JMB

DYNAMIC MODELLING OF WASTEWATER TREATMENT PROCESSES

A Survey of the Literature

M.B. Beck and P. Lessard

Department of Civil Engineering
Imperial College
London SW7 2BU

A report prepared for the Water Research Centre,
Stevenage Laboratory.

October, 1987.

DYNAMIC MODELLING OF WASTEWATER TREATMENT PROCESSES

A Survey of the Literature

TERMS OF REFERENCE

Object

To review and classify the literature on the dynamic modelling and control of the major unit processes of wastewater treatment.

Scope of the Survey

The survey should include the following unit processes:

(a)	Primary settlement and storm tanks;	
(b)	Activated sludge	- aeration tanks
		- oxidation ditches
(c)	Biological filters;	
(d)	Rotating biological contactors;	
(e)	Secondary settlement	- activated sludge
		- humus sludge;
(f)	Tertiary treatment	- filtration processes
		- biological processes;
(g)	Sludge treatment	- anaerobic digestion.

Items (a) to (e) are of greatest importance and it is recognised that the extent of the coverage of items (f) and (g) will be dependent upon the time which remains available for their completion.

The survey will be conducted with a view to the operation of sewage treatment plants in the wider context of river quality management, i.e. with particular reference to:

(a) The possible application of variable (e.g. seasonally or river quality dependent) effluent consent conditions;

(b) The possible need to co-ordinate the operation of stormwater discharges and sewage treatment plant operations;

(c) The suitability of existing models for the simulation of the effects of alternative control strategies for both individual unit processes and for complete treatment plants.

The survey is to be concluded by a number of recommendations covering:

(a) Identification of those areas in which significant deficiencies exist in current dynamic process models;

(b) The nature and direction of further developments in the dynamic modelling of sewage treatment processes;

(c) The likely resource requirements for any such developments.

SUMMARY CONCLUSIONS AND RECOMMENDATIONS

CONCLUSIONS

1. The literature on dynamic modelling of the following unit processes of wastewater treatment has been reviewed:

   (i) plant influent;
   (ii) storm retention tanks;
   (iii) primary sedimentation;
   (iv) activated sludge-aerator;
   (v) activated sludge-secondary clarifier;
   (vi) trickling filters and rotating biological contactors;
   (vii) humus tanks;
   (viii) sludge thickening, aerobic digestion, and anaerobic digestion.

2. The greatest concentration of research effort has been directed towards modelling the dynamics of carbonaceous substrate oxidation and nitrification in the activated sludge aerator.

3. In contrast, little or no work has been carried out on sludge thickening, the humus tanks, and the pollutant removal characteristics of storm retention tanks.

4. While substantial research has been completed on bulking sludge, the thickening and clarification properties of the secondary clarifier, and the fundamentals of biofilm kinetics, results so far are largely theoretical and are unlikely to yield practical operational benefits in the immediate future.

5. In general, the problems of model calibration, verification, and validation have not been tackled in a systematic manner (in this the development of wastewater treatment plant models lags significantly behind similar developments in river water quality modelling).

6. Almost all dynamic models have been developed for the purposes of unit process control system design (or for the formalisation of basic understanding). There have been one or two attempts to assemble models of the complete liquid treatment line, although these have been somewhat speculative. Consequently, no serious attention has yet been given to the application of any models for the purposes of analysing plant operations under variable consent conditions, for stormwater control, for the analysis of design-operation interactions, or as personnel training simulators.

RECOMMENDATIONS

7. Our recent experience of work at the Norwich Sewage Works has convinced us that there is no real substitute for intensive, specialised monitoring exercises at full-scale plant if the development of dynamic models is to make significant and relevant progress in the future. Such monitoring exercises expressly for the purposes of modelling are necessary for all the unit processes of wastewater treatment. However, as a matter of priority the following exercises are recommended (in order of importance):

(i) A comprehensive experiment on the secondary clarifier, separately identified from the activated sludge aerator;

(ii) Likewise, for storm retention tanks and primary sedimentation tanks during storm events;

(iii) Routine (hourly) monitoring of trickling filter and humus tanks performance (preferably separately identified), to establish whether these processes have any significant shorter-term dynamic variability relevant to modelling and control.

8. Apart from model verification/validation, the most urgent requirement for further model development is for the dynamics of sludge settleability to be incorporated into the frameworks of existing

activated sludge models. This might involve multi-species models, the consideration of two-phase (solid-liquid) interactions, and possibly the exploitation of ideas from artificial intelligence, in particular, qualitative simulation models. Such research, properly supported by field-experiments, might best be conducted along the lines of a PhD thesis. It is especially pertinent to note that Olsson from Sweden believes a multi-species model and related time-variable DO set-point strategies to be central to further progress in activated sludge process control.

9. The two most important software "products" to have readily available for widespread application are:

   (i) An alternative to the IAWPRC model of the activated sludge process (as a matter of tactics);

   (ii) A package for time-series analysis of plant influent variations, preferably based on recursive estimation algorithms (with a view to subsequent on-line estimation/forecasting).

10. Current knowledge of plant influent characteristics, and the models of the primary sedimentation, activated sludge aerator, and secondary clarifier units are sufficient (with some license for extrapolation of storm tank behaviour) for preliminary simulation studies of stormwater control strategies. In our opinion the issues for research on stormwater control strategies are at present much better defined than they are for operational control under variable consent conditions. The results of the research are more likely to be accepted by the industry and, if carried through to fruition, in due course could establish two important precedents:

   (i) The principle of operational water quality management (or integrated river basin management) <u>in practice</u>, i.e. that control decisions are taken in the plant as a result of changing river conditions;

   (ii) The use of on-line dynamic models for assisting operational decision-making in complex situations where a fast response in real-time is necessary.

It is recommended therefore that (off-line) simulation studies and experimental monitoring directed towards stormwater control strategies should precede studies related to both on-line estimation/forecasting (as part of an implemented control strategy) and operation under variable consent conditions.

11. Operation under variable consent conditions and the more widespread introduction of the relatively advanced algorithms of on-line estimation/forecasting are both topics on which preparatory research in academic institutions might be initiated in the near future. The acceptability of the latter to the industry, and of the related aspects of uncertainty analysis, expert-advisor interfaces, and personnel training simulators, would appear to be crucial, and may therefore benefit greatly from some appropriate "market research" of the industry's attitudes. In support of such market research we note that treatment plants in Malmo and Stockholm, Sweden, already use visual displays of on-line estimates of the specific oxygen utilisation rate. The Swedish Waterworks Association (VAV) is currently preparing a "manual of practice" on computer presentation of data, statistical analysis of data, time-series analysis, and recursive estimation algorithms for the purposes of process control. The objective of the "manual" is to harmonise software requirements throughout the country and to assist users in preparing more relevant specifications for the software manufacturers. The Canadians have already requested an English translation of the manual.

12. As far as possible the research on wastewater treatment plant control within the context of river basin management should be conducted on a case-study basis. While much further thought must be given to the conceptual basis of the subject, it is vital for practical benefits to be demonstrated at every stage, and especially in the first instance.

13. In the course of our review, less emphasis has been placed on the progress and role of dynamic modelling of sludge treatment processes. This does not imply a lack of significance of these processes in operational management; they are known to have important feedback

interactions with the dynamic behaviour of the processes of the liquid treatment train. It would be a distinct oversight of any future research to assume a priori that such interactions are negligible.

PLANNED RESEARCH AT IMPERIAL COLLEGE

14. In the coming year we expect to couple a dynamic model of the liquid line of the treatment plant with dynamic models of the receiving water in order to examine both stormwater control strategies and seasonal treatment policies. Over the next three years SERC-funding for the former is hoped for; there will also be participation in a U.S.-funded project on the development of an integrated dynamic model of the entire wastewater treatment plant. Collaboration with WRc is integral to the former; regular liaison on the latter is recommended.

CONTENTS

		Page
	Terms of Reference	i
	Summary Conclusions and Recommendations	ii
	Contents	vii
1.	INTRODUCTION	1
2.	FRAMEWORK FOR MODEL CLASSIFICATION	3
2.1.	The Operational Control Problem	3
2.2.	Model Types	4
2.3.	Development and Evaluation	16
2.4.	Applications	20
3.	THE REVIEW	27
3.1.	Plant Influent	27
3.2.	Storm Retention Tank	31
3.3.	Primary Sedimentation	41
3.4.	Secondary Biological Treatment: Suspended Growth Processes	49
3.5.	Secondary Biological Treatment: Attached Growth Processes	91
3.6.	Sludge Treatment	105
3.7.	The Plant as a Whole	114
4.	CONCLUSIONS AND RECOMMENDATIONS	116
4.1.	Overall Progress	117
4.2.	Unit Processes	121
4.3.	Model Application	132
4.4.	Programme of Research at Imperial College	134
REFERENCES		136
Appendix 1:	"Identification, Estimation and Control of Biological Wastewater Treatment Processes"	157
Appendix 2:	Further Introductory Notes on On-line Estimation/Forecasting	169
Appendix 3:	"The Effects of Stormwater Runoff on Wastewater Treatment Plant Operations" (SERC Proposal).	175

DYNAMIC MODELLING OF WASTEWATER TREATMENT PROCESSES

1. INTRODUCTION

Research into the development of dynamic models of wastewater treatment processes now has a history of some twenty years, although since the tutorial paper of Andrews (1974) and the review paper by Olsson (1977) no further comprehensive surveys have appeared in the open literature (as far as we are aware). There have, however, been papers of a more specific review nature published in the past ten years, for example on the modelling of the biological unit processes (James, 1982; Olsson, 1986) and on the identification, estimation, and control of these processes (Olsson, 1980; Beck, 1986). For completeness this last paper (Beck, 1986) is attached to this report as Appendix 1; it is complementary to the present survey.

In this report the status of dynamic mathematical modelling of the following unit processes is assessed:

(i) Storm retention tanks;
(ii) Primary sedimentation;
(iii) Activated sludge, aerator basin;
(iv) Activated sludge, secondary clarifier;
(v) Biofilm processes, i.e. trickling filter, and rotating biological contactor;
(vi) Humus tank;
(vii) Sludge thickening tank;
(viii) Aerobic digestion of sludges;
(ix) Anaerobic digestion of sludges.

For any of these unit processes there may be different objectives for the development of a dynamic model, different ways of expressing the model, and a variety of means of testing its adequacy. Section 2 of the report sets out, therefore, the general framework that we have used to classify the published material. Section 2 is, as it were, the "theory" of the review, and Section 3, the review proper, the "practice". The trends established, and the gaps between theory and practice, are summarised in Section 4, which deals with our conclusions and recommendations, including a brief statement of our own plans for research over the next one to three years.

The purpose of the present survey, while covering as much of the literature as possible, has primarily been to evaluate the sorts of models that would be most useful to studying and advancing the operation of wastewater treatment plants in the wider context of river water quality management. In particular, therefore, the potential applications of dynamic models are classified and organised around that specific point of view. In fact it will be apparent that few, if any, studies have had such a broader issue as their objective.

Most dynamic models have been developed for the purposes of control system design for individual unit proceses. Most probably because of this focus the activated sludge aerator and the anaerobic digester have been by far the most common subjects for the development of dynamic models. In contrast, and in spite of a recent, rapidly growing interest in the modelling of biofilm dynamics, there are no process-scale models of either trickling filters or rotating biological contactors. Conventional wisdom, we presume, is that these attached-growth processes do not suffer from operational control problems, a position that should not necessarily be left unchallenged.

There are as yet no well corroborated models for an entire wastewater treatment plant, nor even of the complete liquid treatment line. With regard to the application of such models for the purposes of controlling transient river pollution due to stormwater runoff or of managing river basins by variable effluent consent conditions, it is our opinion that the former is for the present the much better defined and more feasible proposition. Whether the industry is ready to embrace the concept of on-line models suitable for day-to-day operational decision-making is, however, still an open question. Anglian Water already have a model for real-time pollution event forecasting, and it may well be that the greatest scope for such innovative applications will indeed be in the more complex stormwater control situations.

2. FRAMEWORK FOR MODEL CLASSIFICATION

Our objective has been to classify the literature on dynamic models under the three broad headings of:

(a) Model Type, e.g. input/output, distributed-parameter, lumped-parameter;

(b) Development and Evaluation, i.e. method of model calibration, type of field data used (e.g. from laboratory-scale process or pilot plant, sampling frequency, duration of observed record);

(c) Application, e.g. on-line state estimation, study of design-operation interactions.

Since this may introduce terms that are unfamiliar, there is a need to define such terms, for which purpose it will be assumed that the basic problem context is unit process operational control.

This is not restrictive, however. It does not exclude an interest in purely scientific questions of understanding process behaviour, for which a dynamic, unsteady-state model is also the most natural form of representation. Nor does it exclude consideration of design problems, although these have mostly, if not always, been addressed through the use of steady-state, static models, i.e. models in which none of the variables are in any way a function of time.

2.1. The Operational Control Problem

The capacity to control successfully the dynamic behaviour of a unit process depends essentially on three factors:

(i) The ability to observe the state of the process and its response to various perturbations (i.e. monitoring);

(ii) The ability to relate causes (inputs, controls) to effects (outputs, responses) unambiguously;

(iii) The capacity to act, i.e. to manipulate the causes (control inputs), in order to correct undesirable effects or to bring about more desirable effects.

Successful control depends therefore on progress along three broad fronts, respectively: (i) developments in sensors and measuring devices; (ii) improvements in understanding and process knowledge; and (iii) advances in control actuators and (more fundamentally) process designs.

The focus of this review is clearly the second of the above three factors, although for obvious practical reasons it does not make sense to ignore the important interactions among them. For example, a lack of sensors can be compensated by the use of state estimation methods, although this requires a good knowledge of process behaviour; and a lack of knowledge can to some extent be compensated by an adaptive controller, which tracks (or attempts to learn) process behaviour at the same time as it carries out the control function.

To summarise, Figure 2.1 defines the elements of the operational control problem. Given this setting, the following is the framework by which the literature on dynamic models will be classified.

2.2. Model Types

(a) Linguistic Model

Relating cause to effect does not necessarily demand the construction of a mathematical model (in the normal sense of a set of algebraic equations, ordinary differential equations, or

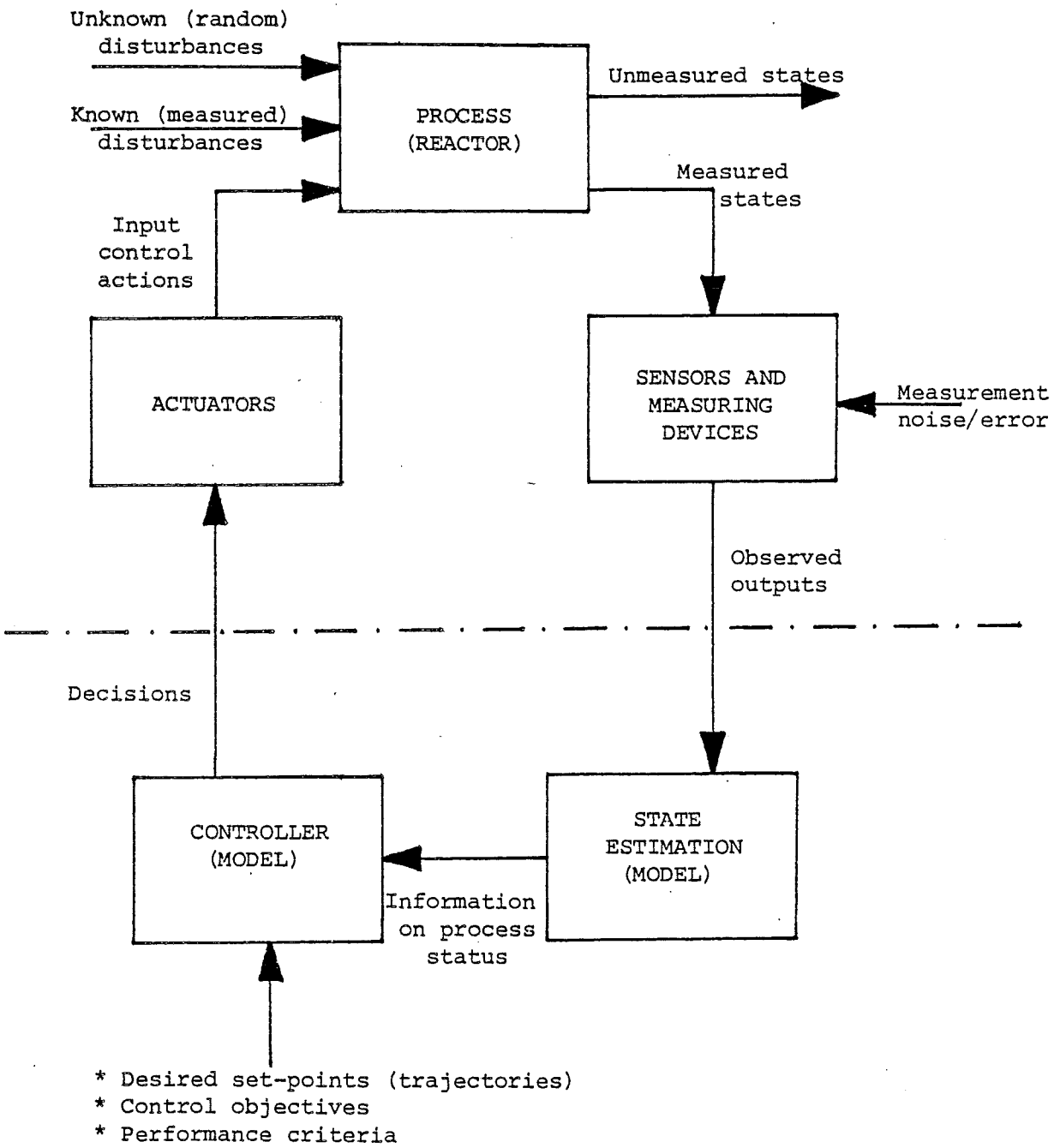


Figure 2.1 Basic elements of the archetypical control system

partial differential equations). At the beginning of every model-building exercise there must be some verbal, or linguistic, form of hypothesis (relationship). For example, if one were to conceptualise the elementary relationships underlying the phenomenon of bulking sludge, initial thoughts might lie along the following lines,

> "IF" bulk dissolved oxygen concentration low "AND" floc particles large "THEN" floc dissolved oxygen concentration very low

> "IF" floc dissolved oxygen concentration very low "THEN" filamentous organisms almost absent from floc

> "IF" filamentous organisms absent from floc "THEN" floc particles break up (i.e. decrease in size)

and so on. Personal reflection shows this to be a most natural means of organising prototype theories about the behaviour of a system; it is integral, albeit in an implicit way, to the development of mathematical models.

Most texts on mathematical modelling and computer simulation would not previously have paid much attention to this form of knowledge representation. The current popularity of artificial intelligence methods is likely to change that situation substantially.

(b) Time-series Model

The collection and interpretation of observations of process behaviour is complementary to the organisation of prototype theories. For the development and evaluation of any type of dynamic model these observations will have been collected in the form of time-series data. Certain inputs to the process,

including both natural (influent) disturbances and control actions, and certain output responses, can be observed at discrete instants of time t_k over a period $t_0 \rightarrow t_N$. Ideally, all the variables would have been observed at essentially the same instant of time, these observations being separated by equal intervals of time, i.e. $(t_k - t_{k-1})$ = constant; the observations would have been made as frequently as possible; and the experiment conducted for as long as possible (although not without regard to the natural time-constants of the process under investigation).

The data will usually comprise the following sets of time-series

$$[\underline{u}(t_0),\ \underline{u}(t_1), \ldots,\ \underline{u}(t_k), \ldots, \underline{u}(t_N)]$$
$$[\underline{y}(t_0),\ \underline{y}(t_1), \ldots,\ \underline{y}(t_k), \ldots, \underline{y}(t_N)] \qquad (2.1)$$

where, according to Figure 2.2,

$\underline{u}$ is a vector of input variables (measured disturbances and/or control actions)

$\underline{y}$ is a vector of output variables (most usually, error-corrupted observations of the process states, see Figure 2.1).

Any relationship between $\underline{u}$ and $\underline{y}$ that can be identified from these data will be referred to here as an input/output form of time-series model, which, for the scalar case (single input; single output) can be defined in general as

$$y(t_k) = \phi_1 y(t_{k-1}) + \phi_2 y(t_{k-2}) + \ldots \phi_n y(t_{k-n}) +$$
$$\theta_1 u(t_{k-1}) + \theta_2 u(t_{k-2}) + \ldots \theta_n u(t_{k-n}) +$$
$$\text{(random error/noise terms)} \qquad (2.2)$$

in which

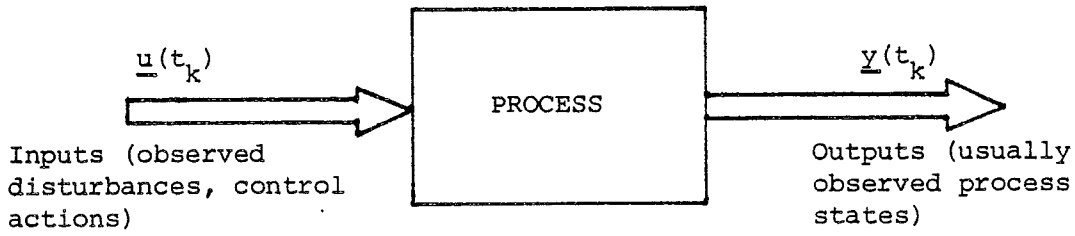


Figure 2.2 Process representation for an input-output time-series model

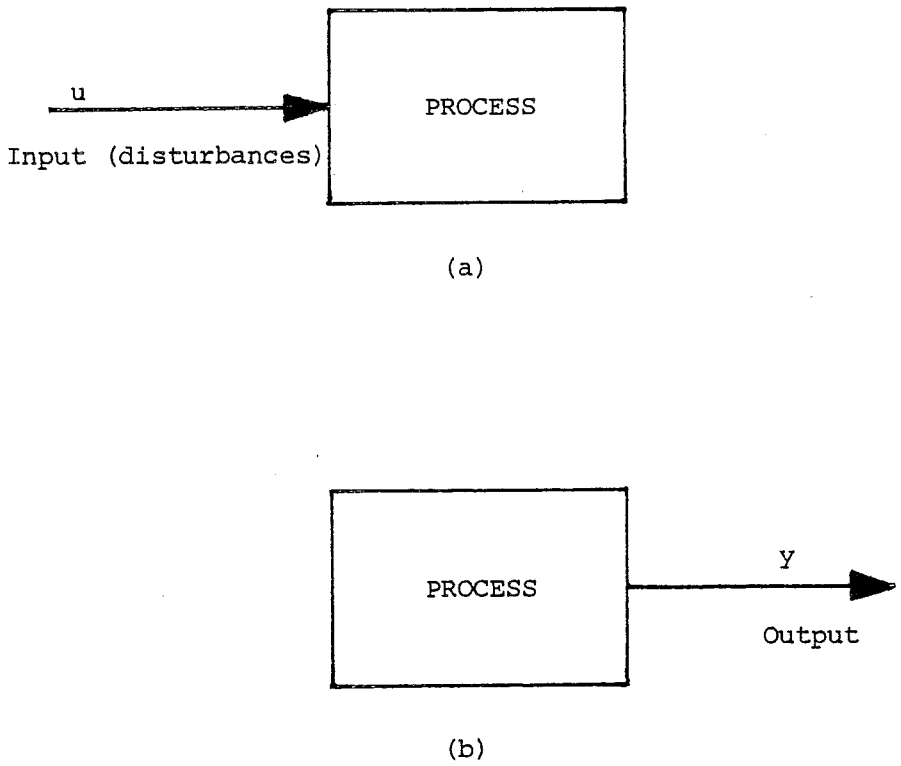


Figure 2.3 Univariate time-series models: (a) for the input; (b) for the output.

Φ_i, θ_i are the unknown <u>model parameters</u> (coefficients) to be estimated from the data.

Such a relationship might otherwise be referred to as a black box model or a transfer function model. It has the form of a discrete-time, algebraic, difference equation; and it is the form of dynamic model most commonly encountered in control theory.

Various assumptions can be made about the nature of the random error/noise terms in equation (2.2), including a model of the collective effects of these random, uncertain elements, the sources of which are usually considered to be measurement error and unknown input disturbances (see Figure 2.1). Without the explicit inclusion of these random terms, equation (2.2) reflects a straightforward <u>deterministic</u> relationship between input and output. Nonetheless, suitable assumptions about the nature of the random elements of the data will influence a proper identification of the form of the deterministic component of equation (2.2).

Two special cases of equation (2.2) are those in which interest centres not upon the relationship <u>between</u> input and output, but upon relationships that characterise the sequence of temporal variations observed in the behaviour of the input or the behaviour of the output. Thus, in line with Figure 2.3,

$$y(t_k) = \Phi_1 y(t_{k-1}) + \Phi_2 y(t_{k-2}) + \ldots + \Phi_n y(t_{k-n}) + \text{(random terms)} \tag{2.3a}$$

$$u(t_k) = \theta_1 u(t_{k-1}) + \theta_2 u(t_{k-2}) + \ldots + \theta_n u(t_{k-1}) + \text{(random terms)} \tag{2.3b}$$

The purpose of such <u>univariate</u> time-series models would most obviously be for the generation of synthetic data on the plant influent or effluent. There will be cases where it is not necessary to have a detailed sewer network model for generating the inputs to a treatment plant model, nor likewise a plant model for generating inputs to a river model.

(c) Lumped-parameter Model

All unit processes exhibit behaviour that varies with both time and space. The focus of the control problem, however, is its concern with the manipulation of changes with respect to time, and the majority of the methods of control theory reflect this emphasis. They are strongly geared to the use of lumped-parameter models.

If it is assumed that all the properties of the unit process are uniform across a given (spatial) volume of the reactor comprising the process, mass balances for each constituent substance in the volume, with concentration x, yield equations of the following form,

$$V[dx(t)/dt] = Vx(t) = (\text{mass flux})_{in} - (\text{mass flux})_{out} \pm (\text{mass generated/consumed in reactions}) \tag{2.4}$$

where t is the single independent variable of (continuous) time, and V is the volume over which the properties of the reactor are assumed to be uniform. The implied complete mixing of the fluids within this volume permits the designation of this lumped-parameter model as a continuously stirred tank reactor (CSTR) approximation. Frequently it is assumed that the properties of the unit process are uniform across its entire volume.

Given several interacting substances ($\underline{x}$) of interest to the characterisation of the state of the process at any time, and given that observations $\underline{y}$ of the concentrations of some of these substances are made at discrete instants of time t_k, we have the following more general and formal statement of a lumped-parameter model:

$$\dot{\underline{x}}(t) = \underline{f}[\underline{x}, \underline{u}, \underline{\alpha}; t] + (\text{random terms}) \tag{2.5a}$$

$$\underline{y}(t_k) = \underline{h}[\underline{x}, \underline{\alpha}; t_k] + (\text{random terms}) \tag{2.5b}$$

Here

$\underline{x}$ is a vector of <u>state variables</u> quantifying the internal state of the process, almost exclusively in terms of a set of concentrations of dissolved or suspended substances,

$\underline{\alpha}$ is a vector of <u>model parameters</u> (coefficients), typically the rate constants for the various biological and chemical reactions,

$\underline{f}$ is a vector of nonlinear expressions for the mass flux and reaction terms influencing the dynamics of each state variable, and

$\underline{h}$ is a vector of expressions indicating essentially how the states $\underline{x}$ are related to the observations $\underline{y}$.

Equation (2.5a) may be referred to as the state equation, and equation (2.5b) as the observations equation.

The significance of the lumped-parameter model is that it involves nothing more complex than the set of ordinary differential equations of equation (2.5a) and is not <u>explicitly</u> dependent upon the independent variable of space. It is clearly quite possible, however, for a unit process to be simulated by multiple CSTR elements, in which case the state vector $\underline{x}$ would contain within its definition concentrations of the same substance in the various component volumes of the overall process. Spatial variations would in this case be implicit in the model.

Conceptually the model can be represented as in Figure 2.4. The choices of state variables and the parameters embedded in the expressions describing the interactions among the state variables collectively define what can be called the <u>structure</u> of the model. Not all of the state variables may be measurable. Some, denoted $\underline{x}_u$, may be unobservable. For example, the aerator basin of an activated sludge unit might be approximated by a sequence of four

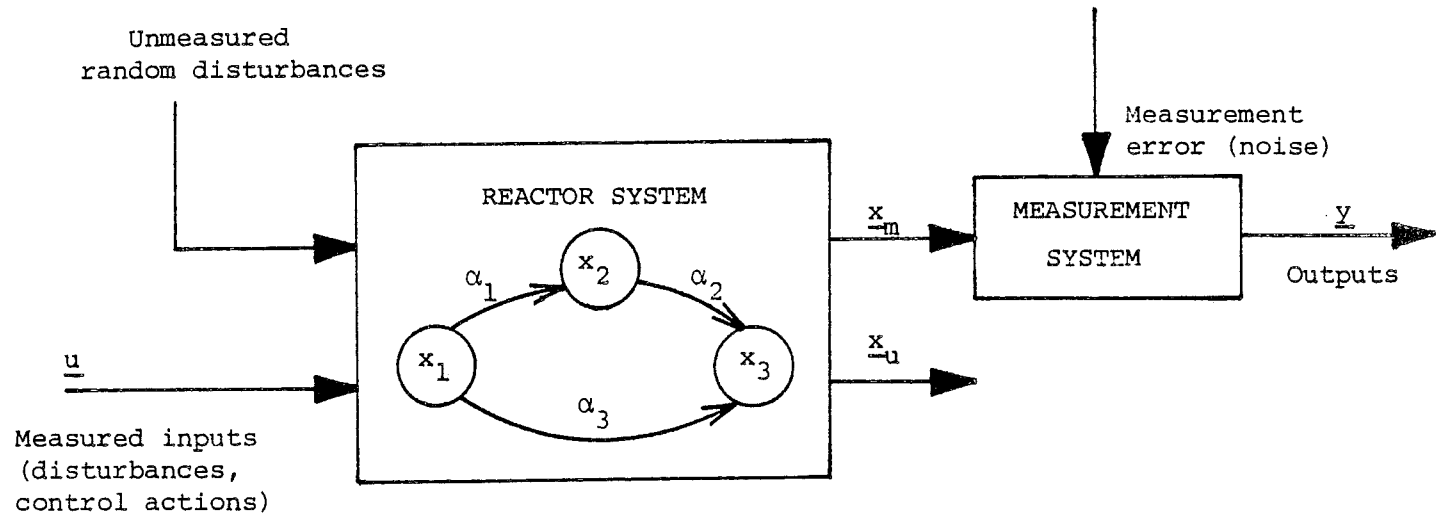


Figure 2.4 Process representation for the lumped-parameter (state-space) model.

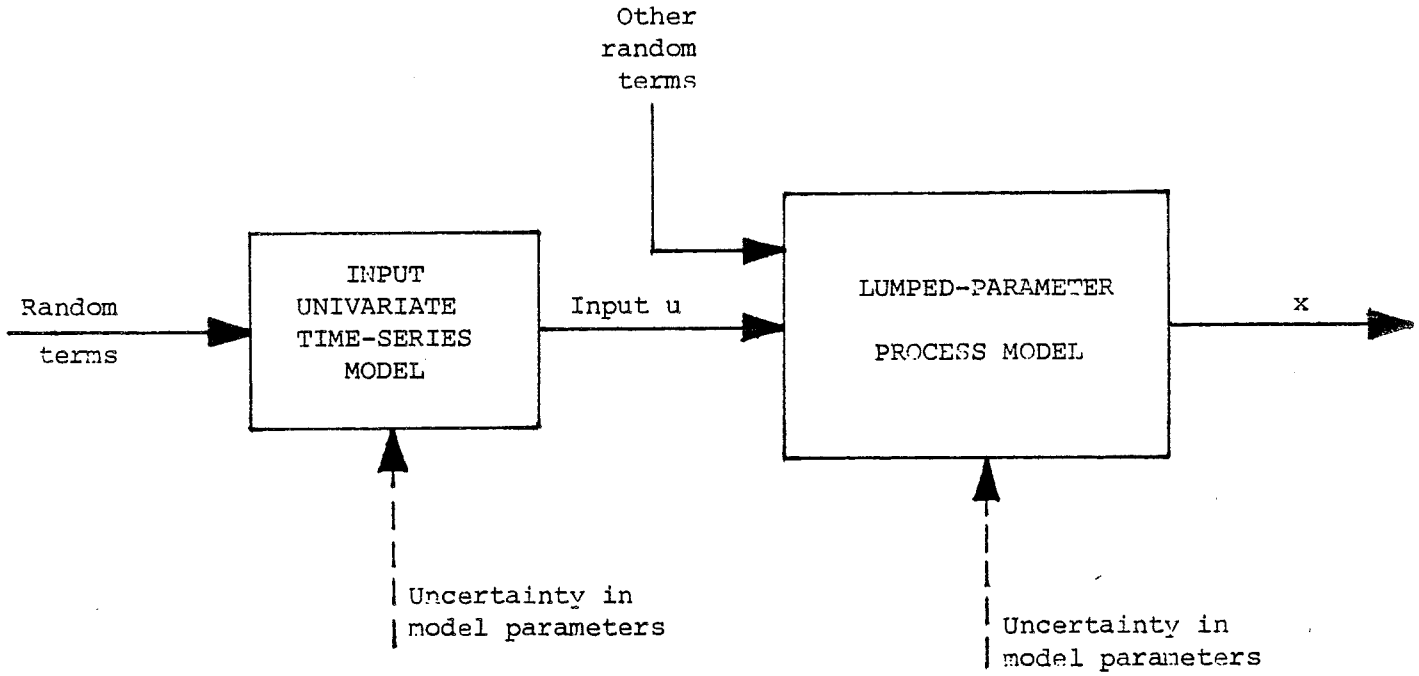


Figure 2.5 Sources of uncertainty and random terms for Monte Carlo simulation

CSTR elements in each of which the dissolved oxygen (DO) concentration is chosen to be a state variable, whereas in practice DO might be monitored at just one location along the channels, e.g. the tail end, equivalent to the fourth CSTR element alone. Alternatively, in a single-CSTR approximation the model might characterise the state of the biomass in terms of three variables, stored, active, and inert masses, with only the single corresponding observation of mixed liquor suspended solids (MLSS) concentration.

Uncertainty, errors, or noise may also be included in the definition of the model: as the unmeasured, random disturbances of Figure 2.4 (and Figure 2.1), which appear in equation (2.5a); and as the measurement errors associated with the output observations in equation (2.5b). Whether these terms need to be included in the development or application of the model will depend on factors such as the detail of the data, the accuracy of the model, the purpose of the model's application, and so forth. Should they be excluded, the model of equation (2.5) would simply quantify deterministic relationships among the inputs, states, and outputs.

In summary, the lumped-parameter model has been the primary vehicle for the development of chemical, biochemical, and microbiological hypotheses about the behaviour of the unit processes of wastewater treatment. This, together with its important role in control theory, determine the special emphasis that will be placed on the lumped-parameter model in later sections of the review. The natural progression from a linguistic model is towards a lumped-parameter model. In contrast to the input/output time-series model, the lumped-parameter model seeks explicitly to hypothesise why, and by what mechanisms, the inputs are related to the outputs. It does this by making reference to the internal state of the process: hence too its alternative designation as a <u>state-space model</u>.

(d) Distributed-parameter Model

As already stated, both spatial and temporal variability may be important to the behaviour of a process in practice. For a number of conditions explicit consideration of spatial variability might be necessary, typically:

(i) non-uniform mixing and/or scale-up problems in stirred tank reactors;

(ii) predominantly plug-flow reactor (PFR) designs, or tubular reactors;

(iii) multi-phase systems with liquid-solid interactions or gas-liquid interactions;

(iv) transport of substances in the micro-scale cell/particle environment.

Assuming for illustrative purposes that spatial non-uniformities prevail along just one dimension (z), the distributed-parameter model can be defined, as,

$$\dot{\underline{x}}(t,z) = \underline{f}\,[\underline{x},\ \underline{x}',\ \underline{x}'',\ \underline{u},\ \underline{\alpha};\ t,\ z] + \text{(random terms)} \qquad (2.6a)$$

$$\underline{y}(t_k,z_j) = \underline{h}\,[\underline{x},\ \underline{\alpha};\ t_k,\ z_j] + \text{(random terms)} \qquad (2.6b)$$

where now the dot notation denotes partial differentiation with respect to time ($\partial/\partial t$) and $\underline{x}'$ and $\underline{x}''$ denote respectively ($\partial\underline{x}/\partial z$) and ($\partial^2\underline{x}/\partial z^2$). The argument z_j in $\underline{y}(t_k,\ z_j)$ signifies that process observations would be made at discrete locations along the dimension z.

The obvious motivation for the development of a distributed-parameter model is that fluid and mass transport phenomena are crucial to the dynamic behaviour of a given process. In such

cases momentum and energy balances may form part of the model, with fluid velocity being a constituent element of the state vector. Should the spatial gradients $\underline{x}'$ and $\underline{x}''$ prove to be insignificant over parts (or the whole) of the reactor, it can be seen that the partial differential equations of equation (2.6a) collapse (in principle) down to the ordinary differential equations of the lumped-parameter model (equation (2.5a)).

(e) Considerations of Uncertainty

The most straightforward route to the construction of a mathematical model would not normally necessitate detailed considerations of the uncertainty associated with the data or the hypotheses and parameter values of the model. Such considerations tend to reflect a subject in a more mature stage of development, a stage perhaps only just reached in the dynamic modelling of wastewater treatment processes.

However, there has been much discussion of the topic with respect to river quality modelling, in particular the use of Monte Carlo simulation. While clearly it is not the only aspect of the analysis of uncertainty (see, for example, Beck, 1987), Monte Carlo simulation is undoubtedly a popular and useful method. It is, nevertheless, only a method, and not a type of model. It could be practised with any of the mathematical models defined above in equations (2.2), (2.3), (2.5), and (2.6), and will be briefly discussed below.

Considerations of uncertainty in wastewater treatment plant modelling are likely to be of greatest relevance in the application of models for the evaluation of process control strategies and alternative design configurations. In this there would be two primary sources of uncertainty:

(i) uncertainty in the parameter values ($\underline{\alpha}$) of the model; and
(ii) uncertainty in the sequences of input disturbances ($\underline{u}$) of the process.

Both could be accommodated within the context of Monte Carlo simulation. For example, combining equations (2.3b) and (2.5a) would give (in scalar form: see also Figure 2.5):

$$\dot{x}(t) = f[x,u,\alpha;t] + (\text{random terms}) \qquad (2.7a)$$

$$u(t_k) = \theta_1 u(t_{k-1}) + \theta_2 u(t_{k-2}) + \ldots + \theta_n u(t_{k-n}) + (\text{random terms}) \qquad (2.7b)$$

A comprehensive treatment of the uncertainty associated with equation (2.7) for a simulation period $t_0 - t_N$ would require the following:

(i) At t_0, a random choice of initial values for the input and state variables, $u(t_0)$, $x(t_0)$;

(ii) A random choice of parameter values θ_i for the input time-series model, and a random choice of α, the parameter in the process model, both to be held constant over $t_0 - t_N$.

(iii) Random sequences of numbers for the (random terms) in both (2.7a) and (2.7b) for the period $t_0 - t_N$.

For Monte Carlo simulation each such complete simulation would have to be repeated many times, and the distributions of random values so generated for $x(t)$, where $t_0 \leqslant t \leqslant t_N$, assessed by criteria appropriate to the application of the model.

2.3. Development and Evaluation

(a) Hypothesis Formulation

In our classification the term hypothesis formulation will be used to define collectively all the elements of model development that lead up to, but do not engage in, the analysis of any form of

empirical time-series observations. These are steps that indicate merely the formalisation of existing or preliminary theories in mathematical terms.

(b) System Identification

It is one thing to postulate the form of a hypothetical, mathematical model of a unit process, i.e. to assemble and compose all the hypotheses about the phenomena believed to govern process behaviour. It is quite another to demonstrate whether the model is a good or a poor approximation of reality.

This process of evaluating the model against experimental data is best described as system identification (the term used in control theory), although it is a concept that has several levels of interpretation. At its most superficial it may simply be understood as "curve-fitting". More generally it is understood as calibration of the model against the data, i.e. model parameter (coefficient) estimation. Yet more fundamentally it can be said to be about the use of models for the interpretation of field data, or indeed as part of the process of developing scientific theories about the behaviour of a system (knowledge acquisition; process learning).

Although it is possible to invent a detailed scheme for classifying the literature on system identification, for pragmatic reasons ours will be restricted simply to drawing a distinction between "semi-quantitative" and "quantitative" system identification as follows:

(i) Semi-quantitative. The data are few and sparse; the objective is to demonstrate that the simulated responses of the model are crudely in agreement with observed responses, i.e. changes at about the right time, in the correct direction (increase/decrease), and of the correct order of magnitude; and on balance the preference would be not to modify the structure of the model in the light of its performance against the data.

(ii) Quantitative. The data are sufficient to support more formal statistical measures of the match between the model and the data; and the need to modify the model structure in the light of its performance is clearly acknowledged, if indeed not actively pursued.

The distinction is inevitably somewhat arbitrary; this too, like the considerations of uncertainty, is a matter of the maturity of a subject.

Quantitative system identification may involve several steps, of which the following are the most important:

(a) Model structure identification. Formally, it is necessary to address the question of whether the functional relationships among the inputs, states, and outputs are correctly specified, i.e. the functions $\underline{f}$ and $\underline{h}$ in equation (2.5) are correctly specified, before being in a position to estimate accurately the unknown parameters appearing in those relationships.

(b) Parameter estimation. This can be conducted in various ways and with increasing sophistication: by familiar trial-and-error simulation without any formal criterion of goodness-of-fit; and by using the methods of optimisation, against such a criterion, either without or (less straightforwardly) with assumptions about the sources of uncertainty in the data and model.

(c) Verification. Ultimately it must be established whether the statistical properties of the residual errors of mismatch between the model and the data confirm an "acceptably good fit" of the model to the data. The criteria defining "acceptably good" are bound to be relative in nature, and the arguments for or against the model having been verified in this way, somewhat circular. Nevertheless, verification of

the model implies such statistical properties of the response errors as a mean value of zero, a low standard deviation, no temporal correlation structure, and so on.

(c) Validation

Confidence in a model grows with its (successful) application to an increasing variety of situations: to the same piece of plant over periods of experimental observation different from that for which the model was first identified; or to the same type of unit process, but at different field sites with differing capacities. Any one of these, or similar, tests of a model will be referred to as validation (other definitions of the term are possible).

(d) Assessment Data

Like validation, the form of the data against which a model has been assessed matters greatly to the credibility of that model. Four attributes of the assessment data are of special significance:

(i) The type of facility from which the data derive, i.e. laboratory replica, pilot-plant, or full-scale plant;

(ii) The sampling frequency of the data, i.e. continuous, on an hourly basis, daily, or weekly;

(iii) The duration of the observed record, where, because of the dominance of the diurnal cycle, the important differentiation is between a period of greater than 24h or otherwise;

(iv) The quality of the observations themselves, which will be a function of such factors as the method of sampling, the form of laboratory analysis, and instrument reliability.

2.4. Applications

According to the terms of reference stated in the introduction, the survey is to be conducted with a view to the operation of sewage treatment plants in the wider context of river quality management. This dictates to a large extent the logic of our scheme for classifying the applications of dynamic models of treatment processes.

In the broadest sense it is assumed that the objectives of studying wastewater treatment plant dynamics are to achieve a reliable, cost-effective operation, subject to probabilistic and possibly time-varying consent conditions for the plant effluent, and in the presence of hydrologic and biological variability across the river basin as a whole.

It is beyond the scope of this review to discuss either the desirability of, or the means of calculating, the various forms the effluent consent conditions could take. Since variations with time are the essential concern, we note that the form of the consent conditions could be one of the following: time-invariant; seasonally varying; variable on a monthly basis; or dependent upon ambient river conditions. Otherwise the consent conditions are assumed to be given.

(a) Target Operating Policies

Satisfying the given consent conditions for the effluent implies the specification of values for certain operating variables (their "target" set-points) about which the plant manager is free to decide, and over which he can exercise active control. Examples of such variables might be a desirable aerator DO profile, sludge age, degree of recycle, or recirculation schedule. The trajectories for these targets will reflect periods where the objectives for individual unit process performance remain constant, and periods where there should be deliberate change, for example, scheduling a change of desired DO profile to move from a non-nitrifying to a nitrifying mode of operation.

What is needed here is the development and evaluation of target operating policies, i.e. courses along which, all other things being equal, it would be desirable to steer the performance of the treatment plant throughout the year. More specifically, the term "course" should be interpreted as the set of desirable set-point values for the various operating (state) variables of all the individual unit processes. There may be more than one such set of values, i.e. more than one target operating policy, that is capable of allowing the plant to achieve its consented effluent conditions. What precise form the dynamic model might take in such an application is open to debate, but the distinctive features of the supporting analysis would be:

(i) The nature of the relationship between the given consent conditions (or river conditions) and the feasible range of target operating policies;

(ii) Operation in the face of normal disturbances and fluctuations;

(iii) A concern with the longer-term, i.e. days, weeks, months; and

(iv) If there is a choice among alternative target operating policies, then it would clearly be important to determine which policies appear to be more/less costly, adaptable, reliable, and so on.

(b) Contingencies

Essential to the definition of target policies is the use of the word "normal" in describing the disturbances to which any treatment plant or river basin is subject and which would tend to push performance away from its target. Certain disturbances, however, such as storm events, equipment failure, loss of solids from a secondary clarifier, or the accidental spillage of a toxic substance to the sewer network, are better described (and

analysed) as major, or even "abnormal" disturbances. (There is a semantic difficulty here as to whether rain events are considered to be contingencies, abnormal events, or simply non-dry-weather-flow conditions. The more important point is to draw a distinction between trying to meet target operating states and handling significant, short-term excursions away from those desirable states). For these contingencies, or set-piece events, the control objective is to return the system to its target policy in the minimum time with the minimum "damage" to the performance of the plant and the river basin.

The analysis of such problems cannot be divorced from that of the problem of target operating policies. Actions to handle a given contingency may differ among the target policies; damages quantified for an individual contingency need to be integrated into aggregate measures of the costs, reliability, and adaptability of the various target policies. The emphasis of the analysis would be different, however, most notably in its time-scale (minutes, hours, days) and the need to consider more detailed and specific sequences of control actions.

(c) Control System Design

River basin management has a natural hierarchical structure of control. So too does a wastewater treatment plant, and so too do individual unit processes. The orientation of the analysis of target operating policies and contingencies is towards the higher plant level. The orientation of what will here be called control system design is toward the lower unit process level. The focus of the problem is the relationship between the process control inputs and the satisfaction of a target operating policy for that process, i.e. satisfaction of the desired set-point trajectories. The associated analysis should not be concerned primarily with the relationship between the consent conditions (or river conditions) and the target operating policy.

The nature of the control system design problem is both extensive and varied. It covers the design of individual control loops, e.g. an aeration-DO loop, and the means of coordinating loops in a multi-loop system, e.g. DO set-point, recycle-ratio set-point, and sludge wastage rate coordination in the activated sludge process. It may exploit the principles of feedforward and feedback; it may be solved using the methods of optimisation (minimising control costs, the time to recovery, etc); and the controller can be made adaptive in its operation (both learning and controlling at the same time), to mention but a few possibilities.

For the present survey the control problem is important in providing the context, but it is not the central issue. The key point is the role played by the dynamic model, and in this respect it is crucial to distinguish between:

(i) the model as a tool for <u>off-line</u> <u>analysis</u> during research and development, i.e. in the analytical design of a controller, or as a simulation of the real plant for evaluation of alternative suggested control strategies;

(ii) the model installed at the treatment plant as a tool for the everyday, practical requirements of <u>on-line operations</u>.

Most applications of models have so far been of the former category. To reflect this balance, and to emphasise the distinction, we have introduced the following definition of on-line estimation/forecasting.

(d) <u>On-line Estimation/Forecasting</u>

The ability to control the dynamic behaviour of a process depends on the capacity to observe (in real-time) to what extent actual performance matches desired performance and the capacity to

implement corrective action in the event of a significant mismatch between actual and desired performance. The success of this function is therefore dependent on whether:

(i) the variables most appropriate to control are readily measurable in real-time; and

(ii) the degree to which the on-line measurements are corrupted by errors and uncertainty.

These problems are the concern of on-line estimation and forecasting.

Definition of these problems is illustrated in Figure 2.4. On the basis of a lumped-parameter model, two groups of state variables have been identified, those that are measurable ($\underline{x}_m$) and those that are not ($\underline{x}_u$), or at least not easily measurable. Fundamentally no variable (x_m) can be observed (as y) in the absence of error (as Figure 2.4 shows). It is possible to define the desired set-point trajectories for the process in terms of either $\underline{x}_m$ or $\underline{x}_u$, as $\underline{x}_m^d$ or $\underline{x}_u^d$, say. The essence of the observational part of the control problem is to be able to have estimates $\hat{\underline{x}}_m$ and $\hat{\underline{x}}_u$ of $\underline{x}_m$ and $\underline{x}_u$ (the true states of the process) in order to base control action on the deviations ($\hat{\underline{x}}_m - \underline{x}_m^d$) and ($\hat{\underline{x}}_u - \underline{x}_u^d$).

The requisite knowledge for the exercise of the control function (in real-time) derives from two sources: the observed information, generally comprising both the measured inputs ($\underline{u}$) and outputs ($\underline{y}$); and the model, which relates these known quantities to the set of unknown quantities $\underline{x}_m$, $\underline{x}_u$ (and indeed the model parameters, $\underline{\alpha}$) according to equation (2.5). If, as may predominantly be the case, the desired set-points are specified in terms of the measurable states alone, $\underline{x}_m$, and $\underline{y}$ is considered to be a sufficiently accurate estimate of $\underline{x}_m$, then the substitution $\hat{\underline{x}}_m = y$ can be made and there is no need of a model within the operational control system.

If either $\underline{y}$ is a highly uncertain measurement or $\underline{x}_u^d$ is to be used, then some form of on-line model may be necessary in order to generate the estimates $\hat{\underline{x}}_m$ and $\hat{\underline{x}}_u$. The degree of uncertainty associated with both the observations and the model will be a decisive factor in determining how to use the model for this purpose. Formally, the model could be used without any explicit considerations of uncertainty, in which case it is being used in an on-line (deterministic) simulation mode. Alternatively, it could be embedded within a Kalman filtering algorithm. Such algorithms are not easily appreciated, not least because they are usually discussed in the arcane mathematics of control theory. Stripped of their sophistication they reduce to the essentials of a model and a set of equations for handling the computations of uncertainty and error propagation. Further, qualitative discussion of the algorithms is given in Appendix 2.

Two extensions of this subject should be noted. First the general state estimation problem can be defined as providing the values of $\hat{\underline{x}}(\tau)$ at time t. In control terms, when $\tau = t$ this is called filtering; when $\tau > t$, forecasting; and when $\tau < t$, smoothing (or hind-casting). The framework of Kalman filtering accommodates all three problems. Second, it is also possible to consider the use of on-line parameter estimation (generating $\hat{\underline{\alpha}}$), a facility closely associated with problems of fault detection.

(e) Design-operation Interactions

The analysis of design-operation interactions could have several interpretations. Given the operational control setting of this review, two interpretations are of special relevance:

(i) An awareness of any implications of the reverse form of the logic used for this classification of model applications, namely that an understanding of contingency control may require a change of target operating policies, which in turn may necessitate

modified consent conditions, and ultimately changes of treatment plant designs.

(ii) The use of dynamic models in assessing alternative plant configurations, with special reference to fluid flows and storage, i.e. the arrangement, sizing, location, and connections among tanks, pipes, and pumps. The objectives of the analysis would include not only improved operational performance but also upgrading of plant to achieve tighter consent conditions or a greater throughput with the lowest additional investment cost.

(f) Training Simulators

This is, quite straightforwardly, a statement of the possibility of using complex dynamic models, with simulated measurement and management information systems, for the training of personnel.

3. THE REVIEW

3.1. Plant Influent

The characterisation and generation of data on plant influent variations have always posed a problem for the simulation of treatment plant performance. The usual procedure is to generate sinusoidal signals in phase in order to simulate the diurnal flow and quality of the influent, such as those characteristics shown in most textbooks (Figure 3.1). Clearly this approach over-simplifies the role the influent can play in determining process performance, especially when, as in many simulation studies, attention has been restricted to short periods, often no more than 24 hours.

Wastewater characteristics are known to vary continuously: daily, weekly and seasonal patterns have been identified. However, industrial discharges and urban runoff entering the sewer network, as well as excessive infiltration and snowmelt, may alter substantially these "typical" patterns. Disturbances resulting from hydraulic shocks (caused by rainfall-runoff entering the sewer network), and from industrial spills, can be quite disruptive for the operation of the treatment plant (Berthouex et al., 1986; Stoltenberg and Horvatin, 1987). They may also cause a subsequent overloading of the receiving water's self-purification capacity.

Even if it were possible to quantify and predict the regularities of industrial discharges (with the cooperation of the industries concerned), this would not be the case for urban runoff, and the resulting storm sewage (sanitary sewage plus urban runoff in combined sewers). The quality of urban runoff and storm sewage depend on so many site-specific variables (rainfall characteristics, type of urban catchment, sewer system, etc.), that they are not only difficult to model, but also any model developed for a specific area may not be applicable at other sites.

Average annual runoff and overflow loads and concentrations can be obtained using long-term records of influent data from treatment plants with combined sewer systems and simple (cumulative) mass balances

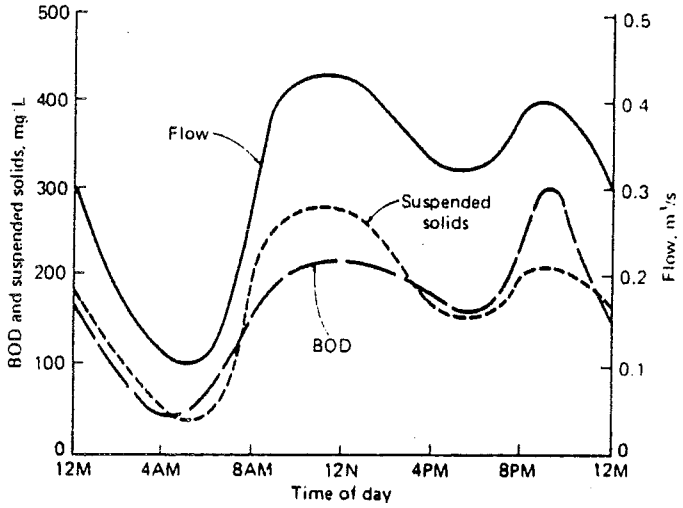


Figure 3.1 Typical variations of the plant influent flow and pollutant concentrations.

(Mueller and Di Toro, 1983). However, estimates of individual daily runoff and overflow concentrations have a high degree of variability due to limitations inherent in a simple mass balance. Long data-bases are required if good estimates of average loads are required.

Huber (1986) has published an extensive review of the literature on conceptual models (i.e. lumped-parameter models) for predicting urban runoff quality, and has concluded that, of the six "usual models" (SWMM, STORM, etc..) all are capable, without exception, of accurate simulation of hydrographs and "pollutographs", given adequate data for calibration. By the same token, none of the models can be assumed a priori to be able to predict accurately the absolute magnitudes of concentrations and loads without such data, although they may be very useful for comparative studies and the estimation of relative effects. Not surprisingly, there is a significantly better capability at present in the simulation of urban runoff quantity as opposed to quality.

A sewer quality model is currently under development at Hydraulics Research Limited in the UK (Moys, 1987), and is to be incorporated within an existing sewer flow quantity model (WASSP-SIM, National Water Council, 1981). The objective of the model is to simulate short-term quality variations of storm sewage overflows and, in conjunction with an appropriate river model, to provide an objective framework for the control of sewerage discharges while maintaining given, desired receiving water uses. The determinands of interest are oxygen demanding load, ammonia, suspended solids, hydrogen sulphide, dissolved oxygen and sediments, and the model is to be capable of simulating the following basic processes:

1) The build-up of pollutants on catchment surfaces;
2) The wash-off of accumulated surface pollutants during rainfall events;
3) Gulley pot performance;
4) Foul sewage in combined sewers;
5) Sedimentation in sewers; and
6) Transport of pollutants through the sewer system.

Finally, the model should also be capable of calibration and verification on the basis of flow survey and sewer inspection procedures employed in drainage area planning. The model is due to be completed by the end of 1988.

Alternatively, "statistical" models might be used to generate plant influent data. Here too, however, the reliability of the field data is of primary importance, i.e. the data must be collected for a rather long period (Hemain, 1986). Patry (1987) has recently compared two urban water quality models for the purposes of on-line forecasting in the real-time control of a combined sewer system, one model being an input-output time-series model, the other a lumped-parameter, conceptual model. The two models yielded similar results, although the time-series model was found to give distinct advantages from a practical operational point of view (subject to the caveat of this model being site-specific). Berthouex et al. (1975) had earlier established that univariate time-series models were capable of giving an adequate characterisation of hourly influent BOD variations. Similar time-series models have also been proposed for the on-line prediction of hourly plant influent flows during dry and wet weather periods. Beck (1977) identified an input/output relationship between wastewater treatment influent flow data and rainfall data (on an hourly basis) for a treatment plant in Stockholm.

In general, while flow records may be available for such detailed analysis, this is not usually the case for data on influent quality, which is generally scarce and available only on a daily-average basis. This restriction might be alleviated by the use of on-line instrumentation, although such instrumentation is neither widely used nor believed to be fully reliable. Much therefore remains to be done in terms of acquiring more experimental observations of the variability of plant influent quality at an hourly time-scale. The knowledge thus gained, together with the trend toward real-time operational control of sewer networks (e.g. Schilling, 1985), must enhance the practical potential of on-line influent forecasting as a component of improved treatment plant control.

3.2. Storm Retention Tank

The use of storm retention tanks can be expected to give some or all of the following advantages (Henderson et al., 1981):

1) The number and total volume of spillages is reduced due to the buffering effect of the storage volume;

2) Sewage passing through the tank will deposit part of its pollutant load, thus becoming more dilute and less damaging to the receiving river;

3) More time is available for the receiving river's discharge to have increased in response to the storm, thus providing greater dilution of any overflow from the retention tank (there is usually a time-lag betwen the storm responses of the river and the corresponding urban sewer system).

Even though storm tanks are widely used in Europe, and always mentioned as one of the most immediate means of solving problems of urban runoff and combined sewer overflows, few dynamic models have been proposed to simulate the behaviour of the retention tank on an event basis. Studies have tended to examine the long-term performance of stormwater detention basins in preliminary assessments of urban-runoff impacts, and for screening treatment alternatives. Probabilistic methods for manipulating empirical or assumed distributions for the characteristics of rainfall-runoff have been used for such purposes (e.g. Di Toro & Small, 1979; Adams and Zukovs, 1987); so too have methods based on long-term simulation of rainfall-runoff, storage/treatment processes (Goforth et al., 1983). The former precludes determination of pollutant removal efficiency, while the latter can include pollutant removal in the form of the following equation:

$$R = R_{max}(1 - e^{-kT}) \qquad (3.1)$$

where: R = pollutant removal efficiency

R_{max} = maximum removal efficiency

k = first order removal rate

T = treatment time (in fact, equivalent to the time-step of the numerical solution procedure).

Amandes and Bedient (1980) have proposed a simple lumped-parameter model to assess the attenuation of flood peaks and the removal of suspended solids by sedimentation that can be achieved by small-scale detention basins. The change in reservoir volume was described by the unsteady-state continuity equation, and sediment removal was computed from a knowledge of the particle-size distribution. For each time step, a critical sedimentation velocity is defined by the quotient of basin depth and instantaneous detention time. Particles with a settling velocity greater than this critical value are assumed to be completely removed from the water column, while those with a lower velocity are removed proportionately according to the ratio of their velocity to the critical velocity. No evaluation of this sedimentation model has been conducted with reference to any field measurements.

Medina et al. (1981a) have also derived models describing the transient response of storage/treatment systems to highly variable forcing functions of flow and concentration. Their work refers to completely mixed systems of both constant and variable volumes, i.e. lumped-parameter representations, and to one-dimensional advective-dispersive systems, i.e. distributed-parameter models. The models have been tested against data collected from a tank in Milwaukee (Medina et al., 1981b), as a result of which it was concluded that the well mixed constant-volume CSTR representation was able to describe closely the observed performance of the basin.

Ferrara and Smith (1982), have suggested a simple dynamic model for the design of a dual-purpose detention basin, i.e. a basin to be used for flood protection and SS removal. The model comprised two ordinary differential equations, one for volume and one for concentration variation, as given below. These authors used a fully mixed basin and

the solids removal rate was a function of settling velocity, stressing the importance of the value assumed for this parameter and the need for further research on this specific subject. Their lumped-parameter model can be sumarised as:

$$\frac{dV_T}{dt} = Q_i - Q_e \qquad (3.2)$$

$$\frac{d(V_T C)}{dt} = Q_i C_i - Q_e C - R \qquad (3.3)$$

where: Q_i = flow in ; Q_e = flow out

C_i = influent concentration

C = concentration in the basin and effluent suspended solids

V_T = volume of water in basin

R = $A\ C\ v_s$

A = surface area

v_s = settling velocity

In other words the model has two state variables, volume, and suspended solids (SS) concentration.

Huber (1986) has briefly described the possible ways of modelling detention facilities for urban runoff. He states that the more challenging problem is usually to simulate the removal processes that may occur through physical means, or through biological or chemical actions. He has concluded that: "The critical information in any such modelling exercise is the treatability of the influent to the detention facility. Pollutants that are in particulate form or adsorbed onto the solids are relatively easy to remove by physical treatment, such as sedimentation (distribution of settling velocities is essential). Dissolved pollutants are typically much more difficult to remove in a high flow setting such as from storm and combined sewage".

In our opinion the essential problem in modelling the behaviour of an off-line retention tank, however, is that this behaviour could change from one physical state to another in a matter of hours. There are four main mechanisms involved in storm retention:

(i) Fill: This is when the tank is filling during the rain event;

(ii) Dynamic sedimentation: When full and overflowing to the receiving body, the behaviour of a storm tank is probably similar to that of a primary clarifier, hence the designation of dynamic sedimentation (sludge is also generated during this step);

(iii) Quiescent settling: Quiescent settling occurs when the tank is partly or completely full and no flow is entering or leaving the basin (again sludge is also produced during this step);

(iv) Draw: This final step corresponds to the pumping of the storm tank contents from the basin back to the plant stream.

Each of these mechanisms could occur when the tank is in a different physical state, i.e. empty, full or partly full. The occurrence of these states depends on such factors as the rainfall, dry-weather plant flow and/or operational decisions. The possible combinations of these states and mechanisms are shown in Figure 3.2.

The storm retention process could thus be modelled by a four-component procedure together with logical statements defining the appropriate components at any given time. However, there are some problems associated with the implementation of this scheme, and they are briefly addressed below.

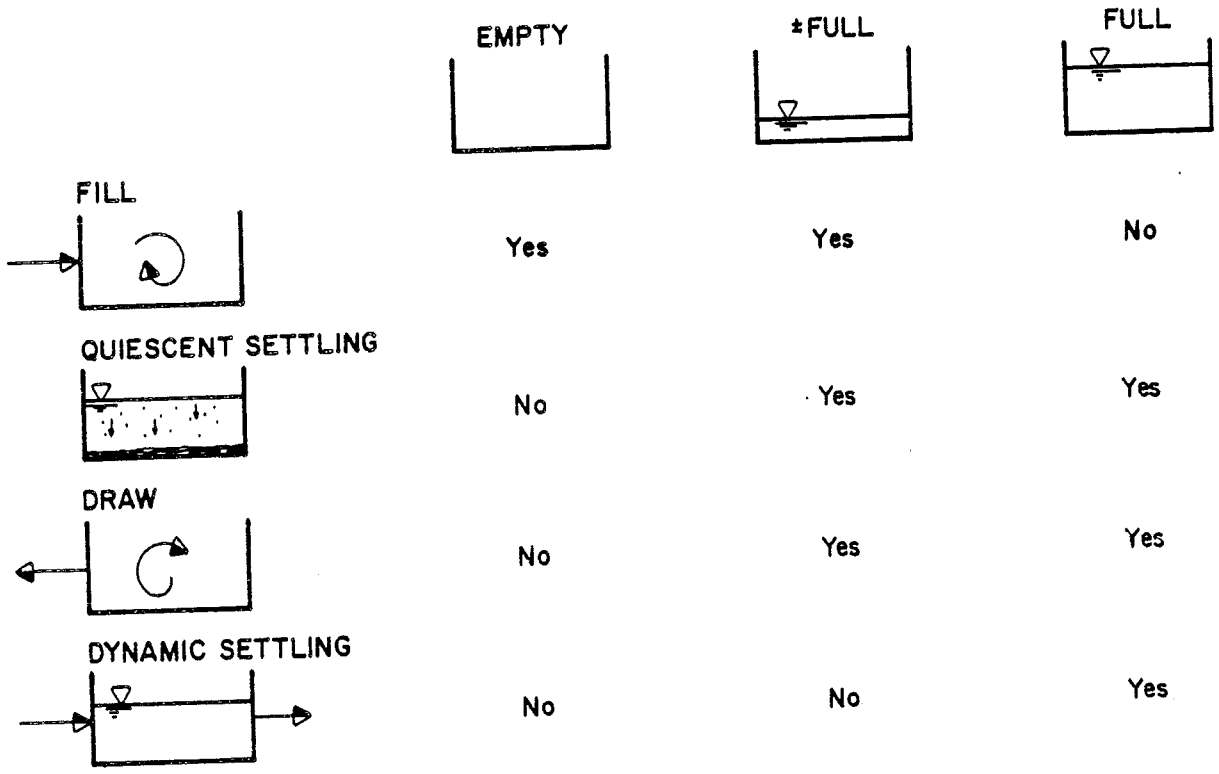


Figure 3.2 Physical states and mechanisms for storm retention tank modelling.

(i) Fill

It can be assumed that during a filling period, the contents of the tank will be completely mixed. This assumption may not be entirely correct (it depends on the basin size, the inlet location, etc.), but can be considered a good first approximation. Accordingly, simple ordinary differential equations can be used to describe volume and concentration variations within the tank.

$$\frac{dV_T}{dt} = Q_i \; ; \; V(0) = V_0 \qquad (3.4)$$

$$\frac{d(V_T C)}{dt} = Q_i C_i \; ; \; C(0) = C_0 \qquad (3.5)$$

(ii) Dynamic sedimentation

A sedimentation model similar to that proposed by Lessard and Beck (1987) for primary clarification (see below) could be used for simulation of this component of the storm tank model. In this case it would be necessary to determine an average settling velocity parameter and the fraction of (each) polluting material considered to be settleable. While the latter might be estimated using simple field tests or published data in the literature, estimation of the former still remains a difficult problem.

(iii) Quiescent settling

Here a model based on discrete settling theory (Stokes' Law) might be postulated. However, it would necessitate knowledge of some form of particle-diameter distribution, together with corresponding settling velocities, and as with the primary sedimentation model, the rate of settling would

be difficult to determine. Alternatively, particle removal could be represented as in the empirical relationships already presented in the literature (e.g., White and Allos, 1976), or extrapolated from urban-runoff settling data or street-solids data (e.g., Whipple and Hunter, 1981; Randall et al., 1982; Klemetson, 1985).

There may be other complications besides those of specifying particle settling properties. Sludges will accumulate in the tank, and should the holding time be too long, hydrolysis of organic nitrogen may lead to the release of ammonia from the sludge. Further, depending on the physico-chemical state of the tank contents, "leaching" of metals from the sediment and changes in the solids settling characteristics may occur. However, these mechanisms are likely to be of relatively minor importance.

(iv) Draw

Modelling the process of pumping back the tank's stored volume should itself be straightforward: one simple equation for volume variation could be used (as in equation (3.4)), assuming that no concentration changes occur during the period of drawing. However, this pre-supposes answers to the following two questions.

(1) What is the pollutant concentration in the tank?

The problem of estimating the pollutant content of the tank is not only a function of quiescent settling (as discussed in point (ii)), but also a function of the sludge handling arrangements.

There exist two possible ways of removing the accumulated sludge generated during the periods of dynamic or quiescent

sedimentation; (a) by collecting and transferring the sludges directly to sludge treatment, and (b) by resuspending the accumulated sludge (usually with a scraper) just prior to pumping the tank contents back to the plant stream. If the latter strategy is adopted, the concentration of pollutants in the pumped stream will result from the complete mixing of the liquid and the sludges. However, vertical stratification of the pollutant concentration could occur since a bottom scraper does not necessarily achieve efficient, complete mixing. It may merely scour the sediments from the bottom, just sufficient for the pump to evacuate them. The vertical concentration profile is difficult to visualise, as too would be the resulting quality of the returned flow. If, on the other hand, removal and transfer of the sludge (strategy (a)) is adopted, the pollutant concentration in the tank just prior to the draw period will be that following quiescent settling. Thus, depending on the choice of operating strategies it may not be necessary to simulate behaviour under quiescent settling conditions, i.e. simplifications are possible.

From our studies of primary clarification, it seems appropriate to simulate the hydraulic behaviour of the storm tank during dynamic sedimentation by a sequence of several CSTR elements. This, however, also complicates the issue of specifying the pollutant concentrations in the return flow at the beginning of the draw period, there being a horizontal distribution of the pollutants along the tank in this case. Again, simplifying assumptions can be made, especially if the horizontal gradient of concentration is small.

(2) <u>When should the tank contents be pumped back to the plant stream</u>?

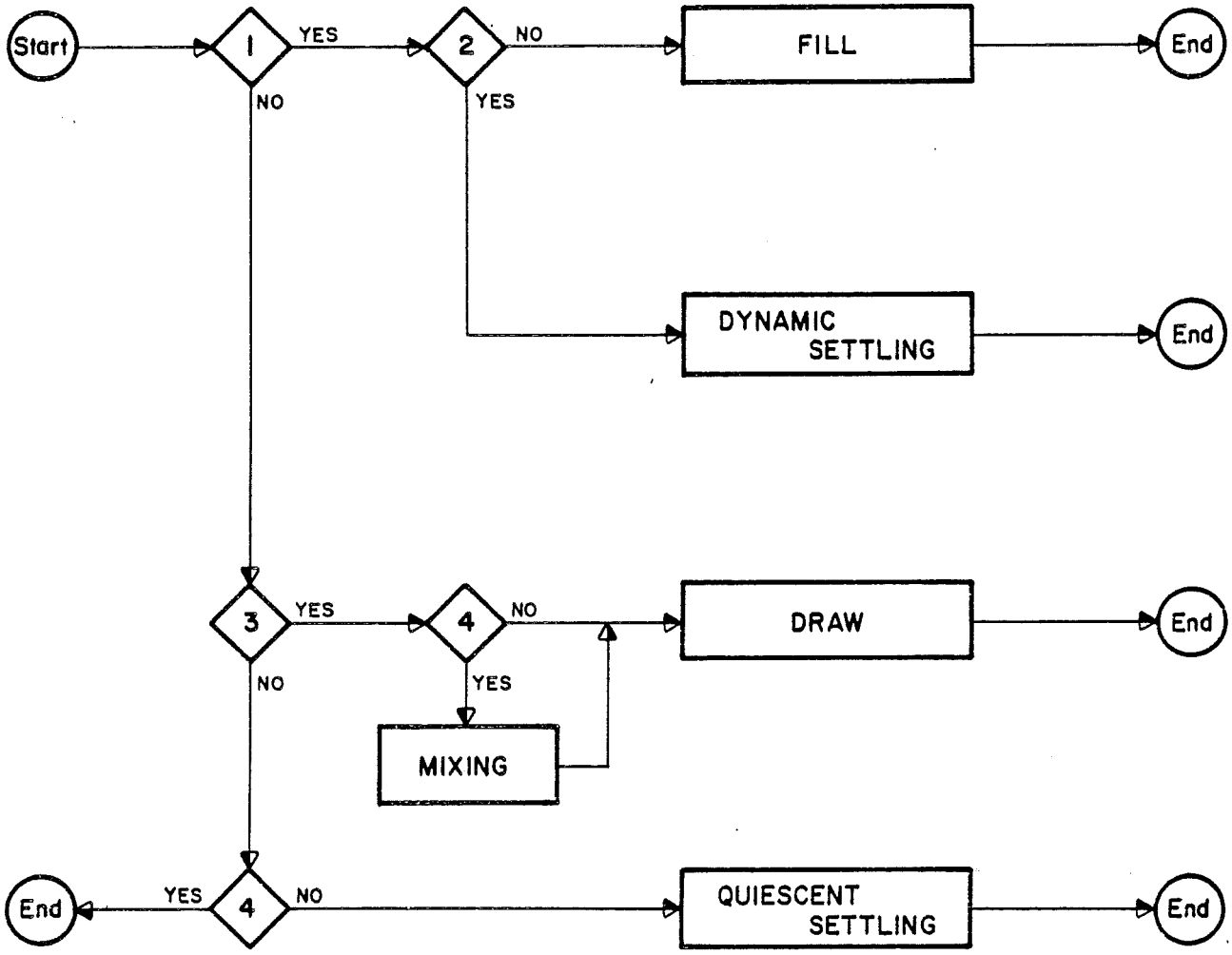
It is well known that control strategies that generate flow transients should be avoided (e.g. Chapman, 1985). The storm tank contents ought therefore to be pumped back during

periods of low flow and at as even a rate as possible. Nevertheless, field practice for the pumping of storm sewage returns may be quite different, since control is often dependent on the availability of a plant operator. Storm sewage might therefore have to be pumped back during peak-flow hours (when the operator is on site), instead of during the night (at low flows) when there would be less risk of impairing normal treatment. The model's use of an automatic control action for pumping returns, based on the influent flow conditions, may not necessarily be valid.

Following these comments our preliminary proposals for a conceptual model for the storm retention tank is shown in Figure 3.3.

It is also feasible for storm tanks, located either in the sewer system or in the treatment plant, to be used as equalization basins. Such an arrangement could prove useful for an overloaded plant, being able to store excess volume during peak hours for subsequent release during low-flow conditions. Literature exists on the benefits to plant performance of equalization tanks (e.g. Andrews et al., 1976; Khudenko, 1985). Simple lumped-parameter models for the simulation of flow and concentration, similar to equations (3.4) and (3.5), have been proposed by Dold et al. (1985) and Therien et al. (1986), among others. Should the storm tank be used as an in-line equalization basin, dynamic modelling of the relevant process mechanisms should not be a problem. However, most probably the tank will be used in a sideline manner, and therefore, if it is accordingly simulated as an off-line equalization basin, problems with the modelling of quiescent settling could arise as indicated earlier.

To summarise, the use of storm tanks has benefits both for the receiving river (preventing and partially treating overflows during rain events) and for the plant (avoiding plant overloading during rain events). Dynamic models of the storm tanks would be useful for operational river basin management in predicting the volumes and quality of overflows during periods of precipitation. Moreover,



1- IS STORM TANK INFLUENT FLOW GREATER THAN 0,0 ?

2- IS TANK FULL ?

3- IS PUMPING BACK POSSIBLE ?

4- IS SLUDGE MIXED WITH TANK CONTENT BEFORE BEING DRAWN OFF ?

Figure 3.3 Proposed conceptual model of the storm retention tank.

dynamic models would be helpful in establishing and optimizing the pumping schedules of storm sewage returns to the influent stream, especially if there are many storm tanks in the sewer network.

Unfortunately, dynamic modelling of the process is still in its infancy, and there is an urgent need for data collection on storm sewage characteristics and storm tank behaviour both during and after the rain event. Quiescent settling of storm sewage and pollutant behaviour in the settled sludge are areas in need of basic research.

3.3. Primary Sedimentation

Primary clarifiers are used in wastewater treatment for the purposes of settling and removing suspended particulate matter in the influent waste stream, which could:

(i) cause operational problems in the activated sludge plant;

(ii) induce a higher energy consumption arising from partial oxidation of settleable solids, as in oxidation ditch systems (von der Emde, 1972);

(iii) reduce the removal rate of soluble substrate in the aerator (Kuo et al., 1972);

(iv) increase the volume of waste activated sludge (Voshel & Sak, 1968).

The performance of settling tanks has usually been assessed from a steady-state point of view. Although performance may vary from one day to the next, it is generally assumed that about 50-60% of suspended solids (SS) are removed, and some 20-30% of the organic matter (BOD, COD). Thus little attention has been given to the behaviour of primary clarifiers in dynamic terms, either as a matter for experimental study or with reference to the development of mathematical models.

To some extent this state of affairs can be explained by the fact that there are limited opportunities for the operational control of clarifiers, the schedule of desludging being the only obvious possibility. However, primary clarification is an important part of a treatment plant, and the efficiency of its operation is closely linked with the efficient operation of the aerator and with the performance of sludge treatment. Clearly the location of the primary clarifier is such that it affords a unique opportunity for the detection of conditions -- either normal or abnormal -- that will inflence the performance of the other unit processes (Pierce, 1960). Since the influent to a primary clarification unit is highly variable in its quantity, quality and settling characteristics, there ought to be considerable interest in the response of the process to these dynamic perturbations. For example, the influent waste is usually described as crude sewage (sanitary and industrial discharges) deriving from the sewer network, is frequently changed to storm sewage (crude sewage with urban runoff), and can from time to time contain recycled flows from within the plant, such as supernatant sludge liquor returns or waste activated sludge (Figure 3.4).

Broadly four types of model have been proposed for the description of primary clarifier behaviour.

(i) Simple, steady-state relationships derived from statistical analysis of field data, quantifying removal efficiencies, and used largely for the purposes of design (e.g. Tebbutt & Christoulas, 1975; Anderson, 1981; Tay, 1982). These models usually correlate removal efficiency with the overflow rate and/or the influent SS concentration, and have found some application in simulation studies (e.g. Tanthapanichakoon & Himmelbau, 1981; Lech et al., 1978).

(ii) Relatively simple, lumped-parameter, dynamic models, expressing component mass balances across a constant volume, continuously stirred tank reactor (CSTR) approximation of the clarifier (Alarie et al., 1980; Shiba et al., 1979).

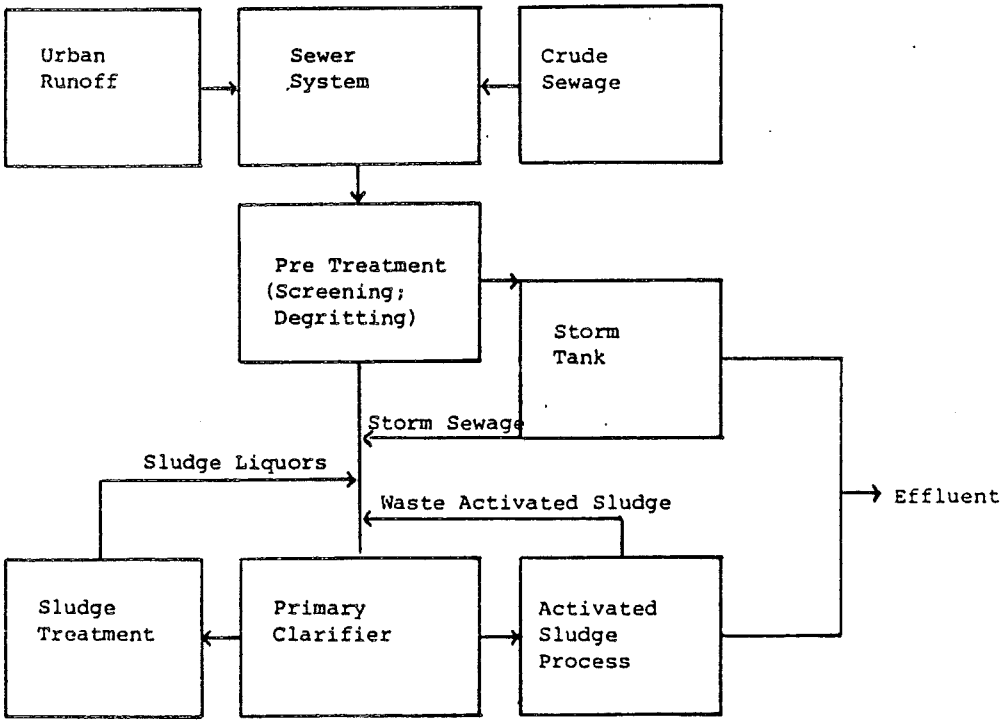


Figure 3.4 Schematic diagram of typical plant configuration.

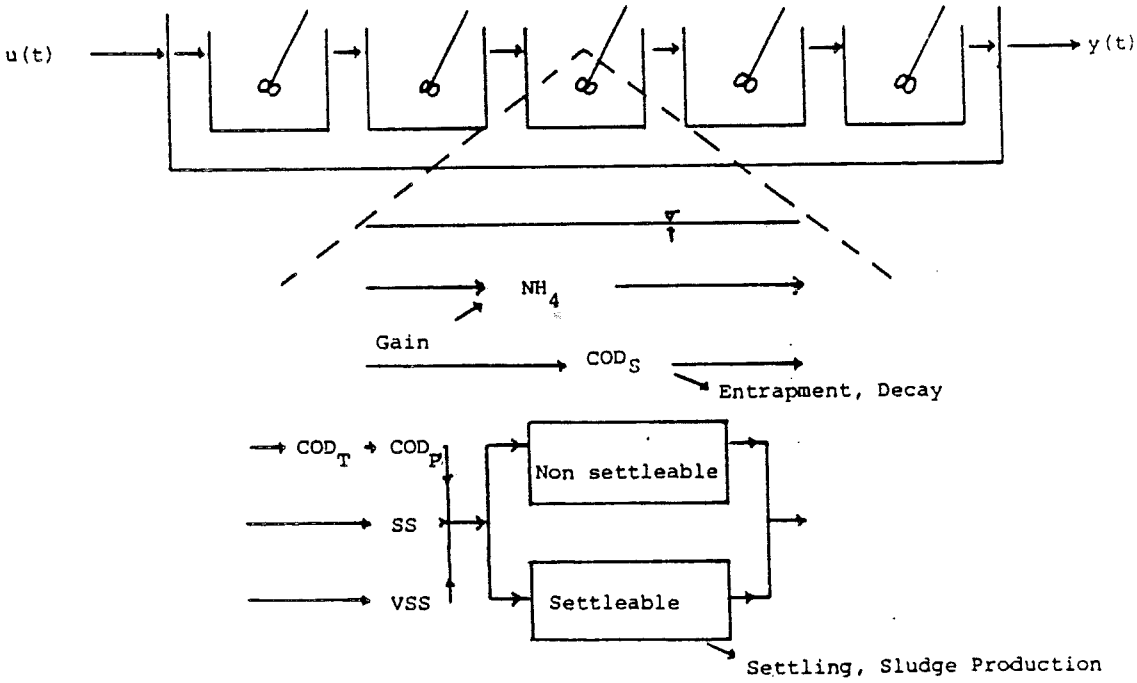


Figure 3.5 Primary clarifier model (from Lessard and Beck, 1987).

(iii) More complex distributed-parameter models, based on the conventional assumptions for advection and dispersion of material passing through the clarifier. These too appear to have been used mostly for design purposes, with special reference to the description of the fluid velocity field and solids concentration within the clarifier (e.g. Schramber & Larock, 1983; Tay & Henke, 1983; Abdel-Gawad & McCorquodale, 1985).

(iv) Crowther et al. (1977) have evaluted the use of univariate and input-output time-series models for the clarifier effluent BOD (using data with sampling intervals of between 20 and 60 minutes). Their later work has explored the use of similar models for the effluent SS characteristics (Dalrymple and Crowther, 1983).

A widespread belief regarding the dynamic simulation of primary sedimentation processes is that they are too complex to be modelled due to the inherent characteristics of a highly variable influent, the distributions of particle-sizes and corresponding settling velocities, flow and density currents in the tank, scouring phenomena, temperature effects, and so on. Simple lumped-parameter models have been criticised on the grounds of not representing the process behaviour satisfactorily (e.g. Barror & Schroeder, 1980), while the distributed-parameter models have been criticized for their excessive complexity (e.g. Hudgins & Silveston, 1984).

There is no doubt that the fundamental nature of particle settling can be considered extremely complex, and possibly even essentially probabilistic, as in the theoretical studies of Iordache and Corbu (1986) on the effects of particle interaction during settling. However, as pointed out by Alarie et al (1982) in response to criticisms of their paper: if the prediction capabilities of a relatively simple mechanistic model in application to real world problems is fairly good, why should the model be made more complicated just for the sake of sophistication?

More recently, Lessard and Beck (1987) have presented a lumped-parameter dynamic model that is somewhat more complex than those presented earlier for the primary clarifier. Their model consists of eight ordinary differential equations, and contains the following improvements over the models of Shiba et al (1979) and Alarie et al (1980):

(i) Eight state variables (settleable and non-settleable SS, settleable and non-settleable volatile SS (VSS), settleable and non-settleable total COD (CODT), soluble COD (CODS) and ammonium-N) are simulated, whereas previously only SS had been included in most models.

(ii) Division of the non-conservative constituents (SS, VSS and CODT) into two fractions, settleable and non-settleable, has been found to be especially useful in the case of settlers with long retention times.

(iii) Use of a variable, average particle settling velocity, depending on the type of influent feeding the settler (crude sewage; storm sewage; crude sewage with return sludge liquors).

(iv) Evaluation of the model against field data has shown that assumed conservative substances (CODS and ammonium-N) can not always be treated as such; loss of CODS and gain in ammonium-N were observed. A term for CODS removal by a first-order kinetic mechanism, and a constant rate term for generation of ammonium-N, were incorporated in the model.

A conceptual picture of the Lessard and Beck model is shown in Figure 3.5, and the relevant equations are given in Table 3.1. Illustrative results for the evaluation of the model against field data are shown in Figure 3.6. Tests of the model were also made with reference to data published in the literature (as presented by Alarie et al, 1980). In general the model performance is quite satisfactory, although (as with all models) it could be improved. In comparison with other such

Table 3.1 State equations for the primary clarifier model of Lessard and Beck (1987)

$$\dot{x}_1(t) = (Q(t)/V)[a_{11}u_1(t)-x_1(t)] - [(1-a_2)a_{31}Sx_1(t)/V]$$

$$\dot{x}_2(t) = (Q(t)/V)[(1-a_{11})u_1(t)-x_2(t)]$$

$$\dot{x}_3(t) = (Q(t)/V)[a_{12}u_2(t)-x_3(t)] - [(1-a_2)a_{32}Sx_3(t)/V]$$

$$\dot{x}_4(t) = (Q(t)/V)[(1-a_{12})u_2(t) - x_4(t)]$$

$$\dot{x}_5(t) = (Q(t)/V)[a_{13}(u_3(t)-u_4(t)) - x_5(t)]$$

$$- [(1-a_2)a_{33}Sx_5(t)/V]$$

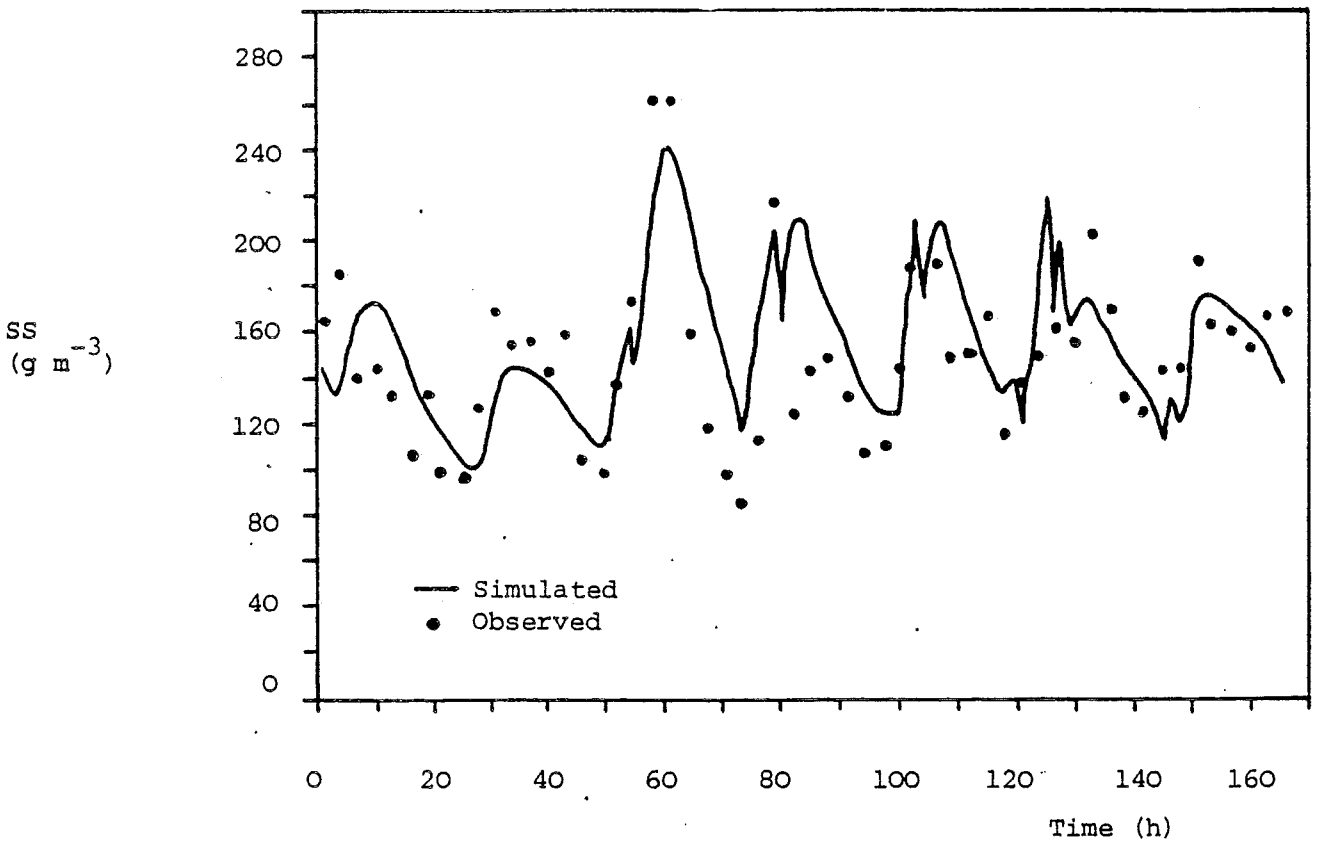
$$\dot{x}_6(t) = (Q(t)/V)[(1-a_{13})(u_3(t)-u_4(t)) - x_6(t)]$$

$$\dot{x}_7(t) = (Q(t)/V)[u_4(t)-x_7(t)] - a_4x_7(t)$$

$$\dot{x}_8(t) = (Q(t)/V)[u_5(t)-x_8(t)] + a_5$$

where

a_1	Settleable fraction
a_2	Scouring parameter (dimensionless)
a_3	Particle settling velocity (mh^{-1})
a_4	First-order decay-rate constant for COD_S(h^{-1})
a_5	Rate constant for generation of ammonium-N (gm^{-3} h^{-1})
a_6	Specific gravity of sludge
a_7	Sludge moisture fraction
C_i	Influent pollutant concentration (gm^{-3})
C_e	Effluent pollutant concentration (gm^{-3})
Q	Settled sewage flow-rate (m_3h^{-1})
S	Cross-sectional area of the clarifier (m^2)
t	Time (h)
u_1	Influent SS concentration (gm^{-3})
u_2	Influent VSS concentration (gm^{-3})
u_3	Influent COD_T concentration (gm^{-3})
u_4	Influent COD_S concentration (gm^{-3})
u_5	Influent ammonium-N concentration (gm^{-3})
V	Volume of sewage in CSTR element (m^3)
V_s	Volume of sludge produced daily (m^3)
x_1	Settleable SS concentration (gm^{-3})
x_2	Non-settleable SS concentration (gm^{-3})
x_3	Settleable VSS concentration (gm^{-3})
x_4	Non-settleable VSS concentration (gm^{-3})
x_5	Settleable, particulate COD concentration (gm^{-3})
x_6	Non-settleable, particulate COD concentration (gm^{-3})
x_7	COD_S concentration (gm^{-3})
x_8	Ammonium-N concentration (gm^{-3})



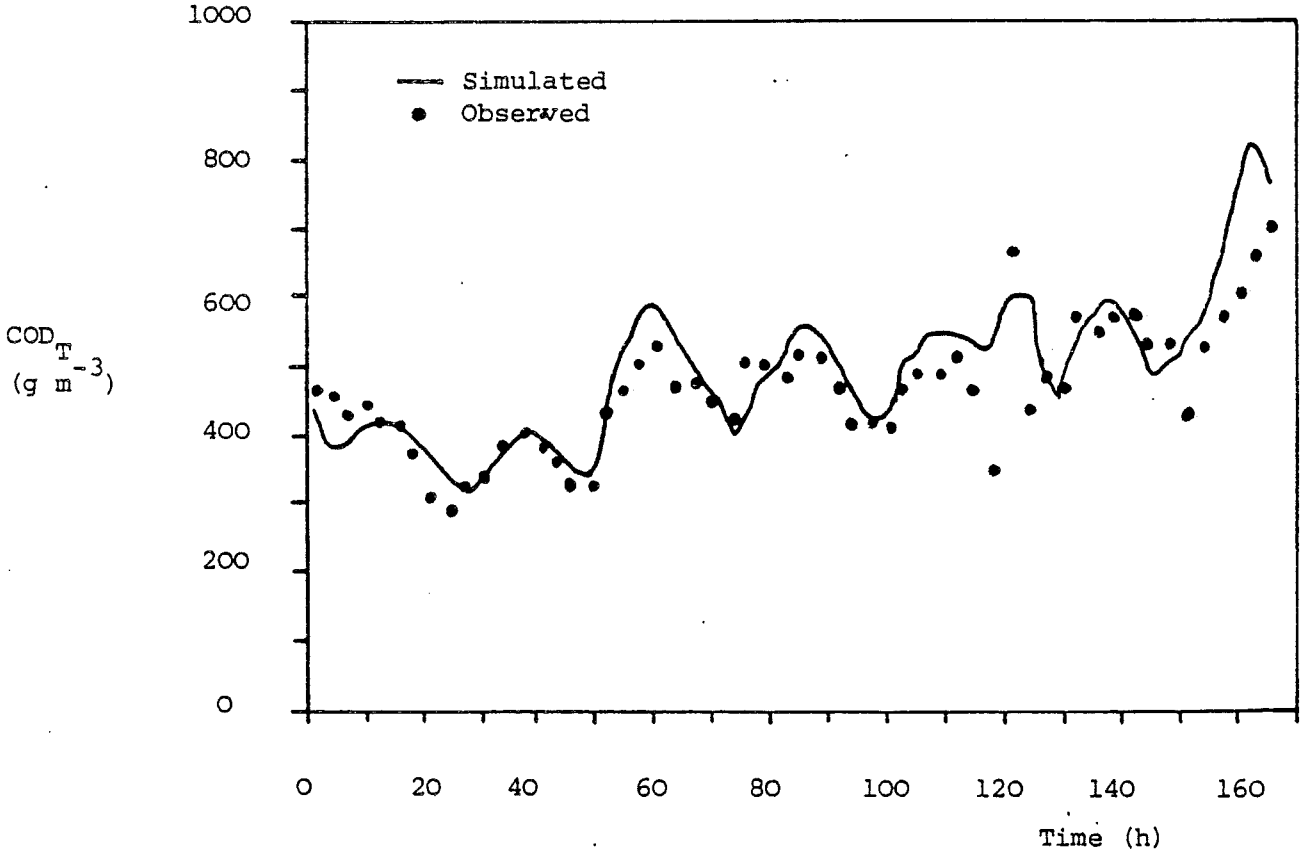


Figure 3.6 Comparison of observed and simulated primary clarifier effluent SS and COD_T concentrations.

models, the extent to which the model is grounded in suitably planned experimental work (at the Norwich Sewage Works) is one of its salient features.

The experimental studies at Norwich are important for other reasons. In addition to the state variables noted above, it is possible that other determinands, such as the concentrations of phosphorus or certain heavy metals, could be of considerable interest in the overall context of operational river quality management. Preliminary results for the evaluation of a lumped-parameter model for the removal of heavy metals in the primary clarifier at Norwich are most encouraging (Hong, 1987). They indicate that the model of Lessard and Beck (1987) can quantify the essential characteristics of metal removal in association with particle settling, even to the extent of suggesting variations among the mean particle sizes on which the different metals might be adsorbed.

The mechanisms of sludge thickening in the primary clarifier (the volume and moisture content of the sludge) have hardly, if ever, been discussed in the literature, despite their importance for the subsequent sludge treatment processes. Torre and Stephanopoulos (1986) have in fact elaborated control strategies for the anaerobic digestion process based on the control and prediction of the primary clarifier underflow. The estimation of sludge volume based on a mass balance equation and using a suitable parameter for the sludge moisture content is certainly possible as evidenced by the study of Lessard and Beck (1987). However, sludge moisture ought to be a characteristic simulated by the model, not "guessed" as a parameter, and more research is needed in this area.

Field data on primary clarifiers (of more than one observation per day) are rare, and further modelling work will depend crucially on the collection of a good data set, especially on sewage settling velocities. However, one could argue that since no significant control options exist for the primary clarifier, little further effort should be invested in the development of more detailed lumped-parameter models, and that research should be concentrated on the development of

input/output time-series models merely to service the simulation requirements of subsequent treatment processes, the activated sludge process in particular. (Managing the desludging schedule is the only control possibility for primary clarifiers. However, there is a tendency to operate the desludging facility of the settler on a continuous basis to avoid operational problems and to save on operating personnel. The resulting diluted sludge is thus thickened in a thickener prior to sludge treatment). Yet the primary clarifier could play an important role in the more global context of river basin management, especially with respect to the control of responses to rain events. More storm flows could be given primary treatment and in part discharged to the river. Moreover, as primary clarifiers are subject to many different dynamic inputs, for example, storm sewage and in-plant recycle, a good model of the process could be central in elaborating strategies for pumping this recycle or for returning the storm tank contents, without disturbing unduly the quality of the settler effluent, which represents the input to secondary treatment and should vary as little as possible. Indeed, a suitable combination of influent and primary clarifier models might enable real-time control of the step-feed activated sludge process during rain events. The use of these models, in an on-line forecasting mode, could provide estimates of flow-rate and concentration at the effluent of the primary clarifier, upon which decisions about suitable feed (and recycle) locations in the aerator basin might be based.

3.4. Secondary Biological Treatment: Suspended Growth Processes

Settleable matter having in theory been removed by primary treatment, it remains for the soluble organic matter to be eliminated by secondary biological treatment. Biological processes can be divided into two broad categories depending on whether the organic matter removal mechanism is by a suspended or an attached biomass. Suspended growth processes are generally referred to as activated sludge and its modifications such as oxidation ditches and contact-stabilisation processes, for example. Fixed-film, or attached-growth, processes include primarily trickling and high-rate filters and biological contactors.

3.4.1. Suspended Growth Processes: Basic Theory

The activated sludge process forms the heart of many wastewater treatment plants. The process consists of three units, these being the biological reactor, solids-liquid separator and air supply system. There are strong interactions between the three units and all must function together for the process as a whole to operate satisfactorily. A vast volume of research has been conducted on the dynamics of the activated sludge unit, dominated mostly by work on the biological reactor (the aeration basin).

In the last 20 years, attention has been focussed on the modelling of carbonaceous oxidation, nitrification-denitrification mechanisms, and the dissolved oxygen profile, although more recently biological phosphorus removal, sludge bulking and heavy metals removal have also started to attract attention. A very wide range of models, for both design and operational purposes, has been proposed for the mechanisms of carbonaceous oxidation and nitrification-denitrification, culminating, as it were, in the publication of the IAWPRC Task Group model (Henze et al., 1986). This latter can be said to represent the state-of-the-art in modelling of the activated sludge process, and it will accordingly be discussed below after a brief literature review setting the IAWPRC model in a historical context.

It is noteworthy to say that most of the research on the dynamic modelling of activated sludge processes has been undertaken by three research groups: one led by Andrews (Houston, USA), one by Marais (Capetown, South Africa), and one by Olsson (Lund, Sweden). While Olsson's group has tackled more or less exclusively the control aspects, Marais's group has examined in more detail the mechanisms involved in the process. Andrews and his co-workers have dealt with both aspects, combining both theoretical and practical considerations.

(a) Carbonaceous oxidation

Many workers have developed simple models of carbonaceous oxidation for design purposes (i.e. steady-state analysis); such models rely on two basic equations, one for substrate utilisation and one for micro-organism growth (Gaudy and Kincannon, 1977). These models describe substrate uptake according to Monod kinetics, and are based on the hypothesis that all substrate is soluble and readily available for oxidation. Steady-state models may be sufficient for the purpose of designing a plant, but they are quite inadequate for its control and operation.

The first major contribution to the dynamic modelling of carbonaceous oxidation is due to Busby and Andrews (1975) who introduced the concept of stored mass. They described the mechanisms by which the carbonaceous oxygen-demanding substrates are entrapped and metabolised by the biological floc, using a series of reactions including storing, assimilation (from stored mass) and decay (endogeneous respiration). The biomass had to be represented by three components (state variables): stored, active and inert mass (Figure 3.7). The mechanism of storage was added to simulate the lag phase in the biological growth of microorganisms. Rapid removal of pollutants from wastewater occurs by a transfer of substrate from the liquid to the floc phase and is a function of several mechanisms including adsorption, absorption and physical entrapment. Although the removal is extremely rapid in the contact tank, the organic material may not be oxidized to any appreciable extent in this reactor. Once the substrate (particulate and soluble), is incorporated in the biological floc, it is described as stored mass.

A second improvement in the understanding of the dynamics of activated sludge was the introduction of the bi-substrate hypothesis (Dold et al., 1980; Clifft and Andrews, 1981).

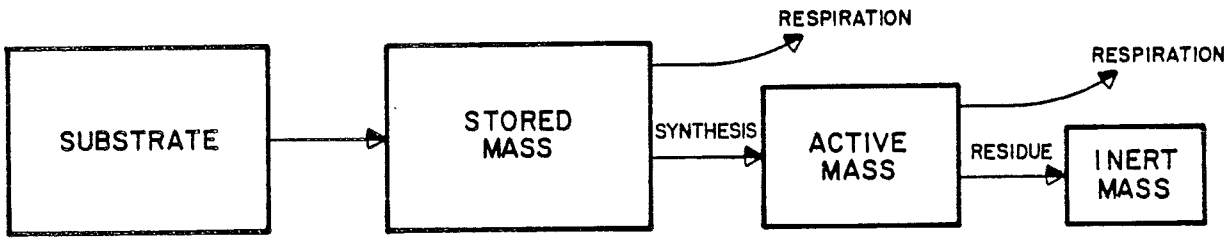


Figure 3.7 Activated sludge model (from Busby and Andrews, 1975).

> "Storage products are defined as all materials that accumulate in the floc as the result of substrate being transported from the liquid to the floc phase more rapidly than it is metabolized. This is assumed to apply to both particulate and soluble substrates. Stored particulate matter is considered to be extracellular whereas storage products of soluble substrates may be either extracellular or intracellular. Because some soluble compounds normally found in municipal wastewater are not stored within the floc prior to metabolism, the model considers that a portion of the soluble substrate is directly metabolized by the active mass". (Clifft and Andrews, 1981).

The bi-substrate hypothesis thus refers to the division of the biodegradable organic substrate into two parts; one that has to pass through storage before assimilation, and one readily (directly) available to the microorganism (see Figure 3.8). The former is generally referred to as particulate and the latter as soluble. Discussions on whether the stored substrate is directly assimilated by the microorganisms, or has to be first hydrolysed to readily available substrate before assimilation, have been presented by Ekama & Marais (1986b).

The use (and promotion) of COD as a measure of the concentration of organic material in wastewater can also be seen as an improvement in the development of these models. COD is undoubtedly a superior measure when compared with BOD or TOC because it alone provides a link between electron equivalents in the organic substrate, the biomass and the oxygen utilized (see, for example, Marais & Ekama, 1976). If the COD is used as a measure of the organic content, it is assumed that a fraction of the total COD (about 20%) is non-biodegradable.

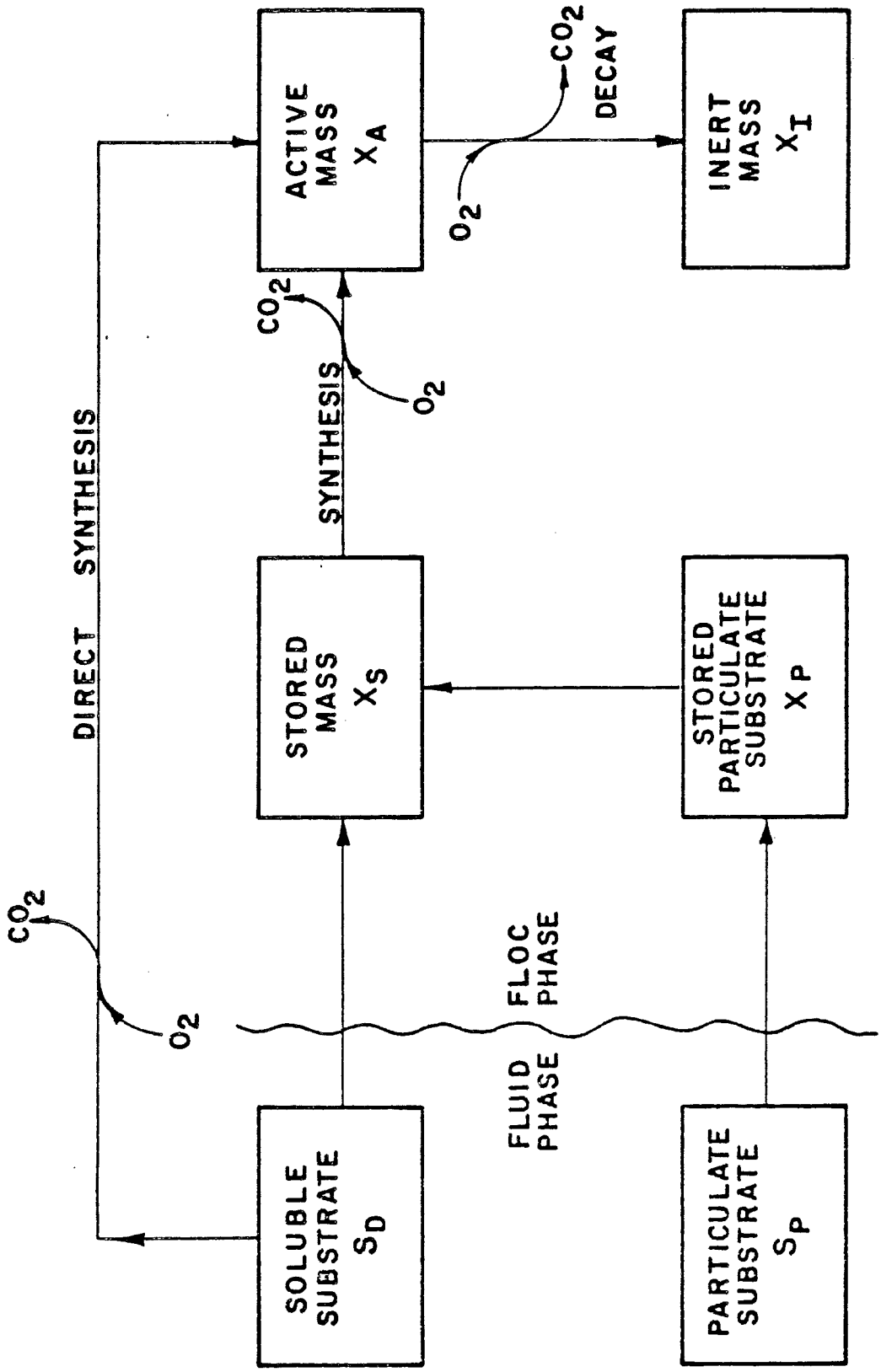


Figure 3.8 Bi-substrate activated sludge model (from Clifft and Andrews, 1981).

> "Endogenous respiration has been attributed to an energy requirement for organism maintenance, where a fraction of the organism mass disappears to provide energy for maintenance of the mass remaining. In order to make the energy available an electron acceptor must be present; in the activated sludge process, either oxygen or nitrate can serve as the electron acceptor. A major problem in modelling the single sludge nitrification-denitrification process was encountered in situations where both the oxygen and nitrate concentrations in a reactor are zero ... When no oxygen or nitrate is present, maintenance energy cannot be released. What happens to the organism under these circumstances? (Dold et al., 1980).

In response to their own question, Dold et al. (1980) proposed a death-generation model to resolve the inconsistencies regarding endogenous respiration in both aerobic and anoxic conditions, and also in situations where aerobically generated organisms are placed in an anaerobic state for a short period. It is hypothesized that the percentage of live mass that disappears in fact dies, lyses biodegradable substrate back into the liquid, and leaves the balance as non-biodegradable endogenous residue. The lysed biodegradable substrate is added to the biodegradable COD in the liquid and passes through the same phases of adsorption, storage and storage abstraction for synthesis. In effect, this approach implies that endogenous respiration per se does not exist and that oxygen is consumed only for synthesis of new cell material from the lysed energy.

(b) <u>Nitrification-denitrification</u>

Because of toxicity problems and undesirable increased oxygen demands in the receiving river, nitrification is often a requirement of wastewater treatment. Several models of the nitrification process have been proposed (e.g. Poduska and

Andrews, 1975; Gujer, 1977; Ekama and Marais, 1979) and have proved to be more "robust" than the models of carbonaceous oxidation, due mainly to the specificity of the biomasses and their substrates, and the possibility of reliable measurement of these substrates (Beck, 1986).

One of the first comprehensive dynamic models of the nitrification process was that presented by Poduska and Andrews (1975). The assumption of Monod kinetics for the rate of growth of the nitrifying bacteria was found to be quite adequate, and five state variables were used, ammonia, nitrite, nitrate, Nitrobacter and Nitrosomonas (see Figure 3.9). Their model was first evaluated for a laboratory activated sludge process, and later by Beck (1981) with reference to daily field data from the Norwich Sewage Works. It was apparent from the latter that the model was limited in its assumption of bacterial growth-rates independent of the DO concentration.

Ekama & Marais (1979) found it necessary to incorporate the biodegradation of organic nitrogen to obtain an adequate description of the nitrification process (in municipal wastewater, approximately 20% of the total nitrogen is in the organic nitrogen form). Their model considered the following concepts:

(1) A fraction of the total Kjeldahl nitrogen (TKN) concentration (probably proteinaceous nitrogen) is not directly available for nitrification, but must first be converted to free and saline ammonia before it can be nitrified. This conversion to ammonia appears to be a relatively slow process.

(2) The conversion from ammonia to nitrate appears to take place at a rate so much faster than that given by theory based on the Monod function for nitrification that, for the purposes of modelling, the conversion can be taken

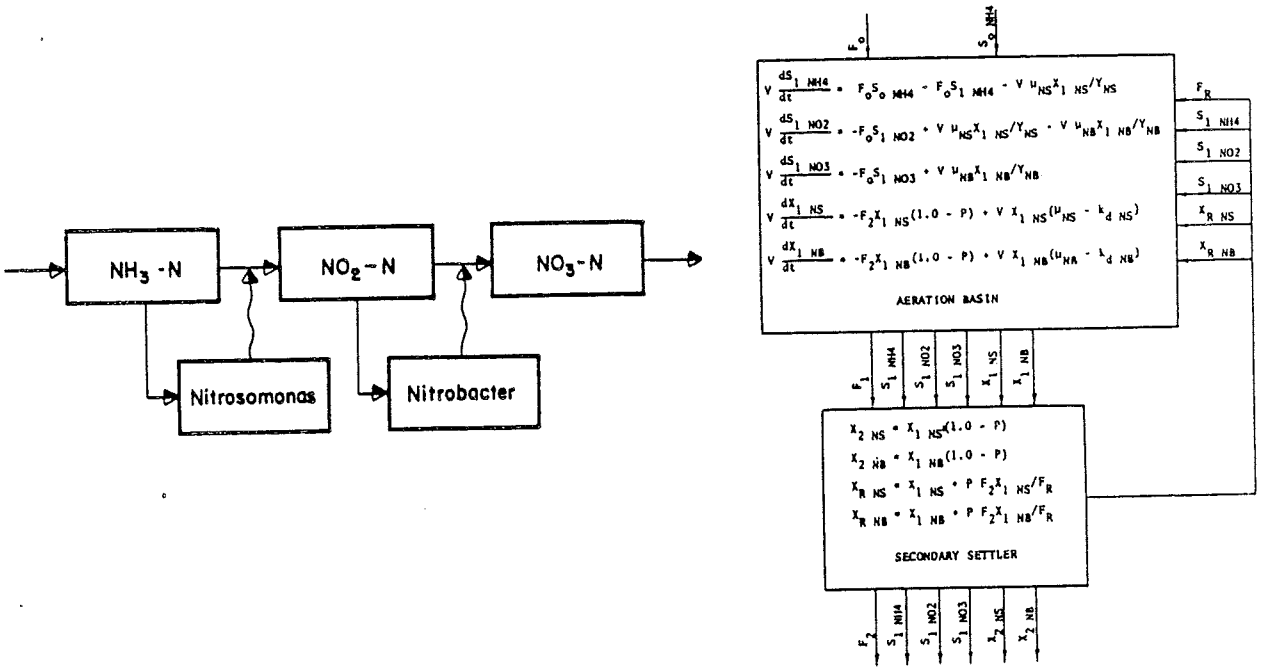


Figure 3.9 Nitrification model (from Poduska and Andrews, 1975).

as instantaneous.

(3) With the storage of COD on the organism, a certain mass of nitrogen is also stored.

In addition to these refinements it is important to note that nitrification is influenced by environmental factors, such as temperature, dissolved oxygen concentration and pH, which affect the growth-rate of nitrifiers. Dissolved oxygen is probably the most important such factor and the least understood, its effect generally being incorporated in dynamic models via a double Monod equation or some form of switching function (for example, Stenstrom and Poduska, 1980).

The production of nitrate in a wastewater treatment plant can also be considered undesirable; at the same time there are energy savings to be gained by the use of oxygen deriving from the nitrate source. Denitrification is therefore practised in a number of wastewater treatment plants. Four conditions are necessary for denitrification:

(i) presence of nitrate;
(ii) absence of dissolved oxygen;
(iii) bacterial mass that can accept nitrate and oxygen as electron acceptors;
(iv) presence of a suitable electron donor (energy source).

There are two basic configurations for the single sludge nitrification-denitrification system: (i) pre-denitrification where the anoxic reactor is placed before the aerobic tank, and (ii) post-denitrification, where the contents of the aerobic reactor discharge to an anoxic vessel.

Van Haandel et al. (1981) extended the model of Dold et al. (1980), which is similar in respect of nitrification to the one presented by Ekama & Marais (1979), to provide a reliable description of the kinetics of the nitrification-denitrification single-sludge process. The extension required no changes in the basic equation describing the aerobic process, merely a change in the value of one of the parameters in the expression defining the substrate utilization rate for slowly biodegradable particulate material when an anoxic environment is required. Nitrification is affected only in so far as growth of the nitrifiers takes place in a strictly aerobic environment, whereas death occurs in both aerobic and anoxic environments.

(c) Dissolved Oxygen Profile

In most wastewater treatment plants the concentration of dissolved oxygen (DO) can routinely be monitored on a continuous basis at one or more locations in the aerator basins of activated sludge units. The dynamic behaviour of the DO level, or of its spatial profile along an aeration channel, is intimately related to the many facets of aerobic waste degradation, either reflecting or determining the asociated reaction kinetics. This is obvious from the preceding discussions of carbonaceous oxidation and nitrification-denitrification, and will also be apparent in the later discussion of bulking sludge problems. From the point of view of modelling the DO profile dynamics, the predominant interests have been not so much fundamental knowledge but rather control system design and on-line estimation/forecasting, in association with the aeration-DO control loop. A suitably low value for the DO set-point ensures a healthy environment for the biomass, and the control problem is then clearly one of avoiding damaging under-aeration and costly over-aeration. It is perhaps significant to note that the IAWPRC Task Group report, which

is concerned so much with basic understanding, reveals little new thinking on the behaviour of the DO profile.

An important contribution to the modelling of DO dynamics has been made by Stenstrom (1975), who, for the first time, brought together descriptions of the various kinetic expressions for the consumption of oxygen in an activated sludge unit. This model, also incorporated into an earlier model of the Norwich Sewage Works (Beck, 1984), accounts for the following six factors:

(i) aeration input;
(ii) mass transport of DO into and out of the aerator in the influent, recycle, and effluent streams;
(iii) consumption of DO in the synthesis of active sludge mass from stored products;
(iv) consumption of DO during decay of active mass;
(v) consumption of DO by the growth of Nitrosomonas;
(vi) consumption of DO by the growth of Nitrobacter.

The kinetic expressions for items (iii) - (vi) are simply proportional to the appropriate growth or decay terms used for describing the biomass dynamics of carbonaceous oxidation and nitrification. In the IAWPRC model a term for (iv) does not appear since (as we have said) the inconsistencies regarding endogenous respiration are resolved by other means (Dold et al., 1980). In a subsequent development of Stenstrom's work, Clifft and Andrews (1981) have added a further term, augmenting item (iii) to account for the DO consumed in the synthesis of active mass directly from extracellular substrate (see also Fig. 3.8).

Stenstrom's lumped-parameter model does not address the issue of the spatial variability of the DO profile, knowledge of whose variations (with time) can be crucial for control purposes, as argued by Olsson and Andrews (1978). Their work incorporates the above source and sink terms in a

distributed-parameter model for the aerator, although their results refer simply to steady-state DO profiles for PFR and advective-dispersive reactor conditions.

Other than these two contributions there does not appear to have been substantial progress in quantifying the basic theory of DO behaviour, except for Clifft and Andrews' (1986) studies of the closed-reactor, oxygen activated sludge process. For this particular form of the process it is necessary to consider both liquid and gaseous phases, gas transfer between the phases, and equilibrium changes in the carbon dioxide-bicarbonate buffering system. Yet this too has not added significantly to the basic understanding of DO dynamics in suspended growth processes. In general the tendency has been to assume that the changes in DO concentration are the net result of almost every other reaction taking place in the aerator, and therefore it is these other reactions that have been the subjects of primary research activities. Neither is it surprising that, for the same reason, the DO dynamics have become the best studied subject of on-line estimation and control applications. Many of these applications have recently been reviewed elsewhere (Beck, 1986), and it suffices here merely to note the following:

(i) For control system design purposes it is not necessary to have a detailed knowledge of the mechanisms governing process behaviour; knowledge in the form of an input-output time-series model relating variations in aeration rate to variations in DO level is quite adequate (both for control and estimation).

(ii) The estimation problem has generally been formulated in terms of a DO balance that is a function of two factors, the oxygen transfer rate

(OTR) and the oxygen uptake rate (OUR), the latter being the aggregated sum of all the oxygen-consuming reactions in the aerator. The problem is then one of estimating on-line the mass transfer rate coefficient of the OTR and of inferring from the estimated OUR such quantities as the activity of the biomass, the state of the biomass, its concentration, the concentration of the substrate, and so on. Quite clearly this kind of inference depends upon a model (as discussed in Section 2 and Appendix 2), and the more information that is to be inferred, so the more detailed and reliable that model must be.

(iii) The degree of interest in on-line estimation using the DO dynamics can be measured by the quite sophisticated forms of estimation algorithms that have been proposed for its realisation (for example, Howell and Sodipo, 1985; Holmberg and Olsson, 1985).

Such popular interest, and the obvious practical successes of on-line DO control, might suggest that there are few problems still to be resolved. This is not the case. Our experience with field data from Norwich -- a situation admittedly complicated by the presence of a supplementary Vitox injection system -- is that while broadly correct performance of the model may be obtained in terms of carbonaceous oxidation and nitrification, the model may fail to characterise DO variations adequately.

3.4.2. The Activated Sludge Process: The IAWPRC Task Group Model

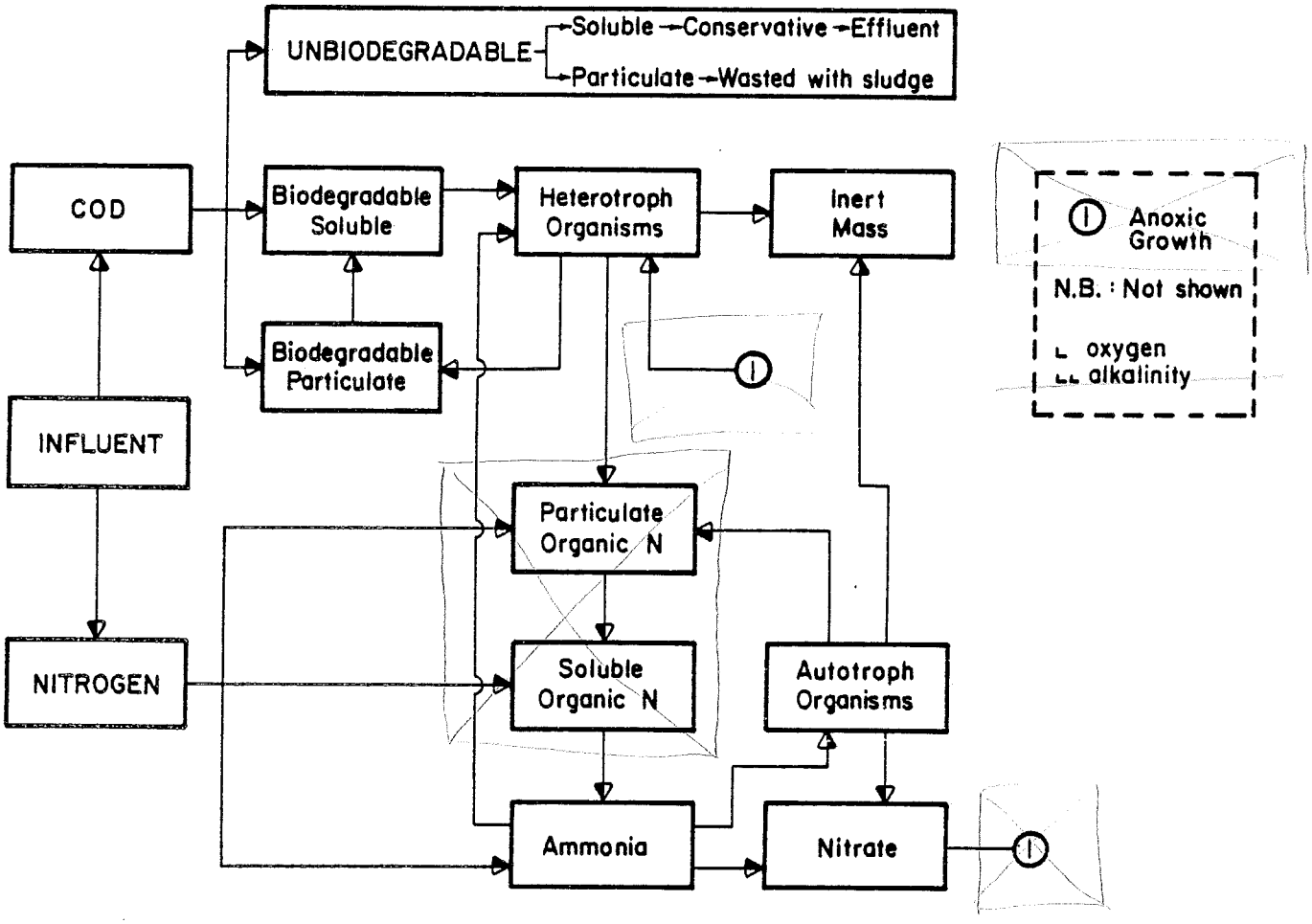
Many lumped-parameter models based on the preceding basic theory can be found in the literature (e.g. Perdrieux & Therien, 1980;

Holmberg, 1981; Tanthapanichakoon and Himmelbau, 1981). At the WRc the work of Jones (1978) appears to have developed along similar lines, although differing in the inclusion of an additional reaction term for substrate utilization by the active biomass (with Michaelis-Menten kinetics). This term accounts for the removal of substrate in cell metabolism but without consequent cell growth, and the biomass is differentiated between viable and non-viable bacteria. Evaluation of the model against pilot plant data has revealed broad success, but with apparent difficulties in simulating returned activated sludge solids concentrations (Jones and Paskins, 1980).

However, in 1983 the IAWPRC formed a task group to promote the development, and to facilitate the application, of practical models for the design and operation of biological wastewater treatment systems. The goal was first to review existing models, and second to reach a consensus concerning the simplest model structure having the capability of realistic predictions of the performance of single-sludge systems carrying out carbonaceous oxidation, nitrification and denitrification. The group's report has been completed and was published in 1986 (Henze et al, 1986). The result is the conceptual model of Figure 3.10. In addition to our brief review, further details can be found in the source reference. Thus, to summarise:

(i) Chemical oxygen demand is used as a measure of the concentration of organic material in wastewater.

(ii) The organic matter can be subdivided conceptually into two important components: biodegradable and non-biodegradable fractions. Non-biodegradable organic matter is biologically inert and passes through an activated sludge system unchanged in form. Two fractions of this component are identified, a soluble (S_I) and a particulate (X_I) fraction. Biodegradable organic matter may also be divided into two fractions, a



Part omitted by Beckermann in Normal model.

Figure 3.10 Schematic representation of the IAWPRC Task Group model of the activated sludge process.

readily biodegradable (S_S) and slowly biodegradable (X_S) form. For the purposes of modelling, the readily available material is treated as soluble, and the slowly degradable as particulate. The division of the degradable substrate into two forms provides a built-in lag in the uptake of the electron acceptor, which allows space-time dependent variations in oxygen and nitrate utilization to be simulated.

(iii) The readily biodegradable material consists of relatively simple molecules that may be taken up directly by heterotrophic bacteria and used for growth of new biomass.

(iv) The slowly biodegradable material, consisting of relatively complex molecules, must be acted upon extracellularly and converted into readily biodegradable substrate (by hydrolysis) before it can be used. The specific rate of hydrolysis of slowly biodegradable substrate is usually considerably lower than the specific rate of utilization of readily biodegradable substrate.

(v) Heterotrophic biomass is generated by growth on the readily biodegradable substrate under either aerobic or anoxic conditions, but is assumed to cease under anaerobic conditions.

(vi) Biomass is lost by decay, which incorporates a large number of mechanisms, including endogenous metabolism, death, predation and lysis. Decay is assumed to result in the conversion of biomass into slowly biodegradable substrate and particulate products, which are inert to further biological attack.

(vii) Nitrogenous matter in wastewater is divided into two categories: non-biodegradable and biodegradable, each with further subdivisions. With respect to the non-biodegradable fraction, the particulate portion is that associated with the non-biodegradable particulate COD; the soluble fraction is usually negligibly small and is not incorporated into the model. The biodegradable nitrogenous matter may be subdivided into: 'ammonia' (both the free compound and its salts), S_{NH}, soluble organic nitrogen, S_{ND}, and particulate organic nitrogen, X_{ND}.

(viii) Particulate organic nitrogen is hydrolysed to soluble organic nitrogen in parallel with hydrolysis of the slowly biodegradable organic matter.

(ix) The soluble organic nitrogen is acted on by heterotrophic bacteria and converted to ammonia-nitrogen.

(x) The ammonia-nitrogen serves as the nitrogen supply for synthesis of the heterotrophic biomass and as the energy supply for the growth of autotrophic nitrifying bacteria. For simplicity, the autotrophic conversion of ammonia to nitrate is considered to be a single step process which requires oxygen. The nitrate formed may serve as the terminal electron acceptor for heterotrophic bacteria under anoxic conditions, yielding nitrogen gas.

The kinetics, source, and sink terms of the model, which has 13 state variables, are shown in matrix form in Table 3.2 (see Henze et al. (1986) for further interpretation of the matrix). The model contains 19 parameters, five of which are stoichiometric, the others being kinetic parameters. They are listed in Table 3.3 with typical values proposed by the Task Group. Methods of determination for these parameters are given in Henze et al.

Table 3.2 Matrix form of the source and sink terms in the IAWPRC Task Group model of the activated sludge process (from Henze et al, 1986)

Component → i / j Process ↓	1 S_I	2 S_S	3 X_I	4 X_S	5 $X_{B,H}$	6 $X_{B,A}$	7 X_P	8 S_O	9 S_{NO}	10 S_{NH}	11 S_{ND}	12 X_{ND}	13 S_{ALK}	Process Rate, ρ_j $[ML^{-3}T^{-1}]$
1 Aerobic growth of heterotrophs		$-\frac{1}{Y_H}$			1			$-\frac{1-Y_H}{Y_H}$		$-i_{XB}$			$-\frac{i_{XB}}{14}$	$\hat{\mu}_H\left(\frac{S_S}{K_S+S_S}\right)\left(\frac{S_O}{K_{O,H}+S_O}\right)X_{B,H}$
2 Anoxic growth of heterotrophs		$-\frac{1}{Y_H}$			1				$-\frac{1-Y_H}{2.86Y_H}$	$-i_{XB}$			$\frac{1-Y_H}{14\cdot 2.86Y_H}-i_{XB}/14$	$\hat{\mu}_H\left(\frac{S_S}{K_S+S_S}\right)\left(\frac{K_{O,H}}{K_{O,H}+S_O}\right)\times\left(\frac{S_{NO}}{K_{NO}+S_{NO}}\right)\eta_g X_{B,H}$
3 Aerobic growth of autotrophs						1		$-\frac{4.57-Y_A}{Y_A}$	$\frac{1}{Y_A}$	$-i_{XB}-\frac{1}{Y_A}$			$\frac{i_{XB}}{14}-\frac{1}{7Y_A}$	$\hat{\mu}_A\left(\frac{S_{NH}}{K_{NH}+S_{NH}}\right)\left(\frac{S_O}{K_{O,A}+S_O}\right)X_{B,A}$
4 'Decay' of heterotrophs				$1-f_P$	-1		f_P					$i_{XB}-f_P i_{XP}$		$b_H X_{B,H}$
5 'Decay' of autotrophs				$1-f_P$		-1	f_P					$i_{XB}-f_P i_{XP}$		$b_A X_{B,A}$
6 Ammonification of soluble organic nitrogen										1	-1		$\frac{1}{14}$	$k_a S_{ND} X_{B,H}$
7 'Hydrolysis' of entrapped organics		1		-1										$k_h\frac{X_S/X_{B,H}}{K_X+(X_S/X_{B,H})}\left[\left(\frac{S_O}{K_{O,H}+S_O}\right)+\eta_h\left(\frac{K_{O,H}}{K_{O,H}+S_O}\right)\left(\frac{S_{ND}}{K_{NO}+S_{NO}}\right)\right]X_{B,H}$
8 'Hydrolysis' of entrapped organic nitrogen											1	-1		$\rho_7(X_{ND}/X_S)$
Observed Conversion Rates $[ML^{-3}T^{-1}]$	$r_i=\sum_j \nu_{ij}\rho_j$													
Stoichiometric Parameters: Heterotrophic yield: Y_H; Autotrophic yield: Y_A; Fraction of biomass yielding particulate products: f_P; Mass N/Mass COD in biomass: i_{XB}; Mass N/Mass COD in products from biomass: i_{XP}	Soluble inert organic matter $[M(COD)L^{-3}]$	Readily biodegradable substrate $[M(COD)L^{-3}]$	Particulate inert organic matter $[M(COD)L^{-3}]$	Slowly biodegradable substrate $[M(COD)L^{-3}]$	Active heterotrophic biomass $[M(COD)L^{-3}]$	Active autotrophic biomass $[M(COD)L^{-3}]$	Particulate products arising from biomass decay $[M(COD)L^{-3}]$	Oxygen (negative COD) $[M(-COD)L^{-3}]$	Nitrate and nitrite nitrogen $[M(N)L^{-3}]$	$NH_4^+ + NH_3$ nitrogen $[M(N)L^{-3}]$	Soluble biodegradable organic nitrogen $[M(N)L^{-3}]$	Particulate biodegradable organic nitrogen $[M(N)L^{-3}]$	Alkalinity – Molar units	Kinetic Parameters: Heterotrophic growth and decay: $\hat{\mu}_H$, K_S, $K_{O,H}$, K_{NO}, b_H; Autotrophic growth and decay: $\hat{\mu}_A$, K_{NH}, $K_{O,A}$, b_A; Correction factor for anoxic growth of heterotrophs: η_g; Ammonification: k_a; Hydrolysis: k_h, K_X; Correction factor for anoxic hydrolysis: η_h

Table 3.3 Typical parameter and characteristic values of the IAWPRC model (from Henze et al, 1986)

Symbol	Name
Y_A	Yield for autotrophic biomass
b_A	Decay coefficient for autotrophic biomass
f_P	Fraction of biomass leading to particulate products
i_{XB}	Mass of nitrogen per mass of COD in biomass
i_{XP}	Mass of nitrogen per mass of COD in products from biomass
$K_{O,H}$	Oxygen half-saturation coefficient for heterotrophic biomass
K_{NO}	Nitrate half-saturation coefficient for denitrifying heterotrophic biomass
$K_{O,A}$	Oxygen half-saturation coefficient for autotrophic biomass

Symbol	Name	Prior information needed
S_{NOI}	Soluble nitrate nitrogen concentration in wastewater	
S_{NHI}	Soluble 'ammonia' nitrogen concentration in wastewater	
S_{II}	Soluble inert COD concentration in wastewater	
S_{NII}	Soluble inert organic nitrogen concentration in wastewater	
S_{NDI}	Soluble biodegradable organic nitrogen concentration in wastewater	S_{NII}
Y_H	Yield for heterotrophic biomass	
S_{SI}	Concentration of readily biodegradable COD in wastewater	Y_H
$\hat{\mu}_A$	Maximum specific growth rate for autotrophic biomass	b_A
K_{NH}	Ammonia half-saturation coefficient for autotrophic biomass	
b_H	Decay coefficient for heterotrophic biomass	Y_H, f_P
X_{II}	Inert suspended organic matter concentration in wastewater	f_P, b_H, S_{SI}, S_{II}
X_{SI}	Slowly biodegradable organic matter concentration in wastewater	X_{II}, S_{SI}, S_{II}
X_{NDI}	Slowly biodegradable organic nitrogen concentration in wastewater	S_{SI}, X_{SI}, S_{NDI}
η_g	Correction factor for μ_H under anoxic conditions	
η_h	Correction factor for hydrolysis under anoxic conditions	
$\hat{\mu}_H$	Maximum specific growth rate for heterotrophic biomass	$Y_H, X_{SI}, X_{II}, S_{SI}, f_P$
K_S	Half-saturation coefficient for heterotrophic biomass	$Y_H, X_{SI}, X_{II}, X_{SI}, f_P$
k_h	Maximum specific hydrolysis rate	
K_X	Half-saturation coefficient for hydrolysis of slowly biodegradable substrate	
k_a	Ammonification rate	

Symbol	Unit	Value at 20 °C	Value at 10 °C
Stoichiometric parameters			
Y_A	g cell COD formed (g N oxidized)$^{-1}$	0.24	0.24
Y_H	g cell COD formed (g COD oxidized)$^{-1}$	0.67	0.67
f_P	dimensionless	0.08	0.08
i_{XB}	g N(g COD)$^{-1}$ in biomass	0.086	0.086
i_{XE}	g N(g COD)$^{-1}$ in endogenous mass	0.06	0.06
Kinetic parameters			
$\hat{\mu}_H$	day^{-1}	6.0	3.0
K_S	g COD m^{-3}	20.0	20.0
$K_{O,H}$	g O_2 m^{-3}	0.20	0.20
K_{NO}	g NO_3-N m^{-3}	0.50	0.50
b_H	day^{-1}	0.62	0.20
η_g	dimensionless	0.8	0.8
η_h		0.4	0.4
k_h	g slowly biodegradable COD (g cell COD · day)$^{-1}$	3.0	1.0
K_X	g slowly biodegradable COD (g cell COD)$^{-1}$	0.03	0.01
$\hat{\mu}_A$	day^{-1}	0.80	0.3
K_{NH}	g NH_3-N m^{-3}	1.0	1.0
$K_{O,A}$	g O_2 m^{-3}	0.4	0.4
k_a	m^3 · COD (g · day)$^{-1}$	0.08	0.04

Symbol	Unit	Denmark	Switzerland	Hungary
S_S	g COD m^{-3}	125	70	100
S_I	g COD m^{-3}	40	25	30
X_S	g COD m^{-3}	250	100	150
X_I	g COD m^{-3}	100	25	70
S_{ND}	g N m^{-3}	8	5	10
X_{ND}	g N m^{-3}	10	10	15
S_{NH}	g NH_3-N m^{-3}	30	10	30
S_{NI}	g N m^{-3}	2	2	3
S_{NO}	g NO_3-N m^{-3}	0.5	1	1

(1986), together with the assumptions, restrictions and constraints related to the model.

Dold and Marais (1986) have evaluated the Task Group model by comparing it with the Dold et al. (1980) model. They found that the changes proposed by the Task Group did not affect the essential processes of their model, although the changes seemed likely to improve its predictive power. Dold and Marais have expressed their confidence that the Task Group report, in consolidating the existing knowledge on the activated sludge process and in promoting the structural presentation of the model via a matrix representation, has enhanced the utilization of these new theories in practical applications and has given new impetus to research.

3.4.3. The Activated Sludge Process: Modelling of Other Mechanisms

With increasingly stringent and comprehensive effluent standards, pollutant removal mechanisms other than carbonaceous oxidation and nitrification-denitrification have attracted attention, specifically biological phosphorus and heavy metal removal, and phenomena such as bulking sludges.

(a) Biological Phosphorus (P) Removal

Marais et al. (1983) have summarised the principal mechanisms of biological removal of excess phosphorus in the activated sludge process. The salient points are as follows:

(i) P release under anaerobic conditions must be observed in order to give rise to P uptake subsequently under aerobic or anoxic conditions, which results in a net P removal.

(ii) For any fixed process configuration the magnitude of the the excess P removal is linked directly to the

magnitude of the readily biodegradable influent COD, S_{bsi}; the higher the S_{bsi}, the higher the P release, uptake and removal.

(iii) The greater the fraction of total sludge mass in the anaerobic reactor, the greater the net P removal.

(iv) Nitrate recycled to the anaerobic reactor has an adverse effect on P removal; the greater the mass of nitrate recycled, the lower the P removal.

Using these principles, Siebritz et al. (1983) have devised an empirical model that, on the whole, reflects the P removal behaviour pattern as observed. The effects of the anaerobic mass fraction and of S_{bsi} are formulated as a multiplicative function with an appropriate parameter whose value is chosen such that the removal is approximately correctly estimated. No account is taken of P release and uptake as individual processes, merely their net effect (i.e. P removal) is simulated.

Wentzel et al. (1985) have subsequently proposed a kinetic model restricted to P release behaviour, i.e. excluding the kinetics of P uptake. They indicated in their conclusions, however, that a linear relationship was observed between the mass of P release in the anaerobic reactor and the mass of P uptake in the anoxic/ anaerobic part of the process. Arvin et al (1985) have also developed a steady-state model and subjected it to a sensitivity analysis in order to identify the crucial design parameters. Comeau et al. (1986, 1987) have presented a biochemical model explaining biological P removal mechanisms occurring during anaerobic, aerobic and anoxic conditions of the process. Their model integrates observations from the literature with the principles of bioenergetics and membrane transport, and with batch experimental results.

It seems, then, that despite proven, practical technology for biological phosphorus removal (Tetrault et al., 1986), no

complete dynamic model has yet been elaborated. Inspection of the titles of papers presented at the third conference on biological phosphate removal (in Rome, September 1987) suggests too that a dynamic model for this process has not yet emerged. In view of these difficulties it is perhaps not surprising that some preliminary suggestions have been made regarding the possible use of linguistic models for assessing biological P removal (Pinheiro and Camara, 1985).

(b) Heavy metals removal

Heavy metals can play a crucial role in the efficiency of the activated sludge process. In spite of an impressive literature on the empirical behaviour of heavy metals within that process (e.g., Lester, 1983; Stephenson and Lester, 1987), no work had until very recently been conducted on the modelling of any such removal mechanisms. Here again, the experimental studies at Norwich Sewage Works are proving invaluable. Although the problem of modelling these processes in an activated sludge unit is substantially more difficult than that for a primary clarifier, current indications are that, while not initially yielding insights into removal mechanisms, models of the strongly adsorbed metals will give useful results for the dynamics of the sludge mass around the aerator, clarifier, and thickener units (Chan, 1987).

(c) Bulking sludge

> "Bulking sludge is one of the most common and widespread sludge separation problems. It is characterised by a high sludge volume index (SVI 150 mL/g). The sludge settles slowly and thickens poorly, but -- provided there are no problems -- produces a clear supernatant (good clarification). Effluent quality typically is high, unless the sludge blanket begins to escape over the weirs ...

Bulking sludge is associated with the excessive growth of filamentous microorganisms, and mainly caused by either low organic loading or low dissolved oxygen" (Strom and Jenkins, 1984).

Even though bulking sludges cause difficult operational problems, modelling of the phenomenon is still in its infancy. The diversity of the causes as well as the number of microorganisms involved in sludge bulking are undoubtedly responsible for this lack of progress.

Beck (1984) has attempted to provide a quantification of the processes believed to be associated with the occurrence of bulking and rising sludges. Following the early work of Curds (1973), equations representing three different groups of microorganisms associated with bulking (swimming and attached protozoa and dispersed bacteria) were adjoined to a lumped-parameter model similar to that proposed by Busby and Andrews (1975); rising sludge was described by a mechanism related to nitrate concentration and sludge retention time in the clarifier. The model was developed essentially for the purpose of evaluating a novel form of process control, and has not been tested against field data.

Based on batch experimental data under normal and high sludge-loading conditions, Takamatsu et al. (1984) have elaborated a model of the activated sludge process simulating both the kinetics and settleability of the two populations of organisms, namely the floc-forming and the bulking-sludge populations. It was assumed that no direct interaction occurs between the two types of organisms, and that the growth-rate of each population is dependent upon the organic loading applied to the aerator. The concentration of the recycled sludge was computed using an empirical relationship based on the ratio of bulking to floc-forming organisms. Attempting various control strategies, they concluded that a nonlinear state feedback regulator would be able to prevent bulking in the activated sludge process. Like so

many control studies, this may be fine in theory, but it is still some considerable distance from practice; the controller implies accurate estimates of the state variables for the bulking and floc-forming organisms.

Lau et al. (1984) have proposed a micro-scale mathematical model describing the diffusion and utilization of both a carbonaceous substrate (glucose) and DO in a floc composed of a single strain of a floc-forming microorganism (Citrobacter sp.) and a single strain of a filamentous microorganism (S. natans). Their model was based on a combination of double Monod microbial growth kinetics and the diffusion of substrates from the bulk solution through a floc. Clearly this is therefore not a model of a unit process, but a (distributed-parameter) model aimed at quantifying the micro-scale distribution of substrate in an individual organism's environment. The model was applied in order to determine the effects of the bulk substrate concentration, floc size and floc shape on the competitive growth of the two bacteria. The general conclusions were that a low DO concentration, high bulk glucose concentration, large floc sizes and spherical floc shapes tended to favour S. natans and therefore increase the tendency to bulk.

More recently, Hermanowicz (1987) has studied the effects of a variable bulk DO concentration on changes in microbial population composition. He has proposed a simple lumped-parameter model (at the process scale) describing a microbial system composed of two populations: filamentous and floc-forming bacteria. The model has been tested against experimental data, giving encouraging results. He concluded moreover that it might be possible to control the composition of the mixed bacterial community through the control of the dissolved oxygen levels, by alternate periods of low and high DO conditions. It is apparent, however, that low DO concentrations required for the growth of filamentous microorganisms may occur inside a microbial aggregate while the DO in the bulk of the liquid continues to be at a high level. There is thus scope for incorporating some of the micro-

scale mass transfer consideratins of Lau et al. (1984) into Hermanowicz's macro-scale process model. But again, there is evidence of a gap between theory and practice:

> "Dr. Hermanowicz is to be commended for his recognition of the potential offered to control engineers by the sensitivity of microbial kinetics to the dissolved oxygen concentration. Due to the many unknowns, however, the main utility of his proposal at this time is as a springboard to new insights into microbial interactions. Further exploration of his ideas using the tools developed by others should yield important new knowledge and should eventually result in an expanded array of control techniques for those operating activated sludge systems". Grady (1987).

3.4.4. The Activated Sludge Process: Other Design Configurations

The hydraulic characteristics of an activated sludge unit are usually neither plug flow nor completely mixed. Completely mixed reactors (CSTR's) in series could thus serve to simulate the mixing properties of the aerator basin, as already noted in Section 2 (Levenspiel, 1972). Hill (1985), for instance, conducted dye-tracer experiments and found that for the four biological reactors used at the Sagemont treatment plant in Houston, three could be modelled with two CSTR's in series and one with five. The recycle sludge stream also has a significant influence on the mixing characteristics of the basin, and Bode and Seyfried (1985) have observed that tracer-response curves of completely mixed tanks with increasing return sludge ratios approach the theoretical response curves for reactors experiencing plug flow. Olsson et al. (1986) have also observed that conventional wastewater treatment plants have almost no damping capacity for hydraulic shocks, except for a minor degree of damping afforded by the aerator basin.

We note that a number of other design configurations for the activated sludge process are possible, the most common being step-feed forms, contact-stabilization and oxidation ditches.

Whereas the conceptual model for the biological mechanisms of these configurations should be similar to the IAWPRC model, their mixing/hydraulic characteristics can be expected to differ considerably from case to case. Some examples, taken from Henze et al. (1986), are shown in Figure 3.11.

A series of four tanks, with the option of influent distribution (part or total) to each tank, usually serves to simulate the step-feed process (e.g. Busby & Andrews, 1975). The contact-stabilization process can be modelled as two tanks in series, the first receiving the returned activated sludge, the second the settled sewage influent and the discharge of the first reactor. Lastly, for the oxidation ditch Henze et al. (1986) have proposed a series of CSTR elements, the first of which receives the returned sludge and the influent, and is oxygenated, while the other tanks simply receive the preceding tank's discharge and are not oxygenated. Cook (1983) has suggested the use of CSTR and PFR elements in series to simulate this process, i.e. a distributed-parameter model, although he subsequently concluded that such a model would be overly complex given the nature of the avaialble field data, and that a single CSTR approximation would suffice. Wishart (1986) has evaluated with some success a model of this latter form using a time-series of 35 daily field observations; there were, however, substantial difficulties in replicating the thickening behaviour of the clarifier associated with the ditch under study.

3.4.5. The Activated Sludge Process: Other Dynamic Models

Most research on activated sludge modelling treats the microbial floc and liquid as a single-phase system whereas, in reality, the two phases are separate. In theory, therefore, the transfer of mass between and within the phases ought to be considered in the development of models. Working along these lines Benefield and Molz (1983, 1984) have developed a mathematical model describing organics removal, oxygen utilization, ammonia-nitrogen removal, ortho-phosphate removal and biomass production in an aggregated

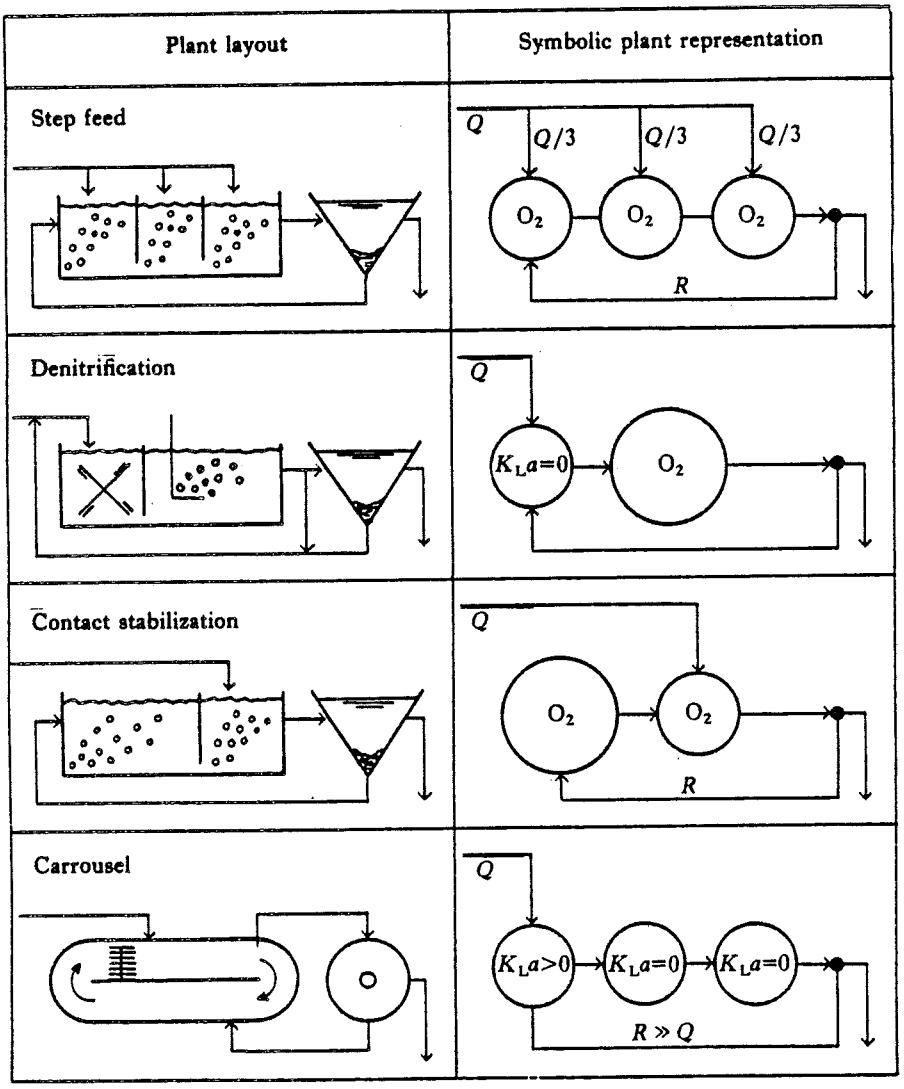


Figure 3.11 Different configurations of the activated sludge process model (from Henze et al, 1986).

microbial suspension containing organics as a soluble biodegradable material and having a uniform floc size. This distributed-parameter model, consisting of five partial differential equations and four ordinary differential equations, takes into account the flow pattern in the reactor, intraparticle mass transport of oxygen, organics, ammonia-nitrogen and ortho-phosphate, and biochemical reactions by the individual cells embedded in the floc. Not surprisingly, their model remains as yet untested against experimental observations, although it does reveal interesting features of the competition between autotrophic nitrifying bacteria and heterotrophic bacteria for oxygen and ammonia-nitrogen substrates.

At the other end of the spectrum, simple input/output time-series models have been proposed and identified for the dynamics of the activated sludge process. Such models, as we have said, assume no prior knowledge of the internal mechanisms governing the relationships between the observed input and output time-series. In this respect the work of Berthouex and his colleagues has played a leading role. For example, from the analysis of hourly time-series data, Berthouex et al. (1978) have identified and successfully verified a simple model relating the effluent 5-day BOD to the influent 5-day BOD for a municipal wastewater treatment plant. Later work has focussed on the P removal behaviour of wastewater treatment processes (Bolman et al., 1987). There has also been some analysis of time-series models for nitrifying systems (using hourly data; Murphy et al, 1977), and similar models, using daily data of COD and SS, have been developed for industrial activated sludge plants (Hansen, et al., 1980; Debelak et Sims, 1981).

It is also possible to develop input/output (or transfer function) models in the frequency-domain, as opposed to the time-domain. In this review, we have not indicated a specific focus on such models, and will therefore note only in passing Corder and Lee's (1986) studies on relating the dissolved oxygen concentration, at the end of the aeration tank, to the input applied biological

load, and (in a separate model) to the air flow rate to the aeration tank. Their work has been based on full-scale treatment plants and enabled the design of a feedforward controller capable (in principle) of savings of up to 20% in the aeration input.

3.4.6. The Activated Sludge Process: Closing Remarks

The IAWPRC Task Group model must certainly be considered as a milestone in the development and promotion of models for the activated sludge process. Nevertheless, there are many other aspects of the behaviour of the process that will need innovative thinking if they are to be incorporated in this model (Dold and Marais, 1986):

(i) Virtually every process mechanism and its associated kinetic formulation need critical re-assessment.

(ii) The cause of an increase in heterotrophic growth rate in systems that operate under plug flow conditions, or have "selector" reactors, should be established.

(iii) The growth of filamentous organisms with reference to bulking sludge needs to be incorporated.

(iv) The growth of organisms that give enhanced biological P removal should be characterised, i.e. the breakdown of readily biodegradable COD to lower fatty acids in the anaerobic reactor, the use of fatty acids as a source for substrate sequestration, and growth of the polyp organisms in anaerobic-anoxic-aerobic systems, and so on.

The dynamic modelling of heavy metals removal in the activated sludge process is yet another possible growth area of research.

It is clear from this review that the dominant thrust of the research on modelling the activated sludge process has been towards the development of lumped-parameter models. This largely reflects the dominance of an interest in the microbial nature of the process, with a concern for detailed mass transfer considerations (necessitating distributed-parameter models) developing only in the more recent years. In any application of a model there are bound to be tradeoffs among the needs of accuracy, simplicity, and expediency (see also Scheffer et al., 1984; Olsson et al., 1985). It is a truism that the purpose of a model should be clearly stated. The detail, completeness, and insights generated by the development of the IAWPRC model have undeniably been a most significant contribution. Nevertheless, the IAWPRC model is just as surely not appropriate for every purpose of model application, and in a number of practical respects its development thus far leaves omissions. Such models contain many parameters that are unknown and must be estimated with reference to field data; in technical terms the IAWPRC model will in this respect suffer from a lack of identifiability, i.e. many combinations of parameter values will fit the data equally well (Beck, 1986).

Above all, most of these lumped-parameter models for the activated sludge process need now to be confronted with rigorous tests against comprehensive sets of field data from full-scale plant installations. Our own experience in the case of the Norwich Sewage Works has emphasised not only the benefits of doing just this, but also the acute and fundamental difficulties thereof. There is a danger of a widening gap between theory and practice in the modelling of the activated sludge process; a gap that sooner or later becomes apparent (and unacceptable) in almost all areas of mathematical modelling (for example, Beck, 1987).

3.4.7. Secondary Clarifier

The activated sludge secondary settler has two main functions:

1) the production of a good effluent, free of settleable solids, and

2) production of an underflow, high in solids (biomass) concentration for recycle to the aerator.

It could also serve as a temporary sludge storage volume during rain events (Ditsios, 1984). The good performance of the secondary settler is vital, since each mg of solids lost over its weir may increase the effluent BOD by about 0.6 mg (Dick, 1970).

Because of the small difference in solid (bacterial floc) and liquid densities, this process is susceptible to failure due to shifting flow rates or changing operating conditions in the biological reactor (Lev et al., 1986). Several studies have investigated the influence of such process variables on the performance of secondary clarification (e.g. Lumley, 1985). Among these, changes in MLSS concentration, the feed flow-rate, and sidewater depth seem to be the most important variables (Chapman, 1983). Lev et al. (1986) have suggested that changes in the settling properties of the floc are the main reason for failure of the activated sludge process, settling velocity being affected by the presence of filamentous bacteria and the concentration of polysaccharide (an exocellular polymer that serves as a flocculant).

Although not comparable with that on the activated sludge aerator, literature on the dynamics of secondary sedimentation is abundant. Much research has been carried out across a whole spectrum of topics dealing with sedimentation in the activated sludge process, yet still no theory adequately describes the dynamics of sedimentation despite advances in basic understanding (Lumley, 1985).

The behaviour of the secondary settler is generally viewed as a function of two independent mechanisms, clarification and thickening (the term "independent" means here that no obvious

relation exists between the two mechanisms). It is possible for the sludge blanket level to be seen as an element that links these two component mechanisms, although some have demonstrated the importance of the sludge blanket to clarification (Ghobrial, 1978; Lumley, 1985), while others flatly deny this importance (Crosby & Bender, 1984; Hill, 1985). There is obviously scope for further research. Our review will distinguish between the modelling of clarification and the modelling of thickening.

(a) Clarification

Most models of the clarification process are empirical and based on statistical relationships identified from field data or bench-scale experiments. Pflanz's (1969) equation, relating the effluent SS to the influent solids loading of the clarifier, has often been quoted, although a more detailed consideration of hydraulic characteristics has led others to propose CSTR elements in series for simulating this relationship (e.g., Tanthapanichakoon & Himmelbau, 1981).

Hill (1985) has reviewed all the major models, as summarised in Table 3.4, where it can be seen that the most important variables are flow-related variables and the mixed liquor suspended solids concentration. From these six models only one suggests the importance of the sludge blanket level. With reference to field data, Hill (1985) finally opted for a decay model described by the following three equations (3.6, 3.7, 3.8):

$$(X_e)_{ss} = K_1 + K_2 F_e \qquad (3.6)$$

where if the value estimated by equation (3.6) is less than the current effluent solids concentration (as with a sudden decrease in flow), the decay (rate of change) of X_e is assumed to be proportional to the difference between the current solids concentration and the steady-state value, i.e.

Table 3.4 Summary of models for the clarification function of the secondary clarifier (from Hill, 1985)

Parameter	Model					
	1	2	3	4	5	6
Influent Flow Rate	X	X	X		X	X
Recycle Flow Rate	*		X	X	X	X
MLSS Concentration	X	X	X	X	X	
Activated Sludge Organic Loading		X				
Activated Sludge Detention Time		X				
Sludge Blanket Level			X			
Activated Sludge Air Flow Rate				X	*	
Clarifier Surface Area	X	X	X		X	
Clarifier Side Wall Depth					X	
Flow Rate Variations						X

1 Pflanz (1969), full-scale data
2 Agnew (1972), full-scale data
3 Ghobrial (1978), bench-scale data
4 Tuntoolavest et al, (1983), bench-scale data
5 Chapman (1983), pilot-scale data
6 Collins and Cosby (1980), no experimental data

x Parameter incorporated into model
* Found to be insignificant

$$\text{if } (X_e)_{ss} > X_e \quad \text{then} \quad X_e = (X_e)_{ss} \tag{3.7}$$

$$\text{if } (X_e)_{ss} < X_e \quad \text{then} \quad \frac{dX_e}{dt} = K_3 \, (X_e - (X_e)_{ss}) \tag{3.8}$$

where

X_e = final effluent suspended solids (M/L^3)
$(X_e)_{ss}$ = X_e at steady-state conditions (M/L^3)
K_1, K_2, K_3 = parameters, F_e = effluent flow.

The model is capable of simulating both a rapid increase and a slow decay of effluent suspended solids concentration in response to flow transients. Hill (1985) has also evaluated a number of other models (see Table 3.5), although rejecting them on the grounds that they gave no significant improvement over the model of equations (3.6) - (3.8).

Using time-series analysis of continuously monitored effluent data (described in Chapman, 1985), Olsson and Chapman (1985) have identified some simple input/output model forms. The parameter values of these models reveal the time-varying character of settling sludge, indicating different behaviour during a decreasing or an increasing flow-rate. The model parameters were found to change gradually as a function of both changing floc properties and instrument drift. Despite the fact that the parameters of such models do not strictly have any physical interpretation, the model can nevertheless be used for monitoring purposes; Olsson and Chapman (1985) have demonstrated in particular that on-line estimation is a powerful tool for detecting parameter variability, thus giving the operator an automatic warning that the clarifier behaviour may have deviated from that of its normal operation.

(b) Thickening

On balance, the thickening function of the secondary clarifier has received greater attention than clarification. The majority of

Table 3.5 Additional models of clarification evaluated by Hill (1985).

$$Xe = K_1 + K_2 \cdot Fe + K_3 \cdot MLSS$$

$$Xe = K_1 + K_2 \cdot Fe + K_3 \cdot MLSS + K_4 \cdot Fe \cdot MLSS$$

$$Xe = K_1 + K_2 \cdot (Fe+Fr) + K_3 \cdot (Fe+Fr) \cdot MLSS$$

$$Xe = K_1 + K_2 \cdot Fe + K_3 \cdot \frac{d(Fe)}{dt}$$

$$\frac{d(Xe)}{dt} = K_1 \cdot Xe + K_2 \cdot Fe + K_3 \cdot \frac{d(Fe)}{dt}$$

$$\frac{d(Xe)}{dt} = K_1 \cdot Fe \cdot Xe + K_2 \cdot Fe \cdot MLSS + K_3 \cdot \frac{d(Fe)}{dt}$$

$$\frac{d(Xe)}{dt} = \frac{(Fe+Fr)}{K_3} \cdot (K_1 + K_2 \cdot Fe - Xe)$$

$$\frac{d(Xe)}{dt} = \frac{(Fe)}{K_3} \cdot (K_1 + K_2 \cdot Fe - Xe)$$

$$\frac{d^2(Xe)}{dt^2} = K_1 \cdot \frac{d(Xe)}{dt} + K_2 \cdot Xe + K_3 \cdot Fe + K_4 \cdot \frac{d(Fe)}{dt}$$

the models proposed have only accounted for convective transport in the thickening region, and are based on the assumptions that there exists a limiting solids flux and that the clarifier is capable of accommodating any accompanying increase in sludge blanket volume. Through this assumption the dynamics of thickening have been rendered independent of the clarification behaviour.

> "The essence of the flux theory is that it predicts that under overloaded conditions there is a critical zone settling layer of sludge which limits the solids flux to the bottom of the tank. When the applied flux is greater than the limiting flux, the mass of sludge in the zone settling layer increases and causes the sludge blanket level to increase, the rate of increase depending on the severity of the overload (i.e., the magnitude of the difference between the applied and limiting fluxes)". (Ekama & Marais, 1986a).

Field data on the secondary settler's behaviour have indeed provided support for the fundamental principle on which the flux theory is based (Ekama & Marais, 1986a). The solids flux in a thickener has two components: a settling flux due to gravitational floc settling and a bulk flux due to downward movement of the tank contents as the underflow is withdrawn for recycle and wasting. A straightforward first-order partial differential equation results, i.e. a distributed-parameter model:

$$\frac{\partial C}{\partial t} = \frac{\partial G_s}{\partial z} + \frac{\partial G_u}{\partial z} \qquad (3.9)$$

where: C = solids concentration in the thickener

G_s = gravity flux = $V_s C$

G_u = underflow flux = $V_u C$

V_s = hindered settling velocity

V_u = downward velocity (Q_u/A)

Q_u = underflow rate; A = settler area

z = distance along the vertical axis

The gravity flux is the product of the concentration and the hindered settling velocity, where the latter has always been considered a function of the local solids concentration, and empirical relationships have been developed to describe the velocity-concentration function. Table 3.6, adapted from Hill (1985), shows most of the equations proposed for this function. Of these the logarithmic and exponential equations have proved the most popular, the latter leading to a more intuitively satisfying flux theory model for analysis of the secondary sedimentation tank behaviour (Smollen & Ekama, 1984).

The first major contribution to the description of continuous thickening is due to Bryant (1972), whose model divided the sludge blanket into a series of some 20 constant-volume CSTR elements with equation (3.9) used to account for thickening. Most later models have been based on Bryant's model and have simulated the blanket as a series of n layers with variable or fixed volumes.

Tracy and Keinath (1974) upgraded Bryant's model by including an upper dilute concentration zone and a completely stirred compression zone. Provisions were made in their computer solution to limit the maximum flux transmitted throughout the thickener to the limiting flux predicted by the flux theory; the SS concentration of the top (or any) layer of the sludge blanket could not exceed that limiting concentration. Once at that concentration, the volume of the top layer was allowed to expand or contract with loading conditions.

Stenstrom (1975) further developed Bryant's work and presented a model based on a one-dimensional continuity equation and incorporating solids flux theory. His distributed-parameter model consists of one partial differential equation with boundary conditions imposed at the top and bottom of the settler. Hill

Table 3.6 Relationships between solids concentration and settling velocity (adapted from Hill, 1985).

Equation
$v = v_0 (1-KC)^{4.65}$
$v = v_0 (1-KC)^2 \cdot 10^{-1.82KC}$
$v = v_0 10^{-aKC}$
$v = aC^{-b}$
$v = v_0 [1 - 2.78 (KC)^{2/3}]$
$v = v_0 \exp(-aC)$
$v = v_0 [1 + \frac{3}{4} KC (1 - \frac{8}{KC-3})]$
$v = v_0 (1-KC)^a$
$v = v_0 (1-aKC) [1 - b(KC)^{1/3}]$
$v = \frac{v_0 [3 - \frac{9}{2} (KC)^{1/3} + \frac{9}{2} (KC)^{5/3} - 3 (KC)^2]}{3 + 2 (KC)^{5/3}}$
$v = a/(C-b)$

v = interface velocity of solids concentration, C
v_o = Stokes settling velocity for a single discrete particle
K = conversion variable so that KC = volume fraction of solids
a,b = coefficients (unique to each equation)

(1985) based his model on Stenstrom's work, but modified it to take into account the effect of a conical section in the clarifier.

Forster (1985) too has developed a simple model based on Bryant's work, and evaluated it with limited success against data deriving from a short field experiment of 24 hours' duration.

Other notable contributions are those of Attir et al. (1976) and George & Keinath (1978). Their work has concentrated on achieving a more accurate representation of the theoretical behaviour of an ideal slurry, giving rise to a complex model with relatively complicated boundary constraints, and whose computation requires the manipulation of varying numbers of layers of varying thickness and concentration (Forster, 1985). Such models would certainly not be appropriate for the design of control systems, as observed by Stehfest (1984), who has accordingly developed a lumped-parameter model consisting of two variable-volume CSTR's separated by a discontinuity and using three state variables, sludge blanket height, mean concentration above the blanket, and mean concentration below the blanket.

The main drawback of schemes based on equation (3.9) is the necessity of artificially imposing limiting fluxes, normal gradients and stable discontinuities (Lev et al., 1986). Numerical solution of the equation may well lead to "multiple" sludge blankets, i.e. the sludge concentration jumps between two solutions (Olsson, 1987). Olsson has therefore suggested a model structure that inherently gives a solution with a uniquely defined blanket.

> "In the mass balance that leads to equation (3.9) there has been no consideration of the increasing solids/liquid interaction as the concentration approaches the compaction zone. As a result some "second order" effects have been neglected. One is the friction between the solids and the liquid, and another is the upflow of liquid to compensate

for the mass of solids flowing downwards. These phenomena have been modelled by a diffusion term that has been added to the model". (Olsson, 1987).

$$\frac{\partial C}{\partial t} = \frac{\partial G_s}{\partial z} + \frac{\partial G_u}{\partial z} + D\,\frac{\partial^2 C}{\partial z^2} \qquad (3.10)$$

where D = diffusion coefficient.

Anderson and Edwards (1981) had earlier also indicated that dispersive effects should not be neglected in peripheral feed/peripheral withdrawal settlers. Their modification of the convective equation is the same as that proposed later by Olsson, and they adopted a fixed-grid finite-difference scheme for numerical solution of their model. Lev et al. (1986) proposed also a similar approach using a simple finite-element scheme incorporating a small Fickian dispersion term. They argue that one advantage of their scheme over existing dynamic models is that it does not need the imposition of artificial constraints for agreement with a steady-state analysis.

(c) Closing Remarks

It is difficult to define which models of clarification and thickening would constitute the state-of-the-art in dynamic simulation of the secondary clarifier tank. Some observations, however, are that:

(i) Settlers are usually conceptualised as two different independent processes (clarifier and thickener), an assumption that has yet to be unequivocally supported.

(ii) The models have not generally been evaluated with reference to sound experimental work at the plant-scale, and especially so for the thickener models.

(iii) Clarifier models are mostly based on empirical relationships, and are therefore strongly site-specific.

(iv) Solids flux theory, known to describe the process fairly well, albeit qualitatively (Ekama & Marais, 1986a), cannot predict the sludge blanket level adequately if formulated with equation (3.9) (Olsson, 1987).

Overall the tendency has been to focus on the development of distributed-parameter models of the thickening function, and in this respect the model of Lev et al. (1986) might serve as a guide to the future. Hill's (1985) model might similarly be used as a point of departure in the modelling of the clarification process.

The fact that no simple and reliable model has yet emerged is probably a reflection of the nature of the process itself. So many factors (side water depth, inlet position, sludge removal mechanism, etc.) can influence its behaviour, including fundamentally the behaviour of the biological reactor, which influences sludge characteristics. In addition, there is simply a lack of decent experiments on the process as an individual unit and in tandem with the aerator.

Since the biological reactor plays such an important role in the clarification-thickening process, one could argue that for as long as the aerator model does not incorporate items to differentiate between floc-forming and bulking sludges (as discussed earlier), attempts to elaborate more sophisticated models of the clarifier are doomed to failure. It has been observed that:

> "Since activated sludge is not an ideal slurry the search for more accurate models of the behaviour of such a slurry seems slightly unnecessary. A more suitable objective might be to develop a simplified model based

on solids flux theory and establish its limitations with respect to (i) its departure from the ideal, and (ii) its applicability to real activated sludge". (Forster, 1985).

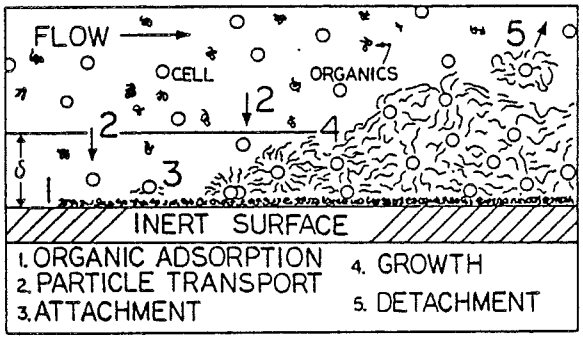
Once again our experience with experimental studies at Norwich indicates that the behaviour of the aerator, such as a low DO affecting sludge settleability, or of the settler, for example, denitrification of anaerobic sludges, makes the process almost impossible to model simply with a single state variable such as suspended solids concentration. On the other hand, we have had some encouraging success with a 3 CSTR representation of thickening; indeed this success has highlighted the failure of our model to simulate observed clarification behaviour.

3.5. Secondary Biological Treatment: Attached Growth Processes

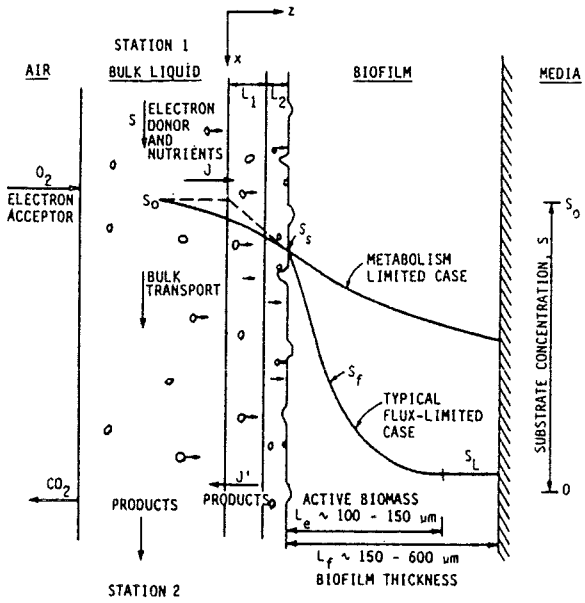
Fixed-film processes found in wastewater treatment plants include (most commonly) trickling filters (TF) (high- or low-rate), and rotating biological contactors (RBC). As with the behaviour of suspended growth processes, there is first some basic theory to be summarised.

3.5.1. Attached Growth Processes: Biofilm Modelling

The various mechanisms contributing to biofilm development, including the potentially rate-limiting phenomena in biofilm reactions and substrate uptake, are shown in Figure 3.12. Grady (1983) has completed an earlier state-of-the-art review on the modelling of biological fixed films, and his principal observations are given as follows:



Processes contributing to biofilm development as presented in Trulear and Characklis (1982).



Rate-limiting phenomena in biofilm reactions and substrate concentration profile (Gullicks and Cleasby, 1986).

Figure 3.12 Conceptual representation of some of the processes of biofilm growth.

(i) Many of the models for biological fixed films fall into the category of empirical models.

(ii) In RBC's and TF's, the radius of curvature of the support medium is large with respect to the biofilm thickness so that the support may be considered to be flat.

(iii) Biofilm modelling requires basic knowledge in the following areas:

(1) transport of material in the liquid phase,
(2) characterisation of the biofilm (thickness, density and composition),
(3) transport and reaction within the biofilm, and
(4) techniques for solving the resulting equations.

(iv) Most models treat the biofilm-liquid interface as if it were analogous to the interface between a flowing fluid and a solid support.

(v) There are two methods of modelling external mass transport: one assumes that transport across the liquid layer is by molecular diffusion, and the other uses a mass transfer coefficient.

(vi) It is important that a distinction be made between the total film thickness and the active film thickness, this being a consequence of transport limitations within the biofilm.

(vii) The general situation for most biofilm reactors in wastewater treatment is one in which continual attrition is not sufficient to balance the net

growth, with the result that the film grows until conditions develop near the support/biofilm interface that cause adhesion to be lost and the film to slough away.

(viii) Because until recently relatively little was known about biofilm detachment, the vast majority of models have assumed a constant film thickness. This may not be a good assumption for fixed-film processes such as TF's and RBC's, because the concentration gradients within them could cause wide variations in film thickness.

(ix) Because the rates of reaction are functions of the mass of microorganisms present, their density must be coupled with the thickness and area of the film to allow computation of the reaction rate. Although it has generally been assumed that the density is constant and independent of film thickness, there is evidence that this is not the case.

(x) Most models treat the biofilm (microbial composition) as if it were homogeneous throughout. This is because most models have sought to predict the fate of one constituent.

(xi) Because of the gelatinous character of the biofilm matrix it is thought that convective transport contributes little to the movement of reactive constituents within the film and that the electron donor and acceptor reach the organisms by diffusion, which may be characterized by Fick's Law.

(xii) The two most widely used reaction rates (relationship between the specific growth rate of a bacteria and the concentration of a single nutrient) are those of Monod and Blackwell.

(xiii) Because of the concentration gradients within the biofilm, it is likely that the concentrations of both the electron donor and the electron acceptor could limit the rates of reaction. Thus, it would be desirable to have a general model which handles all types of double substrate limitation. Two philosophies exist on this subject, and have been labelled as non-interactive and interactive.

A non-interactive model is based upon the concept that the specific growth-rate of an organism can only be limited by one substrate at a time. Therefore, the specific growth rate will be equal to the lowest rate that would be predicted from the separate single substrate model.

An interactive model is based upon the assumption that if two substrates are present in less than saturating concentration, then both must affect the overall specific growth rate of the organisms (e.g. a double Monod equation).

(xiv) If the attached-growth reactor is approximated as a plug flow reactor with the biofilm distributed along its length, the model of this reactor must combine mass balance equations in the axial direction with mass balance equations in a direction perpendicular to that axis (i.e. into the film). Three approaches have been used for solving the equations in such an overall process model: direct, indirect and with an effectiveness factor.

In the direct approach appropriate numerical techniques are employed to solve the two sets of partial differential equations simultaneously.

In the indirect approach, the equation representing reaction and transport within the biofilm is solved for various concentrations of reactants in the bulk liquid and the result is expressed as the flux of material into the biofilm as a function of the bulk fluid concentrations, transport characteristics, and so on. This flux relationship can then be used during solution of the process in an iterative manner.

The effectiveness factor approach is similar to the indirect technique but results in a somewhat more general solution. The effectiveness factor is defined simply as the ratio of the actual, observed reaction rate in the presence of mass transport limitations to the theoretical rate in their absence (i.e., the intrinsic reaction rate).

(xv) On film thickness and sloughing substantially more experimental work is needed before these important aspects of fixed-film reactors can be adequately modelled.

(xvi) Unless evidence to the contrary is overwhelming, double substrate limited models should be employed.

(xvii) In conclusion, it is clear that a complete and general "mechanistic" model for biofilms, which can be used to simulate the performance of a broad range of fixed film processes, has not yet been developed.

Since Grady's review a significant amount of research on the modelling of fixed-film processes has been published. For example, Kissel (1986) has examined the mass transfer phenomena of biological processes and concluded that: (i) the mass transfer and diffusion coefficients required in modelling are subject to significant uncertainty; (ii) in fixed-film processes, significant external mass-transfer resistance is likely for large-media, low-flow systems, but unlikely for small-media, high-flow systems, and possible in the intermediate range of conditions. Similarly Muslu (1986a,b) has studied experimentally the distribution of solute retention times in a biological filter.

Other than these more recent studies of mass-transfer and transport characteristics, Sarner (1986) has examined the removal mechanisms of particulate and dissolved organic matter in fixed-film processes. He points out that certain important areas of research have been neglected, in particular the transport of metabolic end-products from the biofilm to the bulk water, and the interaction between particulate and dissolved organic matter at the interface between the bulk fluid and the biofilm. The concentration of a given species of microorganism in the biomass is also known to be non-uniform, being irregular or patchy in directions perpendicular to the depth of the biofilm (Gujer, 1987). This challenges the validity of the customary assumptions that microscopic gradients are significant in one dimension only, along the depth of the biofilm, and that macroscopic gradients along a single reactor dimension suffice to describe overall process performance.

The inherent two-phase nature of fixed-film processes may, among other reasons, have slowed the progress of dynamic model development. It is pertinent to note that considerations of two-phase representations for the micro-

scale organism environment of suspended growth processes are also only recent innovations. It might indeed be argued that in these respects there is a growing area of common ground between the modelling of attached- and suspended-growth processes. Thus Kissel et al. (1984) have postulated a mixed-culture biofilm model incorporating mass balances on solutes and active and inert bacterial masses within a biofilm in a completely mixed reactor approximation. A system of ordinary differential equations is used to describe growth and decay of the active bacterial mass of several species (aerobic and facultative anaerobic heterotrophs, Nitrosomonas and Nitrobacter) and accumulation of inert mass of these species. The bulk concentrations of five substrates are also represented in lumped-parameter form, with transport/diffusion of these substrates into the depth of the biofilm described by a set of second-order partial differential equations. Loss of mass by periodic sloughing or continuous shearing was, however, not included in the model. Needless to say, a model of such daunting complexity (a total of 19 ordinary and 10 partial differential equations) can at present be regarded as merely a hypothetical formulation for generating preliminary insights.

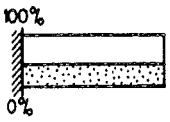
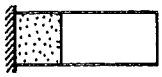
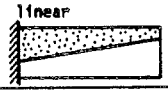
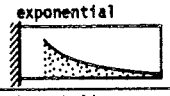
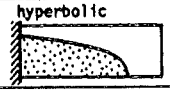
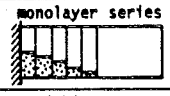
The model of Benefield & Molz (1985) is similar, although somewhat simpler. The bulk concentrations of two substrates are generated from a PFR approximation, a distributed-parameter model, and transport/uptake of these substances in the biofilm is assumed to be instantaneously at steady-state (thus yielding ordinary differential equations quantifying their spatial distribution). As an advance on the model of Kissel et al, the Benefield and Molz model incorporates the loss of the biofilm due to sloughing. When any section of biofilm becomes sufficiently thick to prevent organic substrate from penetrating to the surface of the support medium, the model can allow that section of biofilm to slough, leaving a thin film of 5 μm in depth. At that point the growth process is assumed to resume.

This in turn is similar to the earlier work of Howell and Atkinson (1976) whose model assumed an intermittent sloughing of biomass triggered when substrate concentrations in the depth of the film reach a critical minimum. Trulear and Characklis (1982) have found that the biofilm shear loss-rate depends on the shear stress, the biofilm thickness and biofilm density. This then raises the question of simulating the dynamics of biofilm thickness, a point addressed in two more recent papers, on the particular aspects of film growth before a steady-state is reached (Annachhatre & Khanna, 1987), and on biofilm shearing mechanisms (Speitel & DiGiano, 1987). Nevertheless, sloughing mechanisms and detachment due to hydrodynamic conditions remain poorly understood and continuing serious weaknesses of fixed-film modelling.

It has been observed that "it will probably be impossible to develop a mathematical model that completely describes the trickling process because of its extreme complexity", (Sarner, 1986), and this is a common comment in the literature. Nevertheless, simplifications are possible, and some of these are summarised in Table 3.7 (from Wanner and Gujer, 1986). Typically it might be assumed that the film thickness is either known a priori or time-invariant (at steady-state, by assumption), or that the film composition is homogeneous. It has been argued that such simplifications restrict the applicability of the resulting models to specific process configurations, hence Warner and Gujer's (1985, 1986) attempt to develop a more general treatment of the subject. Their model has established a relationship between biomass production and volume expansion of the fixed film (see Table 3.8).

It is apparent, as in the discussion of the models for the secondary clarifier of suspended-growth systems, that the more complex biofilm models suffer from certain difficulties of accurate and stable numerical solution (Kissel et al, 1984).

Table 3.7 Summary of features of biofilm models (from Wanner and Gujer, 1986).

ref. no	microbial spatial distribution	basis for spatial distribution	microbial system studied	reactor	kinetics	type of model: substrates	type of model: biomass	biomass detachment	solution technique	experiments
1	uniform	a priori assumption	3 groups of anaerobic organisms	submerged anaerobic filter	Monod	stst	stst	-	numerical	yes
2		a priori assumption	BOD degradation, nitrification	RBC	multiple Monod	dyn	stst	yes, production calculated	numerical	yes
3		a priori assumption	BOD degradation, nitrification	RBC	zero order	stst	stst	-	analytical	yes
4	layered	experimental results	glucose degradation, nitrification	RBC	zero order	dyn	stst	-	numerical	yes
5		microscopic observation	allegedly Desulvovibrio and Beggiatoa	RBC	-	-	-	-	-	yes
6	linear 	experimental observations of microbial activity	Nitrosomonas, Nitrobacter	fluidized bed	double Monod	dyn	stst	-	numerical	yes
7	exponential 	analytical mathematical model	organics degradation, nitrification	-	zero order	stst	pseudo stst	not possible	analytical	no
8	hyperbolic 	data fitting	organics degradation (aerobic, denitrification), nitrification	RBC	zero order	dyn	stst	-	numerical	yes
9	monolayer series 	conceptual model for replication of individual bacteria	any two competing species	-	-	dyn	dyn	-	-	no
10	variable	microbial growth in a defined series of film segments	organics degradation (aerobic, denitrification), nitrification	CSTR	Monod	pseudo stst	dyn	may be included	numerical	no

References

1. J. C. Young and P. L. McCarty, "The Anaerobic Filter for Waste Treatment," Technical Report No. 87, Department of Civil Engineering, Stanford University, Stanford, Ca., 1968.
2. J. A. Mueller, P. Paquin, and J. Famularo, "Nitrification in Rotating Biological Contactors," presented at the 51st Annual Conference WPCF, Anaheim, Ca, 1978.
3. I. E. Gönenç, "Nitrification on Rotating Biological Contactors," Department of Environmental Engineering, The Technical University of Denmark, LtH 82-42, 1982.
4. Y.Watanabe, K. Nishidome, Ch. Thanantaseth, and M. Ishiguro, "Kinetics and Simulation of Nitrification in a Rotating Biological Contactor," presented at the 1st International Conference on Fixed-Film Biological Processes, Kings Island Oh., 1982.
5. J. E. Alleman, J. A. Veil, and J. Th. Canaday, *Water Res.*, **16**, 543 (1982).
6. H. Tanaka and I. J. Dunn, *Biotechnol. Bioeng.*, **24**, 669 (1982).
7. P. Harremoës, *Wat. Sci. Tech.*, **14**, 167 (1982).
8. Y. Watanabe, S. Masuda, K. Nishidome, and C. Wantawin, *Wat. Sci. Tech.*, **17**, 385 (1984).
9. P. L. McCarty, M. Reinhard, and B. E. Rittmann, *Env. Sci. Technol.*, **15**, 40 (1981).
10. J. C. Kissel, P. L. McCarty, and R. L. Street, *J. Env. Engin.*, **110**, 393 (1984).

Table 3.8 Summary of the biofilm model of Wanner and Gujer (1986) and its parameter values.

Table II. Stoichiometry and rate laws for microbial processes.

Process j	Biomass compound i: 1 (Het)	2 (Aut)	3 (Inert)	Substrate compound i: 1 (Organics)	2 (NH_4^+)	3 (O_2)	Process rate P_j
Heterotrophic							
1. Growth	+1	—	—	$-\frac{1}{Y_1}$	—	$-\frac{\alpha_1 - Y_1}{Y_1}$	$\hat{\mu}_1\rho_1 f_1 \frac{S_3}{K_{O_1} + S_3} \frac{S_1}{K_{S_1} + S_1}$
2. Endogenous respiration	−1	—	—	—	—	−1	$b_1\rho_1 f_1 \frac{S_3}{K_{O_1} + S_3}$
3. Inactivation	−1	—	+1	—	—	—	$k_1\rho_1 f_1$
Autotrophic							
4. Growth	—	+1	—	—	$-\frac{1}{Y_2}$	$-\frac{\alpha_2 - Y_2}{Y_2}$	$\hat{\mu}_2\rho_2 f_2 \frac{S_3}{K_{O_2} + S_3} \frac{S_2}{K_{S_2} + S_2}$
5. Endogenous respiration	—	−1	—	—	—	−1	$b_2\rho_2 f_2 \frac{S_3}{K_{O_2} + S_3}$
6. Inactivation	—	−1	+1	—	—	—	$k_2\rho_2 f_2$
Observed growth and conversion rates	$\mu o_i = \sum_j \frac{P_j \nu_{ij}}{f_i \rho_i}$			$r_i = \sum_j P_j \nu_{ij}$			

Table III. Parameters for microbial kinetics.

	Heterotrophs	Autotrophs
Maximum growth rate	$\hat{\mu}_1 = 4.8\ d^{-1}$	$\hat{\mu}_2 = 0.95\ d^{-1}$
Substrate saturation constant	$K_{S_1} = 5$ g COD/m³	$K_{S_2} = 1$ g N/m³
Oxygen saturation constant	$K_{O_1} = 0.1$ g O_2/m³	$K_{O_2} = 0.1$ g O_2/m³
Biomass yield	$Y_1 = 0.4$ g COD/g COD	$Y_2 = 0.22$ g COD/g N
Endogenous rate constant	$b_1 = 0.2\ d^{-1}$	$b_2 = 0.05\ d^{-1}$
Inactivation rate constant	$k_1 = 0.1\ d^{-1}$	$k_2 = 0.1\ d^{-1}$
Conversion factor	$\alpha_1 = 1$ g O_2/g COD	$\alpha_2 = 4.57$ g O_2/g N
Biomass density: $\rho_1 = \rho_2 = \rho_3 = 5000$ g COD/m³		

Table V. Initial conditions for biofilm growth.

Biofilm thickness	$L = 5 \times 10^{-6}$ m
Volume fraction of heterotrophs	$f_1 = 0.65$
Volume fraction of autotrophs	$f_2 = 0.35$
Volume fraction of inerts	$f_3 = 0.0$
Organics concentration	$S_1 = 3$ g COD/m³
Ammonium concentration	$S_2 = 13$ g N/m³
Dissolved oxygen concentration	$S_3 = 8$ g O_2/m³

Table IV. Substrate diffusion coefficients D within biofilm (80% of values D_i in water at 20°C).

Organics (acetate)	$D_1 = 83 \times 10^{-6}$ m²/d
Ammonium	$D_2 = 149 \times 10^{-6}$ m²/d
Dissolved oxygen	$D_3 = 175 \times 10^{-6}$ m²/d

3.5.2. Rotating Biological Contactor and Trickling Filter Processes

To the best of our knowledge, no dynamic models have yet been developed for the operation and control of either rotating biological contactors or trickling filters. There exist merely empirical models for design purposes:

> "The present state of knowledge is such that there is no single best design procedure or set of relationships that are universally applicable. No well defined theory of RBC design and operation is accepted by all RBC manufacturers ....Design engineers who have selected RBC technology are extremely dependent upon proprietors' design curves. The situation is compounded by the fact that each manufacturer has a different approach to media fabrication, configuration, shaft attachment, design and covers, etc." (Smith, 1987).

There has, however, been some assessment of the dynamic performance of these processes. For example, using a steady-state biofilm model developed by Ritmann and McCarty (1980), Tanaka et al. (1987) have simulated soluble BOD_5 removal patterns for municipal wastewaters. They tested their model against data collected at three different plants in Japan (daily data, 6-10 times each month over 6 to 12 month periods). Three basic processes are included in this model:

(i) Diffusion of substrate from the bulk liquid across a stagnant film and into the biofilm;

(ii) Substrate utilization by active bacteria within the biofilm; and

(iii) The growth and loss of biofilm mass.

With regard to the biological filter processes, James (1984) has proposed models for both the high-rate and the conventional trickling filter designs. Matters are simplified in the former because the high hydraulic loadings permit the assumption of a continuous feed pattern and enable the exclusion of biofilm growth kinetics (the film is assumed to have uniform, constant properties). For the trickling filter the feed pattern must be regarded as intermittent, giving rise to periods of quiescence in the filter bed alternating with periods of downward advective transport of the bulk fluid. As with the above RBC model, diffusion of the organic substrate from the bulk fluid to the biofilm is considered to be an important rate-limiting mechanism.

These models of the trickling filter, and RBC, however, are not dynamic (although James (1978) has referred to day-to-day simulation over a winter period), and it is important to speculate on the possible reasons for this significant omission. It may be due to: (i) the difficulty in modelling the biofilm (as previously discussed); (ii) the general belief in the relative stability and good performance of these processes; and (iii) the lack of internal control options.

This last point is perhaps most important. Control of the RBC and TF processes has been viewed largely as a matter handling merely contingencies (Wang et al., 1984). The plant operator has relatively little direct control of any of the process state variables, either independently or concurrently, with the result that he can hardly sense how the process is performing other than by checking the effluent quality (in the activated sludge process MLSS, DO, sludge blanket, etc., can be checked). In the absence of any form of internal control, the operator is limited to providing some external means of control, for example: altering the angular velocity of an RBC unit; changing the hydraulic loading of either an RBC or a TF unit; providing forced

draft ventilation of a TF unit or in-channel aeration of an RBC unit; or recycling effluent to decrease the influent soluble BOD concentration, if not the mass loading of organics (Bell, 1983). This apparent lack of control alternatives is no doubt a reason for the complete lack of control-oriented studies and thus an absence of the innovation of input-output time-series analysis.

3.5.3. Closing Remarks

The dynamic modelling of biofilm processes is still in its infancy compared with the developments in activated sludge modelling. Although further research has been undertaken and other dynamic models proposed since Grady's (1983) review, a complete and reliable model for biofilm dynamics, capable of emulating the role of the IAWPRC model for the activated sludge process, has not yet emerged. The models of Kissel et al. (1984), as well as those by Wanner and Gujer (1986) and Benefield and Molz (1985), all show promise, but are still incomplete and need urgently to be evaluated against field data. However, the operation of fixed-film processes is certainly not similar to that of the activated sludge processes, and there is ample scope for creative thinking on this point.

3.5.4. Humus Tank

Humus sludges are generated by the sloughing mechanisms occurring in fixed-film processes of trickling filters (TF) and biological contactors (RBC). The settling of these sludges is similar to the types 1 and 2 settling (discrete and flocculent) found in primary sedimentation tanks. An adequate performance of the final liquid-solid separation device for an RBC is assumed in most designs, although not always realized in practice (Lumbers, 1983). Takana et al. (1987) have found that approximately 30% of the RBC effluent solids were fine particles with a settling velocity of less

than 10-15 m/day. They considered that the small amount of flocculated SS in the RBC effluent could not entrap these fine particles, which remained in the clarifier effluent at an overflow rate of 10 to 15 $m^3/m^2/day$ and, as a result, contributed the majority of the secondary effluent total BOD_5. Humus sludge tanks are generally designed on the same basis as primary settlers (Metcalf & Eddy, 1979).

No dynamic models of humus tank performance have been found in the literature reviewed. Humus tanks are not normally suspected of giving rise to any particular operational problems, at least not when compared with secondary activated sludge settlers. For example, whereas the performance of secondary settlers for activated sludges during rain can be disastrous, this is not the case for the settling tanks following trickling filters (Henze, 1987). Given the hydraulic limitations in the construction of the trickling filter, humus tanks are unlikely ever to be overloaded; and even if overloading were to occur, the consequences should not be disastrous, since the influent suspended solids are themselves low (100-200 g/m^3) (Henze, 1987).

So again, the absence of serious operational difficulties may explain why so little interest in modelling has been focussed on these units. Yet the absence of operational difficulties does not mean that there is no scope for operational improvements.

3.6. Sludge Treatment

The unit processes of sludge treatment usually found in a sewage works are as follows: thickening (gravity and/or flotation), digestion (aerobic or anaerobic), dewatering, and final disposal. Much less research has been devoted to the dynamic behaviour of sludge treatment processes than to the liquid line processes. It could be said that almost exclusively the anaerobic digestion process has attracted the

attention of dynamic modellers; its reputation of being difficult to operate, and of frequently exhibiting process failures, is probably responsible for this interest. By comparison, no literature has been found on the dynamics of unit processes such as flotation, dewatering and final disposal, and thus only gravity thickening and biological digestion will be reviewed here.

3.6.1. Thickening

The sludge thickener (or consolidation tank) serves as an interface between the fluid and solids processing trains. Its purpose is to reduce the volume of sludge to be handled by the digester and to smooth out fluctuations in sludge flow-rate and concentration. It could also serve to improve the performance of primary sedimentation and the solids-liquid separator in the activated sludge process since a portion of the thickening function of these units can be transferred to the sludge thickener. The economic advantages of using sludge thickeners have been demonstrated by Price and Alder (1985) at the Avonmouth plant in the UK.

Two types of thickeners are in common use: the gravity thickener, for handling primary sludges or a mixture of primary and waste activated sludges; and flotation, generally used only for waste activated sludges. The operation of either is still very much an art, with neither dynamic models nor control strategies (based on fundamental principles) being available at present, although much of the basic theory of solids-liquid separation in the activated sludge process is also applicable to gravity thickening.

Thickening occurs primarily as a function of compression mechanisms, although zone settling can also be present. The compression zone can be defined as "that portion of the pulp where the flocs considered as integral bodies, have settled to a point where they rest directly one upon another. After pulp enters the compression zone, further separation of liquid must come through liquid pressed out of the flocs and the out of the interstitial

spaces between the flocs" (Coe and Clevenger, 1916).

Thickening theory (compression) and the different models used for the design of thickeners have been reviewed extensively by Fitch (1975) and Lockyear (1977). The approaches used to model the thickening process can be separated into four basic methods (Lockyear, 1977):

(i) the rate of release of water in the compression zone is proportional to the amount that can be released (empirical approach);

(ii) the sludge blanket consists of a large number of small tubules and tubes, and Poiseuille's Law for the flow of fluid through a capillary may be applied (rational approach);

(iii) the flow of liquid can be considered as though through a porous bed rather than through capillarities (Carman-Kozeny equations), and the effects of wall and sludge blanket can also be considered;

(iv) the mass and liquid force balances across the sludge blanket are considered.

Hoyland (1986) has used an approach similar to the latter to develop a model for the design of thickeners. However, no dynamic simulation of the process has been found in the literature, although models such as those by Hoyland (1986) and Kos (1977) could probably be used on a dynamic basis.

Such models were designed for estimating the quality of the underflow from the thickener, and there appear to be no models for simulating the behaviour of the supernatant effluent quality, which is usually recycled within the plant. The quality of the supernatant liquor can be important. If the liquor returned to

the head of the works has a high BOD and suspended solids content, then much of the advantage to be gained by thickening may be lost because of the increased load to the liquid train of the plant, which may give rise to increased sludge production during secondary treatment (Lockyear, 1978). Thus Stenstrom (1975) has used an empirical relationship to evaluate the solids concentration in the supernatant, although here again, as with the secondary clarifier, this kind of model may prove unsatisfactory since many phenomena can influence the supernatant quality (for example, sludge flotation due to denitrification). In short, given the role of the thickener as an interface between the liquid and solids processing trains, and the feedback of supernatant liquors to the liquid train, a knowledge of the thickener dynamics could be especially important to the control of sludge production and digestion within the plant. Such knowledge is not readily available at present.

3.6.2. Digestion

Digestion of sludges can be achieved either aerobically or anaerobically. Aerobic digestion is primarily used in small plants where its disadvantage of higher energy consumption is offset by its simplicity of operation and lower capital cost. Anaerobic digestion is mainly used when its capacity for energy production (methane gas) offsets its difficulty of control and operation.

(a) Aerobic digestion

Aerobic digestion of sludges became popular in the 1960's when energy sources were inexpensive and easily available. Its popularity accordingly decreased rapidly in the 1970s, and its counterpart, anaerobic digestion, became the preferred treatment process for sludge digestion.

> "A recent thrust in aerobic digestion development has arisen in the area of thermophilic aerobic digestion .... Continued effort in the area of aerobic digestion development may yield a significant breakthrough in reactor configuration and efficiency but in view of the more attractive current and potential energy status of its anaerobic counterpart and its recent capital costs reductions it is the author's view that aerobic systems still face a difficult task in achieving significant market penetration ... Whilst developments in aerobic systems rightfully continue, other papers at this symposium seem to however indicate a trend away from aerobic to anaerobic systems for both liquid and solids phase effluents". (Noone, 1986).

Wolinski (1985), having studied the practical implementation problems related to the use of aerobic digestion with air input, concluded that there are several problem areas that must be resolved before the process can be "optimized", primarily: (i) pumping costs; (ii) the reliability of the process in achieving 40% removal of the volatile material present; (iii) foam control; and (iv) the amenability of all sludges to aerobic, thermophilic treatment.

Very few dynamic models of the process have been discussed in the literature, although they have a relatively long history, deriving from the early work of Kambhu and Andrews (1969). These authors elaborated a simple model comprising three state variables: biodegradable volatile solids, dissolved oxygen concentration and heat content. Their results can only be considered semi-quantitative in nature since reliable values were not available at that time for the effect of temperature on process rate-constants. In addition, the assumption of first-order kinetics (for solids degradation) is unlikely to be valid at high rates of conversion of volatile solids since these residual compounds may be more resistant to biological degradation.

More recently, Warner et al. (1986) have shown that the activated sludge model of Dold et al. (1980) could be used to simulate the aerobic digestion of waste activated sludge, although they have not tackled the problem of thermophilic aerobic digestion. Using the same model, Dold et al. (1985) have developed control strategies capable of pH control without chemical addition and with cost savings in aeration requirements (the control comprises a sequence of alternating periods of aeration and non-aeration).

(b) Anaerobic Digestion

Anaerobic digestion of sludges is normally required to satisfy the following objectives (Frostell, 1985):

(i) consistent achievement of a high degree of waste stabilization;

(ii) as high a conversion as possible of waste to methane; and

(iii) as low a production as possible of excess solids at the highest possible concentration.

The functioning of the process involves three steps as shown in Figure 3.13. The fundamentals of the process have been recently reviewed extensively (Parkin and Owen, 1986; Harper and Poland, 1986), and will not be discussed here. It is sufficient merely to point out the following, taken from Parkin and Owen (1986).

> "The most important message to be extracted from this discussion is the importance of adequate SRT in anaerobic digestion. This fundamental point is not sufficiently realized by design and operation personnel. Proper attention to SRT helps to ensure the desired process efficiency and minimizes adverse effects caused by temperature fluctuations, introductions of toxic substances, variations in hydraulic and organic loadings, and changes

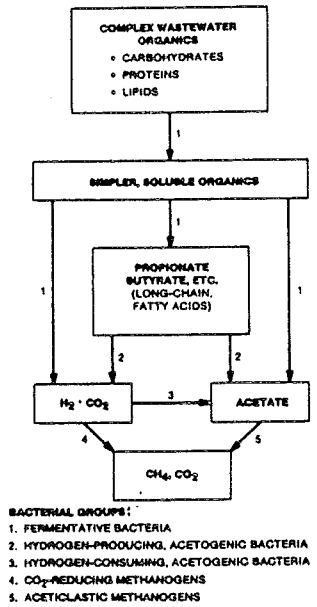


Figure 3.13 Methane formation in anaerobic digestion (from Parker and Owen, 1986).

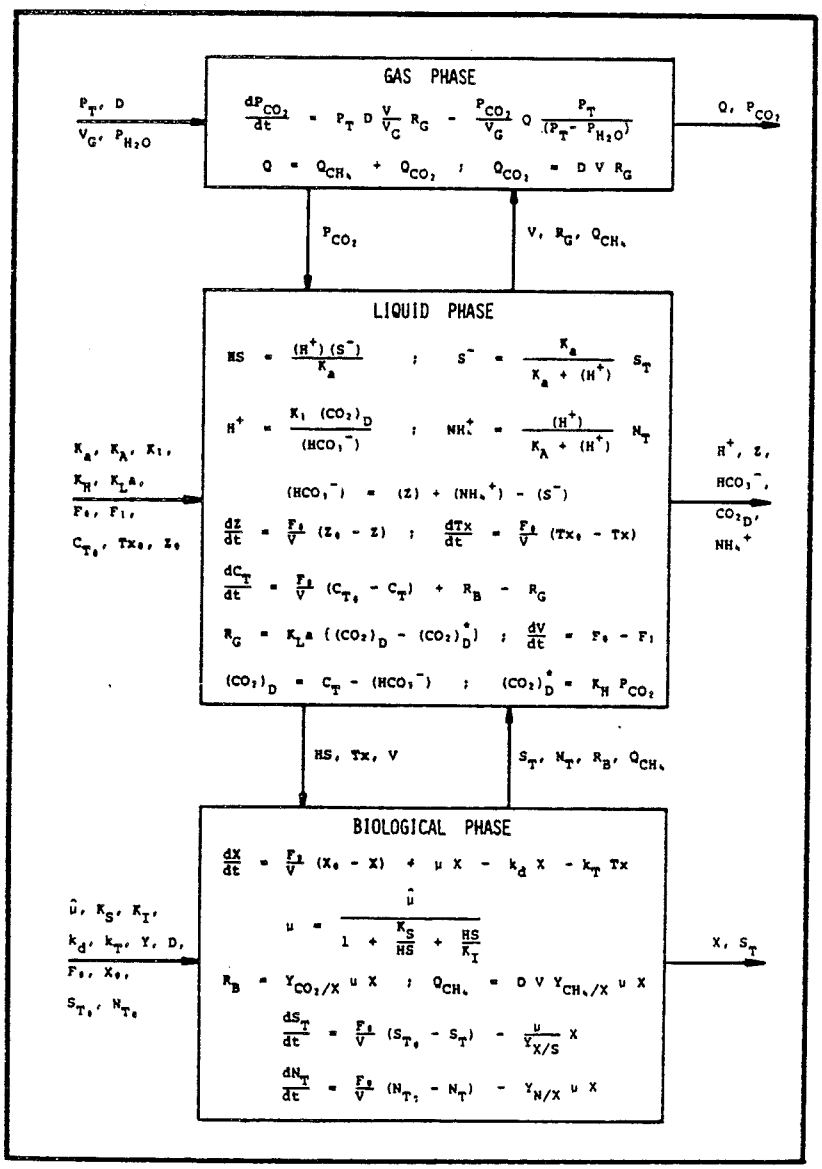


Figure 3.14 Anaerobic digester model (from Buhr and Andrews, 1977).

in feed sludge characteristics".

Despite the fact that SRT (solids retention time) is such a key parameter in anaerobic treatment, current designs of anaerobic digesters permit only very poor control of it (Frostell, 1985). A good control of SRT means that the operator should be able to increase or decrease the solids retention time as desirable, and this is at present not the case (Frostell, 1985). In practice, anaerobic process control is concerned primarily with pre-empting contingencies for which purpose dynamic modelling ought to be essential.

All the pioneering research on dynamic modelling of anaerobic digestion has been made by Andrews and his co-workers (Andrews, 1969; Graef and Andrews, 1974 a,b; Buhr and Andrews, 1977). The resulting model consisted of three phases (biological, liquid and gas), and eight state variables representing the organisms, substrate (volatile acids), nitrogen, cations-anions, total carbonic species, digester volume, conservative toxic material concentration and the partial pressure of CO_2 in the gas phase. The model is summarised in Figure 3.14.

Many authors have since used the Andrews model. Collins and Gilliland (1974), for example, have tested three different control strategies with it: (1) pH control by addition of a basic solution; (2) organism concentration control by recycle of digested sludge; and (3) influent flow-rate control with a holding tank. On the basis of experimental studies Carr and O'Donnel (1977) have proposed some minor modifications of the Andrews model, first, by adding a term to account for the effect of separation and retention of solids in the digester and, second, by inserting a lag-term in the cell growth expression. More recently, Rozzi (1984) has studied pH control strategies with a similar version of Andrews' model; his minor variations on the theme concern changes to the Monod kinetics, pH-inhibiting function, and the gas-liquid equilibrium characteristics.

The Andrews model considers that the digester feed has already been solubilized and converted to volatile acids, such that only the methanogenic stage is simulated. It is, therefore, a single-species model. Given the rapidity of this solubilization and volatile acid formation reaction, the methanogenic culture in the reactor, in effect, functions in an environment equivalent to that of a digester being fed a soluble volatile acid substrate (Graef and Andrews, 1974a). However, this simplification may not be appropriate in all cases and Hill and Barth (1980), in their studies of the digestion of animal wastes, have developed a two-species model composed of acid-formers (facultative heterotrophs) and methane-formers (obligate anaerobes). In the initial conversion of insoluble organics to soluble organics, ammonia and other cations are released from the raw waste, and since poultry waste typically has an abundance of cations and is particularly high in nitrogen, it was found necessary to simulate this initial conversion step. Other two-stage models have been proposed in various simulation and control studies (Kleinstreuer and Poweigha, 1982; Hirotsuji et al., 1985; Moletta et al., 1986).

Still more advanced than these models is the multiple-species, mixed-culture model of Torre and Stephanopoulos (1986). Its distinctive features are as follows:

(i) all the basic steps of anaerobic digestion are accounted for (hydrolysis-acidogenesis-methanogenesis):

(ii) commensal relationships between acid formers (lumped into a single species) and methane formers are identified, and the basic properties of commensalism are considered;

(iii) three distinct groups of methanogenic bacteria are identified, each acting on a different substrate (acetic, propionic, and butyric acids).

Torre and Stephanopooulos examined the customary pH control strategies, but in addition paid special attention to density control of the primary clarifier underflow as a means of controlling the digester feed (a point already noted and emphasised in Section 3.3).

For the most part these models have only been evaluated against data from bench-scale digesters. There has been little work with data from full-scale plant, and consequently few studies in time-series analysis. Some results for the gas production dynamics of the digesters at Norwich Sewage Works are reported elsewhere (Beck, 1986).

Andrews' model can still be considered the point of departure for contemporary modelling exercises, although additional features from the work of others could easily be incorporated, for example:

(i) reactions involving hydrogen (Mosey, 1983);

(ii) the concentrations of individual volatile acids as additional state variables (Buswell and Morgan, 1962);

(iii) solubilization of organic solids (Ferguson and Eastman, 1981); and

(iv) the effects of toxic material (Parkin and Speece, 1982).

3.7. The Plant as a Whole

The extent of this review is sufficient evidence of the relative abundance of dynamic models of the unit processes of wastewater treatment. In comparison there are few models linking all or even most of the treatment processes found in a conventional wastewater treatment plant (primary sedimentation, activated sludge process, disinfection, sludge thickening and digestion).

There are well known examples of such steady-state models, these having been developed for plant design and cost estimation, both capital and operational costs (WRc, 1981; USACE, 1980). Lindholm (1976) has modelled a combined sewer system and a conventional treatment plant in order to evaluate the plant's performance when receiving both sewage and runoff over given periods of the year. His plant model employed only empirical relationships to simulate the removal functions of the different processes, and therefore cannot really serve as a basis for elaborating dynamic control strategies.

Again, it is Andrews and his co-workers who have played a leading role in developing dynamic, lumped-parameter models of the plant as a whole (Bryant, 1972; Busby & Andrews, 1975; Stenstrom, 1977; Vitasovich & Andrews, 1987). Their models have usually described three or more unit processes, and these usually from the liquid line. Other wastewater treatment plant models have been prsented in the literature (e.g. Tanthapanichakoon and Himmelbau, 1981; Lech et al., 1978); have similarly accounted only for the processes of the liquid line; and have in any case been strongly based on the models of Andrews and his co-workers. Despite the importance of the interactions between unit processes within a plant, in particular between the solids and liquid treatment lines, there has thus been little work on the use of dynamic models for the development of overall plant control strategies.

All the models cited above, while they are necessary and interesting from a theoretical point of view, suffer from what might be called practical flaws. This is in no sense a denial of the value of previous research; such work has been crucial in the understanding of plant behaviour. However, since advances in modelling can appear so impressive, it is necessary to reiterate that there still exists a great gap between theory and practice.

4. CONCLUSIONS AND RECOMMENDATIONS

The objectives of this review have been straightforward. They were:

(i) To survey and assess the literature on the dynamic modelling of wastewater treatment processes;

(ii) To identify trends, gaps, strengths, and weaknesses in these past developments;

(iii) Accordingly, to make recommendations for future studies.

The review has been conducted within the overall context of river basin management.

In setting up the basic framework for the review in Section 2 it has been assumed that the primary concern in developing and applying dynamic models would normally be that of unit process operational control. Nevertheless, the survey has balanced this primary concern with the need to maintain an appreciation of the role of dynamic models in both the acquisition of basic understanding and the design of processes. The framework for model classification includes three broad sections:

(a) Model type;

(b) Model development and evaluation;

(c) Model application.

Given the context of river basin management, the scope for model application has deliberately been defined as greater than what might previously have been understood as the natural domain of treatment plant models. Our intention has been to illuminate more sharply the distinction between what has already been achieved and what remains to be done. The concept of operating wastewater treatment plants within the wider, global context of river basin management is a relatively recent innovation, and

some of the items thought desirable for model application must be viewed as somewhat speculative, and themselves matters for further research.

Section 3 is the review proper, but conditioned upon the framework of Section 2. According to the terms of reference, greatest emphasis has been placed on the liquid treatment line, that is: the plant influent; storm retention tanks; primary sedimentation; the activated sludge process; trickling filters; rotating biological contactors; secondary clarifier; and humus tanks. In this it has been necessary to assess in some detail the basic theories of suspended-growth and attached-growth biological systems, irrespective of particular plant configurations (e.g. contact-stabilization unit, oxidation ditch, trickling filter, biological contactor, and so forth). Tertiary treatment processes (filtration and biological processes) have not been included, and the processes of sludge treatment addressed only in relatively superficial terms. However, in addition to anaerobic digestion, aerobic digestion and the thickening aspects of sludge treatment have been discussed, the latter especially because of its role as a focal point of the interactions between the liquid and solid treatment lines.

Our conclusions and recommendations are organised as follows. First, overall progress in the development and application of dynamic models is summarised. This is presented in a matrix format: by unit process; and by model type, development, evaluation, and application. More specific details are then given on a unit-process by unit-process basis, and this is followed by comments on the various forms of model application. Lastly, we make a brief statement of the current programme of related research being undertaken (or planned) at Imperial College.

4.1. Overall Progress

The principal findings of the survey are summarised in Table 4.1. This is necessarily a simplification, and should be interpreted as a reflection of the relative balance of progress (and effort) for each unit process and (less precisely) across the spectrum of unit processes. A number of broad trends and gaps are apparent:

Table 4.1 Summary status of dynamic modelling: ● denotes substantial work; o denotes exploratory/preliminary studies.

	Storm Tanks	Primary Sed.	AS Carb. Oxidn.	AS Nitrification	AS Diss. Oxygen	AS Bulking	AS P Removal	AS Heavy metals	AS Clarification	AS Thickening
Model Type										
Linguistic							o			
Time-series		●	o		●		o		o	
Lumped-parameter	o	●	●	●	●		o	o		●
Distributed-parameter	o	o			●	●				●
Considerations of uncertainty				o	o					
Development and Evaluation										
Hypothesis Formulation	o	●	●	●	●	o	o	o	o	●
System Identification:										
(i) Semi-quantitative	o	●	●	●	●			o	o	●
(ii) Quantitative		●	o	●	●				o	o
Validation		o		o	●					
Assessment Data										
(i) Lab; pilot, full-scale		f	l,p,f	l,p,f	l,p,f			f	p,f	p,f
(ii) Cont., hourly, daily		h	c,h	h,d	c			h	c,h	c,h
(iii) Duration (<24h; >24h)		>24	<24	>24	<24			>24	>24	>24
Applications										
Target Operating Policies						o				o
Contingencies	o					o				o
Control System Design		o	●	o	●					
On-line Estimation/Forecasting			o	●	●				o	
Design-operation Interactions										
Training Simulators										

Table 4.1 (contd.) Summary status of dynamic modelling: ● denotes substantial work; o denotes exploratory/preliminary studies

	RBC	Trickling Filter	Humus Tank	Sludge Thickening	Aerobic Digestion	Anaerobic Digestion		Plant Influent
Model Type								
Linguistic						o		●
Time-series					●	●		o
Lumped-parameter	●	●						
Distributed-parameter								
Considerations of uncertainty								
Development and Evaluation								
Hypothesis Formulation	●	●			●	●		
System Identification:								
(i) Semi-quantitative					o	●		o
(ii) Quantitative						o		o
Validation						o		
Assessment Data								
(i) Lab, pilot, full-scale;					p	p,f		f
(ii) Cont., hourly, daily						c,h,d		h
(iii) Duration (>24h; <24h)						>24		>24
Applications								
Target Operating Policies		o						
Contingencies						o		
Control System Design						o		
On-line Estimation/Forecasting								●
Design-operation Interactions								
Training Simulators								

(i) Work on dynamic mathematical modelling has been dominated by an interest in the activated sludge process and, to a lesser extent, anaerobic digestion.

(ii) Of the dominant interest in the activated sludge process, arguably the most thorough and successful research has been that on the dynamics of the DO profile (at least with regard to practical control applications).

(iii) In spite of its operational importance, work on modelling the behaviour of bulking sludge remains largely at a theoretical/conceptual stage with a focus on detailed (micro-scale) distributed-parameter models that are ill-suited to control studies.

(iv) As for (iii), so the concentration of effort in the modelling of attached-growth processes (trickling filter, biological contactor) has been on hypothetical, distributed-parameter models, with an even more pronounced tendency towards considerable mathematical complexity.

(v) There has essentially been no work on the dynamic modelling of humus tanks and thickener performance, and little on the pollutant removal (as opposed to hydraulic) characteristics of storm retention tanks.

(vi) There has been relatively little progress in model validation (as defined in Section 2), and little work on the systematic identification of models with reference to observations from full-scale plant and for records of longer than 24h (although this latter is not so immediately apparent from Table 4.1).

(vii) Only a passing interest has been shown in considerations of uncertainty, and certainly nothing comparable with the substantial attention this issue has received in river and lake water quality modelling (Beck, 1987).

(viii) The dominant form of model application has undoubtedly been that of control system design for individual unit processes.

(ix) As expected, there has been little or no research on the application of models for the purposes of developing and evaluating target operating policies, contingency control, design-operation interactions, and personnel training simulators.

(x) There has been the first, speculative application of a linguistic model in connection with the removal of phosphorus in the activated sludge process.

Lastly, although not itemised in table 4.1, there is no existing model capable of successful simulation of plant behaviour as a whole, in spite of two or three attempts at achieving this by Andrews and his co-workers in the U.S.A. It is notable that Andrews now has funding from the (US) National Science Foundation to undertake comprehensive research on this problem, including integration of the treatment plant model with compatible models of the sewer network and receiving water.

4.2. Unit Processes

(a) Plant Influent

It is important to note first that in reviewing the models of plant influent variability it has been assumed that a comprehensive model of the dynamic behaviour of a sewer network would not necessarily be available, and the following conclusions and recommendations are consistent with this assumption. Thus:

(i) Univariate or input/output time-series models have been, and will continue to be, the only feasible, simple type of model of the plant influent perturbations. Such models are site-specific, and success with them has been restricted

almost exclusively to the simulation of flow variations. The normal procedure has been to represent these variations as a function of a known repeating cycle of dry-weather flows (diurnal, weekly patterns) and a rainfall-runoff response. The same features could be expected to be valid for the variations of the quality constituents, although other patterns (correlated with industrial production cycles) might be necessary for simulating perturbations in, for example, heavy metal concentrations.

(ii) Given that the models are site-specific, what is of strategic significance for the future is not so much the availability of a specific model but rather the availability of the software of time-series analysis for the rapid development of such models from on-site monitoring. There are many software packages for time-series analysis; almost all of those deriving from control theory are now based on recursive methods of estimation, and these would be essential to any work in the area of on-line estimation and forecasting (see point (iv) below).

(iii) There is, therefore, no substitute for the steady accumulation of a rich variety of experimental observations of the plant influent behaviour on a case-study by case-study basis. This would probably require a sufficient series of hourly, composite samples for the analysis of quality characteristics (SS, oxygen-demanding substances, ammonium-N, etc.). At present the response of the influent during precipitation events is especially poorly characterised, and it is worth noting that simulation studies of control strategies for stormwater surges will require the coordination of plant influent and upstream receiving water conditions (see also Howard et al, 1986).

(iv) The plant influent is an obvious target for on-line estimation/forecasting applications. This raises at least

two questions for further research: what forecasting horizons are feasible, and which are useful? Present capabilities may well be sufficient to answer the first of these questions (for flow, but not quality). Analysis of the second problem (usefulness) would be premature, it being dependent upon some prior assessment of suitable lead times for the practical arrangement of pumping schedules for moving sewage around the plant (see below).

(b) Storm Retention Tanks

An interest in dynamic models of the storm retention tanks implies an interest in the control of contingencies arising from storm events. It is significant to note therefore that little work has been done on this subject. Storm retention tanks are not part of customary U.S. teatment plant design practice, and where the behaviour of such tanks has been investigated, the objective has been to study flow equalization. Hence, our recommendations are as follows:

(i) For the immediate future, the extension of existing primary sedimentation models to the description of storm tank behaviour may suffice, at least for preliminary studies of storm sewage control strategies. Such models should be capable of handling issues associated with the probable nature of any storm tank effluent and with the timing of storm sewage diversions to the tank and subsequently of returns to the main stream of the plant. It would be especially important to establish whether storm tank returns are likely to have any impact on the plant other than in terms of hydraulic load, i.e. settled storm sewage might be identical in quality to normal settled sewage.

(ii) A modest study with the existing conceptual models of how a storm tank is likely to behave could clarify which among several problems require further, more substantial

research effort. One such candidate problem is that of quiescent solids settling, a process not presently incorporated in primary sedimentation models (for obvious reasons). The study might be conducted in the form of a sensitivity analysis of the model, and might also be used to formulate appropriate experimental sampling strategies (which undoubtedly are desirable).

(iii) Already, however, certain practical implications for both process operation and design are apparent. For example, operational sludge removel policies may, or may not, require the installation of a bottom scraper, and this in turn may make it desirable to return the storm tank liquors to either the head- or the tail-end of the primary sedimentation unit (see the appropriate discussion in Section 3).

(c) Primary Sedimentation

In view of the suspected complications of modelling the behaviour of a primary sedimentation unit, the success of our experience with the Norwich Sewage Works has been a satisfying surprise. Despite the unique nature of this case study, there are some general conclusions to be drawn:

(i) While the experiment at Norwich is well worth repeating (if not essential) elsewhere, the current primary sedimentation model is felt to be adequate for applications to the study of storm sewage control strategies. This position is likely to be strengthened given the expected results of further, more comprehensive analysis of the existing heavy-metals data-base for Norwich. Any model is always capable of improvement, and an obvious limitation of the present model is that no major storm event is included in the experimental record on which the model is based. Experimental studies elsewhere are possible, and our

discussions with the industry have revealed the potential to generate artificial hydraulic shocks. Nevertheless, for preliminary control studies, the present model should suffice.

(ii) The experiment at Norwich has confirmed (for us) the crucial role of specialised, intensive monitoring programmes in dynamic model development. It has also emphasised the possibly dominant role of seemingly "minor" recycle loops within a sewage works (in this case the return of liquors from sludge treatment); such loops may not be apparent from textbook descriptions of the process or pilot-scale experiments.

(iii) No model is at present capable of simulating the quality of the sludge produced in primary sedimentation or its thickening behaviour. Sludge quality is of obvious importance, for example, to the control of the anaerobic digestion process, and this, together with the feedback of sludge liquors, illustrates the more general importance of interactions between the solids and liquids treatment lines. These interactions merit research in their own right, although it is not clear whether they should be a priority issue for a research programme on river basin management.

(d) Activated Sludge: Aerator

The pre-eminence of the IAWPRC Task Group model of the activated sludge process, although having contributed substantially to developments hitherto, presents challenges of a different dimension for the future:

(i) To begin with, by its very existence the IAWPRC model has become a benchmark, against which the performance of all other (future) models is likely to be judged. It may also have the effect of retarding, if not

suppressing, the development of "competitor" models, especially if the computer software of the IAWPRC model were to be made readily available. There is a tendency for a commitment to the software to dominate over a commitment to the understanding of process behaviour, since the latter may require the inconvenience of a radical rethinking of basic hypotheses. The IAWPRC model is the first of its kind in wastewater treatment plant modelling. It is likely to stimulate a market for such "packaged" models, in much the same way as the US EPA's model QUAL-II has done for river quality modelling. The measure of control that these models can exercise can be gauged by recent criticism of our own work for not having used QUAL-II (a steady-state model) in a dynamic modelling context for which it was entirely inappropriate. The ease of access to such models can be gauged by the considerable degree of development work that is now being invested in an expert advisor interface for QUAL-II. An uncertainty analysis has already been incorporated into the ancillary software of QUAL-II, and an expert advisor for model calibration and verification is in hand (Barnwell et al, 1987).

(ii) In our opinion, the IAWPRC model has not been adequately or rigorously calibrated, in particular with regard to its description of the removal and metabolism of carbonaceous substrate (a process that has dominated the research leading up to the development of the model). This weakness is not unique. It is in part a function of the nature of the experimental observations. Most records normally available show relatively little variation in the MLSS and effluent BOD concentrations, and there is thus little dynamic information against which to identify dynamic process models. Moreover, when these records do exhibit significant temporal variation it is often as a result of problems with sludge settling and thickening in the secondary

clarifier unit, whose behaviour the IAWPRC model does not accurately quantify. In part this weakness is also a function of the complexity of the model structure, which contains a surplus of parameters to be calibrated against field observations. Indeed, the complexity of a model can for this reason be counter-productive in its "transferability" from one site to another (there is published evidence, for instance, of more than a year being necessary to adapt a U.S. EPA lake model to Finnish conditions).

(iii) Apart from the IAWPRC model, if it is accepted that sludge settling difficulties, and in particular bulking sludge, are key operational problems, then it can be argued that the dynamic modelling of bulking sludge behaviour <u>at the process scale</u> is a major challenge for the future. Put briefly, the challenge is to translate past fundamental research on the micro-scale behaviour of an individual piece of floc into the macro-scale terms of everyday process operational observations. This could well be a candidate problem for linguistic models and the ideas of qualitative simulation current, for example, in biomedical engineering. It could also incorporate the emerging concepts of two-species models (floc-forming and filamentous bacteria) and two-phase models (as for attached-growth systems), and utilise the algorithms of on-line state estimation. However, research of this nature is unlikely to yield practical dividends in the short term, and would probably be best implemented along the lines of a PhD thesis.

(iv) Much success has been obtained in the modelling of DO dynamics, especially inasmuch as this reflects the demand for aeration that would result from the various substrates undergoing model reactions. There seems,

conversely, however, to have been little work on how these reaction rates and degradation pathways would change as a result of deliberate or inadvertent variations in the DO regime. "Variations" in this sense could mean either variations in time of a point DO concentration, or variations of a spatial character as the organism/substrate mass circulates round the aerator/clarifier system. Aerobic, anaerobic, and anoxic conditions have already been considered in biofilm modelling, and there is potential here for a useful cross-fertilisation of ideas.

(v) Table 4.1 shows that there have been no applications of activated sludge models for the purposes of evaluating control strategies for handling storm sewage. The existing models, either the IAWPRC model or simplifications of it, should suffice for this purpose. The most important parts of the model with regard to storm sewage control would presumably be floc-uptake of suspended matter, organism washout, and the clarification and thickening behaviour of the secondary settler. The least secure of these hypotheses are without doubt the last two (on which further comments are made below). However, whatever the constraints this might impose in the immediate future, there are still a number of design configurations that can be tested and compared with regard to their ability to control storm sewage responses; these configurations include the step-feed aerator, the possibility of sludge storage, and even the concept of placing a small balancing tank between the aerator and clarifier.

(vi) Since the activated sludge process has been considered the focal point of operational control possibilities within a wastewater treatment plant, on-line estimation/forecasting of its influent variations must be considered as an important potential innovation. One Works manager has been heard to comment that

advanced warning of the effects of a stormwater surge would permit a pre-emptive rearrangement of aeration rates (and profiles). Such a capability for influent forecasting, based on a combination of existing plant influent time-series models and lumped-parameter models of the primary clarifier and storm tanks, should not take long to develop. However, we see the importance of this as secondary to the more strategic analysis of basic stormwater control options outlined in point (v) above.

(vii) Lastly, our experience at Norwich requires us to reiterate the need for models to be properly related to the actual nature of full-scale plant. For example, at Norwich there are effectively two recycle looops around the activated sludge unit: the normal recycle, and waste activated sludge returns to the settled sewage flow via a thickening/holding tank. This latter is significant, both in terms of substrate uptake prior to entry into the aerator and in terms of accurately simulating heavy metals removal.

(b) Activated Sludge: Secondary Clarifier

It has long been acknowledged that the complexity and apparent accuracy of activated sludge aerator models has not been matched by any comparable success in modelling the secondary clarifier. Since the interactions between the two component parts of the process are so strong, it is somewhat illogical for this imbalanced situation to have developed. Thus:

(i) The primary objective must be to conduct specialised monitoring exercises in which the behaviour of the aerator and the behaviour of the clarifier can be separately identified. This means comprehensive monitoring at the full-scale plant of the following streams: the influent settled sewage to the aerator;

the mixed liquor effluent passing from the aerator to the clarifier; the clarifier overflow; and the clarifier underflow. At the same time it is recommended that frequent and regular observations be made of the sludge blanket level, the SSVI, the DO concentration in both the blanket and the clarification zone, and (if at all possible) microscopic examination of the biological floc. This may seem an excessively demanding objective. Nevertheless, it is our firm belief that sooner or later such an exercise will become unavoidable if further progress is to be made.

(ii) In the meantime, current models of the clarifier appear to be more successful in their characterisation of the thickening function, as opposed to the clarification function. At present, it is our intention to work with a simplified lumped-parameter model of the clarifier (comprising four CSTR elements) for the simulation and evaluation of storm sewage control strategies. Despite the fact that rather complex distributed-parameter representations have often been proposed for the secondary clarifier, in our opinion considerable simplification is possible.

(iii) Such control strategies, specifically for manipulating the thickening and clarification performances of the clarifier, have not been widely discussed in the literature other than in the thesis of Hill (1985), which can accordingly be considered a useful starting point.

(f) <u>Trickling Filter and Rotating Biological Contactor</u>

Our survey of the literature has revealed a perhaps surprisingly wide and intensive interest in the theoretical aspects of biofilm modelling. The further progress of this work could profit from first some consideration of the following questions:

(i) Noting the extreme gap beween theory and practice, what precisely is the sigificance of the micro-scale biofilm models (accounting for film thickness and diffusion across the film boundary layer, for example) for the description of overall, macroscopic, input-output process behaviour? Indeed, even hourly field observations of input and output over a period of a few days would be a welcome piece of original research, and might reveal whether there is any dynamic behaviour of interest to be modelled. Moreover, if there are believed to be no operational control problems of major importance, why is so much work on biofilm models currently in progress?

(ii) In fact it has been argued by Powell and Selby (1987), and notably in the context of a "dynamic approach" to river basin management, that trickling filters can be operated in a number of modes, with switches of performance between these modes determined by short- and longer-term fluctuations in river conditions. This has great significance, being a comment from the industry; it also raises questions about the transient behaviour of processes between switches (and of possible transients in the start-up of other unit processes, such as sand filtration units in tertiary treatment).

(g) Humus Tanks

Given the complete absence of interest in the modelling of humus tank behaviour, there can only be one recommendation regarding this process. Like the trickling filter, its performance should be properly monitored in order to reveal the possibility of any dynamic behaviour of note. It is not in the nature of scientific enquiry for "stones to be left unturned", and the humus tank is no exception. It is, however, worth reflecting on whether it would be prefereable for an academic institute to follow up such enquiry

in the first instance.

4.3. Model Application

It has already been observed (from Table 4.1) that hitherto the most common form of dynamic model application has been for the purposes of (individual) unit process control system design, and that this has been almost exclusively restricted to the activated sludge process. These studies will continue to be integral to future developments. But since little has been said of the other forms of application, it is important to speculate on some strategic choices among them as priority areas for research, and in the following order:

(i) Contingencies (storm sewage control). In this case the issues are well defined, an adequate model of (notably) the complete liquid treatment line is available (but only for plants based on the activated sludge process), and there is a real practical possibility of implementing the resulting control strategies (since they depend essentially on the pumping of flows around the plant).

(ii) On-line Estimation/forecasting. This too is feasible, and according to the literature most appropriate in problems of forecasting influent variations and detecting sludge bulking and settleability characteristics. We are also aware of research recently begun at McMaster University in Canada on the use of the Kalman filter in anaerobic digester control (possibly in conjunction with the Canadian Wastewater Technology Centre). While attractive, we think these concepts should be secondary to the more strategic requirements of point (i) above. In particular, they raise certain very important practical questions: (a) does on-line estimation/forecasting yield information that is both unique and essential to the control function; and (b) is it realistic to imagine such sophisticated techniques being used on-site by the plant operating personnel? The water industry will not be the first to confront these questions. There is already work in progress in Finland on the development of

"user-friendly" interfaces for Kalman filtering software applied to industrial fermentation processes.

(iii) Target Operational Policies. Almost all control system design studies appear to have assumed a priori that it is known to what desirable set-points the performance of individual processes should conform. The basis of these assumptions will need a more formal re-assessment, and especially so in the light of seeking desirable performance specifications that can be coordinated across the plant as a whole to meet either effluent consent conditions or in-stream river conditions. In this global context (of integrating a plant model with a river model) it is not at all clear what the issues for analysis should be, and preparatory research to clarify them is much needed. There are, however, certain principles according to which this analysis might proceed, and certain methods with which to implement it. For example, it would seem best for the research to be problem-driven, on a case-study basis, with subsequent generalisation to a more coherent set of procedures. It is likely to require simplified model structures, e.g. dynamic models representing average daily behaviour, or steady-state models, and these simpler models might be embedded in formal optimisation algorithms for the derivation of optimal target operating policies. (Some work along these lines has been completed for the derivation of optimal aeration profiles for an industrial wastewater treatment process, Johnson and Voetter, 1985). It may accordingly be most acceptable to ally such research to a new concept of "best operational management practice".

(iv) Personnel Training. On this point we note merely in passing that we are aware of research on the development of personnel training simulators at McMaster University.

(v) Design-operation Interactions. Again, there is little more to be said on this other than to remark that it is a subject for long-term research, perhaps initially approached through the use of dynamic models to evaluate alternative plant configurations --

the locations and connections of tanks -- for , say, controlling storm sewage responses.

4.4. Programme of Research at Imperial College

Our situation at present can be summarised as follows. We have available a dynamic model of catchment hydrology and river water quality, which is capable of simulating both day-to-day and seasonal variations in water quality, and which in principle can be adapted to simulate behaviour over shorter time-scales. We have virtually completed a model for the whole of the liquid treatment line of an activated-sludge plant; this is geared to a basic time-scale of hourly variations. Complementing these two models there is software available for time-series analysis and Kalman filtering.

In the coming year we plan to carry out the following research:

(i) To complete the wastewater treatment plant model, in particular implementing the present conceptual model of storm tank operations.

(ii) To apply the treatment plant model for simulating and evaluating within-plant operating strategies for the control of storm sewage responses.

(iii) To couple the plant model with an adapted river model (working at an hourly time-scale) in order to evaluate jointly the in-plant and in-stream consequences of storm sewage control strategies (a study based loosely on the Derby/Derwent System).

(iv) To couple the plant model with the existing river model in order to assess the operational costs of seasonal nitrification policies (a study based specifically on the Milton Keynes/Bedford Ouse system).

(v) To construct a prototype expert system, or qualitative simulation model, for the bulking of activated sludge.

(vi) If suitable personnel are available, to begin a more comprehensive examination of the Norwich data base (COD, SS, VSS, NH_3-N, and heavy metals) using the existing software for time-series analysis and Kalman filtering.

In the longer-term, the following is planned:

(vii) The research as stated in our SERC proposal (re-submitted on October 15, 1987; see Appendix 3).

(viii) Subject to further funding by the U.S. National Science Foundation, continuation of the research itemised as point (iv) above.

(ix) Participation in the Rice University project (1988-1990), primarily (as presently understood) in the provision of advice on dynamic river water quality modelling.

REFERENCES

Abdel-Gawad, S. and McCorquodale, J.A., (1985), "Simulation of particle concentration distribution in primary clarifiers", Can. J. Civl Eng., 12 (3), pp. 454-463.

Adams, B.J. and Zukovs, G., (1987), "Probabilistic models for combined sewer system rehabilitation analysis", in Systems Analysis in Water Quality Management. (Ed. M.B. Beck), Advances in Water Pollution Control, Oxford: Pergamon, pp. 357-368.

Agnew, R.W., (1972), "A mathematical model of a final clarifier", Water Pollution Control Research Series, No. 17090 FJW 02/72, U.S. Environmental Protection Agency, Washington, D.C.

Alarie, R.L., McBean, E.A. and Farquahr, G.G., (1980), "Simulation modeling of primary clarifiers", J. Env. Eng. Div., Proc. Am. Soc. Civ. Eng., 106 (EE2), pp. 293-309.

Alarie, R.L., McBean, E.A. and Farquahr, G.J., (1982), "Simulation modeling of primary clarifiers", closure of discussion in J. Env. Eng. Div., Proc. Am. Soc. Civ. Eng., 108 (EE2), pp. 427-428.

Amandes, C.B. and Bedient, P.B., (1980), "Stormwater detention in developing watersheds", J. Env. Eng. Div., Proc. Am. Soc. Civ. Eng., 106 (EE2), pp. 403-419.

Anderson, H.M. and Edwards, R.V., (1981), " A finite differencing scheme for the dynamic simulation of continuous sedimentation", AIChE Symposium Series, Water-1980, no. 209, vol. 77, pp. 227-238.

Anderson, J.A., (1981), "Primary sedimentation of sewage", Wat. Pollut. Control, 80, pp. 413-420.

Andrews, J.F., (1969), "Dynamic model of the anaerobic digestion process", J. Sanit. Eng. Div., Proc. Am. Soc. Civ. Eng., 95 (SA1) pp. 95-116.

Andrews, J.F., (1974), "Dynamic models and control strategies for wastewater treatment processes", Wat. Res., 8, pp. 261-289.

Andrews, J.F., Stenstrom, M.K. and Buhr, H.O., (1976), "Control systems for the reduction of effluent variability from the activated sludge process", Prog. Wat. Tech., 8 (1), pp. 41-68.

Annachhatre, A.P. and Khanna, P., (1987), "Unsteady-state biofilm kinetics", J. Env. Eng., Proc. Am. Soc. Civ. Eng., 113 (2), pp. 429-433.

Arvin, E., Henze, M., Kristensen, G.H. and Petersen, G., (1985), "A model for biological phosphorus removal based on biological and chemical mechanisms", in Proceedings of International Conference on Management Strategies for Phosphorus in the Environment, Lisbon, Portugal, (Eds. J.N. Lester and P.W. Kirk), London: Selper, pp. 32-40.

Attir, U., Denn, M.M. and Petty, C.A., (1976), "Dynamic simulation of continuous sedimentation", AIChE Symposium Series, No. 167, vol. 73, pp. 49-54.

Ball, R.O., (1983), "The effects of hydraulic variation on fixed-film reactor performance", in Fixed-Film Biological Processes for Wastewater Treatment. (Eds. U.C. Wu and E.D. Smith), Noyes Data Corporation, Park Ridge, NJ, USA,pp. 174-205.

Barnwell, T.O., Jr., Brown, L.C., and Whittemore, R.C. (1987). "QUAL2E - A case study in water quality modeling software", in Systems Analysis in Water Quality Management (Ed. M.B. Beck), Advances in Water Pollution Control, Oxford: Pergamon, pp. 377-387.

Barror, R.F. and Schroeder, E.D., (1980), "Simulation modeling of primary clarifiers", discussion in J. Env. Eng. Div., Proc. Am. Soc. Civ. Eng., 106 (EE6), pp. 1209-1211.

Beck, M.B., (1977), "The identification and adaptive prediction of urban sewer flows", Int. J. Control, 25(3), pp. 425-440.

Beck, M.B., (1981), "Operational estimation and prediction of nitrification dynamics in the activated sludge process", Wat. Res., 15, pp. 1313-1330.

Beck, M.B., (1984), "Modelling and control studies of the activated sludge process at Norwich sewage works", Trans. Inst. Meas. Contr. 6(3), pp. 117-131.

Beck, M.B., (1986), "Identification, estimation and control of biological waste-water treatment processes", IEE Proceedings, Part D, 133(5), pp. 254-264.

Beck, M.B., (1987), "Water quality modeling: A review of the analysis of uncertainty", Wat. Resour. Res., 23(8), pp. 1393-1442.

Benefield, L.D. and Molz, F., (1983), "A kinetic model for the activated sludge proces which considers diffusion and reaction in the microbial floc", Biotech. Bioeng., 25, pp. 2591-2615.

Benefield, L.D. and Molz, F., (1984), "A model for the activated sludge process which considers wastewater characteristics, floc behaviour, and microbial population", Biotech. Bioeng., 26, pp. 352-361.

Benefield, L.D. and Molz, F., (1985), "Mathematical simulation of a biofilm process", Biotech. Bioeng., 27, pp. 921-931.

Berthouex, P.M., Hunter, W.G., Pallesen, L.C. and Shih, C.Y., (1975), "Modeling sewage treatment plant input BOD data", J. Env. Eng. Div., Proc. Am. Soc. Civ. Eng., 101 (EE1), pp. 127-138.

Berthouex, P.M., Hunter, W.G. and Pallesen, L., Shih, C.Y., (1978), "Dynamic behaviour of an activated sludge plant", Wat. Res., 12, pp. 957-972.

Berthouex, P.M. and Fan, R., (1986), "Evaluation of treatment plant performance: causes, frequency and duration of upsets", J. Wat. Pollut. Control Fed., 58(5), pp. 368-375.

Bode, H. and Seyfried, C.F., (1985), "Mixing and detention time distribution in activated sludge tanks", Wat. Sci. Tech., 17, Amsterdam, pp. 197-208.

Booman, K.A., Pallesen, L., and Berthouex, P.M. (1987), "Intervention analysis to estimate phosphorus loading shifts", in Systems Analysis in Water Quality Management (Ed. M.B. Beck), Advances in Water Pollution Control, Oxford: Pergamon, pp. 289-296.

Bryant, J.O., (1972), "Continuous time simulation of the conventional activated sludge wastewater renovation system", Ph.D. dissertation, Clemson University, Clemson, S.C., USA.

Buhr, H.O. and Andrews, J.F., (1977), "The thermophilic anaerobic digestion process", Wat. Res., 11, pp. 129-143.

Busby, J.B. and Andrews, J.F., (1975), "Dynamic modeling and control strategies for the activated sludge process", J. Wat. Pollut. Control Fed., 47 (5), pp. 1055-1080.

Buswell, A.M. and Morgan, G.B., (1962), "Paper chromatographic method for volatile acids - III. Toxicity of propionic acid", Proceedings 17th Purdue Industrial Confeence, pp. 377-378.

Carr, A.D. and O'Donnell, R.C., (1977), "The dynamic behaviour of an anaerobic digester", Prog. Wat. Tech., 9 (5/6), pp. 727-738.

Chan, R., (1987), "Dynamic modelling of heavy metals removal in the activated sludge process", M.Sc. Thesis, Department of Civil Engineering, Imperial College, London, UK.

Chapman, D., (1983), "The influence of process variables on secondary clarification", J. Wat. Pollut. Control Fed., 55 (1), pp. 1425-1434.

Chapman, D., (1985), "Final settler performance during transient loadings", J. Wat. Pollut. Control Fed., 57(3), pp. 227-234.

Clifft, R.C. and Andrews, J.F., (1981), "Predicting the dynamics of oxygen utilization in the activated sludge process", J. Wat. Pollut. Control Fed., 53(7), pp. 1219-1232.

Clifft, R.C. and Andrews, J.F., (1986), "Gas-liquid interactions in oxygen activated sludge", J. Env. Eng., Proc. Am. Soc. Civil Eng., 112 (1), pp. 61-77.

Coe, H.S. and Clevenger, G.H., (1916), "Methods for determining the capacities of slime-settling tanks", Trans. Am. Inst. Min. Eng., 55, pp. 356-384.

Collins, A.S. and Gilliland, B.E., (1974), "Control of anaerobic digestion process", J. Env. Eng. Div., Proc. Am. Soc. Civ. Eng., 100 (EE2), pp. 487-506.

Collins, M.A. and Crosby, R.M., (1980), "Impacts of flow variations on secondary clarifier performance", 53rd Annual WPCF Conference, Lasa Vegas, Nevada, USA.

Comeau, Y., Hall, K.J., Hancock, R.E.W. and Oldham, W.K., (1986), "Biochemical model for enhanced biological phosphorus removal", Wat. Res., 20(12), pp. 1511-1521.

Comeau, Y., Rabionwitz, B., Hall, K.J. and Oldham, W.K., (1987), "Phosphate release and uptake in enhanced biological phosphorus removal from wastewater", J. Wat. Pollut. Control Fed., 59(7), pp. 707-715.

Cook, S.C., (1983), "Systems identification and control of wastewater treatment processes in river management", PhD Thesis, Department of Civil Engineering, Imperial College, London, UK.

Corder, G.D. and Lee, P.L., (1986), "Feedforward control of a wastewater plant", Wat. Res., 20,(3), pp. 301-309.

Crosby, R.M., and Bender, J.H., (1984), "Hydraulic characteristics of activated sludge secondary clarifiers", Report USEPA, EPA-600/2-84-131, USA, 259pp.

Crowther, J.M., Dalrymple, J.F., Coackley, P. and Hamilton, I.M., (1977), "The use of time series models in the control of wastewater treatment plants", Prog. Wat. Tech., vol. 9 (5/6), pp. 579-586.

Curds, (1973), "A theoretical study of factors influencing the microbial population dynamics of the activated sludge process - I: The effects of diurnal variations of sewage and carnivorous ciliated protozoa", Water Research, 7, pp. 1269-1284.

Dalrympe, J.F. and Crowther, J.M. (1983), "Theory and practice of time domain techniques", in Mathematics in Microbiology (Ed. M. Bazin), London: Academic.

Debelak, K.A. and Sims, C.A., (1981), "Stochastic modeling of an industrial activated sludge process", Wat. Res., 15, pp. 1173-1183.

Di Toro, D.M. and Small, M.J., (1979), "Stormwater interception and storage", J. Env. Eng. Div., Proc. Am. Soc. Civ. Eng., 105(EE1), pp. 43-54.

Dick, R.I., (1970), "Role of activated sludge final settling tanks", J. San. Eng. Div., Proc. Am. Soc. Civ. Eng., 96 (SA2), pp. 423-436.

Ditsios, M., (1984), "Relationships between surface loading, depth and effluent suspended solids at a rectangular, horizontal, pilot scale secondary clarifier", Wat. Sci. Tech., 16, Vienna, pp. 293-302.

Dold, P.L., Ekama, G.A. and Marais, G.v.R., (1980), "A general model for the activated sludge process", Prog. Wat. Tech., 12, Toronto, pp. 47-77.

Dold, P.L., Buhr, H.A. and Marais, G.v.R., (1985a), "An equalization control strategy for activated sludge process control", Wat. Sci. Tech., 17, pp. 221-234.

Dold, P.L., Ekama, G.A. and Marais, G.v.R., (1985b), "pH control and costs savings in aerobic digetion", in Instrumentation and Control of Water and Wastewater Treatment and Transport Systems (Ed. R.A.R. Drake), Advances in Water Pollution Control, Oxford: Pergamon, pp. 375-381.

Dold, P.L. and Marais, G.v.R., (1986), "Evaluation of the general activated sludge model proposed by the IAWPRC task group", Wat. Sci. Tech., 18(6), pp. 63-90.

Eastman, J.A. and Ferguson, J.F., (1981), "Solubilization of particulate organic carbon during the acid phase of anaerobic digestion", J. Wat. Pollut. Control Fed., 53, pp. 352-366.

Ekama, G.A. and Marais, G.v.R., (1979), "Dynamic behaviour of the activated sludge process", J. Wat. Pollut. Control Fed., 53(3), pp. 534-556.

Ekama, G.A. and Marais, G.v.R., (1986a), "Sludge settleability and secondary settling tank design procedures", Wat. Pollut. Control, 85, pp. 101-113.

Ekama, G.A. and Marais, G.v.R., (1986b), "The implications of the IAWPRC hydrolysis hypothesis on low F/M bulking", Wat. Sci. Tech., 18 (6), pp. 11-19.

Ferrara, R.A. and Hildick-Smith, A., (1982), "A modeling approach for storm water quantity and quality control via detention basins", Wat. Resour. Bull., 18(6), pp. 975-981.

Fitch, B., (1975), "Current theory and thickener design (1975) - part 2, Compression theory", Filtration & Separation, 12, pp. 480-488.

Forster, D.M., (1985), "The dynamic modelling of the secondary settlement of activated sludge", MSc Thesis, Department of Civil Engineering, Imperial College, London, UK, 134p.

Frostell, B., (1985), "Process control in anaerobic wastewater treatment", Wat. Sci. Tech., 17, Tampere, pp. 173-189.

Gaudy, A.F. and Kincannon, D.F., (1977), "Comparing design models for activated sludge", Wat. Sewage Works, 124(2), pp. 66-70.

George, D.B. and Keinath, T.M., (1978)., "Dynamics of continuous thickening", J. Wat. Pollut. Control Fed., 50(11), pp. 2560-2572.

Ghobrial, F.H., (1978), "Importance of the clarification phase in the biological process control", Wat. Res., 12(11), pp. 1009-1016.

Goforth, G.F., Heaney, J.P. and Huber, W.C., (1983), "Comparison of basin performance modeling techniques", J. Env. Eng., Proc. Am. Soc. Civ. Eng., 109(5), pp. 1082-1098.

Grady, C.P.L., (1983), "Modeling of biological fixed films - A state-of-the-art review", in Fixed-Film Biological Processes for Wastewater Treatment. (Eds. Y.C. Wu and E.D. Smith), Noyes Data Corporation, Park Ridge, NJ, USA, pp. 73-134.

Grady, C.P.L., (1987), discussion of "Dynamic changes in population of the activated sludge community: effects of dissolved oxygen variations", in Wat. Sci. Technol., 19, Rio, pp. 1149-1151.

Graef, S.P. and Andrews, J.F., (1974a), "Mathematical modeling and control of anaerobic digestion", Water-1973, Am. Inst. Chem. Eng. Symp. Series 70, no. 136, pp. 101-131.

Graef, S.P. and Andrews, J.F., (1974b), "Stability and control of anaerobic digestion", J. Wat. Pollut. Control Fed., 46(4), pp. 666-683.

Gujer, W., (1977), "Design of a nitrifying activated sludge process with the aid of dynamic simulation", Prog. Wat. Tech., 9, pp. 323-336.

Gujer, W., (1987), "The significance of segregation of biomass in biofilm", Wat. Sci. Tech., 19, Rio, pp. 495-503.

Gullicks, H.A. and Cleasby, J.L., (1986), "Design of trickling filter nitrification towers", J. Wat. Pollut. Control Fed., 58(1), pp. 60-67.

Hansen, J.L., Fiok, A.E. and Hovious, J.C. (1980), "Dynamic modeling of industrial wastewater treatment plant data", J. Wat. Pollut. Control Fed., 52(7), pp. 1966-1975.

Harper, S.R. and Pohland, F.G., (1986), "Recent developments in hydrogen management during anaerobic biological wastewater treatment", Biotech. Bioeng., 28, pp. 585-602.

Hemain, J.C., (1986), "State of the art, Statistically Based Modelling of urban runoff quality", Proceedings of a NATO conference on Urban Runoff, Montpelier, France. (Eds. H. Torno, J. Marsalek and M. Desbordes), Springer Verlag, Heidelberg.

Henderson, R.J.A., McBain, W. and Pettigrew, F., (1981), "The performance of an off-sewer storm-sewage tank", Wat. Pollut. Cont., 80, pp. 582-599.

Henze, M., (1987), "Storm water handling in wastewater treatment plants", EWPCA-Symposium '87.7f. Europaisches Abwasser-und-Abfallsymposium, Munchen, May, pp. 173-194.

Henze, M., Grady, C.P.L., Gujer, W., Marais, G.v.R. and Matsuo, T., (1986), "Activated sludge model no. 1", Scientific and Technical Reports, no. 1, IAWPRC, London, UK, 33p.

Hermanowicz, S.W., (1987), "Dynamic changes in populations of the activated sludge community, effects of dissolved oxygen variations", Wat. Sci. Tech., 19, Rio, pp. 889-895.

Hill, D.T.. and Barth, C.L., (1977), "A dynamic model for simulation of animal waste digestion", J. Wat. Pollut. Control Fed., 49, pp. 2129-2143.

Hill, R.D., (1985), "Dynamics and control of solids-liquids separation in the activated sludge process", Ph.D. thesis, Rice University, Houston, Texas, USA, 275p.

Hirotsuji, J., Maeda, K., Tanaka, H. and Honda, T., "Two phase kinetic model and operational guide for anaerobic digestion process" in Instrumentation and Control of Water and Wastewater Treatment and Transport Systems (Ed. R.A.R. Drake), Advances in Water Pollution Control, Oxford: Pergamon, pp. 637-640.

Holmberg, A., (1982), "Modelling of the activated sludge process for microprocessor-based state estimation and control", Wat. Res., 16, pp. 1233-1246.

Holmberg, U. and Olsson, G. (1985), "Simultaneous on-line estimation of oxygen transfer rate and respiration rate", in Modelling and Control of Biotechnological Processes (Ed. A. Johnson), Preprints 1st IFAC Symposium, Oxford: Pergamon, pp. 185-189.

Hong, R., (1987), "Dynamic modelling of heavy metals removal in primary clarification", M.Sc. Thesis, Department of Civil Engineering, Imperial College, London, UK.

Hoyland, D., (1986), "A new design procedure for consolidation tanks", IChemE Symposium Series (G.B.), No. 96, pp. 119-136.

Howard, K., Saul, A.J., Thornton, R.C., and Pearson, L.G. (1986), "The effects of combined sewer overflows upon receiving waters".

Howell, J.A. and Atkinson, B., (1976), "Sloughing of microbial in trickling filters", Wat. Res., 10, pp. 307-315.

Howell, J.A. and Sodipo, B.O. (1985), "On-line respirometry and estimation of aeration efficiencies in an activated sludge aeration basin from dissolved oxygen measurements", in Modelling and Control of Biotechnological Processes (Ed. A. Johnson), Preprints 1st IFAC Symposium, Oxford: Pergamon, pp. 191-198.

Huber, W.C., (1986), "Deterministic modeling of urban runoff quality", Proceedings of a NATO Conference on Urban Runoff, Montpelier, France. (Eds. H. Torno, J. Marsalek and M. Desbordes), Springer Verlag, Heidelberg.

Hudgins, R.R. and Silveston, P.L., (1984), "Particle concentration predictions in settling basins", discussion in J. Env. Eng., Proc. Am. Soc. Civ. Eng., 110(4), pp. 862.

Iordache, O. and Corby, S., (1986), "A stochastic approach to sedimentation", Chem. Eng. Science, 41(10), pp. 2589-2593.

James, A. (1978), "An ecological model of percolating filters", in Mathematical Models in Water Pollution Control (Ed. A. James), Chichester: Wiley, pp. 303-318.

James, A., (1982), "Some perspectives in the modelling of biological treatment of wastewaters", Wat. Sci. Tech., 14, pp. 227-240.

James, A., (1984), "An Introduction to Water Quality Modeling", Chichester: Wiley, 234pp.

Johnson, A. and Voetter, M.H. (1985), "Optimization of the operating variables of an industrial waste-water treatment process", in Modelling and Control of Biotechnological Processes (Ed. A. Johnson). Preprints 1st IFAC Symposium, Oxford: Pergamon, pp. 205-210.

Jones, G.L., (1978), "A mathematical model for bacterial growth and substrate utilisation in the activated sludge process", in Mathematical Models in Water Pollution Control (Ed. A. James), Chichester: Wiley, pp. 265-279.

Jones, G.L. and Paskins, A.R., (1980), "Mathematical model of activated sludge. A comparison of predictions and experimental results", Technical Report, TR134, Water Reserach Centre.

Kambhu, R. and Andrews, J.F., (1969), "Aerobic thermophilic process for the biological treatment of wastes - simulation studies", J. Wat. Pollut. Control Fed., 41 (5 part 2), pp. r127-r141.

Khudenko, B.M., (1985), "Development and analysis of equalization basins", J. Env. Eng., Proc. Am. Soc. Civ. Eng., 111(6), pp. 907-922.

Kissel, J.C., (1986), "Modeling mass transfer in biological wastewater treatment processes", Wat. Sci. Tech., 18, Copenhagen, pp. 34-45.

Kissel, J.C., McCarty, P.L. and Street, R.L., (1984), "Numerical simulation of mixed-culture biofilm", J. Env. Eng., Proc. Am. Soc. Civ. Eng., 110(2), pp. 393-411.

Kleinstreuer, C. and Poweigha, T., (1982), "Dynamic simulator for anaerobic digestion processes" Biotech. Bioeng., 24, pp. 1941-1951.

Kos, P., (1977), "Continuous gravity thickening of sludges", Prog. Wat. Tech., 9, pp. 291-309.

Kuo, M.C., Fan, L.T. and Erickson, L.E., (1974), "Effects of suspended solids on primary clarifier size optimization", J. Wat. Pollut. Control Fed., 46(11), pp. 2521-2535.

Lau, A.O., Strom, P.F. and Jenkins, D., (1984), "The competitive growth of floc forming and filamentous bacteria: a model for activated sludge bulking", J. Wat. Pollut. Control Fed., 56(1), pp. 52-61.

Lech, R.F., Lim, H.C., Grady, C.P.L. and Koppel, L.B., (1978), "Automatic control of the activated sludge process -1. Development oa simplified dynamic model", Wat. Res., 12, pp. 81-90.

Lessard, P. and Beck, M.B., (1987), "Dynamic simulation of primary sedimentation", submitted to J. Env. Eng., Proc. Am. Soc. Civ. Eng.

Lester, J.N., (1983), "Singificance and behaviour of heavy metals in wastewater processes: 1. Sewage treatment and effluent discharge", Science Total Env., no. 30, pp. 1-44.

Levenspiel, O., (1972), "Chemical reaction engineering", New York: Wiley.

Lev, O., Rubin, E. and Sheintuch, M., (1986), "Steady state analysis of a continuous clarifier-thickener system", AIChE J., 32(9), pp. 1516-1525.

Lockyear, C.F., (1977), "Gravity thickening of biological sludges", Technical Report, TR 39, Water Research Centre.

Lockyear, C.F., (1978), "Pilot-scale experiments on continuous gravity thickening of activated sludge", Technical Report, TR 97, Water Research Centre.

Lumbers, J.P., (1983), Rotating biological contactors: current problems and potential developments in design and control", Pub. Health Eng., 11(3), pp. 41-45.

Lumley, D.G., (1985), "Settling of activated sludge: study of limiting factors and dynamic response", Publication 6:85, Chalmers University of Technology, Goteborg, Sweden, 116p.

Marais, G.v.R. and Ekama, G.A., (1976), "The activated sludge process, Part 1 - Steady state behaviour", Wat. SA, 2(4), pp. 164-200.

Marais, G.v.R., Loewenthal, R.E. and Siebritz, I.P., (1983), "Observations supporting phosphate removal by biological excess uptake - a review", Wat. Sci. Tech., 15, Capetown, pp. 15-41.

Medina, M.A., Huber, W.C. and Heaney, J.P., (1981a), "Modeling stormwater storage/treatment transients: Theory", J. Env. Eng. Div., Proc. Am. Soc. Civ. Eng., 107(EE4), pp. 781-797.

Medina, M.A., Huber, W.C. and Heaney, J.P., (1981b), "Modeling stormwater storage/teatment transients: Applications", J. Env. Eng. Div., Proc. Am. Soc. Civ. Eng., 107(EE4), pp. 799-816.

Metcalf & Eddy, (1979), "Wastewater engineering: treatment/disposal/ reuse", McGraw Hill, 2 ed., USA, 920p.

Moletta, R., Verrier, D. and Albagnac, G., (1986), "Dynamic modeling of anaerobic digestion", Wat. Res., 20(4), pp. 427-434.

Mosey, F.E., (1983), "Mathematical modeling of the anaerobic digestion process: regulatory mechanisms for the formation of short-chain volatile acids from glucose", Wat. Sci. Tech., 15, 209-232.

Moys, G.D., (1987), "Modelling of stormwater quality including tanks and overflows (MOSQITO) - Design specification", Hydraulics Research Limited, Wallingford, UK, report no. SR 127.

Mueller, J.A. and DiToro, D.M., (1983), "Combined sewer overflow characteristics from treatment plant data", U.S. Environmental Protection Agency, Report EPA-600/2-83-049.

Murphy, K.L., Sutton, P.M., and Jank, B.E., (1977), "Dynamic nature of nitrifying biological suspended growth systems", Prog. Wat. Tech., 9, pp. 279-290.

Muslu, Y., (1986a), "Distribution of retention times in biological filters, I: Mathematical modelling", J. Env. Eng., Proc. Am. Soc. Civ. Eng., 112(6), pp. 1122-1137.

Muslu, Y., (1986b), "Distribution of retention times in biological filters, II: Experimental confirmation", J. Env. Eng., Proc. Am. Soc. Civ. Eng., 112(6), pp. 1138-1150.

National Water Council, (1981), "Design and analysis of urban storm drainage - the Wallingford procedure", report no. 28.

Noone, G.P., (1986), "A review of sludge treatment processes", IChemE Symposium Series (G.B.), No. 96, pp. 101-117.

Olsson, G., (1977), "State of the art in sewage treatment plant control", AIChE Symposium Series, no. 159, 72, pp. 52-76.

Olsson, G., (1980), "Estimation and identification problems in wastewater treatment", in Real Time Forecasting/Control of Water Resources Systems. (Ed. E.F. Wood), Oxford: Pergamon, pp. 93-108.

Olsson, G., (1986), "Control strategies for the activated sludge process", in Comprehensive Biotechnology (Ed. M. Moo-Young), Oxford: Pergamon, Chap. 65, pp. 1107-1119.

Olsson, G., (1987), "Dynamic modelling, estimation and control of activated sludge systems", in Systems Analysis in Water Quality Management (Ed. M.B. Beck), Advances in Water Pollution Control, Oxford: Pergamon, pp. 349-356.

Olsson, G. and Andrews, J.F. (1978), "The dissolved oxygen profile - a valuable tool for control of the activated sludge process", Water Research, 12, pp. 985-1004.

Olsson, G. and Chapman, D., (1985), "Modeling the dynamics of clarifier behaviour in activated sludge systems", in Instrumentation and Control of Water and Wastewater Treatment and Transport Systems (Ed. R.A.R. Drake), Advances in Water Pollution Control, Oxford: Pergamon, pp. 405-412.

Olsson, G., Holmberg, U. and Wikstrom, A., (1985), "A model library for dynamic simulation of activated sludge systems", in Instrumentation and Control of Water and Wastewater Treatment and Transport Systems (Ed. R.A.R. Drake), Advances in Water Pollution Control, Oxford: Pergamon, pp. 721-728.

Olsson, G., Stephenson, J. and Chapman, D., (1986), "Computer detection of the impact of hydraulic shocks on plant performance", J.Wat. Pollut. Control Fed., 58(10), pp. 954-959.

Parkin, G.F. and Speece, R.E., (1982), Modeling toxicity in methane fermentation systems", J. Env. Eng. Div., Proc. Am. Soc. Civ. Eng., 108(EE3), pp. 515-531.

Parkin, G.F. and Owen, W.F., (1986), "Fundamentals of anaerobic digestion of wastewater sludges", J. Env. Eng., Proc. Am. Soc. Civ. Eng., 112(5), pp. 867-920.

Patry, G.G., (1987), "Recursive water quality forecasting models for urban catchments", Can. J. Civil Eng., 14, pp. 221-229.

Perdrieux, S. and Therien, N., (1980), "Modelling the dynamics of the activated sludge wastewater processes in terms of the carbon variable", Wat. Res., 14, pp. 1333-1344.

Pflanz, P., (1969), "Performance of secondary sedimentation basins", in Advances in Water Pollution Research (Ed. S.H. Jenkins), Oxford: Pergamon, pp. 569-581.

Pierce, D.M., (1960), "The primary sedimentation tank - Control center of the treatment plant", J. Wat. Pollut. Control Fed., 32(9), pp. 1015-1022.

Pinheira, M. and Camara, A., (1985), "TROPHIT - A Linguistic based system for improved biological phosphorus removal", in Proceedings International Conference on Management Strategies for Phosphorus in the Environment, Lisbon, Portugal, (Eds. J.N. Lester and P.W. Kirk), London: Selper, pp. 394-398.

Poduska, R.A. and Andrews, J.F., (1975), "Dynamics of nitrification in the activated sludge process", J. Wat. Pollut. Control Fed., 47(11), pp. 2599-2619.

Powell, E. and Selby, K. (1987), "The potential for the application of a dynamic approach to the management of river systems", Water Pollution Control, 86(1), pp. 43-50.

Price, G.J. and Alder, M., (1985), "The economic advantages of sludge thickening at Avonmouth sewage treatment works", Wat. Pollut. Control, 84, pp. 394-406.

Randall, C.W., Ellis, K., Grizzard, T.J. and Knocke, W.R., (1982), "Urban runoff pollutant removal by sedimentation", in Proceedings of the Conference on Stormwater Detention Facilities (Ed. W. de Groot), ASCE Publication, New York, U.S.A.

Rittmann, B.E. and McCarthy, P.L., (1980), "A model of steady state biofilm kinetics", Biotechnol. Bioeng., 22, pp 2243-2357.

Rozzi, A., (1984), "Modelling and control of anaerobic digestion processes", Trans. Inst. Meas. Control, 6(3), pp. 153-159.

Sarner, E., (1986), "Removal of particulate and dissolved organics in aerobic fixed-film biological processes", J. Wat. Pollut. Control Fed., 58(2), pp. 165-172.

Schilling, W., (1985), "A survey on real time control of combined sewer systems in the United States and Canada", in Instrumentation and Control of Water and Wastewater Treatment and Transport Systems (Ed. R.A.R. Drake), Advances in Water Pollution Control, Oxford: Pergamon, pp. 595-600.

Schramber, D.R. and Larock, B.E., (1983), "Particle concentration predictions in settling basins", J. Env. Eng., Proc. Am. Soc. Civ. Eng., 109(3), pp. 753-862.

Sheffer, M.S., Hiraoka, M. and Tsumura, K., (1984), "Flexible modelling of the activated sludge system - theoretical and practical aspects", Wat. Sci. Tech., 17, Amsterdam, pp. 247-258.

Shiba, S., Inoue, Y. and Ueda, Y., (1979), "Dynamic estimation of settling basin performance", J. Env. Eng., Proc. Am. Soc. Civ. Eng., 105(EE6), pp. 1113-1129.

Siebritz, I.P., Ekama, G.A. and Marais, G.v.R., (1983), "A parametric model for biological excess phosphorus removal", Wat. Sci. Tech., 17, Capetown, pp. 127-152.

Smith, E.D., (1987), discussion of "Performance evaluation of rotating biological contactor process", in Wat. Sci. Tech., 19, Rio, pp. 1108-1109.

Smollen, M. and Ekama, G.A., (1984), "Comparison of empirical settling velocity equations in flux theory for secondary settling tanks", Wat. SA, 10(4), pp. 175-184.

Speitel, G.E. and DiGiano, F.A., (1987), "Biofilm under dynamic conditions", J. Env. Eng., Proc. Am. Soc. Civ. Eng., 113(3), pp. 464-475.

Stehfest, H., (1984), "An operational dynamic model of the final clarifier", Trans. Inst. Meas. Contr., 6(3), pp. 160-164.

Stenstrom, M.K., (1975), "A dynamic model and computer compatible control strategies for wastewater treatment plants", Ph.D. Thesis, Clemson University, Clemson, S.C., USA.

Stenstrom, M.K. and Pokuska, R.A., (1980), "The effect of dissolved oxygen concentration on nitrification", Wat. Res., 14, pp. 643-649.

Stephenson, T.N., and Lester, J.N., (1987), "Heavy metals behaviour during the activated sludge process: 1. Extent of soluble and insoluble metal removal, 2. Insoluble metal removal mechanisms", Sci. Total Environment, no. 63, pp. 199-230.

Stoltenberg, D.H. and Horvatin, P.J., (1987), "Results from Great Lakes POTW optimization project", J. Env. Eng., Am. Soc. Civ. Eng., 113(4), pp. 736-751.

Strom, P.F. and Jenkins, D., (1984), "Identification and significance of filamentous microorganisms in activated sludge", J. Wat. Pollut. Control Fed., 56(5), pp. 449-459.

Takamatsu, T., Shioya, S. and Kurome, H., (1984), "Dynamics and control of an activated sludge process as a mixed culture system", AIChE J., 30(3), pp. 368-376.

Tanaka, K., Oshima, A. and Rittmann, B.E., (1987), "Performance evaluation of rotating biological contactor processes", Wat. Sci. Tech., 19, Rio, pp. 483-494.

Tay, J.H., (1982), "Development of a settling model for primary settling tanks", Wat. Res., 16, pp. 1413-1417.

Tay, J.H. and Heinke, G.W., (1983), "Velocity and suspended solids distribution in settling tanks", J. Wat. Pollut. Control Fed., 55(3), pp. 261-269.

Tanthapanichakoon, W. and Himmelbau, D.M., (1981), "Simulation of a time dependent activated sludge wastewater treatment plant", Wat. Res., 15, pp. 1185-1195.

Tebbutt, T.H.Y. and Christoulas, D.G., (1975), "Performance relationships for primary sedimentation", Wat. Res., 9, pp. 347-356.

Tetreault, M.J., Benedict, A.H., Kaempfer, C. and Barth, D.F., (1986), "Biological phosphorus removal: A technology evaluation", J. Wat. Pollut. Control Fed., 58(8), pp. 825-837.

Therien, N., Marcos, B., Muratet, G. and Domenbech, S., (1986), "Amelioration des performances du procede de boues activees par l'ajustement temporel du debit de l'effluent d'un bassin d'egalisation", Revue International des Sciences de l'Eau, 2(3), pp. 79-89.

Torre, A.D. and Stephanopoulos, G., (1986), "Simulation study of anaerobic digestion control", Biotech. Bioeng., 28, pp. 1138-1153.

Tracy, K.D. and Keinath, T.M., (1974), "Dynamic model for thickening of activated sludge", AIChE Symposium Series, No. 136, 70, pp. 291-308.

Trulear, M.G. and Characklis, W.G., (1982), "Dynamics of biofilm processes", J. Wat. Pollut. Control Fed., 54(9), pp. 1288-1301.

Tuntoolavest, M., Miller, E., and Grady, D.P., (1983), "Factors affecting the clarification performance of activated sludge final settlers", J. Wat. Pollut. Control Fed., 55(3), pp. 234-248.

USACE, (1980), "Design of Wastewater Treatment Facilities ". CAPDET. Program user guide, Dept. of the Army, Washington, DC, USA, EM 1110-1-501, Part III.

van Haandel, A.C., Ekama, G.A. and Marais, G.v.R., (1981), "The activated sludge process(3); single sludge denitrification", Wat. Res., 15, pp. 1135-1152.

von der Emde, W., (1972), "To what extent are primary tanks required?", Wat. Res., 6, pp. 395-396.

Vitasovich, Z. and Andrews, J.F., (1987), "A rule-based control system for the activated sludge process", in Systems Analysis in Water Quality Management (Ed. M.B. Beck), Advances in Water Pollution Control, Oxford: Pergamon, pp. 423-432.

Voshel, D. and Sak, J.G., (1968), "Effect of primary effluent suspended solids and BOD on activated sludge production", J. Wat. Pollut. Control Fed., 40(5-part 2), pp. r203-r212.

Wang, K., Wang, M.H. and Poon, C.P.C., (1984), "Rotating biological contactors", Eff. Wat. Treatment J., 24(3), pp. 93-97.

Wanner, O., Gujer, W., (1985), "Competition in biofilms", Wat. Sci. Tech., 17, Amsterdam, pp. 27-44.

Wanner, O. and Gujer, W., (1986), "A multispecies biofilm model", Biotech. Bioeng., 28, pp. 314-328.

Warner, A.P.C., Ekama, G.A. and Marais, G.v.R., (1986), "The activated sludge process, part 4. Application of the general kinetic model to anoxic-aerobic digestion of waste activated sludge", Wat. Res., 20, pp. 943-958.

Wentzel, M.C., Dold, P.L., Ekama, G.A. and Marais, G.v.R., (1985), "Kinetics of biological phosphorus release", Wat. Sci. Tech., 17(11/12), pp. 57-73.

Wishart, S.J., (1986), "Dynamic modelling of oxidation ditch solids", M.Sc. Thesis, Department of Civil Engineering, Imperial College, London.

WRc (1981), "Sewage treatment optimisation model - User manual and description", Technical Report, TR144, Water Research Centre.

Wolinski, W.K., (1985), "Aerobic thermophilic sludge stabilization using air", Wat. Pollut. Control, 84, pp. 433-445.

APPENDIX 1: "Identification, Estimation and Control of Biological Waste-water Treatment Processes", IEE Proceedings, 133, Part D, No. 5, pp. 254-264.

Identification, estimation and control of biological waste-water treatment processes

M.B. Beck, B.Sc., M.A., Ph.D.

Indexing terms: *Biomass, Control equipment and applications, Biotechnology*

Abstract: The biological processes of waste-water treatment have often been regarded in practice as largely self-controlled and somewhat inflexible in their operation. The paper reviews some of the reasons why it has been difficult to apply the methods of control theory to these processes, and what the future role of control in waste-water treatment might be. Attention is restricted to three processes: the activated sludge process, the oxidation ditch and anaerobic digestion. In contrast to many biotechnical processes, these processes are characterised by a heterogeneous culture of micro-organisms and by multiple substrates degraded along several reaction pathways. Their dynamics are extremely difficult to identify, their observed input/output relationships frequently exhibiting substantial stability punctuated by abrupt instabilities. Other aspects of their behaviour are relatively well defined, in particular the dynamics of dissolved oxygen in the activated sludge process, and have been equally well studied from the point of view of online state-parameter estimation and adaptive control. There is still, however, a large gap between practice and theory, and, in looking briefly at an agenda of problems for the future, the paper places special emphasis on, *inter alia*, issues of fault detection and diagnosis and the use of plant operator experience in generating novel approaches to control.

1 Introduction

In spite of a long history of practical experience (e.g. Reference 1) and well over a decade of academic study (e.g. Reference 2), the control of biological waste-water treatment process dynamics seems still to be an elusive object. There is, and must be, some form of 'control' of these processes, but it is not the same control as would usually be understood by the control engineer. Such control as there is, is essentially influenced by the personal expertise of the plant manager/operator. It occasionally incorporates one or two automatic, single-loop controllers, which regulate a fraction of the process state and some of the faster modes of response. All in all, one could easily imagine that things seem to work fairly well, and of their own accord, until abrupt failure occurs.

So there are clearly control problems to be solved, and there are many reasons for persevering with their solution. The policy issues of waste-water treatment plant operation have been widely discussed, and most have argued a case in favour of its desirability and inevitability [3–7]. If therefore the control of biological treatment processes seems both possible and desirable, why has it not come about? There are no new answers to this question, merely those that are already only too familiar [8]: a lack of online instruments; a widely held view that the available instruments are not reliable; a lack of a truly compelling output performance specification for the exercise of operational control; and a limited technological capacity with which to implement control actions (waste-water treatment plant designs are generally regarded as intrinsically inflexible). Accurate process models simply do not exist in the same sense as they do in electrical, mechanical, aeronautical, and (less so) chemical engineering systems. Nor are control concepts in general taught to civil engineering students, the traditional source of professional staffing of the water industry.

In short, there are scant grounds for believing that the straightforward application of control theory to the analysis of waste-water treatment processes, including adaptive and self-tuning control, will yield practical dividends. But the problems are not intractable, and indeed the position of this paper would be untenable if it were assumed that they are so. It is rather that the application of control theory may have to be devious, and that the sort of theory applied may not necessarily be conventional. This may well be true of biotechnology more generally, and a recent limited survey has lent support to the view that the difficult problems of waste-water treatment have spawned some quite novel approaches to process modelling and control [9]. However, the problems of controlling biological waste-water treatment processes must be acknowledged, in some important respects, as quite distinctive. They have much to do with excessive input variability, a dependence on a heterogeneous culture of organisms, and an industry whose motivation to innovate has not been high (as discussed by Tomlinson *et al.* [10]).

The paper is divided into five Sections dealing, in turn, with process description, identification, state estimation, control and, lastly, the questions arising from a review of these topics. In particular, we shall be concerned to elicit the distinction between the relatively well defined and relatively ill defined facets of process dynamics. From other perspectives, it might be said that this distinction is more or less synonymous with the distinctions between quantitative and qualitative process knowledge, stability and instability, the dominant fast and slow modes of process dynamics, the macroscopic and microscopic state of the process, and the nonbiological and biological elements of the state vector. These distinctions, it is argued, are central to the way in which control systems are to be designed for biological waste-water treatment processes.

2 The processes

We shall restrict the discussion to three unit treatment processes: the activated sludge process, the oxidation ditch and anaerobic digestion. Of these, the first has received by far the most study, from the point of view of process dynamics and control, and unless otherwise stated or apparent, it will be this process to which the discussion refers.

Paper 4764D (C9, C11), first received 7th November 1985 and in revised form 30th June 1986

The author is with the Department of Civil Engineering, Imperial College of Science and Technology, Imperial College Road, London SW7 2BU, United Kingdom

An elementary conceptualisation of waste-water treatment is given in Fig. 1. In the very simplest terms, the

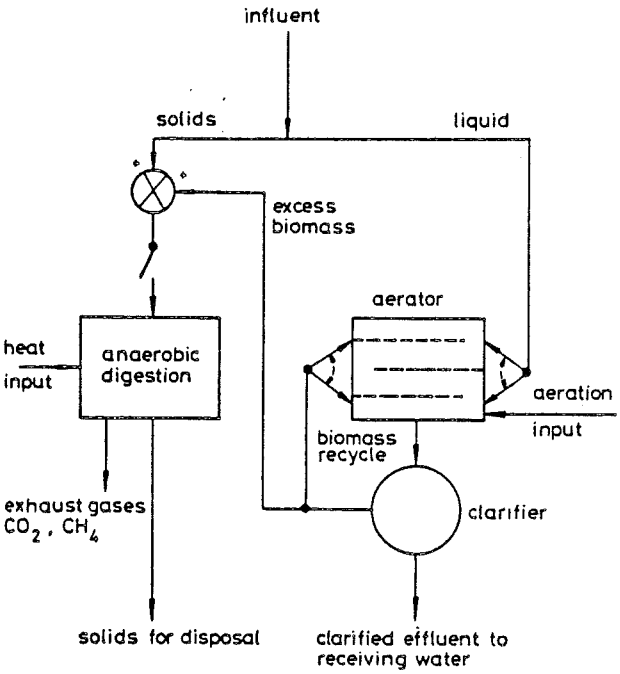


Fig. 1 *Basic functions of a waste-water treatment plant*

overall objective of the treatment plant is to separate the liquid and solids fluxes that enter the plant. It then follows that the liquid effluent should be as free of pollutants as possible and that the solids for ultimate disposal should be biologically 'stable', contain as little water as possible, and be free of pathogenic micro-organisms. The activated-sludge and oxidation-ditch processes are two alternatives for the biological treatment of the liquid flux; their 'mixed liquors' receive aeration and they are aerobic processes. Anaerobic digestion refers to the treatment of solid wastes (although, in fact, about 95% of the slurry is water). All three processes operate on the principle that a microbial biomass can be grown on the waste substrates, and, in so doing, complex organic waste materials are broken down into simple end products such as water, carbon dioxide, and, in the case of anaerobic digestion, methane. In both of the liquid treatment processes the biomass itself should not be allowed to pass out in the final clarified effluent and must be separated and transferred to treatment as a solid waste.

The interaction between biomass and substrate is clearly fundamental to the performance of these treatment processes, as it is in biotechnical processes in the broader sense. However, rarely would the objective of process control be to maximise biomass production (which it might be elsewhere), except in so far as this bestows stability of performance in attenuating large perturbations in the input disturbances. Indeed, excessive biomass production is distinctly undesirable in the oxidation ditch and activated sludge processes, as this creates additional volumes of solid wastes to be disposed of. It is more usual, therefore, to seek a maximal removal of waste substrate, to minimise operating costs, and to maintain a favourable physical and chemical environment in which the various microbial populations can prosper. As such, this favourable environment would normally be defined in terms of pH, temperature, dissolved oxygen (DO) concentration and mixing characteristics; to some considerable extent it determines the functioning of the biological cell matter.

One of the distinctive features (and one of the most difficult problems) of waste-water treatment plant control is that the biomass and the substrate are heterogeneous quantities. Several types of substrate are undergoing degradation by several different microbial populations in several stages and usually in parallel with one another, as Table 1 shows. A favourable environment for one population can be unfavourable to the other populations.

Table 1: Principal reactions and degradation pathways of biological waste-water treatment processes

Aerobic			Anaerobic
Primary objective	Secondary objective	Tertiary objective	Primary objective
(1) *Carbonaceous substrate degradation*	(2) *Nitrification*	(3) *Denitrification*	(1) *Carbonaceous substrate degradation*
substrate capture	ammonium-N	nitrate-N	hydrolysis
↓	↓	↓	↓
metabolism	nitrite-N	nitrogen gas	volatile acid generation
↓	↓		↓
water, carbon dioxide	nitrate-N	(4) *Phosphorus removal* (mainly chemical precipitation)	(acid conversion)
			↓
			methane, carbon dioxide

Moreover, complex predator-prey (food-web) interactions may exist within these heterogeneous cultures, and quite antipathetic environments may be required in one and the same process. For example, to achieve nitrification in the aerator, a relatively high DO level is necessary; yet to achieve denitrification in the aerator (to suppress its occurrence in the clarifier, where it is detrimental to sludge settling and separation), an anaerobic environment must, in part, prevail.

2.1 Process models

We shall assume that, in general, any of the three biological treatment processes can be represented by the following lumped-parameter state-space model:

$$\begin{bmatrix} \dot{x}_b(t) \\ \dot{x}_e(t) \end{bmatrix} = f\{x_b, x_e, u, d, \alpha; t\} + \xi(t) \tag{1a}$$

with sampled observations

$$y(t_k) = h\{x_b, x_e, \alpha; t_k\} + \eta(t_k) \tag{1b}$$

where x_b is that portion of the state vector representing the biological state of the process, x_e is the state of the physical/chemical environment of the biomass, u is a vector of input control variables, d is a vector of measured input disturbances, y is a vector of observed outputs, ξ is a vector of random, unmeasured input process disturbances, η is a vector of random output measurement errors, α is a vector of model parameters and t_k is the kth discrete sampling instant in time t.

The environment x_e subsumes the definition of the substrate x_s and product x_p concentrations. For practical purposes, the controls u for each of the processes are as follows:

(*a*) The activated-sludge and oxidation-ditch processes:
- (i) aeration rate
- (ii) sludge (biomass) recycle rate
- (iii) excess sludge (biomass) wastage rate
- (iv) spatial distribution of the influent to the aerator (step-feed form of activated-sludge process only)

(v) spatial distribution of the recycle biomass to the aerator (step-feed form of activated-sludge process only).

(*b*) Anaerobic digestion:

(i) influent (substrate) feed rate

(ii) thermal energy input.

The lumped form of eqn. 1 obscures the fact that the two aerobic processes are essentially distributed-parameter systems. Indeed, the dynamics of the spatial DO profile can be extremely important to the control of the process [11–13], and it is the spatial variability of the DO environment that permits accommodation of both nitrification and denitrification in the aerator as discussed above. The step-feed form of activated-sludge process, in which the aerator is divided into several (3 or 4) compartments, is the physical analogy of the customary finite-element approximation of a distributed system by a series of continuously stirred tank reactors (CSTR). Even in the case of the extended circulation channels of the oxidation ditch, where it might be appropriate to include a plug flow reactor element (PFR; or dead time), this too can be transformed to an ordinary differential equation form by the method of characteristics [14, 15].

3 System identification

The earliest interest in the development of dynamic models for the purposes of process control is due to the seminal studies of Andrews and his co-workers on anaerobic digestion and activated sludge [2, 16–19]. The fact that their models still mark the boundaries of contemporary research is both a complement to these authors and evidence of a lack of progress in the more rigorous exercises in system identification. Many of these early ideas have since been incorporated, without major structural changes, into the current consensus view of the activated-sludge process, a model developed by a Task Group of the International Association on Water Pollution Research and Control (IAWPRC) [20, 21]. This model, expressed in the form of the state-space representation of eqn. 1, is typical of the dominant approach to waste-water treatment process modelling. It is a complex assembly of many constituent hypotheses, many of which are somewhat speculative; it is strongly nonlinear; the states x_b and x_e are differentiated into effectively unobservable substates; and as such this archetypal model form will almost certainly suffer from a lack of identifiability. There is nothing unusual in this, for the same problem is widespread in the environmental sciences more generally [22–24] and in the adjacent disciplines of pharmacokinetics and biomedical systems analysis [25–27].

For these mechanistic state-space models the crux of the problem of identifiability is that what one would like to know about the internal state of the system (x_b, x_e, α) is of a substantially higher order than what can be observed about the external description on the system (u, d, y). It is well known that there are difficulties with the structural identifiability of biochemical process models, specifically in association with the use of the Monod expression for micro-organism growth [28], although these might be overcome, at least in theory, by an appropriate choice of sampling frequency [29]. The problems of biological waste-water treatment process identification may, however, be more subtle, less obvious, and perhaps more profound.

Consider, first, the properties of the experimental observations constituting the normal operating records to which one frequently has access. Fig. 2 shows daily observations of the influent (input) and effluent (output) concentrations of suspended solids (SS) and biochemical oxygen demand (BOD) for the activated-sludge unit at Norwich Sewage

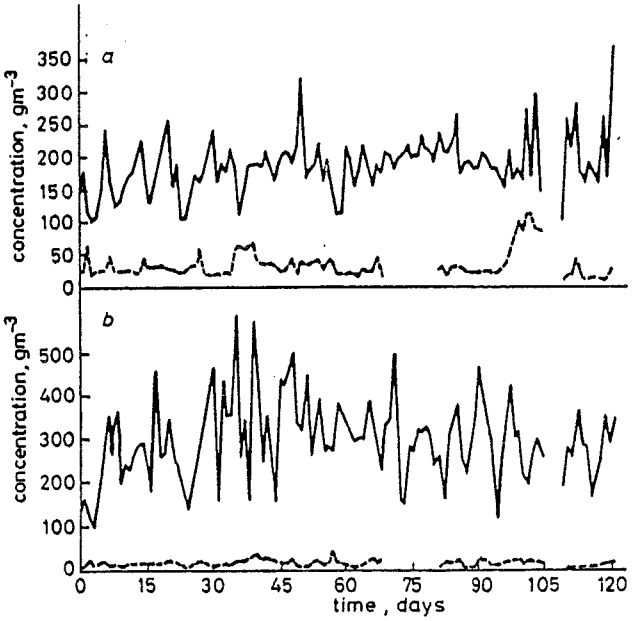


Fig. 2 *Normal operating records for the activated-sludge unit at Norwich Sewage Works*

a Influent and effluent suspended solids concentration

b Influent and effluent 5-day BOD concentration (the influent is measured as total BOD; the effluent as carbonaceous BOD only)

——— influent

- - - - effluent

Works [30]. Both are macromeasures of the degradable organic material (substrate) in waste water. The output SS response is remarkably stationary for most of the 120-day period, in spite of some apparently substantial perturbations in the input SS. The exception is a significant excursion in the output during $t_{94} \rightarrow t_{104}$ which, although it may have been partly exacerbated by the relative high-input SS at t_{101} and t_{103} is more probably a response to a complex sequence and combination of events and control actions over a period perhaps as long as the preceding 10 days. This failure of the process almost certainly reflects the consequences of a poorly settling sludge, the causes of which could be many, perhaps in this instance over-aeration, and therefore over-agitation, of the activated sludge. The biomass would have been finely dispersed and fragmented by this action; possibly too, the process environment was at the time favourable to the enhanced growth of filamentous bacteria, which would have impaired adequate compaction of the biomass in the settling unit (a condition known as bulking sludge).

From their input/output behaviour, then, these processes can appear to be highly stable until failure occurs. On the one hand, apparently significant input disturbances do not excite any significant output response, while, on the other hand, a very significant response can occur in the absence of any obvious motivating input perturbation. These conditions are just as true for hourly time series as they are for the daily observations of Fig. 2 [31]. Both are precisely what one would wish to avoid for the purposes of identifying a model such as eqn. 1. Nevertheless, stability of performance is a highly desirable control objective, and it must be acknowledged that control actions were being taken over the period of Fig. 2 to achieve this. In fact, the plant was being commissioned at the time and was exhibiting somewhat unstable behaviour (see the following). Under more normal operating conditions the stationarity of the output BOD would have been still more pronounced, and the mean level closer to zero.

It is clearly the case that the slower, and arguably more interesting, modes of process behaviour have yielded little to the assault of system identification. The distinction between what is possible in theory (for example, Howell [32] and Cook [15]) and what is identifiable, in practice, is doubtless due to the inherently awkward character of the field observations. Except for one example [33], identification of the dynamics of carbonaceous substrate degradation (Table 1), central though they are to the purpose of these processes, has remained an essentially intractable problem. The IAWPRC model of the activated-sludge process most obviously fails to match observed behaviour with respect to the carbonaceous substrate degradation. This model best succeeds in its description of the nitrification dynamics and the dynamics of oxygen uptake, about which more will be said in the following Section [21]. The reasons for the greater success with the dynamics of nitrification are that the substrate and biomass are more specific, more homogeneous, and, in the case of the substrate, more specifically measurable; under normal operation too, the process can be reasonably unstable and, therefore, identifiably dynamic [14, 34]. Cook's analysis of nitrification dynamics in the oxidation-ditch system is perhaps the best example, so far, of the identification of an appropriate structure $f\{\cdot\}$ for a model within the class of multivariable representations defined by eqn. 1.

Of course, adequate control system design does not depend entirely on a state-space description of the process dynamics. Let us consider then the role of the alternative discrete-time input/output models: these would be no more identifiable from data of the form of Fig. 2 than is

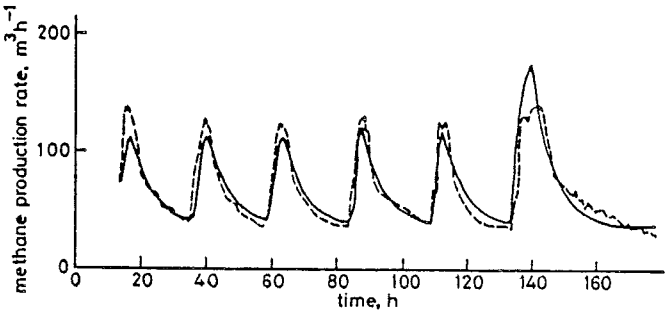


Fig. 3 *Observed and estimated output methane gas production rate for a full-scale anaerobic digester at the Norwich Sewage Works*

- - - - observed
——— estimated

the state-space representation, and indeed they are not [31, 35–38]. However, planned experiments of a kind are not impossible. For instance, most anaerobic digesters are subjected to what amounts to a pulse test of biomass activity, following their daily batch-feeding pattern. Fig. 3 shows the rate of methane production from a 4100 m^3 digester at the Norwich Sewage Works. The batch-feeding pattern is usually distributed over a five-hour period of each day, beginning at the times where the digester gas production rate can clearly be seen to increase rapidly. It is a very straightforward exercise to identify a simple first-order ARMAX model of these dynamics, i.e.

$$y(t_k) = a_1 y(t_{k-1}) + b_0 u(t_k) + b_1 u(t_{k-1}) \qquad (2)$$

where u is the input feed volume, y the output methane production rate, and a_1, b_0 and b_1 parameters. In fact, the results of the model in Fig. 3 are based on the use of a recursive, refined instrumental variable algorithm for parameter estimation [39]. The apparently instantaneous relationship between u and y is partly a function of the sampling frequency (1 hour), but there is also no doubt that gas production increases as soon as feeding commences, simply because of the physical agitation of the digester contents. The model fits the data rather well. But this reveals nothing of the underlying state of the biomass, or whether the feeding pattern is ideal for maximising the performance of the methanogenic bacteria, or, furthermore, whether the output gas production rate should be used (via feedback) to control the feeding pattern. There ought, however, to be some curiosity about why the model response exceeds the peak gas response only at the last feed shown, and why the subsequent observed decline in gas production exhibits a 'knee' (which itself would not be so apparent without the model).

To summarise, nearly all the attempts at the identification of biological treatment systems have attacked the relatively well defined (macroscopic) fringes of the process dynamics, without much impact on the less well defined (microscopic) core of microbiological kinetics. This is true irrespective of whether one is using the more popular state-space representation of eqn. 1 or the simpler input/output models of the form of eqn. 2. We know that the input/output data typically reflect strongly nonlinear relationships. To begin to interpret such data, it is necessary to postulate *a priori* virtually the correct form of (strong) nonlinearity. This is at least possible in the case of the state-space description, but improbable at present, and for many reasons likely to lead to a largely intractable identification problem (including for reasons of a lack of suitable identification methods; Reference 40). The preponderance of identification methods are geared to the use of input/output ARMAX models, best suited to the identification of linear dynamics, and only with difficulty capable of uncovering relatively weak nonlinearities [39]. Above all, there is an absence of simple procedures for inferring the nature of the state vector dynamics from an identified external description of input/output transfer functions. The immediate consequence is that potentially useful microbiological theory cannot easily be refuted or corroborated at the process scale. It may suffice the short-term needs of control-system design to accept an adequate input/output representation of process dynamics. But this serves neither the urgent need of process learning for its own sake (not subordinate to process control), nor the long-term needs of control.

4 State estimation

Given that knowledge of the inaccessible internal state of the biomass (x_b) would seem to be especially useful to control, and given too that only a few variables can be measured reliably online, it follows that the available observations should be made maximally informative via state estimation, or, more specifically, state-parameter reconstruction [11, 41, 42].

In the aerobic treatment processes, the DO dynamics can be observed on a continuous basis and with relatively little measurement noise. They are intimately related to the many facets of aerobic waste degradation (as in Table 1), either reflecting or determining the associated reaction kinetics, and can be effectively manipulated at will. It is hardly surprising, therefore, that they have become, by far, the best studied problem of online state reconstruction [8, 43–49].

A simple conceptual model of the DO dynamics shows that they are a balance between the oxygen transfer rate (OTR), i.e. the controlled input of oxygen to the mixed liquors, and the oxygen uptake rate (OUR), i.e. the consumption of oxygen by the biomass in metabolising the

substrate. In the latter, the relevant biomasses and substrates of both the carbonaceous waste degradation and nitrification are implied. The estimation problem has, therefore, the following structure: (i) the OTR must first be estimated in order that (ii) the OUR can be identified, from which (iii) the substrate and (iv) biomass concentrations can then be reconstructed [43, 50, 51]. Each step requires either a further assumption, or an additional independently observed variable, to avoid the inevitable trap of a loss of identifiability. Any, or all, of the last three quantities might be used for control purposes. This is clearly not a trivial estimation problem to solve, especially when the DO is to be controlled under the customary objective of a constant set-point for long periods of time. In this case Holmberg [43] and Howell *et al.* [44] have illustrated appropriate procedures for introducing the necessary, intermittent pulse- and step-disturbances of the input aeration rate to ensure good identifiability of the OTR parameters. Holmberg's procedure, for example, employs a degraded, high-gain controller with slow (control) sampling frequency to generate the requisite pulse inputs.

In contrast to the impoverished theoretical study of the identification problems of the preceding Section, the proposed solutions to the present problem of state estimation exhibit a notable richness of variety and detail. This includes: recursive least squares [43] (Holmberg, 1981); a recursive extended least-squares algorithm [48]; a recursive instrumental variable algorithm [46]; the extended Kalman filter [46]; a Bierman-type factorised filter [45]; an observer for processes with inaccessible input disturbances [51]; and a Kalman filter formulated in the parameter space with a compensating 'deadbeat' estimator to overcome certain difficulties of insufficient input excitation [49].

What the DO dynamics are to the aerobic treatment processes, so the gas production dynamics are to anaerobic digestion. It is, however, less obvious to discern to what practical control use an online estimate of biomass growth rate could at present be put, and Bastin and Dochain's [52] proofs of convergence properties seem somewhat premature.

Perhaps it is more fair to be critical of one's own contribution to this subject. Unlike either the DO or gas production dynamics, the dynamics of nitrification in the activated-sludge process are relatively slow, and usually monitored by manual sampling on a day-to-day basis, albeit with little noise. Both the substrate (x_s), ammonium-N, and the end product x_p, nitrate-N, are measurable, and it is, in theory, a relatively well posed problem to reconstruct estimates of the nitrifying population biomass x_b from the assumed dynamic interaction among x_b, x_s and x_p. Typical results using an extended Kalman filter are shown in Fig. 4 for the activated-sludge unit at Norwich Sewage Works [34]. The timing is identical with that of the observations of Fig. 2 and the instability of nitrification, already noted earlier, is quite apparent. The growth of the nitrifying population makes more or less steady progress over two 50-day periods ($t_4 \rightarrow t_{57}$; $t_{67} \rightarrow t_{111}$), but is punctuated by relatively sudden collapses of the population at t_{58} and t_{112}. Such behaviour is not unrelated to the behaviour of carbonaceous substrate degradation observed in Fig. 2; the two microbial processes are in a sense 'competitors' for the same resources of dissolved oxygen, and it is the carbonaceous substrate degradation that is most likely to be the victor in this competition.

The results of Fig. 4 may be interesting, and certainly, together with the other results quoted here, they establish the practical possibility of an everyday use of online state estimation. We know that the slower, low-frequency changes in the state of the biomass can have a profound

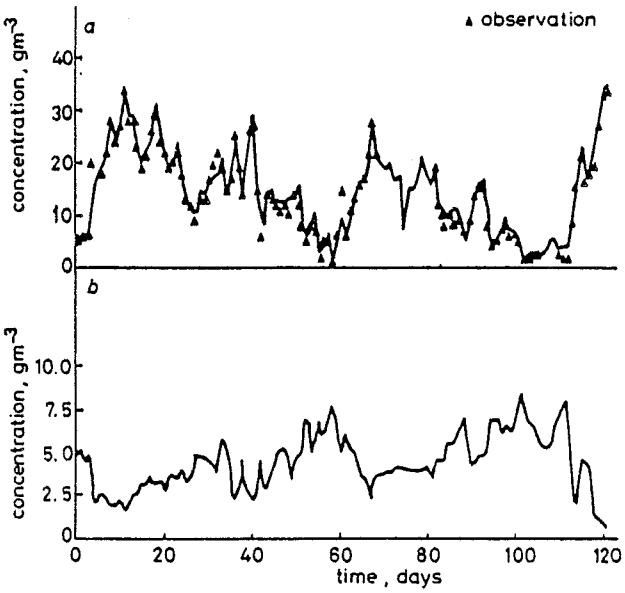


Fig. 4 *State estimates for (a) the aerator ammonium-N concentration and (b) the aerator Nitrosomonas (biomass) concentration for the activated-sludge unit of Norwich Sewage Works*

influence on process stability, but there are some very basic questions still to be answered. For instance:

(i) Is an important event such as biomass collapse detectable *only* via changes in the *inaccessible* process states and parameters?

(ii) How is observed and estimated information most readily available at a fast sampling frequency (minutes, hours) to be translated into the possibly crucially important longer-term time scale (days, weeks)?

(iii) If this information at a slow sampling frequency were made available, would the plant manager be concerned to evaluate and act on trends that are almost imperceptible on a day-to-day basis?

(iv) Are we really about to launch on an unsuspecting audience of treatment plant managers sophisticated methods of control theory that are quite opaque to the user [53]?

5 Control

The dynamics of biological waste-water treatment processes are characterised by a wide range of time constants, nonlinearities, imprecision, substantial stability punctuated by abrupt failures, and a sensitive, readily adaptable community of organisms. As in the foregoing discussion, the earliest applications of control theory showed little appreciation of the practical constraints to which these processes are subject [54, 55], and they were politely but roundly criticised for this [56].

There must now be many examples of conventional three-term (PID) controllers being used in practice, and there is also a substantial literature. A good impression of the scope and detail of the first decade or so of research can be distilled from the papers by Flanagan and Bracken [57], Flanagan [58], Olsson [8, 105, 106], and Tanuma [59]. Virtually all the analysis has been conducted in the time domain, although the frequency-domain analysis of Lech *et al.* [60, 61] is a notable exception.

The essential feature in the design of an activated-sludge unit is the requirement for achieving a given growth rate of the biomass [62]. The clear priority for operational

control is therefore to ensure that this growth rate is maintained in the face of dynamic disturbances to the plant. Under normal conditions this translates into the regulation of a constant level of biomass in the unit (aerator and clarifier together), a constant interaction between the substrate and biomass, and a constant environment for the biomass x_e. Broadly speaking, these three surrogate control loops reflect, respectively, the slow, medium and fast speeds of process perturbation and response, of which the first two are sufficiently slow to have been implemented traditionally by the plant manager. It is this that, in the large part, has given the impression that 'things seem to work fairly well, and of their own accord' (until abrupt failure occurs). With the basic control objective thus assured, it follows that attention can be turned to the minimisation of operating costs. As neither of the slower loops are likely to be excessively expensive in terms of their control inputs, efficient, high-performance control of the DO level emerges as a natural target of analysis. It is the only really controllable element of the biomass environment, its dynamics are fast and relatively well defined, and its control input (aeration) is expensive. The need and economic return of closed-loop DO control have made it a focus for research, and its performance, in practice, is suitably illustrated by Stephenson *et al.* [63], Holmberg [50] and Olsson *et al.* [64].

The primary motivation for the control of anaerobic digestion has always been rather different: a concern with the problems of instability and the development of elementary control procedures for recovery of the system after failure [17, 38, 65, 102, 103, 108]. The goal of maximising the rate of methane production has been of secondary importance, and it is only very recently that the process has been examined from the point of view of gas production rate control [66].

Given its purpose and the standards to which a wastewater treatment plant is subject, i.e. the removal of a polluting substrate to below an acceptable level, the concept of output substrate-error feedback is conspicuous by its absence. This too reinforces the impression of a system operating almost in an open-loop fashion under normal circumstances.

5.1 Optimal control

The first formal treatments of optimal control appear to have been those of Sincic and Bailey [67] and Yeung *et al.* [68] in the context of periodic control of the activated-sludge process. The process is subject to natural diurnal oscillations in its input hydraulic and substrate disturbances, and it is common practice to operate the recycle sludge (biomass) flow rate in sympathy with these oscillations via ratio control. The same sort of control can be augmented by control of the distributions of the influent sewage and recycle flows to the various compartments of the step-feed form of aerator. The intention is to maintain a constant interaction between the substrate and biomass by manipulating the pattern of contact between the two. Both the earlier study of Sincic and Bailey [67] and the more recent work of Stehfest [33] are essentially similar in their application of optimal periodic control to the more advanced form of step-feed process design. Marsili-Libelli [69], pointing out that these solutions lead to prescheduled open-loop controllers, has introduced a three-term feedback controller parameterised via the minimisation of a quadratic performance index expressed as a function of a control cost and set-point errors for output substrate and biomass (with recycle flow as the single control input). In the event, as we have said, penalising excessive control action turns out to be largely irrelevant for this type of control. It is much more pertinent to control of the aeration-DO loop: a suitably low value for the DO set-point ensures a healthy environment for the biomass, and the control problem is then clearly one of avoiding damaging under-aeration and costly over-aeration. It can easily be solved as a linear-quadratic problem (for perturbations about an equilibrium point) and with integral action incorporated, although not necessarily as a periodic solution [70].

None of these optimal controller designs, however, have been evaluated in practice, and some of their implications would be decidedly undesirable. For instance, a near bang-bang form of control, in which the recycle flow from the clarifier to the aerator may be shut off completely for up to six hours at a time [67], might be fine for the performance of the aerator yet disastrous for the performance of the clarifier. But then this neglect of the clarifier's behaviour, and its crucial role in overall process stability, is not restricted to the studies in optimal control.

5.2 Adaptive control

For a nonlinear system in which the dynamics of the biomass are continually being adapted to changing input perturbations and a changing environment, adaptive control would seem ideal. There is currently substantial interest in the application of self-tuning control to biological waste-water treatment processes, and to two areas in particular: control of the DO loop in the activated sludge unit [64, 71, 72, 73], and output substrate and product (gas) control in anaerobic digestion [66, 74]. The controllers of Ko *et al.* [72] and Dochain and Bastin [74] are virtually identical in the sense that the process parameters are estimated by recursive least squares and a minimum-variance deadbeat form of control is obtained by setting the one-step-ahead prediction of the output to the desired set point. In the Dochain-Bastin example certain convergence problems were overcome by resorting to a Clarke-Gawthrop form of controller in which a quadratic cost on the incremental control action was introduced. The other two studies [64, 73] were motivated by a desire to avoid steady-state offset due to unobserved perturbations in the oxygen uptake rate (OUR), and have both opted for PI controller structures, with an extension in the case of the latter to a Cameron-Seborg form of controller with PID features. But whatever the variations on a theme, self-tuning control can be, and has been, demonstrated in practice. The results of Fig. 5 are from Olsson *et al.* [64] and refer to a full-scale plant in Stockholm.

Attractive, though these applications of self-tuning

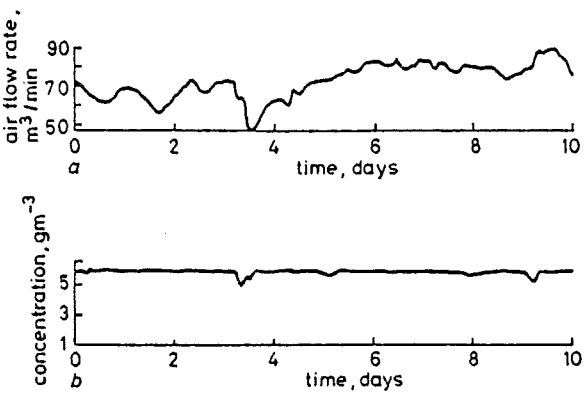


Fig. 5 *Self-tuning control of DO concentration in one of the aeration channels at the Käppala Sewage Works, Stockholm (after Reference 64)*
a Input air flow rate, $m^3 min^{-1}$
b DO concentration, gm^{-3}

control undoubtedly are, they do not strike at the essence of the adaptive nature of the processes, i.e. the adaptability of the biomass activity. They are, at least for DO control in the activated-sludge process, controllers that seek to keep the biomass environment constant; they respond to changes in the behaviour of the biomass, but do not seek to change that behaviour; and they operate on time scales considerably shorter than those over which the adaptability of the biomass operates. At this low level of the control hierarchy the adaptive character of the controller is being exploited for the identification of a process whose dynamics can be modelled as linear in the parameters and where the parameters are unknown but *constant*. In order for overall control to be adapted to the variability of the biomass activity it is apparent that this must be implemented by a combination of altering lower-level loop set points and/or adaptive controllers for the slower loops whose dynamics are truly time-varying at a rate consistent with microbial adaptivity.

There are three such controllers that we must consider in this spirit, although it is highly unlikely that this would have been the original motivation for their development. First, in an adaptive stochastic controller proposed by Cheruy *et al.* [75] the substrate biomass interaction is separated into three linear small-perturbation models for three regions of the state space; the probability of the process being in any of these regions at a given sampling instant is computed from a Kalman filter and Bayes' rule; and expected values of the controls are enabled through the slower loops of biomass recycle and wastage. It is an elegant solution, but one whose practical viability is debatable. The second example, due to Yust *et al.* [76], stems in contrast from a fairly long history of practical attempts to use the specific oxygen utilisation rate (SCOUR), i.e. the oxygen uptake rate per unit of biomass, as a basis for the control of biomass activity [62]. Tight, effective control of the fast DO loop is exploited to provide an estimate of the oxygen uptake rate, which then in cascade is used to derive a (time-varying) set point for a slower loop regulating the biomass concentration. A third study [77] combines elements of both the above: the fast DO dynamics are again used to generate an estimate of OUR, but this information is then used in a second observer that generates estimates of the biomass and substrate concentrations to be used, in turn, for the control of a slower (recycle) control loop. Strictly speaking, this last is not, therefore, an adaptive controller, but together with the two other studies it completes the picture of three isolated examples of the principle of state reconstruction being employed directly within a control loop.

5.3 Hierarchical and knowledge-based control

Clearly the rudimentary co-ordination and cascading of control loops that has surfaced in the discussion of adaptive control demands now a more detailed treatment of the natural hierarchical structure of an overall control system. We have so far examined essentially fine-tuning forms of control for normal conditions in which to some extent the control system can respond and adapt to minor changes in the process dynamics. Even the SCOUR controller of Yust *et al.* [76] is a case where command within the hierarchy is from the process 'upwards'. There are, of course, the complementary problems of using the control system deliberately to adapt the process behaviour (of switching performance from one set of normal conditions to another set) and of making major changes to the process dynamics in the event of quite abnormal operating conditions, typically the abrupt failures already illustrated in Figs. 2 and 4.

Given the lack of progress in the precise formal quantification of process dynamics (especially identification of the central characteristics of biomass activity and substrate-biomass interaction), given the slow dynamics of the process (and thus the natural and integral role of the plant manager in the upper levels of the control structure), it is perhaps inevitable that a knowledge-based (or expert) system would sooner or later be called for. By no means have all the hierarchical control structures used for multiple-loop co-ordination been conceived as expert systems (for example, Flanagan [58], Tanuma [59], Hiraoka *et al.* [78] and Sekine *et al.* [79], but then it could be a matter of semantics as to what constitutes an expert system. For instance, in some of the earliest automated waste-water treatment plants in the UK [80] the computer software incorporated simple rules for control action in the event of failure, such as the solids settling problem of Fig. 2*a* [30]. They were based on past experience and, in essence, relied upon interaction with the plant operator for a final decision. A similar, but more comprehensive form of rule-based empirical logic was developed over a decade ago for computer-assisted control of anaerobic digestion at a plant in the USA [81]. The fact that one can find ten years later a 'knowledge-based system' encoding much the same logic [82] is good for one's sense of perspective, but detrimental to one's confidence in venturing an exclusive definition of such a system.

Suffice it to say, therefore, that the notion of using the accumulated practical knowledge of an expert to design a control system for biological waste-water treatment was already apparent by 1977 when it appeared again in two slightly different forms: the linguistic rules of fuzzy (or multiple-valued) logic [41], and qualitative operational states each associated with a specific combination of control actions [83]. Among subsequent studies, those of Holmberg [50] and Maeda [82] are similar to Gillblad and Olsson's in their use of operational states and a Boolean-type logic. Maeda's controller does not appear to possess a hierarchical structure, but it certainly extends the original view of an expert controller. The plant manager is requested to define an operating state and the conditions of the external disturbances; he is then given a recommended control action; and given his specific choice, the current disturbance and the current operating state, he is advised of the predicted operating state at the next sampling instant. Like Holmberg's [50] analysis, the case studies of Tong *et al.* [84] and Flanagan [13] are clearly directed at the problem of an upper level controller for the co-ordination of the set-points of low-level automatic closed loops. They both, however, employ fuzzy logic for this purpose and, indeed, are comparable in their use of a truth qualification for the support given to any of the control rules at any sampling instant. There are, of course, differences between the two: the controller of Tong *et al.* is eclectic in its use of process observations; Flanagan's controller is driven solely by the plant manager's qualitative interpretations of the dynamics of the spatial DO profile in the aerator, thus encoding (perhaps with some irony) much of the precise, quantitative analysis of Olsson and Andrews [11].

Expert systems probably do not yet qualify for the label 'conventional'. If the wider application of more advanced conventional control to the faster, better-defined aspects of biological process dynamics is to liberate and enhance the role of the plant manager in developing and utilising unconventional control logic for the slower, more awkward aspects of behaviour, overall control can only benefit. As such, much remains to be done. But of all the

problems that might be candidates for the development of a knowledge-based controller, biological waste-water treatment still seems eminently suitable.

6 Some elementary questions for control

Of the three areas of identification, estimation and control there has been the least success in the first and the greatest in the last. This balance of progress is likely to continue, and, indeed, in this Section we shall argue backwards from some elementary questions of control objectives, through estimation and identification to the required nature of future process descriptions (i.e. the steps traced through Sections 2, 3, 4, and 5 will be reversed).

6.1 Control

The primary goal in the application of control to biological waste-water treatment processes has hitherto been to attenuate low-amplitude, high-frequency input load disturbances to maintain a constant set-point performance (however, broadly that may be interpreted). This motivation is enshrined in the design criteria of the processes (a constant level of biomass; a constant environment for the biomass) and is typified by the control of the DO level in the aerobic treatment processes.

In the wider context, however, a waste-water treatment plant and its unit processes are merely parts of the global system of river basin management, and, as such, must be responsive to possible strategic switches of desired performance [85–88].† We have already mentioned such switching of performance in Section 5. It is also well known that nitrification activity can be 'switched' on or off by manipulation of the biomass environment, specifically by changes of the DO concentration [79]. These same changes can be used to modulate the species composition of the biomass to suppress sludge settling problems (rising and bulking sludges).

If the view is widely held that biological waste-water treatment processes perform well of their own accord, it might be that facilitating more flexible and adaptable operation is the major contribution that further control can make to their performance. And, as this desired variability of performance is most apparent at the upper, more strategic levels of the control hierarchy, it can be expected in the first instance to be less concerned with precise changes of set point, and more with the imprecise terms in which plant operator experience is expressed. At this level, the control rules might be structured in terms of broad operating domains; for example, with nitrification, at high throughput, at low throughput, and with default safe operation; rules for the transfer between domains and for the recognition of often-repeated, set-piece failure events would have to be established; and performance in any domain might be gauged by an 'ability to resist shock disturbances' and a 'capacity for easy transfer' out of one domain to another.

6.2 Estimation and identification

A fundamental difficulty in understanding, and therefore controlling, biological waste-water treatment process behaviour is that macroscopic failure (or some other undesirable response) may occur while the observable state of the process appears to be constant, and not as a result of an external disturbance, but in response to low-frequency internal perturbations at a microscopic scale. Access to, and interpretation of, these internal perturbations in the state of the biomass must be a priority.

Some of these difficulties will undoubtedly be resolved by the development of better sensors. It is unreasonable to assume that there will be no improvements in instrumentation and that online observation will remain obscured by uncertainty and imprecision. For example, instrumentation of a sense of smell, an integral but imprecise element of the control of the activated sludge process [30] is already underway in the development of an 'electronic nose' [89]. More precise and reliable instruments are likely to be developed for the macroscopic measures of performance, such as BOD concentration in the aerobic processes [90, 91], and total volatile acids concentration in anaerobic digestion [65]. And in the longer term there may be instruments providing a substantially better resolution of the more microscopic, specific aspects of the biomass and its activity [92].

Alternatively, and in line with Section 4, much has already been written on the subject of fault detection using the methods of state and parameter estimation [93]. Similar schemes, in particular, those exploiting recursive parameter estimation in which the identification of parametric variations is the indicator of the fault, have been suggested for various aspects of waste-water treatment plant operation [34, 37, 74, 94]. These problems of fault detection and fault diagnosis exhibit clear conceptual parallels with the problem of model structure identification. For instance, detection of the fact that a fault has occurred is equivalent to exposing the failure of a constituent model hypothesis; inferring the cause of the fault corresponds with the problem of inferring an improved (posterior) model structure from a failed (prior) structure; and a lack of model identifiability (or over parameterisation) is roughly the same as setting up a scheme for fault detection in which it is not possible to distinguish among several possible causes of the same fault. What has been said elsewhere, then, on the development of novel algorithms for solving the problem of model structure identification for state-space process representations of the form of eqn. 1 in Section 2 would also seem to have some validity here [40].

6.3 Process models

At the very centre of all these issues of control, estimation and identification, however, is the fact that we have the least quantitative knowledge of those state dynamics we would most want to control.

There are models describing, for example, the microscale phenomena of substrate diffusion and transport in the cell environment [95, 107], but it is unclear how one could aggregate upwards from this basis to the macroscopic requirements of control system synthesis. Most attempts at gaining access to the necessary detailed knowledge of the physiological state of the biomass, or its activity, or its species composition, have come from disaggregation of the biomass and biomass environment state vectors (x_b, x_e) in the macroscopic representation of eqn. 1. The procedure of 'structuring the biomass' (x_b) into a number of separate substates x_{bi} ($i = 1, \ldots, n$) has been widely discussed [18, 50, 62, 68, 87, 96–98, 101]; that owing to Busby and Andrews classifies the biomass into $n = 3$ substates, a 'stored' mass, an 'active' mass, and an 'inert' mass. It reflects the commonly encountered observation that the substrate (x_s) is initially incorporated by physical mechanisms into the biological floc (as stored mass), where it is subsequently metabolised by the active

† BECK, M.B., and FINNEY, B.A.: 'Operational water quality management: a case study of the Bedford Ouse river system', *Water Resour. Res.*, 1986 (under consideration)

mass, which itself will eventually decay into the inert mass. It is also widely acknowledged that cell age can affect the process of oxygen consumption via endogenous respiration, and Ranta's work with cell-age-distribution models [28, 96] is a logical extension, as it were, of the Busby-Andrews model. Howell and Jones' [104] work on the control of an industrial waste-treatment process subjected to occasional input overloads of inhibitory substrate arguably comes closest to the biochemical foundations of substrate-biomass interaction. In spite of the fact that their model invokes a detailed discussion of an induction mechanism for the activation of substrate uptake, it is not unduly complex (it comprises six state variables covering two groups of micro-organisms).

There can be little doubt about the long-term goal of this trend, which must presumably be the quantitative understanding (and exploitation) of process kinetics at the molecular or enzyme level. But there are other, more immediate, and quite different goals for the development of process models that are equally challenging. In the past few years there have arisen legitimate concerns about the value of 'conventional' approaches to the identification and description of processes whose behaviour is inherently imprecise and given observations that are also fundamentally imprecise [46, 85]. An algebraic, or differential equation, representation of such processes perhaps simply should not be entertained. The alternative would be some type of qualitative knowledge representation, encoded, for example, in the format of fuzzy logic. This has much appeal (and some philosophical difficulties), and there are already examples of its application in the adjacent field of environmental system simulation [99]. There is also a wealth of empirical expertise on some of the most difficult problems of activated-sludge process fault diagnosis and correction that would lend itself ideally to this approach [100].

7 Conclusions

There is a paradox in the control of biological waste-water treatment processes: they have for a long time been viewed as processes that are largely 'self-controlled'; yet in certain respects they are extremely difficult to control. They can appear to be remarkably stable, but are known to exhibit conditions of failure, and the typical macroscopic observations of either type of performance have yielded little progress in the identification of process dynamics. Some of their instabilities can be attributed to high-frequency large-amplitude external disturbances, although most cannot, and are most probably due to the propagation of low-frequency, initially low-amplitude, internal perturbations in the largely inaccessible microbiological state of the system. Of central importance in this respect are the biomass activity and the species composition of the heterogeneous culture of organisms. Among various applications of the concepts of optimal, adaptive and hierarchical control structures, there has been a clear focus on regulation of the relatively well defined fast dynamics of the dissolved oxygen concentration in the aerobic treatment processes. The motivation for such control has been primarily that a constant environment for the biomass at some desired level ensures a constant rate of biomass growth and polluting substrate degradation.

The present models of process dynamics are not sufficiently well identified to permit precise analysis of the conditions promoting stability or leading to instability of performance. There is an increasing trend towards the less precise, more qualitative classification and evaluation of performance using fuzzy logic and the ideas of expert systems. This is expected to continue, and indeed is likely to cover not only the encoding of control decision rules, but also representation of the knowledge of basic microbiological process dynamics. The first indications of a growing interest in fault detection are apparent. Coupled with advances in the development of biosensors and of intelligent sensors (which latter in effect incorporate tightly controlled, microscale bioreactors) this too is likely to be an area ripe for analysis. Lastly, we have argued that the previous motivation for control is a narrow view of its potential. More specifically, the biomass environment can be manipulated to switch performance from one stable operating region to another. At the macroscopic level this reflects an unconventional view of the waste-water treatment plant as a subordinate part of the larger system of river basin management. At the microscopic level it reflects an, as yet, only vague appreciation of the nonlinear threshold characteristics of basic microbiological functions.

Hitherto, the application of control to biological waste-water treatment processes has been required to answer (in a rather subdued fashion) the questions set for it within the confines of accepted process designs. It might do well to take the initiative, and conversely set the questions to be answered by modifications of treatment process design.

8 Acknowledgments

The author is indebted to Anglian Water, and in particular Tony Latten, for access to the facilities at the Whitlingham Sewage Works, Norwich. Thanks are also due to Dr. Brad Finney of the Humboldt State University, California, for his collaboration on the analysis of the anaerobic digester gas production dynamics.

9 References

1 GARRETT, M.T.: 'Hydraulic control of activated sludge growth rate', *Sewage & Ind. Wastes*, 1958, **30**, pp. 253–261
2 ANDREWS, J.F.: 'Dynamic models and control strategies for waste-water treatment processes', *Water Res.*, 1974, **8**, pp. 261–289
3 HEGG, R.A., RAKNESS, K.L., and SCHULTZ, J.R.: 'Evaluation of operation and maintenance factors limiting municipal wastewater treatment plant performance', *J. Water Pollut. Control Federation*, 1978, **50**, pp. 419–426
4 HILL, W.R., REGAN, T.M., and ZICKEFOOSE, C.S.: 'Operation and maintenance of water pollution control facilities: a WPCF white paper', *ibid.*, 1979, **51**, pp. 899–906
5 'Water industry control systems'. Report prepared by Environmental Resources Limited and Watson Hawksley for the UK Department of Trade and Industry, 1979
6 'Final report of the working party on control systems for the water industry'. Standing Technical Committee Report 27, National Water Council, London, 1981
7 BECK, M.B.: 'Operational water quality management: beyond planning and design'. Executive report 7, International Institute for Applied Systems Analysis, Laxenburg, Austria, 1981
8 OLSSON, G.: 'State of the art in sewage treatment plant control', *Am. I. Chem. E. Symp. Ser.*, 1977, **72**, pp. 52–76
9 YOUNG, P.C., SPRIET, J.A., and BECK, M.B.: 'Requirements for information technology and related techniques in the development and design of advanced biotechnological process control systems', Report prepared for the Commission of the European Communities, 1985
10 TOMLINSON, E.J., HAMILTON, I.M., and WILLIAMSON, K.: 'Process management systems at sewage-treatment works: Witney evaluation and demonstration facilities', *Water Pollut. Control*, 1984, **83**, pp. 172–183
11 OLSSON, G., and ANDREWS, J.F.: 'The dissolved oxygen profile — a valuable tool for the control of the activated sludge process', *Water Res.*, 1978, **12**, pp. 985–1004
12 OLSSON, G., and ANDREWS, J.F.: 'Dissolved oxygen control in the activated sludge process', *Water Sci. & Technol.*, 1981, **13**, pp. 341–347.
13 FLANAGAN, M.J.: 'On the application of approximate reasoning

to control of the activated sludge process'. Preprint, Joint Automatic Control Conference, San Francisco, 1980
14 COOK, S.C.: 'Nitrification dynamics in the oxidation ditch wastewaster treatment process', *Water Sci. & Technol.*, 1984, **16**, (5–7), pp. 595–611
15 COOK, S.C.: 'Parameter estimation in oxidation ditch modelling', *Trans. Inst. Meas. & Control*, 1984, **6**, (3), pp. 132–141
16 ANDREWS, J.F.: 'Dynamic model of the anaerobic digestion process'. *Proc. Am. Soc. Civil Engrs., J. Sanit. Eng. Div.*, 1969, **96**, (SA3), pp. 848–853
17 GRAEF, S.P., and ANDREWS, J.F.: 'Stability and control of anaerobic digestion', *J. Water Pollut. Control Federation*, 1974, **46**, (4), pp. 666–683
18 BUSBY, J.B., and ANDREWS, J.F.: 'Dynamic modelling and control strategies for the activated sludge process', *ibid.*, 1975, **47**, pp. 1055–1080
19 PODUSKA, R.A., and ANDREWS, J.F.: 'Dynamics of nitrification in the activated sludge process', *ibid.*, 1975, **47**, pp. 2599–2619
20 DOLD, P.L., EKAMA, G.A., and MARAIS, G.R.: 'A general model for the activated sludge process', *Prog. Water Technol.*, 1980, **12**, pp. 47–77
21 DOLD, P.L., and MARAIS, G.R.: 'Evaluation of the general activated sludge model proposed by the IAWPRC Task Group', *Water Sci. and Technol.*, 1986, **17**, (6)
22 YOUNG, P.C.: 'General theory of modeling for badly defined systems', *in* VANSTEENKISTE, G.C. (Ed.): 'Modeling, identification and control in environmental systems' (North-Holland, Amsterdam, 1978), pp. 103–135
23 BECK, M.B., and VAN STRATEN, G.: 'Uncertainty and forecasting of water quality' (Springer, Berlin, 1983)
24 SOROOSHIAN, S., and GUPTA, V.K.: 'Automatic calibration of conceptual rainfall-runoff models: the question of parameter observability and uniqueness', *Water Resour. Res.*, 1983, **19**, (1), pp. 260–268
25 GODFREY, K.R., and DISTEFANO, J.J.: 'Identification of model parameters', *in* Barker, H.A., and Young, P.C. (Eds.): 'Identification and system parameter estimation'. *Preprints 7th IFAC Symp.*, York, pp. 89–114
26 AL-DAHAN, M.I., LEANING, M.S., CARSON, E.R., HILL, D.W., and FINKELSTEIN, L.: 'The validation of complex, unidentifiable models of the cardiovascular system', *in* Barker, H.A., and Young, P.C. (Eds.): 'Identification and system parameter estimation'. *Preprints 7th IFAC Symp.*, York, pp. 1213–1218
27 FLOOD, R.L., LEANING, M.S., CRAMP, D.G., and CARSON, E.R.: 'Clinical time series: analysis, modelling and recursive estimation', *in* Barker, H.A., and Young, P.C. (Eds.): 'Identification and system parameter estimation'. *Preprints 7th IFAC Symp.*, York, pp. 1613–1618
28 HOLMBERG, A., and RANTA, J.: 'Procedures for parameter and state estimation in microbial growth process models', *Automatica*, 1982, **18**, pp. 181–193
29 VIALAS, C., CHERUY, A., and GENTIL, S.: 'An experimental approach to improve the Monod model identification', *in* Johnson, A. (Ed.): 'Modelling and control of biotechnological processes'. *Preprints 1st IFAC Symp.*, Noordwijkerhout, The Netherlands, pp. 155–159
30 BECK, M.B., LATTEN, A., and TONG, R.M.: 'Modelling and operational control of the activated sludge process'. Professional paper PP-78-10, International Institute for Applied Systems Analysis, Laxenburg, Austria, 1978
31 BERTHOUEX, P.M., HUNTER, W.G., and PALLESEN, L.: 'Dynamic behaviour of an activated sludge plant', *Water Res.*, 1978, **12**, pp. 957–972
32 HOWELL, J.A.: 'Parameter estimation for biological waste treatment dynamic models', *in* MOO-YOUNG, M., and ROBINSON, C.W. (Eds.): 'Advances in biotechnology, Vol. II' (Pergamon, 1981), pp. 571–578
33 STEHFEST, H.: 'Optimal periodic control of step-feed activated sludge plant', *Environ. Technol. Lett.*, 1985, **6**, (12), pp. 556–565
34 BECK, M.B.: 'Operational estimation and prediction of nitrification dynamics in the activated sludge process', *Water Res.*, 1981, **15**, pp. 1313–1330
35 BECK, M.B.: 'An analysis of gas production dynamics in the anaerobic digestion process'. Report CUED/F-CAMS/TR135, University Engineering Department, Cambridge, 1976
36 DALRYMPLE, J.F., and CROWTHER, J.M.: 'Theory and practice of time domain techniques', *in* Bazin, M. (Ed.): 'Mathematics in microbiology' (Academic, London, 1983)
37 OLSSON, G., and CHAPMAN, D.: 'Modelling the dynamics of clarifier behaviour in activated sludge systems', *in* Drake, R.A.R. (Ed.): 'Instrumentation and control of water and wastewater treatment and transport systems' (Advances in Water Pollution Control, Pergamon, 1985), pp. 405–412
38 SKILTON, J.M., HAWKES, D.L., and RICHARDS, A.H.: 'Control of the anaerobic digestion of livestock wastes', *Environ. Technol. Lett.*, 1985, **6**, (12), pp. 620–628
39 YOUNG, P.C.: 'Recursive time-series analysis: An introduction' (Springer-Verlag, Berlin, 1984)
40 BECK, M.B.: 'Structures, failure, inference and prediction', *in* BARKER, H.A., and YOUNG, P.C. (Eds.): 'Identification and system parameter estimation'. *Preprints 7th IFAC Symp.*, 1985, York, pp. 1443–1448.
41 BECK, M.B.: 'Modelling and control in practice', *Prog. Water Technol.*, 1977, **9**, (5/6), pp. 557–564
42 OLSSON, G.: 'Estimation and identification problems in wastewater treatment', *in* WOOD, E.F. (Ed.): 'Real-time forecasting/control of water resource systems' (Pergamon, Oxford, 1980), pp. 93–108
43 HOLMBERG, A.: 'Microprocessor-based estimation of oxygen utilization in the activated sludge wastewater treatment process', *Int. J. Syst. Sci.*, 1981, **12**, (6), pp. 703–718
44 HOWELL, J.A., YUST, L.J., and REILLY, P.: 'On-line measurement of respiration and mass transfer rates in an activated sludge aeration tank', *J. Water Pollut. Control Federation*, 1984, **56**, (4), pp. 319–324
45 HOWELL, J.A., and SODIPO, B.O.: 'On-line respirometry and estimation of aeration efficiencies in an activated sludge aeration basin from dissolved oxygen measurements', *in* JOHNSON, A. (Ed.): 'Modelling and control of biotechnological processes'. *Preprints 1st IFAC Symp.*, 1985, Noordwijkerhout, The Netherlands, pp. 191–198
46 COOK, S.C., and JOWITT, P.W.: 'Investigation of dissolved oxygen dynamics in the activated sludge process', *in* BARKER, H.A., and YOUNG, P.C. (Eds.): 'Identification and system parameter estimation'. *Preprints 7th IFAC Symp.*, 1985, York, pp. 241–248
47 GOTO, M., and ANDREWS, J.F.: 'On-line estimation of oxygen-uptake rate in the activated-sludge process', *in* DRAKE, R.A.R. (Ed.): 'Instrumentation and control of water and waste water treatment and transport systems' (Advances in Water Pollution Control, Pergamon, 1985), pp. 473–480
48 OLSSON, G.: 'Control strategies for the activated process', *in* MOO-YOUNG, M. (Ed.): 'Comprehensive biotechnology' (Pergamon, 1985), Chap. 65, pp. 1107–1119
49 HOLMBERG, U., and OLSSON, G.: 'Simultaneous on-line estimation of oxygen transfer rate and respiration rate', *in* JOHNSON, A. (Ed.): 'Modelling and control of biotechnological processes'. *Preprints 1st IFAC Symp.*, 1985, Noordwijkerhout, The Netherlands, pp. 185–189
50 HOLMBERG, A.: 'Modelling of the activated sludge process for microprocessor-based state estimation and control', *Water Res.*, 1982, **16**, pp. 1233–1246
51 MARSILI-LIBELLI, S.: 'On-line estimation of bioactivities in activated sludge processes', *in* HALME, A. (Ed.): 'Modelling and control of biotechnical processes' (Pergamon, Oxford, 1983), pp. 121–125
52 BASTIN, G., and DOCHAIN, D.: 'Adaptive estimation of microbial growth rates', *in* BARKER, H.A., and YOUNG, P.C. (Eds.): 'Identification and system parameter estimation'. *Preprints 7th IFAC Symp.*, 1985, York, pp. 1161–1166
53 HALME, A.: 'The industrial application of modern measurement and estimation techniques in biotechnology', *in* 'Modelling and control of biotechnological processes'. *Preprints 1st IFAC Symp.*, 1985, Noordwijkerhout, The Netherlands
54 D'ANS, G., KOKOTOVIC, P.V., and GOTTLIEB, D.: 'A nonlinear regulator problem for a model of biological waste treatment', *IEEE Trans.*, 1971, **AC-16**, pp. 341–347
55 DAVIS, J.J., KERMODE, R.I., and BRETT, R.W.J.: 'Generic feed forward control of activated sludge', *Proc. Am. Soc. Civil Engrs., J. Env. Eng. Div.*, 1973, **99**, (EE3), pp. 301–314
56 WELLS, C.H.: 'On the practical application of a "nonlinear regulator problem for a model of biological waste treatment"' (Discussion), *IEEE Trans.*, 1971, **AC-16**, (4), pp. 385–388
57 FLANAGAN, M.J., BRACHEN, B.D., and ROESLER, J.F.: 'Automatic dissolved oxygen control', *Proc. Am. Soc. Civil Engrs., J. Env. Eng. Div.*, 1977, **103**, pp. 707–722.
58 FLANAGAN, M.J.: 'Upgrading the activated sludge process through automatic control', *A.I.Chem.E. Symp. Ser. 190*, 1979, **75**, pp. 232–242
59 TANUMA, M.: 'Water quality management in a wastewater treatment plant', *in* Beck, M.B. (Ed.): 'Real-time water quality management'. Proceedings of a Task Force, Collaborative Paper CP-80-38, International Institute for Applied Systems Analysis, Laxenburg, Austria, pp. 241–260
60 LECH, R.F., LIM, H.C., GRADY, C.P.L., and KOPPEL, L.B.: 'Automatic control of the activated sludge process — I Development of a simplified dynamic model', *Water Res.*, 1978, **12**, pp. 81–90
61 LECH, R.F., GRADY, C.P.L., LIM, H.C., and KOPPEL, L.B.: 'Automatic control of the activated sludge process — II Efficacy of control strategies', *ibid.*, 1978, **12**, pp. 91–99

62 STENSTROM, M.K., and ANDREWS, J.F.: 'Real-time control of activated sludge process', *Proc. Am. Soc. Civil Engrs., J. Env. Eng. Div.*, 1979, **105**, (EE2), pp. 245–260

63 STEPHENSON, J.P., MONAGHAN, B.A., and LAUGHTON, P.J.: 'Automatic control of solids retention time and dissolved oxygen in the activated sludge process', *Water Sci. & Technol.*, 1981, **13**

64 OLSSON, G., RUNDQWIST, L., ERIKSSON, L., and HALL, L.: 'Self tuning control of the dissolved oxygen concentration in activated sludge systems', *in* DRAKE, R.A.R. (Ed.): 'Instrumentation and control of water and wastewater treatment and transport systems' (Advances in Water Pollution Control, Pergamon, Oxford, 1985), pp. 473–480

65 ROZZI, A., DI PINTO, A.C., and BRUNETTI, A.: 'Anaerobic process control by bicarbonate monitoring', *Environ. Technol. Lett.*, 1985, **6**, (12), pp. 594–601

66 DOCHAIN, D., and BASTIN, G.: 'Stable adaptive algorithms for estimation and control of fermentation processes', *in* JOHNSON, A. (Ed.): 'Modelling and control of biotechnological processes', *Preprints, 1st IFAC Symp.*, 1985, Noordwijkerhout, The Netherlands, pp. 1–6

67 SINCIC, D., and BAILEY, J.E.: 'Optimal periodic control of activated sludge processes — I. Results for the base case with Monod/decay kinetics', *Water Res.*, 1978, **12**, pp. 47–53

68 YEUNG, S.Y.S., SINCIC, D., and BAILEY, J.E.: 'Optimal periodic control of the activated sludge process — II. Comparison with conventional control for structured sludge kinetics', *Water Res.*, 1980, **14**, pp. 77–83

69 MARSILI-LIBELLI, S.: 'Optimal control strategies for biological wastewater treatment', *in* RINALDI, S. (Ed.): 'Environmental systems analysis and management' (North-Holland, Amsterdam, 1982), pp. 279–287

70 MARSILI-LIBELLI, S.: 'Optimal aeration control for wastewater treatment', *in* Cuenod, M.A. (Ed.): 'Computer aided design of control systems' (Pergamon, Oxford, 1980), pp. 511–516

71 McINNIS, B.C., LIN, C.Y., and BUTLER, P.B.: 'Adaptive microcomputer dissolved oxygen control for wastewater treatment'. *Preprint 5th IFAC Symp. Identification & Syst. Parameter Est.*, 1979, Darmstadt, West Germany

72 KO, K.Y.-J., McINNIS, B.C., and GOODWIN, G.C.: 'Adaptive control and identification of the dissolved oxygen process', *Automatica*, 1982, **18**, pp. 727–730

73 MARSILI-LIBELLI, S., GIARDI, R., and LASAGNI, M.: 'Self-tuning control of the activated sludge process', *Environ. Technol. Lett.*, 1985, **6**, (12), pp. 576–583

74 DOCHAIN, D., and BASTIN, G.: 'Adaptive identification and control algorithms for nonlinear bacterial growth systems', *Automatica*, 1984, **20**, (5), pp. 621–634

75 CHERUY, A., PANZARELLA, L., and DENAT, J.P.: 'Multimodel simulation and adaptive stochastic control of an activated sludge process', *in* Halme, A. (Ed.): 'Modelling and control of biotechnical processes' (Pergamon, Oxford; 1983), pp. 127–138

76 YUST, L.J., STEPHENSON, J.P., and MURPHY, K.L.: 'Control of the specific oxygen utilisation rate for the step-feed activated-sludge process', *Trans. Inst. Meas. & Control*, 1984, **6**, (3), pp. 165–172

77 MARSILI-LIBELLI, S.: 'Activated sludge process control using dissolved oxygen measurements', *Water Sci. & Technol.*, 1984, **16**, (5–7), pp. 613–620

78 HIRAOKA, M., TSUMURA, K., FUJITA, I., and KANAYA, T.: 'Modeling and control of the activated sludge process by use of autoregressive model'. *Preprints 9th IFAC World Congress*, 1984, Budapest, Hungary, Vol. IV, pp. 170–175

79 SEKINE, T., IWAHORI, K., FUJIMOTO, E., and IHOMORI, Y.: 'Advanced control strategies for the activated sludge process', *in* DRAKE, R.A.R. (Ed.): 'Instrumentation and control of water and wastewater treatment and transport systems' (Advances in Water Pollution Control, Pergamon, Oxford, 1985), pp. 269–276

80 COTTON, P., and LATTEN, A.: 'Initial operating experience of the automated sewage treatment works, Norwich', *Prog. Water Technol.*, 1977, **9**, (5/6), pp. 499–506

81 KOCH, C.M., and WANKOFF, W.: 'An experiment in computer assisted control of the anaerobic digestion process at Philadelphia's Northeast Pollution Control Plant'. *Proc. 47th Ann. Water Pollut. Control Federation Conf.*, Oct. 1974

82 MAEDA, K.: 'A knowledge based system for the wastewater treatment process'. *Prepints 9th IFAC World Congress*, 1984, Budapest, Hungary, Vol. IV, pp. 89–94

83 GILLBLAD, T., and OLSSON, O.: 'Computer control of a medium-sized activated sludge plant', *Prog. Water Technol.*, 1977, **9**, (5/6), pp. 427–434

84 TONG, R.M., BECK, M.B., and LATTEN, A.: 'Fuzzy control of the activated sludge wastewater treatment process', *Automatica*, 1980, **16**, pp. 695–701

85 JOWITT, P.W.: 'Risk analysis, fuzzy logic and river basin management', *Water Sci. & Technol.*, 1984, **16**, (5–7), pp. 579–585

86 JOWITT, P.W., LUMBERS, J.P., BECK, M.B., and JENKINS, W.O.: 'Operational river water quality management', *ibid.*, 1984, **16**, (5–7), pp. 381–392

87 BECK, M.B.: 'On the development and application of models for water quality management', *ibid.*, 1984, **16**, (5–7), pp. 541–560

88 HIRAOKA, M., TSUMURA, K., TERAO, Y., and OTA, M.: 'Systematic approach to the control of an activated sludge wastewater treatment plant', *in* DRAKE, R.A.R. (Ed.): 'Instrumentation and control of water and wastewater treatment and transport systems' (Advances in Water Pollution Control, Pergamon, Oxford, 1985), pp. 661–664

89 'Directory of Research in Biotechnology'. Biotechnology Directorate, Science and Engineering Research Council, Swindon, UK, 1984

90 KÖHNE, M.: 'Practical experiences with a new on-line BOD measuring device', *Environ. Technol. Lett.*, 1985, **6**, (12), pp. 546–555

91 HARITA, K., OTANI, Y., HIKUMA, M., and YASUDA, T.: 'BOD quick estimating system utilizing a microbial electrode', *in* DRAKE, R.A.R. (Ed.): 'Instrumentation and control of water and wastewater treatment and transport systems' (Advances in Water Pollution Control, Pergamon, Oxford, 1985), pp. 529–534

92 TSUMURA, K., and HIRAOKA, M.: 'Digital image processing for measuring the length of filamentous micro-organisms in activated sludge', *in* DRAKE, R.A.R. (Ed.): 'Instrumentation and control of water and wastewater treatment and transport systems' (Advances in Water Pollution Control, Pergamon, Oxford, 1985), pp. 741–744

93 ISERMANN, R.: 'Process fault detection based on modeling and estimation methods — a survey', *Automatica*, 1984, **20**, (4), pp. 387–404

94 STEHFEST, H.: 'An operational model of the final clarifier', *Trans. Inst. Meas. & Control*, 1984, **6**, (3), pp. 160–164

95 LAU, A.O., STROM, P.F., and JENKINS, D.: 'The competitive growth of floc-forming and filamentous bacteria: a model of activated sludge bulking', *J. Water Pollut. Control Federation*, 1984, **56**, (1), pp. 52–61

96 RANTA, J., KAITALA, V., and HALME, A.: 'Modelling and simulation of bacterial population age distribution in activated sludge plant', *in* CHICHOCKI, K., and STRASZAK, A. (Eds.): 'Systems analysis applications to complex programs' (Pergamon, Oxford, 1977), pp. 93–101

97 HOWELL, J.A., and JONES, M.G.: 'Problems in on-line parameter estimation for a structured model', *in* HERRMANN, J.P.R. (Ed.): 'Proceedings of ICCAFT-3' (Society of Chemical Industry, London, 1981), pp. 57–65

98 THANTHAPANICHAKOON, W., and HIMMELBLAU, D.M.: 'Simulation of a time dependent activated sludge wastewater treatment plant', *Water Res.*, 1981, **15**, pp. 1185–1195

99 CAMARA, A.S., PINHEIRO, M.D., ANTUNES, M.P., and SEIXAS, M.J.: 'Linguistic simulation in planning theory and application'. *Preprint Int. Conf. Syst. Dynamics Soc.*, Keystone, Colorado, July, 1985

100 JONES, G.A., and FRANKLIN, B.C.: 'The prevention of filamentous bulking of activated sludge by operation means at Halifax Sewage-Treatment-Works', *Water Pollut. Control*, 1985, **84**, pp. 329–346

101 BECK, M.B.: 'Modelling and control studies of the activated sludge process at Norwich Sewage Works', *Trans. Inst. Meas. & Control*, 1984, **6**, (3), pp. 117–131

102 CARR, A.D., and O'DONNELL, R.C.: 'The dynamic behaviour of an anaerobic digester', *Prog. Water Technol.*, 1977, **9**, (5/6), pp. 727–738

103 COLLINS, A.S., and GILLILAND, B.E.: 'Control of anaerobic digestion process', *Proc. Am. Soc. Civil Engrs., J. Env. Eng. Div.*, 1974, **100**, (EE2), pp. 487–505

104 HOWELL, J.A., and JONES, M.G.: 'The development of a two-component operon-type theory model for phenol degradation', *in* 'Water — 1980'. *A.I.Chem.E. Symp. Ser. 209*, 1982, **77**, pp. 122–128

105 OLSSON, G.: 'Automatic control in wastewater treatment plants', *Trib. Cebedeau*, 1980, **436**, pp. 121–130

106 OLSSON, G., and HANSSON, O.: 'Stochastic modelling and computer control of a full scale wastewater treatment plant'. *Proc. Symp. Syst. & Models Air & Water Pollut.*, London, Institute of Measurement and Control, 1976

107 PALM, J.C., JENKINS, D., and PARKER, D.S.: 'Relationship between organic loading, dissolved oxygen concentration and sludge settleability in the completely-mixed activated sludge process', *J. Water Pollut. Control Federation*, 1980, **52**, (10), pp. 2484–2506

108 ROZZI, A.: 'Modelling and control of anaerobic digestion processes', *Trans. Inst. Meas. & Control*, 1984, **6**, (3), pp. 153–159

APPENDIX 2: Further Introductory Notes on On-line Estimation/ Forecasting.

Further Introductory Notes on On-Line Estimation/Forecasting

To reiterate what has already been stated in the main body of the text, the essence of the on-line estimation problem is to be able to determine the values of quantities that cannot be easily measured from values of those variables that can be easily measured. Sludge age, for example, is one such quantity, being determined freom observable quantities such as various flow-rates and suspended solids concentrations. It should also be apparent that measurement errors can significantly corrupt the accuracy (and reliability) of the computed value of sludge age.

Put another way, and in the more sophisticated terms of dynamic modelling, the estimation problem can be stated as follows:

> Given a relationship (the model) between the measured input disturbances, the process state variables (and model parameters), and the measured output responses, determine a set of unknown, unobservable quantities (the process states and model parameters) from a set of observable quantities (the measured inputs and outputs).

In its elementary form this is little more than the age-old mathematical problem of: given a set of "knowns", a set of "unknowns", and a relationship between the "knowns" and "unknowns", calculate the "unknowns". And as such this is a problem whose importance is neither difficult to appreciate and whose solution can be obtained in several (more or less sophisticated) ways.

If we take the customary problem of a reactor in which we have a model of the interactions among the substrate, biomass, and product concentrations, and where the substrate and product concentrations can be measured on-line, then the current state of the biomass (the unmeasured quantity) can be estimated from this prudent combination of the model with the data. The reasoning is, of course, that current knowledge of the unmeasured state would be highly desirable for the ultimate purpose of process control, and has a unique significance not directly substitutable by knowledge of any observed quantity. a specific example of this more advanced problem might be the on-line estimation of nitrifier biomass given real-time observations of the ammonium-N and nitrate-N concentrations (Beck, 1981).

Texts on control theory do not make it easy to appreciate the basic concepts and functioning of an algorithm such as the Kalman filter. Simply, there are two salient features of the filter:

(i) It embodies a model of the reactor;

(ii) It acknowledges that any observation and any description of the real world's behaviour is subject to uncertainty, and it embodies therefore the necessary computational effort to accommodate this uncertainty.

It is this second feature that is the most distinctive, and potentially the most powerful attribute of the filter, yet also perhaps the cause of its obscurity (and of much obfuscation).

If we consider the conceptual picture of Figure A2.1 it is possible to see how the filter receives measured information from the bioreactor, and exploits a model to provide reconstructed information about unmeasured (and indeed measured) quantities. Essentially three types of uncertainty are acknowledged:

(a) Uncertainty in the specified relationships of the model;

(b) Uncertainty in the input disturbances of the reactor; and

(c) Uncertainty in the observed outputs of the reactor (although here it may be convenient to imagine the outputs to be simply error-corrupted observations of some of the process states).

The crux of the computations that manipulate the propagation (with time) of these uncertainties is the way in which they enter into the second of the two following steps for the (recursive) determination of the state estimates:

(i) Prediction of the values of the state variables across the measurement interval, i.e. between two consecutive instances in time at which process observations are made;

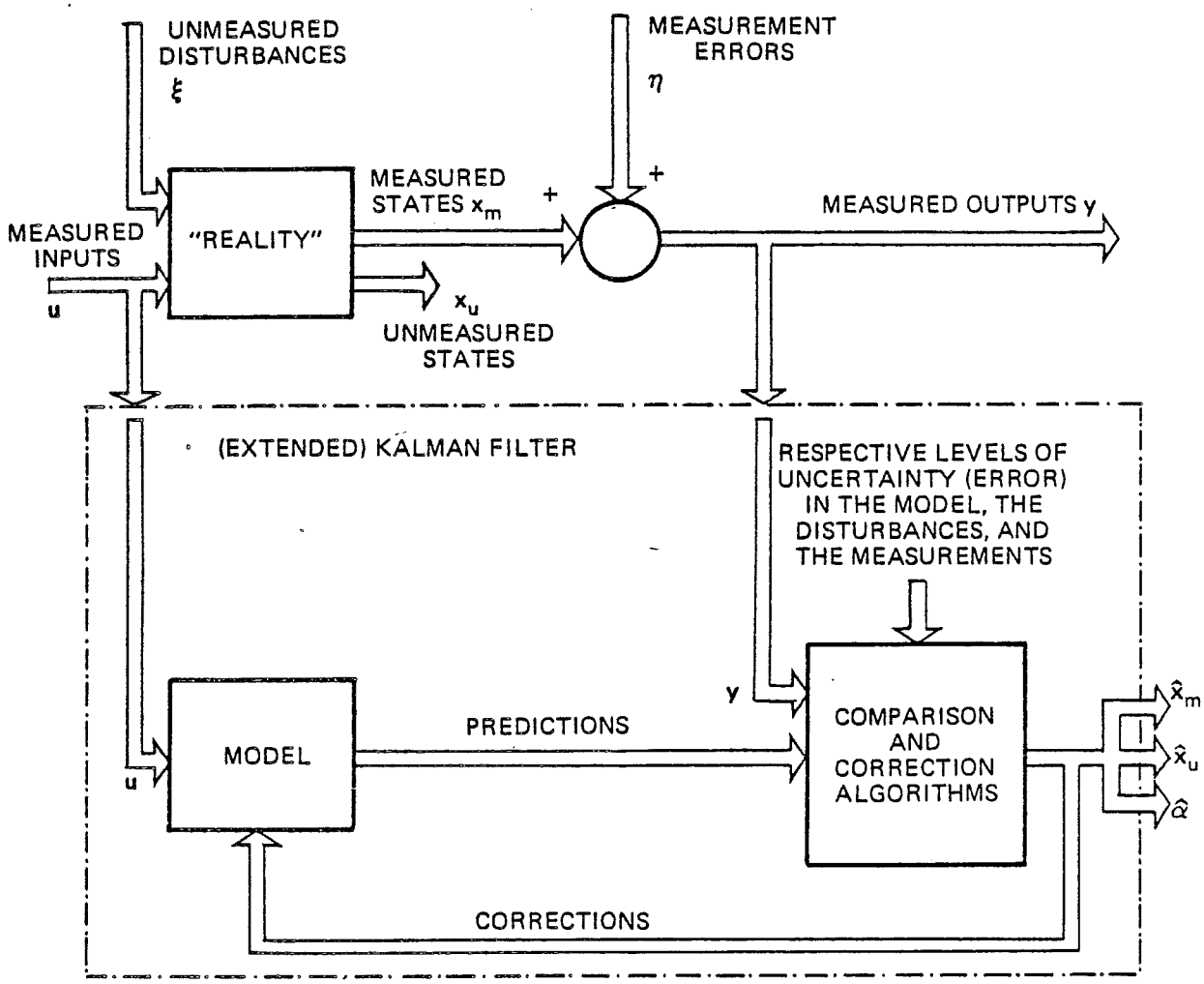


Figure A2.1 Conceptual picture of the (extended) Kalman filter. The notation ^ denotes estimates generated by the filter of the measured state variables, the unmeasured state variables, and the model parameter values.

(ii) Correction of the predicted state estimates at the next sampling instant on the basis of the mismatch between the predicted and observed output response of the reactor.

The specified model carries out the first of these two functions, while the second rests upon a computation of the following form:

Corrected estimate = Predicted estimate + (weighting factor) x (mismatch between model and reality).

The focal point of the computations regarding uncertainty is the role of the "weighting factor" in this relationship. Thus in figurative terms:

(a) If the model is "known well" relative to the accuracy of the process observations, the weighting factor is small and relatively little account is taken of any mismatch between the model and reality in correcting the predicted state estimates (the mismatch is attributed to chance, spurious errors in the observations).

(b) If the model is not "known well" relative to the accuracy of the process observations, the weighting factor is large and accordingly much account is taken of mismatches between the model and reality (in fact, it is clear that there is here a not inconsiderable, in-built capacity of the filter to carry out an adaptive form of process learning).

At the same time, the filter is able to provide measures of the confidence (or, conversely, uncertainty) that should be attached to the computed values of the state estimates.

This, then, is the primary function of the filter. It is worth noting, however, that it can be put to other purposes. For instance, the principle of one-step-ahead prediction embraced by the above (qualitative) discussion can be naturally extended to the generation of state estimates either further forwards, i.e. predictions over several time steps into the future, or backwards in time (smoothing, or at a finer time-scale than the sampling interval (i.e. interpolation)). It is also possibly of

substantial significance to note that the filter can be used as a method of system identification in the development and evaluation of lumped-parameter models, and that these notions of system identification and state-parameter estimation are closely allied with the concepts of fault detection.

Additional discussion of some of these concepts of on-line estimation/forecasting can be found in Beck (1981), a paper dealing with nitrification dynamics in the activated sludge process at Norwich Sewage Works.

APPENDIX 3: "The Effects of Stormwater Runoff on Wastewater Treatment Plant Operations", Supporting Case for an Application for a Research Grant from the SERC, October 15, 1987.

THE EFFECTS OF STORMWATER RUNOFF ON WASTEWATER TREATMENT PLANT OPERATIONS

1. OBJECTIVES

The objectives of the project are as follows:

(i) To develop a dynamic mathematical model for simulating the major unit processes of wastewater treatment, with special reference to the performance of these processes when disturbed by surges of stormwater runoff.

(ii) To design and implement intensive, short-duration experiments at selected sites in order to obtain field data for guiding the development of the model, and for calibrating and evaluating its performance.

(iii) To apply the model for assessing current and novel control strategies for the management of surges of stormwater runoff passing into a wastewater treatment plant.

2. MOTIVATION

Previous Work

The management of water quality in highly developed, intensively used river basins has been moving slowly over the past decade towards a growing concern with shorter-term operational problems. This concern is most apparent in the increasingly widespread discussion of seasonal consent conditions, particularly in respect of the nitrification of wastewater discharges, and in the attention now given to the effects of transient pollution events, such as stormwater overflows from sewers (Clifforde et al, 1986). If these new questions of operational water quality management are to be fully explored, then it will be necessary to relax the normal assumption of steady-state conditions that has underpinned most of the earlier planning studies of river basin management. New mathematical models are required -- models capable of handling stochastic and dynamic conditions -- and flexible, adaptive, cost-effective strategies of wastewater treatment plant operation will need to be developed.

The main thrust of the proposed research is the intention to develop a set of mathematical models so that the costs, benefits, and performance of these new operating strategies can be thoroughly evaluated before their implementation in practice.

The project builds upon several previous contributions of the Principal Investigator to the subjects of water quality modelling and operational water quality management:

(i) In the late 1970's a policy study was conducted clearly identifying the need, desirability and feasibility of the concepts of operational water quality management (Beck, 1981).

(ii) Following the conclusions of this policy study, a more detailed engineering development study was defined for investigation specifically of the behaviour of the Milton Keynes Sewage Works and the effects of its discharge on a 54km stretch of the Bedford Ouse River. During 1985/6 this work was supported in part by SERC (Research Grant GR/D 35155). A dynamic model for flow, runoff, and water quality interactions in the river has been developed and calibrated against a comprehensive field data base (Beck and Finney, 1987a). The model has been applied to the preliminary exploration of seasonal nitrifying policies for wastewater teatment plant operation (Beck and Finney, 1987b), and for assessment of the relative importance of this factor with respect to the contributions to in-stream nitrate-N levels of nonpoint agricultural runoff (Beck et al, 1987). The dynamic model at present quantifies the interactions among the biochemical oxygen demand (BOD), dissolved oxygen (DO), ammonium-N, nitrate-N, and chlorophyll-a concentrations in the river.

(iii) Since the mid-1970's the Principal Investigator has been involved with the development of dynamic models for the biological unit processes of wastewater treatment, principally with reference to the Norwich Sewage Works (Beck, 1984, 1986). In 1985/6 this work assumed a new direction, and has since been concerned with the development of a dynamic mathematical model for simulating and managing the effects of stormwater runoff on both the operating performance of the wastewater treatment plant and the quality of the receiving water body. A specialised, intensive experimental programme was implemented in May, 1986, at the Norwich Sewage Works in support of this modelling effort; the experiment was conducted in collaboration with, and under contract to, the Norwich Division of Anglian Water. A model of the dynamic behaviour of the primary clarifier has been successfully identified from these observations (Lessard and Beck, 1987), and research is currently in progress on the development of similar models for the aerator basin and secondary clarifier of the Norwich plant. These models are designed for description of the interactions among the suspended solids (SS), volatile suspended solids (VSS), total chemical oxygen demand (CODT), soluble COD (CODS), and ammonium-N concentrations in each of the unit processes. A comprehensive and consistent data base for the removal of heavy metals in wastewater treatment was assembled during the Norwich experiment, and the models are capable of being extended to the analysis of these data.

In addition to these previous (and ongoing) contributions to the subjects of the proposed project, the Principal Investigator is currently responsible (with colleagues at Imperial College) for an SERC-supported project on the Control of Wastewater Treatment Processes Using Expert Systems (Research Grant GR/D 51247).

Research Experience

The research experience of the Principal Investigator is in part summarised above. It is also relevant to note that he is the Secretary of the Specialist Group on Systems Analysis in Water Quality Management of the International Association of Water Pollution Research (IAWPRC) and a member

of the IAWPRC Task Group on the Real-Time Control of Urban Drainage Systems.

3. PROGRAMME OF RESEARCH

The proposed project is to be conducted as a part of the Water Research Centre's programme of research on River Basin Management. It therefore has a quite precise remit:

> To concern itself with modelling the dynamic behaviour of the wastewater treatment plant, under the assumption that a model for sewer flow and quality variations and a model for river impacts will be provided elsewhere in the WRc programme.

The steps in the proposed programme of research are as follows:

(i) Completion of analysis of the Norwich experiment (May, 1986). Of particular note in this experiment is the way in which the observations of heavy metals concentrations (e.g. mercury, cadmium, copper, lead, etc.) suggest a quite unexpected means of access to the detailed settling characteristics of sediments passing through the unit processes of wastewater treatment. A modelling analysis of these data (of limited duration) is proposed, to be extended should the WRc Programme as a whole find the preliminary results to be of more general interest.

(ii) An additional experiment at Norwich, with special reference to the secondary clarifier unit. In the May 1986 experiment, the secondary clarifier was only partially included in the sampling programme. This is also the least well understood unit process of wastewater treatment. An additional experiment is therefore proposed, to be of ten days' duration, to include the sampling of process streams once every three hours, and to be implemented during the late spring/early summer when the plant is likely to be disturbed by surges of stormwater runoff.

(iii) Specialised experiments at the Derby Sewage Works. Powell and Selby (1986) of Severn-Trent Water have recently discussed a dynamic approach to river basin management. As part of this approach to practical problem solving the Authority is establishing a comprehensive instrumentation and monitoring programme to assess the effects of sewer overflows on the performance of Derby Sewage Works and the quality of the River Derwent. Experimental work and data analysis for the present project is proposed at this site in three phases:

(a) A short duration, intensive experiment along the lines already implemented at Norwich in 1986. The present project would be required to supplement (with, in particular, manual sampling and analysis of COD and SS variations) the data to be obtained from the automatic monitoring system. For this first experiment attention is to be restricted to observation of the behaviour of the primary clarifier and storm tank units.

(b) In the interim, application of the model derived at (a) above for analysis of data obtained over a longer period from the automatic monitoring system.

(c) A second short duration experiment, designed to permit the development of a dynamic model of percolating filter behaviour under storm surge conditions (collaboration with the Water Research Centre is planned for the development of the model).

(iv) Application of the models for development and evaluation of stormwater control strategies. This will comprise several component parts:

(a) Assessment of simulated performance over a range of different observed sequences and magnitudes of precipitation events.

(b) Evaluation of threshold flows for the diversion of storm flows to storm tanks prior to primary settling and prior to secondary biological treatment.

(c) Development of optimal schedules for the return of storm flows from the storm tanks to either the primary clarifier units or the biological unit processes.

(d) Assessment of flow splitting and partial bypassing of the aerator basin, with special reference to their effects on the maintenance of nitrifying conditions. Studies of the activated sludge process are to include the evaluation of possible design modifications, for example, the step-feed form of that unit.

(e) Analysis of the effects of storm surges on the production, quality and treatability of sludge.

(f) Study of the influence of returned sludge liquors on the settleability of solids entering the primary clarifier.

This part of the programme will be closely coordinated with the Water Research Centre for the pooling of knowledge on stormwater control strategies for wastewater treatment plants.

In broad terms steps (ii) and (iii) of this programme could be carried out in parallel, with the first three steps to be completed within the first two years of the project. The final year of the project would therefore be reserved entirely for the exercise of the model developed by the start of that year.

Should time and resources permit, it would be desirable for the project to move from its initial concern with the Norwich plant, through the analysis of the Derby plant, and then (with suitable generalisation) to the other case studies of the WRc research programme.