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INTEGRATION OF WASTEWATER TREATMENT PLANT DESIGN AND OPERATION – A SYSTEMATIC APPROACH USING COST FUNCTIONS

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ABSTRACT

A general framework for the formulation and analysis of an overall decision support index is discussed. It is indicated that such an index allow evaluation of the combined effects of both design and operation (i) during the planning phase of new WWTPs, as well as (ii) for the evaluation of new operational strategies versus traditional expansions of plants already in operation. Attention is drawn to the problems of incorporating such factors as plant flexibility and robustness against failures in the index. These factors become especially relevant when decisions are to be made that affect the life span of a WWTP. Spatial and temporal separation of the optimisation problem are proposed to make the approach operational. It is not the intention of the authors to provide an exhaustive list of objective criterion functions, but rather to give the reader some examples illustrating the approach. Copyright © 1996 IAWQ. Published by Elsevier Science Ltd.

KEYWORDS

Cost factors; economic optimisation; integration; mathematical modelling; sustainability; wastewater treatment.

INTRODUCTION

In the coming years it is believed that the focal point in wastewater treatment will be the recipient. The wastewater treatment of the future is expected to be dependent on the requirements from the local recipient and not some common effluent standards (Tyson *et al.*, 1993). Fact is that current standards have been controlled to a large extent by present technologies ("BATNEEC: Best Available Technology Not Entailing Excessive Costs"). In some countries, effluent requirements have been explicitly connected to a particular treatment technology. Nowadays wastewater treatment is able to cope with almost any effluent quality objective. The problem is that the cost of such treatment becomes prohibitively large since costs increase rapidly with the effluent requirements. If the cost of treatment and the impact of discharging to the recipient could be compared on a common scale, a meaningful cost/benefit analysis could be performed for different treatment alternatives.

Objective methods for evaluating the overall design and operation of wastewater treatment plants (WWTP) are therefore of importance, for both economical and environmental reasons. However, the authors have experienced, among others in the framework of some successful workshops under the European COST-682 project (Dochain *et al.*, 1995), that in many instances, difficulties arise in defining a specific objective of WWT in a quantitative manner and, moreover, quantifing all efforts required for its accomplishment (Carstensen *et al.*, 1994). The pursued methodology must allow us not only to formalise and include all traditional criteria related to design, maintenance and operation, but it should also allow us to include more elaborate aspects of WWT such as plant flexibility and robustness against failure. The latter aspects will start to play an increasing role as the time horizon over which the evaluation is made increases towards the life span of a treatment works. It is also considered an omission from many previous optimisation studies (Tyteca *et al.*, 1977) that little attention is paid to the potential cost reductions of real-time control as exposed by dynamic simulation. Special focus is to be put on this aspect of cost optimisation.

NEED AND OPPORTUNITIES FOR A SYSTEMATIC APPROACH

There are a number of reasons indicating that it is now needed but also possible to tackle the problem of creating a relevant cost function from a systematic point of view.

1) Information technology and computing power have reached such levels that large amounts of data can now be efficiently analysed using sophisticated interpretation methods. Moreover, new sensors and measurement techniques are available that produce data, that are both reliable and informative, and new means of digital communications make it possible to distribute the data in a simple way.

2) Large research programmes initiated during the last decades have resulted in a substantial increase in understanding of the different unit processes applied in WWTPs. However, knowledge is still packaged around unit processes and does not focus enough on the importance of process interactions. Still, gradually more attention is drawn to these aspects as evidenced, for instance, by the work of Sheintuch (1987) and Nyberg *et al.* (1994).

3) Increasingly complex combinations of these unit processes have evolved to meet ever more demanding standards. Currently, one also observes a trend towards more attention for the interactions between (i) the WWTP, (ii) the sewer system or the production scheduling, and (iii) the receiving waters (Vanrolleghem *et al.*, 1996). This further increases the complexity of the system under study, the interactions to consider and the degrees of freedom to evaluate in an optimisation exercise.

4) Sustainability of the solutions for today's problems is a central theme of putting forward requirements and has therefore generated a lot of research interest. Although the definition of sustainability has hardly settled, the holistic view on which one has to rely for sustainability assessment requires an even broader view than the traditional economic cost approach. Moreover, looking for answers to sustainability questions makes the time horizon over which decisions have to be evaluated increase from the life span of a WWTP to the time span covering a number of generations of the human race.

AN INTEGRATED PERFORMANCE INDEX

Based on the above, it is concluded that an integrated (holistic, overall) performance index is a valuable tool for the design and operation of treatment plants. In this index a trade-off should be made between the pursued quality of the process outputs (liquid: effluent, solids: waste sludge, gas: off-gas) and the associated efforts (investments, operation) required to achieve this with the inputs (wastewater) as starting point. In order to make this trade-off, however, a common framework is needed to quantitatively compare the different objectives. Typically, an economic index could be applied, but it might prove useful to apply a sustainability index instead. In the former approach all factors to consider are translated into a corresponding cost or value, while in the latter it may, for instance, be considered to translate all factors in the corresponding land area needed to produce (among others) the materials, energy and manpower for treatment in a sustainable way (Moser, 1995).

A very important issue when evaluating these types of holistic indices is to exactly define what the system of interest consists of, i.e. it is essential to specify the system boundaries and the exchange of material and energy across the boundaries before any analysis is initiated (Figure 1). Especially when sustainability issues are considered, the problem of a reasonable system boundary definition is difficult to solve as a global view should be considered in the analysis. In particular it is important to define the boundaries towards agriculture and receiving waters. While it is important to have a holistic view, at the same time it is important not to make the task overwhelming of defining the system performance.

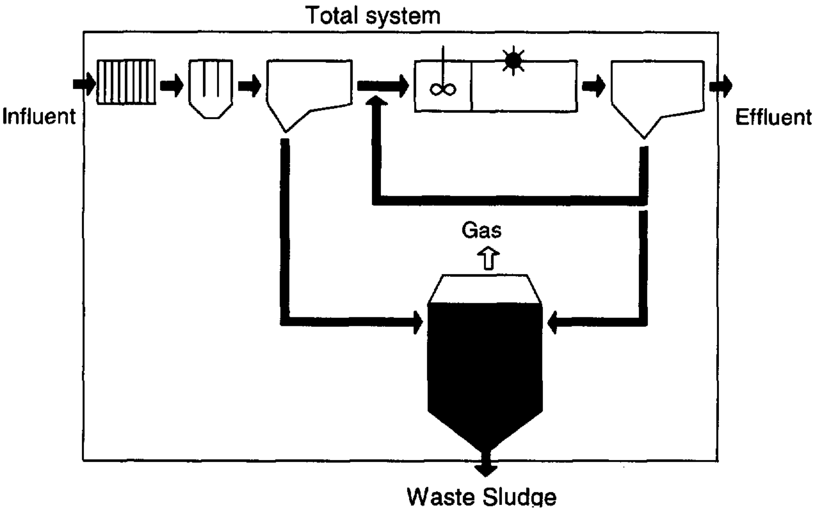

Figure 1. System boundaries for a typical WWTP.

An objective performance index (*J*) for a WWTP is proposed to be described within the following general framework (in analogy to a mass balance):

J + Input - Output + Conversion

Where the *Input* term is the (negative) quality/value associated with the incoming wastewater, the *Output* term is the quality/value of the effluent wastewater, solids, gasses and energy produced at the WWTP, and the *Conversion* term takes the applied efforts (investments and operational costs) into account.

Example

To clarify the concept the question is addressed of how to evaluate the possibility to earn money by providing wastewater treatment.

First the *input* is considered. Here, the negative value of the wastewater must be quantified. In an industrial context and with third-party treatment of for instance tank cleaning wastewaters, this value could be considered to be the amount of money a company would pay for the treatment of such specific wastewaters.

Similarly in public sewage works, one could consider the taxes inhabitants pay for their wastewater discharge, typically a few tens of ECU's per person per year.

Second, the *output* is translated into the proper units. Where levies are applied to the effluent, and where the sewage disposal costs are known and the value of the electricity produced from biogas can be quantified, the output value can be calculated. With current practice, this is mostly a negative value (it is a cost factor), but the increasing recycle of treatment outputs such as purified water, sludge derived chemicals and nutrients will eventually make these outputs valuable (Liessens *et al.*, 1995). One may also look into the heat content of effluents. Heat pumps may be quite a profitable investment, if the heated water can be added to a district heating system. This is implemented in many municipal plants in Sweden. In a sustainability context one may also "pay" special attention to the fact that mineral phosphate may run out in 200 years (Finnson and Peters, 1996) and the role that phosphate lost via the effluent to the environment (in a very dilute form!) plays in this.

Third, one must quantify the *conversion* efforts one has made to reach a certain output value. Here, the amortisation of all investments (reactors, piping, instrumentation, etc.), maintenance, labour, consumed energy and chemicals etc. is to be included. Evidently, all these elements lead to conversion costs but care should be taken because some of them, for instance the use of certain waste chemicals during the treatment process (e.g. a waste carbon source), could give a positive return as a third party might pay the treatment provider for using these waste chemicals in his process.

Assuming that wastewater with a value of -750,000 ECU is taken in yearly, material is produced with a value of -325,000 ECU and efforts are invested for a value of 600,000 ECU, the cost J is given by:

$$\mathbf{J = -750{,}000 - (-325{,}000) + 600{,}000}$$

Hence, no money was earned with this treatment process as a net cost of 175,000 ECU had to be incurred per year. It must be stressed that in this example no exhaustive search was made to include all elements in the cost index as this would make the example prohibitively complicated. Overall, this is a major problem of holistic approaches that is addressed in subsequent sections when the methodology is made operational by appropriate simplifications.

OPERATIONAL METHODOLOGY

For the different aspects mentioned above, models exist that translate measurable variables into a common denominator, either an economical index or a sustainability index. Software tools have been developed such as STOM (Spearing, 1987) or CAPDET (Wright, 1988) that deploy such models for decision support in treatment plant design. However, these tools do not consider the dynamic behaviour of treatment plants nor the advantages flexible operation (e.g. using real-time control) might have on actual plant capacity and operating costs (Coen *et al.*, 1995). In some initial cases, even the importance of including operating (so-called variable) costs themselves in the analysis has been questioned (Smeers and Tyteca, 1984).

To put the different cost factors into perspective, some illustrative data are given below. In addition, the type of function that is used for a number of terms in the cost functional J is illustrated with some examples.

Examples of cost factor distributions

In the literature a number of exercises have been presented in which the total costs of a wastewater treatment plant are quantified and allocated to different cost factors. For instance, Vanderhaegen *et al.* (1994) made an inquiry into number of specific industrial treatment plants and concluded that, next to the investment, sludge treatment forms a major part of the costs associated with activated sludge systems (Figure 2). It is striking that the operating costs form the majority of the overall treatment costs of these industrial plants. The opposite seems to hold for the municipal plants considered.

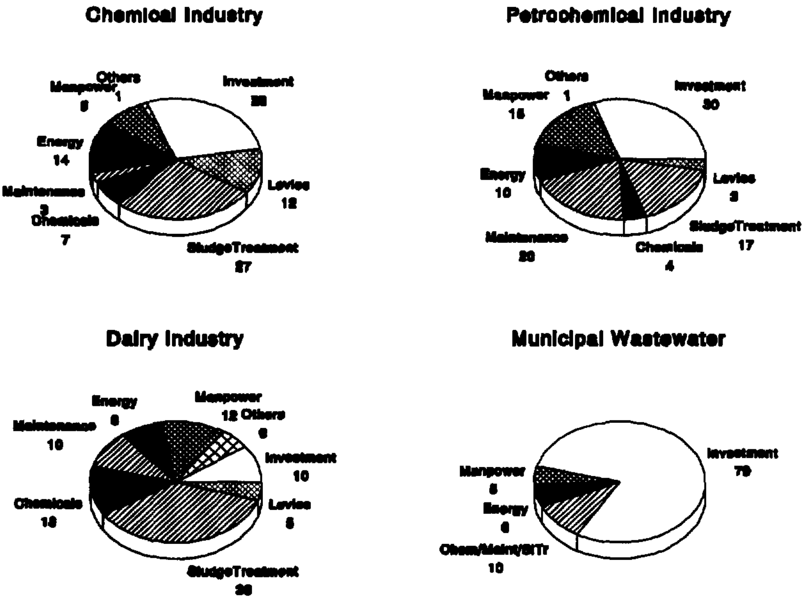

Figure 2. Examples of Cost Factor Distributions for some specific industrial and municipal WWTP's (after Vanderhaegen *et al.*, 1994).

Pullammanappalil *et al.* (1995) performed a similar exercise for anaerobic digestion but focused on the operating costs only. These authors found that usage of chemicals for pH control was responsible for half the operating costs (Figure 3). Evidently they proposed to improve on the pH-control performance to reduce the costs.

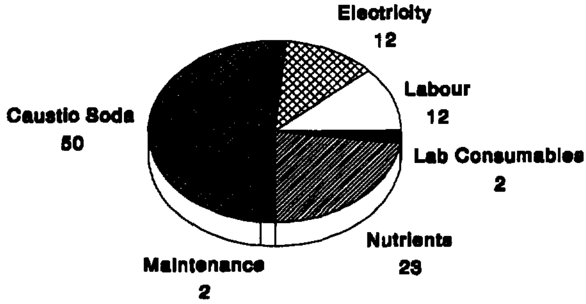

Figure 3. Cost factor distribution for operation of a high rate anaerobic digestion (Pullammanappallil *et al.*, 1995).

Spearing (1987) made a cost analysis between the activated sludge and trickling filter option for a treatment facility and confirmed that a clear difference existed in the investment and operation portions of the overall costs for these designs. It was concluded that different optimal solutions may exist depending on the priorities given in weighting operating and investment costs.

Examples of cost relationships for different cost factors

In general, power laws are applied to calculate investment costs as function of the process size:

$$\mathbf{COST = \Theta\ (Process\ Size)^n}$$

The process size is typically chosen to be a relevant and easy to measure plant characteristic such as the volume or area of a process unit, the design flow rate, installed mechanical power or pumping capacity. Depending on the process unit, *n* ranges between 0.25 and 1 (Smeers and Tyteca, 1984; Wright and Woods, 1993, 1994).

With respect to effluent levies, many cost functions in different countries are specific cases of the function given in Figure 4. Here, $\Delta\alpha$ and $\Delta\beta$ are the additional costs per volume of discharging an additional 1 mg/l below and above the regulatory discharge limit concentration $C^l_{discharge}$. β_0 is the additional cost when the discharge limit concentration is exceeded. For violation of the permit limit concentration C^l_{permit} a very high cost β_1 should be associated as this violation may lead to closure of a company or a production facility. To give some examples, the values of a number of these parameters are given for the Danish and Flemish situations.

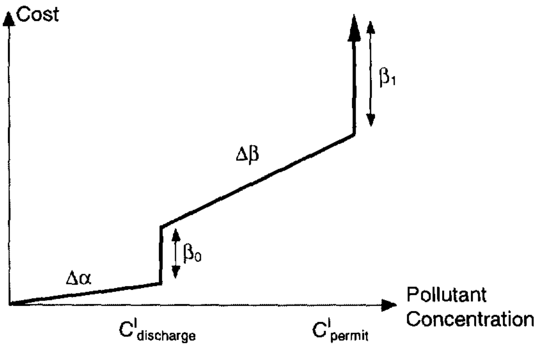

Figure 4. Suggested Cost Function for the discharge of nutrients. (For more details, see text).

In 1994 , Danish legislation imposed a cost of discharging ammonia and nitrate to the recipient of 4 ECU/kg NH_4-N and 2.5 ECU/kg NO_3-N when below the discharge limit ($\Delta\alpha$). When the discharge limits $C^l_{discharge}$ for ammonia (1.5 mg NH_4-N/l) or total nitrogen (8 mg N/l) are exceeded, the additional costs increase threefold. Some plants may have to comply with even stricter demands, depending on the recipient.

In Flanders the fines are based on a weighted sum of different variables that are considered important in the definition of effluent quality:

$$\text{Fine} = U_{nitfine} \bullet (k_{organic} \bullet N_{organic} + k_{metals} \bullet N_{metals} + k_{nutrients} \bullet N_{Nutrients} + N_{heat})$$

in which the *Unitfine* in 1996 is approximately 25 ECU per year. The weighting factors k_i are set depending on the discharge permit given. The values of N_i are given by the following functions of discharge flow and concentrations:

$$N_{organic} = \frac{Q_{day}}{180} \cdot \left(\frac{0.35 \cdot SS}{500} + 0.45 \cdot \frac{2 \cdot BOD_5 + COD}{1350} \right) \cdot (0.4 + 0.6 \cdot d)$$

$$N_{metals} = \frac{Q_{year}}{1000} \left(40 \cdot Hg + 10 \cdot (Ag + Cd) + 5 \cdot (Cu + Zn) + 2 \cdot Ni + As + Cr + Pb \right)$$

$$N_{nutrients} = \frac{Q_{year}}{10000} (N + P)$$

$$N_{heat} = 3.3 \frac{Q^{cool}}{10000}$$

where Q_{year} and Q^{cool} are the yearly effluent and cooling water discharge (m^3/year) respectively. Q_{day} is the daily discharge flow rate (l/d) and *d* is the yearly number of days discharge is taking place divided by 225. In case discharge occurs during less than 225 days per year the value of *d* is set to 1. All concentrations are given in mg/l.

OPEN PROBLEMS

A number of issues still have to be resolved. For instance, it may be useful to include a risk term in the objective function as more complex treatment systems may result in better performance at reduced costs, but may be prone to increased risk of failure (Beck, 1996). Many ways exist to reduce the risk of failure of a treatment facility, for instance, by inclusion of a storm tank or a calamity basin in a plant to accommodate for a storm event or a toxic input. An example at the level of plant operation is the preparation of a microbial community for new chemicals (to be produced in the near future in an industrial production facility) by supplying low amounts in the influent. Similarly, investing a considerable amount of carbon source in a low load period to maintain the levels of internal storage products needed for efficient biological phosphate removal can be seen as a means of reducing the risk of failure of a facility (Temmink *et al.*, 1996). Somehow in the analysis of a plant over its lifetime, it should be possible to quantify the risk or the costs associated with risk reduction and to account for these in the systematic performance index discussed in this paper.

Similarly, it may be important to stimulate flexibility in the design of a treatment plant although it will probably increase the investment and operating costs. However, over a long period of time, high flexibility may prove an advantage. In this respect the ideas and concepts put forth by Geldof (1995) are worth considering. With long-term decision making as a starting point, he develops the idea that optimal solutions must be regarded within their time perspective, i.e. what is optimal now may not be as such in the future. Moreover, an optimal solution now may be of such rigidity that future adjustments to track a changing optimum are no longer feasible because of the reduction in flexibility required to attain the current optimum solution. One way of reducing the risk of "getting stuck" in a rigid optimum is to consider trends and predictions of future requirements and to use this information to keep all options open, probably giving away part of short-term profit.

A second concept Geldof suggested is even more related to the flexibility issue, where he states that a middle course needs to be sought between static, rigid but reliable solutions and dynamic, flexible but vague (i.e. associated with high uncertainty) answers. This can be translated into an area that lies between too much order and too much chaos, which Geldof (1994) terms the area of complexity. The main characteristic of decision making is that the direction of change towards an optimum is clearly identified, but sufficient flexibility is built in to allow for adaptations to changes that will undoubtedly manifest themselves in the cost functional of a treatment facility.

TIME-SCALE SEPARATION

One may say that the formulation of a criterion has to be independent of the particular technology being used in the treatment system. The criterion may be formulated in a hierarchical manner. At the top level criteria should be given for sustainable effluent water quality. This leads of course to design criteria that have to be matched with operational costs. In the operation one can talk about a process planning stage (over weeks, considering the type of load to the plant). More specific process schemes can be a result of this, e.g. how to take care of small or large disturbances. The overall operational schemes subsequently lead to a number of local controllers for concentrations, flow rates etc.

From the above discussion it is apparent that decisions on a WWTP's operation and design should take into account that these are not only interlinked but that their evaluation can be performed in different time scales. A decision at the level of the design phase has impacts over the whole lifetime of a treatment plant, whereas a decision on, for instance, the dissolved oxygen control is effective within minutes. Therefore, a decision making process needs to address the whole span of time constants, which is not practical to include within the same analysis. It is proposed that the index is evaluated according to a hierarchical scheme in which the decisions in the long term (time scale = years, e.g. the design) are first evaluated, then the choices made in the medium term (time scale = months, e.g. installation of a chemical addition pump with controller) are addressed for a given long-term decision (a particular plant design). Finally, the short-term actions are considered within the boundaries imposed by the decisions taken at the higher levels (in terms of time resolution). For particular operations there may even be considerations in different time scales. For example, for DO control one will have a set-point, and the DO control system will adjust the DO concentration to it within a minutes (short-term decision). On the other hand, the set-point may be slowly changed - within days or weeks - in order to match the water quality, the balance between anoxic operations and aerobic operations, etc. (medium-term decision). At the lowest level, dynamic models play a key role as they assess the impact of operational decisions. In the medium and long time scales, it is advisable to use steady-state models for evaluation of the objective performance index.

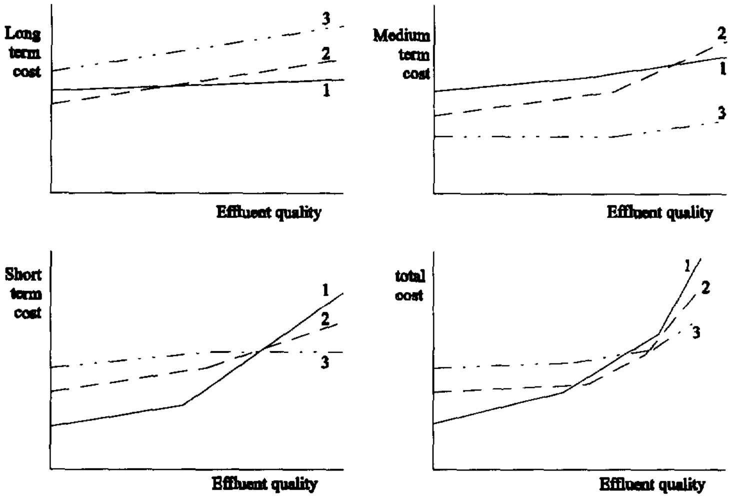

Figure 5. Evaluation of the objective performance index over different time scales.

The approach is illustrated in Figure 5, where three different designs are assessed in terms of desired effluent quality. The long-term costs of the designs are associated with the cost of building the plant. Based on the cost of design only, designs 1 and 2 should be chosen depending on the desired effluent quality. However, based on the medium-term costs, which include such costs as a change of operation mode to meet new standards or changes in wastewater load or incorporation of new technologies, design 3 should now be the most appropriate, since this more flexible design allows for modifications at the plant at a low additional cost. The short-term costs should also be taken into account, since these include the cost of day-to-day operation of the treatment plant, such as aeration, chemicals, maintenance, pumping, etc. The total costs of the different designs are shown in the last graph, where the costs of long, medium and short term have been aggregated. Each of the designs is optimal with respect to different effluent qualities. The effluent quality should be chosen according to present standards, but should also consider expectations of future standards.

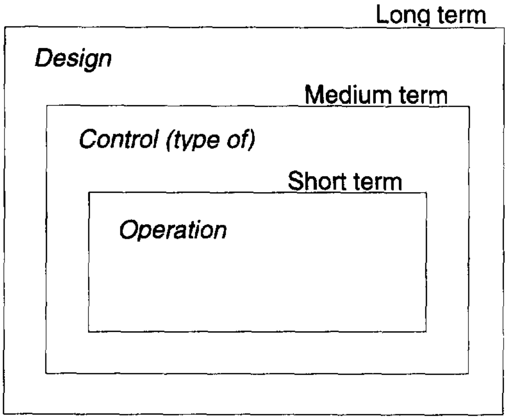

Figure 6. Limitations on decision making imposed by different time scales.

The three time scales must be considered together, since design is limiting the flexibility in choice of control, which again is limiting operational performance. This is shown in Figure 6. Once a particular design is chosen, the flexibility of the plant to change to new types of control can be limited. For instance, it can be difficult to apply step-feed control to a completely mixed reactor without changing the design. Flexibility in design is often more expensive in the beginning, but it is often beneficial when considered over the life-span of a treatment plant. Similarly, the medium-term decision variables which include the types of control applied to the plant are found to be limiting the operation. For instance, a biological phosphorus removal plant will often face larger variations in the effluent phosphorus level than a plant with chemical precipitation, albeit that both are controlled at their best. The choice of control strategy could of course also include a choice of flexibility level in order to allow adaptation to a change in disturbance characteristics. For instance, an alternating plant that is controlled with constant phase lengths is suboptimal in performance compared with a plant in which the control system allows for variations in phase length.

SPATIAL SEPARATION

While the discussed time-scale separation is one way of reducing the complexity of the problem, it is considered necessary to simplify the approach further. It is proposed that the problem is also separated in space by considering each unit process as a starting point. For instance, the index may first be evaluated for a final clarifier, an aeration tank, etc. Then the combination of the unit processes is evaluated by combining the results of each single analysis and by adding a modification term due to the interactions between the unit processes. Indeed, a spatial separation needs to be done with care since the unit processes interact closely with each other. Optimising each unit process individually may give a total result which is far from optimal. It appears essential that some further work is performed to address this problem systematically.

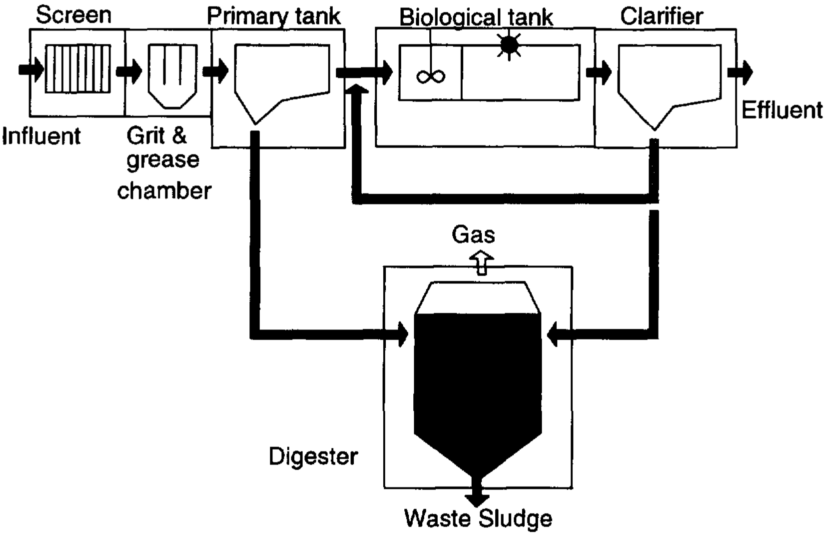

Figure 7. Spatial separation of unit processes in a WWTP.

Reconsider the WWT system in Figure 1. The total system is given by a single input-output-conversion cost relation. This rather complex system where decisions have to be made on the basis of the total system, can be segregated into several unit processes - each given by its own input-output-conversion cost relation. This is illustrated in Figure 7. Therefore, a set of cost relations can be formulated:

$$\begin{aligned}
\mathbf{J_1} &= \mathbf{Input_1 - Output_1 + Conversion_1}\\
\mathbf{J_2} &= \mathbf{Input_2 - Output_2 + Conversion_2}\\
\mathbf{J_3} &= \mathbf{Input_3 - Output_3 + Conversion_3}\\
&\bullet\\
&\bullet\\
&\bullet\\
\mathbf{J_N} &= \mathbf{Input_N - Output_N + Conversion_N}
\end{aligned}$$

Many of these unit processes have common boundaries such that the output from one unit process is the input to the next unit process. Hence, many input/output terms cancel out. This set of equations can be solved w.r.t. unknown terms. Probably the most natural way of dealing with this set of equations is to set up the costs of discharge to the recipient ($Output_N$) and find the costs of conversion ($S\ Conversion_i$). Solving this system with the aim of finding a WWTP which is non-profitable (i.e. setting all left-hand side terms to zero) would reveal the cost of the input term ($Input_I$), i.e. the levies that tax payers and industry have to pay for treatment of their wastewater produced. Another example of the use of the index could be given on the choice between several clarifier designs. Here the *input* term is the output of the activated sludge tank, the *output* term is the effluent discharge and the return sludge, the quality (and value) of which depends on the type, size and operation of clarifier, and the *conversion* costs that also depend on the choice of clarifier. For instance, a large clarifier would result in better effluent quality, larger receiving capacity of activated sludge (sludge loading) but larger conversion cost. Thus, the choice of clarifier design may be judged on the basis of the cost of the clarifier unit process.

DISCUSSION

The given criterion has to be a basis for evaluation of alternative designs or operational schemes. With a reasonable criterion a tool is available that will make it possible to study how the system will depend on a number of parameters, i.e. the sensitivity to various changes can be studied. Typical questions are:

- what would a significant increase in energy cost mean for the design and operation of the plant?
- what would a water consumption reduction mean for the treatment plant?
- what would happen if the P consumption (due to other detergents) were reduced to half?
- how would the water quality change if the influent flow rate were reduced to half?
- how is the operation changed, if nutrient recycling is taken into consideration?

A number of studies have appeared in which the problem of evaluating the combined costs of treatment plant investment and operation is approached from a quantitative (cost) point of view.

Vanderhaegen *et al.* (1994) applied simulation to evaluate the dependence of overall costs on plant size and operation with the particular goal of reducing the sludge treatment costs as these were found to be a major contribution to the overall costs of treatment in an industrial context.

In the work of Coen *et al.* (1995) an evaluation was made to assess whether the inclusion of real-time control could allow us to omit a foreseen expansion of the reactor volumes. It was found that an investment in new reactor volumes could be replaced by a 7-fold lower costing upgrade with a more flexible aeration system, investment in variable flow recycle pumps and a few cheap sensors.

In another example of Nielsen and Önnerth (1996) the introduction of phase length (intermittent aeration), varying recirculation rate and oxygen setpoint control were evaluated. In simulation, reductions of 25 percent of the energy costs, complete elimination of the lime dosage and reduction with 45 percent of the ethanol addition in the predenitrification zone were found to be possible. Implementation in practice led to even better results as it was found that the investment in nutrient sensors and a real-time control system led to a 40 percent reduction in energy consumption, while all external carbon and lime dosage could be omitted. Evidently, substantial cost savings could be obtained.

The cost contributions of levies and aeration were evaluated by Carstensen *et al.* (1994) and the impact of more sophisticated control systems was assessed. In Table 1 costs calculated through simulation for three levels of control complexity are compared. In scenario 1, the traditional operation of the type of nutrient removal plant studied with fixed 4-hour phase lengths and a fixed dissolved oxygen setpoint at 2.0 mg/l, allows to comply with effluent standards. However, similar effluent quality can be obtained under scenario 2 which is characterised by an optimised but static operation with 2-hour phase lengths and a (still fixed, but optimised) setpoint of 1.7 mg/l. Finally; the third, most complex operating scenario, works with dynamically changing phase lengths and oxygen setpoints that are calculated from on-line influent load and biological activity measurements. From the simulation results (Table 1) it is obvious that the considered costs diminish, but one must also consider the additional costs for instrumentation, more capable actuators and better qualified personnel, as required for the implementation of these more complex operating schemes. With the simulation model and the advanced control scheme, the effect of a reduction of the volumes of the aeration tanks was calculated. It was found that a volume reduction of 20 percent was feasible without incurring more operating costs than the ones found under Scenario 1. Hence, large savings on the cost of construction may be obtained by introduction of advanced control systems.

Table 1. Operating costs (ECU/day) for an alternating nitrogen removal plant (Carstensen *et al.*, 1994)

	Scenario 1	Scenario 2	Scenario 3
Aerating Costs	260	242	192
Discharge Costs	405	298	254
Total	665	540	446
Mean Ammonia Concentration (mg NH_4-N/l)	0.67	0.71	0.95
Mean Nitrate Concentration (mg NO_3-N/l)	4.75	3.60	2.97

However; support of such exercises by dynamic models and validated cost functions is a prerequisite for widespread acceptance of the methodology as an optimisation approach during design of new or upgrade of existing treatment plants. More specifically, the performance index can direct optimisation studies. If one is in the design phase of a new WWTP, an a priori performance index could be used:

$$\mathbf{J_{ap} = Input_{ap} - Output_{ap} + Conversion_{ap}}$$

where Input_{ap}: the expected negative value of the input
Output_{ap}: the required output quality
Conversion_{ap}: the expected conversion costs

The optimisation problem then boils down to finding the optimal Conversion_{ap} given Input_{ap} and Output_{ap}. After the plant has been built (and the constraints have been imposed !) the actual input, operating costs and output values can be evaluated and a local optimisation study can be initiated that has to evaluate a "running" performance index.

$$\mathbf{J_r = Input_r - Output_r + Conversion_r}$$

where Input_r: the quality of the input
Output_r: the obtained output quality
Conversion_r: the obtained conversion costs

It might not be needed to include investment costs here since these are fixed. However, taking into account that it may sometimes be fruitful to invest in an instrumentation and automation programme, one should leave that flexibility open for such optimisation at "run-time". Examples of degrees of freedom to consider and decisions to be made are for instance the education level of personnel, increase of carbon dosage, change of maintenance programme (e.g. reduction of maintenance efforts at the expense of an increased risk of failure), etc. It is worth mentioning that a cost-effective (in the sense of the criterion above) treatment of wastewater may require sophisticated "production planning" since the minimisation of a criterion like the one above is nontrivial.

The tradition of "best available technology" has been used extensively. This philosophy is advisable when there is insufficient knowledge about the environmental impact. However, environmental protection laws typically state that the costs for treatment have to be compared with the impact of the pollution disturbances. A best "available technology", however, also implies that too little importance is given to finding out the impact of the pollution disturbances. A philosophy where pollution impact is compared with treatment costs requires more knowledge. A sustainable development further emphasises the need for better knowledge. Therefore, it is stressed that realistic criteria are crucial elements for such analysis.

In the future one has also to look for a more positive use of treatment works. Indeed, production of energy, (bio)chemical consumption goods and recirculation of nutrients may become tasks of WWT. It could be possible to create positive consumer products that would be produced within the treatment process. For example, one must realise that biodegradation works with micro-organism cultures that may at the same time be (en)able(d) to synthesise substances from influent wastewater into useful products, e.g. PHB, a bioplastic. For this reason, there is a challenge to obtain a "tailor-made" water for the actual location and its receiving water. From this stems the recent interest in a concept termed "waste design" and the sustainability issues are certainly relevant in this context.

BENCHMARK PROPOSAL

One way to stimulate work and discussions in the area discussed in this paper may be to create a benchmark problem for the control of a WWTP. The overall goal could be to minimise a cost function that can include effluent quality, energy costs, costs for chemicals and maybe also the additional investment cost for actuators (pumps and valves) and sensors etc. For this, a specific plant layout would be given together with some other crucial prior information. The problem for a benchmark study is then to design a pertinent control/design strategy that achieves a low value on the cost function.

A suitable form for a benchmark problem like this is to use a scrambled simulation code of the plant under study. It is important to have a well defined and clear user interface. Given measurement requests and control signals etc. from the user, the code should simulate a plant response and also maybe calculate the cost function. The benchmark problem may be supplied to interested investigators by using e.g. email, ftp or ordinary mail with a floppy disk. The development of a benchmark for control and design of wastewater treatment plants may be performed by a small task group, possibly after a call for problem suggestions.

It is worth mentioning that benchmark problems in the area of automatic control and identification have generated significant interest and discussion. For example in the 6th IFAC World Congress in 1993, one of the most popular sessions was the presentation of benchmark problem results. At this moment a Task Force (in which the authors are involved) has been created within the European COST-682 action (Dochain *et al.*, 1995) that as its mission has the development of a benchmark suite for evaluation of optimal wastewater treatment design and operation. A first prototype of the benchmark is expected to be available by the end of 1996.

CONCLUSIONS

The importance of using a systematic approach to formulate an integrated performance index (cost function) for design and operation of WWT plants has been stressed in this paper. A cost function is put forth as a fundamental support tool in the context of efficient treatment of wastewater and is hence of considerable importance. In order to simplify the evaluation of a cost function, both time-scale and spatial separations were suggested. Furthermore, some simple examples were provided to highlight and clarify the concept.

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