Cost optimisation and minimisation of the environmental impact through life cycle analysis of the waste water treatment plant of Bree (Belgium)

K. De Gussem, T. Wambecq, J. Roels, A. Fenu, G. De Gueldre and B. Van De Steene

ABSTRACT

An ASM2da model of the full-scale waste water plant of Bree (Belgium) has been made. It showed very good correlation with reference operational data. This basic model has been extended to include an accurate calculation of environmental footprint and operational costs (energy consumption, dosing of chemicals and sludge treatment). Two optimisation strategies were compared: lowest cost meeting the effluent consent versus lowest environmental footprint. Six optimisation scenarios have been studied, namely (i) implementation of an online control system based on ammonium and nitrate sensors, (ii) implementation of a control on MLSS concentration, (iii) evaluation of internal recirculation flow, (iv) oxygen set point, (v) installation of mixing in the aeration tank, and (vi) evaluation of nitrate setpoint for post denitrification. Both an environmental impact or Life Cycle Assessment (LCA) based approach for optimisation are able to significantly lower the cost and environmental footprint. However, the LCA approach has some advantages over cost minimisation of an existing full-scale plant. LCA tends to chose control settings that are more logic: it results in a safer operation of the plant with less risks regarding the consents. It results in a better effluent at a slightly increased cost.

Key words | activated sludge model, full-scale, LCA, WWTP optimisation, energy saving, impact reduction

K. De Gussem (corresponding author)
T. Wambecq
J. Roels
A. Fenu
G. De Gueldre
B. Van De Steene
Department of Research and Development,
Aquafin, Dijkstraat 8,
B-2630 Aartselaar,
Belgium
E-mail: *Kris.DeGussem@aquafin.be*

INTRODUCTION

Surface water quality is of concern in many urbanised areas, and progressively more waste water treatment plants have been built as a result. In the 1970s and 1980s, waste water treatment plants (WWTPs) were built for carbon removal with mostly manual control. Later on with more stringent consents on nutrient removal, amongst others due to the urban waste water framework directive and European Water Framework Directive in Europe, plants were upgraded for nutrient removal. A parallel trend was to install online control systems in the WWTPs, which allow to fine-tune the treatment process for effluent quality. However, a well balanced online control system simultaneously allows significant cost reduction as well as reduction of the environmental footprint. This paper focuses on the optimisation of the full-scale Aquafin WWTP of Bree through PC-based Activated Sludge Modelling (ASM) using both the classic cost optimisation approach and a newer Life Cycle Assessment (LCA) approach.

Activated sludge models are nowadays widely used in merely academic research but also in operations. Since an IAWPRC task group was formed in 1982, mathematical modelling took off and was generalised. This resulted in several consecutive models capable of simulating the complex biological behaviour of biological processes in waste water. (IWA 2000; Gernaey *et al.* 2004b) Examples of studies published in literature include amongst others parameter calibration for studying biological processes (Gernaey & Jorgensen 2004a; Fall *et al.* 2009), benefit/risk analysis as a design support tool (Benedetti *et al.* 2006; Devisscher *et al.* 2006), control design (Ebner *et al.* 2010), etc.

There is a tendency towards lowering the environmental footprint as shown by public interest, the Intergovernmental Panel on Climate Change (IPCC 2007), emission reductions commitments, the broad adoption of the Kyoto protocol, etc. These are primarily focused towards the carbon footprint and

doi: 10.2166/wst.2011.027

global climate change. Life cycle assessment or LCA is a widely used mathematical approach to calculate the total environmental impact, including carbon footprint, eutrophication, acidification, materials usage, ecotoxicity, etc. (Doka 2007; Foley 2010) This environmental impact approach is gaining importance in the waste water treatment sector, for example as a support tool for decision making. It has to be noted though that there exist several methodologies for LCA such as EDIP97, EDIP2003, ReCiPe, LUCAS, IMPACT 2002 +, etc and how to apply them for waste water treatment is still under investigation.

The Aquafin waste water treatment plant (WWTP) of Bree (Figure 1) is located in the east of Flanders, Belgium. The plant has a treatment capacity of 24300 person equivalents, but is partly overloaded. WWTP Bree is able to treat 6 times the dry weather inflow biologically. It is designed to fulfil the effluent norms, but is not able to perform complete removal of nitrogen in case of peak loads. The control system is currently designed to only meet the legal norms. To investigate the opportunities for reductions in cost and environmental impact, a full scale ASM2da plant model, including all online controls, is made. This work is based on a broad range of research studies (APHA 1999; Vanrolleghem *et al.* 2003; Gernaey *et al.* 2004b; Benedetti *et al.* 2006; Devisscher *et al.* 2006; Insel *et al.* 2009), but extends the scope towards combined modelling of nutrients, power consumption, chemicals dosing, cost, and environmental footprint calculation.

The aim of this study is to evaluate LCA for ASM based WWTP optimisation and compare this strategy with classic cost optimisation. Six scenarios for plant optimisation are evaluated and discussed for this purpose.

METHODS AND IMPLEMENTATION

Plant model creation

The ASM 2da plant model is made in MATLAB/Simulink R2009a with an in-house written toolbox. An influent generator is used to generate the ASM2da compatible influent vector. To be able to predict the biological treatment process (see also Figure 1 for the plant layout), the anoxic and aeration tank, both approximately 1500 m^3 in volume, are modelled as continuous stirred tank reactors (CSTR). The two clarifiers are implemented according to the Takacs model (Daigger & Roper 1985; Takacs *et al.* 1991). All current and proposed controls for the plant are implemented in Simulink or C + + mex files for Simulink.

The model was calibrated and validated with the operational data of 2007. Where data series of up to three months for calibration are used in the literature, (Gernaey & Jorgensen 2004a; Ingildsen *et al.* 2006; Sin *et al.* 2008) we chose to use data covering a full year, 2007, for model calibration. The idea is that random events in a full-scale plant, such as external discharges, sensor failure, etc. could significantly deteriorate the model accuracy if a short reference period for calibration is used. In our modelling studies, it was observed that ASM models are more robust and accurate when using 1 year data for calibration. In addition, this is necessary to catch seasonal effects, which are frequently present, in plant operation.

Daily composite samples of influent and effluent flows were collected approximately every two weeks. The samples were analysed in the lab for BOD_5, COS, suspended solids,

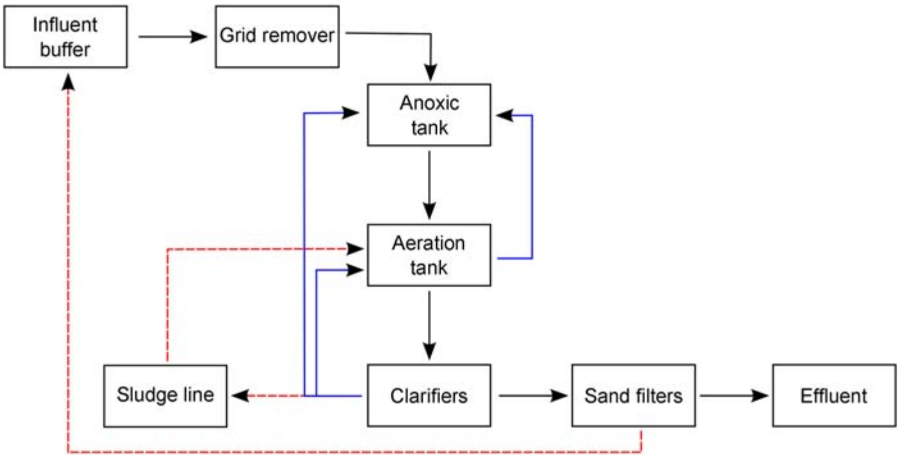

Figure 1 | Plant layout. Full lines represent water streams, whereas dashed lines represent sludge or other streams. Subscribers to the online version of *Water Science and Technology* can access the colour version of this figure from http://www.iwaponline.com/wst.

total nitrogen, Kjeldahl nitrogen, ammonia, nitrate, orthophosphate, and total phosphorus. All samples were analysed according to *Standard Methods* (APHA 1999). The sample filtration was performed with a 0.45 m polyester filter. Mixed liquor suspended solids concentration (MLSS) was also calculated by the *Standard Methods* (APHA 1999). Influent loads for each moment of the day are generated from these lab analyses of the daily composite samples, influent flows, and characteristics of the sewer system. Relevant parameters, such as MLSS in the aeration tank, flows, sludge volume index, etc. were monitored either with on-line detectors or off-line, and these measurement data incorporated in the model. Generally, the degrees of freedom that can be used for the model calibration include (i) influent fractionation, (ii) all ASM parameters, (iii) oxygenation efficiency of the aeration system. Hence, the model is first calibrated roughly for sludge production through the influent fractionation. Subsequently the aeration efficiency and influent fractionation are iteratively fine-tuned for optimal model accuracy.

Extension for the quantification of energy, cost, and environmental footprint

One of the big advantages of carefully calibrated ASM models is their ability to predict energy consumption, cost, and impact with few extra data. Hence, we made an inventory of all pumps and engines in the waterline of WWTP Bree. In essence for each engine an extra block is added to the model to include its power consumption. For each pump on the plant, pump characteristics from the technical data sheets, flow, and the water level difference were utilised to handle pump efficiency. In addition, power consumption of the main consumers (influent screw pumps, sand filter pumps, sludge recirculation, mixers in buffers, etc.) were measured and checked with a Kyoritsu Power Analyser 6310 power logger. These values are directly used in the model to quantify the power consumption for the different scenarios under investigation. In addition, devices not strictly related to the waste water treatment must sometimes be included in an ASM model calibrated for power prediction. This is the case for devices that are also included in the total monthly energy bill of the waste water treatment site. For this purposes a model of the electrical heating of the buildings was included, without which strong seasonal deviations in power prediction could be observed.

The actual local costs of electricity, iron chloride, and carbon source (sodium acetate) are used to model the total cost. To include the sludge treatment cost for the situation in Flanders, the cost per ton of dry matter for the different sludge treatment processes are summed proportionally.

Environmental impact is calculated through LCA-parameters determined in the EU FP6 Neptune project (after Doka 2007; Larsen 2009) (parameters used are shown in Table 1). A detailed discussion of environmental footprint modelling is not part of the scope of this paper and can be found in other reports, (see e.g. Doka 2007). In short the effluent loads are multiplied with the impact coefficients of Table 1 to calculate the impact of effluent discharge. Similarly the footprint of electricity consumption, amount of waste sludge and dosed chemicals are calculated. The environmental impact of the infrastructure is indirectly taken care of through the amount of treated flow (Larsen 2009). The system boundary ranges from the raw influent to the effluent, and includes secondary impact of how energy and chemicals for dosing are produced.

RESULTS AND DISCUSSION

Very good model accuracy in predicting the effluent concentrations is observed (see Figure 2). Observable differences between the model predicted values and lab measurements, are all linkable to external discharges.

Effluent concentrations (see Figure 2), sludge production (105% of reality), and carbon source dosing (93% of reality), match the reality very well. Sodium acetate is used as carbon source in the post denitrifying sand filters. The 7% deviation of carbon source is accumulated in a period of approximately 2.5–3 weeks following a toxic discharge around day 35. The low autotrophic biomass (X_{aut}) concentration in the model resulted in a low nitrate effluent concentration, and hence

Table 1 | LCA coefficients used in this study for impact modelling of the Bree WWTP (Larsen 2009; Roels 2010)

Parameter	Impact*
Nitrogen	37,23 mPET/kg N
Phosphorus	269,2 mPET/kg P
Electricity consumption	0,12324 mPET/kWh
Sludge production	0,1 mPET/kg 37% ds
Infrastructure	0,127 mPET/m treated influent
$FeCl_3$ 40% dosing	2,611 mPET/kg
Carbon source dosing (sodium acetate)	0,7781 mPET/kg NaOAc

*mPET = milli people equivalent target

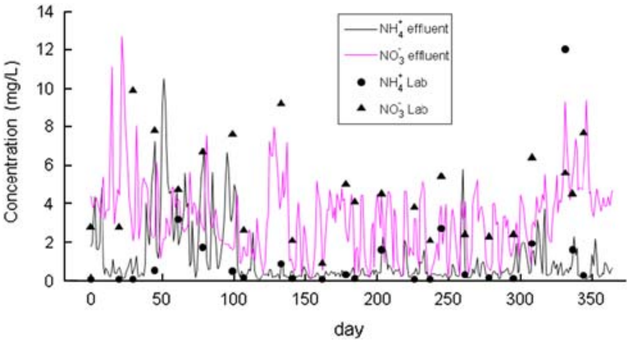

Figure 2 | Effluent concentrations: model predicted concentrations versus lab measurements of daily composite grab samples. Subscribers to the online version of *Water Science and Technology* can access the colour version of this figure from http://www.iwaponline.com/wst.

carbon dosage, which was too low. As this could be linked to external non-modellable factors, also carbon source dosage prediction is considered accurately.

Figure 3 lists the consumption of the different categories of consumers. Remarkable is the consumption of the electrical heaters with 5.4% (12–15% in winter months). Sludge line consumption is shown as well but is out of scope for the rest of the discussion. The *not modelled* fraction is the deviation from reality (98.8% of real consumption was predicted through the model).

Scenario evaluation

In the following sections six different scenarios for plant optimisation will be discussed. The same period, January to December 2007, as used for the model calibration is used here for the comparison of operational and simulated results. Both the well-known optimisation approach for lowest cost meeting the consent and the newer environmental impact approach are applied separately, but their application to a single scenario is discussed in the same paragraph. When treating a scenario, the optimal settings of the previous scenarios are used with the optimisation approach under discussion. This is shown in Figure 4 where point one is the reference situation, points 2 up to 6 represent the consecutive situations for the cost optimisation meeting the consent, whereas the points 7 up to 12 represent the LCA approach. The reported dynamic model results are thus the additional savings over the previous scenarios and the differences of both approaches are discussed.

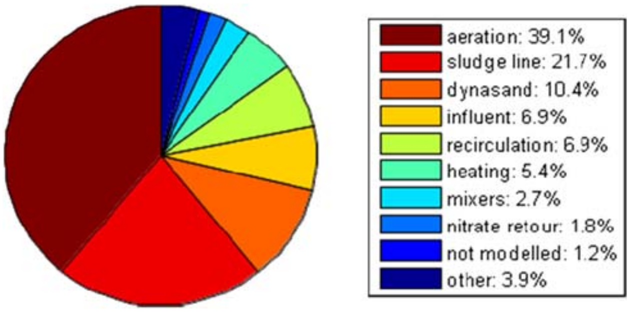

Figure 3 | Energy consumption for the different categories of consumers. Subscribers to the online version of *Water Science and Technology* can access the colour version of this figure from http://www.iwaponline.com/wst.

Online aeration control

Aeration of WWTP Bree is up to now controlled manually and checked every few days using field tests. An online rule-based control utilising ammonia and nitrate concentrations is proposed. SCADA settings include amongst others setpoints for upper limit of ammonia concentration at which aeration is started ($NH_{4,\ high}$) and a lower ammonia limit ($NH_{4,\ low}$) at which aeration is halted. An additional upper limit for nitrate concentrations as a fallback scenario is used in reality. $NH_{4,\ low}$ was checked for values between 0.5 and 2.5 mg/L, whereas $NH_{4,\ high}$ was checked for 1 to 3 mg/L.

Cost optimisation will lead to the highest allowable values for the ammonium setpoints, because no aeration is necessary for the oxidation of the ammonium load in the effluent: 1.5 mg/L for $NH_{4,\ low}$ and 3 mg/L for $NH_{4,\ high}$ were obtained as safe setpoints towards the effluent consents. This would lead to a reduction of 13% in the cost of energy consumption, dosing of chemicals, and sludge treatment (point 2 versus

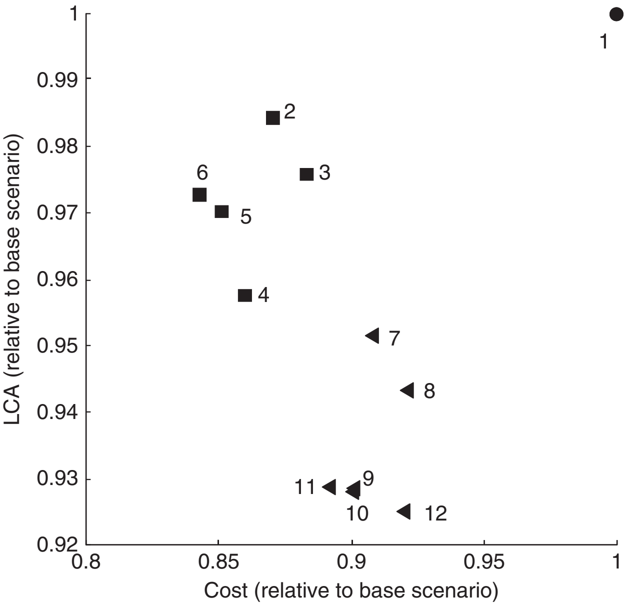

Figure 4 | Cost and LCA value per scenario: circle: base scenario, squares: cost optimisation, triangles: optimisation towards environmental footprint.

point 1 in Figure 4). So while nitrogen removal is not influenced significantly, environmental impact is reduced by a mere 6.2%, electricity consumption of the water line is reduced by 4.9% and carbon dosing by an astonishing 39%.

The *LCA approach* proposes 0.5 mg/L for $NH_{4,\ low}$ and 1.5 mg/L for $NH_{4,\ high}$. This results in an environmental impact which is much lower, namely 9.3% (point 7 versus point 1 in Figure 4). The cost is in this case lowered by 9.5%, electricity consumption is reduced by 4.9%, whereas carbon dosage is reduced by approximately 27%.

Thus, both the cost and environmental impact approach allow for a significant cost reduction. Logically the cost-optimisation approach allows for a bigger cost reduction (point 2) and the LCA approach allows for a higher impact reduction through an extra 2.5% nitrogen removal (up to 84.7%) (point 7). Cost optimisation for an automatic aeration control also allows a roughly 50% higher saving in carbon dosage (39% versus 27%). An important remark is that with the classic cost approach one needs to consider effluent consents as the ultimate limit for optimisation, whereas LCA handles effluent quality automatically. In addition, LCA in optimisation studies tends to mimic the behaviour in which operators operate the plant, at least in this case.

New online control on sludge concentration in the aeration tank

As the sludge concentration has a major impact in the speed of BOD and nitrogen removal, the impact of a simple sludge concentration controller is evaluated. When sludge concentration in the aeration tank would exceed the setpoint, sludge waste pumps are activated. Sludge concentrations of 2 to 6 g/L are studied.

Cost optimisation would be in favour of high sludge concentrations (faster removal of pollutants). However, prolonged periods of sludge concentrations above 4 g/L result in sludge washout and concentrations above 3.5 g/L pose an enhanced risk for the effluent consents on rainy days. When choosing a continuous fixed sludge concentration of 3 g/L (point 3 versus point 2 in Figure 4) would result in 1.5% higher operational costs, whereas carbon source dosing may rise with approximately 3–4%, after implementation of the online aeration control (see previous section).

Interestingly, the *LCA optimum* (0.9% impact reduction; point 8 versus point 7 in Figure 4) is 3 g/L MLSS in the aeration basin. This illustrates that LCA automatically takes effluent quality into account (no risk of sludge wasting) without intervention from the modeller to check whether the effluent consents are met.

Hence, optimisation of costs would not result in the implementation of an online sludge control aiming 3 g/L of MLSS in the aeration tank, because of the higher cost. On the other hand, as it would significantly lower the risk of sludge washout in case of peak inflows, it was decided to elaborate further in a subsequent study on a sludge control and implement it if possible. For both approaches a fixed sludge concentration of 3 g/L is considered for further optimisations.

Investigation of internal recirculation flow

An internal recirculation flow of 200 m^3 h from the aeration tank to the anoxic tank is present on WWTP Bree. To quantify the effect of recirculating nitrate rich water, no internal recirculation and double recirculation flow are evaluated. Both *cost optimisation* and *environmental impact assessment*, respectively situation 4 and 9 in Figure 4, favour double nitrate recirculation flow over the current situation in the plant. Cost would drop by approximately 2.6%, environmental impact by 1.6%, and carbon dosage by approximately 14 to 18%. Energy consumption rises approximately by 1.5% and nitrogen removal by 0.5–0.8%.

Alteration of the oxygen setpoint

Yearly fixed oxygen setpoints of 1, 1.5, 2, 2.5, and 3 mg/L for the aeration tank are studied.

Cost optimisation results in an oxygen setpoint of 1 mg/L for the aerated periods. This would lead to a reduction of 1% in the cost of energy, chemicals, and sludge treatment (point 5 versus 4 in Figure 4). Energy consumption would drop by 2.2%, and nitrogen removal would drop as well (to 81.9% if cost optimisation is used for section 3.3.1 to 3.3.4).

The *optimal LCA* value is obtained for the 2.5 mg/L fixed oxygen setpoint, which is the average value in reality in the simulated period (point 10 in Figure 4).

Remarkably LCA optimises the oxygen setpoint towards the very same value that the operators of the plant experienced as the best oxygen setpoint to operate the plant. On the contrary, cost optimisation favours very low oxygen concentrations (1 mg/L) in the aeration tank. Unfortunately these low oxygen concentrations may favour filamentous bacteria and degrade sludge settleability. (Martins *et al.* 2003).

Installation of mixers in the aeration tank

The aeration basin of WWTP Bree is equipped with three 30 kW point aerators. Hence, in aerated periods enough mixing is present through the aeration system. To prevent sludge settling in the aeration tank, a maximum of 5 minutes unaerated time is allowed after which shortly an aerator is switched on. This is the current strategy at WWTP Bree. In this section the installation of three 5 kW mixers is investigated to quantify eventual higher nitrogen removal.

The *cost optimisation approach* does not show an additional reduction in energy consumption, but costs are roughly 1% lower in comparison with the previous scenario (point 6 versus 5 in Figure 4). Since the installation of mixers results in better denitrification in the aeration tank, the observed cost reduction is due to an additional 6% reduction of sodium acetate dosing in the post denitrifying sand filters. However, the payback time of 10 years for new mixers is considered too long.

The *LCA approach* (point 11 in Figure 4 is the situation with mixers) does not show an improvement in environmental impact over the previous situation (point 10).

Hence, both the cost and environmental impact approach produce the same outcome for this scenario: they agree on the fact that installation on mixers is not highly beneficial.

Nitrate target concentration of denitrifying sand filters

Sodium acetate is proportionally dosed on the post denitrifying sand filters to reach a certain nitrate concentration in the effluent within a certain range. The current nitrate effluent concentration setpoint is 1.5 mg NO_3^-.

Raising the setpoint to 2 mg/L (*cost approach*) would reduce the cost with 2.2%. However, there is no safety anymore in case of incidents, such as defects or external discharges. Hence, the effluent nitrate setpoint cannot be raised because of the effluent consents.

The *LCA approach* could lower the environmental impact by an insignificant 0.4% when reducing the setpoint from 1.5 to 1 mg NO_3^-/L (point 12 versus 10 in Figure 4), the cost penalty would be five times as high. This cost penalty is considered inappropriate for insignificant reductions in environmental footprint.

Hence, both approaches produce the same outcome.

General comparison

Table 2 lists the overall savings that can be obtained with LCA (point 5 in Figure 4) versus the classical cost optimisation (point 10).

When optimising towards *environmental footprint*, the footprint reduction is 2.4 times higher compared to cost optimisation. The reason is a 2.5 per cent increase of the nitrogen removal versus a decrease of 0.8%. Although less euros can be saved with the LCA approach, the savings quantified at 9.9% are still important.

Table 2 | Overall improvements that can be obtained

Impact parameter	Optimisation towards *minimal costs*	Optimisation towards *minimal environmental footprint* (*LCA value*)
Environmental impact	−3.0%	−7.2%
Cost	−15%	−9.9%
Power consumption	−5.6%	−2.3%
Nitrogen removal	−0.8%	2.5%
Carbon dosage	−47%	−35%
Sludge production	1.6%	2.2%

The 15% savings of *cost minimisation* where the effluent consent is met are primarily due to the reduction in carbon dosage, which is almost reduced by a factor of 2. In addition power consumption is 5.6% lower (2.4 times higher when compared to the value for footprint optimisation).

Interestingly, implementation of the proposed aeration control only requires the installation of an ammonium sensor. The conservative payback period of the investment would be 7 months in case of the LCA approach and less for the cost meeting the consent approach.

When optimising towards environmental impact using LCA, we noticed that LCA as a criterion inherently leads to safer operating conditions as compared to classical cost minimisation. The reason is that one always needs to take care of the consents when trying to lower the costs. As opposed to cost optimisation, LCA showed no risks towards excessive nutrient discharges or sludge washout, and no checking of consents by the modellers was necessary (consents were always met within a safe margin). An additional advantage is that LCA tends to optimise towards settings that plant operators would use automatically. Hence, better effluent quality is reached at a slightly higher operational cost (5% at most). The question can be raised whether the cost penalty is affordable for the receiving water and community.

CONCLUSIONS

An accurate ASM2da model of the full scale WWTP of Bree was built. Significant reductions in costs, power consumption, chemicals dosing, and environmental impact can be obtained through the installation of new sensors and optimal settings for the online control system. It is shown that LCA inherently optimises towards safe operating conditions, whereas the cost minimisation approach allows a higher cost reduction with lower nitrate removal. The settings obtained for the LCA approach are also those that operators tend to use themselves. There is a minor cost penalty for these conditions in comparison to widely used minimisation of costs while meeting the effluent consent.

ACKNOWLEDGEMENTS

This study was part of the EU Neptune project (Contract No 036845, SUSTDEV-2005-3.II.3.2), which was financially supported by grants obtained from the EU Commission within the Energy, Global Change and Ecosystems Program of the Sixth Framework (FP6-2005-Global-4).

REFERENCES

Standard Methods for the Examination of Water and Wastewater 1999. 20th edition. American Public Health Association (APHA).

Benedetti, L., Bixio, D. & Vanrolleghem, P. A. 2006 Benchmarking of WWTP design by assessing costs, effluent quality and process variability. *Water science & technology* **54**(10), 95–102.

Daigger, G. T. & Roper, R. E. 1985 The relation between SVI and activated sludge settling characteristics. *Journal WPCF.* **57**(8), 859–866.

Devisscher, M., Ciacci, G., Fe, L., Benedetti., L., Bixio, D., Thoeye, C., De Gueldre, G., Marsili-Libelli, S. & Vanrolleghem, P. A. 2006 Estimating costs and benefits of advanced control for wastewater treatment plants - the MAgIC methodology. *Water science & technology* **53**(4–5), 215–223.

Doka, G. 2007 Ecoinvent report nr. 13: Life cycle inventories of waste water treatment services. Ecoinvent Centre, Dübendorf.

Ebner, M., Stalph, P., Michel, M. & Benz, R. 2010 Evolutionary parameter optimization of a fuzzy controller which is used to control a sewage treatment plant. *Water science & Technology* **61**(1), 53–65.

Fall, C., Loaiza-Navia, J. & Esparza-Soto, M. 2009 Full activated sludge model no. 1 calibration experience at a medium-size WWTP in Mexico. *Water science & technology* **60**(12), 3069–3081.

Foley, J., de Haas, D., Hartley, K. & Lant, P. 2010 Comprehensive life cycle inventories of alternative wastewater treatment systems. *Water Research* **44**(5), 1654–1666.

Gernaey, K. V. & Jorgensen, S. B. 2004a Benchmarking combined biological phosphorus & nitrogen removal wastewater treatment processes. *Control Engineering Practice* **12**(3), 357–373.

Gernaey, K. V., van Loosdrecht, M. C. M., Henze, M., Lind, M. & Jorgensen, S. B. 2004b Activated sludge wastewater treatment plant modelling & simulation: state of the art. *Environmental modelling & software* **19**(9), 763–783.

Activated Sludge Models ASM1, ASM2, ASM2d, ASM3. Scientific & technical report No 9 2000 IWA Task group on mathematical modelling for design & operation of biologic wastewater treatment.

Ingildsen, P., Rosen, C., Gernaey, K. V., Nielsen, M. K., Guildal, T. & Jacobsen, B. N. 2006 Modelling & control strategy testing of biological & chemical phosphorus removal at Avedøre WWTP. *Water Science & Technology* **53**(4–5), 105–113.

Insel, H. G., Gorgun, E., Artan, N. & Orhon, D. 2009 Model Based Optimization of Nitrogen Removal in a Full Scale Activated Sludge Plant. *Environmental Engeneering Science* **26**(3), 471–480.

IPCC 2007 Climate Change 2007 - The Physical Science Basis (fourth assessment report) 2007 Intergovernmental Panel on Climate Change.

Larsen, H. F. 2009 Personal communication.

Martins, A. M. P., Heijnen, J. J. & van Loosdrecht, M. C. M. Effect of dissolved oxygen concentration on sludge settleability. 2003 *Applied Microbiology & Biotechnology* **62**(5–6), 586–593.

Roels, J., Wambecq, T., De Gussem, K. & Fenu, A. 2010 LCA & nutrient removal (Neptune FP6 end user conference presentation). http://www.eu-neptune.org/End%20User%20Conference/05_Roels.pdf.

Sin, G., De Pauw, D. J. W., Weijers, S. & Vanrolleghem, P. A. 2008 An efficient approach to automate the manual trial & error calibration of activated sludge models. *Biotechnology & Bioengineering* **100**(3), 516–528.

Methods for Wastewater Characterization, Inventory, & Guidelines (in Dutch). 1996 STOWA-report 97–23. Hageman Fulfilment, Zwijndrect, The Netherlands.

Takacs, I., Patry, G. G. & Nolasco, D. 1991 A dynamic model of the clarification-thickening process. *Water Research* **25**(10), 1263–1271.

Kyoto Protocol To The United Nations Framework Convention On Climate Change 1998 United Nations http://unfccc.int/resource/docs/convkp/kpeng.pdf.

Vanrolleghem, P. A., Insel, G., Petersen, B., Sin, G., De Pauw, D., Nopens, I., Weijers, S. & Gernaey, K. 2003 A comprehensive model calibration procedure for activated sludge models. In proceedings: 76th Anual Technical Exhibition & Conference. WEFTEC 2003. October 11–15, 2003. Los Angeles, California, US.