0.191	$kg CO_{2e} m^{-3}$
BOD oxidation	0.212	0.219	0.219	0.219	$kg CO_{2e} m^{-3}$
Credit nitrification	−0.012	−0.012	−0.012	−0.012	$kg CO_{2e} m^{-3}$
N_2O emissions	0.236	0.146	0.158	0.202	$kg CO_{2e} m^{-3}$
Total direct secondary treatment emissions	**0.616**	**0.544**	**0.555**	**0.599**	**$kg CO_{2e} m^{-3}$**
CO_2 emissions from digestion	0.079	0.078	0.078	0.078	$kg CO_{2e} m^{-3}$
CH_4 emissions from digestion	0.152	0.150	0.150	0.150	$kg CO_{2e} m^{-3}$
Total Sludge processing GHG emssions	**0.231**	**0.228**	**0.228**	**0.228**	**$kg CO_{2e} m^{-3}$**
Power	0.311	0.276	0.276	0.287	$kg CO_{2e} m^{-3}$
Credit Power	−0.310	−0.306	−0.306	−0.306	$kg CO_{2e} m^{-3}$
Total Net power GHG emissions	**0.001**	**−0.030**	**−0.030**	**−0.019**	**$kg CO_{2e} m^{-3}$**
Total Embedded GHG emissions from Chemicals use	0.099	0.099	0.099	0.099	$kg CO_{2e} m^{-3}$
Total Sludge disposal and reuse GHG emissions	0.194	0.191	0.191	0.191	$kg CO_{2e} m^{-3}$
Total GHG emissions (GHG total)	**1.142**	**1.032**	**1.044**	**1.100**	**$kg CO_{2e} m^{-3}$**

limitation of NOB activity. Again, an increased nitrite concentration favors N_2O production.

A more efficient aeration system in A_1, A_2 and A_3 reduces the off-site CO_2 emissions due to lower power consumption (as already mentioned for the operational cost index). Note that the implementation of these three basic controllers does not suppose any change in the addition of external carbon source. Surprisingly, the different waste flow patterns in the closed-loop strategies do not have a big impact on the CO_2 derived from sludge treatment and reuse. For this reason the GHG emissions due to sludge processing, sludge disposal and reuse and the embedded emissions from chemicals use remain almost at the same value. Finally, it has to be mentioned that the GHG estimations obtained in this study (0.87–1.00 kg $CO_{2e}\cdot m^{-3}$) are within the range of values presented in Bridle et al. (2008) (0.9, 1.6 and 2.2 kg $CO_{2e}\cdot m^{-3}$) and in Pagilla et al. (2009) (from 0.34 to 1.25 kg m^{-3}).

4. Scenario analysis

A scenario analysis, which focuses on analyzing GHG emissions as well as the previously defined environmental, economical and legal criteria, is also included to investigate which variables are worth looking at for control. Therefore, for exemplary purposes, we analyze how the results of alternative A_2 are affected by changing some of its settings in three scenarios. *Scenario 1* evaluates the plant performance at the DO set-points of 1 and 3 g $O_2\cdot m^{-3}$. *Scenario 2* changes the sludge retention time by either increasing (Q_W = 400/550 $m^3\cdot day^{-1}$) or decreasing (Q_W = 200/350 $m^3\cdot day^{-1}$) the winter/summer waste flow patterns. Finally, in *Scenario 3* the COD/N ratio is changed by modifying the dosage of external carbon source (Q_{carb}) from 5 $m^3\cdot day^{-1}$ to 0 and 10 $m^3\cdot day^{-1}$. First, the effect on the GHG emissions of the three scenarios is studied in subsection one, two and three. In the fourth subsection, the conflicting results for the other criteria are discussed.

4.1. *Scenario 1: effect of the DO concentration in the aerated section*

The airflow rate in the aerobic section (AER) is essential in activated sludge treatment because it promotes the growth of the heterotrophic and autotrophic bacateria that will oxidize organic matter and nitrogen. Hence, *Scenario 1* examines the effect of the overall GHG emissions when the dissolved oxygen set-point is above and below the default value of 2 g O_2 m^{-3}. In order to do such analysis, the plant is simulated under different airflow rates and the emitted GHG quantified for each simulation. The results of the simulations show that low DO concentrations (Fig. 2, DO = 1 g O_2 m^{-3}) lead to a reduction of the CO_2 production thanks to the lower energy consumption but a very slight increase of the direct secondary treatment emissions compared to the default A_2. The responsible of the increase are the higher N_2O emissions caused by the accumulation of NO_2^- (Fig. 3b) due to incomplete nitrification (Fig. 3a). Lower DO concentrations in the nitrification reactor cause growth limitation of AOB and especially NOB. Thus, the resulting high NO_2^- concentrations in the anoxic reactor transported by the internal recycle lead to lower denitrification rates and accumulation of NO and N_2O. Fig. 3 presents one year of data starting July 1st (day 245) and with the summer holiday period (where the load from industries is significantly reduced) between days 270 and 300. The yearly dynamics also show an increase in the ammonia concentration in the winter period with a consequent increase of the nitrite accumulation. At a DO set-point of 3 g O_2 m^{-3} (Fig. 2) the production of CO_2 is increased due to higher energy consumption (see net power values). In addition, more N_2O is released due to incomplete denitrification caused by recirculation of DO from the aerobic to the anoxic reactor. As a consequence there is another increase of the overall secondary treatment emissions compared to the base case. No substantial changes are observed in GHG due to sludge treatment, sludge reuse or embedded use of chemicals. The large impact of the dissolved oxygen concentration in both N_2O and CO_2 emissions indicates that process control is required in the

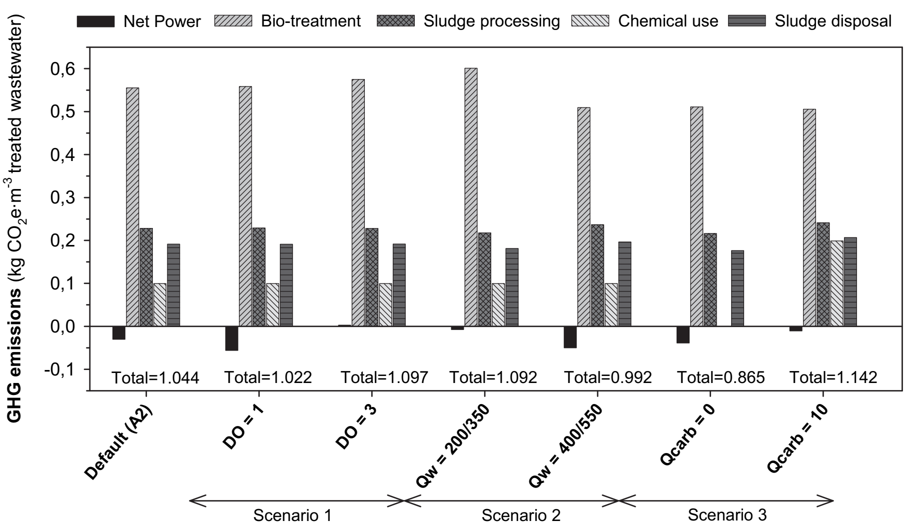

Fig. 2 – Breakdown of the GHG emissions for the different evaluated scenarios.

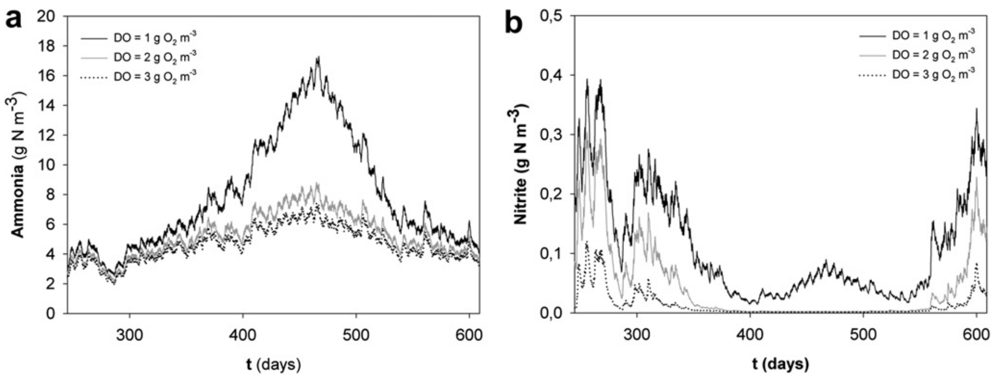

Fig. 3 – Dynamic evolution of the effluent NH_4^+ (a) and NO_2^- (b) in AER1 when the DO set-point is changed.

nitrification tanks with a "moderate" set-point around 2 g O_2 m^{-3}. In that way, complete nitrification is ensured, at the same time avoiding oxygen returns via internal recycle.

4.2. Scenario 2: effect of the sludge retention time (SRT)

The second scenario (*Scenario* 2) analysis investigates how the emissions of GHG change when the winter/summer waste flow rate (Q_W) pattern is modified. At lower SRT (Q_W = 400/550 m^3 day^{-1}, SRT around 12 days) there is a slight increase of the GHG emissions due to sludge treatment and disposal because the amount of TSS going to the sludge line increases (Fig. 4a) compared to the default A_2 (Q_W = 300/450 m^3 day^{-1}) It is important to highlight that the additional CH_4 in the digester (Fig. 4b) comes with higher energy credit, *i.e.* less off-site CO_2 generation, because more energy can be produced from the digester biogas. Note that it is possible to appreciate the changes in the quantity of TSS going to the sludge line and the methane produced in the digester during the summer and winter periods. In this specific case, SRT (together with dissolved oxygen concentration) is long enough for nitrite oxidation to proceed. Thus, a potential increase of N_2O production is avoided as it has been reported in other studies (Hanaki et al., 1992). At higher SRT (Q_W = 200/350 m^3 day^{-1}, SRT around 18 days) there is an increase of the non-N_2O GHG emission produced in the bio-reactor, off-site emission due to electrical use i.e. higher aeration intensities and less energy credit produced in the anaerobic digester (see Fig. 2). Thanks to these two factors it can be concluded that for this specific system lower sludge ages (high waste flow patterns) allow reducing GHG emissions i.e. N_2O is non crucial, less energy consumptions, more energy credit.

4.3. Scenario 3: effect of the COD/N ratio

The last scenario (*Scenario* 3) examines plant performance at different COD/N ratios in the biological reactor by modifying the addition of the external carbon source flow rate (Q_{carb}). As in the previous cases, higher (Q_{carb} = 10 $m^3 \cdot day^{-1}$) and lower (Q_{carb} = 0 $m^3 \cdot day^{-1}$) values are compared to the default conditions (Q_{carb} = 5 m^3 day^{-1}). A higher COD/N ratio (Q_{carb} = 10 $m^3 \cdot day^{-1}$) substantially increases GHG emissions in the secondary treatment (see Fig. 2) *i.e.* from biomass respiration and BOD oxidation. Due to the increased COD load the quantity of TSS produced is higher and thus GHG emissions due to sludge processing and disposal are also increased. Finally, the off-site CO_2 emission from chemical and energy use are very high, which make this scenario the worst in terms of GHG emissions. At low COD/N ratio (Q_{carb} = 0 $m^3 \cdot day^{-1}$) the total emissions are extremely low (17,861 kg $CO_2 \cdot day^{-1}$) compared to the default A_2 (21,558 kg $CO_2 \cdot day^{-1}$). This is due to the fact that zero emissions are associated to chemicals and there is a significant decrease of the direct secondary emissions and the energy-related emissions. However, secondary treatment GHG emission related N_2O are higher. From this study some important points can be put forth. It is confirmed

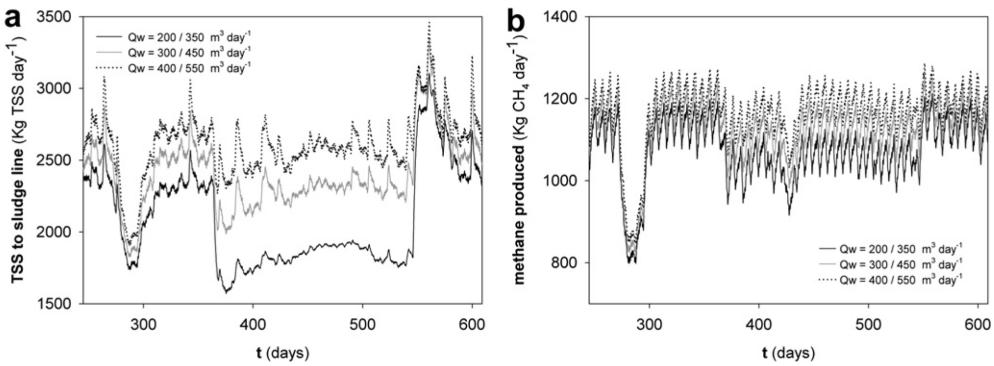

Fig. 4 – Dynamic evolution of the quantity of TSS going to the sludge line (a) and the CH4 produced in the anaerobic digester (b) when the waste flow pattern (Q_W) is changed.

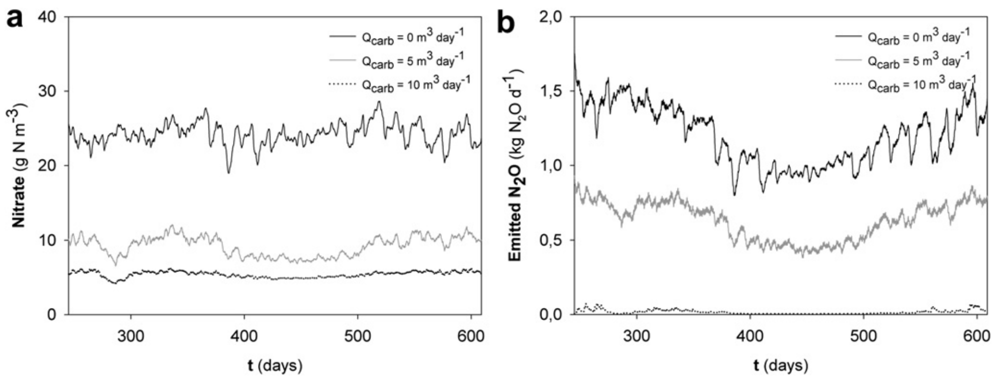

Fig. 5 – Dynamic evolution of the effluent NO_3^- (a) and N_2O released from ANOX2 (b) when the Qcarb is changed.

that limited availability of biodegradable organic carbon increases N_2O emissions during incomplete denitrification (Schultness and Gujer, 1996; Hiatt and Grady, 2008b; Kampschreur et al., 2009) (see Fig. 5a and b). However, in overall terms the decrease in endogenous respiration and BOD oxidation is more important than the increase in N_2O emitted. Thus, in terms of GHG emissions it seems that is better not to add carbon.

4.4. *Synergies and trade-offs amongst ghg emissions and effluent, cost and legal criteria*

After analyzing the influence of some operational parameters and identifying low emission control strategies, it is necessary to study their synergies and trade-offs amongst the "traditional" effluent (EQI), economic (OCI) and legal criteria (TIV).

Table 3 presents the results of the total GHG emissions with EQI, OCI and TIV for the previously defined scenarios. From these results it can be seen that the scenario with the lowest GHG emissions ($Q_{carb} = 0\ m^3 \cdot day^{-1}$) is the worst in terms of effluent quality (see values of EQI). This is mainly due to the limited biodegradable organic carbon making denitrification almost impossible (check the high values in TIV_TN). Conversely, the scenario with the highest denitrification rates promotes the highest GHG emissions and it is extremely expensive to operate due to the high carbon addition (see OCI values). The scenario with the best effluent quality involves a high DO set-point (DO = 3 g O_2 m^{-3}), but again it implies a high operation cost and GHG emissions due to increased aeration energy (see Table 3 and Fig. 2). Finally, when it comes to sludge retention time, a low waste flow rate enhances nitrification but also the operation cost due to aeration (see TIV_NH_4^+ and OCI values). On the other hand, when a lower SRT is selected it is possible to achieve relatively low operational cost and GHG emissions but the effluent quality will suffer due to a reduced nitrifying activity. It is important to highlight that for this specific case, the SRT is long enough for nitrification to proceed (see that TIV_NH_4^+ is around 33%). Thus potential problems with NO_2^- accumulation and consequently N_2O emissions are avoided. From the above it is clear that a balancing act is necessary to find the best operating conditions to satisfy the four categories of criteria evaluated here, i.e. effluent quality, economic, legal and GHG criteria.

5. Discussion

The results of this study provide several points of discussion. Firstly, from a process engineering point of view it is possible to see that to ensure sustainable wastewater treatment operation; one must guarantee a sufficiently long SRT and an adequate dissolved oxygen concentration in the biological reactor for AOB and NOB to grow. Too high oxygen set-points and SRT increase the operational cost and GHG emissions. On the other hand, too low oxygen levels and SRT provoke nitrification failures. In this case study, COD is limiting the denitrification process and addition of organic carbon is necessary to prevent eutrophication. However, too high external carbon dosage rates in the biological reactor increase the cost of operation as well as the overall GHG emissions.

Secondly, the potential adverse effects of certain operational procedures are highlighted. These effects are normally not considered with current state-of-the-art evaluation methods because they do not consider dynamics during quantification of GHG emissions in the simulation procedure. For example, the ammonium controller implemented in A_3 substantially improved the overall nitrification efficiency by adapting the airflow rate to the oxygen demand for organic matter and nitrogen removal. However, when the GHG emission were quantified, it was possible to observe an increased N_2O emission due to the long periods the cascade controller set the DO to low values. A low DO concentration in the nitrification tank will lead to increased N_2O production. This fact has important implications from a process engineering point of view. As previously stated by Kampschreur et al. (2009) the trend of WWTP to decrease their energy consumption could be adverse towards the greenhouse effect: even though it decreases CO_2 emissions from aeration energy, this could be countered by increased N_2O emissions due to its 300-fold stronger greenhouse effect.

Another interesting point comes from *Scenario 3*, where it was possible to see – just in terms of GHG emissions – that a higher external carbon source reduced the overall N_2O emissions. Nevertheless, when all the other potential GHG emissions were evaluated as well, the undesirable side-effect of such addition, i.e. an increase of the off-site CO_2 emissions due to chemical and energy use, was easily detected.

The results of such analysis depend largely on the model selected to perform the study. When modeling activated sludge plants, there is often a disagreement about the best

Table 3 – Total GHG emission, EQI, OCI and TIV values for the different evaluated scenarios.

	A_2	Scenario 1		Scenario 2		Scenario 3		Units
		DO = 1 g O_2 g m^{-3}	DO = 3 g O_2 g m^{-3}	Q_W = 200/350 m^3 day^{-1}	Q_W = 400/550 m^3 day^{-1}	Q_{carb} = 0 m^3 day^{-1}	Q_{carb} = 10 m^3 day^{-1}	
GHG total	1.044	1.022	1.097	1.092	0.992	0.865	1.142	kg CO_{2e} m^{-3}
EQI	6172	7653	5915	6484	6602	8405	6143	kg poll·day^{-1}
OCI	13,323	12,743	14,059	13,391	13,102	6515	20,393	–
TIV_TN	1.09	11.13	1.8	1.18	2.62	99.52	1.09	%
TIV_COD	0.06	0.06	0.06	0.81	0.05	0.06	0.45	%
TIV_NH_4^+	20.83	51.39	6.43	9.65	33.27	6.45	27.73	%
TIV_TSS	0.30	0.31	0.30	1.65	0.26	0.26	1.29	%
TIV_BOD_5	0.23	0.23	0.23	0.95	0.23	0.22	0.67	%

model to apply in each case. The representation of the biomass decay (Siegrist et al., 1999), oversimplification of settling models (Bürger et al., 2011; De Clercq et al., 2009) or the possible aerobic/anoxic mechanisms leading to N_2O emissions (Ahn et al., 2010; Yu et al., 2010; Lu and Chandran, 2010) are still issues under discussion. In view of the study at hand the latter is probably the most important modification to be expected from future research. No attempt was made at this stage to model AOB-related N_2O production because no consensus exists yet on the way the observations should be modeled.

In the same way, the list of possible GHG sources is not complete. For example the methane generated in the anaerobic digester that remains in the liquid phase and it is recycled to the inlet of the WWTP. Upstream and downstream sources are not considered either. For instance it is known that CH4 can be formed in the sewer system (Guisassola et al., 2009) and afterwards be stripped in the treatment plant (influent, pumping station, aeration tank). Other potential sources are N_2O emissions due to nitrogen discharges with the effluent or from sludge disposal (Ahn et al., 2010). For this reason, the reader should be aware that the results of this study depend on the assumptions made by the authors and presented in the methods section.

In that respect it should be emphasized that the objective of the approach presented in this paper is not to predict GHG emissions with absolute accuracy. The main objective of this paper is to provide a better picture of the overall WWTP performance with this new dimension dealing with GHG emissions. With the use of this platform it is now possible to see how effluent standards, economic considerations and the causes of GHG emissions are entangled. Given this complexity, the authors advocate use of multi-objective/multi-criteria evaluation techniques (Flores-Alsina et al., 2008) in order to include all these different factors during the decision making process. Also, the simulation values need to be interpreted with care. Scenarios create combinations of conditions and, for instance, the limitation of denitrification capacity may be due to a combination of SRT and DO limitations. The absolute values obtained in the scenario analyses should not be taken as such, but the qualitative results obtained can, as presented above.

Lastly, including this type of analysis in WWTP simulation studies is actively encouraged because it can give better guidance to decision makers, process engineers and wastewater professionals on the sustainability of different treatment options. Thus, it is possible to evaluate the capabilities of the control strategy or operational procedure to handle effluent, economical, legal and GHG related issues.

6. Conclusions

This paper has complemented the traditional effluent quality, operational cost and legal criteria used for evaluation of control strategies in WWTP with a new dimension dealing with GHG gases. The authors have applied this approach that evaluates and quantifies the different sources of GHG gases using dynamic modeling in different control strategies and scenarios. The key findings are summarized in the following points:

- By implementing controllers it is possible to reduce the overall GHG emissions as well as to improve effluent quality and reduce operational costs.
- A sufficiently high DO set-point is required to ensure complete nitrification, but it should not be too high as this would (i) increase oxygen recycle to the anoxic zone with N_2O emissions due to incomplete denitrification and (2) energy consumption for aeration with concomitant increase in off-site CO_2 emissions. Low concentration of oxygen could lead to high N_2O emissions due to nitrite accumulation although off-site CO_2 are reduced.
- A rather low SRT should be favored because it reduces GHG emissions by increasing the CO_2 credit due to energy production by the anaerobic digester CH_4 even though that also increased the sludge treatment related emissions. A high SRT substantially increases GHG emissions from secondary treatment.
- Increasing the carbon source addition increases the GHG emissions due to increased sludge production that results in increased endogenous respiration, sludge treatment and disposal and both chemicals and energy use. However the N_2O emissions are reduced.
- It is necessary to find a compromise solution between effluent quality, costs, legal and GHG criteria to reach sustainable modes of operation. In this case study, it was possible with a DO set-point of 2 g O_2 m^{-3} and moderate SRT and external carbon source addition rates.

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