Benchmarking of WWTP design by assessing costs, effluent quality and process variability

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Abstract Process choice and dimensioning of WWTPs is a particularly sensitive step to cost-efficiently comply with regulatory standards. This step accounts only for a small fraction of the upfront costs, but it can lead to substantial savings. This paper illustrates the results of a systematic methodology to evaluate system design/upgrade options. In contrast to conventional practice, this approach allows the choice between the most appropriate trade-off between cost of measures and effluent quality, and to assess the reliability of a process layout. It is therefore a flexible instrument to cope with the flexibility and complexity of integrated water management regulations. Results show good agreement of the simulations with extensive benchmarking studies on actual plants. For that reason, the suggested methodology can provide valuable support also to such practices.

Keywords Benchmarking; cost-effectiveness; mathematical modelling; Monte Carlo simulation; probabilistic design

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Introduction

With the new water quality based approach introduced with the European Water Framework Directive (WFD), the design of the systems is by far less predetermined and the options to meet the goals become much more numerous. Therefore new design methodologies must be developed in order to be able to cope with such increased complexity in a cost-efficient way. The EU project CD4WC (www.cd4wc.org) is currently tackling these problems, focussing on the urban wastewater system.

This paper illustrates some results of an innovative methodology to identify and quantify the costs and benefits of an urban wastewater system with regard to its environmental and economic consequences, as required by the Water Framework Directive. Criteria to assess the environmental consequences are – besides the water quality – also secondary resource inputs such as energy, materials and chemicals (Benedetti *et al.*, 2006a, b).

The developed methodology has been introduced in Benedetti *et al.* (2005) showing its merits for the dimensioning of a selected process. This paper focuses on its suitability in comparing process options.

Methods

The methodology consists of four steps: (1) data collection and reconstruction, (2) model building and calibration, (3) evaluation of alternatives and (4) uncertainty assessment.

(1) Data collection and reconstruction

Data collection is probably the most challenging task. In order to evaluate upgrade alternatives, long-term (several representative years) dynamic influent data are advisable. In the absence of these data, influent data can be reconstructed using available measurements and making assumptions on the influent properties (Bixio *et al.*, 2002a; Devisscher *et al.*, 2006). In the proposed procedure, a dynamic influent is generated using a simple

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model of the draining catchment, which considers the number of inhabitants, the presence of industry, the loads per capita of households and industry, the size of the catchment, the length of the sewer system, rainfall data, and the interactions with groundwater (Gernaey *et al.*, 2005).

(2) Model building and calibration

The most used models for biological treatment are the IWA activated sludge models (ASM) (Henze *et al.*, 2000).

In the case of the design of a new plant (i.e. no infrastructure is in place yet), standard parameter values should be applied for the used models, while the modeller decides on the operational variables, and their optimal values will be identified by performing several simulations with different values, or applying an optimisation algorithm. In this case there is no calibration of the model to be performed, but only some adjustments to parameter values; e.g. if for any reason (temperature, influent characteristics, etc.) the expected Sludge Volume Index is higher than average, some parameters in the secondary settling model should be modified.

(3) Evaluation of scenarios

The comparison of alternative scenarios can be based on performance criteria that can be grouped into two categories: environmental and economic criteria. The weight attributed to them in the decision making process depends on the specific situation of the project.

Environmental performance. The proposed methodology adopts the approach set out by IWA/COST (Copp *et al.*, 2002). It consists of the evaluation of three indicators: the effluent quality index, the time and number of effluent violations, and operating costs.

The effluent quality index (EQI) is meant to quantify the effluent pollution load to a receiving water body in a single variable. The EQI is the weighted sum of the pollution loads due to (1) total suspended solids (TSS), (2) chemical oxygen demand (COD), (3) biological oxygen demand after 5 days (BOD5), (4) total Kjeldahl nitrogen (TKN), (5) nitrates and (6) total phosphorus (TP) over one complete year. The used weights are based on Vanrolleghem *et al.* (1996) that cited a Flanders' effluent quality formula for calculating fees.

Economic performance. The evaluation of costs for wastewater treatment is very complex. Costs can differ among countries or regions because of different specific conditions and also because of differences in planning and building procedures (Bode and Lemmel, 2001). This complexity makes the approach to calculate costs in order to compare different plant configurations and operational strategies very difficult. Detailed cost calculations should in general be preferred over the use of cost functions, which can only be useful for rough estimations. Most WWTPs are tailored to specific conditions/needs, i.e. plants with the same treatment performances do not inevitably incur the same costs. The use of cost functions is feasible only for process options screening (Gillot *et al.*, 1999), i.e. as is the case here. In this project, operating costs have been estimated with the benchmark assessment procedure (Copp *et al.*, 2002) and with prices representative for Northern Europe, while the capital costs were provided by Ruhrverband (Germany).

The cost categories used in this paper are: aeration energy cost (AEC); energy cost (EC) including aeration, pumping and mixing costs; sludge cost (SC) which comprises sludge treatment and disposal; variable cost (VC) incorporating energy, sludge and chemicals cost; total cost (TC) which includes variable, personnel, maintenance and annualised capital costs.

Operational costs are considered through five terms: sludge production, pumping energy, aeration energy, mixing energy and chemicals use.

(4) Uncertainty assessment

An issue when dealing with deterministic models is the degree of uncertainty linked to their predictions. Probabilistic design, which is the combination of probabilistic modelling techniques with the currently available deterministic models, provides a solution to this issue (Bixio *et al.*, 2002b). By building a probabilistic shell around the deterministic models one can quantify the uncertainty of the model predictions. For example, a goal can be to determine the probability of exceeding the legal effluent standards of a WWTP. This percentage of exceedance should be accompanied by confidence intervals indicating the uncertainty due to the variability of influent characteristics and due to the uncertainty in model parameters.

The quantification of the uncertainty of the system as a whole may be carried out by running Monte Carlo simulations (Rousseau *et al.*, 2001), which generate a probability density function or cumulative density function of the output. Based on the distribution of the output, a risk level representing the high end (e.g. 95th percentile), central tendency (median or mean), or any other desired level of probability can be identified. See Figure 1 for two examples of uncertainty visualisation; on the left side of Figure 1 for all of the ten configurations a cloud of 100 dots is plotted, each dot representing the yearly average of the EQI and of the TC for one particular Monte Carlo simulation; on the right side, each cloud is summarised by a polygon joining the 5th and 95th percentiles for the two variables and by a marker for the 50th percentile.

A measure used in this study to summarize the model output uncertainties is the relative reliability index (RRI) which is inversely proportional to the sum of the standard deviations of the two considered variables (e.g. EQI and TC in Figure 1) and it gives a measure of how stable the performance is when subjected to variations in model parameters.

The number of necessary simulations for scenario analysis tends to be large, especially with Monte Carlo-based uncertainty assessment. To reduce this computational burden, tools that distribute simulations over idling PCs available in a local network are under development and were used in this study (Claeys *et al.*, 2006).

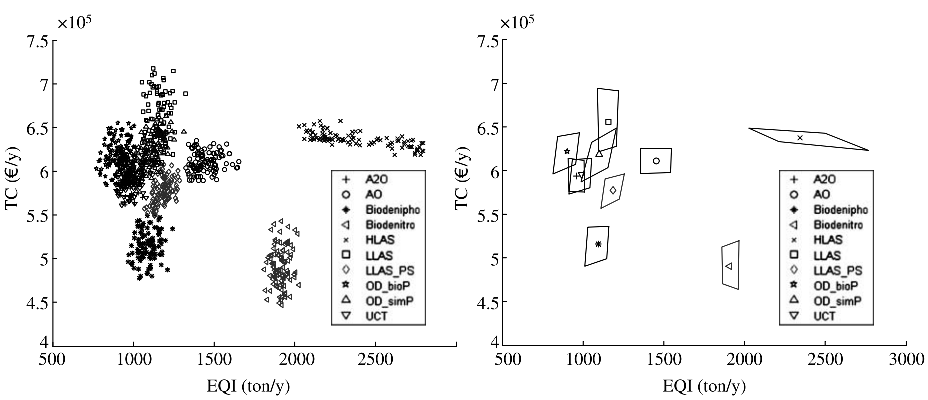

Figure 1 Two options to visualise Monte Carlo simulation results: all results as a cloud of markers (left) and polygons joining the 5th and 95th percentiles for the two variables and the 50th percentile as a marker (right); the data show yearly average effluent EQI and TC for 10 different configurations; the explanation of the legend can be found in the Results section

Results and discussion

The benchmarking methodology was applied to WWTP design according to ten conventional process layouts with three different sizes (3,000, 30,000 and 300,000PE) and in four different climatic conditions (Alpine, Continental, Mediterranean and Oceanic). Each scenario was characterized by an actual rainfall series for a representative year with 10 minutes interval, and a daily temperature series. The ten process configurations are:

- anaerobic-anoxic-oxic (A2O);
- anaerobic-oxic (AO);
- Biodenipho;
- Biodenitro;
- high loaded activated sludge (HLAS);
- low loaded activated sludge with bio-P removal (LLAS);
- LLAS with primary settler (PS);
- oxidation ditch with bio-P removal (OD_bioP);
- OD with chemical P removal (OD_simP);
- University of Cape Town process (UCT).

Such processes have been dimensioned according to the guidelines contained in ATV (2000), in order to represent existing treatment plants, and were modelled by using ASM2d (Henze *et al.*, 2000). In this work the number of parameters considered as uncertain was 13 (11 belonging to ASM2d and 2 to the influent fractionator model), and for each configuration 100 Monte Carlo simulations were run; this number was sufficient to reach convergence of the output frequency distributions. The natural variability of the influent is assumed to be represented in the one year time series with 15 minute intervals.

The process comparison is illustrated in this paper only for one combination of climate and plant size. The chosen catchment characteristics (generating the wastewater to be treated) refer to a combined sewer system collecting 30,000PE (with 15% of the COD load to be industrial), located in an area with Oceanic climate conditions, with infiltration as 30% of dry weather flow, having a weekend load increase of 12% and a summer holiday loads decrease of 25%. The daily average load per population equivalent (PE) is taken from ATV (2000). The pollutant load is increased with nutrients to consider the return to the water line of reject water from the sludge treatment line. The influent time series was generated with the model of Gernaey *et al.* (2005).

The performance of the ten simulated processes can be summarised by the EQI and by the TC (Figure 1). Three regions are easily identifiable:

- the HLAS process, not removing nutrients, on the right hand side of the graph;
- the Biodenitro process, not removing phosphorus, in the lower middle zone;
- all other systems, on the left hand side.

The first conclusion is that the HLAS has full costs comparable to nutrient removing systems, but poorer environmental performance. There is therefore no use in implementing such a system. The second conclusion is that the Biodenitro system shows the lowest TC but relatively high EQI, i.e. lower effluent quality. So it is a question of receiving water quality requirements whether the savings can be better invested in more cost-effective measures, as possible in the flexible context of the WFD. With conventional emission limits approach, like the one introduced by the EU Urban Wastewater Directive, this configuration would have been discarded from further consideration. Third, it can be noticed that the P-removal plant (AO) has TC similar to the N- and P-removal plants and slightly worse environmental performance.

Table 1 gives the RRI for the ten configurations, from which it appears that the least reliable configuration for EQI and TC is the HLAS system. The main reason is its wide variability of effluent TP values (Figure 3). The most reliable systems for EQI-TC are

Table 1 Relative reliability index (RRI) for the ten processes; TC = total cost, VC = variable cost, EC = energy cost, SC = sludge cost, AEC = aeration energy cost; the RRI is relative to the two variables in consideration (e.g. the first row considers EQI and TC)

Configuration	A2O	AO	Biodenipho	Biodenitro	HLAS	LLAS	LLAS_PS	OD_bioP	OD_simP	UCT
EQI − TC	1.37	1.00	1.05	1.06	0.39	0.86	1.23	1.02	0.86	1.17
COD - VC	1.02	1.14	0.99	0.92	1.22	0.59	1.14	0.99	0.98	1.01
TN − EC	1.27	0.58	0.85	0.67	2.64	0.77	1.01	0.76	0.51	0.94
TP − SC	0.91	1.32	0.82	0.67	0.22	0.88	2.20	0.82	1.30	0.86
NH_4 - AEC	0.98	0.11	0.93	0.89	1.25	0.94	2.44	0.80	0.66	1.01

A2O, LLAS_PS and UCT which perform consistent nutrient removal. On the other hand, HLAS is very stable for the TN-EC combination as it does not denitrify; OD_simP and AO are less stable in this regard. From Table 1 one can also notice that the addition of the primary settler to the LLAS enhances the stability of COD removal.

In Figures 2 and 3 the performance of the plants regarding NH_4, AEC, TN, EC, COD, VC, TP and SC are illustrated.

Concerning the environmental performance, all configurations achieve excellent levels due to their generous dimensioning. Very low NH_4 effluents are achieved for the two LLAS layouts, while TP is kept very low by bio-P plants. The TP effluent of chemical P precipitation plants is deliberately closer to the limit of 2 mg/L but shows very small variations due to the fact that the dosage of P-precipitant is controlled by measuring PO_4 in the effluent. For all configurations large variability appears in effluent TN, and the performance in TN removal is inversely proportional to the performance in NH_4 removal, due to the difference in effluent nitrate. Looking at the economic performance of nutrient removal plants, the lowest TC is associated to Biodenipho, which has both low VC and capital costs. The lowest VC is achieved for N- and P-removing plants by the LLAS_PS configuration. These two configurations have both low EC and SC. The high aeration cost of OD_simP is due to the higher DO set-point than for the other configurations.

Figure 4 shows the removal efficiencies of the 10 plants, assessed by the ratios of VC on COD removed and of TC on pollution removed, which is expressed by a quality index (QI) calculated as the weighted sum of pollutant removed, using the same weights as for the EQI.

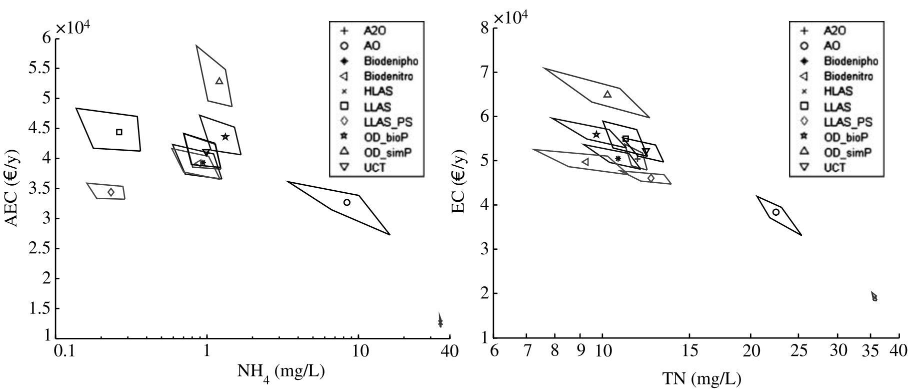

Figure 2 Comparison of alternatives based on NH_4 and AEC (left) and on TN and EC (right); logarithmic scale for NH_4 and TN

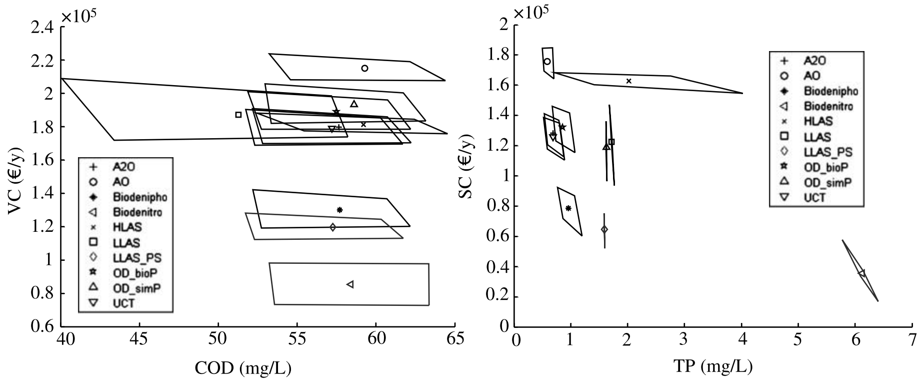

Figure 3 Comparison of alternatives based on COD and VC (left) and on TP and SC (right)

Alternating systems (Biodenipho and Biodenitro) show better unitary cost performance (TC per QI removed). Overall, for nutrient removal the Biodenipho plant might be considered as a configuration with low costs, good effluent quality and the best treatment efficiency. The LLAS_PS plant requires high capital costs but performs very well in terms of VC, effluent quality and RRI. Of course it must be kept well in mind that such results are a function of the input characteristics and the applied cost data. In particular, the comparative analysis with regard to the TC is very sensitive to the sludge disposal policy that is taken into consideration. In this study, we have considered a sludge disposal cost typical of western European countries, where the excess sludge may not be spread on land and where the disposal to landfill is highly taxed. In those places where the sludge can be disposed of in agriculture, sludge disposal costs are up to one order of magnitude lower and therefore the high loaded systems (HLAS and AO), would be bettered placed in the analysis (Figure 3).

In Figure 5, TC and VC (values averaged on the 10 configurations) are shown relative to the plant PE, for all climates and plant sizes. It appears that TC is highly influenced by the plant size, while VC is only to a lesser extent. The figures are well in the range of benchmarking studies performed on actual WWTPs (Balmér, 2000; Bode and Grünebaum, 2000; Stemplewski *et al.*, 2001; Lindtner *et al.*, 2004; Benedetti *et al.*, 2006b), giving confidence in using the proposed methodology to benchmark wastewater systems without the need to perform extensive and detailed data collection on existing systems.

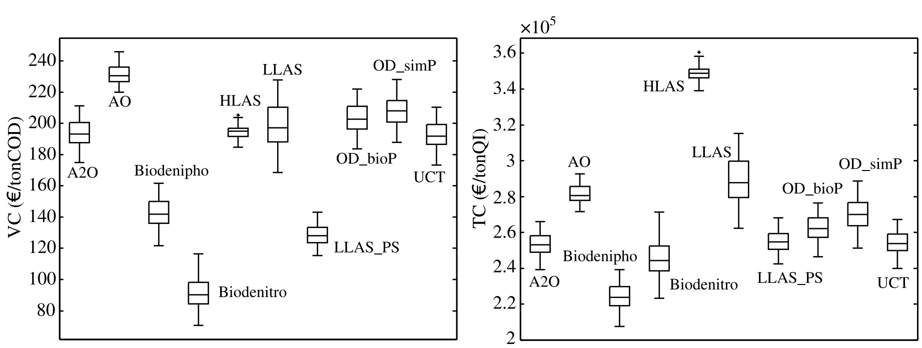

Figure 4 Comparison of alternatives based on VC per ton of COD removed (left) and on TC per ton of QI removed (right)

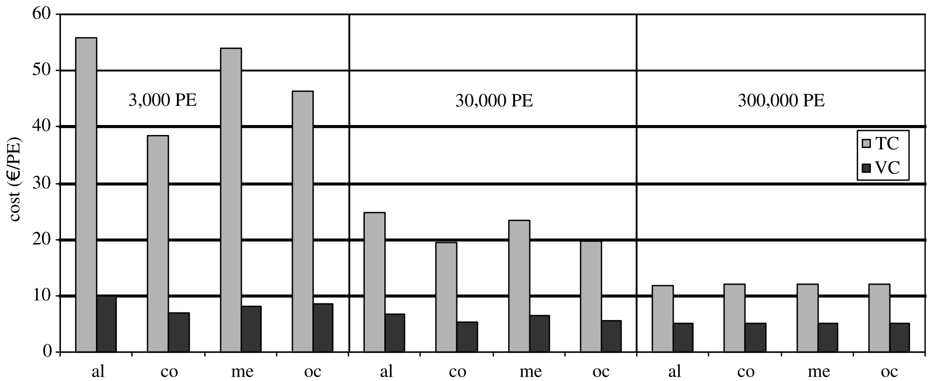

Figure 5 TC and VC per PE; al = Alpine, co = Continental, me = Mediterranean, oc = Oceanic

Conclusions

The proposed benchmarking methodology shows promising results for the systematic comparison of urban wastewater treatment measures. Under the considered boundary conditions, alternating systems show the best cost-benefit performance (TC per QI removed) while high loaded systems show the lowest. The comparison results are dependent on the boundary conditions and on the cost data used for the study, but the methodology is general.

The availability of well-accepted models, risk assessment techniques and sufficient computational power should move the design practice from conventional procedures suited for a relatively stiff context as imposed by emission limits, to more advanced, transparent and cost-effective procedures appropriate to cope with the flexibility and complexity introduced by integrated water management approaches like the WFD.

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