

SYSTEMS IDENTIFICATION OF FACULTATIVE
WASTE STABILIZATION PONDS

A thesis submitted to the University of London in
partial fulfilment of the requirements for the
degree of Doctor of Philosophy in the
Faculty of Engineering

by

R.Y.G. Andoh B.Sc., M.Sc., D.I.C.

SEPTEMBER 1986

Public Health & Water Resources Engineering
Department of Civil Engineering
Imperial College of Science & Technology
London SW7

CONTENTS	PAGE
ABSTRACT	8
ACKNOWLEDGEMENT	10
FOREWORD	11
CHAPTER ONE: INTRODUCTION	13
1.1 GENERAL	13
1.2 UNCERTAINTIES IN DESIGN AND PERFORMANCE	14
1.2.1 GENERAL	14
1.2.2 ON CRITERIA FOR ASSESSMENT OF PERFORMANCE	17
1.3 NEED FOR A SYSTEMS STUDY	21
1.4 THE STUDY	24
1.4.1 PRIMARY OBJECTIVE	24
1.4.2 SCOPE OF STUDY	24
1.4.2.1 GENERAL	24
1.4.2.2 USE OF MATHEMATICAL MODELS	25
1.4.2.3 EXPECTATIONS	25
1.4.2.4 APPROACH TO THE STUDY	26
1.5 END POINT	27
CHAPTER TWO: DESCRIPTION OF POND SYSTEMS	29
2.1 GENERAL	29
2.2 WASTE WATER TREATMENT IN PONDS	31
2.3 PROCESSES OCCURRING	37
2.3.1 GENERAL	37
2.3.2 CHEMICAL PROCESSES	37
2.3.3 BIOLOGICAL PROCESSES	40
2.3.3.1 GENERAL	40

2.3.3.2	ALGAE	41
2.3.3.3	BACTERIA	44
2.3.3.4	HIGHER PROTISTS AND SMALL ANIMALS	45
2.3.4	PHYSICAL PROCESSES	46
2.3.4.1	GENERAL	46
2.3.4.2	TRANSPORT	47
2.3.4.3	METEOROLOGICAL FACTORS	47
2.3.4.4	STRATIFICATION	49
2.4	THE BENTHIC SUB-REGION	52
2.5	PATHOGENS	54
2.6	END POINT	56
CHAPTER THREE: DATA REQUIREMENTS TYPE AND PRELIMINARY ANALYSIS		58
3.1	POND DATA	58
3.1.1	DATA REQUIREMENTS	58
3.1.2	DATA TYPE	61
3.1.3	RELIABILITY OF POND DATA	62
3.1.4	DATA SETS ACQUIRED	63
3.2	PRELIMINARY ANALYSIS OF DATA	65
3.2.1	GENERAL	65
3.2.2	SUITABILITY OF DATA IN RELATION TO PROCESS DESCRIPTION	65
3.2.3	SUITABILITY OF DATA FOR DYNAMIC ANALYSIS	68
3.2.3.1	GENERAL	68
3.2.3.2	MISSING DATA AND INTERPOLATION	69
3.2.3.3	RAW DATA PLOTS	70
3.2.3.4	SIGNALS CONTENT	74
3.2.4	SIGNALS ENHANCEMENT	80

3.2.5	TIME SERIES ANALYSIS (TIME DOMAIN)	83
3.2.5.1	GENERAL	83
3.2.5.2	AUTOCORRELATIONS	85
3.2.5.3	CROSS-CORRELATIONS	87
3.3	MULTIVARIATE ANALYSIS	91
3.3.1	GENERAL	91
3.3.2	PRINCIPAL COMPONENTS AND FACTOR ANALYSIS	92
3.3.2.1	GENERAL	92
3.3.2.2	RESULTS FOR POND 1 DATA	94
3.3.2.3	RESULTS FOR POND 2 DATA	98
3.3.2.4	SUMMARY	101
3.4	END POINT	102
CHAPTER FOUR: REVIEW AND EVALUATION OF PROCESS MODELS FOR DESIGN		106
4.1	GENERAL	106
4.2	NEED FOR REVIEW AND EVALUATION OF MODELS	107
4.3	RECOMMENDED LOADINGS	108
4.3.1	GENERAL	108
4.3.2	GLOYNA (1971)	109
4.3.3	CANTER AND ENGLAND (1970)	109
4.3.4	COMMENTS ON RECOMMENDED LOADINGS	109
4.4	EMPIRICAL FORMULATIONS	111
4.4.1	INDIAN METHOD	111
4.4.2	GLOYNA AND CO-WORKERS	112
4.4.3	SOLAR METHODS	117
4.4.4	THE REGRESSION APPROACH	122
4.4.5	DIMENSIONLESS METHOD	126
4.4.6	MARAIS AND SHAW PROCEDURE	128

4.4.7	DISCUSSIONS AND CONCLUSIONS - EMPIRICAL FORMULATIONS	129
4.5	MODELS BASED ON PROCESS RATES	130
4.5.1	GENERAL	130
4.5.2	STEADY-STATE MODELS	131
4.5.2.1	GENERAL	131
4.5.2.2	MARAIS' MODELS	132
4.5.2.3	THIRUMURTHI'S MODELS	135
4.5.2.4	UHLMAN'S RATE EQUATION	138
4.5.2.5	DISCUSSIONS AND CONCLUSIONS - STEADY-STATE MODELS	139
4.5.3	DYNAMIC MODELS	140
4.5.3.1	GENERAL	140
4.5.3.2	OBJECTIVES OF SIMULATION EXERCISE	140
4.5.3.3	MARAIS (1970)	142
4.5.3.4	FERRARA (1978)	146
4.5.3.5	FRITZ (1978)	153
4.5.3.6	SUMMARY	157
4.6	END POINT	157
	CHAPTER FIVE: IDENTIFICATION OF POND SYSTEMS	163
5.1	GENERAL	163
5.2	INTERNAL DESCRIPTIVE MODELLING	165
5.2.1	CONVENTIONAL APPROACH	165
5.2.2	NEED FOR NON-CONVENTIONAL APPROACH	167
5.2.3	GENERAL PHILOSOPHY OF ADOPTED APPROACH	168
5.2.4	PROCESS DESCRIPTION AS INPUT-STATE-OUTPUT	169
5.2.5	STOCHASTIC PROCESSES	171
5.2.6	GAUSS/MARKOV PROCESSES	173
5.2.7	STATE-SPACE REPRESENTATION	173

5.2.8	STATE ESTIMATION FOR STATE-SPACE MODELS	174
5.2.8.1	GENERAL	174
5.2.8.2	BLOCK ESTIMATORS	177
5.2.8.3	RECURSIVE ESTIMATORS	179
5.2.8.4	ASSESSMENT PROCEDURE	185
5.3	HYPOTHESES EVALUATION SCHEME	186
5.3.1	BASE MODEL: HYPOTHETICAL COMPLETE DESCRIPTION	186
5.3.2	EXPERIMENTAL FRAMEWORK (SIMPLIFICATIONS)	187
5.4	THE IDENTIFICATION EXERCISE	189
5.4.1	GENERAL	189
5.4.2	CHOICE OF MODELLED OUTPUT VARIABLES	190
5.4.3	ZERO RATES MODEL	193
5.4.3.1	MODEL FORMULATION	193
5.4.3.2	POND 1 SIMULATION RESULTS	194
5.4.3.3	POND 2 SIMULATION RESULTS	196
5.4.3.4	SUMMARY AND CONCLUSIONS	197
5.4.4	GROSS REACTIVE TERM MODELS	197
5.4.4.1	MODEL FORMULATION	197
5.4.4.2	GENERAL DISCUSSION ON ESTIMATION RESULTS	203
5.4.4.3	POND 1 SIMULATION RESULTS	207
5.4.4.4	POND 2 SIMULATION RESULTS	211
5.4.4.5	SUMMARY AND CONCLUSIONS	214
5.4.5	CARBONACEOUS ORGANICS: WORKING MODELS	215
5.4.5.1	GENERAL	215
5.4.5.2	CARBONACEOUS ORGANICS: WORKING MODEL 1	218
5.4.5.3	CARBONACEOUS ORGANICS: WORKING MODEL 2	223
5.4.5.4	CARBONACEOUS ORGANICS: WORKING MODEL 3	228
5.4.6	NUTRIENTS: WORKING MODELS	234

5.4.6.1	GENERAL	234
5.4.6.2	POND 1 NUTRIENTS: WORKING MODELS	235
5.4.6.3	POND 2 NUTRIENTS: WORKING MODELS	242
5.5	END POINT	248
CHAPTER SIX: OVERVIEW AND USE OF MODELS IN DESIGN		251
6.1	GENERAL OVERVIEW	251
6.2	UNCERTAINTIES IN SYSTEMS REPRESENTATION	254
6.3	POND DESIGN AND OPERATIONAL REQUIREMENTS	257
6.4	USE OF MODELS IN DESIGN	259
6.5	END POINT	261
CHAPTER SEVEN: SUMMARY, CONCLUSIONS AND RECOMMENDATIONS		262
7.1	GENERAL	262
7.2	SUMMARY AND CONCLUSIONS	263
7.3	RECOMMENDATIONS	265
REFERENCES		267
TABLES AND FIGURES		284
APPENDIX A: NUMERICAL INTEGRATION SCHEME AND MODEL ASSESSMENT PROCEDURE		532
APPENDIX B: RECURSIVE ESTIMATION ALGORITHMS		539

ABSTRACT

The concept of waste water treatment in ponds is introduced and the need for a systems approach to the study of ponds identified. A description of facultative pond processes is presented covering the chemical, biological and physical processes responsible for the observed transformations in ponds. The effects of both endogenous and exogenous (mainly meteorological factors) variables are also indicated.

The interactions between the planktonic and benthic sub-regions are described together with the importance of the benthic sub-system in the overall treatment process. The viability of ponds as biological waste water treatment systems is discussed in relation to observed homeostatic (self-regulatory) mechanisms which help maintain a stable pond environment.

A review of the various pond process models developed to date, including, empirical, steady-state and dynamic models, is presented and the main conclusions drawn are that the empirical and steady-state models are inadequate representations of pond systems and that the dynamic models based on pond ecology yield functional forms which are too complex to be verified with the currently available data.

Results of the identification exercise which centres around the use of exploratory models to evaluate hypotheses about overall

effect pond mechanisms are presented. Both Recursive and Block procedures for the estimation of states in state-space models are used for parameter estimation. Data from monitoring exercises on ponds located in Albuquerque, New-Mexico and Corrine, Utah in the USA are used for model verification.

Internal carbonaceous organics loading components are identified for both data sets. These are found to be closely related to the temperature/solar radiation annual cycle. The soluble portion of the internal loading is attributed to feedback from the benthic sub-region and the particulate portion is attributed to increases in algae population in the hot months.

A 2 cycles in the year component is found to be important in describing nutrient dynamics and this component is attributed to a combination of zoo-plankton/phytoplankton interactions and the spring and autumn instabilities observed in temperate ponds.

Areas of uncertainty in pond process representation are identified and recommendations on process descriptors to be measured in future pond monitoring exercises are presented.

The overall effects of pond dynamics are found to be well represented by simple linear models.

ACKNOWLEDGEMENT

I would like to thank my supervisors Dr. P.W. Jowitt and Mr. J.P.L. Lumbers for all the advice and encouragement in the preparation of the thesis.

I also wish to thank Dr. S. Cook for all the useful discussions, Mrs J. Andoh for her support and help with the typing work and all who in their own way contributed towards the successful completion of the work.

I would also like to acknowledge my indebtedness to the WHO who provided the necessary financial support.

FOREWORD

The natural world is not a haphazard jumble of constitutive components eg. soil, plants and animals, but rather a highly organised structure made of a multitude of interdependent resources interacting in such a way as to preserve stable forms.

Man by his activities interfere with these balancing forces and where the resulting perturbations are small, these are usually absorbed rapidly and the disturbance easily go unobserved. Pollution problems arise when the disturbance results in a marked shift from equilibrium conditions.

Only relatively recently has the need for conservation of natural resources, the abatement of pollution and consideration of environmental factors in development and planning become an important issue. It is more now than ever before in the history of man becoming evident that ecological systems cannot cope with many of man's activities (Odum, 1971).

By application of his technological capability to the use of resources, man has become an important force in global ecology.

To stem the growing environmental crisis, some control action needs to be implemented. The only way to maintain our planet is to keep a proper balance between ourselves, our resources and nature.

Negative feedback controls or homeostatic mechanisms are very evident in nature. Man has traditionally operated and still to an extent, operates under a feedback law. A control action is sought

and implemented only when an undesirable effect or state of affairs is observed. The trend in more recent times though has been that of purposeful application of knowledge and insight acquired from the study of natural systems and the dynamic interactions of their sub-systems.

Uncertainties produce a bias towards conservatism, towards routine ways of solving problems and towards doing nothing new. It is deemed that deliberate alterations or disturbances will be carried out now and in the future with less uncertainty at the same level of risk due to the better insights acquired from systems study. It must be stated however that an operational description of a system is required for the effective regulation of that system.

CHAPTER ONE: INTRODUCTION

1.1 GENERAL

Waste generation is associated with living and where people live in communities, some means has to be found for the effective disposal of the wastes generated within the community.

The philosophy behind wastewater treatment is that of effecting a change in the characteristics of the inputs to a waste water treatment facility such that the resulting outputs (effluents) from the system do not interfere with the intended use of the medium to which they are disposed off.

This work involves a systems identification of the Facultative Waste Stabilisation Pond System (FWSPS). This is a waste water treatment facility which can be considered to be a microcosm of nature in that it is an open system, consisting of a multitude of interconnected feedback loops, influenced considerably by exogenous variables which in the main are uncontrollable by man.

Waste Stabilisation Ponds have evolved as viable waste water treatment systems as a result of observations of improvement in the quality of ponded wastewaters. They have found widespread use in the United States (currently over 7,000 pond systems) and in developing countries where climatic conditions are particularly suitable for the use of such systems.

Pond systems have been used to treat a variety of wastewaters from domestic wastewater to complex industrial wastes and have been found to function under a wide range of weather conditions (from tropical to arctic). They enjoy some important advantages over more conventional waste water treatment systems because of their minimal power consumption, minimal operational and maintenance costs, relatively higher pathogen inactivation together with lower capital costs in general.

Owing to their immense volumetric capacities ponds are able to smooth peak hydraulic loads, deal adequately with seasonal industrial and agricultural wastes and produce effluents virtually free from pathogenic organisms.

1.2 UNCERTAINTIES IN DESIGN AND PERFORMANCE

1.2.1 GENERAL

It is perhaps because ponds are simple to operate and require very little in terms of routine maintenance that, until recently, they have suffered from a lack of comprehensive data on which to base any evaluation.

From results of the US Environmental Protection Agency (EPA) survey on the performance of Waste Stabilisation Pond Systems and their ability to meet EPA secondary effluent discharge standards, Mayo (1977) concludes that well-designed and operated ponds can meet various levels of effluent requirements and still retain the features attractive to small communities ie. low capital, operating

and maintenance costs. "Well-designed" here, however, does not refer to any specific design procedure.

Reports that ponds are not able to achieve required levels of treatment (Barsom and Ryckman, 1970) have usually been attributed to poor design and lack of adherence to rudimentary maintenance and operation requirements (Mitchell and Williams, 1982; McKinney, 1977).

Studies in the late fifties and early sixties (Neel et-al., 1961) showed that there was a dearth of information on basic principles and pointed out the need for controlled field-scale studies to resolve a number of fundamental considerations in pond technology. Since that, there has been considerable work done on both laboratory and field scale WSP units to determine how influencing factors mediate stabilisation. However, relatively little has been done to incorporate this knowledge into design and performance assessment of pond systems.

There are currently several unverified procedures for the design of pond systems giving vastly differing results for the required pond capacity under similar circumstances (Lumbers, 1979). It is clear from the literature that there is much variability in the design and performance of pond systems.

Finney and Middlebrooks (1979) in their work on the evaluation of facultative WSPS design found that none of the recommended procedures in the literature ranging from empirical organic and hydraulic loadings through to more rational design based on kinetics and estimated hydraulic characteristics yielded predictions that

could be substantiated by published pond performance data.

The lack of incorporation of pond information in the evolution of design procedures probably stems from the fact that often there are conflicting results present in the literature regarding the effects of the variables studied. Barsom (1973) has suggested that this lack of design evolution probably results from the general incorrect notion on the part of design engineers that the currently used procedures are adequate.

There is considerable doubt as to the usefulness or representativeness of most work on ponds, especially work using laboratory scale units. Reports on pond performance given at the 2nd International Conference on Ponds were based mostly on physical observations as assessment of performance.

A seminar held at the University of Leeds (UK) in April 1983 indicated that there is still considerable uncertainty on the part of both design engineers and researchers on pond systems regarding the most reliable design methods.

While there remains uncertainty as to the design and performance of pond systems owing perhaps to the fact that treatment objectives have not been well defined, there is no question that they remain viable alternative secondary waste water treatment facilities in many circumstances (Tariq and Ahmad, 1985; Oswald, 1972).

1.2.2 ON CRITERIA FOR ASSESSMENT OF PERFORMANCE

Conventionally the performance of a waste water treatment facility is determined from a comparison of the levels of biodegradable organic matter in the influent and effluent streams. The typical measure used is the 5-day BOD which is a measure of the amount of dissolved oxygen used in 5 days by bacteria converting the degradable organics into inorganic substances and new cells. This measure is an indication of the likely demand on the oxygen resources of a typical disposal medium - a water course.

In terms of the conventional measure, total BOD, the performance of pond systems has been found to be inadequate (Barsom & Ryckman, 1970). However, when comparison is made between the influent total BOD and effluent soluble BOD, pond performance is comparable to that of other biological wastewater treatment systems such as the activated sludge process. This implies that the major portion of pond effluent BOD arises from solids in the effluent. This in part accounts for the conflicting and contradictory results reported in the literature.

With regards the effectiveness of ponds as biological wastewater treatment systems, the important question that should be asked is what the solids are and what their effects are on the disposal medium (eg. receiving waters).

Though pond systems sometimes have an unacceptably high level of suspended solids in their effluents, these solids comprise mostly live algae cells which are much more stable than solids of sewage origin. The effect of these algae cells on a watercourse depends to

a great extent on the characteristics of the watercourse such as, flow rates, volumes, quality and on the state of the algae cells.

In terms of demand on the oxygen resources of receiving waters, these effects have been found to be small in some cases (Oswald and Ramani, 1976; Patrick, 1976). This is because the algae have survived in these waters and have continued to photosynthesize. Death and decay has been gradual resulting in the distribution of the ultimate oxygen demand load over a relatively long stretch of the water course. However in other cases, increase in BOD of receiving waters has been observed and this has been found to be caused by senescent or dead algal cells in effluents (Ward and King, 1976).

The relaxation of the effluent suspended solids limitations for wastewater treated in ponds by the EPA in 1977 was partly based on the evidence that in many instances, the release of live algae into receiving waters did not appear to create odours or adversely affect the receiving stream.

Also, the release of nutrients accumulated in algae to land irrigated with algal laden pond effluents is bound to be beneficial for agriculture or grazing (Strom, 1979; Searle and Kirby, 1973).

Most pond design procedures have been based solely on the consideration of the activity in the overlying waters in the pond basin. This has resulted in the erroneous conclusion that the worst performance (critical conditions) can be expected to always occur in the coldest month of the year. The complexity of the interactions of physical, chemical and biological processes with natural

environmental phenomena, eg. solar radiation, temperature and wind speed make it difficult to predict the quality of pond effluents on a day to day basis and even on a seasonal basis, the worst performance will not necessarily occur at one particular time of the year.

Critical conditions with regards pond performance are actually dependent on the criteria for assessment of performance and as such the worst pond performance will not necessarily always occur at a particular time of the year (Lumbers & Andoh, 1985). Examples of the likely times of the year where critical conditions may occur dependent on the criteria for assessment are given below:

BOD REMOVAL

The critical period for removal of carbonaceous organics for example, might correspond to a warm windless summer night. Under such conditions, there is no photosynthetic oxygen production. Dissolved oxygen saturation concentrations decrease with increasing temperature and heterotrophic activity increases with increasing temperature. Also, material feedback from the bottom deposits (the benthos) increases with increasing temperature and this coupled with the respiratory demand by algae (which bloom with increasing pond temperature) results in an additional internal loading to the pond system. It is interesting to note that of those who reported on pond performance for a worldwide survey conducted by the WHO (Gloyne, 1971), 36% reported problems in the summer months.

ODOUR CONTROL

Ponds located in different climatic environments are likely to have odour problems at different times of the year. Ponds located in temperate and much colder climates for example, tend to have odour problems in the fall and spring due to overturn whereas ponds located in tropical regions usually have odour problems in the summer (hot) months.

PATHOGEN REMOVAL

If the criteria for assessment are based on pathogenic organisms then critical conditions would generally occur in the colder months. Inactivation of enteropathogenic microorganisms decreases with decreases in both temperature and incident solar radiation.

NUTRIENTS REMOVAL

Pond performance has mainly been evaluated in terms of organics removal/transformation and pathogens inactivation/removal. Since algae for example use nutrients such as nitrates and phosphates from the waste waters in their photosynthetic activity thus removing these from the pond liquid, the quality of pond effluents in terms of nutrients is also important. This is especially the case when effluent discharge is to an eutrophying body of water. Nutrients removal would generally be expected to be greatest in the summer months when the maximum photosynthetic activity occurs.

The above suggests that pond performance should be related to the major treatment objectives and constraints imposed by the intended disposal medium eg. reuse for irrigation.

The potential for biological control of algae cells in pond effluents exists (Mitchell, 1980) and has been practised in the Far East for well over a century (Ghosh et al., 1985). China has succeeded in the treatment and utilization of waste water in the form of farmland irrigation and fish ponds on a considerably large scale. Algae represents a potential food source for organisms such as prawns and coarse fish higher up the aquatic food chain. These higher organisms can be used to control algae in the pond systems. Their grazing pressure results in the generation of healthy standing crops of algae which have greater survival potentials when discharged to a water body.

There are potential health problems associated with direct human consumption of prawns and fish produced in waste stabilization pond systems and as a result, direct human consumption is not envisaged. Rather these products can be used as a source of protein in the meals prepared for livestock. The success of such ventures represent the generation of recoverable by-products of significant economic value especially in the developing countries. This suggests that a useful additional measure of performance of pond systems would be its productivity in terms of the zooplankton occupying higher trophic levels in the aquatic food chain.

1.3 NEED FOR A SYSTEMS STUDY

If the view point adopted is that design formulations are derived from an understanding of process behaviour and that rationalisation of design is based on the evolution of this understanding enjoying greater reliability and less uncertainty with regards prediction of

system performance, then it can be appreciated that design has evolved from empiricism based largely on experience - to more rational (theoretical/mechanistic) approaches as knowledge has become available.

Development of better process models for design requires a better understanding of the system behaviour and the effect the system environment has on this behaviour. The vast number of contradictory design procedures for ponds suggests that:

- o Design and performance assessment criteria have not been well defined.
- o Relatively little has been done in terms of incorporation of knowledge acquired from pond studies in the evolution of better models.
- o No attempts have been made towards the generation of an information/data base on ponds consisting of up to date verified concepts on pond functioning; and towards unifying the approach for pond design.

It is recognised in general that for the majority of developing countries, pond systems are often the best practicable waste water treatment process available. Taking into account such factors as technological development, the level of operator skill, availability of land, climatic type and the particularly high degree of pathogen die-off occurring through a series of ponds, pond systems stand out as the first choice for biological waste water treatment in developing countries.

It stems to reason that in the long term considerable investments are going to go into the installation of pond systems in several developing countries in the world. Taking into consideration the severe financial constraints these countries face, it is clear that over-design, resulting mainly from lack of information, will not be tolerated. This clearly spells out the need for the development of design procedures which have more reliability with regards prediction of pond performance with the changing seasons of the year.

The above deliberations suggest that there is a need for the adoption of a systematic approach to the study of pond systems with the view to the evolution of more meaningful process models which will help identify data needs in a progressive fashion and aid in the rationalisation of design.

The general methodology broadly classified as the "SYSTEMS APPROACH" to a study is the application of the branch of science concerned with the identification of systems and optimal utilization and allocation of resources. The systems orientation may be described as a way of looking at the system as a whole. This is presently not a clearly defined discipline but rather takes on a combination of theories developed in applied fields of "Systems Analysis", "Operational Research", "Estimation and Control Theory" and others. It is usually not apparent a priori which disciplines will make the major contribution to the study.

The systems approach results in the gain of information on the system under study resulting in improved knowledge about the system

and its processes. This improved knowledge enables a better description of the system and this is usually reflected in improved design or approach to design.

Since ponds are dynamic systems, the systems study should evolve around the use of systems analysis techniques for investigating the dynamics of processes.

1.4 THE STUDY

1.4.1 PRIMARY OBJECTIVE

In very broad terms, the research work involved the application of a SYSTEMS APPROACH to the study of FACULTATIVE WASTE STABILIZATION POND SYSTEMS. The primary objective was that of evolving more rational approaches to the design and evaluation of pond systems based on the better insight acquired on the behaviour of the system resulting from the systems study.

1.4.2 SCOPE OF STUDY

1.4.2.1 GENERAL

In the study of systems, the ideal would be to investigate complete full scale representations of the system under consideration. This however involves considerable cost and in some cases, is quite impractical. Often either scaled down versions called physical models or mathematical algorithmic representations called mathematical models are used in the study of the system.

1.4.2.2 USE OF MATHEMATICAL MODELS

Mathematical models have more appeal in that they save on costs, as compared to physical models, and present a more flexible approach to a modelling exercise. The development and utilization of mathematical models provides a versatile superstructure for integrating the contributions from different disciplines and for incorporating data and research results from many fields. Mathematical modelling permits the comparison of a wide range of candidate models and the selection of the more appropriate from the range based on the objectives of the exercise. Mathematical models are therefore used in this study.

1.4.2.3 EXPECTATIONS

When a piece of scientific description is very highly developed it may be thought of as a model; successful models not only describe the phenomena occurring but also appear to explain them. If the mathematical model of a system is fully representative, it may be held to be a law of the nature of the system.

To date, none of the applications of mathematical modelling to ecological systems, which by their very nature are exceedingly complex large scale systems, has reached the status of say Ohm's law for electrical systems. They are nonetheless in the tradition of scientific modelling and more insight on these systems has been obtained from attempts at subjecting them to established mathematical classification schemes.

It cannot be hoped therefore from one systems study effort, to

produce the optimum design for pond systems based on the evolution of a complete description of the system. Taking into consideration the present level of knowledge on pond systems, which can be described as ill-defined exceedingly complex ecological systems, we are far from that situation. There is however the need for a concerted effort at unifying the search for better models. Insights gained from recognising the limitations of the existing models should provide pathways for the evolution of more representative systems models.

1.4.2.4 APPROACH TO THE STUDY

Below is a schematic presentation of the adopted approach to the systems study. The basic underlying philosophy being that: "Decision making takes place under conditions of uncertainty and uncertainty can only be reduced by obtaining unbiased information".

THE STUDY

ACQUISITION OF TOTAL A PRIORI KNOWLEDGE (REVIEW OF AVAILABLE LITERATURE AND ATTAINMENT OF OBSERVATIONAL DATA SETS)...

ANALYSIS OF DATA FOR INPUT-OUTPUT RELATIONSHIPS THAT MAY THROW LIGHT ON CAUSAL RELATIONSHIPS NOT DIRECTLY EVIDENT FROM A PRIORI KNOWLEDGE...

REVIEW AND APPRAISAL OF MODELS OF POND SYSTEMS DEVELOPED TO

DATE...

INTERNALLY DESCRIPTIVE ANALYSIS BASED ON A CONCEPTUAL
DETERMINISTIC MODELLING APPROACH AND STOCHASTIC ESTIMATION
TECHNIQUES...

UPDATE OF KNOWLEDGE ON POND SYSTEMS BASED ON INFORMATION ACQUIRED
FROM STUDY...

UTILISATION OF THE TOTAL KNOWLEDGE ACQUIRED IN THE DEVELOPMENT OF
GUIDELINES FOR THE EVOLUTION OF MORE RATIONAL DESIGN CRITERIA AND
RECOMMENDATIONS FOR MORE EFFECTIVE OPERATION AND MAINTENANCE...

1.5 END POINT

Formulation of a model of the pond system will provide a basis for understanding the pond system and more broadly, of its effects on the ecology of disposal media. If an identified model is a valid expression of a typical pond system, it can be used in predicting consequences of expected range of conditions to which such systems are likely to be subjected to.

It is hoped that results of simulations based on scenarios construed from likely distributions of quality and design parameters would be instrumental in determining the future courses of action in pond experiments, pond evaluation and operational needs to effect the discharge of acceptable effluents throughout the changing seasons of the year.

In a systems study of ponds with the view towards optimization of design, it is pertinent to first describe in a very simplistic fashion what waste stabilization pond systems are, and then review the various design procedures and mathematical models that have been proposed as models of pond systems.

CHAPTER TWO: DESCRIPTION OF POND SYSTEMS.

2.1 GENERAL

"Waste Stabilization Ponds" WSP is a term that is used to classify a broad range of waste water treatment facilities and generally describes any shallow man-made basin utilizing natural biological processes for the reduction of organic pollutants and the destruction of pathogenic organisms in waste water (Marais, 1970; and Gloyna, 1971).

The various pond types that have been developed include:

- o Anaerobic ponds :- In which anaerobic fermentation processes predominate.
- o Facultative ponds :- In which anaerobic fermentation, aerobic oxidation and photosynthetic oxygen producing and reduction processes occur.
- o Aerobic ponds :- In which aerobic conditions are maintained throughout the entire pond basin.
- o Aerated lagoons :- In which artificial aeration is used to supplement the oxygen resources of the pond. These can either be facultative or aerobic, fully or partially mixed.

These pond types are classified primarily according to the nature of the biological activity taking place.

A description of terms used to classify the various types of pond systems is given by Oswald (1975) and a discussion on them can be found in Eckenfelder and O'Connor (1960, 1961), Eckenfelder (1966) and Gloyna (1971). The majority of pond systems in use fall into the category of facultative ponds. Typical characteristics of the various types are given in Table 2.1.

In a sense, facultative ponds are extensions of the natural environment. Their function is to allow to proceed in a controlled volume what would normally occur in a receiving water course. They can also be viewed as similar to organically enriched natural ponds and lakes and as such considered as aquatic ecosystems.

Both aerobic and anaerobic processes take place in the typical facultative pond system and as a result, facultative ponds feature the dominant treatment mechanisms in the other pond types. Being the most common type of pond system, they are the pond type on which most information is available.

This systems study is based on facultative waste stabilization ponds (FWSP), and subsequent reference to the term "pond system" would actually mean "facultative pond system".

The physical, chemical and biological processes which control the transport and transformation of constituents in pond systems are many, varied and highly complex. This complex nature allows a large number of possible interactions between the biotic and abiotic components and is compounded by the effects of natural environmental variations. Unlike other wastewater treatment processes eg. activated sludge, stabilization ponds are subject to significant

influence by meteorological conditions which are in the main, stochastic in nature and not completely understood in their effects.

The development of a meaningful process models of pond systems requires understanding and familiarization with the pertinent information available on pond systems.

A description of ponds as a waste water treatment facility is given together with a review of the available information in the literature on the processes occurring and how they are deemed to be influenced by both endogenous and exogenous factors.

2.2 WASTE WATER TREATMENT IN PONDS

The primary objectives of biological wastewater treatment are to reduce the organic pollutorial load and pathogenic micro-organisms in the waste stream to acceptable levels. In broad terms, the following are the processes that occur in treatment works using more conventional biological treatment systems eg. activated sludge and percolating filters.

a) Primary solids-liquid separation processes eg. screens, grit chambers and primary sedimentation units. These result in the removal of gross particulate material from the waste stream and the resulting primary sludge is usually sent to anaerobic treatment units such as digesters.

b) The waste stream is then brought into contact with micro-organisms in either a suspended growth or attached growth environment. These micro-organisms are able to transform the unstable organics to innocuous inorganic material and new microbial

cells.

c) The new cells together with dead and dying cells are then subsequently separated from the waste stream by solids-liquid separation devices eg. secondary settling tanks. The resulting sludge is partly recirculated in the case of the activated sludge system or is sent to the anaerobic treatment units previously mentioned.

d) The sludge from the digesters are usually either incinerated or dried and then transported for ultimate disposal.

In contrast, all the above stages except for transportation of sludge for ultimate disposal, occur in one unit (the basin) for pond systems. A flow through diagram showing the fate of the organic pollutional load through a typical pond system is shown in Figure 2.1.

A significant proportion of the suspended material in the influent stream is settleable and settles out to the bottom of the basin. Dead microbial cells and the faecal pellets of the higher protists (grazers) also settle out to the bottom of the pond system where together they form a sludge layer. Depending on the arrangement of the inlet section, there is the tendency for the formation of sludge banks at the inlet region. Materials settling out in regions further away from the inlet tend to be made up mostly of microbial cells and faecal pellets of higher protists.

The resulting sludge layer or sediments accumulating at the bottom constitutes a separate subregion, "the benthos", which in its

function and behaviour is analagous to the anaerobic digesters in the more conventional treatment processes (Rich,1982; Rich and Connor, 1982). Conditions in this layer range from facultative at the sediment-water interface to anaerobic in the deeper layers (see Fig 2.2). Products of the degradation activity in the benthic region (intermediate fermentation products) slowly diffuse from the deeper layers to the sediment-water interface as the sludge consolidates.

The intermediate fermentation products are rich in both carbonaceous organics and inorganic nutrients and depending on the transport characteristics established in the hydraulic regime, these often constitute an additional source of pollutional load to the overlying planktonic region.

Degradation of organic material is brought about by a complex system of nutritional inter-relations between the organisms colonizing the planktonic region and the underlying benthic region. The soluble and colloidal portion of the organic pollutional load in the planktonic region resulting from both the influent flow and feedback from the benthos, are degraded by pelagic organisms, mostly bacteria, which with the aid of exogenous enzymes use the organics for the derivation of the energy required to maintain life functions and for the regeneration of new cells. In the process, carbon dioxide, inorganic nitrogenous and phosphorous compounds are released which together with the nutrients in the influent stream and those fed back from the benthos add on to the nutrient store of the pond system. These nutrients are used by the primary producers, mostly algae, in their photosynthetic activity producing oxygen and new

algal cells.

Reduction of pathogenic organisms in some conventional treatment processes is usually achieved by disinfecting the resulting effluent with chlorine. This is because the flow through times are relatively short and as such significant decrease in numbers of pathogenic organisms through natural die-off is not achieved. In contrast, stabilization ponds have long liquid detention times and this results in amoebic cysts and helminthic ova settling out into the benthic region. The adverse environment created by the competition for food by the diverse organisms colonizing both the benthic and planktonic regions and the disinfecting power of the ultraviolet component of sunlight, results in the considerable die-off of enteropathogenic organisms.

The degree to which the original organic material in the waste stream is transformed and the reduction in numbers of pathogenic organisms achieved depends, amongst other factors, on the average liquid detention time in the pond system. This detention time depends on the hydrodynamic regime established in the pond system and is influenced considerably by factors such as convective mixing, wind induced mixing, thermal stratification and short circuiting.

Because of the long detention times in ponds the community structure developed depends to a great extent on the waste characteristics. The micro-organisms that grow in the pond system are a function of the organic matter in the wastes (Walker and Leclerc, 1975) and the succession of the various populations is influenced considerably by the cycling of nutrients within the system and by meteorological factors eg. solar radiation and

temperature (Young and King, 1980).

The dominant transformations occurring in the planktonic region of the typical facultative pond system exhibit significant diurnal variations. This is partly because photosynthetic activity which is the major source of dissolved oxygen required for aerobic stabilization, occurs only in the daylight hours. The continual alteration of aerobic/anoxic conditions enables a wide range of microorganisms to survive such that; aerobic carbonaceous bio-oxidation, aerobic ammonia nitrification and anoxic denitrification can all occur in the pond basin.

Lack of significant amounts of dissolved oxygen during the night time results in the creation of microaerophilic conditions throughout most of the planktonic region. The resulting reduction in the oxidation-reduction potential (ORP) results in the use of nitrate as the terminal electron acceptor giving rise to denitrification.

Denitrification most probably accounts for the unexplained high level of nitrogen gas found in gases from pond sediments by Brockett (1977) and also must be the mechanism that accounts for the increased reduction of total nitrogen in deeper ponds observed by Oswald (1972).

Redy (1983) using the ammonium isotope ($^{15}\text{NH}_4^+$) as a tracer found that 21% of the total nitrogen went to the bottom sediments, 5% was contained in the algae, 21% remained in the liquid and 53% was unaccounted for. The 53% unaccounted for represents more than half of the total nitrogen removed in the pond system and probably

resulted from the loss of nitrogen gas through denitrification.

In cases of severe reduction in the ORP, sulphate reduction takes place resulting in the production of sulphides with associated odour problems. It has however been observed that as long as a microlayer rich in oxygen exists at the top of the pond system, odours are not produced.

Ponds exhibit considerable spatial and temporal heterogeneity on account of such factors as:

- o The separate regions in the pond basin viz. planktonic and benthic.
- o Vertical migratory movement of motile algae resulting in algal bands and the creation of a photic zone.
- o Stratification (Thermal/Nutrients)
- o Settling of readily settleable suspended solids in the vicinity of the inlet region resulting in more rapid bioactivity in that region.

The above exposition of ponds as wastewater treatment systems clearly shows that ponds combine the primary, secondary, tertiary and sludge treatment processes in conventional treatment processes using eg. activated sludge and trickling filters in one process unit, the pond basin.

2.3 PROCESSES OCCURRING

2.3.1 GENERAL

The chemical, biological and physical processes occurring within the pond system; how they effect organic and nutrient transformations; the role of the sludge layer; the influence of thermal stratification and mixing and how these processes are deemed to be influenced by meterological factors together with the fate of pathogens in ponds, is presented below.

2.3.2 CHEMICAL PROCESSES

Chemical reactions are relatively rapid and as a result, inorganic species in pond systems are usually in chemical equilibrium. The typical domestic/municipal type wastewater contains innumerable numbers and diversification of species of chemical entities. The micronutrients essential for cell metabolic activity and synthesis are usually present in such quantities that they are not growth limiting.

In the absence of an adverse environment created by the lack of micronutrients or the presence of toxic chemicals, the macronutrients (inorganic carbon, nitrogen and phosphorous compounds) used by the primary producers (mostly algae) in their autotrophic activities together with the influencing environmental phenomena (eg. solar radiation) are the factors which interact to limit oxygen production through photosynthesis.

The inorganic species present in pond systems result from the contributions of the influent waste stream and that resulting from the recycling of nutrients within the pond system. Mineralization of

organic substances within both the planktonic and benthic subregions results in the release of inorganic chemical species.

Cell lysis and excretory products of microbial cells result in the release of low molecular weight compounds (Juither and Matuschek, 1978) which may be physiologically active and can either stimulate or inhibit growth and development (King, 1976).

The range and type of chemical species present determine the ionic properties of the pond fluid such as its conductivity and pH and also the buffering capacity of the pond system. The three nutrient systems forming the predominant chemical buffer are: $\text{CO}_2 - (\text{HCO}_3)^- ; \text{NH}_3 - (\text{NH}_4)^+ ;$ and $(\text{HPO}_4)^{2-} - \text{H}_2(\text{PO}_4)^-.$

The carbonate buffering system in particular plays a very significant role in pond functioning since it influences the availability of dissolved carbon dioxide which is deemed to be the main inorganic carbon species used by algae in their photosynthetic activity (Nichols, 1973). This system also helps maintain pH within the neutral to slightly alkaline range (7-10) and prevents excessive algal growth in blooms through a reduction in the availability of carbon dioxide (King, 1970; Novack and Brune, 1985). This is achieved by the uptake of available carbon dioxide to satisfy chemical equilibrium reactions which are relatively more rapid than the physical uptake involved in the utilization of CO_2 by algae. Also as pH rises above 9, bacterial activity begins to diminish causing a reduction in CO_2 production and thereby limiting subsequent algal growth.

Comparing the stoichiometry of an algal cell with the relative

abundance of macronutrients in domestic/municipal type waste waters (see Table 2.2), it is realised that carbon is the most likely growth limiting nutrient. Carbon limitations on photosynthetic activity has been suggested (Bartsch and Allum, 1957) and observed in nutrient enriched waters (Wright and Mills, 1967; and Goldman et al., 1974).

During periods of intense photosynthetic activity, high pH results in the precipitation of phosphates (as hydroxyapatite) and the hydroxy complexes of some heavy metals. These precipitates tend to enmesh other suspended particles as they settle to the bottom of the pond system, resulting in a further removal of material from the planktonic region. The anoxic conditions in the sediment/water interface however results in dissolution and desorption of these precipitates. The extent to which they are removed from the pond liquid thus depends on the net effect of these opposing mechanisms and the hydrodynamic regime established in the pond basin. Significant removals of heavy metals has been observed in pond systems (Hussainy, 1979) and this tends to suggest that the bias is towards permanent entrainment in the benthos.

Ammonia is of particular importance in pond systems because, though it is a macronutrient for algae, in its unionized form, it is toxic to algae and zooplankton at high concentrations (Abeliovich & Azov, 1976). The distribution of species between the free (NH_3) and ionised (NH_4^+) forms is dependent on the pH. At high pH (> 10) virtually all of the ammonia present is in the unionised form. Thus a combination of high ammonia and pH inhibits photosynthesis resulting in a reduction in pH (since bacterial activity

predominates then), and a consequent shift from the toxic unionised form to the ionised form.

2.3.3 BIOLOGICAL PROCESSES

2.3.3.1 GENERAL

The biological entities responsible for transformations occurring within the pond system are mainly bacteria, algae, higher protists and small animals (Bopardikar, 1969; and Goulden, 1976).

The main role of bacteria is that of conversion of unstable organic substances to relatively inert inorganic substances, the energy released in the process partly being used for the synthesis of new cells and the maintenance of life functions.

Algae are mostly responsible for the photosynthetic production of the oxygen required by the aerobic organisms (Bartsch, 1961). In the process, they use up the inorganic nutrients such as carbon dioxide, ammonia and phosphates.

The role of the higher protist and small animals is more that of clarification than of purification. They achieve this by grazing on the bacterial and algal populations.

Traditionally, the efficacy of wastewater treatment units has been measured in terms of the reduction in the energy content of the influent stream achieved. This has resulted in the description of pond functioning in terms of the symbiotic relationship between algae and bacteria. The role of zooplankton, macrophytes and

detritus feeders higher up in the food chain has thus largely been considered as of secondary importance. This, most invariably, has been because the higher organisms command only small amounts of the total energy and matter flow through the pond system. A typical flow through diagram for a pond system showing the energy pathways is shown in Figure 2.3.

In terms of nutritional relationships in the typical food web, it takes quite a considerable number of organisms lower down the food web to effect a significant change in mass in higher organisms. This in effect means that though these higher organisms produce very little energy destruction compared to the decomposers, they are important in as far as control of the lower organisms are concerned. This fact is well demonstrated in the reciprocity law of causal effect by Pattern and Cale (1974).

Microbial metabolic activity can be classified as aerobic, anoxic or anaerobic depending on the presence or degree to which oxygen acts as a terminal electron acceptor. Aerobic and anoxic (microaerophilic) processes are more efficient than anaerobic processes in terms of energy destruction and as such as long as there is oxygen present, anaerobic processes do not occur.

In pond systems oxygen for promoting aerobic stabilization is derived mainly from that produced by the photosynthetic activity of the producers (mostly algae).

2.3.3.2 ALGAE

There are a wide range of algae types reported in the literature

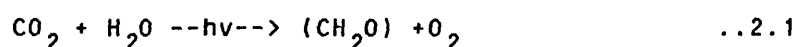
on pond systems. The most common types are listed in Table 2.3. The list though representative is by no means comprehensive. Climatic conditions have little influence on the occurrence of species but the diversity of algal flora depends quite definitely on the physical and chemical conditions prevailing in the system. The algae types found in ponds belong to three general groups: The motile algae; the non-motile small algae; and the filamentous algae.

The motile algae are capable of vertical migratory movements which enables them to reside in regions where conditions are favourable and as such can compensate for varying light conditions. Euglena and Chlamydomonas are the most frequent motile algae in ponds. The non-motile species eg. Chlorella and Scenedesmus have no powers of locomotion and as such drift passively. They tend to be small and usually have a large surface area to mass ratio so that they remain suspended with very slight fluid motion. The filamentous algae usually form floating mats which are undesirable in that they block light transfer, prevent surface reaeration and wind induced mixing. The most common type found in pond systems are, Oscillatoria and Anacystis.

Though it is quite impossible to predict the numbers and type of algae that will be present in a pond system at any given time, general trends and patterns in the succession of the various species with the changing seasons have been observed.

Algae synthesize new cells from the inorganic nutrients available in the planktonic region of pond systems. The energy required for this synthesis is derived from solar energy (when light intensities

are in excess of about 0.002 ly/min). The resulting process can be represented by the following simplified chemical equation:



Photosynthesis does not usually occur until a threshold irradiance, the compensation point, where photosynthetic oxygen production equals that required for respiration. Thereafter, the rate is proportional to irradiance until this reaches a region of a twentieth of full sunlight when light saturation occurs and the rate becomes independent of irradiance. At higher irradiance, usually over half full sunlight, photosynthesis becomes inhibited and the rate falls off. The typical response of an algae cell to solar radiation is shown in Figure 2.4.

As it penetrates through water light becomes attenuated and altered in its spectral characteristics, according to the Beer-Lambert law. Suspended material tends to reflect and absorb light depending on their optical properties and this results in a further attenuation. On account of these, photosynthesis varies with depth and a typical curve showing this variation for phytoplankton uniformly distributed in the vertical direction is as shown in Figure 2.5. The layer within which photosynthesis is possible is termed the photic zone and it cannot be defined precisely since the lower limits continuously shifts according to the time of the day and other conditions.

Dissolved oxygen concentrations vary accordingly throughout the day in accordance with the variations in photosynthetic activity.

Hartley and Wiess (1970) observed diurnal vertical migration of Euglena rostrifera with maximum populations being centered at those depths where the light intensity had diminished to 75 cft^{-2} . This depth selection appeared to be in order to avoid higher light intensities. There was very little light penetration beyond the algal bands and the vertical distribution of dissolved oxygen tended to be related to the vertical migratory movement of the algae.

Investigations into the typical variations of dissolved oxygen with depth and time of day for varying weather conditions have been undertaken by King (1976) and Marais (1970). Typical distributions in the planktonic subregion of pond systems are shown in Figs 2.6 and 2.7.

2.3.3.3 BACTERIA

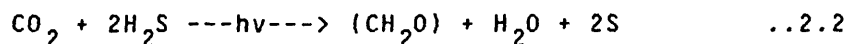
From the literature on the bacteriology of pond systems (Gann et al., 1968) it would seem that the majority of bacteria belong to the Achrombacter, Pseudomonas and Flavobacterium groups. The overall environment determines which bacteria grow and to what extent. The bacteria in the planktonic region are able to metabolize aerobically as long as there is dissolved oxygen. They are essentially maintained in a dilute, dispersed condition and the rate at which organics are utilized depends on the opportunities for contact between the bacterial cells and the molecules of the organics.

Under aerobic conditions conversion of soluble organics to microbial cells is usually complete in 3 to 5 days with endogenous respiration and predation by protozoa, rotifers and crustaceans

accounting for the reduction of bacterial cell mass.

Anoxic/microaerophilic conditions occurs below the photic zone and usually in the interface between the planktonic and benthic subregions. Facultative organisms under these conditions, use nitrate as their electron donor for deriving the energy required for cell synthesis and maintenance. Denitrification occurs in the anoxic zones and this has tended to mask the effects of nitrate production through nitrification in pond systems. The resulting low levels of nitrate observed in pond effluents probably accounts for the widespread notion that nitrification does not occur in ponds.

Overloaded pond systems are usually typified by the creation of anoxic conditions throughout the entire planktonic region (McKinney, 1968; Miller and Belfort, 1977). Under such conditions, photosynthetic bacteria eg. the green sulphur bacteria carry out photosynthetic activity with the production of sulphides. This process can be simply represented by:



2.3.3.4 HIGHER PROTIST AND SMALL ANIMALS

The higher protist and small animals eg. protozoa, rotifers and crustaceans reduce the turbidity of the effluent stream by feeding on the bacterial and algal cells. This results in a reduction in both the BOD and suspended solids concentration of the effluent stream.

The population dynamics of the higher protist and small animals in pond systems has not been studied extensively because their role in pond treatment has all along been considered to be secondary to the algal/bacterial symbiotic interactions (Mitchell, 1980). Grazing pressure by *Daphnia*, a crustacean, for example is known to terminate algal blooms in the summer months resulting in good quality effluents (Mitchell and Williams, 1982).

Studies on zooplankton in ponds (Wennstrom, 1955; and Afanas 'Eva and Messineva, 1981) found protozoa to be dominant in the cold months whereas the multicellular animals grew readily during the remainder of the year. Vorticella was the major stalked form of protozoa while Paramecium was the primary free swimming ciliated protozoa. Glaucoma, Euplotes and Colpidium were the other types also found to be present in large numbers.

Rotifers and the Crustacean, Daphnia were found to be active in the summer months. Daphnia ate most of the algae in the final pond producing an effluent essentially free of algae.

2.3.4 PHYSICAL PROCESSES

2.3.4.1 GENERAL

Most of the physical processes influencing pond behaviour, do so through their effect on transport. The interactions between biotic and abiotic components of the waste stream are dependent on the opportunities for contact in the pond basin and are thus dependent on the general transport characteristics established in the suspending fluid.

Stratification, on account of the special role it plays in pond functioning, is discussed separately whereas the other physical processes are discussed under meteorological factors.

2.3.4.2 TRANSPORT

The physical processes effecting transport in the pond basin include such processes as advection; longitudinal, lateral and vertical dispersion, sedimentation, resuspension of particulates, seepage and atmospheric reaeration (or gaseous diffusion).

These considerably influence the nature of the hydraulic regime in the pond system. The hydraulic regime (level of mixing and flow pattern established) determine to a large extent the transport pathways of the macromolecular constituents of the waste stream as they flow into, through and out of the physical boundaries of the pond system.

The hydraulic regime also determines the effective retention time of the system, the degree of short-circuiting and backmixing and rate at which surface waters are renewed; and hence, the surface reaeration rate. To a large extent, the hydraulic regime is also influenced by the arrangement of inlet and outlet structures and the nature and types of baffles (Mangelson and Walters, 1972; Kilani and Ogunrombi, 1984).

2.3.4.3 METEOROLOGICAL FACTORS

The main meteorological factors influencing pond performance are:

Solar Radiation; Temperature; Evaporation; Wind; and Precipitation. Solar radiation and temperature, on account of their strong seasonal character and their effects on the biological processes occurring, exert the most significant influence on pond systems. The relationship between sunlight and algal activity has already been discussed under biological processes (section 2.3.3.2). The influences of the other meteorological factors on pond performance are briefly discussed individually.

Temperature:- Temperature is the single most important factor influencing the performance of pond systems. Biological activity generally increases with increases in temperature, with the optimum growth for the different biological entities colonizing pond systems occurring at different temperature ranges. This results in the succession of the various species of algae, bacteria and zooplankton. Temperature also determines to an extent the nature of the hydraulic regime established. This is because of its role in stratification and in the creation of convective currents. Temperature also has a significant effect on the anaerobic activity in the benthos. Increases in sludge temperature above 15 °C generally results in an increase in the generation of methane.

Evaporation:- Evaporation is of particular importance in ponds located in arid and semi-arid regions. Excessive evaporation results in a considerable loss of water to the atmosphere resulting in the concentrating of constituents of the pond liquid. This results in a higher concentration of these constituents in pond effluents than would otherwise be observed. Evaporation results in the cooling of pond surface waters and hence aids in the creation of convective

currents. Together with precipitation, evaporation determines to a large extent the effective retention times of the non-volatile components of the waste stream.

Wind:- High wind speeds create surface waves which produce turbulence at the water surface and tend to stretch out the pond surface. This increases the rate of transfer of gases across the pond surface (Banks and Herrera, 1977). The waves created also induce mixing countering the effects of stratification. Wind however, cannot be relied upon as a source of dissolved oxygen through surface reaeration because of its stochastic nature.

Rainfall:- Any mechanism which creates turbulence at the pond surface will tend to increase the rate of oxygen transfer across the surface. Oxygen transfer due to rainfall, is directly proportional to the power of the rainfall. For example, a rainfall of intensity 10mm/hr establishes the same value of the oxygen transfer coefficient as would a wind with velocity 6.5m/sec (Banks et al., 1984). Rain dilutes the contents of the pond basin and in this respect has an opposite effect to that of evaporation.

2.3.4.4 STRATIFICATION

Water bodies in general stratify when stable density differences are created. This often results from surface heating (with the establishment of a thermocline). A thermocline greatly inhibits mixing between the surface layers (where oxygen is being produced), and the zones beneath. Thermal stratification also brings about stratification in the other chemicals (nutrients) and in the physical characteristics of the water including O_2 , pH, alkalinity

and conductivity (Danos et al., 1983).

If the thermocline persists for long enough, the region below is exhausted of its oxygen resources resulting in anaerobic conditions. The organisms that proliferate under such conditions include the various chemosynthetic bacteria which can use methane, hydrogen sulphide, sulphur and other reduced compounds, and inorganic substrate in the fixation of carbon.

When thermal stratification is extreme, almost all of the drop in temperature occurs in the upper 30cm of water and the region below the thermocline assumes the long term average temperature of the earth in the particular environment. Theoretically, a thermally stratified pond contains less heat than an unstratified pond and as such its mean temperature is less if other conditions are equal (Hendricks and Pote, 1974).

On account of stratification, a significant portion of the pond fluid does not experience the extremes in daily temperature especially in the hot periods (see Table 2.4). Fortunately the region where the extremes of temperature are manifested (ie. above the thermocline) is usually where active photosynthesis results in supersaturation of dissolved oxygen (O_2).

Since the activity of heterotrophic bacteria about doubles with every $10^{\circ}C$ rise in temperature, it would seem that stratification is a mechanism whereby the pond system gears itself into the creation of ecophysically diverse habitats each properly suited to its role: High pH and O_2 at the top (above the thermocline); neutral pH and a range of low O_2 to microaerophilic conditions in the planktonic

region below the thermocline; and neutral to low pH and anoxic/anaerobic conditions at the bottom (in the benthos).

Usually, incoming sewage has a temperature higher than the long term average temperature for the locality. This results in the hotter, less dense inflow sewage streaming on top of the water beneath the thermocline. The soluble readily degradable portion of the influent waste stream is rapidly taken up by the bacteria which have an abundant supply of dissolved oxygen. The suspended portion settles out of the top layers into the lower layer where the neutral to low pH conditions prevailing aid in their hydrolysis into more readily useable forms.

Also, the prevention of mixing between the oxygen rich upper layers and the pond bottom region helps maintain methane fermentation. Dissolved oxygen inhibits the methane formers since they are strict anaerobes.

Depending on the inlet outlet arrangement, if severe short-circuiting or channeling of flow does not exist, then the detention time required for the conversion of the readily degradable portion to innocuous inorganic material is achieved.

Critical conditions with regards to the carbonaceous organics quality are likely to occur in the night where cooling of the surface waters results in the generation of a convective current leading to overturn. This results in the mixing of the entire contents of the planktonic region of the pond system. If sufficient nitrate or other terminal electron acceptors are present such that the ORP does not fall to levels where sulphate reduction is possible, then problems

of odour barely exist.

The turnover, however, is a necessary mechanism because it is a means by which the nutrient depleted surface waters are replenished with an internal supply of the inorganic nutrients required by the algae. Also the suspended organics, which have had time to hydrolyse, and the intermediate fermentation products, are transported into the surface region where they are used up in the daytime by aerobic and facultative bacteria.

This cycling between a stratified pond in the day and overturn in the night is observed on a daily basis in tropical ponds whereas this occurs only in the summer in temperate ponds.

Thermal stratification enables aerobic conditions to be produced even when the total oxygen demand of the pond is greater than the oxygen produced. As a result, odoriferous compounds produced by anaerobic decomposition in the lower layers are oxidised in the aerobic zone.

2.4 THE BENTHIC SUBREGION

Early concepts on pond functioning were based on the notion that the majority of the compounds settling out from the planktonic region to the benthic region were buried in the bottom with practically no return to the water columns. The effects of the benthic subregion over the overlying planktonic region has been recognised for over two decades but not taken into consideration in the design of pond systems. It is only in recent years that attempts are being made to develop quantitative expressions for the activity

in this layer and to link its effects on the overall treatment efficiency of ponds and other ecological systems (Bryant, 1983; Walker and Snodgrass, 1986; and Spies, 1984).

The benthic subregion essentially acts as a source and sink of organic and inorganic constituents of waste in the pond system. This region is made up of gross particulate material settling out from the overlying planktonic region namely; the settleable solids in the influent waste stream, microbial cell mass and faecal pellets of organisms colonizing the planktonic region.

The interface between the planktonic region and the top of the benthic layer is a region where compression settling takes place, the lower layers undergoing consolidation and subsequent compaction.

Conditions in this region are basically anaerobic except for the occasional microaerophilic conditions that may exist in the interface when significant mixing occurs within the planktonic region. Mortimer (1976) suggests that the contents of dissolved oxygen in the bottom waters is the main factor regulating the rate of release of nitrogen, phosphorous, iron and manganese from sediments into water. When the ORP of the water adjacent to sediments reaches about 0.24V, active diffusion of phosphates and also of divalent iron and manganese as a consequence of reduction of the hydroxides and breakdown of complex compounds begins. In this period, the sediments can be a substantial "source" of release of materials into the water.

Organic material in the benthos undergo anaerobic decomposition resulting in the release of intermediate soluble fermentation

products which diffuse into the overlying planktonic region and exert an oxygen demand. Consolidation of the deposits as they decompose also results in the feedback of soluble fermentation products by forcing interstitial waters into the overlying waters.

During periods of intense anaerobic activity, gas production causes further disturbances via sludge lifting. The gases produced, namely methane and carbon dioxide, tend to buoy settled organics resulting in the resuspension of particulate organic material.

The rate at which the settled organics are decomposed is very dependent on the activity of the benthic microbial community and macro-invertebrate population of borers and detritus feeders. This activity is highly temperature dependent and below 15°C is negligible.

2.5 PATHOGENS

Several studies have been conducted to identify the causes of bacterial reduction in ponds and a review is given by Amin and Ganapati (1972). These include the high pH levels common in ponds (Parhad and Rao, 1974); the production of toxic extracellular compounds by algae (Davis and Glyona, 1972); the exposure to the sun or ultraviolet light (Moeller and Calkins, 1980) and depletion of nutrients (McGarry and Bouthillier, 1966). It would seem from these studies that all the above factors contribute to the significant reductions in pathogenic organisms in waste streams through pond systems.

Natural die-off is temperature dependent and on that basis, the

worst conditions pertaining to removal of pathogenic organisms occurs in the colder periods. Balieux and Troussellier (1983) in their work on changes in faecal indicator bacteria, Pseudomonas aeruginosa and Aeromonas hydrophilia in pond systems found that the maximum rate of bacterial reduction occurred during the summer periods.

The difference in the rates of reduction between the hotter and colder periods is fortunately not very much. This is because the effective retention time in the colder periods is greater due to the absence of stratification. Also, the majority of the pathogenic organisms are obligate anaerobes, and the proportion of the planktonic region that is aerobic for a typical facultative pond system is usually greater in the colder months than in the hotter periods (Marais, 1970).

Parhad and Rao (1976) tested the performance of two lagoon systems located at Nagpur, India, with respect to reduction of Salmonella, Coliforms, E. Coli, and Fecal Streptococci. Salmonella were not completely eliminated from the effluent of a two-celled lagoon that had submerged connections between the cells, while none were found in 1 litre samples of effluent from the third cell of a three-celled lagoon with the cells connected by a surface overflow arrangement. The authors concluded that apart from loading and detention time, the number of cells and interconnecting arrangements are important design features that have to be considered.

Based on the description of mechanisms in operation (see section 2.3.4.4), it would seem that the high pH and dissolved oxygen in the

surface waters in the day time in the typical tropical/semi-tropical pond system are the likely factors accounting for the differences in Salmonella reduction observed in the two pond systems by Parhad and Rao (1976).

2.6 END POINT

Ponds are natural habitats or ecosystems with homeostatic interactions tending to maintain dynamic equilibrium (eg. self shading, stratification and inorganic carbon limitation). As a result, they have assimilative capacities dependent on size, climate and other factors. As long as this assimilative capacity is not exceeded by internal or external loadings or a combination of both, satisfactory performance can be expected.

Because all the processes occur in one unit, optimization of the individual treatment processes is not achieved, however, when the overall objectives of waste water treatment are taken into consideration, ponds represent the complete treatment facility and are viable treatment systems.

The description of pond mechanisms given in this chapter are summarised in Fig. 2.8. It is clear from this description that ponds are truly complex ecological systems with a multitude of interacting mechanisms in operation some of which are not completely understood or identified. Undoubtedly this lack of information or incomplete description must be a significant factor accounting for the lack of evolution in design procedures for pond systems.

Based on the description given, it is clear that a sampling or

monitoring scheme for pond systems should take into consideration the fact that, pond behaviour has a unique daily as well as seasonal character and as such the issue of representativeness of samples taken should be clearly addressed. For example grab samples taken at a fixed location of the pond system on a daily basis will not give information on the likely impact of the pond system on the disposal media. On the other hand, grab samples taken at different locations at different times in the day will give information on the daily characteristics of pond behaviour in the particular season for which the sampling activity was undertaken.

Daily composited samples on the other hand will give information on the seasonal characteristic behaviour of the pond system and reveal the likely medium to long term impact of the pond effluent on the disposal medium.

The process description presented, clearly shows that, both the planktonic and benthic subregions are important to pond functioning and that ponds are influenced considerably by the diurnal cycle in solar radiation. It is obvious therefore that physical models (usually laboratory scale units) which neither have a clearly established benthic layer nor attempt to simulate both day time and night time conditions produce results which are not representative of pond systems.

CHAPTER THREE: DATA REQUIREMENTS; TYPE; AND PRELIMINARY ANALYSIS

3.1 POND DATA

3.1.1 DATA REQUIREMENTS

A mathematical modelling exercise requires a "DATA BASE"; based on which the suitability of the model as a description of the system in question can be assessed. Since the model identification scheme adopted generally depends to a great extent on the nature and type of data acquired, the ideal situation is where the modeller undertakes planned experiments, based on some a priori knowledge from concepts and physical insights into the process, to obtain the required data.

In the case of data acquisition on pond systems, it is practically impossible to obtain field data from controlled experiments. The inputs to, and outputs from an operating pond system are largely not within the control of the experimenter and as such, the level of excitation of the system cannot be considerably influenced.

In circumstances where it is practically impossible to experiment with field scale units in operation, pilot plants or laboratory scale units are usually investigated.

To obtain dynamic information about a continuous time process that is meaningful out to some frequency $f_{\max}$, the sampling frequency f_s

must be set at a rate greater than twice $f_{\max}$ and it is also of vital importance that the dominant modes of the underlying system processes be excited. Ideally, information relating to these dominant processes should be present in the data.

As dynamic systems, ponds can be considered to have large time constants and dynamic lag times on account of their immense volumetric capacity and changing hydraulic regime. The exogenous variables influencing pond behaviour eg. solar radiation and pond temperature vary seasonally and as a result, pond behaviour also varies seasonally. Therefore to obtain meaningful information on pond dynamics, they have to be observed over relatively long periods (in the order of years).

The time scale of reference in relation to the sampling frequency, however, is dependent on the level of detail sought. Because ponds have long detention times, dramatic changes in their effluent quality on a day to day basis is not expected and since initial attempts at identification of the behavioural characteristics of ponds relate more to seasonal variations than variations on a daily basis, observations made on a daily basis over a period of at least one year should be adequate for the systems study.

The uncontrollable exogenous factors (meteorological conditions), which influence pond behaviour considerably, are difficult to simulate under laboratory conditions and results of investigations into the separate effects of these influencing variables cannot generally be extrapolated to apply to the complex interactions occurring in the field. This is because the influencing factors do

not interact in a mutually exclusive fashion (Young and King, 1980). Stated simply; "the whole is more than the sum of its parts studied in isolation" (Bertalanff, 1933).

The hydraulic regime established in a pond system is very dependent on the pond's dimensions (Mangelson and Walters, 1972) and phenomena such as stratification which alter the hydraulic characteristics of pond systems are dependent among other factors on the areal and volumetric characteristics of the pond system (Stahl and May, 1967). Because of the problems of geometric scaling-up and the fact that a number of pond mechanisms (especially those controlled by exogenous variables) are often not simulated under laboratory conditions, data resulting from laboratory scale experimentation are mostly not representative of field-scale systems.

The interactions between the benthic subregion and the overlying planktonic region, for example, are usually not simulated in laboratory scale ponds in which experimentation is carried out for a limited period (eg. a couple of months). This is because most often, there is no clearly established benthic subregion for such systems. The activity in the benthic subregion is very difficult to simulate in laboratory scale units because this activity depends amongst other factors on the sludge temperature, which varies seasonally, and on the ORP at the sludge-water interface. Material feedback from the benthos into the overlying planktonic region is also dependent on the hydraulic regime established in the pond basin. These effects are however often not well reproduced in laboratory scale units.

On the basis of the above and the fact that ponds are not closed systems, there is the need for caution in the interpretation of results from the analysis of laboratory scale pond units. Laboratory scale representations of pond systems most invariably present a limited scope of enquiry and as such data resulting from such studies cannot be used for the complete systems study.

Results of laboratory scale investigations are however useful in that they help define the range of plausible behavioural characteristics based on the study of sub-component processes in pond systems.

3.1.2 DATA TYPE

Most data sets on pond systems have been collected during the normal operation of the system and result from performance evaluation exercises. This work was based on such data.

Typically these observational data sets consist of measurements of specific water quality characteristics of the influent and effluent streams and the measurements made usually reflect the objective of the performance evaluation study. Generally because the underlying objective is that of assessment against effluent standards, the sampling scheme adopted usually relates to the established criteria for effluent quality standards.

The EPA pond evaluation exercise conducted in 1977, for example, clearly shows how the existing effluent quality standards determine to an extent, the sampling scheme adopted. The EPA Secondary

Treatment Standards recommendation in the 1973 Federal Register states in part that: "The five-day biochemical oxygen demand (BOD_5) and suspended solids (SS) shall not exceed an arithmetic mean value of 30 mg/l for effluent samples collected in a period of 30 consecutive days nor 45 mg/l for samples collected in seven consecutive days". The pond evaluation studies encompassed 12 full months of data collection including four separate 30-consecutive days sample periods in months representative of the four seasons.

3.1.3 RELIABILITY OF POND DATA

The reliability of most of the reported data on pond systems is questionable. This is evidenced by the Barsom report (Barsom, 1973). Flagrant violations of sampling and reporting were enumerated in this report and it was observed that most lagoon performance evaluations were based on physical observation as assessment of performance.

Numerous evaluations of the performance of lagoon systems have been reported and reviews and synopses of these studies are available in the literature (Willford and Middlebrooks, 1967; Reynolds et al., 1967 ; McKinney, 1977). These observational data sets comprise mostly of infrequent grab and composited samples with considerable missing data points. Missing data points here refers to scheduled measurements which were not taken usually for either an influent or effluent constituent. This results in a considerable loss of information on the system under study.

In many circumstances, the choice of sampling interval and the scheme adopted are severely limited by administrative considerations

such as the normal working hours. Sampling schemes consisting of obtaining grab samples at prespecified periods of the day (say 9 - 10 am) result in aliased data (Ward and Loftis, 1983) which strictly cannot be used for the dynamic analysis of the system.

Information about the system at other times in the day is clearly lost. Errors arising from attempts at evaluating pond systems from such data is compounded by the fact that these grab samples are often taken at a fixed depth and location in the pond system.

Because pond systems exhibit significant spatial heterogeneity, gross errors result from the use of such grab samples. For example, sampling at a depth where an algal band resides will result in an effluent sample with a very high concentration of both BOD and suspended solids. Depending on the outlet arrangement, this would probably not be representative of the pond effluent BOD and suspended solids.

3.1.4 DATA SETS ACQUIRED

The data base for this exercise comprised monthly averaged data for four different pond systems and near daily data for two pond systems. Four of the pond systems used in this study formed part of the USA EPA (EPA, 1977a,b,c,d) pond performance evaluation exercise conducted in different geographical regions. A description of the characteristics of the four pond systems: Corrine, Utah; Eudora, Kansas; Peterborough, New Hampshire; Kilmichael, Mississippi; is given in Table 3.1. The monthly averaged data for the primary cells of these four pond systems was used for the evaluation of proposed

empirical design formulations.

Near daily data for two pond systems were used for the identification exercise and for the evaluation of the dynamic models for ponds. These two sets were made up of, year long pond monitoring exercises conducted in: New-Mexico, Albuquerque (Larsen, 1974); and Corrine, Utah, USA respectively (EPA 1977a). These two were the sets that had more consistent samples in the one year period for which the ponds were monitored.

In both monitoring exercises, analytical determination of constituents were done on composited samples. The minimum sampling interval was 24 Hr. Summaries of the characteristics of the waste streams, and the environmental factors measured for the New-Mexico and Corrine pond systems are given in Table 3.2 and 3.3 respectively.

The New-Mexico system, a one-cell unit, treated a combination of domestic and animal wastes of particularly high strength from the Inhalation Toxicology Research Institute, Kirtland Air Force Base, New Mexico. No analytical determinations were made for the inorganic nitrogen and phosphorous compounds in the first 80 days. The Corrine facility, a 7-cell lagoon system located in Corrine, Utah treated domestic waste of medium to low strength.

Since the future use of data are often impossible to anticipate, these sampling exercises cannot lay claim to optimality. Also, these data result from passive observation and as such are not the ideal a systems analyst would want to work with. They are however deemed to contain more information than has been abstracted from

evaluation and modelling exercises. A systems study aimed at retrieving as much information as possible from these data sets would contribute significantly towards the improvement of knowledge on pond systems.

Taking into consideration logistics, time and expense normally associated with data collection schemes, it can be appreciated why comprehensive and reliable data on ponds is not widely available.

3.2 PRELIMINARY ANALYSIS OF DATA

3.2.1 GENERAL

The preliminary analysis of the data involved the use of a combination of qualitative and quantitative procedures to determine the suitability of the acquired data for a systems study. The qualitative procedures included an assessment of the measurements made in relation to the process description given, and the quantitative procedures centred around the use of time series analysis techniques to assess the consistency and representativeness of the data sets.

These were followed by the use of multivariate analysis techniques to investigate the inherent structure in the data matrix and to determine plausible direct linear input-output relationships in the data sets.

3.2.2 SUITABILITY OF DATA IN RELATION TO PROCESS DESCRIPTION

It is impractical to attempt to obtain measurements for all the

descriptive variables representative of the ultimate elements and processes interacting in the typical pond system. Attempts at identifying pond systems based on deductive reasoning cannot as yet be clearly stated. Though some generalizations can be made, the interrelationships existing are not fully understood and some are even probably not recognised.

Characterization of pond behavioural characteristics at the present level of knowledge is more in terms of macroscopic properties than microscopic ones. In relation to the process description given in the previous chapter, the ideal would be to obtain measures pertaining to the various individual descriptive variables of interest in defining the macroscopic properties of the pond system. This is generally only possible, to an extent, in the case where the analyst is responsible for planning and executing the sampling programme. Even there, he is constrained by the range of available and reliable analytical techniques for the determination of the variables of interest.

For example attempts at measuring benthic activity and sediment oxygen demand have utilised in-situ measurement techniques (Bowen and Delfino, 1980) which either disturb the sediment in some kind of way or isolate the column of benthos under study. These result in sample measurements which are not truly representative with considerable uncertainty as to the nature of the resulting sampling errors.

Attempts at measuring algal activity (ie. primary production) have related this to the chlorophyll-a content of the biomass, or numbers of algal cells (Barnes and Mann, 1978). Numbers of algae

cells gives a poor idea of photosynthetic capacity since the cells counted may range from flagellates only $1\mu\text{m}$ or so in diameter to larger diatoms. Small cells have a high surface/volume ratio which favours rapid exchange of materials between protoplasm and environment, and as such, they are more active in photosynthesis.

Many organisms in the planktonic region are principally osmotrophic (absorbing dissolved material) or holozoic (eating particulate material) in nutrition but retain sufficient photosynthetic capacity to suffice maintenance. Thus the photosynthetic capacity of different species varies. The assimilative number or ratio (mg C fixed per mg chlorophyll per hour at light saturation) of phytoplankton may vary between 0.1 and 50 according to conditions. As a result, chlorophyll-a is not a reliable measure of primary production.

The pond data comprises of generic non-specific measurements such as biochemical oxygen demand (BOD), chemical oxygen demand (COD) and volatile suspended solid (VOLSS). These are surrogate measures which do not represent the descriptive variables, truly representative of the states of the pond system of interest in defining behaviour. They are however what are called "sufficient parameters" (Levins, 1966) because they describe gross effects. Both BOD and COD for example reflect the kind of changes that occur in the total carbonaceous organics quality of the waste stream though they do not give an indication of the relative proportions of dissolved and suspended organics.

A measure such as VOLSS incorporates both particulate organics of

the waste stream and cell mass of the biological entities; algae, bacteria and zoo-plankton. It is practically impossible to partition the VOLSS measure into its constituent sub-components. This limits considerably, information retrieval on the role the individual species play in the observed transformations and how they interact in the complex pond environment. However, as measures of macro or aggregate effects these surrogate measures suffice.

In terms of the description of pond systems given in Chapter 2, the absence of measurements which pertain directly to the activity of algae and bacteria; the activity in the benthic subregion and indication of numbers and types of pathogenic organisms clearly limits to an extent the information retrieval process. However, the availability of measurements representative of the inorganic nutrients and meteorological factors deemed to influence pond behaviour partly makes up for this lack of information.

In relation to the process description given, though there are limitations on what can be reconstructed, there is enough information available on which a systems study can be based. The above discussion however clearly demonstrates the need for the critical review of pond monitoring and the need for the evolution of measures which are more representative of the pond process mechanisms described in Chapter 2.

3.2.3 SUITABILITY OF DATA FOR DYNAMIC ANALYSIS

3.2.3.1 GENERAL

Prior to the use of any data for the dynamic analysis of a

system, it is necessary to find out whether the data is suitable and contains information such that the dynamics of the system can be investigated.

The data sets for dynamic analysis are composed of analytical measurements of characteristics for flow proportioned composite samples and daily average measures for the monitored environmental factors. The sampling frequency ranged from bi-weekly to daily measurements and in each case, the monitoring was done over a period of at least one year. Thus in terms of the objectives of identifying gross seasonal type behavioural characteristics, as long as the measurements made are representative, the acquired data sets are suitable for the dynamic analysis of pond systems.

To assess the consistency and representativeness of the data, time series analysis techniques were used. Many of the techniques for the analysis of time series eg. autocorrelation and harmonic analysis require data that are equally spaced in time; ideally therefore, the observations must be taken at regular intervals.

The acquired data is made up of irregularly spaced data together with some missing data points. In order to apply time series techniques for the preliminary analysis of these data sets, there was the need for some means of obtaining regularly spaced data from the irregular spaced sample values. Interpolation techniques were used for this purpose.

3.2.3.2 MISSING DATA AND INTERPOLATION

There are a wide range of interpolating techniques available, the

simplest being linear interpolation. This however has the disadvantage that most of the original data is ignored because only two surrounding points determine an interpolated value. Linear interpolation is a special case of the Lagrangian polynomial interpolation scheme for which a polynomial of any order less than the number of data points can be fitted to the data.

Interpolation techniques such as: Higher order lagrangian polynomials; Cubic Spline (Burden, Faires & Reynolds, 1981); and Generalised Continuous Parabolic Interpolation (Org and Lee, 1985) attempt to overcome the disadvantages of linear interpolation by considering more than two of the original data points. Experience with high order polynomial interpolation showed that, because the original data possessed large random components which caused values to fluctuate widely, the interpolated points fluctuated unacceptably and produced unrealistic estimates.

A bi-cubic spline interpolation was therefore used for the restructuring of the original data into data sets comprising of regularly spaced data points. It must be borne in mind however that data cannot be created by estimation using any method and that the validity of the results of any interpolation scheme depends to a great extent on the density of the original values used.

3.2.3.3 RAW DATA PLOTS

Sample plots of the resulting reconstructed data showing influent and effluent characteristics together with the measured environmental factors for the two pond systems are shown in Figs 3.1-3.28 and

Figs. 3.29-3.51 respectively. Summaries of the first two statistics (mean and standard deviation) together with the data range for each of the individual time series in the New-Mexico (Pond 1) and Corrine (Pond 2) pond data are given in Tables 3.2 and 3.3 respectively and a discussion on the visual interpretation of the raw data plots is presented below.

NEW-MEXICO POND SYSTEM (POND 1)

The resulting time series plots for the measured characteristics of both the influent and effluent streams for pond 1 data show very rapid fluctuations between subsequent data points which makes it virtually impossible to discern any similar trends/patterns between influent and effluent series.

Not much information about the signals content (eg. seasonal variations or medium and long term trends) can be made out from the plots suggesting that the data is very noisy. An examination of the computed mean for the influent and effluent series clearly shows that both dissolved and suspended carbonaceous organics as well as most of the inorganic nutrients constituents of the influent stream went through transformation processes which in their overall effects resulted in a reduction of their concentrations in the effluent stream.

Plots of measures of the environmental factors are however not as noisy as those of the characteristics of the influent and effluent stream. The mean of air and pond temperature (Figs. 3.22 & 3.23) are not significantly different and pond temperature follows the same bell shaped seasonal pattern as air temperature. The resulting

plot for solar radiation (Fig. 3.21) however does not depict a uniform sinusoid over the annual cycle as observed for temperature and evaporation (Fig. 3.24). There is actually a spike in the solar radiation series around day 280 and this, most probably, is not a representative measure and could have resulted from some gross measurement or transcription error. Precipitation (Fig. 3.25) and wind speed (Fig. 3.26) show no discernable seasonal characteristics.

If it is assumed that the resulting constituent characteristics for the effluent stream results from the pond system acting on their respective components in the influent stream, and that standard deviation is an indicator of the variations in a time series and hence is some measure of its signals strength, then the computed standard deviations suggest that the constituents of the influent stream undergo significant damping.

There are indications of consistency in the data. For example the plots of measures representative of the influent solids content (see Figs. 3.7 and 3.9) shows that both the total and volatile suspended solids have a spike around day 200. This however is not manifest in their respective measures in the effluent stream (Figs. 3.8 and 3.10) suggesting either that the influent signal is considerably damped in the pond system, or that the major portion of the influent solids settle to the pond bottom resulting in the effluent solids not being driven considerably by the influent solids.

Both measures of the effluent solids concentration (TOTSS and VOLSS) reveal a general pattern of higher concentrations in the hotter months than in the colder months. There are however no

trends in the influent series to suggest that this could have resulted from increasing influent solids.

CORRINE POND SYSTEM (POND 2)

The fluctuations in the time series plots for this data are considerable less than for Pond 1. This could be due to the fact that this data set was made up of a smaller set of data points. Also it is quite evident from the mean levels of measures of influent carbonaceous organics (BODs and CODs) and total solids concentrations (see Table 3.3) that the organic loading to this pond system was considerably less than that to Pond 1.

The mean and standard deviations for measures of both the influent and effluent characteristics show that there were reductions in concentrations of the constituents of the waste stream, and that some damping of the influent series occurred. The removals for measures representative of carbonaceous organics were however considerably less in this pond system compared to the observed removals in Pond 1. Plots of pond temperature (Fig. 3.50) and solar radiation (Fig. 3.51) have the expected bell-shaped seasonal pattern.

There are spikes in the individual plots which look like outliers. An inspection of measures equally representative of the same descriptive variable reveals however that some of these spikes are consistent. This suggests that if these outliers are gross errors, they most probably did not result from the analytical determination procedure employed but rather could have resulted from sampling errors.

The plot for influent total COD (Fig. 3.29) for example, has a spike around day 270 which is also present in the plots for both influent total BOD (Fig. 3.33) and influent solids (Fig. 3.37). This spike is however absent from the influent soluble COD plot (Fig. 3.31) suggesting that it is a solids associated measure. Another example is the spike in the effluent soluble BOD (Fig. 3.36) around day 50 which is also observed in the effluent total BOD plot (Fig. 3.34).

The spike in the influent soluble BOD plot (Fig. 3.35) around day 300, for example, could be considered to be true outlier resulting from some gross error. This is because this spike is not present in either the influent total BOD or soluble COD plots (Figs. 3.33 and 3.31).

3.2.3.4 SIGNALS CONTENT

The apparently rapid fluctuations in the data are almost certainly due to a combination of measured input disturbances, random process behaviour and random measurement error. It cannot be determined before hand what kind of information can be abstracted from a noisy data set but then prior to its use for any analysis, there is the need to find out whether "signals" are really present.

When a time series is referred to as being noisy, this implies that the observations consist of two parts; these being, an underlying signal or meaningful pattern of variation and a superimposed random variation (noise) which could arise from various dependent and independent sources.

Noise is short-term and fluctuates rapidly whereas signals in contrast tend to be long term. The successive values of signals are usually autocorrelated whereas noise at one point is mostly independent of noise at adjacent points. Noise contaminating process signals can arise from diverse sources eg. analytical errors, sampling errors and transcription errors. A malfunctioning measuring instrument, usually produces structured noise terms.

The raw data plots showed the data to be very noisy. There was therefore the need to analyse the data sequences for their signals content. In this work, the adopted criteria for determining the signals content was to find out whether the observed time series differed significantly from random wide band process signal with equal power at all frequencies within the band pass of interest (typically represented by white noise).

The basic premise here was that if the observed series were found to be significantly different from white noise, then it could be speculated that the signals present in the data contain information on the system under study. If signals were found to be present but however had low signals-to-noise ratios, then there would certainly be the need to enhance these signals such that they would possibly shed light on the dynamic interactions in the system.

A random signal can be decomposed into orthogonal components ranging from the fundamental (base) frequency to the Nyquist frequency. For a sample data sequence comprising N measurements in time, the process signal can be represented by:

$$X(t) = u + \sum (a_p \cos f_p t + b_p \sin f_p t) + a_{N/2} \cos \pi t \quad \dots 3.1$$

where:

$$t = (1, 2, 3, \dots, N)$$

$$u = \sum X_t / N$$

$$a_{N/2} = \sum (-1)^t X_t / N$$

$$a_p = 2(\sum X_t \cos(f_p t)) / N$$

$$b_p = 2(\sum X_t \sin(f_p t)) / N$$

$$f_p = 2\pi p / N$$

$$p = 1, 2, \dots, N/2 - 1$$

This is the Fourier series representation (Chatfield, 1977). The respective frequencies f_p are called the harmonics and the contribution of each harmonic to the total variance of the series is given by:

$$(R_p)^2 / 2 = ((a_p)^2 + (b_p)^2) / 2 \quad \dots 3.2$$

Analysis of series along these lines is called harmonic analysis. Figure 3.52 is a plot showing the resulting analysis for a psuedo-random sequence generated on the computer. The spectrum is a plot showing the contribution of each harmonic to the total variance and the cummulative periodogram shows the cummulative contribution from the base frequency to the Nyquist.

The section in the top right hand corner of the spectrum shows the six most significant cycles (w_p) and their contributions to the total variation in the series (R_p^2). The component accounting for the largest variation in the series accounts for less than 4% of the total variation and the cumulative power plot is virtually a straight line with unit slope showing the near equal contribution at all frequencies within the band width.

A harmonic analysis was carried out to determine whether or not the individual series differed significantly from white noise. Plots of the resulting contribution to total variance and the cumulative power for Pond 1 and Pond 2 data are presented in Figs. 3.53-3.78 and 3.79-3.101 respectively. No statistical analysis was performed to determine the level of significance of the test of the hypothesis that the signals differed significantly from white noise. The test for this was simply based on visual inspection of the spectrum and cumulative periodogram obtained.

A discussion of the resulting plots of the spectrum and derived periodogram for the two pond systems is presented below.

NEW-MEXICO POND SYSTEM (POND 1)

The relative similarity of the frequency distributions for measures of the influent carbonaceous organics characteristics and also of measures of the effluent carbonaceous organics characteristics eg. COD, ROD and VOLSS (Figs. 3.53-3.54; 3.57-3.58; and 3.61-3.62), suggests that the data has a consistent information content.

The influent carbonaceous organics series all appear to be composed of a signal at the base frequency and one in the region of

52 cycles in the year. The latter corresponds to a weekly cycle. The weekly cycle observed in the influent series is not surprising considering the fact that Pond 1 treated waste from a Research Institute. Small establishments are more likely, than municipalities, to produce wastewater discharges following a characteristic weekly pattern.

The effluent series suggests that the pond system behaved like a low pass filter. Frequencies contributing significantly to the observed variations in the individual effluent series are all low and the weekly cycle observed in the influent series appears to have been filtered out. This suggests the likelihood of the non-existence of direct linear input-output relationships between measures of carbonaceous organics quality of the waste stream. This is not surprising considering the complex nature of pond interactions and the long detention times referred to earlier.

The process description presented in the previous chapter showed that the planktonic region of pond systems receives both an external and internal loading of both organic and inorganic material. The observed pond effluent characteristics are therefore likely to result from transformations of both of these loading components and as such, a direct relationship between the measured influent and effluent characteristics cannot be expected.

It is evident from the power spectrum for temperature, solar radiation and evaporation (Figs. 3.72, 3.74 and 3.75) that the main observed variation in these environmental factors is an annual cycle.

CORRINE POND SYSTEM

The results of the harmonic analysis for Pond 2 data suggest that this data is essentially composed of low frequency signals compared to Pond 1 data. The influent and effluent series for measures of the nitrogen compounds appear to be composed of the same dominant frequencies. The components for the influent series are 1 and 3 cycles in the year whereas the effluent series are made up essentially of 1 and 2 cycles in the year.

The 2 cycles in the year is apparently a dominant component in all the effluent series for this pond data. Measures of total carbonaceous organic content eg. total COD and BOD (Figs. 3.81 and 3.87) have 2 cycles in the year as the frequency component contributing the most towards total variance in the observed series. This component however is not present in their corresponding influent measures.

The above observation suggests in agreement with observations made for Pond 1 data, that the effluent organics are not driven solely by the influent organics. The dominant frequency in the effluent carbonaceous organics measures appears to be solids associated. The 2 cycles component in the effluent solids series explains the most variance in that series (see Fig. 3.88).

Both the influent and effluent total phosphorus appear to be composed of the same dominant cycles (1 & 2) which suggests the likelihood of some direct input output relationship for total phosphorous (see Figs. 3.89 and 3.90).

3.2.4 SIGNALS ENHANCEMENT

Assuming that contributions to total variance less than 3 percent are indistinguishable in their effects from the effects of white noise, an ad-hoc procedure for the determination of the signal-to-noise ratio of the individual series can be derived from results of the harmonic analysis. This comprises of classifying all frequency components contributing less than 4% of the total observed variation as noise.

If the resulting periodograms for both Pond 1 and Pond 2 are assessed on the basis of the above criteria, the conclusion that can be drawn is that the raw data is very noisy (has a low signal-to-noise ratio. This means in effect that the fast process dynamics will be difficult to isolate from the effects of noise.

Since ponds are ill-defined systems and identification exercises at this stage relate more to identifying the slow dynamics of the system, the noise masking the fast process signals does not impose a major restriction in the use of the data for identification purposes.

The harmonic analysis on both pond data showed that frequency components above 52-cycles in the year contribute less than 3 percent to the total observed variation. This suggests that filtering out frequencies above 52-cycles will not result in a considerable loss of information on the slow dynamics characteristics of the signals. Rather, this would result in signals

enhancement.

In attempting to filter out the noise, some information on the dynamics of the system is lost. There is the need therefore for some kind of compromise in the selection of filtering scheme for signal enhancement. The choice of scheme is often determined by the fastest dynamics of interest and the likely time constants of the system assuming it can be represented by a first order process.

In this study, a moving average type filter (of least order 7) was chosen for signal enhancement. The order 7 was chosen primarily because the smallest consistent sampling frequency in the original raw data was made of three data points in the week and one week was considered to be smaller than the likely time constants of significance in the determination of the slow dynamics of the pond process.

The concept of using an averaging type filter is that, as the signals tend to be the same from point to nearby point and the noise does not, the average of several adjacent points will tend to converge on the value of the signal alone.

The moving average filter estimate for a signal at time t of a series Y_t ($Y_t: t \in T$) is given by:

$$\hat{Y}_t = \sum_{j=t-k}^{t+k} Y_j / M \quad \dots 3.3$$

where:

$$k = (M-1)/2 \quad \text{for odd } M$$

and:

$$\bar{Y} = \sum_{j=t-k}^{t+k} Y_j / M + 1/2 (Y_{t-M/2} + Y_{t+M/2}) \quad \dots 3.4$$

where:

$$k = (M-2)/2 \quad \text{for even } M$$

The plots of the resulting influent and effluent series for the enhanced signals using a 7-days moving average filter, are shown in Figs. 3.102-3.112 and 3.113-3.122 for Pond 1 and Pond 2 respectively. The influent series are represented by the solid line and the effluent series by the bars. At the top right hand corner of each plot is shown the mean and standard deviation for both influent and effluent series. It is quite evident that these series do not show as much variability as the original data and trends are more evident.

The similarity in the trends in influent and effluent BOD over the colder periods, for Pond 1 data (Fig. 3.104) for example, becomes more apparent. The general trend of higher carbonaceous organics and solids concentration in the hotter periods with effluent solids concentration exceeding the influent concentration in the hottest months is also evident from these plots (see Figs. 3.102-3.106). Effluent nitrite concentrations are greater than the influent concentrations (see Fig. 3.112) suggesting the possibility of either an excessive first stage nitrification process; considerable denitrification or some combination of both.

The plots of the influent and effluent series for Pond 2 data clearly show that significant reductions in concentrations were mostly observed for the inorganic and organic nitrogen compounds represented by ammonia nitrate and total kjeldahl nitrogen (see Figs. 3.119-3.121).

Measures of the effluent carbonaceous and total solids concentration for Pond 2 eg. total COD and solids (see Figs. 3.113 and 3.117), exceed their respective influent concentrations in the hotter periods.

In both Pond 1 and Pond 2 data, there are no trends in the influent series suggesting that the observed higher effluent concentrations could have resulted from the influent series even when considerable lag times are taken into consideration. The high concentrations observed in the effluents most likely resulted from the increased productivity of phytoplankton with increasing temperatures and incident solar radiation.

The results of the analysis of the signals content clearly shows that both data contain enough indication of meaningful signals with some information content on which a systems identification exercise can be based.

3.2.5 TIME SERIES ANALYSIS (TIME DOMAIN)

3.2.5.1 GENERAL

The time domain analysis techniques namely: autocorrelation and cross-correlation, were used to help evaluate further, the

consistency of the data sets. Frequency domain (harmonic analysis) and time domain analysis techniques are complementary with time domain techniques focussing on relations/properties of the time series in time.

The correlation between two time series is a measure of the strength of their linear relationship. It is the ratio of the covariance of the two series to the product of their standard deviations. The correlation coefficient can range between -1 and $+1$ and a value of ± 1 indicates a perfect linear relationship between the two series. A correlation coefficient of zero implies linear independence but does not necessarily imply a lack of any other type of dependence.

The structure of the moving average type filter is such that serial correlation at low lags is introduced into the filtered series. This results in spurious correlations when serial correlations are determined for the individual series. The original data was therefore used for autocorrelation analysis.

Sampling of the enhanced data at intervals equivalent to that used for the moving average filter results in time series which theoretically have no induced correlations. This however results in a significantly reduced data set with a possible loss of information. For example for an original data set comprising of 365 data points, a 7th order moving average filter results in a data set comprising of 359 data points. When this data is sampled at 7 day intervals, the resulting data set is made up of 52 data points. Depending on the starting position for the sampling procedure, there are 7 such different data sets that can be derived.

Determination of the cross-correlograms for seven such data sets is a daunting task. On account of this, the original series were also used for the cross-correlation analysis. The enhanced signals would have no doubt made any lag effects more evident. However, because the enhanced signals have induced serial correlations, the chances of obtaining spurious correlations are quite high.

3.2.5.2 AUTOCORRELATIONS

Autocorrelation is essentially a self comparison process in which two parts of the same time series are compared to determine their degree of similarity and hence the extent to which the sequence is repetitive. It is complementary to harmonic analysis except that here, there is no indication of power contribution.

The autocorrelation at any lag k is computed as the linear correlation between the time series and the same series offset by k intervals of time. For a series $X (X: X_1, X_2, \dots, X_n)$, this is given by:

$$r(k) = \frac{\sum_{i=1}^{n-k} X_i X_{i+k} - \sum_{i=1}^{n-k} X_i \sum_{i=1}^{n-k} X_{i+k} / (n-k)}{\sqrt{\left(\sum_{i=1}^{n-k} X_i^2 - \left(\sum_{i=1}^{n-k} X_i \right)^2 / (n-k) \right)} \cdot \sqrt{\left(\sum_{i=1}^{n-k} X_{i+k}^2 - \left(\sum_{i=1}^{n-k} X_{i+k} \right)^2 / (n-k) \right)}} \quad \dots 3.5$$

Examination of the autocorrelogram, which is a plot of the autocorrelations at successive lag positions, reveals intervals of

time at which the time series has a repetitive nature. The autocorrelograms for the individual time series in the two data sets bore out the inferences made from the harmonic analysis.

An example of the autocorrelogram for the white noise sequence generated on the computer is shown in Figure 3.123. It is clear from this plot that the white noise sequence has no significant autocorrelation coefficients and hence no obvious structure.

Though there are statistical procedures for testing the hypothesis that a series is significantly different from white noise, such procedures were not used because a visual inspection of the resulting correlograms clearly indicated that apart from the influent and effluent organic phosphorous, precipitation, and wind speed for Pond 1 data, and precipitation for Pond 2 data (see Figs. 3.125-3.129), all the series differed significantly from white noise.

The 52 cycles component in the influent series for Pond 1 data eg. ammonia, total BOD and total solids (see Figs. 3.130, 3.132 and 3.134), are reflected as high serial correlation coefficients at lag multiples of 7 (ie. 7, 14, 28 etc). The autocorrelograms for their respective effluent series (see Figs. 3.131, 3.133 and 3.135) all have relatively high serial correlation coefficients at low lags as would be expected from systems with large time constants (hence considerable damping potentials).

3.2.5.3 CROSS-CORRELATIONS

Cross-correlation is a measure of the linear correlation between two individual time series at given lags. The cross-correlation between two time series X ($X: X_1, X_2, \dots, X_n$) and Y ($Y: Y_1, Y_2, \dots, Y_n$) at a given lag k is given by an equation similar to Eq. 3.5 with X_{t+k} replaced by Y_{t+k} .

Cross-correlation analysis was used in this work to investigate plausible direct input-output relationships and to determine if the pond behaviour resulted in pure time delays between input and output signals where linear input-output relationships were indicated.

Preliminary analysis of data using cross-correlation can give some indication on likely range and order of magnitude of time delay for a system. Heuristically, this is defined as the time elapse between an input going into the system and its effect being manifest in the output. For conservative substances, it is practically equal to the hydraulic detention time of the system. The resulting cross-correlogram peaks at that time displacement equal to the time required for the signal to pass through the system. In a typical non-linear system with dynamic lags, the rate of transmission and path of an input signal through the system is not constant. This results in the masking of the input signal and thus no observation of significant cross-correlations.

The resulting correlograms for the cross-correlations between the individual series in the two data sets are discussed below.

POND 1 DATA

The correlograms for measures of the carbonaceous organics quality eg. total COD and BOD (see Figs. 3.136 and 3.137) all showed no significant correlation between their respective measures for influent and effluent both on the same day and at low lags reinforcing the claim that no direct linear relationship between the influent and effluent carbonaceous organics can be expected for pond systems.

There are significant negative correlations between influent and effluent BOD at positive lags in the region of 48 days suggesting that over the monitoring period, a high influent BOD on a particular day produced a corresponding low effluent BOD about 48 days later. This does not give a clear indication of some kind of lag effect because a high influent BOD would be expected to produce a corresponding high BOD at lag times corresponding to the detention time of the pond system. It could be argued however that if any observed lag effects result solely from a pure time delay, then a high BOD on a particular day could result in the stimulation of growth of heterotrophic organisms in the system leading to the observance of a relatively low effluent BOD at a lag time close to the theoretical liquid detention time of the system. It must be borne in mind however that a phase shift between the general trend of lower influent BOD in the hotter months with the trend of increasing effluent BOD could also have produced the observed correlations.

The correlograms of other measures representative of carbonaceous organics do not reveal this apparent lag effect. As a result, no

real meaning can be attached to the observed correlations.

The cross-correlograms of influent and effluent measures for the various inorganic nitrogen forms eg. ammonia and nitrate (Figs. 3.138 and 3.139) showed significant zero and low lags correlations. The nitrate series showed a peak in the cross-correlogram at about 5 days lag which probably represents the streaming time between inlet and outlet structures. Measures of both organic and inorganic phosphorus showed no significant same day and low lags correlations.

The measured environmental factors which produced significant cross-correlations were those with the bell-shaped seasonal pattern (viz. temperature, solar radiation and evaporation). These produced high zero lag correlations with measures of effluent carbonaceous organics and solids (see Figs. 3.140-3.143).

Precipitation and wind speed produced no significant zero and low lags correlations with the various measures of the effluent characteristics (for example see Figs. 3.144-3.146). This is probably because of the fact that over the observation period for Pond 1, both precipitation and wind speed showed no significant seasonality or characteristic behavioural pattern and as such did not exert any sustained observable influences on pond behaviour.

POND 2 DATA

The correlograms for Pond 2 data also showed no significant zero and low lags correlations between influent and effluent measures of carbonaceous organics and solids (see Figs. 3.148 and 3.149). There

was however a significant zero-lag correlation between influent and effluent soluble COD (Fig. 3.150) and this probably represents the refractory portion of the carbonaceous organics quality of the waste stream (which is likely to be reflected in the soluble COD measure).

The correlograms for the correlations between precipitation and measures of the effluent series for this data also showed no significant correlations to suggest that precipitation had any sustained effect on pond performance. (for example see Figs. 3.150-3.152). The effluent series however all appear to be highly correlated with the temperature related factors at low lags. Both effluent total phosphorus and ammonia for example have significantly high negative zero-lag correlations with temperature and solar radiation (see Figs. 3.153-3.156). The total COD and the total solids have significantly high positive correlations with temperature and solar radiation at low lags (see Figs. 3.157 & 3.158). These cross-correlograms suggest the presence of a phase shift between increased temperature/solar radiation and increased solids.

The correlograms for the temperature related factors and various effluent series for Pond 2 data generally suggest that high temperature and solar radiation were associated with low nutrients concentrations and high carbonaceous organics and solids concentration. This observation is probably a reflection of the increased productivity of algae with increasing temperature and solar radiation.

The measures of nitrogen related compounds all have significant zero and low lags correlations between their influent and effluent

series (see Figs. 3.159-3.161). The total kjeldahl nitrogen for example (Fig. 3.161) has an apparent lag of about 3 days between the influent and effluent.

SUMMARY

The apparent low lags (in the order of 3 to 4 days) observed for cross-correlations for both Pond 1 and Pond 2 data are not important because they will not have a significant influence on the determination of medium term input output relationships. For example if either a bi-weekly or monthly pond output behaviour is being investigated, then a lag of a few days between influent and effluent would practically be an instantaneous effect.

The cross-correlograms obtained for both Pond 1 and Pond 2 data clearly suggest that no linear input-output relationship between the influent and effluent carbonaceous organics is likely to be observed in the identification exercise and that temperature related environmental factors exert observable sustained medium to long term influences on pond behaviour. Also, any pure time delays that may be present in pond system behaviour are masked by the non direct linear input-output relationship.

3.3 MULTIVARIATE ANALYSIS

3.3.1 GENERAL

Multivariate techniques provide methods for describing interdependence between system variables. A typical data set made up of N measurements of p variables can be considered to be an $N \times p$

matrix of information on the system. The maximum information can be retrieved from such a data set if each variable has a separate inherent structure or characteristic (ie. contains information unique to that variable).

The actual pond data however, is not made up of independent data since a number of the surrogate measures eg. COD and BOD reflect the characteristics of the same descriptive variable (though they are in essence different measures).

The multivariate analysis technique used in this work comprised of the principal component and factor analysis procedures. These were used to assess the inherent structure of the data matrices with the objective of a possible reduction in the dimension of the data (ie. number of series deemed to be relevant to the systems study).

3.3.2 PRINCIPAL COMPONENTS AND FACTOR ANALYSIS

3.3.2.1 GENERAL

Principal component analysis basically deals with the issue; "what relationships exist amongst variables in a data matrix such that new orthogonal (mutually independent) variables can be extracted in decreasing order of significance with regard explaining the the total variation in the data matrix". Factor analysis deals with the correlation between a number of prespecified factors and the variables in the original data matrix.

The transformation of an original $N \times p$ data matrix to an $N \times p$ matrix of principal components is a linear transformation and can be

represented by:

$$Z = X.A \quad \dots 3.6$$

Where; X is the N*p data matrix of the original variables ; Z is the N*p matrix of transformed variables; and A is a p*p matrix of coefficients defining the linear transformations. The matrix A is equivalent to the matrix of eigenvectors of the variance-covariance matrix of X.

Because the units of the original variables in the data matrix (X) are non-commensurate, standardized variables were used for the analysis . Standardization results in a data set comprising of individual series of equal power hence relationships that emerge are basically dependent on the inherent structure of the series and not on dimensions or order of magnitude.

The factors used for the factor analysis were the principal components obtained from the correlation matrix of X. The resulting factor loading matrix therefore gives the correlation between the original standardised variables and their principal components. Similarity in structure between any two variables is therefore revealed by their relative high correlation with the same factor.

Because the original data has a low-signal to noise ratio, a correlation coefficient of greater than or equal to 0.60 was assumed to be significant. Though an arbitrary level of 0.60 was chosen as the value depicting a significant correlation, this was only used as a guide and not an absolute value for discrimination.

The cross-correlation analysis showed that in terms of the slow

dynamics of the ponds monitored, no significant lag effects were observable. There was therefore no need for lag effects to be included in the multivariate analysis. Also, because the zero-lag cross-correlations between measures of similar characteristics of the influent and effluent streams did not suggest the likelihood of direct linear input-output relationships, the data sets were not partitioned into influent and effluent sub-sets.

3.3.2.2 RESULTS FOR POND 1 DATA

Because measurements for the nutrients series for this data set commenced after day 80, Pond 1 data was partitioned into a carbonaceous organics related set (set 1) and a nutrients set (set 2). The measured variables taken into account in the analysis of Pond 1 data and their abbreviations are given in Table 3.4. The resulting correlation and factor loading matrices are given in Tables 3.5-3.6 and 3.7-3.8 for set 1 and set 2 respectively.

SET 1

The correlation coefficients in the correlation matrix (see Table 3.5) found to be significant were mostly between measures representative of the same characteristic of either the influent or effluent streams. Influent filtered COD was found to be significantly correlated with the influent total COD and, the influent BOD, TOTSS and VOLSS were all significantly correlated with each other. Their respective effluent series were also found to be significantly correlated with each other but were not significantly correlated with their respective influent series.

Measures of the effluent solids concentration (VOLSS and TOTSS) were highly correlated with both temperature and evaporation.

The factor loading matrix for set 1 (see Table 3.6) shows that above the sixth principal component, there are no significant correlations between the the standardized variables and their principal components. The first six factors account for 79.8% of the total variation in the correlation matrix with the other 12 factors probably accounting for the noise in the data matrix.

The first component can be considered to be an effluent total solids factor because the effluent total solids is the variable most highly correlated with this factor (.93). The other variables significantly correlated with this component are air and pond temperature, evaporation, effluent volatile suspended solids and effluent total BOD.

The correlations between the original variables and the first principal component suggests that the main influencing factors that determined the resulting effluent carbonaceous organics and solids concentrations for Pond 1, over the observation period, were temperature related.

The second component can be described as an organic loading component. The influent total BOD is the variable most highly correlated with this component (0.79) with all the other measures representative of the influent carbonaceous organics and solids also significantly correlated with this component.

None of the standardized variables were significantly correlated

with the third component. The influent and effluent total and filtered COD were however the only variables which had close to significant correlations with this factor. This suggests that the third component probably represents the refractory portion of the carbonaceous organics quality of the waste stream.

The results of the analysis of set 1 suggest that the pond influent and effluent carbonaceous organics quality have inherently different "structures" though the various surrogate measures for the influent and effluent appear to contain the same information.

The emergence of influent and effluent related factors with there being strong correlations between the effluent factor and the temperature related environmental variables suggests that the temperature related exogenous variables could have exerted significant influences on pond behaviour.

SET 2

The correlation matrix for this set (see Table 3.7) shows that there were rather low correlations between the various measures of the nutrient series. The only correlations which are significant in relation to the adopted criteria are those between pond temperature and evaporation (0.91) and between the influent and effluent flows (0.65). The following however appear to be partially correlated: Influent and effluent nitrate; influent ortho-phosphorus and influent ammonia; effluent ortho-phosphorus and effluent ammonia.

Though the influent series for ortho-phosphorus and ammonia are partially correlated, they are not significantly correlated with their effluent series. This suggests that the inorganic nutrients

(represented by these variables) were probably transformed by similar pond mechanisms.

The factor loading matrix for this set (see Table 3.8) shows that above the fifth component, significant correlations do not exist between the principal components and their standardized variables.

The first component can be classified as a temperature/inorganic nutrients factor since pond temperature, evaporation, effluent inorganic phosphorus (ortho) and effluent ammonia are all significantly correlated with this component.

The second component can be termed a flow factor because the only variables found to be significantly correlated with this component are the influent and effluent flows.

Since both the influent and effluent nitrate are the only variables significantly correlated with the third component, it can be described as a nitrate component.

The fourth component was not significantly correlated with any of the variables but appeared to be an influent organic phosphorus component and the fifth, an effluent organic phosphorus factor.

The influent inorganic nutrients measures (ie. ortho-phosphorus and ammonia) did not emerge as some clear cut factor but were however partially correlated with the primary component. This suggests the possibility of some partial relation between the influent and effluent nutrients series which has apparently been overshadowed by the influence of the temperature related exogenous variables on the effluent characteristics.

3.3.2.3 RESULTS FOR POND 2 DATA

Pond 2 data was partitioned into three sets with overlapping variables. The first set (set 1) comprised mainly of the measures representative of the waste stream quality and had no variables pertaining to the environmental factors deemed to influence pond behaviour. The second set (set 2) comprised measures of the carbonaceous organics quality and environmental factors whereas the third set (set 3) was composed of the environmental factors and measures of the nitrogen and phosphorous compounds.

The respective abbreviations used to represent the variables analysed are given in Table 3.9 and the resulting correlation and factor loading matrices in Tables 3.10-3.15.

SET 1

It is evident from the correlations for this set (see Table 3.10) that the influent total COD, total BOD and total solids are all highly correlated with each other. The only variable with which the effluent total COD is correlated is the effluent total solids. This suggests that a significant proportion of the observed effluent total COD could be solids associated.

The influent total kjeldahl nitrogen (TKN) is significantly correlated with the influent total and soluble COD, and influent total BOD. This suggests some consistency in the data matrix since all these measures are to some degree representative of the organic content of the waste stream. The influent TKN is also highly

correlated with the influent ammonia and to a much lesser extent with the influent total phosphorus .

The correlations between effluent total phosphorus , ammonia and TKN are all significant. The effluent ammonia is negatively correlated with the effluent total solids which suggests that over the monitoring period, high effluent solids were associated with low effluent ammonia. This is what is expected if the effluent solids were made up mostly of algae cells.

The effluent soluble BOD and effluent total BOD are significantly correlated with each other but not with any of the influent series. This reinforces the earlier assertion that the biodegradeable fraction of pond effluents is largely independent of the influent fraction.

Influent and effluent nitrate are significantly correlated with each other as in case 2 for Pond 1 data. This suggests the likelihood of pond nitrate dynamics being controlled by the nitrate loading levels in the influent stream.

The resulting factor loading matrix for this set (Table 3.11) shows that components above the sixth are not significantly correlated with the original variables. The primary component for this set can be considered to be an influent loading factor. This is because, though influent ammonia and TKN are the variables most highly correlated with this factor, all the other influent measures (except for nitrate, total solids, and BOD) are significantly correlated with this variable.

The second component can be described as a nitrate factor and the

third, an effluent carbonaceous organics factor. The separation of the data matrix into influent and effluent factors clearly demonstrates, especially for the carbonaceous organics, that pond effluent characteristics are largely independent of their influent characteristics.

The fourth component cannot be described as some clearcut factor in relation to the original variables since none of them are significantly correlated with this component. The fifth component is considered to be a soluble effluent BOD factor and the sixth a soluble effluent COD factor.

SET 2

The resulting correlation matrix for this set (see Table 3.12) showed that the environmental factors were not significantly correlated with the measures of organics and solids content in the pond data. The only additional significant correlations observed were those between pond temperature and solar radiation, and between influent and effluent flows. These are however to be expected since the effluent flow for example is determined to a large extent by the influent flow and over a typical annual cycle, pond temperature and solar radiation are expected to be highly correlated.

The resulting factor loading matrix (see Table 3.13) shows that the first two components are basically an influent loading factor and an effluent carbonaceous organics factor respectively. The significant negative correlation between pond temperature and the first factor shows that this pond system was subjected to a higher loading in the colder months than in the warmer months.

SET 3

The correlation matrix for this set (see Table 3.14) shows that the variables with a significant correlation between their respective influent and effluent series (eg. ammonia and total phosphorus), have higher negative correlations with pond temperature than with their respective influents. This suggests that though there was a general trend of reduced loading with increasing temperatures, temperature effects on the observed transformations were quite significant and not completely masked by the observed trend in the influent series.

The resulting factor loading matrix (see Table 3.15) shows that the primary component is an influent loading factor. The second component can be described as a nitrate factor and the third an effluent total solids factor. Components beyond the sixth are not significantly correlated with the original variables. The sixth component can however be considered to be a precipitation factor if the assessment criteria is slightly relaxed.

3.3.2.4 SUMMARY

The appearance of wind (in Pond 1 data) and precipitation (for both data sets) in the principal components and factor analysis as the least significant factors, but nonetheless as factors, shows that though these environmental variables had a highly stochastic nature over the monitoring period, they were not entirely random sequences. They had unique characteristics which however did not exert significant influences on pond behaviour over the typical

annual cycle.

Pond functioning is dependent on the interaction of several endogenous and exogenous factors (eg. organic loading, temperature, solar radiation, wind etc.). It is clear from the literature on ponds that temperature and solar radiation are the most significant influencing factors over the typical annual cycle and that ponds exhibit seasonal characteristics on account of these influences. This fact is clearly demonstrated by the results of the principal component and factor analysis of the two data sets.

The other exogenous factors (eg. wind speed and precipitation) which are known to influence pond behaviour usually have no clear cut seasonal characteristics. Their influence on pond performance therefore tends to be short term and most probably relate more to the fast dynamics of the pond process.

It must be noted however, that the multivariate analysis undertaken was not a study with the objective of producing regressional type models but rather a study aimed at interpreting the structure of the data matrices and to assess their consistency.

3.4 END POINT

The preliminary analysis of the two data sets showed conclusively that the data though noisy, contains meaningful signals from which dynamic information on the slow dynamics of the pond process can be obtained.

The principal components and factor analysis of the data matrices

revealed considerable consistency in the structure of the various surrogate measures representative of pond process descriptive variables. This no doubt instils confidence with regards the reliability and representativeness of the acquired data. Apart from the resulting confidence, an added bonus is that different representative measures within the same data set can be used for the verification of model structure of dynamic models developed for pond systems.

The classical use of different data sets in modelling exercises has been the partitioning of the data into calibration/verification and validation sets. The calibration/verification set is used for the calibration and verification of prespecified model structures and the validation set which comprises of an entirely different set of data to the one used for model calibration/verification, is used to validate the assumed model structure so that it can be held to be a law of the nature of the system under study.

The calibration/verification set is often subdivided into a calibration portion which usually comprises of the first half of the time series of data and the second half is used for the verification exercise which usually comprises of determining the predictive/forecasting ability of the model.

The pond data cannot be treated in this conventional fashion because, ponds exhibit seasonal behavioural characteristics and, the minimum period over which any seasonal effects can be observed is one year. A one year data set cannot therefore be subdivided into calibration and verification sets.

The conventional procedure is appropriate for the study of systems where knowledge about the system and its processes are well advanced and the system properties under investigation are uniquely identifiable (ie. global properties are the same for different scenarios).

In this study, both data sets for the dynamic analysis were prescribed as learning sets thus one set was not used for calibration/verification and the other for validation. Though both ponds had fairly similar external environmental conditions (eg. operating temperature range), they treated wastewaters which had significantly different characteristics. Each data set is therefore presumed to contain information on pond dynamics not present in the other.

In line with the adopted view point that knowledge on pond systems is not complete or near complete and taking note of the fact that the macroscopic properties of pond systems are dependent on several exogenous variables which may probably not be taken into account explicitly in model formulation, and also that, pond behaviour under different climatic types is bound to be different, it is realised that uniquely identifiable system properties, characteristic of all ponds, cannot be identified from the available data.

The two data sets acquired represent pond performance under essentially different scenarios. Pond 1 represents a typical highly loaded pond whereas Pond 2 represents a lightly loaded pond. Both pond data were thus deemed to describe two overlapping sets of possible pond behavioural modes.

Similarities in process dynamics arising from the identification exercises on both data sets will provide some basis for confidence in mechanisms incorporated in model formulation.

The process description presented in the previous chapter shows that though general behavioural trends are observable, pond behaviour in different environments are characteristically different. This clearly spells out the need for extensive data collection on ponds from characteristic environments to enable the evaluation of pond mechanisms and the development of pond models which simulate essential pond behaviour in the different environments.

It should also be recognised that significant field data will take considerable time to gather. Approximately one year of continuous sampling and analysis is required to obtain a base set of data to evaluate new concepts and a second year of continuous sampling and analysis will be required for the examination of any pond modification. By comparison with other forms of biological treatment, research on pond systems is a slow and difficult process. This clearly spells out the need for collaborative ventures in the collection of pond data and a unified approach in the search for simple and more reliable pond process models for design and performance evaluation.

CHAPTER FOUR: REVIEW AND EVALUATION OF PROCESS MODELS FOR DESIGN

4.1 GENERAL

In this chapter, process models used for pond design are reviewed and the representativeness of these models are assessed in relation to the pond process description presented in Chapter 2. Where appropriate, the models are evaluated using the acquired pond data.

Process models described here include; empirical; steady-state; and dynamic models. The review is conducted in a pseudo-chronological order as this gives a better overall picture of developments in pond theory and process description. This approach also shows the progress that has been made and how changes in concepts have been introduced.

Though pond design procedures have changed considerably over the years, the models upon which these design procedures are based have not been subjected to extensive evaluations to determine how representative and reliable they are as descriptions of pond process mechanisms. This lack of evaluation probably stems from the paucity in comprehensive data on which to base pond evaluation.

Early pond design methods were characterised by rules of thumb, engineering intuition and practical experience. Some of the early guidelines based on recommended loading rates and detention times

have been reported by Towne et al. (1956). These have resulted mainly in a wide range of design standards typically related to the climate of the locality under consideration. This is particularly evident in the review of the design standards used by the various states in USA (Canter et al., 1969; Canter and Englande, 1970).

Some of the most notable contributions to empirical design of pond systems in the past three decades have been those of Oswald (1960, 1972, 1975), and Herman and Gloyna (1957, 1958a,b,c). The empirical formulations include those that relate pond capacity to the influent organics concentration and temperature (Herman and Gloyna, 1957); permissible BOD loading to latitude (Arceivala et al., 1970); ultimate BOD removal to amount of oxygen produced due to solar influence (Oswald et al., 1957); performance prediction based on regression analysis (McGary and Pescod, 1970); and a procedure based on dimensional analysis (Larsen, 1974).

Further attempts at rationalising design following the empirical formulations have ranged from the use of simple steady-state models (Marais, 1966 and Thirumurthi, 1974) to exceedingly complex ecological model representations (Fritz, 1979).

4.2 NEED FOR REVIEW AND EVALUATION OF MODELS

The ideal design module for pond systems should incorporate a dynamic model which produces pond performance profiles for various performance indicators given information on waste characteristics, environmental factors and specific features of the proposed site. These profiles should be representative of actual pond performance,

with an indication of the level of uncertainty associated with the model output.

This is an ideal which can only be realised with improved knowledge on pond processes and a consequent development of more representative process models. The present lack of knowledge on descriptive laws on phenomena effecting transformations in ponds and other ecological systems precludes the development of complete all encompassing deterministic dynamic models. It is clear from the literature that considerable information has been gained from the study of the ecology of pond systems and the influence of environmental factors. However, little effort has been made to integrate this knowledge into an easily workable format for: Pond design; performance evaluation; and maintenance.

There is therefore the need for a concerted effort at unifying the search for better models of pond systems. This predicates the evaluation of existing process models no matter how inadequate they may be because insights gained from recognising the limitations of the existing models should provide the motivation and direction for the evolution of more representative system descriptions.

The process models reviewed are grouped together under headings which either relate to their originator(s) or to some specific feature underlying the procedure for design.

4.3 RECOMMENDED LOADINGS

4.3.1 GENERAL

The simplest pond design procedures are based on recommended

loading rates. These mostly relate to organic and hydraulic loadings with the recommended levels largely based on practical experience acquired from "successfully" operated ponds. Two examples of such recommended loadings are presented below.

4.3.2 GLOYNA (1971)

Based on world-wide data on successfully applied organic loading rates, Gloyna (1971) made generalizations on acceptable organic loading rates related to the various environments in which they were employed. These are summarized in Table 4.1.

These recommended loadings are based on the assumptions that:

- o The total evaporative and seepage loss is not greater than the influent flow volume
- o The per capita BOD contribution is 50gms per day;
- o The waste water production per capita/day is 100 litres.

4.3.3 CANTER AND ENGLANDE (1970)

Canter and Englande (1970), reported on the design criteria for organic loading and/or hydraulic detention time for facultative waste stabilization ponds for most states in the USA. Table 4.2 shows the average organic loadings and detention times for the northern (above latitude 42°), southern (below latitude 37°) and the central regions in the USA.

4.3.4 COMMENTS ON RECOMMENDED LOADINGS

The basic premise of design based on recommended loadings is that

a loading level producing satisfactory pond performance for a pond system in a particular environment can be expected to produce similar results for another pond system in a similar environment.

In line with the rationalisation of design, recommended loadings represent attempts at reducing the ambiguities associated with design based on a rule of thumb procedure since this involves the establishment of definitive levels of loadings for various characteristic environments. Specification of loading levels for different environmental conditions implicitly incorporates information on variations in pond behaviour with changes in environmental conditions.

The criteria for determining whether a particular loading level is acceptable usually depends on whether that loading level produces an adverse nuisance condition (eg. odours when ponds turn completely anaerobic). These loading levels therefore are strictly not related to pond effluents but are rather based on the state of the system.

Considering data from four different pond systems Finney and Middlebrooks (1979) showed that for three of the systems with design and actual surface loadings far below their respective design limits, frequent violations of effluent standards were observed.

Since the recommended loadings are not based on established effluent quality standards but rather relate to the state of pond systems, satisfactory pond performance (in terms of these standards) cannot be expected.

It must be noted that evaluations of pond effluent quality with respect to established secondary effluents standards have placed

pond effluent solids in the same category as solids of sewage origin. As discussed in Section 1.2.2, pond effluent solids are largely composed of algal cells which represent a more stable form of organic material with a characteristically different impact on disposal media.

A loading level resulting in acceptable state(s) of the system throughout the year will however not necessarily produce an acceptable pond effluent in terms of either its bacteriological, nutrients or carbonaceous organics quality.

Since no information on the likely pond effluent characteristics is presented, no assessment can be made of the impact of pond effluents on the disposal medium for a pond designed on the basis of recommended loadings.

Designs based on recommended loadings fail to recognise ponds as dynamic systems with effluent characteristics which change over the seasons. This presents an unacceptable level of uncertainty in information for pond design. As Lumbers (1979) rightly points out, recommended loadings are useful only in situations where limited information precludes the use of more rational methods.

4.4 EMPIRICAL FORMULATIONS

4.4.1 INDIAN METHOD

Based on experience in India, an expression for the permissible surface loading at different latitudes in the range $8-36^{\circ}\text{N}$ has been derived (Arceivala et al., 1970). The expression is of the form:

$$S_o = 375 - 6.25L \quad \dots 4.1$$

where S_o is the permissible loading in Kg BOD₅/ha/day and L is the latitude in °N. Equation 4.1 is said to be valid only in regions where there are clear skies for a least 75% of the year and pond depths fall in the range 0.9 to 1.5m.

COMMENTS

India is predominantly a tropical region with uniformly distributed sunshine and temperature, and no seasonal cloud cover. The resulting recommended loadings for latitude 8°N and 36°N are 325 and 150 kg BOD₅/ha/day respectively. This compares favourably with the range recommended by Gloyna (1971) for tropical regions (see Table 4.1).

Equation 4.1 is a linear equation showing the variation in recommended loading rates with latitude. This represents a formal description of the variation in loading rates with environmental conditions as applied to the Indian situation.

Since the basic design philosophy and principles are the same as for recommended loadings, the comments made on recommended loadings (Section 4.3.4) are equally applicable to design based on the Indian method.

4.4.2 GLOYNA AND CO-WORKERS.

Herman and Gloyna (1957) developed an empirical model for lagoons based on operating experience in Texas. Their proposed design

equation, which was based on an assumed influent organics concentration of 200 mg/l BOD and a minimum pond depth requirement of 1 metre, was of the form:

$$V = \frac{N \cdot q}{200} S_i t_o \exp (T_o - T)c \quad \dots 4.2$$

Where; V is the required pond volume in gallons; N is the number of people served; q is the sewage flow in gallons/capita/day; S_i is the influent BOD (mg/l); t_o is the optimal residence time (days); T_o is the optimal lagoon operating temp. ($^{\circ}\text{C}$); and c is the energy temperature characteristics of the van't-Arrhenius equation (0.0693).

On the basis of results derived from experimental work on ponds in series, they established that the retention time required for a fixed BOD reduction of 90% at 35°C was approximately 3.5 days.

In a later work, Herman & Gloyne (1958) suggested that temperature was the most important variable for pond design and on the basis of that, modified equation 4.2 to incorporate temperature effects on the optimal retention time. This was expressed as:

$$t_{o_T} = t_{o_{35}} \theta^{(35-T)} \quad \dots 4.3$$

Where; $t_{o_{35}}$ is the retention time for 90% reduction at 35°C ; t_{o_T} is the retention time for 90% reduction at $T^{\circ}\text{C}$; and θ is the Arrhenius constant equal to 1.072.

Based on the recognition that wastes containing toxic substances and high levels of sulphate reduced the efficiency of pond systems,

Gloyna and Espino (1967) modified the earlier design formulation to include sulphide concentration and waste water algal toxicity effects. These were incorporated as multiplicative factors resulting in the following representation:

$$V = 3.5 * 10^{-5} QLa(\theta(35-T))ff' \quad \dots 4.4$$

Where; V is the volume of the pond (m^3); Q is the influent flow rate (l/day); θ is the temperature coefficient; T is pond temperature ($^{\circ}C$); f is the algal toxicity factor; f' is the sulphide oxygen demand factor; and La is the influent organics concentration as BOD (mg/l).

The algal toxicity factor (f) is assumed to be equal to 1.0 for domestic and many industrial wastes and the sulphide oxygen demand factor (f') is dependent on the sulphate ion (SO_4^{2-}) concentration and is equal to 1.0 for sulphate concentrations less than 500mg/l.

They suggested that the average temperature of the pond in the critical or coldest month be used in design. Sunshine was not considered to be limiting but could be incorporated by multiplying the pond volume by the ratio of sunlight in the particular area to the average found in South Western USA.

The removal efficiency that can be expected based on unfiltered influent and filtered effluent BOD lies in the region of 80-90%. The recommended pond depth for treating most wastes is 1.0m and should be increased to 1.5m for systems treating waste with significant seasonal variations in daily flow.

COMMENTS AND EVALUATION

The basic principle underlying the empirical formulations by Gloyna and co-workers, is that the optimum detention time required to degrade a typical domestic type waste water of strength 200mg/l BOD is 3.5 days at 35 °C. This is then adjusted by organic loading and temperature factors to account for non-optimal conditions.

The algal toxicity factor is introduced in an attempt to incorporate knowledge that certain types of wastes inhibit algal development hence their ability to generate the dissolved oxygen required for aerobic oxidation. The sulphide oxygen demand factor, on the other hand, is to account for the oxygen demand exerted by reduced forms of sulphur (eg. H_2S). These factors result in increased pond capacity for wastes containing inhibitory toxic substances or sulphates in excess of 500 mg/l (sulphates in excess of 500 mg/l are typically present only in industrial effluents).

Though an expression for the determination of the algal toxicity factor has been suggested by Huang & Gloyna (1968), no recommendations have been presented for a means of evaluating the sulphide oxygen demand. Suggested concentrations of toxic substances producing a 50% reduction in chlorophyll-a are given in Table 4.3.

The factors (f and f') result in increased pond capacity which does not necessarily result in improved performance. There is usually a diminishing return for the improvements in performance achieved with increasing pond capacity for a system treating a waste stream containing material toxic to the biological community in pond

systems.

Since the temperature correcting factor increases exponentially with decreasing temperature, in terms of design, the worst conditions pertaining to the required capacity (volume) corresponds to the coldest period in the year.

Carbonaceous organics, bacterial conversion rates decrease considerably with decreasing temperatures, and based solely on this consideration, worst pond performance would be expected to occur in the coldest month. This is generally true for lightly loaded ponds in cold climates where there is considerable ice cover during the winter. In such ponds, the benthic subregion can be considered to be a sink of material.

For ponds in most parts of the world, the complex interactions occurring (eg. coupling between benthic and planktonic subsystems and the influence of wind and other exogenous factors), result in variable pond performance over the annual cycle. Poor performance can actually occur during any season or time of the year.

Pond performance is largely determined by such factors as:

- o The loading levels
- o The extremes in environmental conditions (eg. temperature, wind speed and solar radiation) which vary throughout the year
- o The inlet and outlet arrangements
- o The age of the pond system

This in effect means that a rigid criterion, such as "worst performance corresponding to the coldest periods", cannot be applied

to ponds in all environments.

The Gloyna formulations do not predict the pond effluent quality. An expected range for the removal of carbonaceous organics from the waste stream is however quoted.

Table 4.4 gives the results of the computed pond areas based on the EPA data (EPA, 1977a-d). For comparison, the actual pond areas are also given in this table. The computed pond areas were derived from the pond volumes in the coldest month of the year. Apart from the Kilmichael pond system which has a depth exceeding that recommended by Herman and Gloyna (1957), the resulting pond areas suggest that the Gloyna method is extravagant in terms of land. Similar observations were made by lumbers (1979).

4.4.3 SOLAR METHODS

Oswald and Gotaas (1957) developed a solar radiation method of design relating the ultimate BOD load that could be applied, to the amount of dissolved oxygen produced by algae. They suggested that pond design should be based on the average radiation for the month with the lowest visible solar radiation. This was calculated as:

$$S_{ave} = S_{min} + p (S_{max} - S_{min}) \quad \dots 4.5$$

where p (the sky clearance factor), is the ratio of the actual hours of sunshine to the total possible sunlight hours; and S_{min} and S_{max} are the monthly minimum and maximum solar radiation respectively.

Based on work on photosynthetic conversion efficiency as a

function of light, time, nutrients and temperature, Oswald (1960) observed that the utilization efficiency of conversion of visible energy by algae, fell within the range, 0.6 to 6.0 percent. This work led on to the following formulation relating algae production to solar radiation:

$$M = \frac{F \cdot S_{ave}}{h_c} \quad \dots 4.6$$

where; M is the algal productivity (kg/ha/day); S_{ave} is the visible radiation (gcal/cm²/day); F is the efficiency of conversion of solar energy by algae; and h_c is the energy required for the formation of one gram of algae (~ 4-6 kilo-calories).

The photosynthetic reaction based on an assumed average stoichiometry for algae gives a theoretical yield of 1.587 kg of oxygen for each kg of algae cell material synthesised. Oswald and Gotaas (1957) have related the organic load that can be applied to a pond system, to the amount of oxygen produced by algae by:

$$BOD = \text{rate of oxygen production} = 1.6 \cdot M \text{ kg/ha.d} \quad \dots 4.7$$

Jayangoudar et al (1970) and Arceivala (1981) have suggested that the ultimate BOD removed should be related to the amount of oxygen produced.

An oxygenation factor ranging from 1.2 to 1.6 (Oswald, 1960) has been recommended for use in pond design. The oxygenation factor which is expressed as the ratio of total oxygen produced to the total oxygen required, was introduced to account for the fact that

not all of the oxygen produced by algae is available for utilization by bacteria.

An oxygenation factor of 1.6 and a sunlight utilization efficiency of 6 percent results in a pond BOD loading (in kg/ha/day) directly related to the solar radiation measured in gcal/cm²/day.

The detention time required in a pond for the complete photosynthetic oxygenation of sewage is given by:

$$D = \frac{h \cdot d' \cdot C_c}{1000 F S_c} \quad \dots 4.8$$

Where, h is the specific heat of combustion of algae in gcal/mg, C_c is the algal concentration in mg/l, d' is the pond depth in cm, F is the photosynthetic conversion efficiency, D the detention period in days, and S_c the sunlight energy in cal/cm²/day.

The formula for predicting the depth of penetration, d (cm) of sunlight into the algal culture is given by:

$$d' = \frac{\ln S_0 - \ln S_d}{C_c a} \quad \dots 4.9$$

Where; S_0 is the visible light intensity at the surface, ft-c; S_d is the visible light intensity at which the photosynthetic rate of oxygen production is equal to the respiratory rate of oxygen use ft-c; C_c is the concentration of uniformly dispersed algae (mg/l) (assuming all algae to be nearly the same size), and a is an absorption coefficient, cm²/mg.

Based on results of outdoor studies on the relation of depth to overall light energy conversion efficiency, Oswald (1960)

recommended that pond basins should be designed with depths three times the value given by Equation 4.9.

Oswald et al. (1970) proposed an extension to the above design procedure to account for the removal of organic carbonaceous material through methane fermentation in regions where the mean pond temperatures in the coldest month exceeds 15°C . Carbon removal through methane fermentation was determined by:

$$L = 44.8 (T - 15) \quad \dots 4.10$$

Where; L is the BOD converted in Kg/ha/day; and T is the average temperature of the pond supernatant with a minimum temperature of 15°C for active fermentation.

COMMENTS

The solar methods are based on the assumption that photosynthetic efficiency is constant throughout the year and that the main factor controlling pond performance is algal activity. The organic loading that can be applied to a pond in a particular environment is then related to the oxygen producing potential for the particular environment.

Pond design based on the solar method presents a framework for incorporating environmental influences through an empirical relationship between solar radiation and algal productivity. In the absence of any locally derived recommended loading rates, this provides a method for evaluating loading rates likely to produce

acceptable state(s) for a pond system in a given climate. However, like design based on recommended loadings, no indication is given of the effluent characteristics that can be expected.

Though the importance of the benthic subregion in the overall stabilisation process is recognised in the formulations for design based on solar radiation, the activity in this region is depicted solely as that of carbon removal through methane fermentation. Equation 4.10 therefore gives the additional loading that can be applied on account of benthic activity. This does not take into account the internal loading to the planktonic region resulting from the feedback of intermediate fermentation products.

The critical conditions for design correspond to periods of low solar radiation and hence low production of dissolved oxygen. Low levels of solar radiation generally corresponds to the cold months where, for ponds in temperate (and colder climates), temperatures are likely to be less than 15°C and as a result, no carbon removal through methane fermentation can be expected.

As stated previously, critical conditions for pond systems cannot generally be classified as corresponding to cold months. The saturation level of dissolved oxygen in water, for example, increases with decreasing water temperature and this in turn results in an oxygenation factor which varies with the different seasons in the year; being greater for the hotter months than the colder months. The internal loading resulting from benthic feedback, and inhibition of mixing due to thermal stratification respectively result in a sediment oxygen demand and a non-uniform distribution of dissolved oxygen in the pond basin. These factors acting together

can result in pond "failure" at any time of the year.

Though solar radiation is a major factor influencing pond performance, other factors such as temperature and nutrients availability are known to control photosynthetic activity in ponds. Temperature in particular has been found to be the single most important variable governing algal blooms in pond systems (Mitchell, 1982). Also, the photosynthetic efficiency is not constant throughout the year and especially during periods of excessive algae production (blooms), self-shading of the algae results in a considerable reduction in the overall photosynthetic efficiency.

Since solar radiation is not the only factor influencing pond performance and the solar method depends on many assumed factors, its use should be restricted to situations where no locally derived loading rates based on operating experience is available.

4.4.4 THE REGRESSION APPROACH

A common empirical relationship that has been developed for pond systems to predict the organic load removal for given influent organics loadings is based on regression analysis.

In a survey of primary facultative ponds in tropical and temperate zones, McGarry and Pescod (1970) found that the areal BOD removal could be estimated from the areal BOD loading using the following equation:

$$L_r = 10.37 + 0.725 L_o \quad \dots 4.11$$

Where; l_r is the BOD removal (Kg/ha/day) and L_o is the BOD loading (Kg/ha/day).

The above equation was obtained by regressing organic load removal against the organic loading. Equation 4.11 was reported to have a correlation coefficient of 0.995, a 95% confidence interval of ± 32.8 Kg BOD/ha/day (see Figure 4.1) and to be valid for a loading range of 34 to 560 Kg BOD/ha/day. Examples of similar equations developed for pond systems in different parts of the world are presented in Table 4.5.

Based on a similar regression analysis, McGarry and Pescod (1970) found that the hydraulic detention time and pond depth had little, if any, influence on percentage areal BOD removal under normal operating ranges.

From results of plots of pond failure loading versus mean ambient temperature in the range 20 to 90°F (see Fig. 4.2), McGarry and Pescod (1970) derived the following formula:

$$L_o = 10 (1.054)^{T_m} \quad \dots 4.12$$

Where L_o is the areal BOD loading kg/ha/day; and T_m is the ambient mean monthly temperature °F. Pond failure was here defined in terms of the facultative pond remaining anaerobic throughout an entire 24Hr. period.

Linear equations reportedly incorporating factors of safety on loading levels predicted by equation 4.12 have been derived by Mara (1976) and Arthur (1981) respectively. These are:

$$L_o = 20 T - 120 \quad \dots 4.13$$

$$L_o = 20 T - 60 \quad \dots 4.14$$

where L_o is the permissible organic loading in Kg/ha/day and T is temperature in $^{\circ}\text{C}$. Equation 4.14 is for application to design of ponds in warm climates.

The typical design procedure based on the Regression Approach involves the determination of areal loading for the pond system based on the mean ambient temperature in the coldest month. The required pond size is determined from the desired effluent quality and the predicted removals based on Equation 4.11 or similar equations derived from local data.

COMMENTS AND EVALUATION

The quoted regression equations used for pond design are basically of the form:

$$L_r = a L_o + b \quad \dots 4.15$$

Where L_r is the areal organic load removal; L_o is the areal organic loading; and a and b are constants obtained from the regression analysis. A rearrangement of Eq. 4.15 results in the following form:

$$L_r/L_o = a + b/L_o \quad \dots 4.16$$

The coefficient a thus represents the average removal efficiency of the system under study when the influent organic loadings are relatively high since the b/L_0 term then becomes negligible.

An examination of Table 4.5 shows that the quoted regression coefficients are quite high. This is not surprising and should actually be expected. The high regression coefficients obtained are spurious (Lumbers, 1982). Apart from the fact that the slope represents the average percentage removal of organics in the particular pond system, the regression equations are meaningless.

Because pond effluent carbonaceous quality is largely independent of the influent quality (see Sections 3.14 and 3.15), regressing removal versus loading is tantamount to regressing two highly dependent variables.

Conducting a regression of removal versus loading is in effect considering ponds as being typical black box devices in which the main considerations of importance are relationships between outputs and inputs. Such regression models cannot be relied upon for the prediction of pond performance and should not be used in pond design.

Equation 4.12 relating pond organic loading to ambient temperature presents a formal procedure for determining pond loading levels likely to produce acceptable pond state(s) throughout the year for a given environment. Like the solar method for evaluating loadings, equations such as Eq. 4.12 are no doubt useful in situations where there is no information on loading levels based on

local experience.

The reported cases of pond failure used in the derivation of Equation 4.12, however, were for ponds in cold climates (ambient temperatures were around freezing, see Fig. 4.2) The solid line described by Equation 4.12, represents the boundary between pond failure and successful operation at the loading for the given environment described by the ambient temperature. It is quite clear that the region enveloped by Eq. 4.12 does not incorporate information on pond failure in regions with relatively high ambient temperature.

The linear approximations by Mara (1976) and Arthur (1981) imply that ponds cannot be used in regions where temperatures fall below 6°C and 3°C respectively. They represent gross simplifications of the relation between ambient temperature and loading.

4.4.5 DIMENSIONLESS METHOD

Larsen (1974) proposed a dimensionless design equation using data from a one year study at the Inhalation Toxicology Research Institute, New Mexico. From results of step-wise regression analysis incorporating environmental factors, he suggested that pond surface area could be estimated using the following formulation:

$$MOT = (2.468^{RED} + 2.468^{TTC} + 23.9/TEMP + 150.0/DRY) \times 10^6 \quad \dots 4.17$$

where the system of dimensionless numbers are outlined as follows:

$$MOT = \frac{\text{surface area} \times (\text{solar radiation})^{1/3}}{\text{influent flow} (\text{influent BOD})^{1/3}} \quad \dots 4.18$$

$$RED = \frac{\text{Influent BOD} - \text{effluent BOD}}{\text{Influent BOD}} \quad \dots 4.19$$

$$\begin{aligned} \text{TTC} &= \frac{\text{Wind speed (influent BOD)}^{1/3}}{\text{Solar radiation}^{1/3}} && \dots 4.20 \\ \text{TEMPR} &= \frac{\text{Pond liquid temperature}}{\text{Air temperature}} && \dots 4.21 \\ \text{DRY} &= \text{Relative Humidity} && \dots 4.22 \end{aligned}$$

The units for the various parameters and their conversion factors to make them dimensionless are given in Tables 4.6 and 4.7 respectively. Larsen (1974) suggested that his equation could be applied to pond design in any geographical region, since it was in a dimensionless form.

COMMENTS AND EVALUATION

Variables included in his final regression equation were obtained from F-test results of the regression of separate variables such as daily averages of evaporation rate, rainfall, wind speed, solar radiation etc. against the ratio of area to daily flow.

Larsen (1974) probably derived the motivation for this type of analysis from the desire to produce a dimensionless equation relating pond area to waste characteristics and environmental factors.

If note is taken of the fact that the real variable in the area/flow quotient is the flow and that the flow to a pond system is rather related to such factors as the pattern of water usage and the environmental factors affecting water use, then clearly, prediction of pond areas using the proposed dimensionless formulation is meaningless.

Table 4.8 shows results of the computed areas for the four pond systems based on the observed BOD reduction and on different assumed values for the RED number.

The relative insensitivity of the area computed by the Larsen (1974) formulation to the level of treatment measured by BOD reduction clearly demonstrates that this formulation is not a representative model of pond systems.

4.4.6 MARAIS AND SHAW PROCEDURE

Marais and Shaw (1961) found from operating experience in Africa, that aerobic conditions will be maintained in a pond systems if the total BOD of the pond effluent does not exceed the value given by:

$$L_e = \frac{700}{0.6d + 8} \quad \dots 4.23$$

Where d is the depth of the pond in feet and L_e is the maximum concentration of BOD in the pond consistent with aerobic conditions. Where there is a need to maintain a greater factor of safety against anaerobic conditions, a lower value of 500 or 600 is recommended for the numerator (Marais, 1966).

This formulation has been used as a check for aerobic conditions and to evaluate the required depth in designs based on the regression approach. Mara (1974) has pointed out however that for practical purposes, L_e is independent of the depth for the range of interest viz. 1 - 1.5m.

A further development by Marais (1966, 1970) based on an assumed

completely-mixed hydraulic conditions and first order kinetics is reviewed under the section on steady-state kinetic models.

4.4.7 DISCUSSION AND CONCLUSION - EMPIRICAL FORMULATIONS

Pond design criteria based on recommended loadings and empirical formulations have tended to account for the general observation that pond performance is closely linked with environmental factors. Recommended loadings have been based on surface loading rates as opposed to volumetric loading rates in recognition of the fact that in relative terms, the effects of the influencing environmental factors is more dependent on a pond's surface area than on its volume.

Attempts to quantify the permissible areal loading on ponds have related this to pond temperature, solar radiation and latitude. The close relationship between average temperature and solar radiation at most sites renders evaluations based solely on either temperature or solar radiation equally representative.

Pond design based on loadings, whether derived from experience or some empirical relationship, consider the systems to be analogous to static structures. This is because the capacity (size) of the system being designed is derived from information on loading levels likely to produce failure.

Design on this basis fails to recognise that ponds are dynamic systems with characteristics influenced considerably by exogenous factors some of which are random in nature. Also, no real indication is given of the effluent characteristics that can be expected from

the designed system.

Pond performance has been related mainly to the carbonaceous organics quality. Other quality parameters of importance such as pathogenic organisms, helminthic cysts and ova, and nutrients have largely been ignored.

It is clear from the developments in empirical formulations for pond designs, that the role of the benthic sub-region also has largely been ignored. The only formulation that recognises the benthic subregion is that by Oswald (1970). This, however, only represents a part description of the interactions between the benthos and the overlying planktonic region. The benthos is depicted as a reactive sink accounting for an additional removal of carbonaceous organics. The benthic sub-region however is not just a sink of material, but also a source of material (see Section 2.4).

The regression of removal versus loading represents a wrong application of statistical analysis. Spurious and meaningless results are derived from such analysis.

As models of pond systems, these empirical relationships are clearly inadequate. They are not representative of pond processes as described by the present level of knowledge.

4.5 MODELS BASED ON PROCESS RATES

4.5.1 GENERAL

This approach to design has progressed from the use of steady-state kinetic models to dynamic models of pond systems. Most

of the work on steady state models has assumed that the overall transformation process occurring with respect to carbonaceous organics removal can be represented by a first order auto-catalytic process. They have differed however in their representation of the hydraulic regime.

The dynamic models have mostly been developed on the conservation of mass principle; based on spatial aggregation of the variables of interest; and on the assumption that the hydraulic characteristics can be represented by a completely stirred tank reactor (CSTR) configuration. The resulting functional forms for the mass balance equations have resulted in the representation of pond dynamics in the form of coupled/uncoupled ordinary differential equations.

In this section, the steady-state and dynamic models for pond systems are reviewed and further evaluated using the acquired data. The conceptual description of ponds adopted for the various models developed are assessed in relation to the process description given and the functional forms used.

4.5.2 STEADY-STATE MODELS

4.5.2.1 GENERAL

The steady-state models developed for pond systems differ in the basic assumptions regarding the hydraulic characteristics. Workers like Marais (1966,1970) have assumed a CSTR configuration for characterizing the hydraulic regime whereas Thirumurthi (1969,1974) has described this regime with dispersive flow models after the work of Whener and Wilhem (1957).

4.5.2.2 MARAIS' MODELS

Marais (1970) presented two steady-state models for pond systems based on a mass balance of soluble carbonaceous organics substrate measured as ROD.

The following were the basic assumptions made in his model development:

MODEL 1

- o That there were no pollutional losses due to seepage
- o No settlement of influent BOD as sludge
- o Evaporation losses could be neglected
- o Organic substances degraded according to first order kinetics.

The resulting mass balance for the fate of organic substrate was of the form:

$$dS/dt = Q_i \cdot S_i / V - Q_e \cdot S_e / V - K \cdot S_e \quad \dots 4.24$$

Where; S_i is the influent ROD concentration (mg/l) ; S_e is the ROD concentration in the pond (mg/l); K is the 1st order decay rate (d^{-1}) ; Q_i is the influent flow rate (l/d); Q_e is the effluent flow rate (l/d); and V is the volume of pond (l).

Under steady-state conditions, Q_i and Q_e can be assumed to be equal and constant (ie. $Q_i = Q_e = Q$). Equation 4.24 therefore reduces

to:

$$S_e/S_i = 1/(1 + KV/Q) \quad \dots 4.25$$

This expression is the rate model most often applied in the design of ponds.

MODEL 2

The second model is a modification of the first incorporating a temperature dependent decay rate coefficient.

Marais (1970) assumed that this temperature dependency could be suitably represented by an Arrhenius-type relationship of the form:

$$K_T = K_{T_0} \theta^{(T-T_0)} \quad \dots 4.26$$

where; K_T is the reaction rate coefficient at the temperature $T^{\circ}\text{C}$ and K_{T_0} is the rate at the reference temperature T_0 ; and θ is the temperature relation constant. The recommended values suggested for θ , K_{T_0} and T_0 were 1.085, 1.2 and 35°C respectively.

COMMENTS AND EVALUATION

If Marais's formulation is a valid representation of pond processes, then the characteristic decay constant K would be expected to be constant throughout the year for a given pond system. In Marais's second model, seasonal variations in the rate constant have been incorporated in the form of the temperature relationship (Eq. 4.26).

To evaluate the validity of the assumption that pond interactions can be represented by a temperature dependent first order decay rate constant, the standard K values required to produce the observed effluent carbonaceous organics concentration (measured as BOD) for the different months in the year for the EPA (1977) data were obtained using Equation 4.25.

The results of the K values obtained from the mean monthly data for the four pond systems showed significant variations in values computed for successive months for the different pond systems. These results are shown in Figures 4.3 to 4.6.

Generally, a wide range of values have been reported for the 1st order decay rate constant K (see Table 4.9). It has been suggested that the observed variability arises partly from the fact that often there is an inconsistency in the measures used for evaluating the decay rate (Arceivala, 1981). The plots (Figs. 4.3-4.6) however, clearly show that even for cases where consistent measures are used, significant variability in the K values are observed.

The variability in the K values observed for data from different pond systems is understandable because this decay coefficient is not a universal constant and incorporates properties specific to the site eg. waste characteristics. However, the variability observed for different months for the same pond system clearly demonstrates that Marais's (1970) steady state model is not adequate.

Marais's assumption that no settlement of influent BOD as sludge occurs, is contrary to the pond behaviour described in Chapter two.

The settleable solids in the waste stream settle out to the benthic sub-region. The only situation where this assumption would hold is in the treatment of a highly soluble industrial effluent with practically no solids in the waste stream.

4.5.2.3 THIRUMURTHI'S MODELS

Thirumurthi (1967) proposed that waste stabilization ponds should be designed using the equations developed for non-ideal mixing in chemical reactors. Drawing upon the work of Whener and Wilhem (1956), Thirumurthi (1969) recommended the use of the following equation, as an approximation for the dispersed flow conditions in pond systems.

$$l_e = l_i . 4ae^{1/2d} / ((1+a)^2 e^{a/2d} - (1-a)^2 e^{-a/2d}) \quad \dots 4.27$$

Where; $a = (1+4Ktd)^{1/2}$; d is the dimensionless dispersion number, D/ul ; t is the retention time (days); u is the mean velocity (m/h); l is the length of travel (m); and D is the dispersion coefficient (m^2/h).

Thirumurthi (1967) suggested that the hydraulic variations of short-circuiting, mixing, and inlet and outlet device influences could all be accounted for by the appropriate choice of d and that the accurate determination of the decay rate constant K , was the key to the design process. Figure 4.7 shows the resulting design chart obtained.

Thirumurthi (1974) proposed further modifications to include factors for effects of toxic wastes and high organic loadings, when

conditions deviate from the standard environmental conditions set below:

- o Pond temperature at 20°C
- o Organic loading at 67.2 kg BOD/ ha-day
- o No toxic chemicals present
- o Minimum solar energy of 100 ly/day
- o No benthic load

The modified decay rate constant (k) was represented as:

$$K = K_s \cdot C_T \cdot C_O \cdot C_{TOX} \quad \dots 4.28$$

Where; K_s is the standard decay rate constant; C_T is the temperature correction factor; C_O is the organic load correction factor; and C_{TOX} is the correction for industrial toxic chemicals

The correction factors are of the following form:

TEMPERATURE

$$C_T = \theta^{T-20} \quad \dots 4.29$$

where T is the average temperature of the pond contents in the critical or coldest month °C; and θ is the temperature coefficient (1.036).

ORGANIC LOAD

$$C_O = 1 - \frac{0.083}{K_s} (\log_{10}(67.2/S_{oi})) \quad \dots 4.30$$

where S_{oi} is the organic load in Kg/day/ha.

TOXICITY

In the absence of industrial waste, the factor C_{TOX} is equal to one, but in the presence of certain toxic chemicals, algal cells cannot grow or do so only at reduced rates. Table 4.10, lists suggested values for the toxicity factor for selected organic chemicals.

COMMENTS AND EVALUATION

The temperature correction factor and organic loading factor both increase with increasing temperature and organic loading. This suggests that pond performance can be expected to be better for higher loadings and higher operating temperatures. This assertion is not generally supported by observations made on full-scale operating pond systems. Ponds in tropical and semi-tropical regions are more likely to have problems during hot periods due to excessive odour production. This is due to overloading resulting mainly from the feedback of organic material from the benthic subregion. Therefore the last of the the set of standard conditions specified by Thirumurthi (1974) is not realistic.

Though the hydraulic regime in pond systems deviates from the ideals of plug flow and complete mixing, a characteristic dispersion number cannot be specified for a pond system based on either its characteristic dimensions or some other physical factors. The spatial heterogeneity in pond basins resulting from effects such as stratification and short circuiting, result in a non-uniform dispersion characteristic for the typical unbaffled pond system. Even where the gross hydraulic flow pattern can be

characterised by some average dispersion coefficient, this coefficient is bound to vary significantly over the year (with the seasons and flow).

Thirumurthi (1967) has suggested that dye tracer studies be used in the evaluation of the dispersion coefficient, d . It is however quite clear from the process description presented that the value of d obtained for a full-scale pond will depend on both the time of the day and the season.

Whener and Wilhem's (1956) equation was derived for reactors which had uniform dispersion characteristics and in which the conversion process could be represented by a first order autocatalytic reaction under isothermal conditions. These conditions are far from being met in the field. Also, the dispersion in ponds is non uniform and because of the effects of stratification, isothermal conditions do not exist throughout the year for pond systems in most regions of the world.

Thirumurthi's (1969, 1974) representation of pond systems clearly has limitations for design.

4.5.2.4 UHLHAM'S RATE EQUATION

Uhlham (1979) has suggested that the decay rate K is dependent on such factors as the loading, the detention period and temperature. He has derived an expression for this relation based on work on laboratory scale ponds in which day-time and night-time conditions were simulated. The resulting relation was expressed as:

$$K = t \frac{-1(0.327+10.277/T+1/(L(0.25+0.476/T)))}{(1.391+ 1.304/T+(0.061+0.5T)/L)} \quad \dots 4.31$$

Where; K is the decay rate (d^{-1}); T is the temperature ($^{\circ}C$); and L is the organic loading ($gm/m^2/d$).

COMMENTS AND EVALUATION

The representativeness of Equation 4.31 is questionable considering the fact that, no indication was given as to whether or not there was an established benthic layer in the experimental pond system. Also, the synthetic waste stream used, is not representative of the characteristics of typical waste water stream to ponds treating domestic/municipal waste waters.

4.5.2.5 DISCUSSION AND CONCLUSION - STEADY STATE MODELS

A considerable amount of the work on steady-state models has centred on the determination of typical values for the first order decay rate constant. While steady state models are simplistic representations of pond systems and present an easy method for pond design, they do not reflect the seasonal characteristics of pond systems. Ponds are dynamic systems with behavioural characteristics that change with the seasons. An adequate pond model therefore should be able to reflect these dynamic characteristics and enable the determination of critical conditions for pond design.

4.5.3 DYNAMIC MODELS

4.5.3.1 GENERAL

Non-steady state mathematical models for pond systems have been developed (Marais, 1970; Ferrara, 1978; Fritz et al., 1979; Ferrara and Harleman, 1980). Functional representation of the various constitutive mechanisms taken into consideration in the formulation of mass balance equations have resulted in coupled/uncoupled simultaneous differential equations with a number of rate coefficients (parameters) which represent kinetic constants and also account for environmental and other non-deterministic and unknown influences.

The various non-steady state models that have been proposed to date for pond systems are presented together with the mechanisms taken into consideration in model build-up. The concepts upon which the models were developed are assessed in relation to the process description presented. The simulation and model assessment procedures adopted are presented in Appendix A.

4.5.3.2 OBJECTIVES OF SIMULATION EXERCISE

Where sufficient information exists in the measurement data for the simulation of the proposed dynamic models, state profiles are obtained using the proposed model structure and recommended parameter values.

The objective of these simulation exercises was to demonstrate the general non-transferability of rate coefficients and to assess the suitability of the models.

The rate coefficients have been assumed by most workers to be constant and independent of the particular system. The view therefore has been that a model that has been successfully verified using data from a pond system in one region can be transferred en bloc and applied to a pond system in another region.

In reality, proposed models together with their recommended parameter values producing an acceptable modelling result for one pond system cannot generally be expected to produce an acceptable modelling result for data from a different pond system. This is mainly because of the non-universality of the rate coefficients (model parameters). They cannot be compared with such physical constants as the universal constant of gravitation, G (Nm^2/kg) and Planck's constant, h (Js) which are essentially independent of system properties.

Most of the coefficients for models of ecological systems, such as ponds, incorporate site specific features. A constant parameter or rate coefficient for a particular pond system is therefore more related to material constants in the physical sciences such as density and modulus of elasticity.

The important issue that should be addressed in any modelling of ecological systems, is the structural representativeness of the functional forms adopted for the component mechanisms. It is also important that the conceptual description forming the framework about which the component mechanisms are derived, is a reflection of knowledge on the system and its interactions with the environment.

Ecological systems are particularly difficult to model because some process rates are subject to change with the changing seasons. This in effect means that the parameters relating to these processes take on different values at different times of the year and as such, a constant parameter value cannot be used for modelling the system over a typical annual cycle.

4.5.3.3 MARAIS (1970)

Marais (1970) proposed a three state model predicting pond effluent BOD, the volume of gas evolved from the benthic subregion, and the total sludge mass. A schematic representation of the mechanisms taken into account is given in Figure 4.8.

Marais (1970) conceptualised the benthic subregion to be both a source and a sink of organic material to and from the planktonic region. He assumed that a portion of the influent organic load (I_g) settled to the benthos, and the remaining portion (I_p) was completely mixed in the planktonic region.

The carbonaceous organics in the planktonic region were deemed to be transformed according to an overall temperature dependent first order decay process; the benthic activity was also represented by a temperature dependent first order decay process; and the rate at which gas evolved, was related directly to the benthic activity.

The functional representations for the mechanisms and the resulting differential equations representing the time rate of change of the three states are presented below:

1) Effluent Carbonaceous Organics (BOD)

$$dS/dt = i_p S_i \frac{Q_i}{V} + s_p K_s \frac{S_t}{V} \cdot 10^6 - \frac{Q_e S}{V} - K_1 S \quad \dots 4.32$$

2) Sludge Mass

$$dS_t/dt = -K_s S_t + i_s S_i (Q_i \cdot 10^{-6}) \quad \dots 4.33$$

3) Gas Evolution

$$dV_g/dt = C_v S_g K_s S_t \quad \dots 4.34$$

Where; S_i = influent BOD concentration, mg/l; S = BOD concentration of the pond liquid and effluent, mg/l; K_1 = 1st order decay coefficient, day^{-1} ; Q_i = influent flow rate, l/day; Q_e = effluent flow rate, l/day; V = pond volume, l; i_p = fraction of influent BOD dispersed throughout the pond; S_p = fraction of BOD resulting from anerobic activity in the benthos, entering the liquid layer; S_t = total sludge mass BOD, kg; K_s = 1st order decay coefficient for sludge mass, day^{-1} ; i_s = fraction of the influent BOD settling to the benthos; V_g = volume of gas produced, m^3 ; C_v = volume of gas liberated per unit mass of BOD destroyed, m^3/kg ; and s_g = fraction of sludge BOD which decomposes into gas.

The temperature relations for K_1 and K_s at the temperature $T^\circ\text{C}$ is given by the Arrhenius type temperature relation:

$$K_T = K_{T_0} \theta^{(T-T_0)} \quad \dots 4.35$$

Where; K_T is the rate in day^{-1} at the temperature T ($^\circ\text{C}$); K_{T_0} is the rate in day^{-1} at the reference temperature T_0 ($^\circ\text{C}$); and θ is the

temperature coefficient.

Marais (1970) suggested the following values for the fractional coefficients and the decay rate constants at 20°C:

$$i_s = 0.4 \text{ to } 0.6; \quad s_p = 0.4; \quad s_g = 0.6; \quad C_v = 0.62 \text{ (m}^3/\text{kg) BOD}; \quad K_s = 0.002 \text{ (d}^{-1}\text{)}; \quad K_1 = 1.2 \text{ (d}^{-1}\text{)}; \quad \theta_1 = 1.07; \quad \theta_s = 1.35$$

The pond effluent BOD obtained from the above rate equations (4.35 to 4.37) was termed the bacterial BOD by Marais (1970).

Recognising that algae contributes to the measured pond effluent BOD, Marais expressed the total effluent BOD by the following empirical relationship:

$$\text{BOD}_{\text{TOT}} = \text{BOD}_B + \text{BOD}_A \quad \dots 4.36$$

Where; BOD_{TOT} is the total pond effluent BOD; BOD_B is the bacterial BOD; and BOD_A is the algal BOD. The algal BOD component is related to the bacterial BOD and temperature (T) as follows:

$$\text{BOD}_A = \text{BOD}_B \quad \text{for} \quad T < 0^\circ\text{C} \quad \dots 4.37$$

$$\text{BOD}_A = 2 * \text{BOD}_B \quad \text{for} \quad T > 15^\circ\text{C} \quad \dots 4.38$$

$$\text{BOD}_A = 2 * \text{BOD}_B * \cos(\theta) \quad \text{for} \quad 0^\circ\text{C} < T < 15^\circ\text{C} \text{ and; } \dots 4.39$$

$$\theta = (15 - T) * \pi / 30$$

COMMENTS AND EVALUATION

Results of the effluent BOD simulations for Marais's model using the two data sets are shown in Figs. 4.9 and 4.11 respectively. The resulting model prediction errors and their autocorrelogram are shown in Figs. 4.10 and 4.12. It is clear from both the simulation

and errors plots that the model together with the recommended values for the fractional coefficients, temperature coefficients and decay rate constants fail to reproduce the observed pond behaviour described by the two data sets.

Marais (1970) used this model together with the coefficients to successfully reproduce pond behaviour based on data from South Africa. The lack of fit to the two data sets demonstrates the non-transferability of model coefficients. A different set of parameters and initial conditions **would** have given better results.

The resulting simulations and errors plots for Pond 1 data (Figs. 4.9 and 4.10) show that the discrepancy between model output and observed pond behaviour is pronounced over the hot periods in the year (see Fig 3.23 for a plot of temperature variations in the year). The seasonality in the resulting errors are shown in the autocorrelogram as a damped sine wave with significantly high serial correlations at low lags.

The resulting profiles for the total gas evolved and the sludge mass in the benthos are shown in Figs. 4.13- 4.16. These two states unfortunately have no corresponding descriptive variables in the data sets. The profiles for both volume of gas evolved and total sludge mass over the year, however, correspond to that which might be expected. The sludge mass for each case shows an accumulation of sludge in the colder periods followed by a reduction in the hot periods whereas gas evolution practically ceases during the cold periods.

No description of nutrient dynamics in pond has been presented by

Marais. He has however presented a model for faecal bacterial kinetics (Marais, 1975). This model was based on a CSTR configuration for the hydrodynamic regime and was of a form similar to his second model described in Section 4.5.2.2. The faecal bacterial kinetics model could not be evaluated because of data limitations.

4.5.3.4 FERRARA (1978)

Ferrara (1978) developed a "biogeochemical model" for pond systems. Biogeochemical, probably because it incorporated biological and nutrient dynamics with rate coefficients implicitly accounting for environmental influences. His conceptual description of the pond system comprised of a CSTR in which there was a cycling of material between organic and inorganic forms of the elements carbon (C), nitrogen (N) and phosphorous (P). The benthic region was envisaged to be a net sink of material removed from the planktonic region.

The transformations within the cycles for each of the individual elements (C:N:P) were assumed to occur independent of each other. Faecal coliform dynamics were presented in a form similar Marais (1975).

A schematic representation of Ferrara's conceptual description of the pond system is shown in Figure 4.17. The main transformations within the individual cycles are shown in Figure 4.18 and summarised as follows:

- o Mineralisation of organic compounds :- represented by a 1st order autocatalytic temperature dependent term.

- o Organism growth :- represented by the classical hyperbolic form of nutrient dependent uptake kinetics - the Monod type expression.
- o Net loss of material by settling of non-biodegradeable organic matter, precipitation and adsorption of inorganic phosphorus , and denitrification of nitrates :- represented by a 1st order temperature independent term.
- o Removal of faecal coliform by death and predation.

The resulting equations for the time rate of change of the states for his model are presented below:

1) Organic Carbon (OC)

$$d(OC)/dt = \frac{Q_i}{V} \cdot OC_i - \frac{Q_e}{V} \cdot OC - (K_{12} + K_{1s}) \cdot (OC) + K_{21} \frac{(IC)}{(K_{sc} + IC)} \cdot (OC)$$

.. 4.40

2) Inorganic Carbon (IC)

$$d(IC)/dt = \frac{Q_i}{V} \cdot IC_i - \frac{Q_e}{V} \cdot IC + K_{12}(OC) + K_{20}(CO_{2s} - CO_2) - K_{21} \frac{(IC)}{(K_{sc} + IC)} \cdot (OC)$$

.. 4.41

3) Organic Nitrogen (ON)

$$d(ON)/dt = \frac{Q_i}{V} \cdot ON_i - \frac{Q_e}{V} \cdot ON - (K_{34} + K_{3s}) \cdot (ON) + K_{43} \frac{(NH_3 - N)}{(K_{sn} + NH_3 - N)} \cdot (ON) + K_{53} \frac{(NO_3 - N)}{(K_{sn} + NO_3 - N)} \cdot (ON) \cdot (X)$$

.. 4.42

4) Ammonia Nitrogen (NH₃-N)

$$d(\text{NH}_3\text{-N})/dt = \frac{Q_i}{V} \cdot \text{NH}_3\text{-N}_i - \frac{Q_e}{V} \cdot \text{NH}_3\text{-N} + K_{34}(\text{ON}) - K_{43} \frac{(\text{NH}_3\text{-N}) \cdot (\text{ON})}{(K_{sn} + \text{NH}_3\text{-N})}$$

.. 4.43

5) Nitrate Nitrogen (NO₃-N)

$$d(\text{NO}_3\text{-N})/dt = \frac{Q_i}{V} \cdot \text{NO}_3\text{-N}_i - \frac{Q_e}{V} \cdot \text{NO}_3\text{-N} - K_{5s}(\text{NO}_3\text{-N}) - K_{53} \frac{(\text{NO}_3\text{-N}) \cdot (\text{ON}) \cdot X}{(K_{sn} + \text{NO}_3\text{-N})}$$

.. 4.44

6) Organic phosphorus (OP)

$$d(\text{OP})/dt = \frac{Q_i}{V} \cdot \text{OP}_i - \frac{Q_e}{V} \cdot \text{OP} - (K_{67} + K_{6s}) \cdot (\text{OP}) + K_{76} \frac{(\text{IP}) \cdot (\text{OP})}{(K_{sp} + \text{IP})}$$

.. 4.45

7) Inorganic phosphorus (IP)

$$d(\text{IP})/dt = \frac{Q_i}{V} \cdot \text{IP}_i - \frac{Q_e}{V} \cdot \text{IP} + (K_{67} - K_{76} \cdot \frac{\text{IP}}{(K_{sp} + \text{IP})}) \cdot (\text{OP}) - K_{7s}(\text{IP})$$

.. 4.46

8) Faecal Coliform (FC)

$$d(\text{FC})/dt = \frac{Q_i}{V} \cdot \text{FC}_i - \frac{Q_e}{V} \cdot \text{FC} - K_{8s}(\text{FC})$$

.. 4.47

The various modelled states are numbered 1-8. Q represents the flow, V the pond volume and the subscripts i and e refer to the influent and effluent streams respectively. The rate constants are labelled with subscripts which show the direction of transfer. For example, the rate at which the organic carbon (state 1) is transferred to the pond sediments is represented by K_{1s} . The half saturation constants for carbon (K_{sc}), nitrogen (K_{sn}) and phosphorus (K_{sp}) were assumed to be equal to 0.5, 0.25 and 0.01 mg/l respectively. The suggested values for the model coefficients are given in Table 4.11.

The temperature relation for the temperature dependent rates was represented by the Arrhenius type relationship (see Eq. 4.38) with a reference temperature of 20°C and temperature coefficients 1.036 and 1.065 for mineralization and synthesis reactions respectively.

COMMENTS AND EVALUATION

Ferrara's (1978) model incorporates a greater number of descriptive variables (states) than Marais's (1970) model and describes the dynamics of eight states of the pond process including measures for nutrients and faecal indicator organisms.

In his conceptual representation of pond processes, he considered the benthic subregion to be a net sink of material. This is characteristically the case for pond systems in extremely cold environments with no significant wind or benthic activity.

It is clear from the current understanding of the process (presented in Section 2.4) that dependent on factors such as the ORP

and the hydraulic regime, the benthos can act as either a net source or a sink of both organic and inorganic material to the overlying planktonic region.

The states described by Ferrara include organic and inorganic forms for the elements carbon (C), phosphorus (P) and nitrogen (N (ie. Eqs. 4.40-4.47). The equations representing the dynamics of inorganic carbon, organic phosphorus and inorganic phosphorus, Eqs. 4.41, 4.45 and 4.46 respectively, have terms representing their contributions via the influent waste stream. The pond data however has no corresponding measures for these states. As a result, no state profiles can be produced. Therefore, the the model structure in relation to these states cannot be verified.

Ferrara presented simulation plots for total phosphorus and total COD but gave no indication as to how these were derived from his set of equations.

If the total phosphorus is assumed to be made up of the sum of the inorganic and organic components, then a new state variable (TP) can be derived for the total phosphorus by adding together Equations 4.45 and 4.46. This results in a first order model for total phosphorus in which the net transformation process is the settling of total phosphorus to the benthos. The resulting dynamic equation for total phosphorus is given by:

$$d(TP)/dt = \frac{Q_i}{V} TP_i - \frac{Q_e}{V} TP_e - K_{ps} TP \quad \dots 4.48$$

The simulation and errors plots for Pond 2 data using the above simplifications for the phosphorus dynamics and Ferrara's full

representation for the nitrogen transforms are shown in Figs 4.19-4.26. The simulation plots are similar to those produced by Ferrara (1978). Based solely on a visual inspection of the simulation plots, the prediction of the effluent total phosphorus ammonia and nitrate (Figs. 4.19, 4.21 and 4.23) would be classified as fairly good. It is clear, however, from Figs 4.25 and 4.26 that the effluent TKN was not modelled adequately.

The simulation plot for effluent total phosphorus (Fig. 4.19) produced an R_{sqd} of 0.95 which indicates a good fit to the data. The errors (Fig. 4.20) appear to be evenly distributed about the mean zero line and based solely on these two criteria, it would seem that the adopted model structure together with the recommended parameter values represents pond total phosphorus dynamics described by Pond 2 data. The autocorrelogram of the errors however clearly shows that these errors are highly structured and suggests some seasonality in the errors. The autocorrelogram appears to be a damped sine wave with a half period of about 11 weeks. This suggests that a pond mechanism with a characteristic of roughly two cycles in the year could be missing from the model.

The simulation and errors plots for effluent ammonia (Fig. 4.21 and 4.22) show that the model produced a good fit for the hot periods of the year but failed to reproduce the observed ammonia profile over the cold periods (see Fig. 3.50 for the temperature profile for Pond 2 data). This suggests the possibility of seasonal variations in the reaction rate constants.

High levels of effluent ammonia in pond effluents in winter

months have been observed for ponds in temperate climates in both the northern and southern hemispheres. Different mechanisms have been postulated to account for this observation and these include: the cessation of nitrification at low pond temperatures; and the production of ammonia from the liquefaction of proteinaceous material derived from algal kills.

The simulation results for effluent nitrate (Figs. 4.23 and 4.24) show no seasonality in the errors. However, the autocorrelogram of the errors are significantly different from white noise but nonetheless show that the slow dynamics of nitrate has been described fairly well.

Though a fairly high R_{sqd} value was obtained for the effluent TKN simulation plot (0.70) it is clear from both the simulations (Fig. 4.25) and errors (Fig. 4.26) plots that this system state was not predicted well by the model. The errors are clearly biased and structured. The autocorrelogram of the errors strongly suggests the absence of a component with a half period of about 12 weeks which apparently is similar to that observed for the total phosphorus,

The apparent 2-cycles in the year observed for the errors plots for total phosphorus and TKN suggested that the missing pond mechanism could possibly relate to the transformation of organic material in ponds.

The states in Ferrara's model simulated using Pond 1 data are inorganic and organic phosphorus. Though measures for the inorganic nitrogen compounds (namely, ammonia and nitrate) were available, no measure pertaining to the organic nitrogen was present in the data

set. And since Eqs. 4.42-4.44 which represents the transformation processes within the N-cycle are coupled, no simulation for N-forms for which measurements were available could be made.

The simulation and errors plots for inorganic and organic phosphorus for Pond 1 are shown in Figs. 4.27-4.30. The simulation plot for the effluent ortho-phosphorus (Fig. 4.27) had a high R_{sqd} and the model outputs lie within the observed effluent inorganic phosphorus range. However, the autocorrelogram of the resulting errors (Fig. 4.28) shows that these errors are highly structured and suggests the absence of a seasonal component within the model having roughly two cycles in the year.

The model underestimates the observed net removals for the organic phosphorus state (see Figs. 4.29 and 4.30). This could probably be due to Ferrara's specification for the rate at which organic phosphorus is removed, being less than the observed.

The resulting simulation and errors plots for Pond 1 data reinforce the claim that generally rate coefficients are not transferable. Although the adopted model structure, together with the prescribed values for the parameters, produced a fair representation of pond dynamics displayed by the Pond 2 data set, the model applied en bloc to the Pond 1 data produced unacceptable results.

4.5.3.5 FRITZ (1978)

Fritz (1978) developed a non-steady state model representing the pond system by 12 state variables. These included , organic

substrate, specifically COD, algal mass, bacterial mass, dissolved oxygen, inorganic carbon, nutrients and detrital buildup.

The following were the energy inputs taken into account and assumed to be the main external driving forces:

- o Solar radiation
- o Wind shear
- o Air temperature

Precipitation, evaporation and seepage were taken into account in a hydrologic balance for the computation of the effective flow through the pond system.

The following assumptions were made:

- a) Pond hydraulics can be represented by a CSTR
- b) The effects of solar radiation and other heat balance components can be represented by a spatially averaged value
- c) The effects of sunlight can be represented by a depth integrated average.
- d) Algal growth is limited by light, temperature, carbon dioxide, inorganic nitrogen and phosphorus availability.
- e) Bacterial growth is limited by oxygen, substrate, inorganic nitrogen, phosphorus availability and temperature.
- f) Pond temperature can be computed via a heat balance between: inflow, outflow and interfacial heat transfer

- g) All biological mass is active
- h) Zooplankton and Macrophyte growth and decay do not exert a significant influence on pond performance
- i) No chemical precipitates such as CaCO_3 or phosphorus are formed
- j) The portion of the influent COD considered to be refractory and non-soluble is temperature dependent
- k) There is no oxygen demand from the benthic sludge layer.
- m) The anaerobic processes in the benthos can be modelled in a "black-box" fashion

The assumption that no precipitates are formed (i) and that there is no oxygen demand from the benthic sludge layer (k) are all contrary to observed pond phenomena and are not supported by the process description presented in Chapter 2.

Fritz's (1978) conceptual description of the pond system is shown in Fig 4.31. The benthic subregion is taken to be a zone to which bacterial and algal cells settle and from which there is a feedback of inorganic nutrients into the overlying planktonic region. This description of the benthic interaction with the planktonic region actually represents a subset of the interactions described in Chapter 2. The feedback component to the planktonic region is not made up of only inorganic nutrients, but also of organic material resulting from the fermentation process occurring in the benthos.

A schematic representation of the interactions between the 12

states modelled by Fritz (1979) is given in Figure 4.32 This clearly shows the complex nature of Fritz's (1979) model.

His equations for the rate of change of the modelled states were of the form:

$$dC/dt = C_i/\theta - C/\theta + \sum r_{c,j} \quad \dots 4.49$$

where; C_i is the influent concentration of the modelled state; C is the concentration of the particular state variable in the pond liquid; θ is the hydraulic residence time; $r_{c,j}$ is the volumetric reaction rates for component mechanisms accounting for transformations in the modelled state (C).

The interactions represented in Fig. 4.32 formed the basis for the derivation of the highly coupled kinetic formulations. The various rates in the equations for the rate of change of the states are shown in Table 4.12. The component reactive processes for algal and bacterial dynamics presented by Fritz, (1978) are based on the multiple nutrient limiting growth kinetics in which each limiting nutrient is represented by a Monod type expression.

Fritz's (1979) representation of pond dynamics is exceedingly complex (12-states and 25-parameters). With such a representation, the number of possible combinations of initial conditions and parameters values is enormous. It was not possible to reproduce Fritz's (1979) modelling results because of lack of information on the input characteristics and initial conditions for the modelled states which had no corresponding measures in the data set viz.

algae, bacteria and dissolved oxygen.

Plots of Fritz's (1979) simulations for Pond 1 data are shown Figs. 4.33 and 4.34. It is quite clear from these plots that apart from total COD, the salient features of the observations for these states are not predicted well by the model.

The simulation plot for filtered COD (see Fig 4.33) shows that the lack of fit between model predictions and observations is pronounced in the hotter months of the year. This suggests the absence of a pond mechanism with a more profound effect in the hotter months of the year possibly the internal loading component.

4.5.3.6 SUMMARY

The simulation exercise showed the usefulness of the adopted model assessment procedure. Quite clearly each of the separate criteria (viz the simulation and errors plots) were insufficient as sole criterion for model assessment and could lead to erroneous conclusions regarding the suitability of the models.

Fritz's model in particular represents the characteristic complex ecological model format in which it is virtually impossible to conduct posteriori analysis to determine the dominant pond mechanisms, sensitivity to parameters and to evaluate the representativeness of the functional forms in the adopted format for process dynamics.

4.6 END POINT

It is evident that the empirical formulations for pond design are

not adequate pond process models.

The lack of information on the likely effluent quality from a pond designed on the basis of the empirical formulations would only be acceptable if the design loadings were such that complete waste treatment was guaranteed.

The benthic subsystem, an essential part of pond systems, has largely been ignored in the empirical formulations for pond design proposed. This may in part be because most experimental ponds, especially those based on synthetic wastes, barely have established benthic subregions. The results obtained from such studies therefore mostly represent only the pond processes occurring in the planktonic region.

Consideration of pond dynamics in only the planktonic region has lead to the general wide spread notion that the critical conditions for pond performance correspond to the coldest month of the year when the activity of colonizing microbial populations is minimal.

The ideal design module described in section 4.2 suggests the development of process models which predict the likely effluent characteristics.

The steady-state models developed for pond systems to date attempt to give an indication of the possible long-term effluent characteristics for a given loading level. However, a comparison of the conceptual description forming the basis for these steady-state models with the process description based on the current knowledge on pond processes reveals that these steady-state models do not incorporate information on important system interactions. Also

since ponds operate in a dynamic environment, they rarely attain steady-states.

The incorporation of toxicity factors in empirical and steady-state formulations for pond design represents a useful information up-dating procedure in which knowledge of the influence of inhibitory substances are incorporated in pond design. The complex pond environment however means that caution should be exercised in the inclusion of knowledge based on the study of the effects of substances acting singly. This is because in admixture, toxic substances can produce antagonistic or synergistic effects.

There is the need for an extensive evaluation of toxicity thresholds for various common toxic substances found in wastewaters. This should be based on treatability tests in the typical complex environment found in pond systems. This would provide an information base on acceptable levels for the various substances.

The development of dynamic models for pond systems represents the right step in the evolutionary process involved in the rationalisation of design. Dynamic models provide a means for predicting effluent quality profiles for different envisaged scenarios and presents a flexible approach to pond design. Dynamic modelling of ponds and other ecological systems is however, both an art and a science in its infancy. The question that has to be addressed at the present stage is the representativeness of the models developed to date. These have ranged from simple models describing only a few states of the system (Marais, 1970), to models which attempt to incorporate all knowledge on pond interactions.

The complex ecological models developed for ponds actually represents a big jump in this evolutionary process. This is because, these basically fail to recognise that given the present level of knowledge on descriptive laws for ponds and other ecological systems (described as ill-defined exceedingly complex systems), completely deterministic process models describing all process interactions cannot be developed.

As more knowledge is acquired on process interactions and with the development of structurally representative descriptive formats for the component mechanisms, more representative process models will be developed.

The complex models proposed to date actually outstrip the present capacity for data collection. They describe system states for which there are no corresponding measures available in the current data sets. These models cannot therefore be verified using the current data and as such, it is virtually impossible to assess the representativeness of the functional relationships adopted for the various constitutive mechanisms.

Dynamic models should be used in a progressive fashion to help identify data needs. The level of complexity of pond models should therefore be compatible with the currently available measurement data.

The rate coefficients or parameters in models for ponds should be viewed more in terms of system dependent properties in a way analagous to material constants (eg. density and Young's modulus). Emphasis should therefore be laid on how adequately the structural

format adopted for pond mechanisms describes the range of behaviour described by the available data.

A structurally adequate model, will produce acceptable reproductions of observed system behaviour for different data sets with realistic parameter values. These would characteristically be different for ponds in different environments. In fact because ponds and other ecological systems exhibit seasonal characteristics, some parameters are likely to change in the typical annual cycle and in the long-term as the system ages. A parameter describing the metabolism of a principal group of organisms such as algae, for example, is likely to change with the seasonal changes in the predominant species.

There is therefore no need for deriving point estimates or a recommended value for system parameters. Rather, the range of values for structurally verified functional forms describing observed pond behaviour in characteristic environments should be sought.

Pond design for a characteristic environment would then involve the use of predefined performance criteria and Monte Carlo methods for producing plausible behavioural profiles for combinations of parameter values within the range for the environment. This would enable the identification of factors combining to produce the worst performance for the different performance assessment criteria used. Pond design, involving such exercises as sizing units and recommending operational strategies, could then be based on these worst conditions.

The next chapter deals with a pond identification exercise which

aims to retrieve information on pond dynamics from the acquired data using both "block" and "recursive" estimation techniques in a hypothesis evaluation framework.

CHAPTER FIVE: IDENTIFICATION OF POND SYSTEMS

5.1 GENERAL

The "experimental testing" procedure for the identification of systems, usually involves obtaining data on the response of the system to test signals (controlled inputs). This procedure is often used in the identification of electrical and chemical engineering systems. These systems are mostly well defined and can easily be subjected to experimentation using these controlled signals.

It is generally recognised however, that even for such well defined systems, it is difficult to differentiate between second, third and higher order systems using such identification techniques and that systems with widely differing time constants are even much more difficult to identify (Coughanowr and Koppel, 1965).

The typical pond data resulting from evaluation of performance exercises comprises of "test signals" which the systems analyst has had no control over what so ever. Information on input or control signals leading on to the state of the system at the time of initiation of the sampling scheme is commonly not available.

Owing to the exceedingly complex nature of ponds and other ecological systems, and the difficulty in obtaining data from controlled field studies, the above identification procedure is practically not applicable to such systems.

Identification of ecological systems by recourse to data resulting from monitoring exercises during routine operation is not

an exact science. Systems representation by mathematical models have generally involved the translation of "valid linguistic descriptions" of the system and its components into functional mathematical expressions. The functional forms adopted represent mathematical interpretations of underlying theories and hypothesis on systems dynamics.

There is a broad spectrum of dynamic mathematical model formats for representing system dynamics. Models at the ends of the spectrum are classified as "Black-box" and "Deterministic" models respectively (Karplus, 1976).

Black-box models simply relate system inputs and outputs presenting virtually no information on the possible internal mechanisms accounting for system behaviour. Deterministic models on the other hand attempt to account for all the internal interactions and mechanisms involved in the observed process.

Since the objective was to obtain a better insight into the laws governing the dynamics of pond processes, internally descriptive models which aim to explain the observed aggregate effect behavioural characteristics were developed.

The critical factor determining the validity of a completely deterministic model is the extent to which it can be derived from basic laws and insight into the internal structure of the process. The problem with developing mathematical models for ecological systems is the fact that knowledge on the basic laws is incomplete. This problem is compounded by the fact that for the observed relations which have "linguistic descriptions", there are usually

quite a number of mathematical relationships which fit these linguistic descriptions.

In this study, uncertainties resulting from incomplete knowledge are incorporated by lumping together mechanisms and considering overall effects.

5.2 INTERNAL DESCRIPTIVE MODELLING

5.2.1 CONVENTIONAL APPROACH

Most internally descriptive mathematical modelling exercises follow a conventional approach described as a methodology based on a "reductionists" philosophy by Young (1978). This deductive approach involves the development of comprehensive deterministic models of the system based on the reconstitution of quantitative expressions for sub-component processes deemed to be operative in the system. This usually results in a model of immense structural complexity with several parameters or rate coefficients.

Most of these parameters are often obtained from the literature or laboratory based experimentation. Parameters are also determined from trial and error or on the basis of some parameter estimation technique. This conventional approach is the modelling strategy that has been adopted in the dynamic modelling of pond systems to date (Marais, 1970; Ferrara, 1978; Fritz, 1979).

Models can be developed to agree or fit field data by incorporating a large number of parameters. This essentially increases the degrees of freedom of the mathematical structural

representation model. By applying an appropriate parameter estimation technique, a fairly good fit to the data set can be obtained. Such models are usually artefacts of the particular modelling exercise (Chapman, 1975) and in cases where the estimated parameters are devoid of any apparent physical meaning, the identification exercise can be compared to a curve fitting procedure. The functional forms for such models cannot usually be validated with an entirely different set of field data.

The stability of parameter estimates arrived at by curve fitting has received considerable attention (Kuczera, 1982 ; and Johnston and Pilgrim, 1976). Mein and Brown (1978) while examining the stability of optimized parameter estimates noted that parameter estimates can be strongly dependent on each other and can also exhibit instability in a fashion analagous to near-multicollinearity in linear regression.

Exceedingly complex model representations result in redundant parameters to which the model output is insensitive. It is not uncommon to observe parameter interactions and compensatory behaviour for the parameters of such models. Often, quite a number of these parameters are unidentifiable.

In this respect the good model is one that is parsimonious in relation to its parameters, has functional components which have direct interpretations with regards to the possible laws governing the process and at the same time mimics the salient temporal behaviour of the system.

5.2.2 NEED FOR A NON-CONVENTIONAL APPROACH

It was realised from the evaluation exercise in Chapter 4, that complex models which describe system states with no corresponding measures in the commonly available data, cannot be evaluated and hence envisaged mechanisms cannot be verified.

It is evident from pond literature, though, that a great deal of information is available on hypotheses of mechanisms deemed to be operative in pond systems. These hypotheses have however, hardly been tested. Data limitations and the complex nature of the model formats adopted for deterministic models has made their testing a daunting task. As a result of the lack of verification of hypotheses, no real confidence can be placed on the process models (empirical, steady-state and dynamic) developed since no assessment can be made of their representativeness.

In view of the factors discussed above, it was concluded that there was a need for the adoption of an approach to internally descriptive modelling which takes into account the lack of information on systems dynamics and the noisy nature of measurement data on which systems identification is based. Ideally, this approach should present a formal way of incorporating the fact that understanding of the system is incomplete.

The process information content of the available pond data has been discussed in Chapter 3 where the need for the initiation of data collection exercises was clearly identified. It is believed that systems identification exercises based on an interaction between a priori information on pond theory and posteriori

information resulting from pond data analysis should help reveal the areas of critical uncertainty in pond information and help identify data needs.

Since it is clear that there are limitations in the presently available pond data, identification exercises using such data should aim at producing models which "explain" the data in the simplest physically meaningful manner, and to a degree which is defined carefully in relation to the uncertainty in the data. The modelling approach adopted, should allow for model update as more reliable information becomes available and model complexity should be dictated by the nature and type of measurement information available. Ideally, behaviour should be predicted within the margins of error to be expected from the imprecisions in the observations.

5.2.3 GENERAL PHILOSOPHY OF ADOPTED APPROACH

The philosophy of the approach adopted for the identification exercise can be stated as one of: Abstracting as much information as possible from the available data on pond systems using the a priori information as a search guide and taking into consideration the fact that at present, knowledge on pond systems is far from complete.

Systems identification relating to the above objective then essentially becomes an information retrieval exercise.

With the realisation that ambiguities exist in the a priori information on pond systems and that observations are noisy, analysis techniques used should be as robust as possible. Ideally

these should be such that it is possible to discern when the maximum possible information has been extracted and should reflect uncertainties resulting from representing the pond system with the chosen model structure. Unfortunately, no unified criteria for discrimination exists and as such ad hoc procedures had to be adopted.

The ad hoc procedure adopted for discerning when the maximum possible information had been abstracted from the data was basically the criterion that residual sequences were not significantly different from white noise.

5.2.4 PROCESS DESCRIPTION AS INPUT-STATE-OUTPUT

On a macroscopic level, the pond system can be represented by the input-state-output diagram shown in Fig. 5.1. The real pond system is visualised as being represented by descriptive variables. These variables are those thought to be relevant in describing the dynamics of the system.

The descriptive variables are classified as observed or unobserved depending on whether they have representative measures in the data sets. The observed variables are further classified as input or output variables. The external input variables ($U; U^*$) are those considered to perturb, influence or affect the pond system from outside its overall boundaries (see Fig. 5.2). The output variables ($Y; G; S$) are those that describe the resulting observed effects when the inputs perturb the states of the system and the states of the system ($X; X'; X_R; \dots$) can be visualised as descriptors of the internal composition of the system at any given

time instant. They describe the current status of the internal cycling processes (IC) resulting in the observed transformations.

Apart from the external inputs some of which are measured ($U_m; U_m^*$), the pond system is controlled to an extent by internal inputs (FB; SB) which are mostly unmeasured and to a large extent not well defined.

Because not all of the input (control) variables influencing pond dynamics are measured or identified, the observations made on the outputs from the system (Y_m) are made up of the response of the system to the deterministic inputs (both observed and unobserved), unknown random influences and the noise (ξ) in the measuring process. The measured inputs are also not perfectly observed and have measurement noises ($\underline{U}$ & $\underline{U}'$).

The data set therefore comprises noise corrupted measurements of the observed inputs to and outputs from the system and the main identification exercise can be stated as: Given observed system characteristics ($U_m; U_m^*; Y_m$) identify how U is transformed to Y and how the transformation processes are influenced by U^* and FB.

Because of noise corruption, the observed variables are stochastic processes and estimation procedures have to take this into consideration and address the likely effects of noise contamination on the information retrieval process.

For example, if the sampled values of the inputs and outputs (see Fig 5.1) are completely dominated by the noise terms ($\underline{U}$; $\underline{U}'$ and ξ) then the resulting data (U_m ; U_m^* and Y_m) on which systems

identification is to be based will contain very little useful information. This is the reason why field data have to be subjected to preliminary analysis to determine their signals content.

5.2.5 STOCHASTIC PROCESSES

If the state of the system is represented by the random n -dimensional vector $\underline{X}$ and the available measurements by the matrix Z , $Z = (Z_1, Z_2, Z_3, \dots, Z_n)$, then to accomplish the objective of obtaining information about the state from the measurements we need to obtain the conditional probability density or distribution defined as:

$$f(\underline{X}/Z) \quad \text{or} \quad f(\underline{X}/Z_1, Z_2, Z_3, \dots, Z_n) \quad \dots 5.1$$

and;

$$F(\underline{X}/Z) \quad \text{or} \quad F(\underline{X}/Z_1, Z_2, Z_3, \dots, Z_n) \quad \dots 5.2$$

respectively.

These represent the probability density and distribution of the state conditioned on knowledge of all the available sets of measurements. If $f(\underline{X}, Z)$ represents the joint density function of $\underline{X}$ and Z , then through Bayes rule:

$$f(\underline{X}/Z) = f(\underline{X}, Z) / f(Z) \quad \dots 5.3$$

Derivation of this conditional distribution requires a priori knowledge of the population statistics of the various measured variables and that for the joint distribution of the states and the measurements. This knowledge however is generally not available.

For most practical systems, where the number of measurements

taken is quite large, the distribution of the mean or expected value can be approximated by a Gaussian distribution (The Central Limit Theorem). The general random $n \times p$ matrix X , is said to be Gaussian (normal), if it can be described through a probability function of the form:

$$f_X(\underline{y}) = (1/(2\pi))^{1/2} |P| \cdot \exp(-1/2 \cdot (\underline{y} - \underline{m})^T \cdot P^{-1} \cdot (\underline{y} - \underline{m})) \quad \dots 5.4$$

where; P is a positive definite ($n \times n$) matrix; $| \cdot |$ denotes determinant of a matrix; $\underline{m}$ is the vector of the means; and the superscript T refers to the transpose of a matrix or vector.

Equation 5.4 shows that the density function is completely defined by two parameters, the mean ($\underline{m}$) and the variance-covariance matrix (P) respectively. This can be compared with the need to evaluate all the moments for processes which cannot be represented explicitly by the known probability distribution functions.

The Gaussian distribution approximation gives an indication of the central tendency and the spread about this central tendency. If the joint probability density function of the states and measurements and the density function for the measurements can be represented by a Gaussian distribution, then the conditional density of the states conditioned on the measurements is also Gaussian (Maybeck, 1979).

5.2.6 GAUSS/MARKOV PROCESSES

Most stochastic processes have been observed to have the Markov property. This is conceptually analagous to the ability to describe a system state for a purely deterministic system. Under this assumption, the value assumed by the state vector at any time instant provides as much information about future system states as do the values of the states at all previous time instants. This in effect means that the state at any time instant embodies all information needed for propagation and the past history leading to that time instant is of no consequence.

This assumption enables identification excercises to be based on data collected at later times in the history of the system or data that does not explicitly include information on the system being forced from some steady-state or initial condition.

5.2.7 STATE-SPACE REPRESENTATION

Systems representations attempting to account for systems property variations in all the four principal coordinates in the time-space continuum, result in unnecessarily complicated mathematical representations which cannot be evaluated using the type of data currently available. Spatial aggregation leads on to the description of variations in system properties with time only.

The state-space representation for the typical internally descriptive model based on spatial aggregation and deductive reasoning, incorporating uncertainties in system representation and measurements for a continous time system with discrete measurements

is given by:

SYSTEMS EQUATION

$$dx_i(t) = f(U_i, X_i, \theta_i, t).dt + d\hat{\theta}_i(t) \quad \dots 5.5$$

for $i = 1, 2, \dots, n$

$$U_i \subset U$$

$$X_i \subset X$$

$$\theta_i \subset \theta$$

$$x_i \in X_i$$

MEASUREMENT EQUATION

$$z_i(t_k) = \underline{h}_i(t_k)^T \cdot \underline{X}_i(t_k) + n_i(t_k) \quad \dots 5.6$$

Where; $z_i(t_k)$ is the measurement of the i^{th} state ($x_i(\cdot)$) at time t_k ; $\underline{h}_i(t_k)$ is a measurement vector; $n_i(t_k)$ is a Gaussian white-noise process with variance, R_i ; and $d\hat{\theta}_i(t)$ is a Brownian motion process with diffusion Q_i . The other terms are as defined in Appendix A1.

A discrete representation for the measurement equation was chosen because measurements on system output variables are usually taken at discrete time intervals.

5.2.8 STATE ESTIMATION FOR STATE-SPACE MODELS

5.2.8.1 GENERAL

The testing of hypotheses on mechanisms for internally descriptive models usually involves the estimation of the system

states from which the response of the model to known inputs can be compared with the systems response. The validity of the hypothesis is then measured by the extent of agreement between real system data and model generated data based on some criteria for assessment.

There is no real difference between estimation of parameters and estimation of the states of the system. Constant parameters in the assumed model structure for example can be defined as time invariant unmeasured states of the system.

The stochastic differential equation represented by Eq. 5.5 can be integrated to derive an estimate for the state $x_i(t)$ of the system at any time (t) given information on the systems forcing functions (U_i) , the parameters (θ_i) and the initial state $X_i(t=0)$. The Brownian diffusion becomes a Gaussian white-noise process with variance Q_i .

Optimal state estimates can be derived based on measurement information and on some prespecified objective function. For pond system models based on the assumption that the hydraulic regime can be characterised by a CSTR configuration, most of the observed outputs correspond to the states of the system. The parameters are however usually unknown. State estimation in this context therefore usually focuses on the estimation of the unmeasured states (mostly the parameters).

Parameters, which are often assumed to be time invariant states, can be estimated using either "One-shot/Block" estimation procedures or "Recursive" estimation procedures. One-shot procedures result in a single-valued estimate for the parameter vector of interest and as

is shown later, gives no indication of the appropriateness of the model structure or the validity of the time invariance assumption for the parameters. Prior to actually assessing simulation results based on the estimated parameter values, the only information that can be derived with regards the suitability of the assumed model structure is for cases where unrealistic parameter estimates are obtained.

Recursive estimation procedures on the other hand process the measurement data serially, such that with each new observation, the states being estimated are updated optimally based on information gained from the new measurement. The resulting time profile or trajectory of parameter estimates gives information on the suitability of the assumed model structure. For an unbiased consistent and minimum variance estimator, the trajectory of truly time invariant parameters will show very little variation about the optimal parameter value (after successive iterations). Time varying parameters or structural deficiency in the model is often depicted by a non-stationary parameter trajectory.

The non-stationarity in the parameter trajectory results from the estimation algorithm adjusting the parameter estimates in such a way as to compensate for structural deficiencies (or time variation). The resulting profile often sheds light on the missing structural components in the model. Recursive estimation procedures are however severely limited for non-linear state estimation.

The most common objective function used for state estimation is the simple least squares criterion (SLS). For the measured states, this is represented by:

$$\text{Min}_{\theta_i} J_i = \sum_{k=1}^m (z_i(t_k) - x_i(t_k))^2 \quad \dots 5.7$$

for $i = 1, 2, 3, \dots, n$

$$x_i(t_k) = x_i(t_{k-1}) + \int_{t_{k-1}}^{t_k} f(u_i, x_i, \theta_i, t) dt + E(\epsilon_i(t_k)) \quad \dots 5.8$$

If $\epsilon_i(.)$ is assumed to be a white-Gaussian noise process, then $E(\epsilon_i(t_k)) = 0$ for all k . The SLS objective function corresponds to obtaining state estimates which minimize the root mean squared error, R_{mse} (see Appendix A2). The SLS criterion however has limitations in that the stochastic properties of the measurement errors are not taken into account in a realistic manner.

Other objective functions based on the Maximum Likelihood Theory have been developed. These however essentially reduce to the SLS criterion when the errors are assumed to be homogenous and uncorrelated. The choice of optimality criteria and objective functions are discussed by Johnston and Pilgrim (1976).

5.2.8.2 BLOCK ESTIMATORS

The commonly used block estimation procedures are basically of two types; the "iterative search" and the "gradient" techniques respectively. The block estimation procedures are generally robust

and for the given objective function, produce a good single-value estimate for the parameters provided the specified model structure is representative. Single-valued estimates however present no information on possible variations accounting for inadequate model representations for situations where the estimated parameters together with the specified model structure fails to produce an adequate description of the observed process.

In general, the iterative search techniques have been found to be superior to gradient techniques. This is probably due to the fact that the explicit partial derivatives of the objective function with respect to the parameters required for the gradient methods, are often very difficult to obtain. The iterative search techniques have however been found to be inefficient for parameter estimation where:

- a) There exists interdependence between model parameters
- b) The objective function is insensitive to parameter values over a wide range; eg. inactive range for threshold type mechanisms.
- c) Discontinuities exist in the response surface
- d) There are local optima due to non-convexity of the response surface.

Generally, their effectiveness is highly dependent on the nature of the response surface generated by the objective function being used.

For parameter estimation using the Block estimation techniques, the Rosenbrock (1960) iterative search procedure was adopted.

5.2.8.3 RECURSIVE ESTIMATORS

Recursive estimation algorithms are often formulated so that the model output is continuously assessed against observations of the system response. The residuals (innovation sequence) are used as feedback information, which is modified by a gain factor, to adjust the estimates of the states. Most are of the form (see Appendix B):

$$\underline{X}(t_k/t_k) = \underline{X}(t_k/t_{k-1}) + K(t_k) \cdot (\underline{Z}(t_k) - \underline{X}(t_k/t_{k-1})) \quad \dots 5.9$$

Where; $\underline{X}(t_k/t_k)$ is the updated estimate of the state at time t_k ; $\underline{X}(t_k/t_{k-1})$ is the prediction of the state based on measurement information at time t_{k-1} ; $K(t_k)$ is a matrix of gains (which depends on the estimation scheme adopted); and $\underline{Z}(t_k)$ is the measurement information available at time t_k .

Basically, what is required of a recursive estimator in parameter estimation is that, while still allowing for time variation in the parameters of interest, it is able to filter out the deleterious effects of noise contamination.

There are several approaches in the literature on recursive estimation techniques and these include: Recursive least-squares estimation (RLS); Kalman filtering (KF); the Extended Kalman filter (EKF); and the Instrumental variable approach (IV). They differ in the amount of a priori information required on the properties of the stochastic processes. Eykhoff (1974) gives a good description of the features, characteristics and a priori information required for the implementation of each kind. A summary description of the recursive estimation procedures used in this work is presented in

Appendix B.

The RLS procedure is the simplest and requires the least a priori information. It is robust but however, gives rise to biased estimates for structured models or for data which contains serially correlated measurement errors. The IV approach is actually a variant of the RLS procedure. Theoretically, it overcomes the shortcomings of RLS for the structured model case, by means of a set of instrumental variables generated through an auxiliary model (Young et al., 1980). The IV approach however is only applicable to state estimation for state-space models with both discrete time systems and measurement equations, linear in the states being estimated (discrete-discrete).

The Kalman filter (KF) can be applied to the various types of linear state-space models, namely; the continuous-continuous, the continuous-discrete, the discrete-continuous and the discrete-discrete. Where the statistics of the stochastic processes are correctly specified, the KF produces the minimum variance, consistent and unbiased estimate for the states. In practice, the noise topology for ecological and other large-scale systems is not known.

Because of this lack of information on noise statistics, states estimated using the KF are not "Globally Optimum". They are however optimal for the assumed set of conditions. On account of this, state estimation for ecological systems, using the KF is sensitive to the assumed properties for the stochastic processes. For linear discrete-discrete systems, the IV is a more robust estimator

(compared with the KF) because it requires less a priori information.

For systems non-linear in the states, extensions to linear systems theory are employed, eg. the Extended Kalman Filter (EKF). The EKF is based on the linearisation of systems and measurement equations by a first-order Taylor-series expansion. System dynamics are represented in terms of perturbations about a nominal reference trajectory. Higher orders in the Taylor series expansion are neglected, based on the assumption that the perturbations are small. This results in a linear perturbation dynamics.

The assumption about the perturbations being small is valid as long as the true state of the system does not deviate far from the reference trajectory. If the perturbations are large however, this leads to filter divergence. To reduce the likelihood of filter divergence, the EKF is usually implemented in an adaptive fashion. Prior to propagation to the next time step, the observed states are reset to their measured values. This is conceptually analogous to assuming that the measured system states have been observed perfectly.

The Fortran source codes for the recursive estimation algorithms used in this work can be found on the microfiche in Appendix B.

THE EFFECTS OF NOISE STATISTICS ON FILTER PERFORMANCE:

Since the correct coefficients of the systems and measurement noise, Q and R respectively are not known, suitable values have to be found through trial and error (subjectively). This is known as "tuning" the filter. Insight into how these "tuning devices" affect

the filter behaviour can result in the judicious choice of suitable values.

Where the recursive estimation procedures are used for parameter estimation, they can be compared to tracking devices. Taking the general case of a systems equation non-linear in the parameters and in which the parameters are assumed to vary in a random walk fashion, the systems and measurement equations can be represented by:

SYSTEMS EQUATION

$$\underline{X}(t_k) = \underline{X}(t_{k-1}) + \underline{\epsilon}(t_k) \quad \dots 5.10$$

MEASUREMENT EQUATION

$$\underline{Z}(t) = \underline{h}(\underline{U}_i, \underline{X}_i, t) + \underline{v}(t) \quad \dots 5.11$$

Where $\underline{H}$ in the EKF algorithms, is given by:

$$\underline{H}(t_k) = \left. \frac{d\underline{h}(\underline{U}_i, \underline{X}_i, t)}{d\underline{X}} \right|_{\underline{X}(t_{k-1})} \quad \dots 5.12$$

Here $\underline{h}(\cdot)$; represents a general non-linear vector-valued function relating the measurements to the system inputs and states; $\underline{\epsilon}(t_k)$ represents the stochastic process with a variance-covariance matrix, $\underline{Q}(t_k)$, accounting for parameter variation; and $\underline{v}(t)$ represents a stochastic process with variance-covariance matrix $\underline{R}(t)$ accounting for systems and measurement noises.

The H matrix is made up of influence coefficients which actually represent the changes in systems output for unit changes in the states being estimated - the parameters. This introduces a non-linear action into the estimator and works mainly by the fact that large values of H increase the estimator gain K . The gain is in this way adapted to the "signal level" of the measurements.

It is clear from the algorithms that Q affects the variance-covariance matrix of the estimates, P , directly. It keeps $P(t_k/t_{k-1})$ from becoming infinitely small. This has the secondary effect that the gain is prevented from going to zero which would result in no updating and ceasure of tracking abilities. Because Q affects P directly, this means that the accuracy of the estimation procedure is a matter of choice to some extent.

If Q is large, the estimator becomes erratic and P indicating this becomes large. When Q is decreased, the variation in the estimated state decreases and so does P . The value of P will be a reasonable measure of estimation accuracy only if Q which represents the rate of change of the state is greater than that for the true state, if the state being estimated is a true system state. If Q is much less than the true rate of change, this results in biased estimates since the estimated value of the state always lags behind the true value.

R can be interpreted as the variance of the residual sequence when the true state is known. The power (noise level) of the high-frequency components of the process not accounting significantly for the total variation can give a clue as to likely

values for R .

Because the true values of the parameters are not known, a high value of $P(t=0)$ is specified to reflect the high degree of uncertainty in the initial value specified.

As the filter processes the data serially, the variance P associated with the estimates decreases and so does the gain. If the rate of change of the state specified (ie. Q) is less than the true rate of change of the state, successive passes through the entire measurement history using the last state estimate as the initial estimate for each successive iteration produces a better estimate than a single pass through the data.

Extremely noisy data often results in filter divergence when the EKF is used. This is mostly because of the violation of the small perturbations assumption. Also, high noise levels on data with a low signal-to-noise ratio can result in very little information gain from the measurement sequence. To overcome this, pre-filtering of noise, and other signals enhancement techniques are often applied to noisy data.

Preliminary work on parameter estimation using synthetic data generated by a pre-specified model structure showed that, both the IV and the Kf were equally good at estimating parameters for cases where the error statistics were specified correctly. They were also good at indicating the nature of the variation in time varying parameters. In the cases where it was assumed that there was no prior information on the error statistics, the IV proved to be more robust and better at indicating parameter variations.

A discussion on the effects of the filter statistics on the estimated state values is presented in later sections where the recursive estimators are used for estimation of model parameters.

5.2.8.4 ASSESSMENT CRITERIA

The criteria outlined in Appendix A2 for model assessment formed the basis for assessing the structural representativeness of the model formats adopted for the identification exercise. With regards to systems properties derived from recursive estimation procedures, these in particular centred around the simultaneous satisfaction of the following criteria by Beck and Young (1976):

1) The recursive estimates of assumed time-invariant parameters are themselves indicative of time-invariance and the estimates of any assumed variable parameters have a direct physical interpretation.

2) The estimated stochastic disturbances ie. for both the process and measurements are purely stochastic in form and have no systematic components attributable to some physical aspects of the system behaviour.

What the second assumption means in effect, is that the conditional density of the residuals (errors) is functionally independent of the entire measurement random variable. This independence says conceptually that **the estimator gleans as much** information from the measurements as possible, and there is nothing left in the measurements that could divulge any information about

the errors. The information retrieval exercise is then optimal in the sense that there are no projections from the measurements into the errors directional modes.

With such an approach, it is necessary to make assumptions about the statistical properties of the stochastic disturbances. This is not an easy task in general terms since the characteristics of the noise influencing the unidentified system is virtually unknown and that for the measurements usually poorly defined if at all.

In the absence of information on noise characteristics, they are assumed to be wide band noise which can be represented by Gaussian white-noise statistics. This in effect means the noise terms have equal power at all frequencies and are not serially correlated. Statistical analysis of the residuals will thus be in terms of verifying these assumptions about the noise statistics.

5.3 HYPOTHESIS EVALUATION SCHEME

5.3.1 BASE MODEL: HYPOTHETICAL COMPLETE DESCRIPTION

The input-state-output description for ponds represented diagrammatically in Fig. 5.1 was based on a priori information from the pond process description presented in Chapter 2. On the basis of this information, a base model which comprises of a complete description of pond behaviour as supported by the current information on ponds can be derived. This will no doubt comprise of a large number of descriptors for the several component processes and interactions, and would be along the lines of conventional internally descriptive modelling. This model will however not be

verifiable with the available pond data.

The section above (5.2.1) on the conventional approach to internal descriptive modelling discussed the difficulties in model structure verification and validation for the complex models given the type of measurement information available. Because of these factors, the systems identification exercise was taken as an information retrieval exercise placing as much confidence on the a priori information in the literature as that gained from posteriori analysis of models using the data acquired.

This centred around a hypothesis evaluation scheme using exploratory models describing aggregate effect pond behaviour. This can be compared to a hypothesis evaluation scheme on base model simplifications.

5.3.2 EXPERIMENTAL FRAMEWORK (SIMPLIFICATIONS)

As was realised from the process description presented in Chapter 2, pond processes are highly interactive and influenced considerably by environmental factors. However, to proceed with the adopted modelling approach, it was assumed that the following notions apply to the system under study, namely: Separability, Selectivity; and Causality.

Separability relates to the assumption that some interactions between the systems components can be ignored and that processes can be studied in isolation.

Selectivity deals with the condition that of all the possible interactions, only a relatively small subset are relevant to the

identification exercise.

Finally, causality relates to the information content of the measurement data, viz. that the acquired data contains information on system dynamics resulting from the system inputs exciting the dominant modes of process behaviour and that this information is present and reflected in the acquired data.

If a cause/effect relationship exists between an input and output variable through some state variable, then if that state variable is not sufficiently excited such that changes in the state variable are reflected in the measurement sequence, the cause effect relationship in question will not be identifiable from the data.

The above conditions are idealistic but are nonetheless necessary for the simplification of the dominant process modes identification. It is known that all system processes interact to an extent and are influenced by system environmental factors. The causality condition for example suggests the notion of some directly observable input output relationship. In fact systems with complex feedback loops may actually respond to an excitation or control signal in a manner that is diametrically opposed to that expected.

The hypothesis evaluation scheme adopted involves grouping component processes and aggregating variables; resulting in lumped model simplifications. The simplifications are regarded as working models or hypotheses about the base model - namely that the omitted structural components are relatively insignificant in explaining the systems behaviour.

This hypothesis is confirmed if in relation to the pre-defined assessment criteria, the model proves to be valid. Where this hypothesis is rejected, the next step involves incorporating base model entities which are believed to be important in describing the observed system behaviour.

It would seem that this approach adopted is appropriate in this study. Workers have found that a basic assumption that can be invoked in the analysis of environmental systems which generally fall into the class of complex systems is that these complex processes can be effectively modelled by linear approximations (Gentil, 1984). It seems that a "Law of Large Systems" applies where the overall response characteristics are often very simple in form and capable of being described by simple linear models (Joergensen et al., 1982).

5.4 THE IDENTIFICATION EXERCISE

5.4.1 GENERAL

The overall pond system was divided into two coupled sub-systems namely; the planktonic and benthic sub-systems. Since no measurements pertaining to the benthic sub-region was available in the acquired data sets, the identification exercise concentrated on the planktonic region with the benthos taken to be a source and sink of materials.

The overall system boundaries were taken to be the walls, floor and water surface of the pond system. The boundaries for the coupled subsystems are shown in Fig 5.2 which also gives the likely sources

of loading to these two subsystems.

Studies on the hydraulic characteristics of ponds (Ferrara and Harleman, 1981; Iwugo and Winnicki (1976) and Morenzo-Grav et al., 1984) have shown that the typical unbaffled pond can be represented by a CSTR configuration. In this study, the hydraulic regime in the planktonic region was represented by a CSTR of volume equivalent to the nominal volume of the pond system.

The assumption of a CSTR configuration for the hydraulic regime is no doubt an idealisation of the actual hydraulics of the pond system. The true hydraulic regime will have non-constant time delay elements due to dead space especially when stratification occurs.

5.4.2 CHOICE OF MODELLED OUTPUT VARIABLES

In any identification exercise, there is the need to define the system state(s). Ideally, a systems state vector comprising of a fully determinate set would characteristically embody all the information necessary for the prediction of the evolution of the system given information on changes in the influencing (or control factors).

The above corresponds to defining the Hamiltonian of an object in classical dynamics (viz. coordinates and momentum). In this regard, the history of the system including its initial conditions are of no consequence.

Since pond systems are not fully defined or identified, a Hamiltonian cannot be defined for systems dynamics. The systems state(s) chosen for the identification exercise were therefore

determined by data limitations and characteristics of the systems output which are important with regards to environmental pollution.

The variables modelled were those that relate to the immediate and delayed depletion of the oxygen resources of the disposal media, those relating to nutrients enrichment, and those specifically related to toxicity and the creation of adverse environmental conditions for higher trophic organisms or in general upsetting the ecological balance of the disposal media.

The classification under which each of the measured variables for the two pond data fall, is shown in Table 5.1. The variables representative of the organic pollutional load are sub-classified into those that relate to the soluble portion of the waste stream and those that relate to the suspended portion. The soluble portion effects a relatively immediate demand on the oxygen resources whereas the effect of the suspended portion are gradual and delayed (Harremoes, 1982). Since these two organic waste types have different impacts on disposal media, they were modelled as two separate output variables.

The inorganic nutrients of interest are mainly the nitrogen and phosphorus compounds. Though inorganic phosphorus was measured as a separate variable for Pond 1 data, it was not measured separately for Pond 2 data and was rather incorporated in a total phosphorus measure.

The measures of organic forms of nitrogen and phosphorus available for the two pond data are not of interest directly as output variables, their dynamics were however modelled with the view

that information on the dynamics of the possible coupling between inorganics and organics in the waste stream would emerge.

The preliminary analysis of the data , particularly the principal component and factor analysis, showed that some measured variables eg. total COD and total BOD had similar information content. There was therefore no need to model them separately. Where two or more measures representative of the output characteristic of interest were present in the data, the measure with a stronger signals content, as indicated by the preliminary analysis of the data, was selected.

In order to investigate the observed macro-transformations of the influent constituents into their respective constituents in the effluent, it was assumed that the interactions between the system states under the observed conditions (routine operation) were subtle and for the purpose of investigating aggregate effects, could be neglected. The modelled states were therefore processed as parallel streams with interactions between the states, lumped together with other sources and sinks.

The first hypothesis on pond dynamics investigated was that; pond systems behave essentially like big tanks with no reactive processes and that the observed transformations were due to dilution and equalization of flow.

5.4.3 ZERO RATES MODEL

5.4.3.1 MODEL FORMULATION

Based on the assumption that the pond system is a physical reactor with no bio-chemical transformation processes occurring, a material balance for the modelled states results in the following representation for the planktonic region :

$$dx_i(t)/dt = U_i(t) - x_i(t)/T(t) \quad \dots 5.13a$$

for $i = 1, 2, \dots, n$

$$U_i(t) = \frac{Q_{in}(t)}{V} \cdot C_i(t) \quad \dots 5.13b$$

$$T(t) = V/Q_{ef}(t) \quad \dots 5.13c$$

Where; $Q_{in}(t)$ is the influent flow rate ($m^3 d^{-1}$); $C_i(t)$ is the concentration in the influent stream of the substance represented by the i^{th} state variable, $x_i(t)$, (mg/l); V is the volume of the planktonic region, (m^3); $Q_{ef}(t)$ is the effluent flow rate ($m^3 d^{-1}$). The subscript i refers to the i^{th} state variable.

Equation 5.13 is a set of independent linear ordinary differential equations which can be integrated either explicitly or numerically to obtain a time profile of the measured states being modelled. The numerical integration scheme outlined in Appendix A1 was used to obtain the model's prediction of the states on a daily basis with the initial state set to the first day value of the corresponding output variable in the data sets.

The model was forced with the raw daily measured values for the

input variables and the weekly average model output was compared with the weekly average measured output data.

The results of the evaluation of the zero-rates hypothesis for pond dynamics are presented below for the two pond data and are discussed under the sub-headings carbonaceous organics and nutrients. The modelling results for the organic phosphorus for Pond 1 and the total phosphorus and nitrogen for Pond 2 are discussed under the sub-heading, *nutrients* although strictly speaking they are not nutrients in the sense as the term is commonly used.

5.4.3.2 POND 1 SIMULATION RESULTS

The resulting simulations and errors plots for Pond 1 data are shown in Figs. 5.3 - 5.12 and 5.14-5.15. These plots generally show that there are reactive processes accounting for removals of both the carbonaceous organics and the inorganic nutrients within the pond system.

CARBONACEOUS ORGANICS

The simulations and errors plots for both filtered COD and the volatile suspended solids (Figs. 5.3-5.6), produced negative R_{sqd} values and high negative mean errors comparable in magnitude to their observed concentrations in the pond effluent stream. It is obvious from these plots that with regards the carbonaceous organics quality, the non-reactive tank model is not representative of the observed pond behaviour.

NUTRIENTS

The plots for the nutrients series also clearly show that the zero-rate model format is not representative of the observed pond behaviour. The simulation plots for ammonia (Fig. 5.7) and inorganic phosphorus (Fig. 5.9) have similar trends and their errors plots (Figs. 5.8 and 5.10) show that these errors have a similar structure. This suggests that a similar dominant mechanism must have accounted for the observed inorganic nutrients removals.

The nitrate plots (Figs. 5.11 and 5.12) indicate that pond interactions over the simulation period resulted in the net production of nitrate for part of the year. This is because for most of the first half of the simulation period, the model predicts a lower concentration of effluent nitrate than that observed.

The literature on ponds suggests that because of the unavailability of surfaces for attachment of nitrifying organisms, nitrification does not occur in ponds. The simulation results for nitrate however show that nitrate is produced within the system. In fact the plot of influent and effluent nitrite for the same pond data (see Figs. 3.112 and 5.13) shows that effluent nitrite concentrations are relatively higher than the influent concentrations. The above observations no doubt suggest that nitrification most probably occurred in the pond over the monitoring period.

The plots for organic phosphorus (Figs. 5.14 and 5.15) indicate that there was a net removal of organic phosphorus over the 285 days period during which pond nutrients were monitored. The errors

plot for this state variable shows a near constant removal over the simulation period.

5.4.3.3 POND 2 SIMULATION RESULTS

The results of a similar exercise for Pond 2 data are shown in Figs 5.16 - 5.27. These plots generally show that the zero-rates model is an inadequate model representation of the pond system.

CARBONACEOUS ORGANICS

Though the simulation and errors plots for the soluble COD (Fig. 5.16 and 5.17) show that essentially, there were net removals throughout the simulation period, there is some indication that the observed removals were less in the hotter months than in the colder months. Those for the total solids (Fig. 5.18 and 5.19) clearly show that there was a net production of solids in the hotter months of the year.

NUTRIENTS

The plots for all the modelled states representative of the phosphorus and nitrogen compounds (Figs. 5.20-5.27) show that there were net removals throughout the entire simulation period. Though their computed effluent concentrations are higher than the observed, their profiles have similar trends.

model are determined by trends in the influent (control) signals. The observed similarity in the modelled and observed profiles suggests that the systems states corresponding to the nitrogen and

phosphorus compounds were driven to an extent by their respective input components in the influent stream.

The errors plots for both total phosphorus (Fig. 5.25) and total nitrogen (Fig. 5.27) have similar trends which suggests that similar dominant mechanisms probably account for their observed removals.

5.4.3.4 SUMMARY AND CONCLUSION

The plots for the zero-rates model simulations for both data sets clearly showed that transformation processes occurred within the pond systems during their monitoring periods. This clearly suggested that the pond system could not be considered as a large tank with no reactive processes. The similarities in trends in the output profiles however, suggest that the system states were excited by their respective input components and that meaningful information on system dynamics was present in the data.

5.4.4 GROSS REACTIVE TERM MODELS

5.4.4.1 MODEL FORMULATION

The evaluation of the zero-rates hypothesis in Section 5.4.3 showed that there were reactive processes in the pond system during the period over which they were monitored. "Gross Reactive Term" models were postulated as being representative of the observed pond behaviour. These models were based on the hypothesis that the net effect of all reactive processes occurring in the pond system, could be represented by either a constant source/sink rate or by a simple first order autocatalytic process.

In terms of system dynamics, the source/sink term model format assumes that the overall pond interactions for the various modelled states can be represented by a zero order rate process. The simple first order decay format on the other hand assumes that the rate at which a substance is removed from the planktonic region of the pond system is proportional to the concentration of that substance in the pond basin. This format, is similar to Marais's (1970) first model and is the most common kinetic formulation assumed to be representative of pond systems.

The resulting deterministic model representation for the systems model based on a mass balance for the planktonic region and on the above assumptions on process dynamics are given below:

SOURCE/SINK MODEL:

$$dx_i(t)/dt = U_i(t) - x_i(t)/T(t) - \theta_i \quad \dots 5.14$$

for $i = 1, 2, \dots, n$

FIRST ORDER DECAY MODEL:

$$dx_i(t)/dt = U_i(t) - x_i(t)/T(t) - \theta'_i \cdot x_i \quad \dots 5.15$$

for $i = 1, 2, \dots, n$

Where $U_i(t)$, $T(t)$, and $x_i(t)$ are as defined above for Equation 5.13; θ_i represent the constant sink rate and; θ'_i , the first order

decay rate for the i^{th} state variable.

If the initial system states are set to their respective first day measured values for their corresponding output variables, then the only unknowns in Equations 5.14 and 5.15 are the parameters θ_i and θ'_i . The above hypothesis on pond dynamics would be confirmed if values for the unknown parameters could be found such the conditions set out in the assessment criteria were satisfied.

Assuming that the forcing function $U_i(t)$ does not vary significantly between unit time steps/intervals results in the following explicit solutions for the continuous time Equations (5.14 and 5.15) in discrete form giving the one-step ahead predictions for the observed states:

SOURCE/SINK MODEL:

$$x_i(t_{k+1}) = x_i(t_k) \cdot A_i(t_k) + C_i(t_k) \quad \dots 5.16a$$

for $i = 1, 2, \dots, n$

$$A_i(t_k) = e^{-1/T(t_k)} \quad \dots 5.16b$$

$$C_i(t_k) = (U_i(t_k) + \theta_i) \cdot T(t_k) \cdot (1 - e^{-1/T(t_k)}) \quad \dots 5.16c$$

FIRST ORDER DECAY MODEL:

$$x_i(t_{k+1}) = x_i(t_k) \cdot B_i(t_k) + D_i(t_k) \quad \dots 5.17a$$

for $i = 1, 2, \dots, n$

$$B_i(t_k) = e^{-(\theta'_i + 1/T(t_k))} \quad \dots 5.17b$$

$$D_i(t_k) = U_i(t_k) \cdot (\theta'_i \cdot T(t_k) + 1)^{-1} \cdot T(t_k) \cdot (1 - B_i(t_k)) \quad \dots 5.17c$$

Where all the variables are as previously defined:

The above Equations (5.14 and 5.15) are linear in the measured states $x_i(t_k)$ and the discrete form for the s/sink model (5.16) is also linear in the parameter θ'_i . The first order decay model (Eq. 5.17) is however nonlinear in the first order decay constant θ'_i .

Based on the assumed CSTR configuration for ponds, the modelled states correspond to the measured outputs. Defining the parameters by a new state variable $X'_i(t_k)$ results in:

$$x_i(t_k) = h_i(t_k) \quad \dots 5.18$$

$$x'_i(t_k) = \theta'_i(t_k) \quad \text{or} \quad x'_i(t_k) = \theta'_i(t_k) \quad \dots 5.19$$

Where; $h_i(t_k)$ represents the measured value for the i^{th} state variable at the k^{th} sampling instant; and $x'_i(t_k)$, the new state variable representation for the parameters.

If the new state variable (parameter), $x'_i(t_k)$, is assumed to vary in a random walk fashion, then the following discrete-discrete state-space representation can be derived:

SOURCE/SINK MODEL:

$$x'_i(t_k) = x'_i(t_{k-1}) + \epsilon_i(t_k) \quad \dots 5.20a$$

$$h_i(t_k) = h_i(t_{k-1}) \cdot A_i(t_k) + C_i(t_k) + \mu_i(t_k) \quad \dots 5.20b$$

FIRST ORDER DECAY MODEL:

$$x'_i(t_k) = x'_i(t_{k-1}) + \epsilon'_i(t_k) \quad \dots 5.21a$$

$$h_i(t_k) = h_i(t_{k-1}) \cdot B_i(t_k) + D_i(t_k) + \psi'_i(t_k) \quad \dots 5.21b$$

Where; $\epsilon_i(t_k)$ and $\epsilon'_i(t_k)$ represent stochastic terms accounting for parameter variations; $\psi_i(t_k)$ and $\mu'_i(t_k)$ represent stochastic terms accounting for systems and measurement noises respectively; and $A_i(t_k), B_i(t_k), C_i(t_k), D_i(t_k)$ are as defined above for Eqs. 5.16 and 5.17 but with the parameters θ_i and θ'_i replaced by $x'_i(t_k)$.

If the term $C_i(t_k)$, involving the input $U_i(t_k)$ and the parameter, $x'_i(t_k)$, is assumed to be an unknown constant term, then Equation 5.20 can be set for an IV estimation scheme involving the estimation of $C_i(t_k)$ from which $x'_i(t_k)$ can be determined since the rest of the terms are known.

If $\epsilon_i(t_k)$ and $\psi_i(t_k)$ are assumed to be Gaussian white-noise, realisations from random processes, with the following statistics:

$$\left. \begin{aligned} E(\epsilon_i(t_k)) &= 0 & E(\epsilon_i(t_k) \cdot \epsilon_i(t_k)) &= Q_i \\ E(\psi_i(t_k)) &= 0 & E(\mu_i(t_k) \cdot \psi_i(t_k)) &= R_i \\ \text{and } E(\psi_i(t_k) \cdot \epsilon_i(t_k)) &= 0 \end{aligned} \right\} \dots 5.22$$

Where; $E(.)$ is the expectation operator; Eq. 5.20 can be set up for

a LKF estimation scheme with H given by:

$$H_i(t_k) = T(t_k) \cdot (1 - e^{-1/T(t_k)}) \quad \dots 5.23$$

The IV scheme for evaluating the source/sink rate constant requires the specification of Q_i whereas the LKF scheme requires the specification of both Q_i and R_i . Because the statistical properties of these stochastic processes are not known, the scheme requiring the least prior knowledge (the IV) was chosen for evaluating the source/sink rate constant.

Since the discrete-discrete representation (Eq. 5.21) for the first order decay model (Eq. 5.15) is non linear in the parameter to be estimated, a non-linear estimation procedure had to be used for estimating the parameters. An EKF was used for the recursive estimation of this parameter.

If $\epsilon'_i(t_k)$ and $\mu'_i(t_k)$ are assumed to be Gaussian white-noise, realisations from random processes, with the following statistics:

$$\left. \begin{aligned} E(\epsilon'_i(t_k)) &= 0 \quad ; \quad E(\epsilon'_i(t_k) \cdot \epsilon'_i(t_k)) = Q'_i \\ E(\mu'_i(t_k)) &= 0 \quad ; \quad E(\mu'_i(t_k) \cdot \mu'_i(t_k)) = R'_i \end{aligned} \right\} \dots 5.24$$

and $E(\mu'_i(t_k) \cdot \epsilon'_i(t_k)) = 0$

Where; $E(.)$ is the expectation operator; Eq. 5.21 can be set up for an EKF estimation scheme with H given by:

$$H_i(t_k) = -h_i(t_k) \cdot e^{-(B_i(t_k))} + D'_i(t_k) \quad \dots 5.25a$$

for $i = 1, 2, 3, \dots, n$

$$D'_i(t_k) = U_i(t_k) / (B_i(t_k))^2 \cdot (e^{-(B_i(t_k))} \cdot (B_i(t_k) + 1) - 1)$$

.. 5.25b

Where all the terms in Eq. 5.25 are as defined for Eq. 5.17.

The sink and first order decay rate constants were also evaluated using the Rosenbrock procedure. A summary of the results obtained for some of the assumed values for the noise statistics are presented in Tables 5.2 to 5.5.

5.4.4.2 GENERAL DISCUSSION ON ESTIMATION RESULTS

The resulting profiles for two assumed values of Q for the sink rate in the modelled states are shown in Figs. 5.28-5.39. The mean for the JV estimates was found to be practically independent of the value of Q_i assumed and actually corresponded to the value obtained if the parameter was assumed to be truly time invariant, ie. $Q_i = 0$.

The mean values were also found to be quite close to the parameter estimates obtained by the Rosenbrock (1960) procedure. The effect of Q_i on the tracking ability of the recursive estimator is clearly seen from the plots of the trajectories for the different assumed values for Q_i for the same modelled state.

As Q_i increases, the trajectory of the estimates becomes more erratic because the filter puts more weight on the information obtained from the residuals. At relatively high Q_i values the filter becomes very sensitive to the noise on the signals and this makes it difficult to determine trends or seasonalities

(non-stationarities) in the resulting profiles. The profiles for very low values of Q_i shows the trends and seasonalities better.

The trends or seasonalities observed at a low Q_i 's could, however, have resulted from the filter estimates lagging behind the true parameter value. On account of this, different values of Q_i ranging from zero to quite high values were used to help determine the "true nature" of the observed trends and seasonalities in the trajectories.

The profiles for Pond 1 nitrate sink rate for example shows how different assumed values for Q_i helped determine the "true nature" of the parameter variations. The profile for a low Q (ie $Q=.0001$; see Fig 5.38) showed that there was an increase in the sink rate around day 250 which gradually decreased towards the end of the observation period. This pattern is indicative of some strong seasonal component and suggests a possible shift in the sink rate. The resulting profile for a much higher Q value ($Q=1$; see Fig 5.32) clearly shows that the observed gradual decrease was due to the filter lagging behind the true parameter value on account of the low Q specified.

The profiles for relatively high Q 's for Pond 1 nitrate (see Fig 5.32) show that the nitrate sink rate was fairly constant throughout the year with an unseasonal high sink rate around day 250 which probably resulted from the high levels of influent nitrate observed at that period (see Fig. 3.19).

The trends and seasonal variations in the profiles of the first order decay rate were found to be quite similar to those obtained

for the sink rate. Sample plots of the resulting parameter trajectories are shown in Figs. 5.40 and 5.41.

The results of the EKF estimation scheme were found to be dependent on the Q_i to R_i ratio. Specification of a high Q_i to R_i ratio resulted in a highly erratic parameter trajectory with trends and seasonalities masked. A low Q_i to R_i ratio on the other hand resulted in a much smoother trajectory from which trends and seasonalities were more evident. It was however apparent from these trajectories that the filter estimates lagged behind the true parameter values. The erratic nature was most invariably due to the low signal-to-noise ratio for the raw data sequence.

Using the enhanced signals (see section 3.2.4) resulted in profiles which depicted the seasonal patterns better. These were found to be quite similar to those obtained for the source/sink rate using the IV.

The results of simulations based on the assumed model structure and the resulting parameter estimates, illustrate how the non-stationarity is indicative of an inadequate model representation. The simulation and errors plots for Pond 1 filtered COD for different assumed Q values are shown in Figs. 5.42-5.47. Comparing the results of simulations based on the mean of the IV estimates (equivalent to $Q=0$) and the resulting parameter estimate from the Rosenbrock procedure (see Figs. 5.42-5.43 and 5.48-5.49) with those for the other assumed Q values, shows that the mean and the Block estimate values tend to compensate for inadequate systems representation by the model.

The mean of the 1V estimates and the Block estimate values represent the average of the observed reactive processes occurring in the pond system. The single value for the parameter obtained using the block estimation procedure however does not give any indication as to the nature of the variations in the reactive processes occurring.

The trajectories for the recursive estimates of both the source/sink rate and the first order decay rate clearly showed that the gross reactive processes that occurred in the pond system were not constant for most of the modelled states.

Since the parameter estimate on the basis of the time invariance assumption compensates for variation in the reactive processes, simulations to assess the validity of the gross reactive term model were based on this parameter value.

The results of simulations for the chosen state variables for both Pond 1 and Pond 2 data based on the gross reactive term models, are discussed below. For each pond data, the results are discussed under the main sub-groupings for the modelled states and observations are linked with the resulting trajectories of the recursive parameter estimates.

The non-stationarities in the parameter trajectories are interpreted in terms of likely systems mechanisms accounting for the observed variations and new working models are developed from results of this posteriori analysis using the base model description as a guide.

5.4.4.3 POND 1 SIMULATION RESULTS

CARBONACEOUS ORGANICS

The simulations and errors plots for the filtered COD and VOLSS (Figs. 5.50-5.57) clearly show that there is some seasonal component missing in the assumed model. The trajectories of the recursive estimates of the gross reactive rate constants (Figs. 5.34 and 5.35) showed that the observed net removals of both the dissolved and the particulate components of the carbonaceous organics was lower in the hotter months than in the colder months. This is illustrated in their respective simulation and errors plots. Removals in the colder months are under-estimated whereas those in the hotter months are over estimated. The autocorrelogram of the errors show that they are highly structured.

Based solely on a consideration of the activity of heterotrophic organisms in the planktonic region of the pond system, the observed seasonal trend is contrary to the expected. If the soluble COD measure is assumed to be truly representative of the soluble organic substrate utilized by these organisms in their metabolic activities, then it would be expected that the uptake rate and hence rate of removal would generally increase with increasing temperature.

It is unlikely that temperature inhibition of the heterotrophic organisms could account for the decreased net removals at higher temperatures since the pond temperatures observed over the pond monitoring periods were not high enough to result in the denaturing of organisms or interference with enzymatic functions.

Figure 5.2 which describes the systems boundaries and likely sources of loading, shows that there is a loading component (FB) to the planktonic region from the benthic subregion of the pond system which in fact is an unmeasured internal loading. The assumed gross reactive rate model does not take this internal loading component into account explicitly but rather, it is accounted for by the gross reactive term.

To obtain estimates for the gross reactive rate constants, the filter adjusts the parameter estimates so as to account for discrepancies between the outputs, controlled by the measured inputs, and the observed outputs. An internal loading component with the characteristic seasonal pattern of a higher loading in the hotter periods than in the colder periods, could possibly account for the observed seasonal patterns in the trajectories of the gross reactive rate constants for both the filtered COD and the VOLSS. There are other possible mechanisms and these are investigated in the posteriori analysis leading on to the development of new working models.

NUTRIENTS

The profiles for the sink term for ortho-phosphorus clearly showed some seasonal pattern (see Figs. 5.30 and 5.36). From around day 85 to day 120, there was an apparent increase in the removal rate which peaked around day 120. Thereafter, it decreased till around day 225 then increased till about day 325 after which there was an apparent decrease until the end of the observation period. A similar pattern was observed for ammonia (see Figs. 5.31

and 5.37) except that the decreases in the removal rates after the peaks were more pronounced and there was an apparent increase in the removal rate around day 215 (ie in the region of the trough).

The ridges in the trajectories for the inorganic nutrients discussed above, indicate some sort of turning point in the gross removal rate. A possible explanation for this effect would be a shift in the dominant algae types with temperature increases beyond some threshold temperature. The above observation could also have resulted from the feedback of inorganic nutrients from the benthos with increased anaerobic activity.

Obviously, because the nutrients series were not observed over the entire annual cycle, it is difficult to place a lot of confidence in the explanations given for the observed non-stationarity in the parameter trajectories.

The simulation and errors plots for ammonia and ortho-phosphorus (Figs. 5.58-5.65) clearly show that the observed effluent concentrations of these nutrients were under estimated in the region of the peaks in the parameter trajectory and over estimated in the region of the trough.

Though significantly high values were obtained for the R_{sqd} for both modelled states, the autocorrelogram of the errors clearly show that these errors are highly structured and indicate that the model does not describe the observed pond inorganic nutrients dynamics.

The trajectories for the gross nitrate removal rates (eg. see Fig. 5.38) showed that there was a net production of nitrate

throughout most of the year with an unseasonal high removal around day 250 which corresponds to the period where significantly higher levels of nitrate were observed in the influent stream. The simulations and errors plots for the gross reactive term models (Figs. 5.66-5.69) clearly reflect this observation. The effluent nitrate concentrations were under-estimated for most of the year.

The simulations and errors plot for organic phosphorus (Figs. 5.70-5.73) suggest that the observed removals of organic phosphorus were constant throughout the simulation period. This is because the model apparently produces an adequate fit to the observed effluent organic phosphorous series. Though the trajectory of the parameter estimates (eg. see Fig. 5.32 and 5.38) would suggest that there was significant variation in the gross organic phosphorus removals, a closer look at the range and the scales of the plot shows that this trajectory is stationary. The mean and the final values are quite close, and the standard deviation of the recursive estimates is very small in relation to the mean value.

Compared with the relatively high R_{sqd} values for the inadequate model representations for the inorganic nutrients, the R_{sqd} value for the organic phosphorous simulation plot suggests that there is still a considerable part of the variations in the effluent organic phosphorus series unaccounted for. However, if note is taken of the fact that in the preliminary analysis of the data, the organic phosphorus series was found to have very weak signals and hence limited information content, then it is appreciated that not much more information can be extracted from the organic phosphorus series.

Probably because of the low proportion of phosphorous incorporated in organic cell material, the amount of organic phosphorus uptake due to heterotrophic activity would remain largely unobservable compared with removal due to the settling of organic material. The first order decay rate constant estimated for organic phosphorous is therefore deemed to be indicative of the likely order of magnitude of the rate at which organic material settled in the pond system.

5.4.4.4 POND 2 SIMULATION RESULTS

CARBONACEOUS ORGANICS

The trajectories for the gross reactive rates for Pond 2 data show a similar overall seasonal pattern to that observed for Pond 1 data; generally a lower removal rate in the hotter months than in the colder months. These are shown in Figs 5.74 and 5.75 respectively. The trajectory for soluble COD (Fig. 5.74) suggests some sort of cut-off temperature after which a decrease in the sink rate is observed with decreasing temperature.

The observed reversal (cut-off described above) suggests that either there was a decrease in the effective removal rate or that soluble carbonaceous organics was generated within the pond system, with decreasing temperatures after this cut-off point.

Although it can be expected that algal kills will occur in the cold seasons, considerable cell lysis with a subsequent release of soluble organics is not expected at low pond temperatures.

The observed profile could possibly have been produced by two superimposed effects. These effects are:

- o The decrease in removal rates with decreasing temperature which results from the general decrease in the uptake of substrate by bacteria with decreasing temperature

and superimposed on this,

- o The effect of an internal loading of soluble carbonaceous organics which increases with increasing temperatures.

The negative mean value for the total solids sink rate for this pond system shows that effectively, there was a net production of solids in the pond system over the annual cycle. Based on the theory on pond processes, this increase in solids probably resulted from algal synthesis.

It is clear from the simulation and errors plots (Figs. 5.76-5.83) that, the observed removals in the cold months are under estimated whereas those for the hot months are over estimated. The seasonal nature of the errors are reinforced by the structure evident in their autocorrelograms.

NUTRIENTS

The profile of the ammonia gross reactive rate constant (eg. see Fig. 5.84) has quite similar trends to that observed for the soluble COD (especially over the second half of the year). The seasonal nature though is less evident for the ammonia removal rate. The observed removals in the first half of the year are considerably less than those observed for the second half of the year.

The influent and effluent nitrate plot (see Fig. 3.120) show that there were considerable concentrations of nitrate in the waste stream over the first half of the year and for virtually all of the second half, the effluent nitrate was actually below detection levels.

Since algae utilize both ammonia and nitrate it is conceivable that the preference of the colonizing community was for nitrate and under nitrate limitations, ammonia was the main source of inorganic nitrogen. The observed profile for the nitrate gross reactive rate constant (Fig. 5.85), apparently lends support to the above explanation for the observed trajectory for ammonia.

The simulation and errors plots for both ammonia and nitrate (Figs. 5.86-5.93) clearly show that the simple sink model does not represent the observed pond dynamics for these two states. The plots for the first order decay model for ammonia (see Figs. 5.88-5.89) show that the ammonia removal is over estimated in the cold months, and under estimated in the hot months. This is suggestive of a temperature dependent decay term for ammonia removal.

The simple first order decay model output for nitrate (Figs. 5.92-5.93) appears to mimic the salient behavioural pattern observed. The errors are close to the zero-mean line and suggest that this state is well modelled by the first order decay model format. The closeness to the zero-mean line is however due to the small order of magnitude of the nitrate concentrations. Though the autocorrelogram of the errors does not depict a clear seasonal structure, it is significantly different from that for a white noise

sequence and suggests that not all the salient information on dominant pond mechanisms contained in the nitrate series is represented by the first order model.

The profiles for total nitrogen and total phosphorus (Figs. 5.94 and 5.95) have similar trends but are not indicative of any clear seasonal pattern in the the gross removal rates. The resulting simulation and errors plots for the s/sink and first order decay models for both of these states (Figs. 5.96-5.103) suggest that the main observed behavioural pattern is well modelled by a gross reactive rate constant model with the first order decay producing a better fit to the data. The errors and their autocorrelograms however reveal quite clearly that a component with a half period of about 13 weeks, which corresponds roughly to two cycles in the year, is unaccounted for.

5.4.4.5 SUMMARY AND CONCLUSION

The results of the evaluation of the gross reactive processes representation for pond dynamics for the two pond data clearly showed that this representation was inadequate in explaining observed pond behaviour. Valuable insights into the possible nature of the transformation processes occurring, have however been gained from the profiles of the recursive estimates of both the simple source/sink and the first order decay rates.

The trajectories of the recursive estimates indicate that there were seasonal variations in the gross reactive processes for state variables representative of the carbonaceous organics. These were

typically different from those for the measures representative of the nutrients quality of the waste stream. Possible explanations for the observed variations have been presented. Considering these, it is obvious that subsequent working models developed will be different for the different modelled states.

Therefore the working models were developed along the lines of the sub-groupings of interest viz: Carbonaceous Organics and Nutrients. The first of these sub-groupings considered was the carbonaceous organics.

5.4.5 CARBONACEOUS ORGANICS : WORKING MODELS

5.4.5.1 GENERAL

The trajectories of the recursive estimates of the gross reactive rates for the measured states representative of carbonaceous organics for both Pond 1 and Pond 2 showed a general seasonal pattern which apparently was closely related to the temperature/solar radiation annual cycle. It was suggested in the evaluations above, that an internal loading component could have accounted for the observed profile.

A further observation made for Pond 2 data was that this internal loading component probably was superimposed on a temperature dependent decay rate.

A critical overview of pond information reveals that there are quite a number of mechanisms which could offer possible explanations for the observed profile. These include:

- o The feedback of soluble carbonaceous organics from the benthic region to the overlying planktonic region resulting from the transfer, by diffusion and convective currents, of the intermediate fermentation products.
- o The feedback of particulate organics from the benthos due to sludge lifting by gases resulting from increased anaerobic activity in the benthic subregion at higher temperatures.
- o The increase in organic solids resulting from increased algae synthesis at higher temperatures with a subsequent increase in the amount of low molecular weight extra-cellular organic compounds released.
- o Stratification in the pond at high temperatures resulting in;
 - a) a reduction in the effective pond volume and b) nutrient limitations in the effective pond volume on account of stratification, also, in the nutrients.

The observed seasonality could have resulted from either of the above mechanisms operating independently or from some combination of these. Based on the pond data available, the hypothesis on the effects of stratification cannot be evaluated since no spatial related measures are available.

There are no measured variables in the data uniquely associated with either the low molecular weight extra-cellular organic compounds or the intermediate fermentation products. These are rather, all incorporated in the soluble organics measures ie. the filtered and soluble COD for Pond 1 and Pond 2 data respectively. On account of this, the effects of these two mechanisms cannot be evaluated solely from, or on the basis of, the COD measures. Pond literature however, strongly suggests that observed increases in pond effluent solids in the warm periods essentially results from excessive algal growths.

On the basis of the above, the internal loading of particulate organics was assumed to result solely from algae synthesis.

Given the fact that algae in ponds exist as a mixed population comprising species of various sizes, the VOLSS measure can be considered to be a more representative gross measure of the total algal concentrations compared with say evaluations based on microscopic counts.

If the soluble component of the increased loading to the planktonic region in the hotter months was due solely to algal activity, a significant phase shift would not be expected between the internal loading components for both the particulate and soluble carbonaceous organics. On the other hand, if it was due to benthic activity, then it would be expected to have a strong relationship with the sludge temperature.

Though sludge temperature was not measured during the monitoring exercises on the two pond data, this would have been expected to be a damped version of the pond temperature with a very small phase shift between the two temperatures; probably similar to that between pond and air temperatures.

Based on the process description presented, a carbonaceous organics removal rate dependent on temperature would be expected. This is supported by the explanation given for the observed trajectory for the gross reactive rates for Pond 2 carbonaceous organics. There is however no evidence in the trajectories for Pond 1 to support this. Rather, the observation that the removal of

organic phosphorus in Pond 1 was fairly constant throughout the year, would tend to suggest that the carbonaceous organics removal for Pond 1 was constant throughout the year.

In fact the simulation plot for the first order decay model for the Total COD for Pond 1 data suggests that the gross removal of carbonaceous organics could well be represented by a first order decay model (see Figs. 5.104-5.105).

The above assumption would appear to be justifiable for the suspended portion of the waste stream where, the constant removal rate would be indicative of the rate at which organic solids settle out.

The removal of soluble carbonaceous organics, through mainly the activity of heterotrophic organisms, would not be expected to remain constant throughout the changing seasons of the year.

It could however be argued that because of the relatively long detention time in pond systems, the soluble portion of the influent stream is completely transformed into microbial cells which undergo subsequent settling at a fairly constant rate throughout the year.

5.4.5.2 CARBONACEOUS ORGANICS: WORKING MODEL 1

On the basis of the above, a working model for pond carbonaceous organics was developed which incorporated a simple first order decay constant and a source term which was some linear function of either pond temperature or solar radiation. The resulting model formulation is presented below:

$$d(x_i)/dt = U_i(t) - x_i(t)/T(t) - \theta'_i \cdot x_i + \theta_i \cdot U'_i(t) \quad \dots 5.26$$

for $i = 1, 2$

where; θ'_i represents the first order decay rate, d^{-1} ; θ_i represents source rate, $d^{-1}/^{\circ}C$, for $U'_i(.) \hat{=}$ pond temperature ($^{\circ}C$) and, d^{-1}/BTU for $U'_i(.) \hat{=}$ solar radiation (BTU) respectively. The other terms are as previously defined.

Equation 5.26 is an ordinary linear differential Equation which can be integrated explicitly to obtain the following discrete equation:

$$x_i(t_{k+1}) = x_i(t_k) \cdot B_i(t_k) + D_i(t_k) \quad \dots 5.27a$$

for $i = 1, 2$

$$B_i(t_k) = e^{-(\theta'_i + 1/T(t_k))} \quad \dots 5.27b$$

$$D_i(t_k) = C_i(t_k) \cdot (\theta'_i \cdot T(t_k) + 1)^{-1} \cdot T(t_k) \cdot (1 - B_i(t_k)) \quad \dots 5.27c$$

$$C_i(t_k) = U_i(t) + \theta_i \cdot U'_i(t) \quad \dots 5.27d$$

Redefining the parameters as new state variables and equating the measured states to their measured values (ie. $h_i(.) = x_i(.)$), results in the following discrete-discrete state space formulation with a nonlinear measurement equation:

$$\underline{X}'_i(t_k) = \underline{X}'_i(t_{k-1}) + \underline{\epsilon}_i(t_k) \quad \dots 5.28$$

$$h_i(t_k) = h_i(t_{k-1}) \cdot B_i(t_k) + D_i(t_k) + \psi_i(t_k) \quad \dots 5.29$$

Where $\underline{X}'_i(.)$ is the state vector with elements $(\theta'_i; \theta_i)^T$. $B_i(.)$ and $D_i(.)$ are as above but with the parameters represented by the new state variable; $\underline{X}'_i(.)$.

For each of the two measured state variables, the state vector has two elements which are the first order decay rate constant and the source rate respectively. Since the measurement equation is non-linear in the state vector to be estimated, an EKF was used for the estimation.

The H vector (matrix) in the EKF algorithms was defined by:

$$H^T(t_k) \hat{=} (H_1(t_k); H_2(t_k)) = (dh_i/d\theta'; dh_i/d\theta)$$

where;

$$H_1(t_k) = -h_i(t_k) \cdot B_i(t_k) + D'_i(t_k) \quad \dots 5.30a$$

where;

$$D'_i(t_k) = U'_i(t_k) / (\ln(B_i(t_k)))^2 \cdot (B_i(t_k) \cdot (1 - \ln(B_i(t_k))) - 1) \quad \dots 5.30b$$

and

$$H_2(t_k) = (\theta' \cdot T(t_k) + 1)^{-1} \cdot U'_i(t_k) \cdot (1 - B_i(t_k)) \quad \dots 5.31$$

Samples of the resulting trajectories are shown in Figs 5.106-5.107 and 5.108-5.109 for the filtered COD and VOLSS for Pond 1 data respectively. It is obvious from these plots that the

parameter trajectories exhibit compensatory behaviour.

Parameter trajectories exhibiting compensatory behaviour usually suggest that either the wrong model format has been specified or that the parameters are not uniquely identifiable from the data. The latter is quite unlikely given that the two parameters have influence coefficients which are structurally different.

The profile of the recursive estimate of the first order rate constant for both the filtered COD and the VOLSS suggest that this rate could be temperature dependent. The filter estimates of the first order decay rate in the cold months is apparently lower than those for the hot months.

The resulting simulation and errors plots based on the above model format and estimated values for the parameter given by Figs. 5.106-5.109, are shown in Figs. 5.110-5.111 and 5.112-5.113 respectively. It is apparent from these that this model structure is not representative of pond carbonaceous organics dynamics. The model's outputs in the hot months are close to the observed but those in the cold months are an over-estimation of the observed removals.

It is clear from the errors plots (Figs. 5.111 and 5.113) that the resulting errors for both filtered COD and VOLSS have a seasonal pattern. In particular, their effluent concentrations in the cold months are grossly under-estimated. The model however produces a fairly good fit to the observed values for the hot periods of the year.

The models' prediction of higher removals than observed in the cold months most invariably must have resulted from the specification of a higher removal rate in the cold months than observed. This suggests that the first order decay rate is actually temperature dependent; in agreement with the accepted fundamental principle that heterotrophic activity decreases with decreasing temperature.

The pronounced effects of temperature on the activity of heterotrophic organisms is widely recognised. The general observation has been that of a doubling in rate of substrate uptake with every 10°C rise in temperature within the normal operating range (Metcalf and Eddy, 1972). This temperature dependency is usually represented by the Arrhenius relationship (see eq. 4.26 section 4.5.2.2). More complex temperature relations have been postulated to account for such observed phenomena as saturation and denaturing at high temperatures which are not represented by the Arrhenius formulation.

For the typical mixed populations colonizing the planktonic region of a pond system, different organisms will be well suited to different temperature ranges within the operating range and it is the aggregate or overall effect that will be observed in any apparent temperature relationship. This overall effect will therefore not correspond to the observed characteristic rate for a particular group of organisms but would probably be well represented by the exponential relationship described by the Arrhenius formulation. On account of this, it was deemed that the Arrhenius type functional form would provide an adequate description of the

overall temperature relationship for the mixed population of heterotrophic organisms.

5.4.5.3 CARBONACEOUS ORGANICS: WORKING MODEL 2

On the basis of the evaluations of the first working model, a second working model was developed which evolved around the hypothesis that:

- a) The gross heterotrophic activity could be represented by a temperature dependent first order decay process.
- b) There was an internal source of soluble carbonaceous organics to the planktonic region which was proportional to the concentration of soluble organics in the planktonic region and was generated at a rate which generally increased with increasing temperature.
- c) There was an internal source of particulate organics to the planktonic region which was proportional to the mass of viable cells -represented by the particulate organics measure- and was generated at a rate which generally increased with increasing temperature.

The Arrhenius type relationship was the specified functional form for the source rate. It was deemed to be appropriate since it is a general exponential function with the linear temperature format (see Eq. 5.26) as a special case (ie. temperature coefficient of one).

A mass balance for the planktonic region results in the following deterministic model equation:

$$dx_i(t)/dt = U_i(t) - x_i(t)/T(t) - (R_i(t_p) - R'_i(t_p)) \cdot x_i(t) \quad \dots 5.32a$$

for $i=1,2$

$$R_i(t_p) = \theta_{(1,i)} \cdot (\theta_{(2,i)})^{(t_p - 20)} \quad \dots 5.32b$$

$$R'_i(t_p) = \theta_{(3,i)} \cdot (\theta_{(4,i)})^{(t_p - 20)} \quad \dots 5.32c$$

where $\theta_{(1,i)}$ represents the first order decay rate constant at 20°C for the i^{th} state; $\theta_{(2,i)}$ represents the temperature coefficient in the Arrhenius type relationship for decay rate; $\theta_{(3,i)}$ represents the first order internal source rate for the i^{th} state at 20°C; $\theta_{(4,i)}$ represents the temperature coefficient for the internal source rate and t_p represents temperature in °C. The other terms are as previously defined.

The differential equation for the soluble and particulate organics dynamics (Eq. 5.32) are linear in these measured states; $(x_1; x_2)$ and as such, they can be integrated explicitly to obtain the following discrete form:

$$x_i(t_k) = x_i(t_{k-1}) \cdot B_i(t_k) + D_i(t_k) \quad \dots 5.33a$$

$$B_i(t_k) = e^{-(1/T(t_k) + R_i(t_k) - R'_i(t_k))} \quad \dots 5.33b$$

$$R_i(t_k) = \theta_{(1,i)} \cdot (\theta_{(2,i)})^{(t_p(t_k) - 20)} \quad \dots 5.33c$$

$$R'_i(t_k) = \theta_{(3,i)} \cdot (\theta_{(4,i)})^{(t_p(t_k) - 20)} \quad \dots 5.33d$$

$$D_i(t_k) = U_i(t_k) / -\ln(B_i(t_k)) \cdot (1 - B_i(t_k)) \quad \dots 5.33e$$

Redefining the parameters as the new state variable; $x'_i(t_k)$ ($x'_{1,i}(t_k); x'_{2,i}(t_k); x'_{3,i}(t_k); x'_{4,i}(t_k)$) such that:

$$x'_{j,i}(t_k) \equiv \theta_{(j,i)} \quad \dots 5.34$$

for $j=1,2,\dots,4$ and $i=1,2$

results in the following discrete-discrete state-space formulation when the measured states are equated to the measured values:

$$\underline{x}'_i(t_k) = \underline{x}'_i(t_{k-1}) + \underline{\epsilon}_i(t_k) \quad \dots 5.35a$$

$$\underline{h}_i(t_k) = \underline{h}_i(t_{k-1}) \cdot B'_i(t_k) + D'_i(t_k) + \underline{u}_i(t_k) \quad \dots 5.35b$$

Where, $B'_i(.)$ and $D'_i(.)$ are similar to $B_i(.)$ and $D_i(.)$ in Equation 5.33 but with the parameters replaced by their new state variable representation.

The measurement Equation 5.35b is non-linear in the states therefore, a non-linear estimation scheme had to be implemented. Equation 5.34 was set up for an EKF estimation where H in the EKF algorithms for each of the modelled states was given by:

$$H^T(t_k) \equiv (H_1(t_k); H_2(t_k); H_3(t_k); H_4(t_k))$$

where;

$$H_j(t_k) = -C_j(t_k) \cdot h(t_k) \cdot B'_i(t_k) + \frac{U(t_k)}{(A'_i(t_k))^2} \cdot C_j(t_k) \cdot (B'_i(t_k) \cdot (1 - A'_i(t_k)) - 1) \quad \dots 5.36a$$

$$A'_i(t_k) = \ln(B'_i(t_k)) \quad \dots 5.36b$$

for $j=1,2,\dots,4$

$$C_1(t_k) = x'_2(t_k)^{(t_p(t_k)-20)} \quad \dots 5.36c$$

$$C_2(t_k) = x'_1(t_k) \cdot (t_p(t_k) - 20) \cdot x'_2(t_k)^{(t_p(t_k) - 21)} \quad \dots 5.36d$$

$$C_3(t_k) = -x'_4(t_k)^{(t_p(t_k) - 20)} \quad \dots 5.36e$$

$$C_4(t_k) = -x'_3(t_k) \cdot (t_p(t_k) - 20) \cdot x'_4(t_k)^{(t_p(t_k) - 21)} \quad \dots 5.36f$$

The EKF estimation scheme for the above model representation did not converge to realistic estimates. Most runs lead to filter divergence. This lack of convergence, could have resulted from the high degree of non-linearity in the measurement equation (Eq. 5.36). Also, the similarities in structure of the influence coefficients and the high degree of noise on the measurement data could have lead to confusion in the gain of information from the residuals.

The Rosenbrock block estimation procedure was used to obtain estimates of the states (parameters). The results of this Block Estimation procedure for both Pond 1 and Pond 2 data are given in Table 5.6. Results of simulations based on these parameter values are shown in Figs. 5.114-5.117 and 5.118-5.121 for Pond 1 and Pond 2 data respectively.

POND 1 SIMULATION RESULTS

The simulations and errors plots for Pond 1 carbonaceous organics (Figs. 5.114-5.117) show that a significant proportion of the observed pond behaviour is explained by the assumed model. It is evident from the simulation plot for filtered COD (Fig. 5.114) that the observed effluent concentrations are under-estimated in the hot

period at the end of the simulation period. Though the autocorrelogram of the resulting errors (Fig. 5.115) are significantly different from that for a white noise sequence, no clear-cut structure is indicated.

The autocorrelogram of the particulate organics (Fig. 5.117) suggests a seasonal structure in the resulting errors. The autocorrelogram appears to be a damped sine wave with a half period of about 12 weeks - indicating a 2 cycles in the year.

Pond 2 SIMULATION RESULTS

The simulation and errors plots for Pond 2 data are shown in Figs. 5.118-5.121. It is evident from these also, that a significant proportion of the observed pond behaviour is explained by the model. The autocorrelogram of the errors however clearly show that the errors are structured and the apparent seasonal structure observed for Pond 1 volatile solids errors are also indicated in these errors.

SUMMARY AND CONCLUSION

The resulting errors for the second working model representation suggest that though a significant proportion of the total variation in the observed pond effluent carbonaceous organics characteristics was explained by the model, some pond mechanism with roughly 2 cycles in the year was unaccounted for.

In order to investigate the possible nature of the unaccounted for pond phenomena, the first order temperature dependent decay

process representation for the gross heterotrophic activity was assumed to be representative of actual pond dynamics and all other effects were lumped together as an unknown time variable source/sink term.

5.4.5.4 CARBONACEOUS ORGANICS: WORKING MODEL 3

On the basis of the results of the evaluation of the second working model, a third working model was developed which was based on the assumption that the functional form for the gross heterotrophic activity could be well represented by the Arrhenius type temperature relationship for a first order temperature dependent rate. Also, the parameter values estimated for this functional representation for the second working model (viz. Eq. 5.32 and Table 5.6) were assumed to be representative. This resulted in the following deterministic model representation for the carbonaceous organics.

$$dx_i(t)/dt = U_i(t) - x_i(t)/T(t) - R_i(t_p) \cdot x_i(t) + \theta_{(3,i)}(t) \quad \dots 5.37a$$

for $i=1,2$

$$R_i(t_p) = \theta_{(1,i)} \cdot (\theta_{(2,i)})^{(t_p - 20)} \quad \dots 5.37b$$

Where, $\theta_{(3,i)}(t)$ represents the time varying lumped internal interactions and the other terms are as previously defined.

Because the model Equation 5.37 is linear in the measured state, it can be integrated explicitly to obtain a discrete form and hence the following discrete-discrete state-space representation in which,

the lumped source/sink term is the unknown time varying state to be estimated:

$$x'_i(t_k) = x'_i(t_{k-1}) + \epsilon_i(t_k) \quad \dots 5.38a$$

$$h_i(t_k) = h_i(t_{k-1}) \cdot B_i(t_k) + D_i(t_k) + \psi_i(t_k) \quad \dots 5.38b$$

for $i=1,2$

$$B_i(t_k) = e^{-(1/T(t_k) + R_i(t_k))} \quad \dots 5.38c$$

$$D_i(t_k) = (U_i(t_k) + x'_i(t_k)) / (-\ln(B_i(t_k)) \cdot (1 - B_i(t_k))) \quad \dots 5.38d$$

The state-space representation (Eq. 5.38) has a systems and measurement equation, both linear in the lumped parameter. An IV was therefore used for the recursive estimation of this state.

Trajectories of the resulting estimates are shown in Figs. 5.122-5.125 for the soluble and particulate carbonaceous organics measures for both Pond 1 and Pond 2 data. It is clear from these plots that the main variation in the lumped component is seasonal; quite similar to the solar radiation/temperature annual cycle.

In order to investigate further the signals composition of the estimated lumped component, a Harmonic Analysis was done on the recursive estimates of the source/sink term for a relatively high Q (ie. $Q=400$). The high Q specified was in recognition that the parameter was time varying and to overcome bias resulting from filter inertia.

The Harmonic Analysis showed that a 1 and 2 cycles in the year were the main signals composition of the lumped source/sink

component. The results of a one cycle fit to the parameter trajectories are shown in Figs. 5.126-5.129 and Figs. 5.130-5.133 for Pond 1 and Pond 2 data respectively. It is quite evident from the autocorrelogram in the errors plots that the residuals of the 1-cycle fit are structured. The autocorrelograms all appear to be sine waves with half periods in the region of 12-13 weeks which corresponds roughly to two cycles in the year.

The results of a 2-cycles fit to the residuals of the 1-cycle fit, are shown in Figs. 5.134-5.137 and Figs. 5.138-5.141 for Pond 1 and Pond 2 data respectively. It is clear from the autocorrelograms that the residuals after the 2-cycles fit contain very little structured information. The general pattern observed for the 2-cycles component is, a peak in the source rate in the region of the coldest and hottest months of the year with a peak in the sink rate in the quarter term periods between the observed peaks in the source rate. The peaks in the sink rate correspond approximately to spring and autumn in the typical temperate climate.

The cosine and sine relations evaluated for the 1 and 2 cycles fit to the lumped component are given in Table 5.7 for both Pond 1 and Pond 2 data. The relations for the 1 and 2 cycles components in temperature and solar radiation for both Pond 1 and Pond 2 are also given in Table 5.7. The resulting plots of the 1 cycle fit and the 2-cycles fit to the residuals of the 1-cycle fit for solar radiation and temperature are shown in Figs. 5.142-5.149.

It is quite evident from the two cycles plots that there is a significant phase shift between the two cycles components of pond

temperature and solar radiation for Pond 2 data. Also, the two cycles component for the lumped source/sink term for Pond 2 particulate carbonaceous organics, corresponds to that for solar radiation. This suggests that the primary input variable that controlled the observed two cycles component for Pond 2 solids was solar radiation.

Partitioning the observed lumped component into 1 and 2 cycles components, and assuming that the 1-cycle component is representative of pond internal loading results in an internal loading component which varies from around zero in the cold months to well over the mean influent loading, for the hot months. A comparison between the the mean internal and external (Influent source) loadings for soluble and particulate carbonaceous organics for both Pond 1 and Pond 2 data is given in Table 5.8. It can be seen that for both Pond 1 and Pond 2, the internal particulate carbonaceous organics exceed the external loading showing the extent to which solids are generated within pond systems.

The relative similarity between the observed 2-cycles component for pond carbonaceous organics dynamics and the 2-cycles components for temperature and solar radiation suggest that the two cycles component in the lumped source/sink term is driven by some solar radiation/temperature related factor.

A positive value for the source sink term estimated, represents a net source and possible production of material whereas a negative value represents a net sink and possible consumption of material. Also, a net source could represent a relative reduction in the rate of removal whereas a net sink could represent a relative reduction

in the rate of production.

Analysing the observed 2-cycles component with the above considerations in mind and with pond process theory as a guide produces several candidate mechanisms which offer possible explanations for the observed 2-cycles component.

Temperate ponds are known to stratify in the summer and as such, the pond has the smallest effective volume then. Effective here refers to the aerobic portion above the thermocline. The increase in the sink rate with decreasing pond temperatures after the hottest period of the year, could possibly represent the increase in effective pond volume with the sinking of the thermocline as pond temperatures decrease. In this regard, the peak in the sink rate would then represent the the spring and autumn (fall) overturns when the thermocline has sunk to such a level that no stable density differences are created.

Spring and autumn overturns in temperate ponds results in the complete mix of the planktonic region of the pond system with a resulting increase in the internal loading of nutrients and carbonaceous organics which stimulate the growth of both algae and bacteria.

The observed trend of an increase in the source rate to a peak in the coldest month would then most likely correspond to the general decrease in the overall removal rate with decreasing temperatures.

The 2-cycles component also could have resulted from the succession of two main microbial communities one well suited to the

relatively high temperatures in the hottest month and the other well suited to the near freezing conditions in the coldest month. Extensive studies on the population dynamics in ponds have not been undertaken thus it is difficult to ascertain the likelihood of the above explanation. The increase in the net source rate in the colder months could also have resulted from algae kills.

SIMULATION RESULTS

The results of simulations based on the temperature dependent first order decay rate and a two component source/sink term are shown in Figs. 5.150-5.153 and Figs. 5.154-5.157 for Pond 1 and Pond 2 data respectively. These simulation and errors plots show that the dominant process behavioural modes identifiable from the data have been explained by the model.

5.4.5.5 CONCLUSIONS

Apart from the effluent filtered COD, however, the autocorrelogram of the errors for the other modelled states all have a significant lag 1 autocorrelation coefficient. The simulation plot for effluent volatile solids for Pond 1 data shows that the models predictions lags behind the observations with the average lag time for the peaks and troughs marked with asterix (see Fig. 5.152) being about 2 weeks. This suggests the presence of time delay elements which are not accounted for by the adopted model structure.

On account of changes in the effective pond volume with the varying seasons, time delays are unlikely to remain constant. These time delay elements could possibly account for the significant lag 1

autocorrelation coefficients observed in the errors plots.

It is clear however from the autocorrelograms of the modelled states that no medium to long term variational pattern exists in the errors for the modelling result based on the third working model. This suggests that the observed slow pond process dynamics for carbonaceous organics has been explained. Therefore, with regards the acquired data, increasing the complexity of the model by incorporating more base model entities and possibly by coupling the states will result only in marginal improvements in the explanation of the observed process behavioural characteristics.

In this regard, the main process behavioural modes for the carbonaceous organics dynamics of the typical temperate region pond treating domestic type waste can be considered to have been identified. The identified model structure incorporates an internal loading component which is not measured in routine pond monitoring exercises. However, in order to develop representative functional relationships between the internal loading component and the main exogenous variables, there is the need for appropriate measures for the internal loading component.

5.4.6 NUTRIENTS: WORKING MODELS

5.4.6.1 GENERAL

The identification of pond nutrients dynamics on the basis of the acquired data was limited by the fact that the measures of the inorganic nutrients for Pond 1 did not comprise of a full years

record. Also, there were no measures in Pond 2 data representative of the inorganic phosphorus.

It is virtually impossible to determine with certainty, seasonal properties from a data set comprising of less than a full year's record.

Because of the above restriction in the data, working models for Pond 1 and Pond 2 nutrients were developed separately.

5.4.6.2 POND 1 NUTRIENTS: WORKING MODELS

The trajectories of the recursive estimates for the gross reactive rates showed the likelihood that similar dominant mechanisms accounted for the observed removals of both inorganic phosphorus (ortho) and ammonia nitrogen. The simulations and errors plots for the simple first order decay model (Figs. 5.60-5.61 and 5.64-5.65) showed that the time invariant parameter estimate resulted in the over-estimation of the observed removals in the winter and the under-estimation of those for the summer.

Based on the observations mentioned above, the autotrophic activity of the primary producers (mostly algae) was assumed to be the main mechanism responsible for ammonia and ortho-phosphorus removals observed in Pond 1.

WORKING MODEL 1

The initial working model for nutrients dynamics assumed that the rate at which each of the two nutrients (viz. ammonia and ortho-phosphorus) were removed was proportional to their

concentrations in the pond liquid. It was further assumed that the overall variation in their removal rates with temperature could be represented by the Arrhenius type exponential function/relation. This resulted in the following uncoupled deterministic model representation:

$$dx_i(t)/dt = U_i(t) - x_i(t)/T(t) - R_i(t_p) \cdot x_i(t) \quad \dots 5.39a$$

for $i=1,2$

$$R_i(t_p) = \theta_{(1,i)} \cdot (\theta_{(2,i)})^{(t_p - 20)} \quad \dots 5.39b$$

Where $\theta_{(1,i)}$ represents the nutrients removal rate at 20°C ; and $\theta_{(2,i)}$ represents the temperature coefficient in the exponential relation.

Integrating Eq. 5.39 explicitly results in discrete equation non-linear in the parameters. Redefining the parameters as new states and equating the measured states to their measured values results in the following discrete-discrete state-space representation:

$$\underline{x}'_i(t_k) = \underline{x}'_i(t_{k-1}) + \underline{f}_i(t_k) \quad \dots 5.40a$$

$$h_i(t_k) = h_i(t_{k-1}) \cdot B_i(t_k) + D_i(t_k) + \underline{w}_i(t_k) \quad \dots 5.40b$$

for $i=1,2$

$$B_i(t_k) = e^{-(1/T(t_k) + C_i(t_k))} \quad \dots 5.40c$$

$$C_i(t_k) = x'_{(1,i)}(t_k) \cdot x'_{(2,i)}(t_k) \cdot (t_p(t_k) - 20) \quad \dots 5.40d$$

$$D_i(t_k) = U_i(t_k) / \ln(B_i(t_k)) \cdot (1 - B_i(t_k)) \quad \dots 5.40e$$

where H in the EKF algorithms is given by;

$$H_i(t_k)^T \triangleq (H_{(1,i)}(t_k); H_{(2,i)}(t_k))$$

where (dropping the subscript i);

$$H_j(t_k) = -C_j(t_k) \cdot h(t_k) \cdot B(t_k) + \\ U(t_k) / (\ln(B(t_k)))^2 \cdot C_j(t_k) \cdot (B(t_k) \cdot (1 - \ln(B(t_k)))) - 1) \quad \dots 5.41a$$

for $j = 1, 2$

$$C_1(t_k) = x'_2(t_k)^{(t_p(t_k)-20)} \quad \dots 5.41b$$

$$C_2(t_k) = x'_1(t_k) \cdot (t_p(t_k)-20) \cdot x'_2(t_k)^{(t_p(t_k)-21)} \quad \dots 5.41c$$

The parameter trajectories for the recursive estimation of the two parameters based on the EKF algorithms resulted in compensatory parameter trajectories. Because of the relatively short length of the record for the nutrients series, no interpretations could be made of the observed non-stationarity.

The Rosenbrock procedure was used to obtain block estimates for these parameter values. The resulting values are given in Table 5.9. Results of simulations based on these parameter values are shown in Figs 5.158-5.159 and 5.160-5.161 for ortho-phosphorus

and ammonia respectively. It is clear from these plots that the model fails to reproduce the observed behaviour in the cold months. The autocorrelogram of the ortho-phosphorus errors (Fig. 5.159) clearly suggests that a 2-cycles in the year component is unaccounted for.

WORKING MODEL 2

The simulations plots (Figs. 5.158 and 5.160) however suggest that the general long term trends are represented by the above model. It was assumed therefore that the above model format represents the observed overall nutrients removal mechanism and that other phenomena were superimposed on this general behavioural pattern. The phenomena accounting for the observed discrepancies were lumped together as a time variable source/sink term which was then estimated.

Based on the above assumptions, the second working model developed for Pond 1 inorganic nutrients was similar to the third working model for Pond 1 carbonaceous organics (Eq. 5.37). The trajectories of the resulting IV estimate of the lumped source/sink term are shown in Figs. 5.162 and 5.163 for ortho-phosphorus and ammonia respectively.

The observed non-stationarity in the trajectories of the source/sink term were similar to those observed for the gross reactive rates. Here, there is however a net source of material in the coldest and hottest months in the year, and a peak in the sink rate in the quarter term periods. The observed pattern

actually resembles the 2-cycles patterns observed for the carbonaceous organics (see Figs. 5.134 and 5.136).

A harmonic analysis of the lumped term showed the main signals content to be 2-cycles in the year. The results of a 2-cycles fit to the trajectories for a relatively high Q are shown in Figs. 5.164-5.165 and 5.166-5.167 for ortho-phosphorus and ammonia respectively. It is clear from these plots that the 2-cycles component is similar to that observed for the carbonaceous organics. The coefficients for the sine and cosine relations for the 2-cycles fit are given in Table 5.10.

The errors plot for the 2-cycles fit to the lumped term for ortho-phosphorus (Fig. 5.165) however shows the likelihood of some other pond mechanisms. Though the autocorrellogram shows that the errors are significantly different from white noise, they are not indicative of some clear-cut seasonal structure.

Results of simulations based on the above model with the 2-cycles representation for the lumped term are shown in Figs. 5.168-5.169 and 5.170-5.171 for ortho-phosphorus and ammonia respectively. These plots show that a significantly greater proportion of the overall behaviour observed has been explained by this model. This model however clearly under-estimates the observed removals around day 250.

The dip in the observed effluent ortho-phosphorus and ammonia concentrations clearly did not result from a reduction in their respective influent concentrations. The plots of the measured influent and environmental characteristics for Pond 1 show that

both wind speed and influent nitrate were high around the day 250 period.

In fact, a smoothed version of the influent wind speed (Fig. 5.172) suggests that the observed errors in the 2-cycle fit to the lumped term could have resulted from two superimposed effects. These are:

- o An increase in wind speed above a threshold level resulting in the increased source of inorganic material to the planktonic region, probably due to wind induced mixing or resuspension,

and superimposed on this

- o A non-seasonal removal of inorganic nutrients due to stimulation of growth resulting from a high influent nitrate source.

The two mechanisms outline above were investigated in the third working model for Pond 1 nutrients.

WORKING MODEL 3

The next working model developed was to test the above hypothesis on the mechanisms responsible for the observed pond behaviour. A threshold effect type functional relation was developed for the influence of influent nitrate and wind. It was assumed that below certain threshold values for both wind and influent nitrate, these control variables had no discernable influence on the states of the system being modelled. When these control variables assume values above these threshold levels, their effects (viz. source or sink) is linearly related to their strength.

The deterministic model representation based on the above

assumption on system dynamics are given below:

$$\begin{aligned} dx_i(t)/dt = & U_i(t) - x_i(t)/T(t) - R_i(t_p) \cdot x_i(t) + R'_i(t) - \\ & \theta_{(3,i)} \cdot (N(t) - \theta_{(4,i)}) + \theta_{(5,i)} \cdot (W(t) - \theta_{(6,i)}) \quad \dots 5.42a \end{aligned}$$

for $i = 1, 2$

$$R_i(t_p) = \theta_{(1,i)} \cdot \theta_{(2,i)} (t_p - 20) \quad \dots 5.42b$$

$$R'_i(t) = \text{2-cycle source/sink term} \quad \dots 5.42c$$

Where; $\theta_{(1,i)}$ and $\theta_{(2,i)}$ are as defined above for Eq. 5.39b; $\theta_{(3,i)}$, is the sink rate of the i^{th} state due to influent nitrate; $N(t)$, is the influent nitrate concentration at time t ; $\theta_{(4,i)}$, is the threshold concentration for influent nitrate; $\theta_{(5,i)}$, is the source rate of the i^{th} state due to wind; $W(t)$, is the wind speed at time t ; and $\theta_{(6,i)}$, is the threshold wind speed.

The estimated values of the parameters $\theta_{(1,i)} - \theta_{(6,i)}$ in the above model using the Rosenbrock block estimation procedure are given in Table 5.11.

The resulting simulations and errors plots based on the above model structure and estimated parameter values are shown in Figs. 5.173-5.174 and 5.175-5.176 for ortho-phosphorus and ammonia respectively. Comparing these plots with the corresponding plots for the second nutrients working model shows that though the third working model incorporates a greater number of functional

components, it does not produce a better model fit to the observations. The errors for the third working model are clearly structured and there is no indication that the assumed threshold type model is representative of the actual pond interactions observed.

5.4.6.3 POND 2 NUTRIENTS WORKING MODELS

The stoichiometric relations between the elements carbon, nitrogen and phosphorous for the typical microbial cell mass suggests that a significant proportion of the observed total phosphorus and nitrogen for Pond 2 data comprises of the inorganic forms. The initial working models for Pond 2 total phosphorus and nitrogen states were thus developed along the lines of the working models for Pond 1 nutrients.

The first hypothesis evaluated was similar to that for the first working model for Pond 1 nutrients. The resulting simulations and errors plots based on this model structure and block estimates of the model parameters are shown in Figs. 5.177-5.178 and Figs. 5.179-5.180 for total phosphorus and total nitrogen respectively.

It is evident from the simulations and errors plots that though a significant proportion of the observed total variation is explained by the model, an apparent 2 cycles in the year component is unexplained.

Considering the estimated parameter values for the gross temperature dependent decay rate as representative, and lumping all other phenomena as a time varying source/sink term results in the

trajectories shown in Figs. 5.181 and 5.182 for the IV estimates of this lumped term for total phosphorus and nitrogen respectively.

A harmonic analysis of the parameter trajectories confirmed that the main frequency component in the lumped terms was 2-cycles in the year. Figure 5.183-5.184 and 5.185-5.185 show the resulting 2-cycles fit to the lumped terms. It is quite clear from these figures that the two cycles component has a similar variational pattern to that observed for the carbonaceous organics for both pond data and that for the inorganic nutrients for Pond 1 data.

Results of simulations based on the 1st order decay and 2-cycles fit to the lumped term for total phosphorous and nitrogen are shown in Figs. 5.187-5.188 and 5.189-5.190. It is evident from these plots that this model format explains the salient observed slow dynamics characteristics for Pond 2 total phosphorus and total kjeldahl nitrogen.

Nitrate and ammonia were also analysed starting off with the first nutrients working model. Results of the EKF estimation of the first order decay rate constant and temperature coefficients are shown in Figs. 5.191-5.192 and Figs. 5.193-5.194 for nitrate and ammonia respectively.

It is evident from the final value, the mean and the standard deviation of the parameter trajectory for nitrate (see Figs 5.191-5.192) that the estimated parameters had a near constant value throughout the year. The results for ammonia (see Figs. 5.193-5.194) however suggest that the model representation is inadequate. The temperature coefficient exhibits a seasonal

variation; being greater in the colder months than in the warmer months.

The results of simulations based on the mean of the parameter values are shown in Figs. 5.195-5.196 and Figs. 5.197-5.198 for effluent nitrate and ammonia respectively. It is clear from the nitrate plots that the first working model for nutrients adequately represents the observed nitrate transformations. Although the autocorrelogram of the errors (Fig. 5.196) is not strictly indicative of a white noise error sequence, no clear-cut structure is indicated.

It must be borne in mind, however, that the analytical procedure for measuring the nitrate concentrations in the waste stream could not detect concentrations less than 0.1 mg/l accurately (EPA, 1977a). Because of this, samples with concentrations less than 0.1 mg/l were reported as: "< 0.1 mg/l" in the results of the monitoring exercise (EPA, 1977a). In this work, the samples with concentrations less than 0.1 mg/l were taken to be 0.1 mg/l. Thus, the prediction of lower nitrate concentrations than observed, from around day 150 to the end of the simulation period where practically all observed nitrate concentrations are 0.1 mg/l, could possibly account for the serial correlations observed.

The plots for ammonia (Figs. 5.197-5.198) show that the observed effluent ammonia is not well modelled. The model gives a fairly good fit to the observed ammonia concentration in the warm months but grossly under-estimates the observed concentrations in the cold months.

The relatively high ammonia concentrations observed in temperate ponds in the cold months (Drew and Denyssen, 1978) has been explained as probably resulting from the liquefaction of proteinaceous material derived from algal kill. Another possible explanation is the temperature effects on nitrification. Randall and Bur (1984) have observed for activated sludge that nitrification decreases significantly below 17°C and is negligible below 10°C.

WORKING MODEL 4

In order to investigate further, the dynamics of ammonia depicted in the Pond 2 data, the temperature dependent first order decay mechanism was assumed to be representative (since it gave a good fit to the summer months) and the mechanisms described above were lumped together into a time variable source/sink term. This resulted in the model formulation presented below (nutrients working model 4):

$$dx(t)/dt = U(t) - x(t)/T(t) - R_1(t_p) \cdot x(t) + R_2(t) \quad \dots 5.43a$$

$$R_1(t_p) = \theta_1 \cdot (\theta_2)^{(t_p - 20)} \quad \dots 5.43b$$

$$R_2(t) = \theta_3(t) \quad \dots 5.43c$$

where; $U(.)$ represents the influent ammonia loading ($\text{mg l}^{-1} \text{d}^{-1}$); $x(.)$ is the pond and effluent ammonia concentration (mg l^{-1}); $T(.)$ is the dynamic detention period; $R_1(t_p)$ is the temperature dependent first order decay rate; $R_2(t)$ is the time varying lumped s/sink term; and t_p is the pond temperature.

Integrating Eq. 5.43 explicitly and assuming that both the inputs and the parameters do not vary significantly between unit time steps

results in the following discrete form:

$$x(t_k) = x(t_{k-1}) \cdot B(t_k) + D(t_k) \quad \dots 5.44a$$

$$B(t_k) = e^{-(1/T(t_k) + R_1(t_k))} \quad \dots 5.45b$$

$$R_1(t_k) = \theta_1 \cdot \theta_2 (tp(t_k) - 20) \quad \dots 5.45c$$

$$D(t_k) = (U(t_k) + R_2(t_k)) / (-\ln(B(t_k))) \cdot (1 - B(t_k)) \quad \dots 5.45d$$

Redefining the parameters as the new state variable, $x'(tk)$:

$$x'_i(t_k) = \theta_i(t_k) \quad \text{for } i = 1, 2, 3.$$

results in the following discrete-discrete state-space formulation when the measured state is equated to its measured values (ie. $h(.) = x(.)$).

$$\underline{x}'(t_k) = \underline{x}'(t_{k-1}) + \underline{f}(t_k) \quad \dots 5.45a$$

$$h(t_k) = h(t_{k-1}) \cdot B'(t_k) + D'(t_k) + \psi(t_k) \quad \dots 5.45b$$

where $B'(.)$ and $D'(.)$ are similar to $B(.)$ and $D(.)$ in Eq. 5.44 except that the parameters are replaced with their new state variable representation.

The measurement equation (Eq. 5.45b) is non-linear in the states (parameters) therefore, an EKF was used for the recursive estimation of the parameters. H in the EKF algorithms is given by:

$$H(t_k)^T \triangleq (H_1(t_k); H_2(t_k); H_3(t_k))$$

where;

$$H_j(t_k) = -C_j(t_k) \cdot h(t_k) \cdot B(t_k) + \frac{U(t_k)}{(\ln(B(t_k)))^2} \cdot C_j(t_k) \cdot (B(t_k) \cdot (1 - \ln(B(t_k)))) - 1) \quad \dots 5.46a$$

for $j = 1, 2$

$$C_1(t_k) = x'_2(t_k)^{(t_p(t_k)-20)} \quad \dots 5.46b$$

$$C_2(t_k) = x'_1(t_k) \cdot (t_p(t_k)-20) \cdot x'_2(t_k)^{(t_p(t_k)-21)} \quad \dots 5.46c$$

and

$$H_3(t_k) = 1 / (-\ln(B(t_k)) \cdot (1 - B(t_k))) \quad \dots 5.46d$$

The trajectories of the resulting recursive estimates for the parameters for the nutrients working model 4 for ammonia are shown in Figs. 5.199-5.201. The results show that the first order decay rate and temperature coefficient do not vary significantly over the tracking period compared with the lumped source/sink term (see the final value, mean and standard deviation shown in the figures).

The lumped source/sink term (see Fig. 5.201) clearly has a seasonal pattern; being higher in the colder months than in the warmer months. In order to further investigate the nature of the lumped source/sink term, the first order decay rate and temperature coefficients were assumed to be constant and could be represented by the values estimated by the EKF. An IV (similar to that used in the carbonaceous organics working model 3) was then used to estimate the lumped source/sink term recursively.

The resulting trajectory of the IV estimation is shown in Fig. 5.202. A harmonic analysis of the trajectory for a relatively high Q (ie. $Q=400$) showed that the lumped s/sink term was mainly composed of a 1 and 2-cycles in the year. The results of a 2-cycle fit to the residuals of the 1-cycle fit is shown in Fig. 5.203. It is evident that the 2-cycles component has the same seasonal variation as that observed for both the carbonaceous organics and nutrients states previously modelled.

The results of simulations based on the estimated values of the parameters in nutrients working model 4 for Pond 2 effluent ammonia are shown in Figs. 5.204 and 5.205. These plots show that ammonia dynamics as described by Pond 2 data is well represented by the model.

The above results suggest that the production of ammonia in the cold months is the more likely mechanism for the high ammonia levels observed in temperate ponds in the cold months.

5.5 END POINT

The identification exercise has shown that the gross behavioural characteristics manifested by the typical unbaffled temperate facultative pond can well be represented by simple linear dynamic models. Though undoubtedly, the model formats developed describe a limited scope of the possible pond behavioural characteristics, there is no doubt that the essential features of the slow dynamics have been identified.

Because other exogenous variables (eg. wind and the diurnal variation in sunlight) which have significant influences on the daily characteristics of pond systems were not taken into account explicitly in the model formulations developed, the models cannot be expected to predict pond daily behaviour with a high degree of certainty. The medium to long term behaviour is however deemed to be well represented.

The use of a hypotheses evaluation scheme about base model simplifications has been demonstrated. Here, base model refers to the complete pond process description as presented by the current level of knowledge on the mechanisms operating. This resulted in simple linear dynamic models describing the overall behavioural pattern.

Information gained from posteriori analysis of the pond data lead to the identification of a possible internal loading component, probably resulting from the feedback of soluble carbonaceous organics and the synthesis of algae cells in the hot months.

Also, a 2-cycles in the year pond phenomena was observed for which possible explanations in terms of pond process description were presented. Because there were no measures in the pond data pertaining to the postulated internal sources and sinks, no evaluations could be made to assess their validity. Also, no functional relations were developed for these internal loading components.

There is the need for extensive studies into the nature of the coupling between the benthic and planktonic sub-regions and the

determination of factors influencing the transfer of material between the two zones.

An overview of the identification exercise is given in the next Chapter where the implications of the work for pond design and future monitoring activities are presented.

CHAPTER SIX: OVERVIEW AND USE OF MODELS IN DESIGN

6.1 GENERAL OVERVIEW

The identification exercise described in the previous chapter revealed that the aggregate effect behaviour of ponds (slow pond dynamics) can be well represented by simple linear models in which the main driving forces are the influent stream and seasonal internal loading components. The main seasonal internal component for the carbonaceous organics was found to have a similar characteristic to the annual temperature/solar radiation cycle. This component therefore was deemed to result from the influence of temperature and solar radiation.

It was postulated that the internal loading component for the soluble carbonaceous organics results from the feedback of soluble intermediate fermentation products and that for the particulate organics, results from the synthesis of algae cells.

The exercise also showed that there was a 2-cycle in the year component in pond dynamics particularly significant in the nutrients dynamics. Possible explanations presented for this observation included:

- o Effective volume reduction through stratification
- o Seasonal turnover in temperate ponds
- o Zoo-plankton-phytoplankton interactions

The explanations for the aggregate effect 1 and 2 cycle components were based on information extracted from the a priori pond process description and not on any direct measure of descriptive variables for these phenomena. As such, they cannot be evaluated using the acquired pond data. These observations however suggest areas of uncertainty in pond information which could possibly be resolved by "experimentation".

Experimentation, here, refers to the specific measuring of descriptive variables representative of these phenomena. This would achieve the dual purpose of verifying the behavioural patterns "identified" from the acquired data and also, present data from which quantitative expressions/relations can be developed.

Though the identification exercise showed that the dominant observed behavioural modes of pond systems are seasonal in character, it is clear from the pond process description presented (Chapter 2) that ponds have diurnal characteristics as well. The diurnal characteristics were however not identifiable from the acquired data. The model structure developed also did not include the influence of other exogenous variables such as wind (which apparently did not exert a significant influence on the slow dynamics). The effects of precipitation and evaporation on the liquid retention time were incorporated in a hydrological mass balance for flow. Seepage was considered to be negligible.

On the basis of the above, it is suggested that future pond monitoring exercises should seek to obtain data which incorporates and describes both the slow and fast dynamics. Typically, this could involve a combination of short-term high frequency (at least

two samples within the day) and long term low frequency (at least two samples within the week) sampling schemes.

Measured characteristics which would help in the further evaluation of the hypotheses presented on pond mechanisms include:

- o Measures of the benthic feedback component eg. SBOD or SCOD.
- o The effective volume at different times within the year (seasons) typically by means of dye tracer studies into characteristic residence time distribution patterns.
- o Measure of phytoplankton activity, including descriptors of the total phytoplankton mass and how species distribution relates to this measure.
- o Measures of heterotrophic activity, including descriptors of the total bacterial mass and how this measure relates to possible seasonal changes in predominant heterotrophic organisms.
- o Characterization of the predominant zooplankton species, their succession and measures representative of either their numbers or mass.
- o The dissolved oxygen in ponds on both a daily and seasonal basis and also the variations with depth to enable the evaluation of a depth integrated average for aggregate effect modelling purposes.
- o Measures representative of faecal enteropathogenic organisms, factors influencing their die-offs and the fate and removals of

helminthic cysts and ova.

On account of the diversity in species type and characteristic dimensions, measures of the total mass of organisms without reference to how changes in this measure relates to the succession of the predominant species may produce non-representative descriptive variables for the aggregate effects of: autotrophic activity, heterotrophic activity, and zooplankton population dynamics.

6.2 UNCERTAINTIES IN SYSTEMS REPRESENTATION

The ideal design module described in Chapter 4 incorporates a dynamic pond model which produces output profiles for different envisaged scenarios. In the case of a completely representative process model, a lot of confidence can be placed on the resulting output profile. Therefore, the degree of confidence is related to both the uncertainty in pond information used for model formulation and the amount of that information encapsulated in the adopted model format.

A systems process behaviour can be described qualitatively using linguistic descriptors or quantitatively using mathematical models. Representative mathematical models for systems behaviour usually represent a "fine tuning" of the linguistic description in which ambiguities in system process behaviour have largely been resolved and , suitable functional forms have been found for describing the process mechanisms.

The models developed in this study can be described as exploratory models in that they are not complete representations of the possible systems behavioural modes even for the range of behaviour likely to be encountered under normal operating conditions. They are working models which were used to help evaluate hypotheses on the macroscopic behavioural characteristics of ponds as presented by the acquired data. This hypotheses evaluation scheme was conducted through an experimental framework in which attempts were made to describe overall behaviour, neglecting the subtle interactions between system states.

The exploratory models used were based on continuum dynamics and on a conservation of mass principle. These are well suited to describing the dynamics of processes at equilibrium or in the "close-to-equilibrium" region. There are several self-regulatory mechanisms tending to maintain stable forms and hence stationary states in the typical facultative waste stabilization pond. As such, aggregate effect pond dynamics can be well described by continuum dynamics. It must be borne in mind however that ponds and other ecological systems in the "far-from-equilibrium" region can exhibit catastrophic behaviour which would not be predicted by a model based on observations of the systems behaviour in the "close-to-equilibrium" region.

Pond technology is currently at a phase in the development of more representative process models where an interaction between qualitative and quantitative descriptions aids in the resolving ambiguities in the characterisation of process behaviour. Prior to the verification/validation of a postulate or hypothesis on systems behaviour, through either experimentation or

hypothesis on systems behaviour, through either experimentation or extensive observations of its effects, the unresolved ambiguity presents uncertainty in systems representation. With the use of mathematical models for process description, additional uncertainties are introduced by the unknown system parameters.

The simple pond models developed in Chapter 5 incorporate parameters which are in a sense system characteristic (dependent). This is because they describe pond system properties which are dependent on both the characteristics of the waste being treated and on the pond system environment. As stated previously, they are therefore not universal constants and typical values for characteristic waste and climatic types have to be evaluated.

Though no deterministic/functional forms were developed for the internal loading components, the benthic feedback term measured as SBOD and SCOD for other ecological systems in which the benthos plays a significant role shows that the SBOD or SCOD is not related to the sludge mass but rather, is controlled to a large extent by temperature (Chiaro and Burke, 1980; Walker and Snodgrass, 1986). Any formal description presented relating temperature to this feedback term, for example, will no doubt also incorporate process parameters with system dependent properties.

Extensive calibration of model structure to pond data from ponds treating waste types of specified characteristics in well defined environment types should result in typical parameter values with known probability distributions.

The pond identification exercise produced estimates of typical

values of these pond parameters for two ponds treating domestic type waste in temperate regions. The values obtained for the two pond systems, Pond 1 and Pond 2, are likely to be representative of the values for a medium to high strength, and low strength wastes respectively.

6.3 POND DESIGN AND OPERATIONAL REQUIREMENTS

Design has traditionally been viewed in terms of specifying the capacity of the system required and little information has been presented on the operational requirements. This is mostly because, after the commissioning of a works, the design engineers usually have very little to do with the system.

Optimum pond design in this context has related to the minimization of a total cost function often derived mainly from the cost of civil engineering works. Hence, the main objective has been that of obtaining the minimum capacity that will result in acceptable performance on the basis of the process model underlying the design procedure used.

Based on the realisation of the inadequacies in process models used for design, it is advocated that optimum design should be based on the "best" utilization of the "Total Knowledge" on pond systems behaviour. This will no doubt involve a mixture of empiricism, relating to the well developed qualitative descriptions, and rationalism relating to verified and validated mechanistic (quantitative) representations of process behaviour.

In this regard, optimum pond design also relates to

recommendations for effective operation and maintenance. Though the specification of pond capacity from quantitative representations of process behaviour is important, the success of a pond scheme depends as much on how well it is maintained and operated.

Pond design should be viewed in this light as an integrated effort, not only involving the specification of the dimensions of the facility required, but also producing operational guidelines for different possible behavioural scenarios (especially guidelines on trouble shooting). The recommendations for operation (manipulation of controllable influencing factors to produce desirable states) should be based on sound knowledge on systems behaviour.

Since knowledge acquired by an individual or consulting group represents a limited experience, an operations manual based on the the knowledge of one design engineer or a consulting group would be sub-optimal. It would not represent the best use of the "available" knowledge on pond systems. On the basis of the above, it is recommended that such knowledge be extracted from a group of "experts" with expert knowledge on process behaviour; operational needs etc. Their knowledge can be pooled together into an information base on ponds which could be updated as ambiguities are resolved and more representative process representations developed.

Since certain operational characteristics cannot be described in terms of readily measureable variables or by precise descriptive languages (such as mathematics), operational modules developed may probably have to be in terms of fuzzy descriptors.

For example, the observation that the pond surface

characteristics and colour can be used to determine pond treatment efficacy cannot be described as yet precisely by a mathematical model. How would a statement such as: "When organic wastes are stabilized in the pond, a light wind creates definite waves which are visible on the surface. When organic matter remains untreated, the surface of the pond is glassy smooth even with a light breeze", be represented in a mathematical model for ponds based on currently used mathematical language.

Another example relating to the need for the maintenance of an adequate pond liquid depth is presented: "Methane fermentation is unlikely to occur in a shallow pond leading to excessive sludge build-up and also, growth of vegetation in and around the the pond. This provides potential breeding sites for mosquitoes, rats and musk rats and can further aggravate the problem by excessive seepage resulting from bottom cracks and damage to banks and dikes".

6.4 USE OF MODELS IN DESIGN

In the typical pond design exercise, a characteristic waste strength is usually derived from the expected waste load contribution per person. To arrive at a value for the expected waste load, factors normally taken into consideration include typical characteristics of any measured waste water flows, and the domestic and socio-cultural activities of the contributing population. The estimate so obtained is then taken to be the design influent characteristic and is assumed to be constant throughout the year.

The actual waste load to a treatment facility will however vary

both on a daily and seasonal basis and it is unrealistic to assume a constant waste load for design. This coupled with the fact that ponds are dynamic systems means that critical conditions will not necessarily correspond to when the external loading is maximum as is the case for static systems. Both the Internal and External loading sources vary throughout the year and some combination of these loadings with the pond system in a particular state could lead to critical conditions and thus failure.

If the overall design objective is stated as that of obtaining the minimum pond volume that will result in acceptable pond effluents throughout the typical annual cycle after the establishment of an active benthic subregion, then in order to evaluate pond performance with regards the stated objectives, there is the need to specify what is meant by an acceptable effluent. This would typically be related to the established secondary effluent standards and the assimilative capacity of the disposal media.

To evaluate whether a design pond detention time/volume will result in acceptable pond effluents throughout the year, a Monte-Carlo type simulation with a pre-specified distribution for the dynamic model parameters and pond influent characteristics can be undertaken.

All sets of conditions leading on to failure, failure here being defined in relation to the prespecified effluent standards can then be identified. This should lead to the establishment of critical conditions for design. Also, this would enable the identified critical conditions to be incorporated in an accompanying operations

manual.

In relation to the specified overall objective, the minimum pond volume giving rise to acceptable effluents throughout the year can then be chosen as the pond design volume.

6.5 END POINT

It is clear from the systems study that descriptive procedures for process representation which permit the integration of both qualitative and quantitative process descriptions in a framework which can be used for design and production of operational guidelines for ponds in different regions, have to be evolved.

Such a procedure would enable the expert knowledge on ponds from diverse fields to be incorporated in an "expert system" for ponds.

CHAPTER SEVEN: SUMMARY CONCLUSIONS AND RECOMMENDATIONS

7.1 GENERAL

This systems study has focussed on facultative waste stabilization ponds. It must be noted however, that though facultative ponds can be used as single units, they are often an integral part of a series of ponds generally called the Waste Stabilization ponds system. The usual set-up comprising of an Anaerobic pond followed by a Facultative pond which in turn is followed by an Aerobic/ Maturation pond. Each of these units plays a special role in the ultimate transformation of the waste load and the choice of units used and the sequence of operation is determined largely by the characteristics of the waste stream and by the treatment objectives.

The facultative pond is the unit in which secondary treatment, responsible for the reduction of most of the organic pollutional load, occurs. As a result, the main identification exercise centred around the carbonaceous organics dynamics of ponds.

Information from performance evaluation surveys clearly suggest that a number of ponds operated in series produces a better quality effluent than a single pond system and that treatment in ponds above the third in a series arrangement achieve marginal removals. It is important therefore that facultative ponds be viewed as part of a larger system and that comprehensive studies be undertaken on the other pond types as well. This should result in the evolution of

representative process models for the various pond types and enable optimal design to be based on overall system representation resulting from the coupling of models for the sub-modules.

7.2 SUMMARY AND CONCLUSIONS

Pond effluents have generally been observed to have relatively high solids content. This has been found to be mainly due to algae cells and high quality effluents result when the algae is removed. Though it has been recognised that the algae constitutes a separate phase with a characteristically different impact on disposal media compared with that due to solids of sewage origin, pond performance evaluations have not taken this fact into consideration.

Algae has been measured as part of the organics quality of the effluent waste stream. Thus compared to the influent solids, periodic increases to as much as 258% have been observed in effluent solids from pond systems in summer months. On account of this recent research efforts have been directed towards developing methods for the removal of dispersed algae (Cullinane and Shafer, 1980; Harrelson and Cravens, 1982 ; Yair and Wacks, 1973). These include: Microscreening, Sand filtration and Chemical precipitation. None of these methods have, however, really proved satisfactory to date either because they have turned out to be too complex or too expensive for routine use.

Very little research has been done on the potential for biological control of algae in ponds though the termination of algae blooms by the grazing pressure of zoo-plankton has been observed in

ponds (Mitchell, 1980). Algae are a potential food source for organisms higher up in the aquatic food chain and the harvesting of these larger organisms is not as difficult as attempting to remove dispersed algal cells from pond effluents. Phase isolation ponds have been found to effect very high algae removals. The severely nutrient depleted waters in such ponds create a pond environment suitable for the growth of attached filamentous algae which tend to form surface mats and prevent light penetration. It has generally been observed that the filamentous algae and suspended algae appear to be incompatible and that the presence of one excludes the other.

The review on pond process models, showed that very little progress has been made in attempts at rationalising the design of these systems and that most of the empirical design procedures are no better than design based on recommended loadings. The design procedures have not incorporated basic concepts on the biological and other treatment processes occurring in the pond system and were found to be inadequate system representations.

Based on the recognition of the limitations in the current procedures for pond design, the need for a concerted effort at unifying the search for better design procedures through improved knowledge was pointed out. In line with this, a systems identification exercise with the objective of developing dynamic mathematical models which would help assess pond concepts and identify pond data requirements for the evolution of more representative process models, was undertaken.

The identification exercise produced results which suggests the importance of pond interactions which have largely been ignored in investigatory work on ponds such as, the benthic feedback, stratification and zooplankton/phytoplankton interactions.

The process description presented showed the facultative waste stabilization pond to be a viable biological waste water treatment unit with potential operational modes which could result in the generation of by-products of significant economic value especially for developing countries. The pond environment was found to have exceedingly complex ecology with homeostatic mechanisms operating in such a way as to maintain stable conditions and to adapt the pond environment to varying levels of stress induced by changes in the systems external environment.

7.3 RECOMMENDATIONS

Phenomena influencing pond performance which need further investigation include:

- 1) Interactions between the benthic and planktonic regions. It is essential that the additional seasonal load (summer periods) be quantifiable with regard waste type and prevailing enviromental conditions.

- 2) Broad classification of the succession of algal types and factors influencing the dorminance of the various types: This could lead to the modification of pond environment such that only desirable types proliferate.

3) Investigations into the use of biological control methods for limiting algal cells in subsequent ponds succeeding the primary units. eg. zooplankton, crustacean, coarse fish.

4) Nutrient/Heavy metal removal capabilities: It seems that the high pH arising in pond systems results in the precipitation of hydroxy complexes of heavy metals and phosphorus complexes.

5) Nitrification/Denitrification occurring in the aerobic and anoxic regimes in the planktonic region respectively. Pond functioning seems to suggest that there is nitrification during the day time when the oxidation reduction potential (ORP) is high and denitrification at night when the ORP is low.

It is recommended that full scale experimental ponds be constructed in different regions of the world to study the various phenomena outlined in this work. Comprehensive guidelines for the monitoring of these experimental ponds should be laid down by a panel of experts on ponds. The variables monitored should be determined by the need to understand the underlying processes better, and to provide the data required for the evaluation of proposed descriptive forms for the underlying mechanisms.

If such ponds are linked with to the small bore sewers being recommended by the world bank, this will eliminate the need for anaerobic pre-treatment ponds.

REFERENCES

- Abeliovich, A., and Azov, Y., (1976) Toxicity of Ammonia to Algae in Sewage Oxidation Ponds . Appl. Envir. Microbiol., 31, 801-806.
- Afanas 'Eva, E.I., and Messineva M.A., (1981) Interactions of bacterial plankton and zooplankton in oligotrophic water bodies . Water Resources, 8, No. 4, 457-461.
- Amin, P.M., and Ganapati, S.V., (1972) Biochemical Changes in Oxidation Ponds, J.W.P.C.F., 44, 183-200
- Aris, R., (1961) The Optimal Design of Chemical Reactors , Academic Press, New York. Wiley, New York
- Arceivala, S.J., Lakshminarayana, J.S., Alaqqasamy. S.R., and Sastry, C.A., (1970) Waste Stabilization Ponds . Design Construction and Operation in India, Central Public Health Engineering Research Institute, Nagpur, India.
- Arthur, T.P., (1981) The Development of Design Equations for Facultative Waste Stabilization Ponds in Semi-Arid Areas. Proceedings of the Institution of Civil Engineers, Vol. 71, (2), 197.
- Arvola, L., (1984) Diel Variation in Primary Production and Vertical Distribution of Phytoplankton in a Polyhumic Lake . Archiv. for Hydrobiologie, 101, No. 4, 503-519.
- Atherton, M.J., and King, D.L., (1971) The Effect of Lagoon Effluent on the Oxygen Budget of a Receiving Stream . Proc. 26th Indust. Waste Conf., Eng. Bull., Purdue Univ., Eng. Ext. Series No. 140.
- Avnimelech, Y., Yamamoto, M, and Menzel, R.G., (1983) Evaluating the release of soluble compounds from sediments . Journal of Enviromental Quality, 12, No. 1, 86-91.
- Bain, R.C., McCarty, P.L., Robertson, J.A., and Pierce, W.H., (1970) Effects of an Oxidation Pond Effluent on Recieving Water in the San Joaquin River Estuary , Proc. 2nd Inter. Sym. for Waste Treatment Lagoons, Kansas City, Mo., 168-180.
- Balieux, B., and Troussellier, M., (1983) Changes of Faecal Indicator Bacteria and of Pseudomonas Aeroginosa and Aeromonas Hydrophilia in a Sewage Treatment Lagoon . Techol. Sci. Municipales,, Vol 70, No. 7, 361-570.
- Ballinger, D.G., and Lishka, R.J., (1962) Reliability and Precision of BOD and COD Determinations . J.W.P.C.F., 34, 470

- Banks, R.B., and Herrera, F.F., (1977) Effect of Wind and Rain on Surface Reaeration . Jour. Environ. Eng. Div., A.S.C.E., 103, 489.
- Banks, R.B., Wickrananayake, G.B., and Lohani. B.N., (1984) Effects of Rain on Surface Reaeration . Journal Environmental Engineering, ASCE, Vol 110, No. 1, 1-14.
- Barnes, R.S.K., and Mann, K.H., (1978) Fundamentals of Aquatic Ecosystems . Blackwell Scientific Publications, pp209.
- Barsom, G., (1973) Lagoon Performance and the State of Lagoon Technology . Enviromental Protection Technology Series, PB 223 129, EPA-172-73-144, 214.
- Barsom, G.M., and Ryckman, D.W. (1970) Evaluation of Lagoon Performance in Light of 1965 Water Quality Act . In. R.F. McKinney (ed.) Proceedings of the 2nd International Symposium for Waste Treatment Lagoons. Kansas City, Missouri Basin Engineering Health Council and Federal Water Quality Administration.
- Bartsch, A.F., (1961) Algae as a Source of Oxygen in Waste Treatment . J.W.P.C.F., 33: 239-249.
- Bartsch, A.F. and Allum, M.O. (1957) Biological Factors in Treatment of Raw Sewage in Artificial Ponds , Limnology & Oceanography, II, 2, 77-84.
- Beck, M.B., and Young, P.C., (1976) Systematic Identification of DO-BOD model Structure. ASCE, Env. Eng., 102, 909-924.
- Bertalanffy, L Von., (1933) Modern Theories of Development: An Introduction to Theoretical Biology . Oxford University Press, London
- Black, M.I., Scarpino, P.V., et al., (1982) Survival Rates of Parasite Eggs in Sludge During Aerobic and Anaerobic Digestion . Applied and Enviromental Microbiology, 44, No.5, 1138-1143.
- Bopardikar, M.F. (1969) Microbiology of a Waste Stabilization Pond. IN: Advances in Water Pollution Research, Oxford: Pergamon., 595-601.
- Borup, M.B., and Adams, V.D., (1985) Upgrading Wastewater Lagoon Effluents with the UV/Sedimentation Process . J.W.P.C.F., Vol 57, No. 3, 196-200.
- Bowan, G.T., and Delfino, J.J., (1980) Sediment Oxygen Demand Techniques: A Review and Comparision of Laboratory and In Situ Systems: . Water Research, Vol. 14, 491-499.
- Bowles, D.S., Middlebrooks, E.J., and Reynolds, J.H., (1979) Coliform Decay Rates in Waste Stabilization Ponds . J.W.P.C.F., Vol. 51, No. 1, 87-99.

- Bowsen, S.P., (1977) Performance Evaluation of Existing Lagoons, Peterborough, NH, . EPA-6/2-77-085, U.S. EPA, Cincinnati, Ohio.
- Brockeh, O.D., (1977) Some Causes of Biological Instability and their Effects on Algal Population Levels in Waste Treatment Lagoons . Prog. Wat. Tech., Vol. 9., 941-948.
- Brockett, O.D., (1977) Nitrogenous Compounds in Facultative Oxidation Pond Sediments ., Water Research, Vol.11, 317-321.
- Bradley, R.M., (1983) BOD Removal Efficiencies in Two Stabilization Lagoons in Series in Malaysia . Water Pollution Control, J. of W.P.C., Vol. 82, No.1.
- Bradley, R.M., and Alvares Da Silva, M.D.S., (1977) Stabilization Lagoons Including Experience in Brazil ., Eff. Wat. Treat. J., 71(1), 21.
- Bradley, R.M., and Senra, M.O., (1976) Stabilization Lagoons Including Experience in Brazil . Effluent & Water Treatment Journal, Vol.16, No.12, 619.
- Bryant, C.W., (1983) Benthic Stabilization of Organic Carbon and Nitrogen . Diss. Abst. Int. Pt. B-Sci. & Eng., Vol. 44, No. 5,227.
- Burden, R.C., Faires, J.D., and Reynolds, A.C., (1981) Numerical Analysis , Wadsworth International Students Edition, 2nd Ed., 598.
- Canter, L.W., and Englands, A.J., (1970) States Design Criteria for Waste Stabilization Ponds . J.W.P.C.F., 42, 1840.
- Canter, L.W., Englands, A.J., Jr. and Mauldin, A.F., Jr., (1969) Loading Rates on Waste Stabilization Ponds . Journal of Sanitary Engineering Division., Proc. A.S.C.E., SA 6.
- Chapman, T.G., (1975) Trends in Catchment Modelling. In Prediction in Catchment Hydrology, ed. T.G. Chapman, and F.X. Dunin, Australian Academy of Science, Canberra.
- Chatfield, C., (1975) The Analysis of Time Series . Chapman and Hall, London, New York.
- Chiaro, P.S., and Burke, D.A., (1980) Sediment Oxygen Demand and Nutrient Release , A.S.C.E., EE1, 177-195.
- Coughanowr, D.R., and Koppel, L.B., (1965) Process Systems Analysis and Control . McGraw-Hill Book Company, New York, 491.
- Cullinane, M.J., and Shafer, R.A., (1980) Reduce Lagoon Algae Problems with Coagulants . Water and Wastes Eng., 17, 6.

19.

- Daigger, G.T., and Grady, C.P.L., (1982) The Dynamics of Microbial Growth on Soluble Substrates ; A Unifying Theory . Water Research, 16, No. 4, 365-382.
- Danos, S.C., Maki, J.S., and Remsen, C.C., (1983) Stratification of Micro-organisms and Nutrients in the Surface Microlayer of Small Freshwater Ponds . Hydrobiologia, 98, No.3, 193-202.
- Davis, E.M., and Glyona, E.F., (1972) Bacterial Die-Off in Ponds . Journal of the Sanitary Engineering Division, ASCE, Vol. 98, No. SA1, 59-69.
- DiToro, D.M., et al., (1970) A Dynamic Model of Phytoplankton Populations in Natural Waters . Manhattan College, Bronx, N.Y.
- Drews, R.J.L.C., and Denyssen, J.H., (1978) Sewage Treatment Works for Small Communities - A Need for Careful Design and Operation . Int. Conf. on Water Poll. Contrl. in Dev. Countries, Bangkok, Thailand, 21-25 Feb, 381-397.
- Eckenfelder, W.W., Jr (1966) Industrial Water Pollution Control . McGraw-Hill Book Publishing Co., New York, 206.
- Eckenfelder, W.W. Jr, and O'Connor, D.J., (1960) Treatment of Organic Wastes in Aerated Lagoons . J.W.P.C.F., 32, 365-376.
- Eckenfelder, W.W. Jr, and O'Connor, D.J., (1961) Biological Waste Treatment . Pergamon Press, New York, 224.
- EPA (1977a,b,c,d) Evaluation of Facultative Waste Stabilization Ponds . US Environmental Protection Agency Reports 600/2-77-085,086,109 and 167.
- Eykhoff, P. (1974) Systems Identification, Parameter and State Estimation . John Wiley and Sons, 555.
- Ferrara, R.A., and Avici, C.B., (1982) Nitrogen Dynamics in Waste Stabilization Ponds . J.W.P.C.F., 54, 361-369.
- Ferrara, R.M., and Harleman, D.R.F., (1981) Hydraulic Modelling for Waste Satbilization Ponds . J. of Env. Eng., A.S.C.E., Vol. 107., No. EE4., 817-830.
- Fillos, J. and Swanson, W.P., (1976) Phosphate Release and Sorption by Lake Mahagan Sediments . J. Environ. Eng. Div., Proc. ASCE 102, No. 2.
- Finney, B.A., and Middlebrooks, E.J., (1979) Evaluation of Facultative Waste Stabilization Pond Design In: Performance and Upgrading of Waste Water Stabilization Ponds, Utah Water Res. Lab. Logan. EPA-600/9-79-011

- Fitzgerald, G.P., and Rohlich, G.A., (1958) An Evaluation of Stabilization Pond Literature .
- Fritz, J.J., (1979) Dynamic Process Modelling of Wastewater Stabilization Ponds . J.W.P.C.F., 51, No.11, 2724.
- Fritz, J.J., Meredith, D.D. and Middleton, A.C., (1980) Non-steady State Bulk Temperature Determination for Stabilization Ponds . Water Research, Vol.14, No.5, 413-420.
- Foster, W.E., (1978) Conserve Energy in Wastewater Systems . Water and Wastes Engineering, Vol.15, No.4, 20-23.
- Fruh, E.G. (1967) The Overall Picture of Eutrophication . J.W.P.C.F., 39, 1449.
- Gann, J.D., Collier, R.E. and Lawrence, C.H. (1968) Aerobic Bacteriology of Waste Stabilization Ponds , J.W.P.C.F., 40,2, 185-191.
- Gentil, S. (1984) Linear and Recursive Identification for Ecological Modelling . Ecol. Modelling, 21, 21-33.
- George, D.B., (1979) Waste-Water Treatment - Lagoons and Oxidation Ponds . J.W.P.C.F., 51, 1197-1200.
- Ghosh, T.K., Raa, A.V., Krishnamanthi, K.P., and Joshi, M.W., (1985) Settled Sewage for Pisciculture . Asian Environ., Vol 7, No. 2, 31-35.
- Glyona, E.F., (1971) Waste Stabilization Ponds . WHO (Geneva) Monograph Series No. 60. 175. X
- Gloyna, E.F., (1976) Facultative Waste Stabilization Pond Design : In Ponds as a Waste Water Treatment Alternative, Water Resources Symposium, No. 9, Univ. of Texas, 143-158.
- Gloyna, E.F., and Herman, E.R., (1957) Algae in Waste Treatment . J.W.P.C.F., Vol. 29, No. 4.
- Goldman, J.D., et al., (1974) The Kinetics of Inorganic Carbon Limited Algal Growth . J.W.P.C.F., 46, 554.
- Gotaas, H.B., (1948) Effect of Temperature on Biochemical Oxidation of Sewage. Sewage and Industrial Wastes, 20, 441.
- Goulden, C.F. (1976) Biological Species Interactions and their Significance in Waste Stabilization Ponds . In: Ponds as a Wastewater Treatment Alternative. (eds.) E.F. Glyona, J.F. Malina Jr. and E.M. Davis. , Water Resources Symposium No. 9., Univ. of Texas: Centre for Research in Water Resources.
- Greenfield, R.E., and Elder, A.L., (1926) The Effect of Temperature on the Rate of Deoxygenation of Diluted Sewage

- . Ind. Eng. Chemistry, 18, 281.
- Gross, R., Gross, U., Ramirez, A., Cuadra, K., Collazos, C., and Feldheim, w., (1978) Nutritional Tests with Green Alga Scenedesmus with Healty and Malnourished Persons . In: C.J. Soeder and R. Bisnsack (eds) Microalgae for Food and Feed, Proc. of German-Israeli Workshop, 17-18 Oct., 1977, Neuherberg., Archiv. Hydrobiol. Ergeb. Limnol., 11, 161-173.
- Harrelson, M.E., and Cravens, J.B., (1982) Use of Microscreens to Polish Lagoon Effluents . J.W.P.C.F., 54, No.1, 36-42.
- Harremoes, P., (1982) Immediate and Delayed Oxygen Depletion in Rivers . Water Research, Vol. 16, No. 7, 1093-1098.
- Hartley, W.R., and Weiss, C.M. (1970) Light Intensity and Vertical Distribution of Algae in Tertiary Oxidation Ponds . Water Res., 4, 751-763.
- Hendricks, D.W., and Pote, W.D., (1974) Thermodynamic Analysis of a Primary Oxidation Pond . J.W.P.C.F., 46, 2, 333-351.
- Hermann., E.R., and Gloyna, E.F., (1958) Waste Stabilization Ponds 1. Experimental and Pilot Plant Studies. Sewage & Ind. Wastes, 30, 511-538.
- Hermann., E.R., and Gloyna, E.F., (1958) Waste stabilization Ponds. II. Field Practices. Sew. & Ind. Wastes, 30, 5, 646-651.
- Hermann., E.R., and Gloyna, E.F., (1958) Waste Stabilization Ponds. III. Formulation of Design Equations . Sew. & Ind. Wastes, 30, 8, 963-975.
- Hill, D.O., and Shindala, A., (1977) Performance Evaluation of Kilmichael Lagoon . EPA-600/2-77-109, U.S. EPA, Cincinnati, Ohio.
- Hill, D.T., and Lincoln, E.P., (1981) Development and Validation of a Comprehensive Model of Large-Scale Production of Micro Algae . Agricultural Wastes, f, 43-64.
- Himmeblau, D.M., and Gloyna, E.F., (1972) Process Modelling of Biological Waste Treatment Final Report U.S. Army Medical Research and Development Command, NTIS: AD-739992, 76.
- Hodgson, H.T., (1964) Stabilization Ponds for a Small African Urban Area . J.W.P.C.F., 36, 1, 51-68.
- Hornberger, G.M., and Spear, R.C., (1981) An Approach to the Preliminary Analysis of Enviromental Systems . J. Envir. Mgmt., 12, 7-18.
- Huang, J., and Gloyna, E.F., (1968) Effect of Organic Compounds

- on Photosynthetic Oxygenation Part II. Design Modification for Waste Stabilization Ponds . Water Research, 2.
- Hussainy, S.U. (1979) Ecological Studies of Lagoons at Werribee: Removal of Biochemical Oxygen Demand, Nitrogen and Heavy Metals. , Prog. Wat. Tech. , 11,, 315-337.
- Ip, S.Y., Chin, C.T., Martin, W.R.B., and Raper, W.G.C., (1982) Algal Growth in Primary Settled Sewage . The Effect of Five Key Variables . Water Research, 16, No.5, 621-632.
- Iwugo, K.O, and Winnicki, R., (1976) A Study of Mixing Characteristics of Sewage Stabilization Ponds with Radioactive Tracers . Publ. Hlth. Engr., Vol.4, No.5, 138-142.
- Jayangoudar, I.S, Kathandaraman, V., et. al. (1970) Rationale Process Design Standards for Aerobic Ponds in Ahmedabad, India. J.W.P.C.F., 42, 8, 1501-1514.
- Joergensen, S.E., Nielsen, L.K., and Mejer, H.F., (1982) Comparision of a simple and a Complex Sediment Phosphrous Model . Ecological Modelling, Vol. 16, No. 2,3,1., pp 99-124.
- Johnston, P.R., and Pilgrim, D., (1976) Parameters Optimization for Watershed Models . Water Resources Res., 12, (3), 477-486.
- Joshi, S.R., Parhad, N.M., and Rao, N.U., (1973) Elimination of Salmonella in Stabilization Ponds . Water Research, 7, 1357-1365.
- Juitner, F., and Matuschek, T., (1978) The Release of Low Molecular Weight Compounds by the Phytoplankton in an Eutrophic Lake . Water Research, Vol 12, 251-255.
- Karplus, W.J. (1976) The Spectrum of Mathematical Modelling and Systems Simulation . ed: I. Delker Proceedings of 8th AICA Congress, Delft, Aug. 23-28, 5-15.
- Kawai, H., (1982) Waste Stabilization Pond Design in Brazil . Companhia de Tecnologia de Saneament Ambiental, Sao Paulo, Brasil.
- Kawai, H., Vano, T., Rocha, M.J.M., Limas, A., and Scheneiderman, B., (1981) Establishment of Design Criteria for Stabilization Ponds . Revista Dae, 41, No. 127, 37-45.
- Kilani, J.S., and Ogunrombi, J.A., (1984) Effects of Baffles on the performance of Model Waste Stabilization Ponds . Water Res., Vol. 18, No. 8, 941-944.
- King, D.I. (1976) Changes in Water Chemistry Induced by Algae ., In: Ponds as a Wastewater Treatment Alternative. (eds.) E.F. Glyona, J.F. Mailina Jr. and E.M. Davis. Water

Resources Symposium No. (. Univ. of Texas: Centre for Research in Water Resources.

- King, D.L., (1976) Changes in Water Chemistry Induced by Algae . In: Ponds as a Wastewater Treatment Alternative (ed.) Glyona et al., Water Res. Symposium No.9, University of Texas at Austin, Austin Texas.
- King, D.L. (1970) The Role of Carbon in Eutrophication . J.W.P.C.F., 42, 2035-2051.
- King, D.L., Tolmseeff, A.J., and Atherton, M.J., (1970) Effect of Lagoon Effluents on a Recieving Stream , Proc. 2nd Inter. Sym. for Waste Treatment Lagoons, Kansas City, Mo., 159-167.
- Klein, S.A, (1974) NTA Removal in Septic Tanks and Oxidation Pond Systems . J.W.P.C.F., Vol. 46, No.1, 78-88.
- Knudson, M.K., Williamson, K.J., and Nelson, P.O., (1982) Influence of Dissolved Oxygen on Substrate Utilization Kinetics of Activated Sludge . J.W.P.C.F., 54, No. 1, 52-60.
- Kucera, G., (1982) On the Relationship Between the Reliability of Parameter Estimates and Hydrologic Time Series Data used in Calibration . Water Resources Research, Vol 18, No. 1, 146-154.
- Larsen, T.B., (1974) A Dimensionless Design Equation for Sewage Lagoons . Ph.D. Thesis, University of New Mexico, Albuquerque.
- Legendre, P., Baleux, B., and Trossellier, M., (1984) Dynamics of Pollution-Indicator and Heterotrophic Batceria in Sewage Treatment lagoons . Applied and Enviromental
- Levenspiel, O., (1962) Chemical Reaction Engineering , Microbiology, 48, No.3, 586-593.
- Levins, R., (1966) The Strategy of Model Building in Population Biology . American Scientist, 54, 421-431.
- Loehr, R.C., and Ehreth, D.J. (1975) Oxidation Ponds.. Research Needs to Meet Effluent Limitations . Univ. of Texas Symposia. Ponds as a Wastewater Treatment Alternative. Austin, Texas, July 1975.
- Lumbers, J., and Andoh, B., (1985) Waste Stabilization Pond Design: Some Myths Behind the Science . Waterlines, Vol 3, No. 4, 12-14.
- Lumbers, J.P., (1979) Waste Stabilization Ponds Design Considerations and Methods J. Inst. Public Health Engineers, Vol 7, 22, 70-78.

- l umbers, J.P., (1982) Discussion: The Development of Design Equations for Facultative Waste Stabilization Ponds in Semi-Arid Areas by J.P. Arthur. Proceedings of the Institution of Civil Engineers, Part 2, Vol 73, 225.
- Mancini, J.L., (1978) Numerical Estimates of Coliform Mortality Rates Under Various Conditions . J.W.P.C.F., 50, 2477
- Mangelson, K.A., and Walters, G.Z., (1972) Hydraulic Treatment Efficiency of Waste Stabilization Ponds . J. of the Sant. Eng. Div. A.S.C.E., Vol. 98, No. SA2, 407.
- Mara, D.D., (1974) A Note on the Design of Facultative Sewage lagoons . Water Research, Vol. 8, 493-495.
- Mara, D.D., (1976) Sewage Treatment in Hot Climates . John Wiley and Sons, London.
- Mara, D.D., and Silva, S.A., (1982) Sewage Treatment in Waste Stabilisation Ponds: Recent Research in North-East Brazil . Progress in Water Technology, Vol 11, No. 2.
- Marais, G.V.R. (1971) Dynamic Behaviour of Oxidation Ponds . In: R.E. Mckinney (ed.) Proc. of the 2nd. Inter. Sym. for Waste Treatment Lagoons., Kansas City, Mo.
- Marais, G.V.R., (1966) New Factors in the Design, Operation and Performance of Waste Stabilization Ponds . Bull. Wld. Hlth. Org., 34, 737-763
- Marais, G.V.R., (1974) Faecal Bacteria Kinetics in Stabilization Ponds . Jour. Env. Eng. Div., ASCE, 100, EE1, 119-139.
- Martin, J.D., Dutcher, V.D., Frieze, T.R., Tapp, M., and Davis, E.M., (1976) Waste Stabilization Experiences at Union Carbide Seadrift, Texas Plant . In: Ponds as a Wastewater Treatment Alternative, Water Resources Symposium No.9. Centre for Research in Water Resources. University of Texas at Austin, 191-206.
- McGarry, M.G., and Bouthillier, P.H., (1966) Survival of S. Typhi in Sewage Oxidation Ponds . Journal of the Sanitary Engineering Division, ASCE, Vol. 92, No. SA4, 33-44.
- McGarry, M.G., Lin, C.D., and Merto, J.L., (1970) Photosynthetic Yields and Byproduct Recovery from Sewage Oxidation Ponds . Preprint: Presented at 6th International Water Pollution Research Conference, session 4, Hall C, Paper No.7, PC/4/7/1-c/4/7/11.
- McGarry and Pescod, M.D., (1970) Stabilization Pond Design Criteria for Tropical Asia . 2nd Int. Symp. for Waste Treatment Lagoons. Missouri River Basin Eng. Health Council, Kansas City, MO.

- McKinney, R.E., (1968) Overloaded Oxidation Ponds - Two Case Studies , J.W.P.C.F., 40, 1, 49-56.
- McKinney, R.E., (1976) Functional Characteristics Unique to Ponds: In Ponds as a Wastewater Treatment Alternative. Gloyna et al. (eds), Univ. of Texas, Austin.
- McKinney, R.E., (1977) Performance Evaluation of an Existing lagoon System at Eudora, KS. . EPA-600/2-77-157, U.S. EPA, Cincinnati, Ohio.
- Maybeck, P.S., (1979) Stochastic Models, Estimation and Control . Vol1, Academic Press, N.Y., 423.
- Mein, R.G., and Brown, B.M., (1978) Sensitivity of Optimized Parameters in Watershed Models . Water Resour. Res., 14, (5), 229-303.
- Meron, A., Rebhun, M., and Sless, J.B. (1965) Quality Changes as a Function of Detention Time in Wastewater Stabilization Ponds ., J.W.P.C.F., 37, 1657-1670.
- Metcalf and Eddy Inc. (1972) Wastewater Engineering . McGraw-Hill, New York.
- Middlebrooks, E.J., et al., (1982) Wastewater Stabilization Lagoon Design, Performance and Upgrading . Macmillan Publishing, New York, N.Y.
- Middlebrooks, E.J., Panagiotou, A.J., and Williford, H.K., (1967) Sludge Accumulation in Municipal Sewage Lagoons , Water and Sewage Works, 112, 2, 63-68.
- Middlebrooks, E.J., and Reed, S.C., (1981) The Flowering of Waste-water Treatment . Water/Engineering & Management, 1128, No. 6, 51-54.
- Middleton, F.M. and Bunch, R.L., (1970) Challenge for Wastewater Lagoons , Proc. 2nd Inter. Sym. for Waste Treatment Lagoons, Kansas City, Mo., 364-366.
- Miller, S., and Belfort, A.A., (1977) Effects of High Organic Loading on Mixed Photosynthetic Wastewater Treatment . J.W.P.C.F., 49, No.3, 436-446.
- Mitchell, D.B., (1980) The Ecology of WasteStabilization Ponds . PhD. Thesis, University of Adelaide, Australia.
- Mitchell, B.D., and Williams, W.D., (1982) The Performance of Tertiary Treatment Ponds and The Role of Algae, Macrophytes and Zooplankton in the Waste Treatment Process . Australian Water Resources Council Ocassional Papers Series No. 2 : Department of National Development and Energy, Australian Water Resources Council.

- Moeller, J.R., and Calkins, J., (1980) Bactericidal agents in waste-water lagoons and lagoon design . J.W.P.C.F., 52, No. 10, 2442-2451.
- Moore, E.W., Thomas, H.A., and Snow, W.B., (1950) Simplified Method for Analysis of BOD Data . Sewage And Ind. Wastes, 22, 1343.
- Moreno-Grav, M.D., Soler, A., Saez, J., and Romera, P., (1984) Study on the Hydrodynamic Behaviour of a deep Waste Stabilization Pond . Tribune du Cebedeau, 37, No. 489/490, 323-328.
- Mortimer, C.H., (1971) Chemical Exchanges Between Sediments and Water in the Great Lakes- Speculations on Probable Regulatory Mechanisms . Limnology and Oceanography, 16, 387-404.
- Moshe, M., Beetzer, N., and Kott, Y. (1972) Effect of Industrial Wastes on Oxidation Pond Performance . Water Research, 6, 1165-1171.
- Murphy, K.R., and Timpany, P.L., (1967) Design and Analysis of Mixing for an Aeration Tank . Jour. San. Eng. Div., ASCE, 93, SA5, Proc. Paper 5496, 1-15.
- Myers, F.A., and Williams, T.C., (1970) A Decade of Stabilization Lagoons in Michigan with Irrigation as Ultimate Disposal of Effluent . In: R.E. McKinney (ed.) Proceedings of the 2nd International Symposium for Waste Treatment Lagoons. Kansas City, Missouri Basin Engineering Health Council and Federal Water Quality Administration.
- Myklebust, R.J., and Harmston, F.C., (1962) Mosquito Production in Stabilization Ponds . J.W.P.C.F., 34, 3, 302-306.
- Neel, J.K., McDermoth, J.H. and Monday, C.A., (1961) Experimental Lagooning of Raw Sewage at Fayette, Missouri . J.W.P.C.F., 33, 6, 603-641.
- Nichols, H.W., (1973) Growth Media for Freshwater Algae: In Handbook of Phycological Methods in Culture Methods and Growth Measurements . ed. J.R. Stein, Cambridge Press.
- Novak, J.T., and Brune, D.E., (1985) Inorganic Carbon limited Growth Kinetics of some Freshwater Algae . Water Res., Vol. 19, No. 2, 215-225.
- Odum, E.P., (1971) Fundamentals of Ecology . Saunders, Philadelphia. 574 pp.
- Ong, S.L., and Lee, G.S., (1985) Data Analysis via Generalised Continuous Parabolic Interpolation . Adv. Eng. Software, Vol. 7, No. 3, 133-136.
- Oron, G., and Shelf, G., (1982) Maximizing Algal Yield in

- High-rate Oxidation Ponds . J. Env. Eng. Division, A.S.C.E., Vol. 108, EE4, 730-738.
- Oswald, W.J., (1960) Fundamental Factors in Stabilization Pond Design . paper No. 44, Conference on Biological Waste Treatment, 357-393.
- Oswald, W.J., (1968) Basis for Waste Stabilization Pond Design. In: Gloyna, E.F., and Eckenfelder, W.W., eds. Adv. in Water Supplies, Washington D.C., U.S. Public Health Service (Publ. No. 657)
- Oswald, W.J., (1972) Complete Waste Treatment in Ponds . Low Cost Waste Treatment Systems, 6th International Conference on Water Pollution Research, Jerusalem, Israel, 153-163.
- Oswald, W.J., (1973) Productivity of Algae in Sewage Disposal . Solar Energy, Vol. 15, 107-117.
- Oswald, W.J., (1975) Waste Pond Fundamentals . Part of an International Bank for Reconstruction and Development Training Programme, USA, 65p.
- Oswald, W.J., et al. (1957) Algae in Waste Treatment . Sew. and Indust. Wastes, No. 29, 437.
- Oswald, W.J., et al. (1970) Designing Waste Ponds to Meet Water Quality Criteria . Proc. 2nd Inter. Sym. for Waste Treatment Lagoons, Kansas City, Mo., 186-193.
- Oswald, W.J., and Gotaas, H.B., (1957) Photosynthesis in Sewage Treatment. ASCE, Transactions, Vol 122, 73-105.
- Oswald , W.J., Gotaas, H.B., Golueke, C.G., and Kellen, W.R., (1958) Algae in Waste Treatment . Sewage and Industrial Wastes, Washinton, D.C., Vol. 29, 437-57.
- Oswald, W.J., and Ramani, R., (1976) The Fate of Algae in Receiving Waters . In: E.F. Glyona et al (eds) Ponds as a Wastewater Treatment Alternative. Water Resources Symposium No.9, Univ. of Texas, Centre for Research in Water Resources.
- Pano, A., and Middlebrooks, E.J., (1982) Ammonia Nitrogen Removal in Facultative Wastewater Stabilization Ponds . J.W.P.C.F., 54, 344.
- Parhad, N.M., and Rao, N.V., (1974) Effects of pH on Survival of Escherichia coli . J.W.P.C.F, Vol. 46, No. 5, 980-986.
- Parhad, N.M., and Rao, N.V., (1976) Decrease of Bacterial Content of Different Types of Stabilization Ponds . Indian J. Environ Hlth., 18, 1, 33-46.
- Parker, C.D., (1962) Microbiological Aspects of Lagoon Treatment . J.W.P.C.F., 134, 149.

- Patrick, R., (1976) The Effect of a Satbilization Pond on the Sabine Estuary . In: E.F. Glyona et al (eds) Ponds as a Wastewater Treatment Alternative. Water Resources Symposium No.9, Univ. of Texas, Centre for Research in Water Resources.
- Pattern, B.C., and Cale, W.G., (1974) Propagation of cause in Ecosystems . In System Analysis and Simulation in Ecology, Vol. 4. (B.C. Pattern ed.), Academic Press - 1971-76.
- Poduska, R.A., (1981) Successful storage lagoon odour control. J.W.P.C.F., 53, No. 3, 299-310.
- Polprasert, C., and Bhattarai, K.K., (1985) Dispersion Model for Waste Stabilization Ponds . Journal of Enviromental Engineering, Vol. 111, No.1, 45-59.
- Polprasert, C., Dissanayake, M.G., and Thanh, N.C., (1983) Bacterial Die-Off Kinetics in Waste Stabilization Ponds . J.W.P.C.F., Vol. 55, No. 3,
- Proceedings of Waste Stabilization Lagoons Symposium, Kansas City, Mo., August 1-5, 1960, PHS Publication 872, (August, 1961).
- Rafi Ahmed, S.M., (1980) Choice and layout of various types of stabilization ponds . WHO/EMRO Technical Publication No. 3, pp 133-154., Report of a seminar on waste stabilization pond design and operation., Lahore.
- Randall, C.W., and Buth, D., (1984) Nitrite Build-up in Activated Sludge Resulting from Temperature Effects . J.W.P.C.F., Vol 56, No. 9.
- Reddy, K.R., (1983) Nitrogen and Phosphrous Interchange Between Sediments and Overlying Water of a Wastewater Retention Pond . Hydrobiologia, 98, No. 3, 237-243.
- Reynolds, J.H., Nelson, S.B., Middlebrooks, E.J., (1975) Biomass Distribution and Kinetics of Baffled Lagoons . Journal of the Enviro. Eng. Div., proceedings of ASCE., Vol. 101, EE6.
- Reynolds, J.H., et al., (1977) Performance Evaluation of an Existing Seven Cell Lagoon System . EPA-600/2-77-086, U.S. EPA, Cincinnati, Ohio .
- Rich, L.G., (1982) Benthall Stabilization: A Low Cost Process for the Stabilization of Waste Activated Sludge Solids . Water Research, 16, No. 9, 1399-1403.
- Rich, L.G., and Connor, B.W., (1982) Benthall Stabilization of Waste Activated Sludge . Water Research, 16, No. 9, 1419-1424.

- Richardson D.L., and Daigger, G.T. (1984) Aquaculture: The Hercules Experience . Journal of Enviromental Engineering Vol 110, No.5, Oct. 1984
- Rosenbrock, H.H., (1960) An Automatic Method of Finding the Greatest or Least Value of a Function . Computer Journal, Vol. 3, 175-183.
- Schneider, R.W., Middlebrooks, E.J., and Sletten R.S., (1983) Cold Region Wastewater Lagoon Sludge Accumulation . Water Res., Vol. 17, No. 9, 1201-1206.
- Searle, S.S., and Kirby, C.F., (1973) Waste into Wealth . Water Spectrum, 4, 15-21.
- Serfling, S.A., and Alsten, C (1978) An Intergrated Controlled Enviroment Aquaculture Lagoon Process for Secondary or Advanced Wastewater Treatment . In: Proceedings of a Conference held August 23-25 1978, at Utah State University Logan, Utah entitled. Performance and Upgrading of Wastewater Stabilization Ponds.
- Shelef, G., Wachs, A.M., and Zoref, A., (1975) Aging of Stabilization Ponds in Series . Progress in Water Technology, Vol. 7, No. 3/4, 715-723.
- Siddiqui, R.H., and Handa, B.K., (1971) Evaluation of some Stabilization Ponds in India . J. Sanit. Eng. Div., Proc. A.S.C.E., 97, SA 1, 91-102.
- Spies, R., (1984) Benthic-Pelagic Coupling in Sewage-Affected Marine Ecosystems . Marine Enviromental Research, 13, No. 3, 195-230.
- Stahl, J.R., and May, D.S. (1967) Microstratification in Waste Treatment Ponds . J.W.P.C.F., Vol. 39, No.1, Jan. pp 72-88.
- Stendahl, K., (1982) Does Conventional Sewage Treatment meet the demands of the 1980's . Effluent and Water Treatment Journal, 22, No. 2, 61-64.
- Strom, A.G., (1979) Use of Sewage as a Resource in Australia. Search, 10, 136-142.
- Tariq, M.N., and Ahmad, K., (1985) Treatment of Wastewater by Waste Stabilization Ponds in Developing Countries . Asian Environ., Vol. 6, No. 5, 15-19.
- Thirumurthi, D., (1967) A New Approach for Designing Waste Stabilization Ponds . Water and Sewage Works, Chicago, Ill., Vol. 114, 208-218.
- Thirumurthi, D., (1969) Design Principles of Waste Stabilization Ponds. ASCE, Sanitary Eng. Div., 95, SA 2, 311-329.

- Thirumurthi, D., (1974) Design Criteria for Waste Stabilization Ponds . J.W.P.C.F., 46, 9, 2094-2106.
- Towne, W.W., Bartsch, A.F., Davis, W.H., and Graves, Q.B., (1956) Waste Stabilization Ponds - A Statment of Cooperative Studies in Progress by the Public Health Service , Robert A. Taft Sanitary Engineering Center.
- Tschortner, U.S., (1968) Biological Parameters for the Operation and Control of Oxidation Ponds - II . Water Research, Vol. 2, 327-346.
- Uhlman, D., (1979) BOD removal rates of waste stabilization ponds as a function of loading, retention time, temperature and hydraulic flow pattern . Water Res., 13, 193-200.
- Walker, J.F., and Leclerc, H. (1975) Experimental Lagoon Purification of Surface Water: Ecology and Biochemical Aspects . Progress in Water Tech., Vol. 7, No. 3/4, 701-713.
- Walker, R.R., and Snodgrass, W.J., (1986) Model for Sediment Oxygen Demand in Lakes . Journal Env. Engr., ASCE, Vol 112, No. 1, 25-43.
- Walker, W.W., (1982) A Sensitivity and Error Analysis Framework for Lake Eutrophication Modelling . Water Resources Bulletin, 18, No. 1, 63-68
- Ward, C.H., and King, J.M., (1976) Fate of Algae in Laboratory Cultures . In. Ponds as a Wastewater Treatment Alternative. Water Resources Symposium No.9. Centre for Research in Water Resources, University of Texas at Austin. 87-103.
- Ward, R.C., and Loftis, J.C., (1983) Incorporating the stochastic Nature of Water Quality into Management . J.W.P.C.F., Vol. 55, No. 4, 408-414.
- Wehner, J.f., and Wilhelm, R.H., (1956) Chem. Eng. Sci., 6, 89.
- Wennstrom, M., (1955) Oxidation Ponds in Sweden . Lunds Universitets Arsskrift, Lund Sweden.
- Wiess, C.M., (1969) Relation of Phosphates to Eutrophication . Jour. Amer. Water Works Assn., 61, 387
- Willford, H.K. and Middlebrooks, E.J., (1967) Performance of Field Scale Facultative Wastewater Treatment Lagoons . J.W.P.C.F., 39, 12, 2008-2019.
- Wong, K.K., and Springer, A.M., (1981) A First-Order Kinetic Model for Designing Anaerobic Ponds in the Treatment of Palm Oil Mill Effluents . Agricultural Wastes, 3, 35-42.

- Wright, J.C. (1960) The Limnology of Canyon Ferry Reservoir III: Some Observations on The Density Dependence of Photosynthesis and It's Cause . Limnol. & Oceanog., 5, 356.
- Wright, J.C., and Mills, I.K., (1967) Productivity Studies on the Madison River Yellowstone National Park . Limnol. & Oceanog., 12, 568.
- Yair, F., and Wacks, A.M., (1973) Removal of Algae from Stabilization Ponds and Effluents by Lime Treatment . Water Research, 1973, 7, 419-435.
- Young, P.C., (1978) General Theory of Modelling Badly Defined Systems . In: Modelling Identification and Control in Environmental Systems ed. G.C., Vansteenkiste, IFIP, North-holland, 103.
- Young, P.C., Jakeman, A.J., and McMurtrie, R.E., (1980) An Instrumental Variable Method for Model Order Identification . Automatica, 16, 281-294.
- Young, P.C., Shellswell, S.H., and Neethling, C.G., (1971) A Recursive Approach to Time-Series Analysis . Technical Report, CVED/8- Control/TR16., Department of Engineering, Univ. of Cambridge, England.
- Young, T.C., and King, D.L., (1980) Interacting limits to algal growth: light, phosphorous, and carbon dioxide availability . Water Research, Vol. 14, 409-412.

TABLE 2-1

TYPICAL CHARACTERISTICS OF WASTEWATER STABILIZATION PONDS

<u>Pond Type</u>	<u>Application</u>	<u>Typical Loading Parameters</u>	<u>Typical Detention Times</u>	<u>Typical Dimensions</u>	<u>Comments</u>
Anaerobic	Industrial wastes	160-800 kg BOD ₅ /1000 m ³ /d	20-50 d	2.5-5 m deep	Odor production usually a problem; Subsequent treatment normally required
Facultative	Raw municipal wastewater Effluent from primary treatment, trickling filters, aerated ponds, or anaerobic ponds	22-67 BOD ₅ /ha/d	25-180 d	1.2-2.5 m deep 4-60 ha	Most commonly used waste stabilization pond type May be aerobic through entire depth if lightly loaded
Aerobic	Generally used to treat effluent from other processes, produces effluent low in soluble BOD ₅ and high in algae solids	85-170 kg BOD ₅ /ha/d	10-40 d	30-45 cm	Application limited because of effluent quality Maximizes algae production and (if algae is harvested) nutrient removal High loadings reduce land requirements
Aerated	Industrial wastes Overloaded facultative ponds Situations where limited land area is available	8-320 kg BOD ₅ /1000 m ³ /d	7-20 d	2-6 m deep	Use may range from a supplement of photosynthesis to an extended aeration activated sludge process Requires less land area than facultative

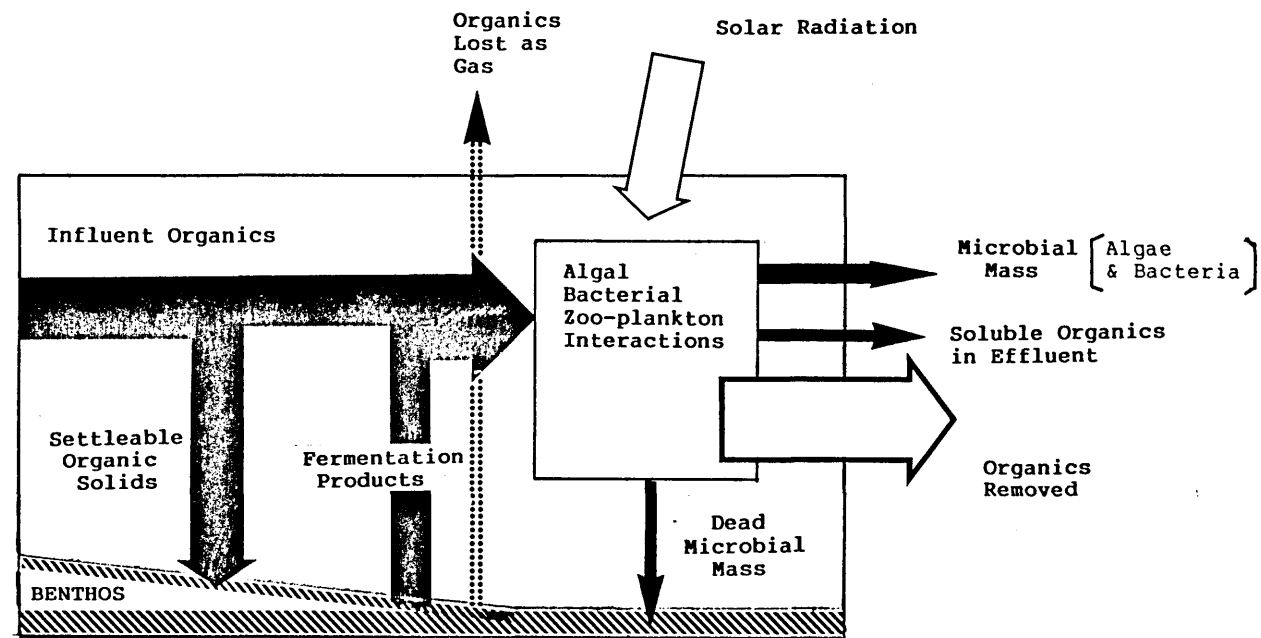


Fig 2.1 Principal pathways of Organics removal in facultative ponds

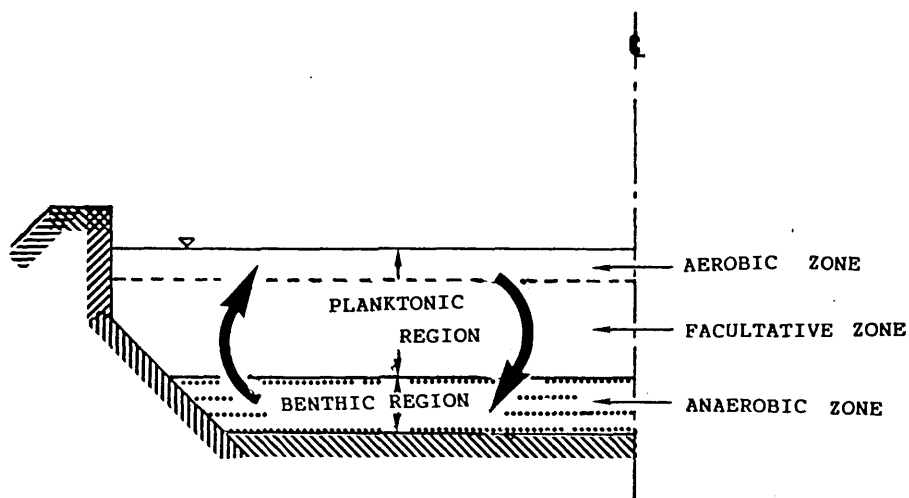


Figure 2.2: Zones in a typical facultative pond system

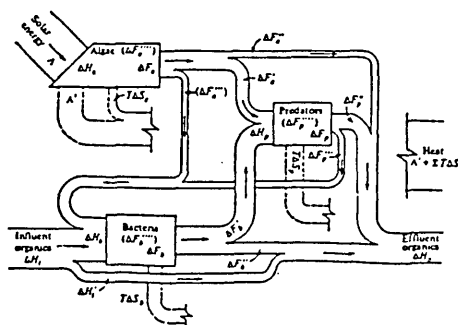


Figure 2.3: Energy Flow through the Biotic Component of Pond Systems
(After Rich, 1970)

TABLE 2.2 Stoichiometry of Domestic Sewage and Algae
SEWAGE[‡]

NUTRIENT	mg/l	Millimole/Litre
Organic Carbon	66-176	5.5-14.7
Total Nitrogen (NH_4^+ , NO_3^- , NO_2^- , organic-N)	20-50	1.4-3.5
Soluble Phosphorous	1-13	0.03-0.41
MEAN STOICHIOMETRIC RELATION, ATOMS C:N:P = 70:17:1		
ALGAE*		
MEAN STOICHIOMETRIC RELATION, ATOMS C:N:P = 106:16:1		

[‡] After: Fitzgerald and Rohlich (1958)

* After: Hoover, S.R. (1954)

TABLE 2.3 Common Algae Types in Pond Systems

CLASSIFICATION		COMMON TYPES
CYANOPHYTA [‡] (Blue-green)		Oscillatoria Anabena Anacystis
CHLOROPHYTA Green Algae)	Non Motile Green	Chorella Ankistrodesmus Microactinium Scenedesmus Actinastrus
	Motile Green (Pigmented Flagellates)	Euglena Phacus Carteria Chlamydomonas Pandorina
CHRYSTOPHYTA		Navicula Nitzchia

[‡] Related to high pH and high Temperatures

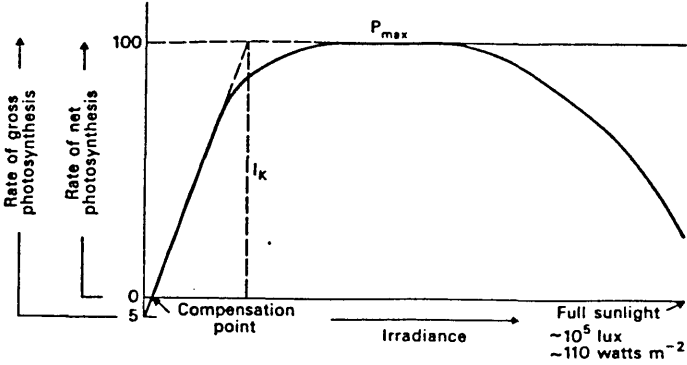


Figure 2.4: Typical Response of Algae to Irradiance

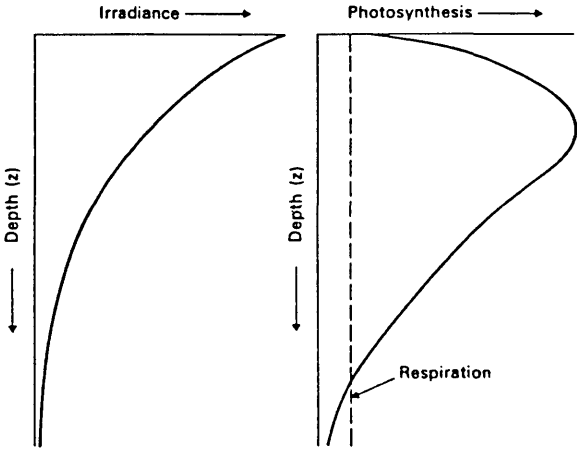


Figure 2.5: Light Penetration and Depth Profile of Photosynthesis

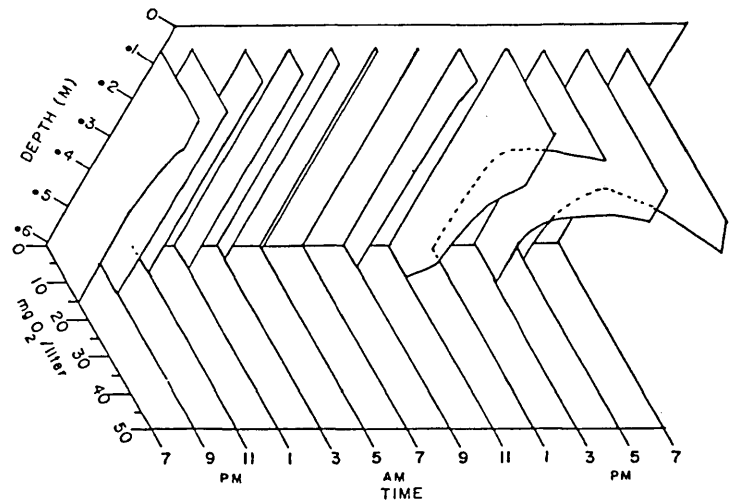


Figure 2.6: Dissolved Oxygen as a Function of Time and Depth
(After King, 1976)

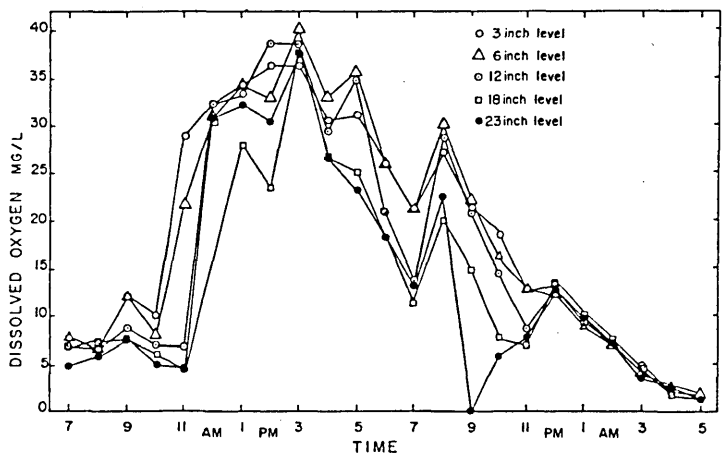


Figure 2.7 Diurnal Variation of Dissolved Oxygen at Five Levels of
(After King, 1976)

Table 2.4 : Diel Range in Temperature at
Different Depths in a Pond System
(After Stahl and May, 1967)

Depth Below Surface (cm)	Temperature Range °C
1	18.3
10	10.3
30	2.2
50	0.4

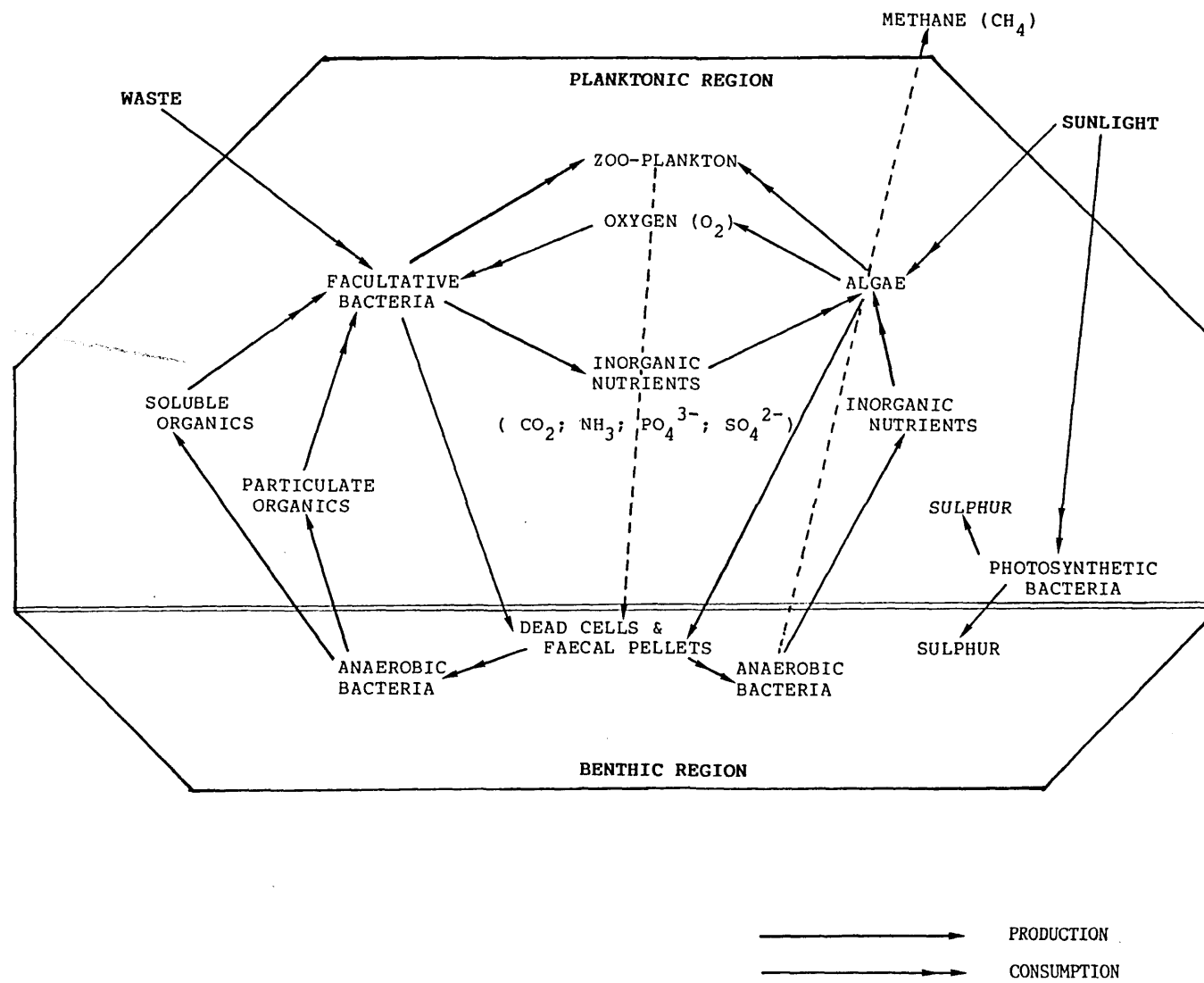


Fig 2.8 Schematic of Dominant Facultative Pond Mechanisms

TABLE 3.1 Characteristics of the four EPA Pond Systems

Facility	No of cells in series	Total Surface Area (ha)	Mean Depth (m)	Mean Hydraulic Residence Time (Days)		Areal Loading (Kg BOD ₅ /ha.d)	
				Design	Actual	Design	Actual
Corrine	7	3.9	1.2	180	70	36.2	14.2
Eudora	3	7.8	1.5	47	231	38.0	19.6
Peterborough	3	8.5	1.2	57	107	19.6	15.6
Kilmichael	3	3.3	2.0	79	214	67.2	27.2

TABLE 3.2: Characteristics of New-Mexico^I Pond Data

MEASURED CHARACTERISTIC	UNITS	MEAN	STANDARD DEVIATION	MAXIMUM	MINIMUM
Influent Total COD	mg/l	919.268	249.386	1665.000	370.000
Effluent Total COD	mg/l	408.178	115.508	854.000	156.000
Influent Filtered COD	mg/l	396.796	177.742	1286.000	121.000
Effluent Filtered COD	mg/l	142.944	54.151	433.000	19.000
Influent Total BOD	mg/l	524.534	237.783	1206.000	119.000
Effluent Total BOD	mg/l	129.670	56.216	321.000	21.000
Influent Total Solids	mg/l	1502.330	649.091	5015.000	669.000
Effluent Total Solids	mg/l	1113.486	156.018	1538.000	800.000
Influent Volatile Solids	mg/l	727.423	428.668	3674.000	227.000
Effluent Volatile Solids	mg/l	395.151	84.468	674.000	232.000
Influent Ortho - Phosphorous*	mg/l	73.581	27.918	294.000	18.800
Effluent Ortho - Phosphorous*	mg/l	36.924	10.509	65.000	16.500
Influent Organic Phosphorous*	mg/l	13.959	9.110	49.000	1.100
Effluent Organic Phosphorous*	mg/l	5.655	3.669	34.100	.100
Influent Ammonia*	mg/l	145.146	64.364	562.200	17.000
Effluent Ammonia*	mg/l	81.420	23.823	166.200	11.300
Influent Nitrite*	mg/l	1.612	1.601	10.400	.115
Effluent Nitrite*	mg/l	3.659	1.075	7.030	.030
Influent Nitrate*	mg/l	.388	.520	3.000	.020
Effluent Nitrate*	mg/l	.376	.171	.930	.100
Influent Flow	gals/Hr	1400.346	288.015	2391.482	502.682
Effluent Flow	gals/Hr	1157.435	422.314	3574.780	146.880
Air Temperature	°F	53.518	16.474	85.100	25.100
Pond Temperature	°F	54.943	11.733	74.400	37.700
Solar Radiation	BTU	2282.363	1459.838	7580.000	236.000
Evaporation	ins/Hr	.007	.005	.023	.001
Precipitation	ins/Hr	.002	.005	.045	.000
Wind	mph	9.172	4.769	33.500	1.700

^I Physical Properties:

Volume $\approx 6000 \text{ m}^3$
Depth $\approx 0.9 \text{ m}$

* Not a full year's record

TABLE 3.3:
Characteristics of Corrine[‡] Pond Data

MEASURED CHARACTERISTICS	UNITS	MEAN	STANDARD DEVIATION	MAXIMUM	MINIMUM
Influent Total COD	mg/l	137.590	93.848	707.000	15.400
Effluent Total COD	mg/l	122.154	42.889	324.000	24.300
Influent Soluble COD	mg/l	52.553	21.500	178.000	13.000
Effluent Soluble COD	mg/l	42.848	8.103	76.000	10.000
Influent Soluble BOD	mg/l	25.262	38.247	449.000	3.000
Effluent Soluble BOD	mg/l	5.682	3.047	29.900	1.800
Influent Total BOD	mg/l	76.521	52.936	337.400	1.500
Effluent Total BOD	mg/l	32.625	18.275	190.400	5.700
Influent Total Solids	mg/l	71.793	66.740	493.500	.100
Effluent Total Solids	mg/l	62.444	26.643	130.900	9.200
Influent Total Phosphorous	mg/l	4.085	1.808	9.500	1.100
Effluent Total Phosphorous	mg/l	3.384	.979	5.600	1.200
Influent Ammonia	mg/l	7.812	4.237	25.600	.300
Effluent Ammonia	mg/l	1.688	1.892	7.500	.000
Influent Nitrate	mg/l	1.089	1.200	5.200	.000
Effluent Nitrate	mg/l	.216	.266	1.200	.000
Influent Total Nitrogen	mg/l	15.248	7.231	37.800	.900
Effluent Total Nitrogen	mg/l	9.138	3.467	30.000	3.000
Precipitation	ins/Hr	.054	.147	1.080	.000
Solar Radiation	gcal/cm ² /d	395.069	219.023	795.600	27.000
Influent Flow	gals/Hr	7689.696	3762.916	15972.100	277.800
Effluent Flow	gals/Hr	6112.589	2942.343	14261.900	653.100
Pond Temperature	°C	10.061	7.352	26.000	-.100

‡ Physical Properties:-

Volume $\approx 17,880 \text{ m}^3$
Depth $\approx 1.2 \text{ m}$

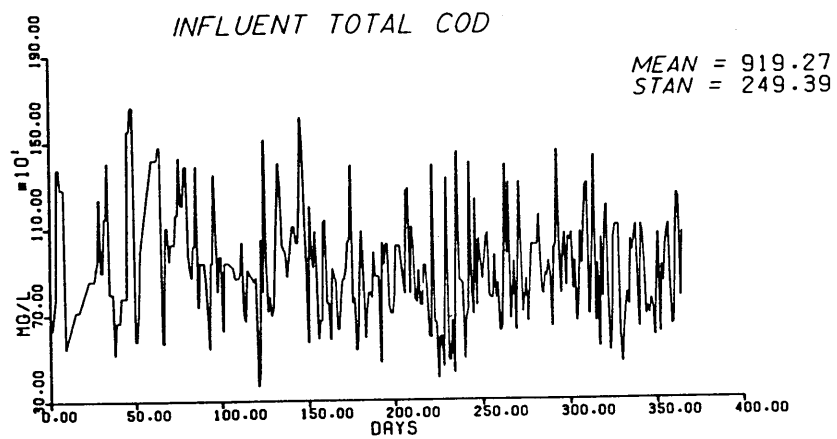


Fig 3.1 Pond 1 Influent Total COD (Raw Data)

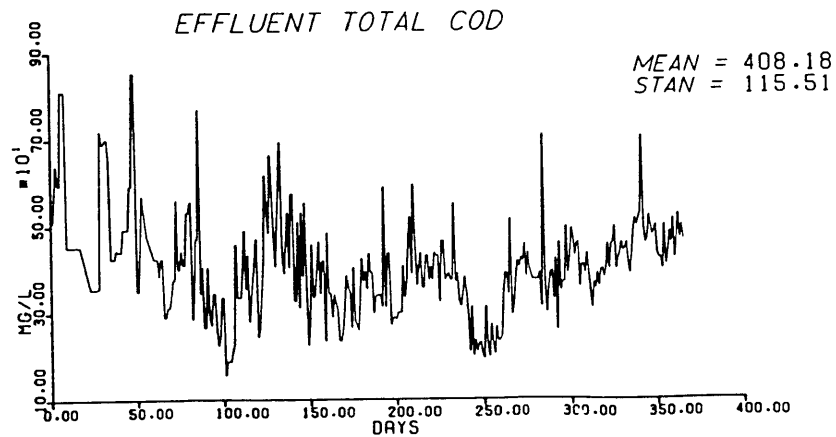


Fig 3.2 Pond 1 Effluent Total COD (Raw Data)

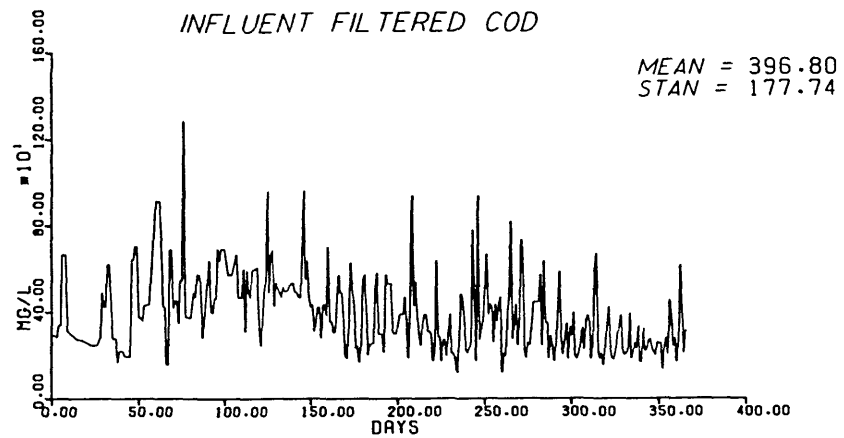


Fig 3.3 Pond 1 Influent Filtered COD (Raw Data)

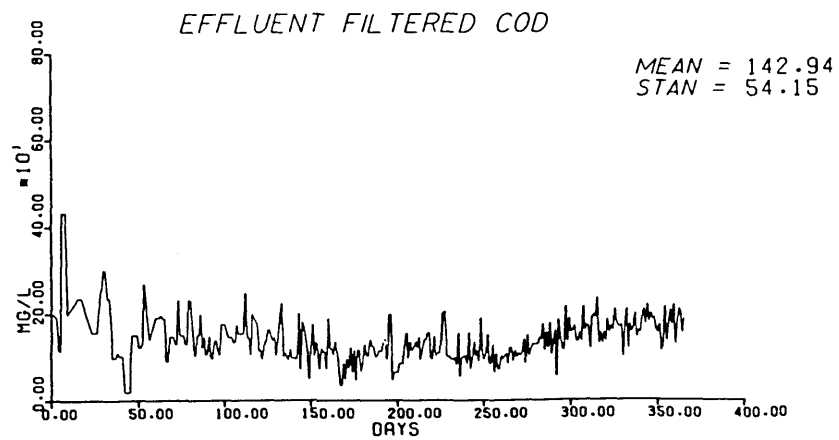


Fig 3.4 Pond 1 Effluent Filtered COD (Raw Data)

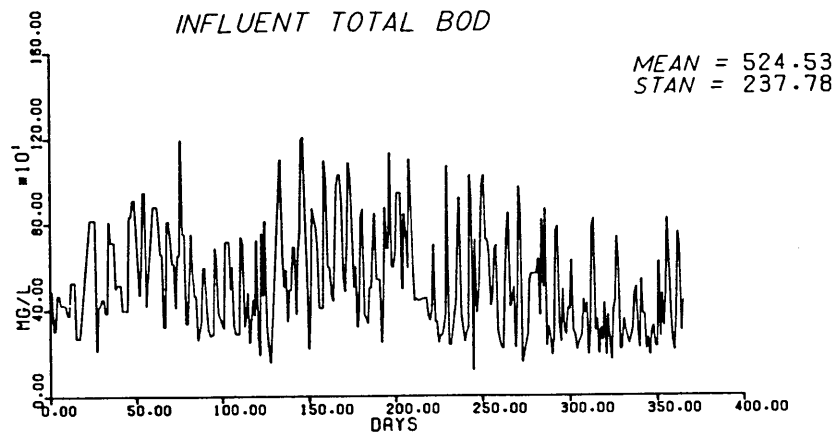


Fig 3.5 Pond 1 Influent Total BOD (Raw Data)

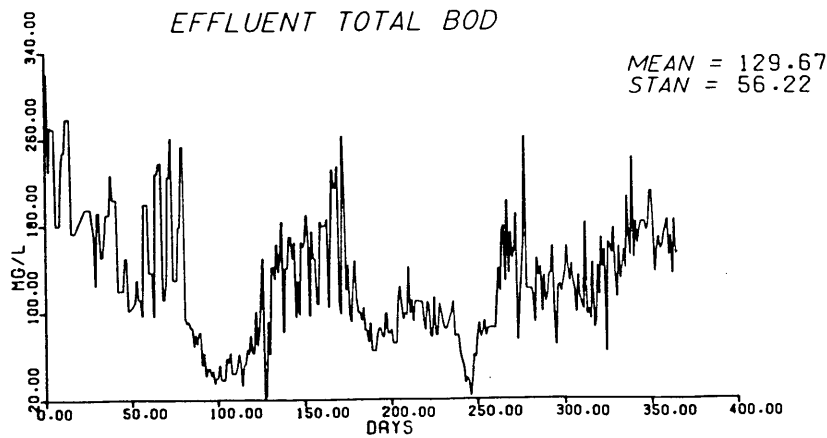


Fig 3.6 Pond 1 Effluent Total BOD (Raw Data)

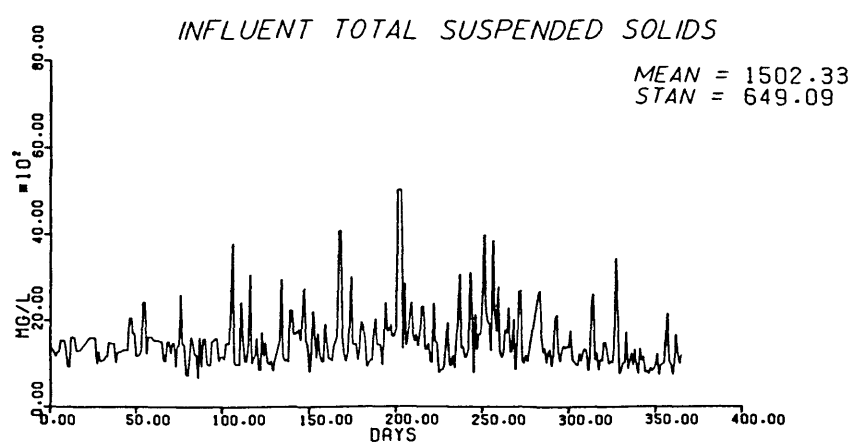


Fig 3.7 Pond 1 Influent Total Suspended Solids (Raw Data)

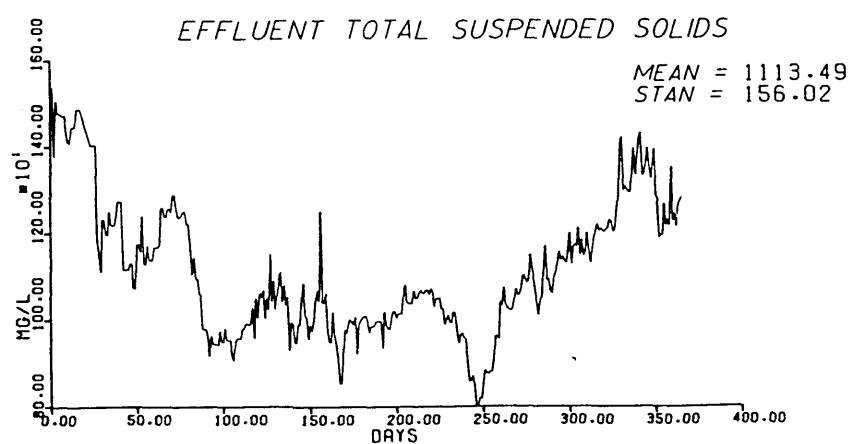


Fig 3.8 Pond 1 Effluent Total Suspended Solids (Raw Data)

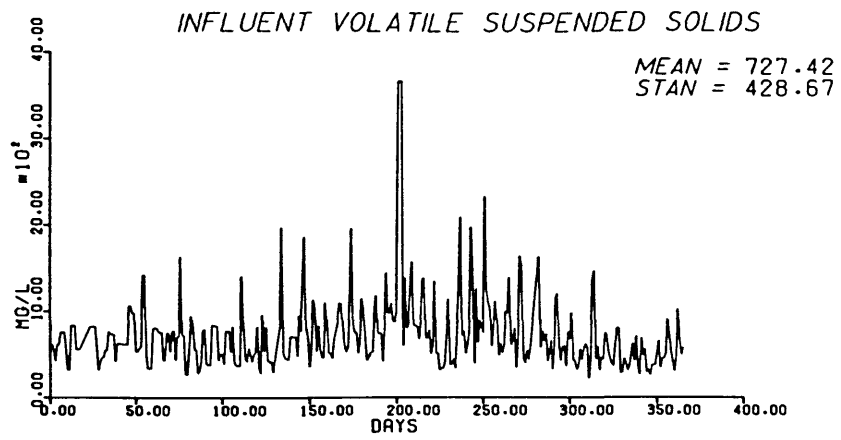


Fig 3.9 Pond 1 Influent Volatile Suspended Solids (Raw Data)

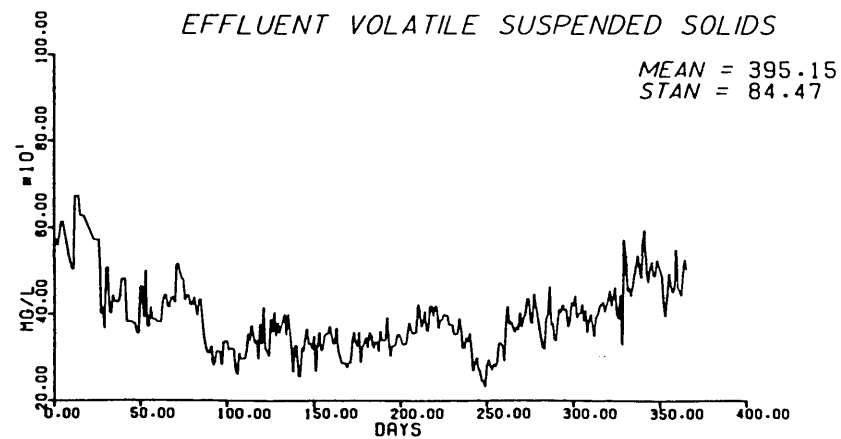


Fig 3.10 Pond 1 Effluent Volatile Suspended Solids (Raw Data)

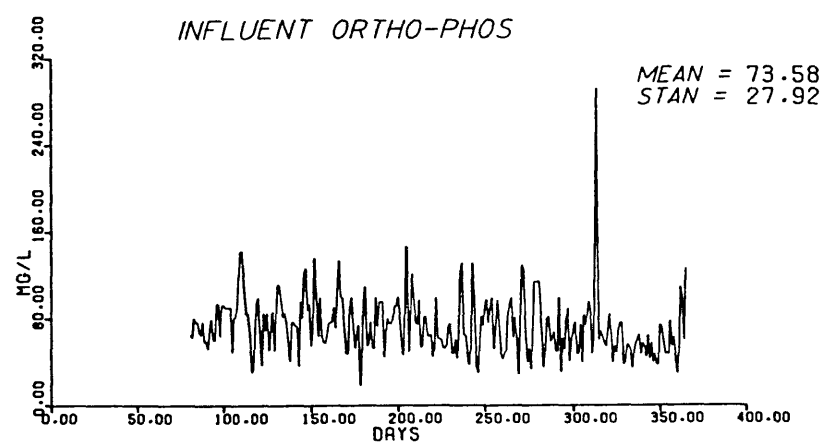


Fig 3.11 Pond 1 Influent Ortho-phosphorus (Raw Data)

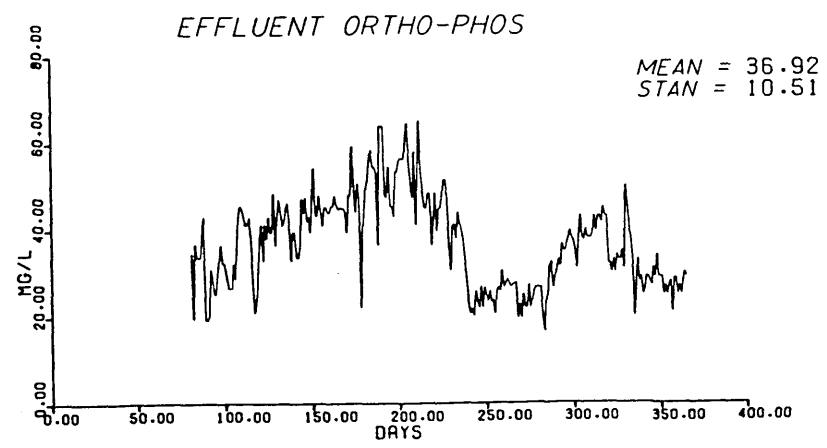


Fig 3.12 Pond 1 Effluent Ortho-phosphorus (Raw Data)

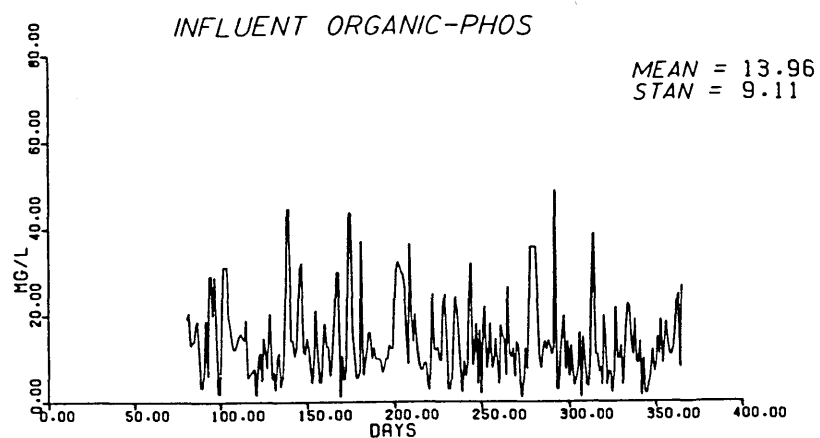


Fig 3.13 Pond 1 Influent Organic-phosphorus (Raw Data)

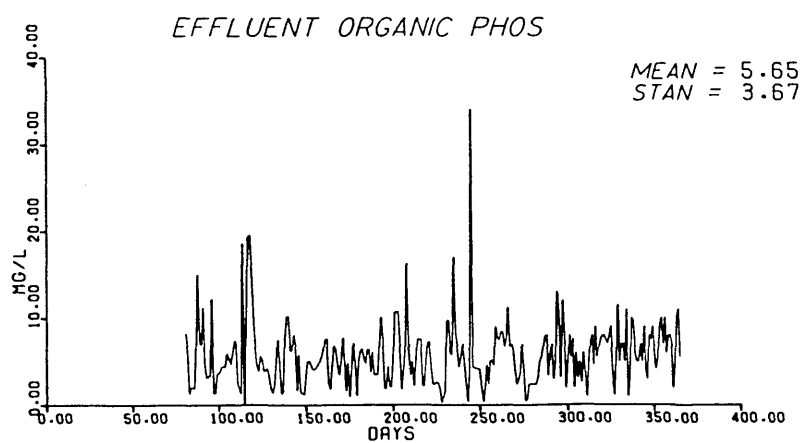


Fig 3.14 Pond 1 Effluent Organic-phosphorus (Raw Data)

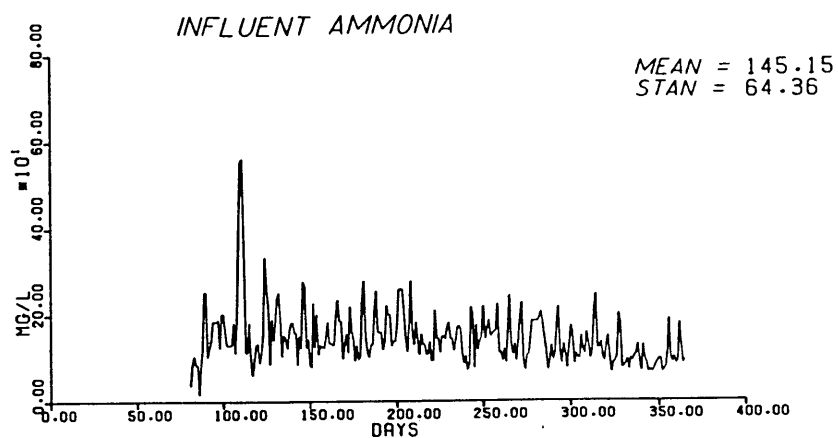


Fig 3.15 Pond 1 Influent Ammonia (Raw Data)

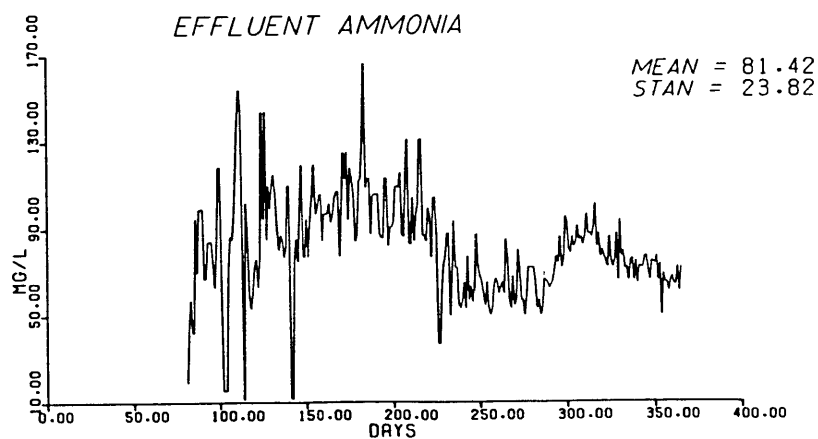


Fig 3.16 Pond 1 Effluent Ammonia (Raw Data)

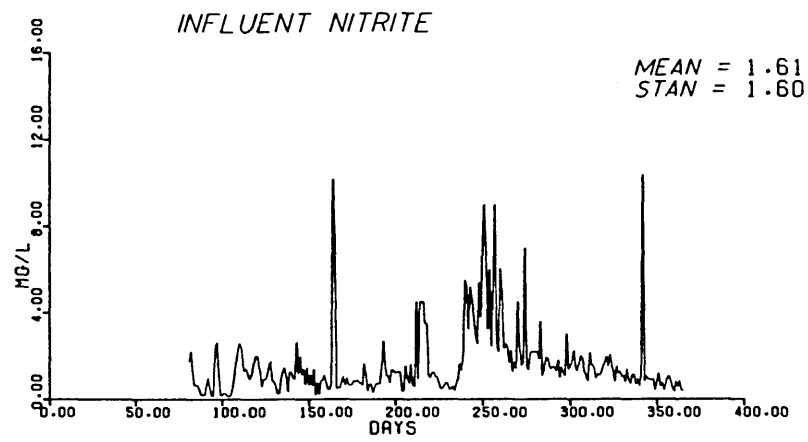


Fig 3.17 Pond 1 Influent Nitrite (Raw Data)

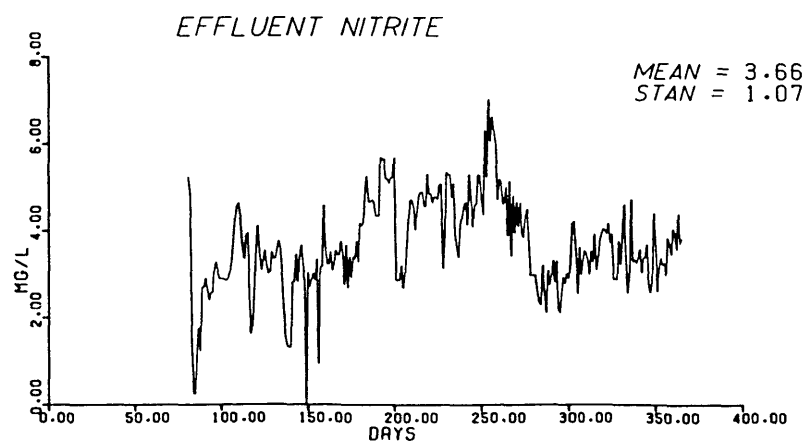


Fig 3.18 Pond 1 Effluent Nitrite (Raw Data)

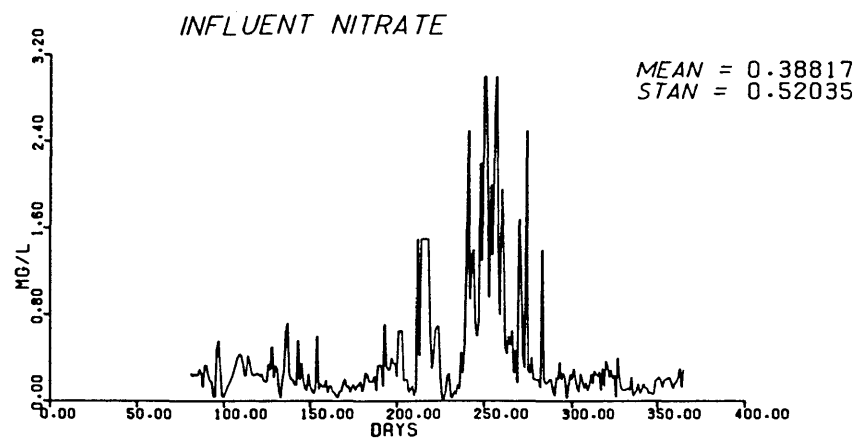


Fig 3.19 Pond 1 Influent Nitrate (Raw Data)

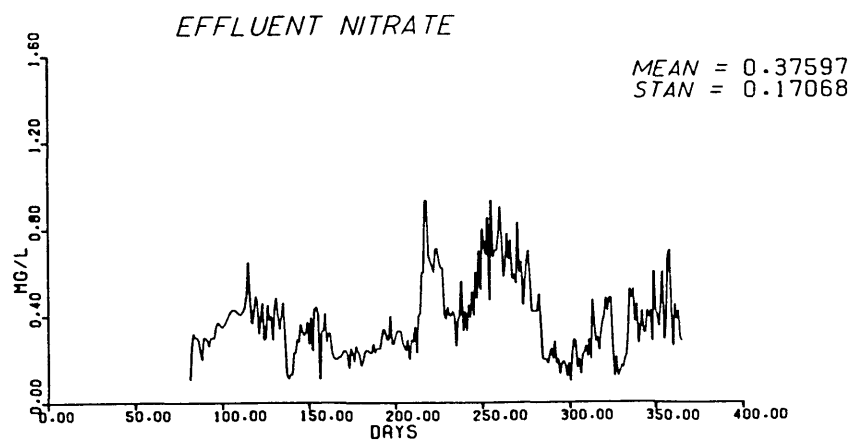


Fig 3.20 Pond 1 Effluent Nitrate (Raw Data)

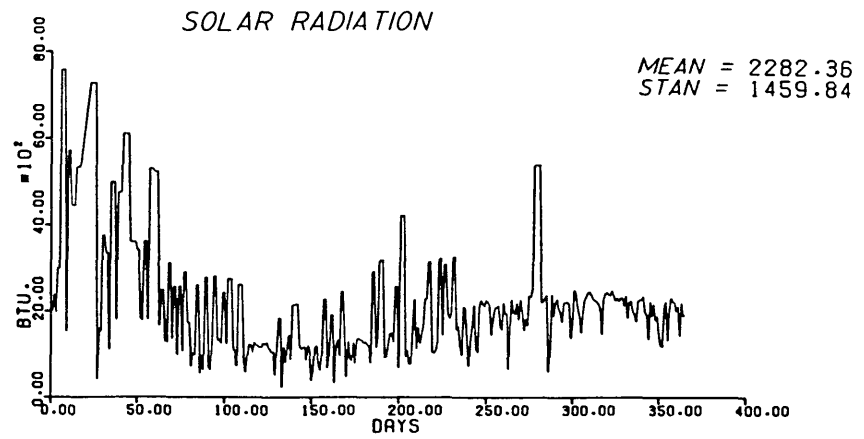


Fig 3.21 Pond 1 Daily Average Solar Radiation (Raw Data)

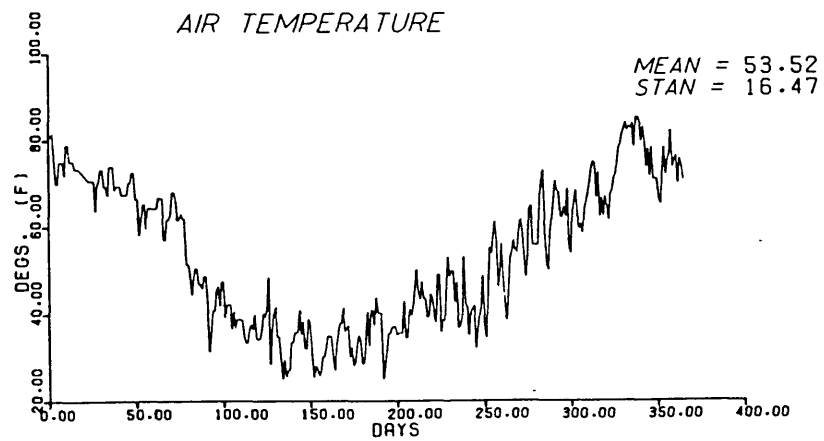


Fig 3.22 Pond 1 Daily Average Air Temperature (Raw Data)

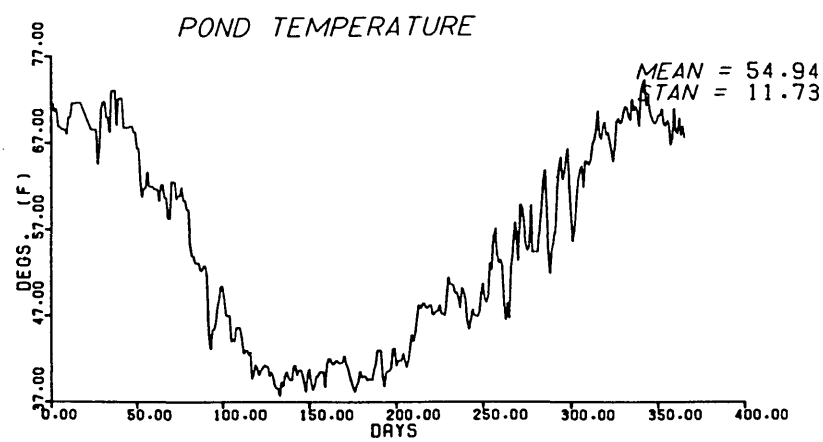


Fig 3.23 Pond 1 Daily Average Pond Temperature (Raw Data)

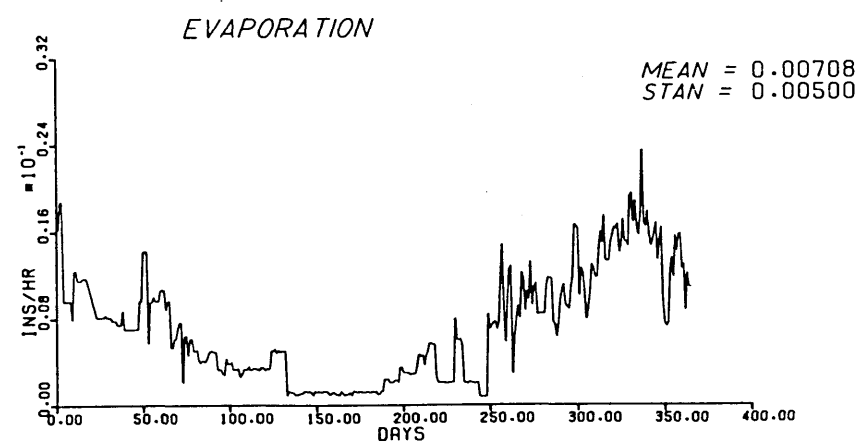


Fig 3.24 Pond 1 Daily Average Evaporation (Raw Data)

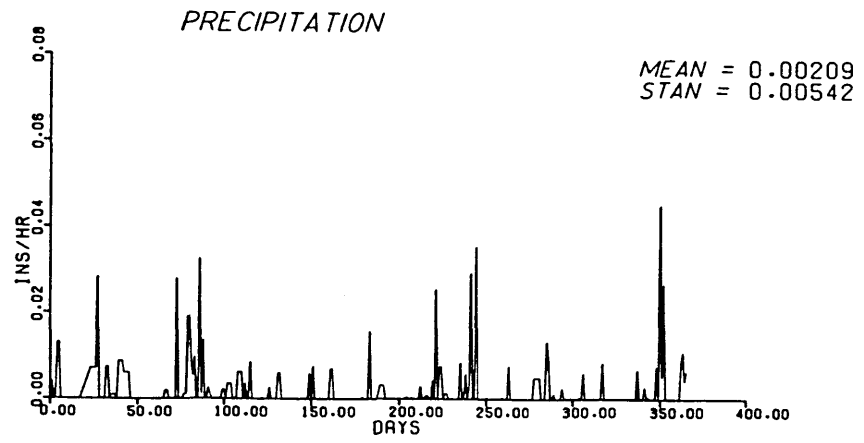


Fig 3.25 Pond 1 Daily Average Precipitation (Raw Data)

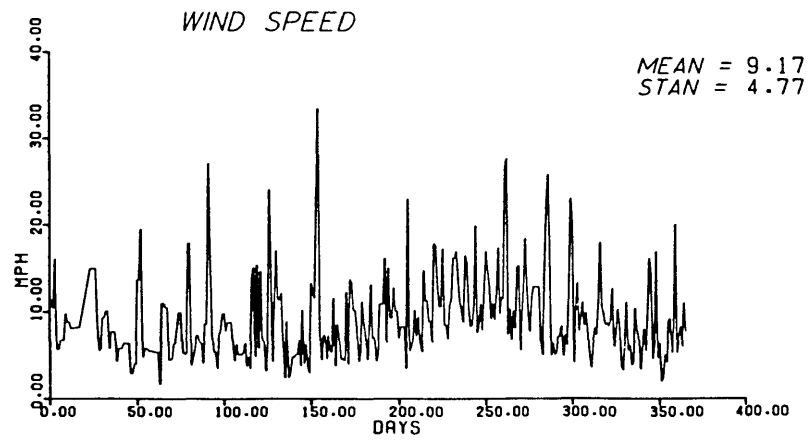


Fig 3.26 Pond 1 Daily Average Wind Speed (Raw Data)

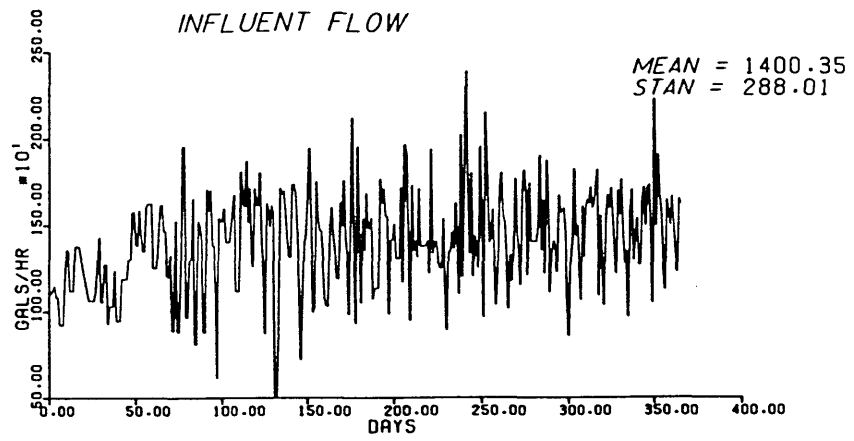


Fig 3.27 Pond 1 Influent Flow (Raw Data)

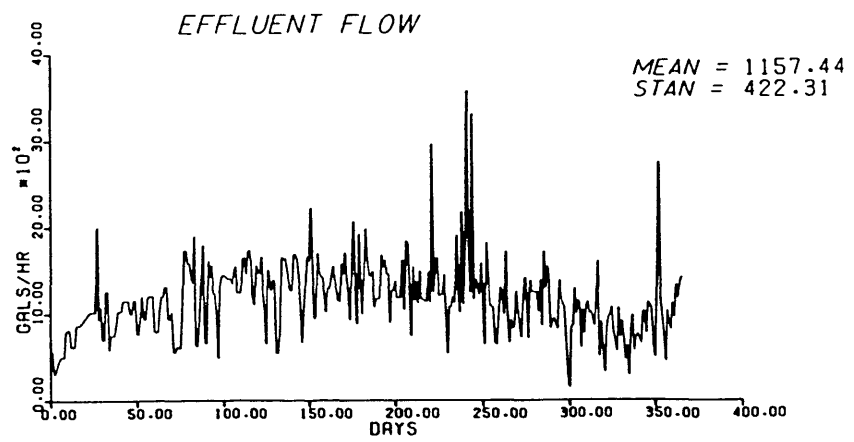


Fig 3.28 Pond 1 Effluent Flow (Raw Data)

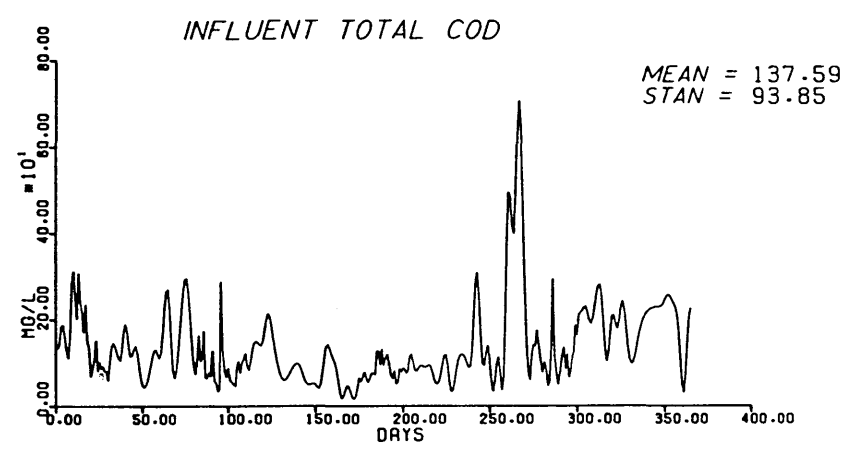


Fig 3.29 Pond 2 Influent Total COD (Raw Data)

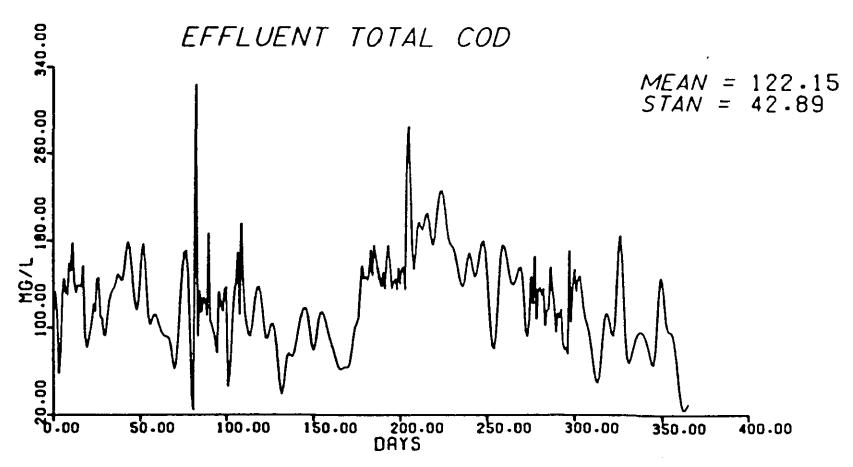


Fig 3.30 Pond 2 Effluent Total COD (Raw Data)

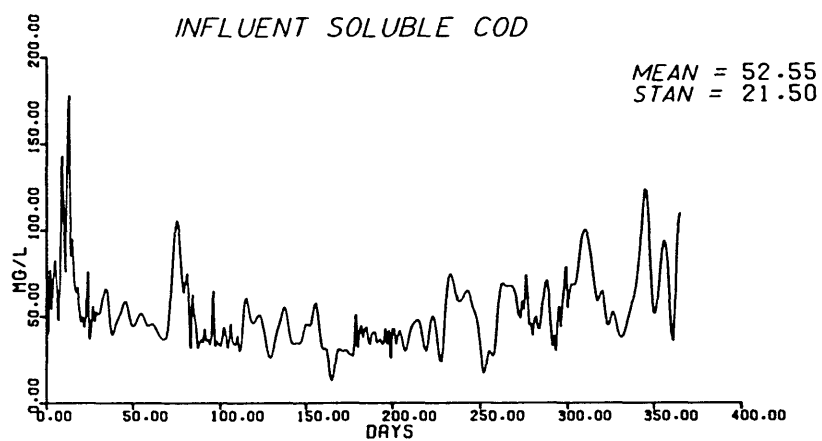


Fig 3.31 Pond 2 Influent Soluble COD (Raw Data)

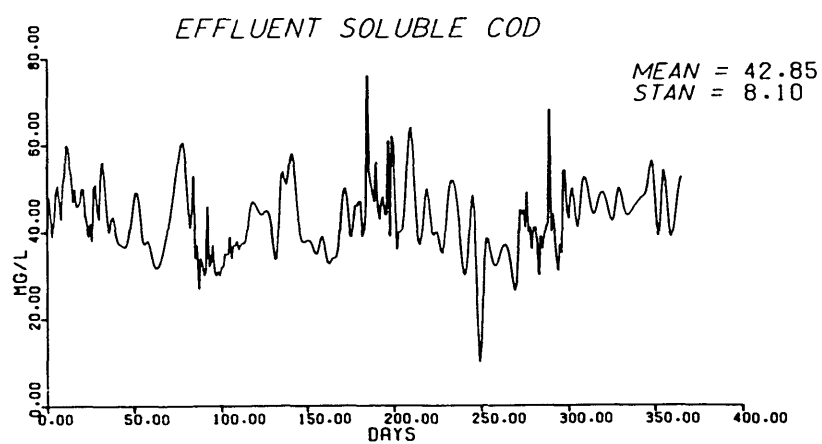


Fig 3.32 Pond 2 Effluent Soluble COD (Raw Data)

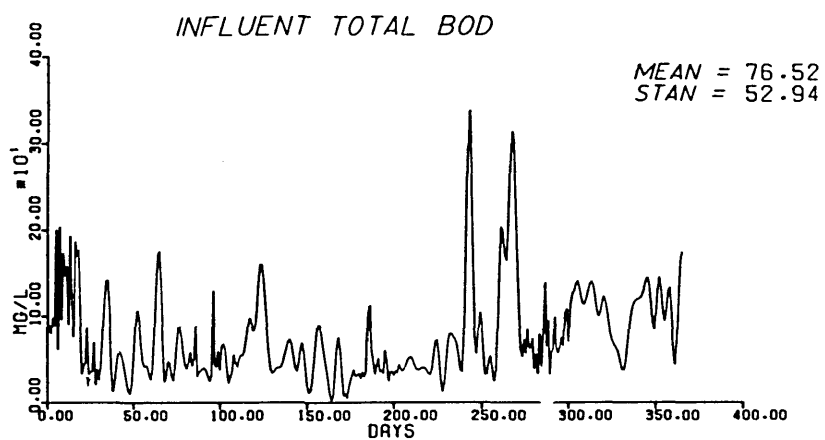


Fig 3.33 Pond 2 Influent Total BOD (Raw Data)

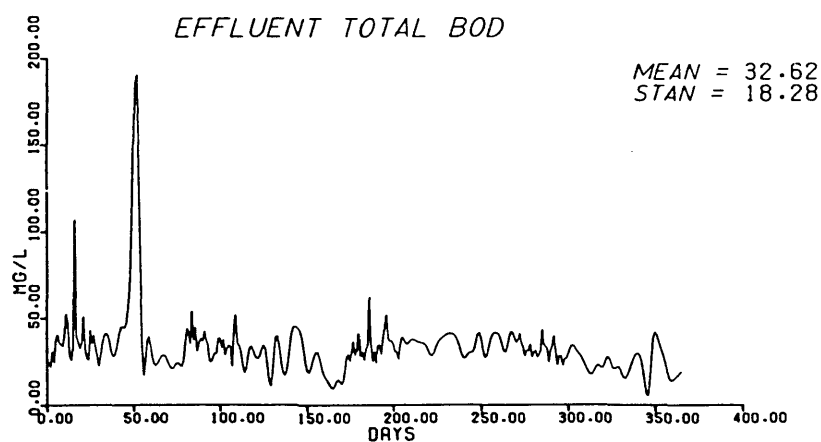


Fig 3.34 Pond 2 Effluent Total BOD (Raw Data)

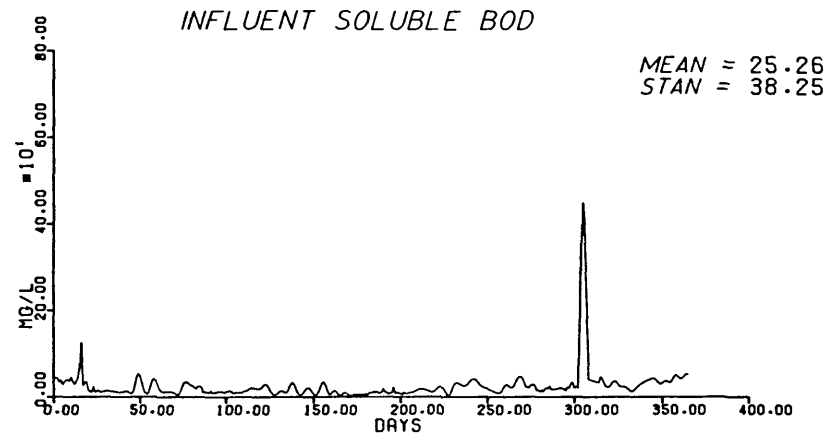


Fig 3.35 Pond 2 Influent Soluble BOD (Raw Data)

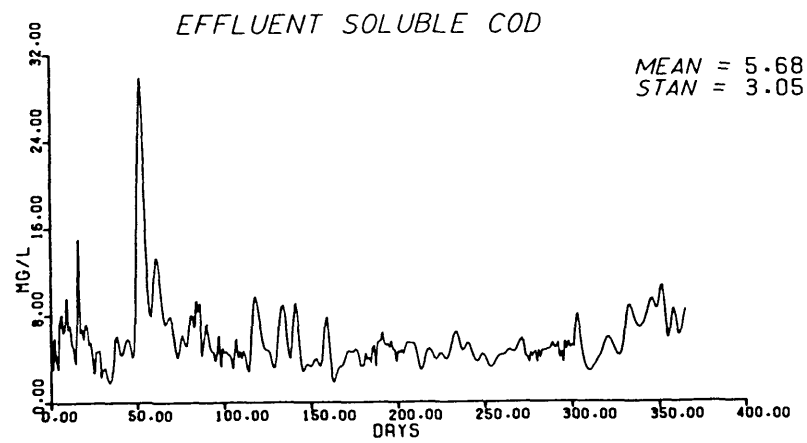


Fig 3.36 Pond 2 Effluent Soluble BOD (Raw Data)

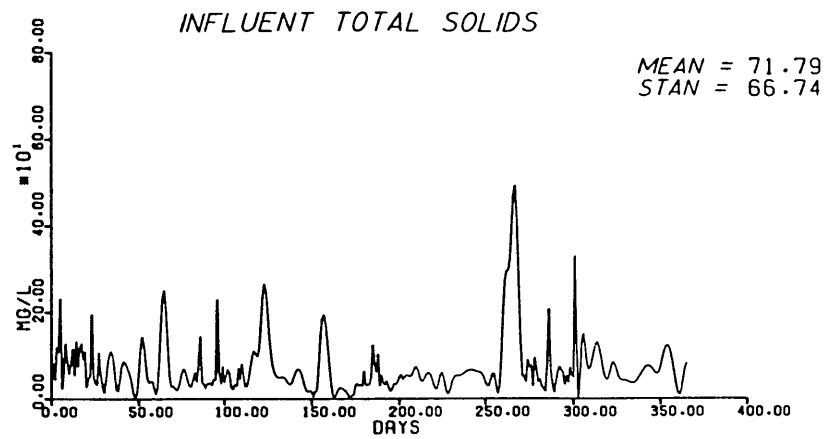


Fig 3.37 Pond 2 Influent Total Solids (Raw Data)

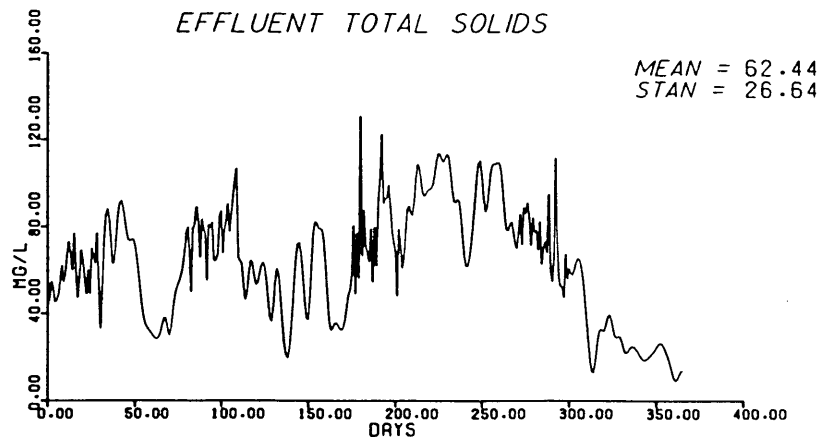


Fig 3.38 Pond 2 Effluent Total Solids (Raw Data)

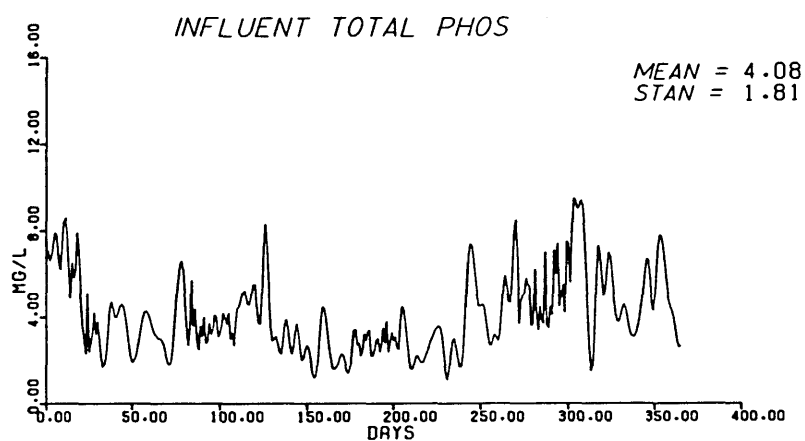


Fig 3.39 Pond 2 Influent Total Phosphorus (Raw Data)

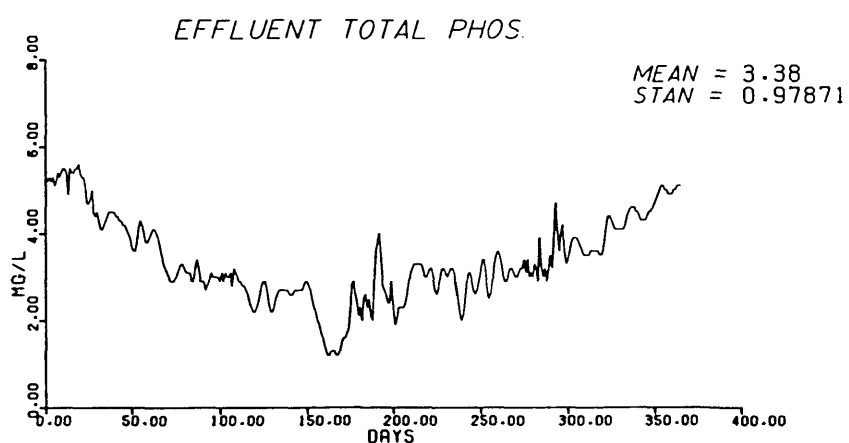


Fig 3.40 Pond 2 Effluent Total Phosphorus (Raw Data)

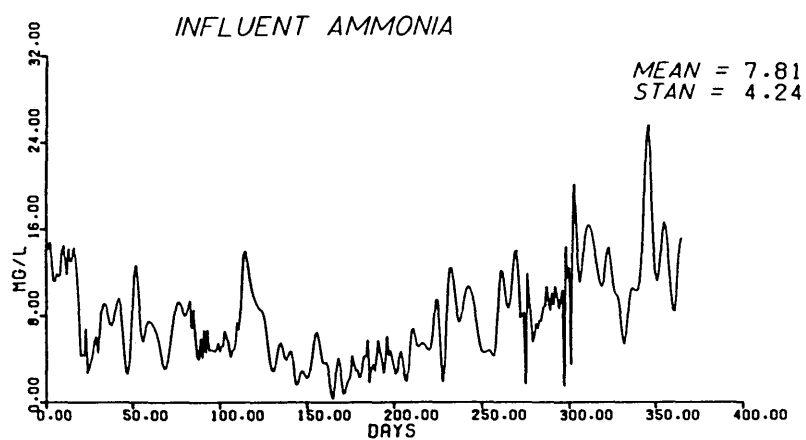


Fig 3.41 Pond 2 Influent Ammonia (Raw Data)

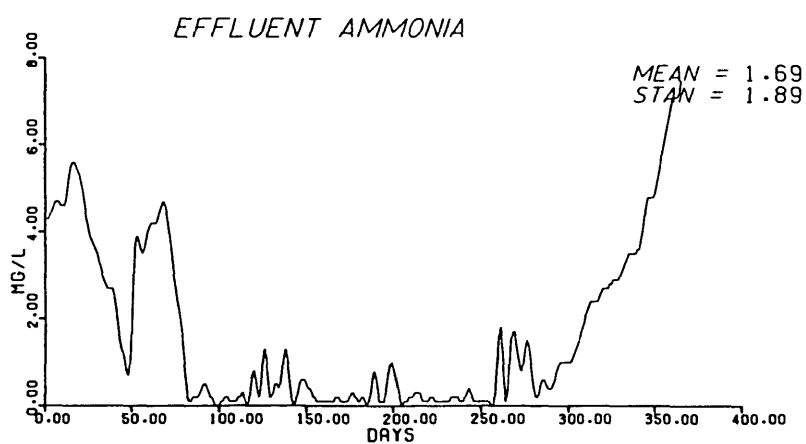


Fig 3.42 Pond 2 Effluent Ammonia (Raw Data)

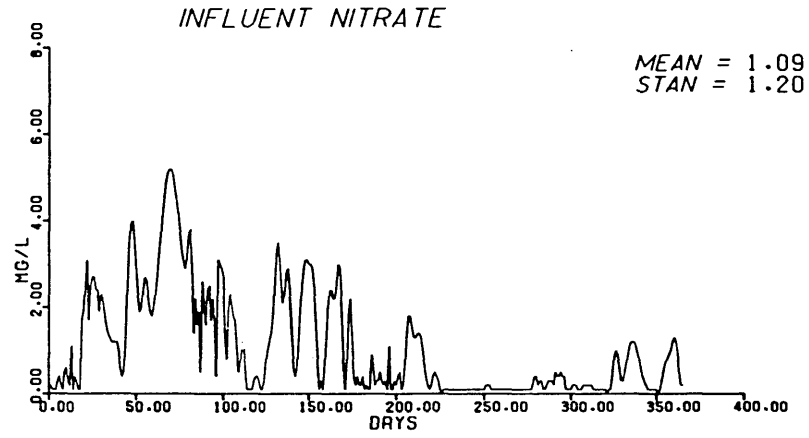


Fig 3.43 Pond 2 Influent Nitrate (Raw Data)

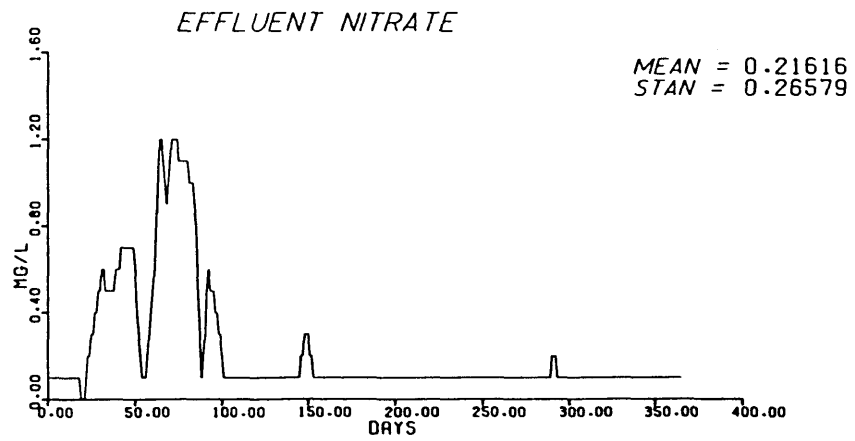


Fig 3.44 Pond 2 Effluent Nitrate (Raw Data)

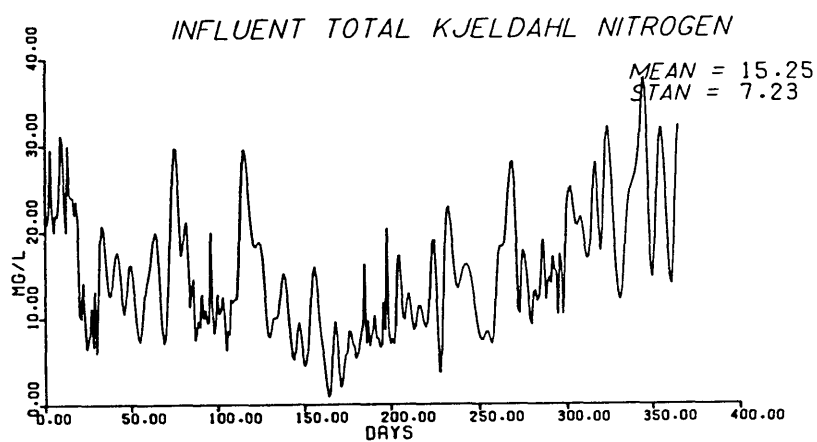


Fig 3.45 Pond 2 Influent Total Kjeldahl Nitrogen (Raw Data)

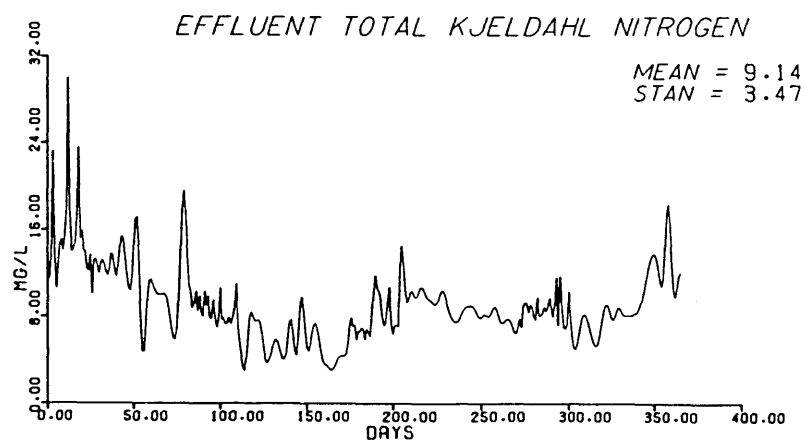


Fig 3.46 Pond 2 Effluent Total Kjeldahl Nitrogen (Raw Data)

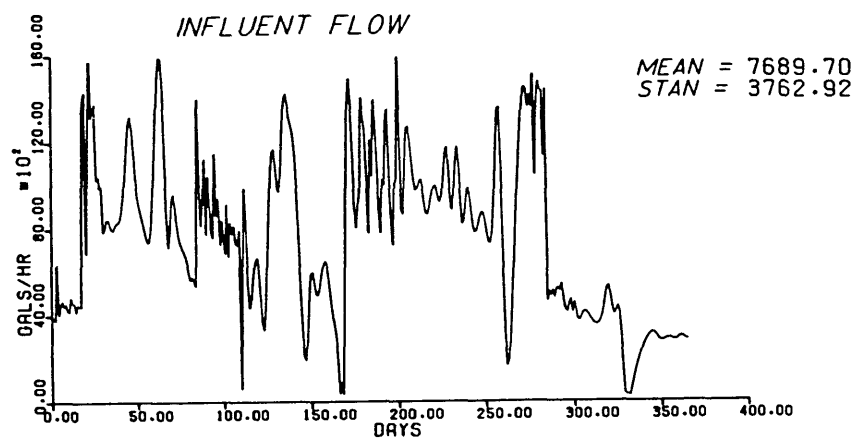


Fig 3.47 Pond 2 Influent Flow (Raw Data)

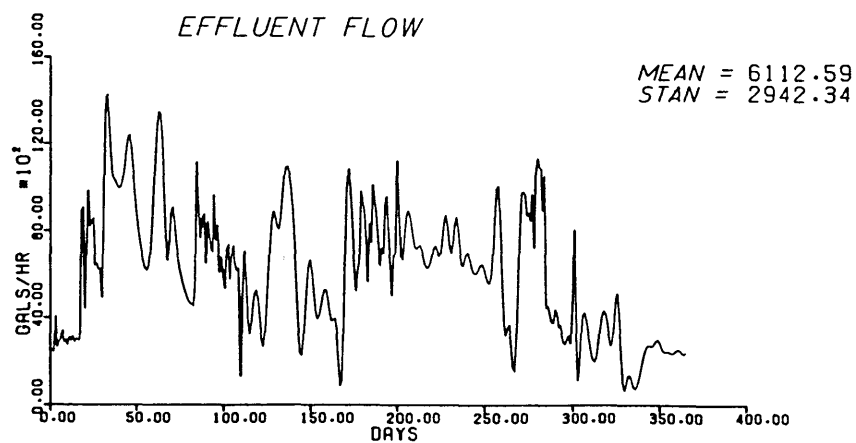


Fig 3.48 Pond 2 Effluent Flow (Raw Data)

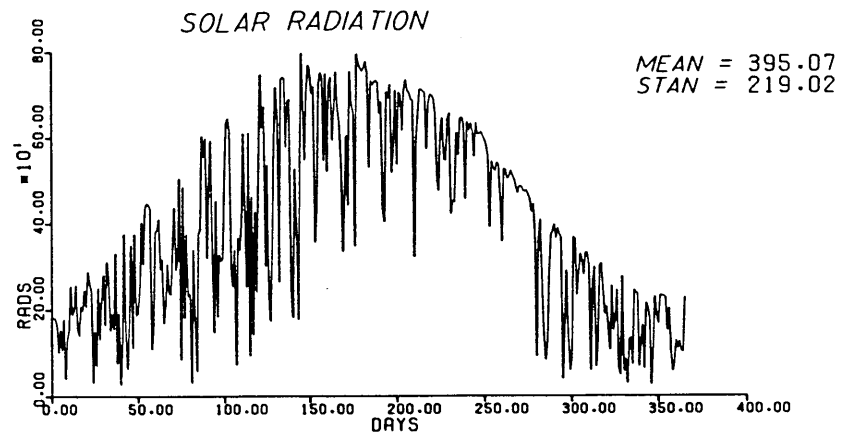


Fig 3.49 Pond 2 Daily Average Solar Radiation (Raw Data)

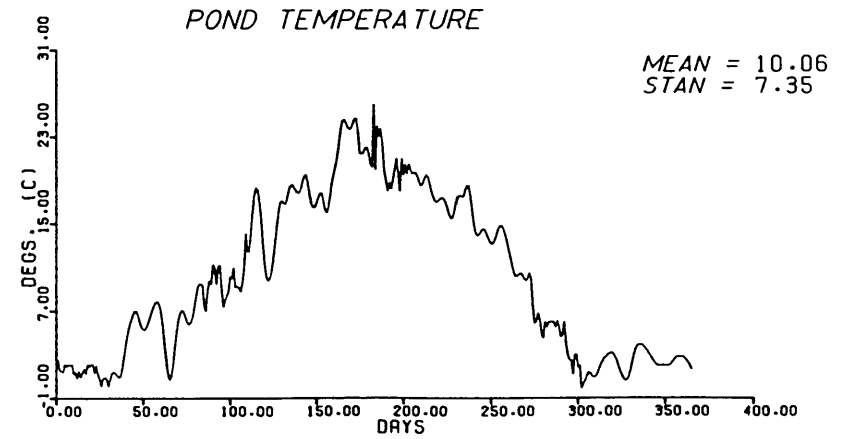


Fig 3.50 Pond 2 Daily Average Pond Temperature (Raw Data)

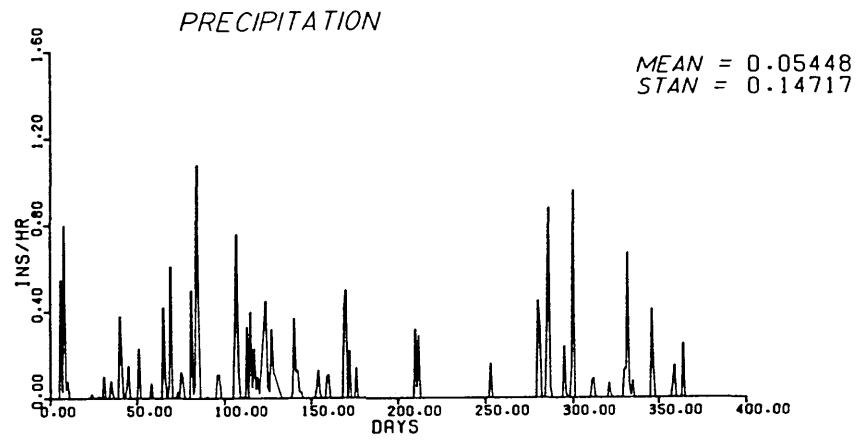


Fig 3.51 Pond 2 Daily Average Precipitation (Raw Data)

HAMONIC ANALYSIS OF RANDOM NUMBERS

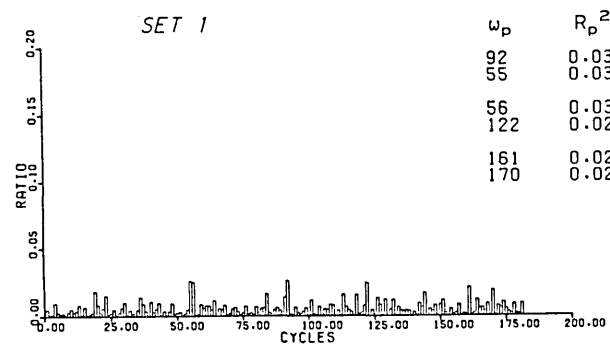


Fig 3.52a Spectrum of Pseudo-Random Sequence (Set 1)

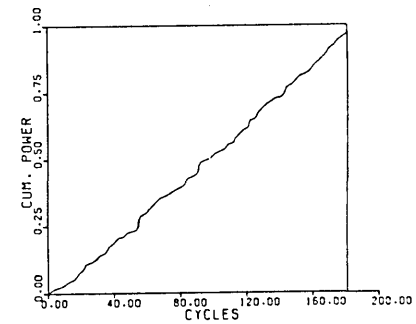


Fig 3.52b Cumulative Periodogram of Pseudo-Random Sequence (Set 1)

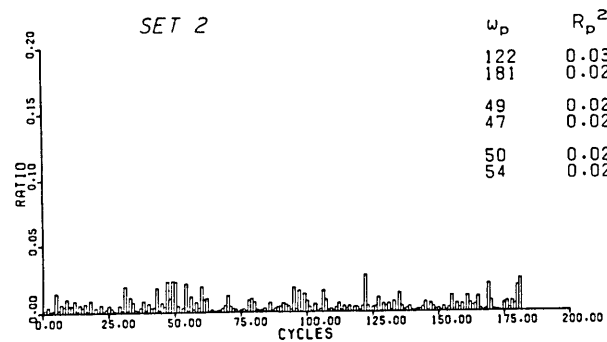


Fig 3.52c Spectrum of Pseudo-Random Sequence (Set 2)

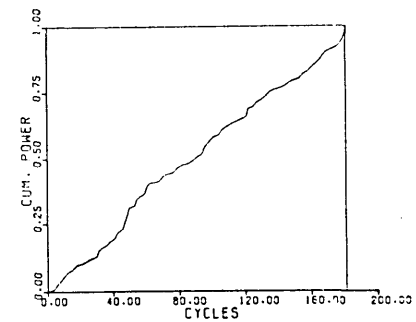


Fig 3.52d Cumulative Periodogram of Pseudo-Random Sequence (Set 2)

HARMONIC ANALYSIS OF POND 1 DATA

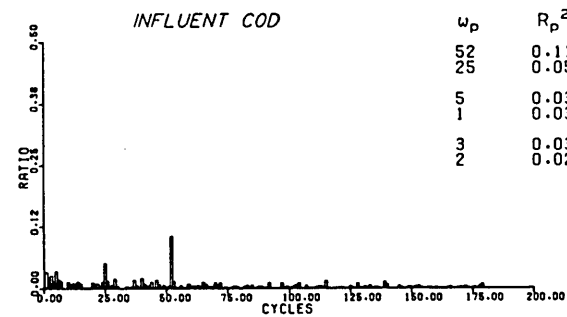


Fig 3.53a Spectrum of Pond 1 Influent Total COD

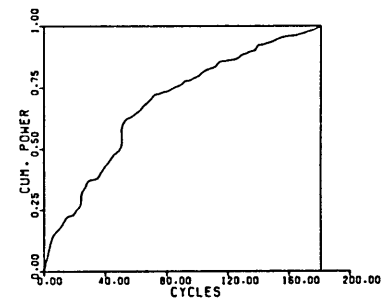


Fig 3.53b Cumulative Periodogram of Pond 1 Influent Total COD

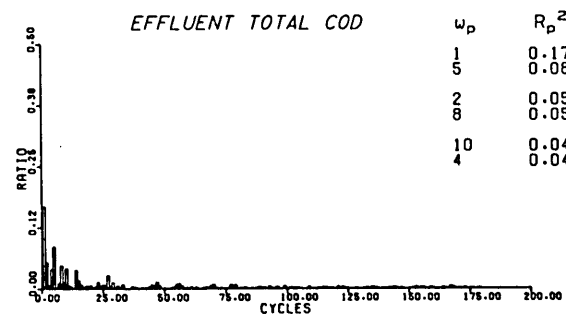


Fig 3.54a Spectrum of Pond 1 Effluent Total COD

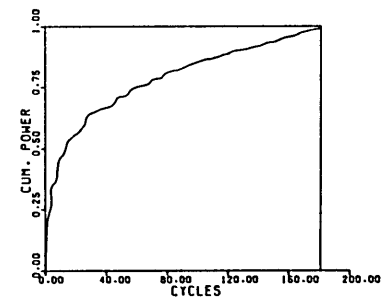


Fig 3.54b Cumulative Periodogram of Pond 1 Effluent Total COD

HARMONIC ANALYSIS OF POND 1 DATA

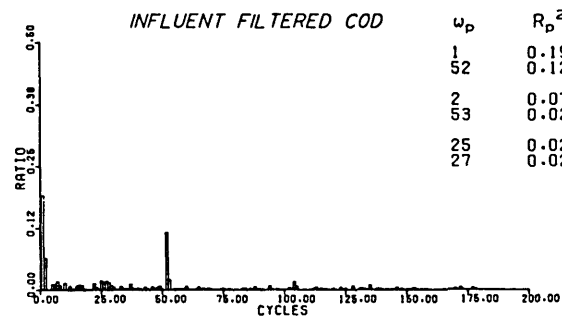


Fig 3.55a Spectrum of Pond 1 Influent Filtered COD

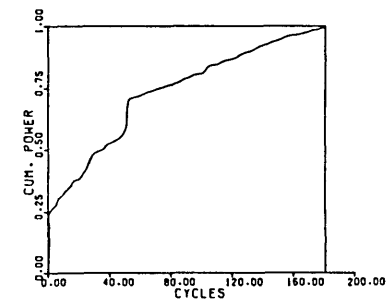


Fig 3.55b Cumulative Periodogram of Pond 1 Influent Filtered COD

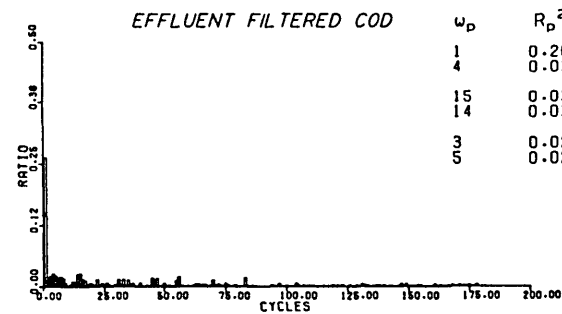


Fig 3.56a Spectrum of Pond 1 Effluent Filtered COD

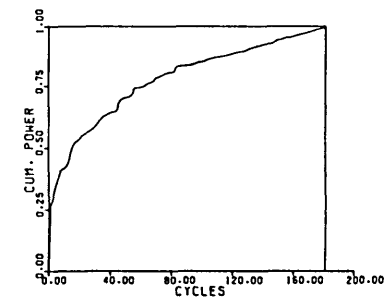


Fig 3.56b Cumulative Periodogram of Pond 1 Effluent Filtered COD

HARMONIC ANALYSIS OF POND 1 DATA

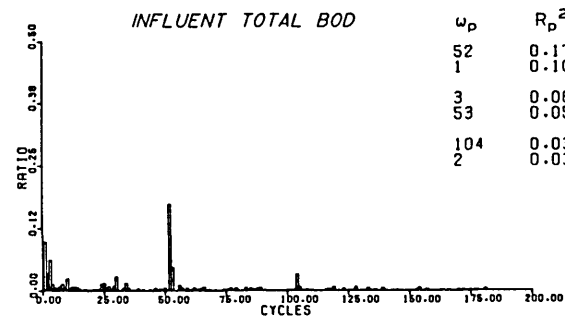


Fig 3.57a Spectrum of Pond 1 Influent Total BOD

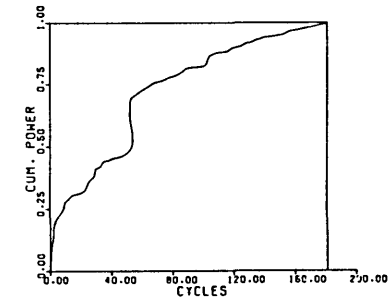


Fig 3.57b Cumulative Periodogram of Pond 1 Influent Total BOD

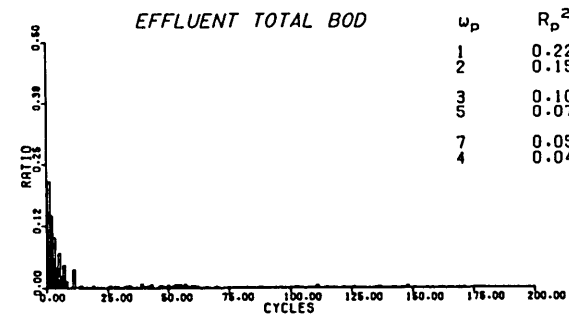


Fig 3.58a Spectrum of Pond 1 Effluent Total BOD

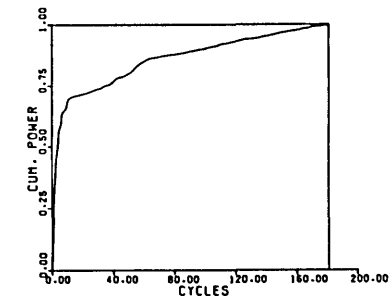


Fig 3.58b Cumulative Periodogram of Pond 1 Effluent Total BOD

HARMONIC ANALYSIS OF POND 1 DATA

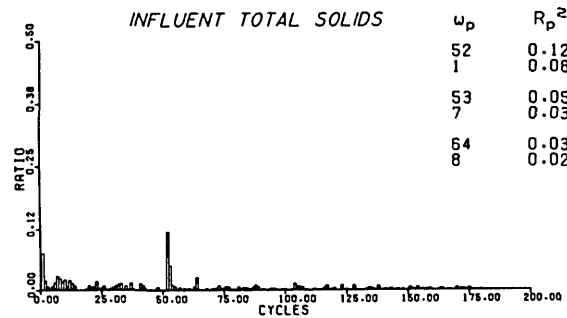


Fig 3.59a Spectrum of Pond 1 Influent Total Solids

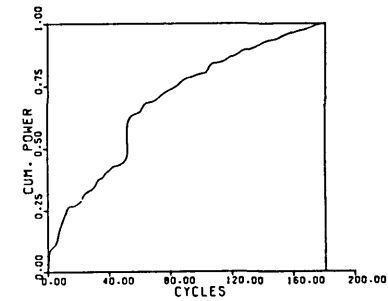


Fig 3.59b Cumulative Periodogram of Pond 1 Influent Total Solids

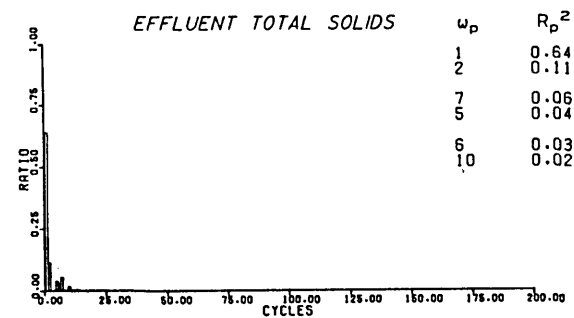


Fig 3.60a Spectrum of Pond 1 Effluent Total Solids

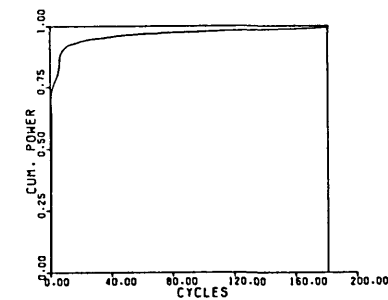


Fig 3.60b Cumulative Periodogram of Pond 1 Effluent Total Solids

HARMONIC ANALYSIS OF POND 1 DATA

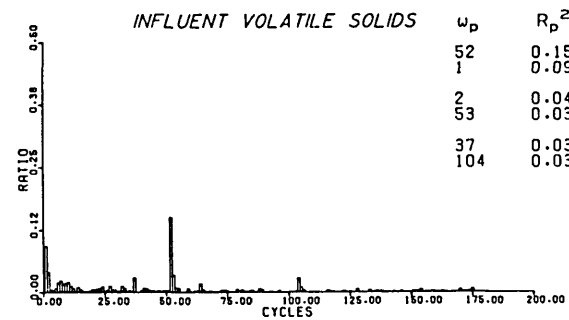


Fig 3.61a Spectrum of Pond 1 Influent Volatile Solids

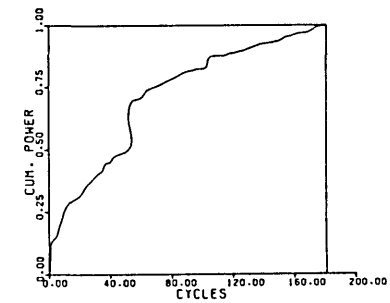


Fig 3.61b Cumulative Periodogram of Pond 1 Influent Volatile Solids

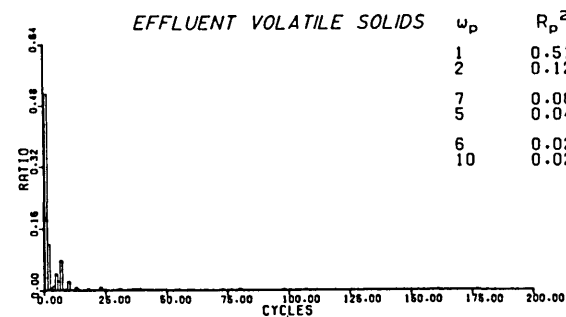


Fig 3.62a Spectrum of Pond 1 Effluent Volatile Solids

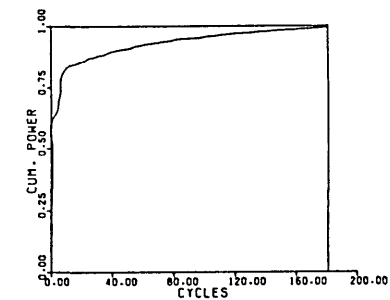


Fig 3.62b Cumulative Periodogram of Pond 1 Effluent Volatile Solids

HARMONIC ANALYSIS OF POND 1 DATA

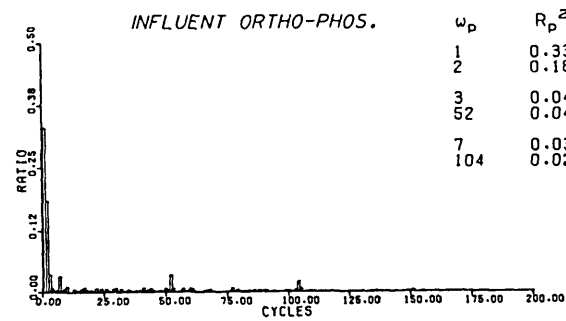


Fig 3.63a Spectrum of Pond 1 Influent Ortho-phosphorous

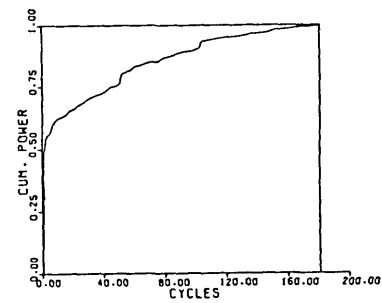


Fig 3.63b Cumulative Periodogram of Pond 1 Influent Ortho-phosphorous

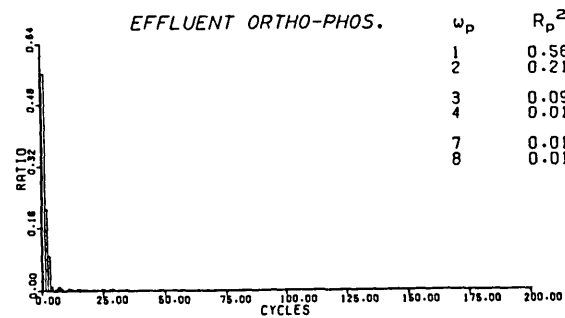


Fig 3.64a Spectrum of Pond 1 Effluent Ortho-phosphorous

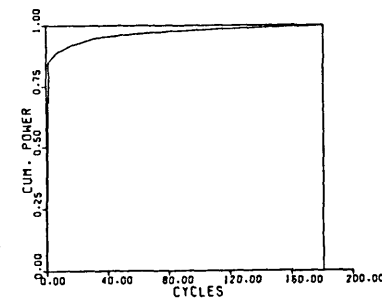


Fig 3.64b Cumulative Periodogram of Pond 1 Effluent Ortho-phosphorous

HARMONIC ANALYSIS OF POND 1 DATA

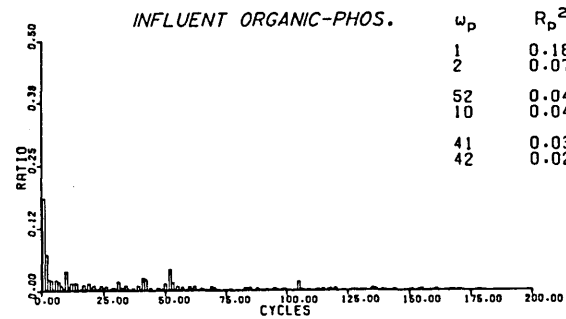


Fig 3.65a Spectrum of Pond 1 Influent Organic-phosphorous

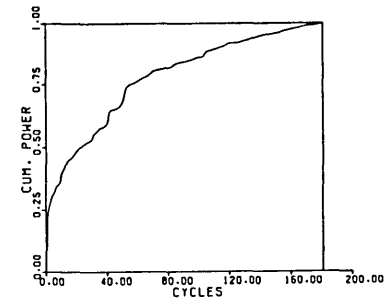


Fig 3.65b Cumulative Periodogram of Pond 1 Influent Organic-phosphorous

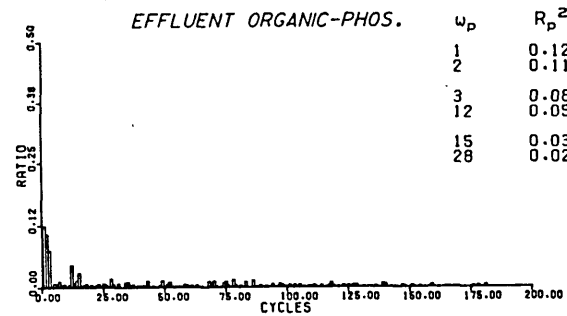


Fig 3.66a Spectrum of Pond 1 Effluent Organic-phosphorous

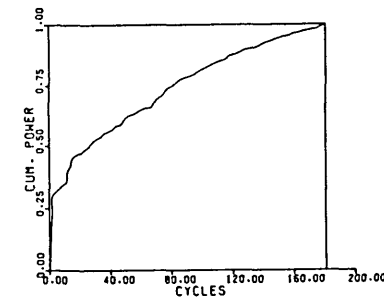


Fig 3.66b Cumulative Periodogram of Pond 1 Effluent Organic-phosphorous

HARMONIC ANALYSIS OF POND 1 DATA

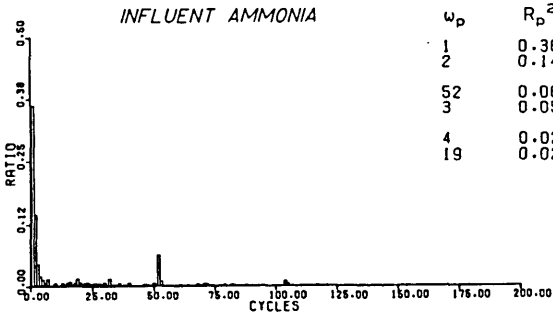


Fig 3.67a Spectrum of Pond 1 Influent Ammonia

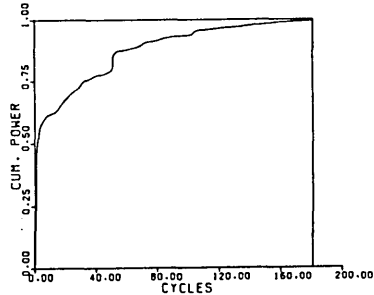


Fig 3.67b Cumulative Periodogram of Pond 1 Influent Ammonia

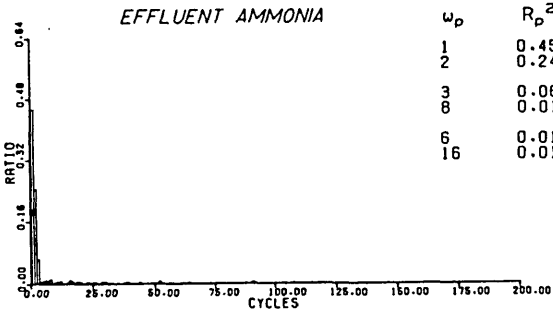


Fig 3.68a Spectrum of Pond 1 Effluent Ammonia

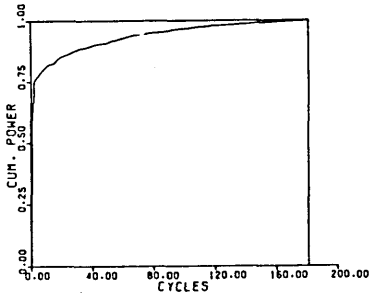


Fig 3.68b Cumulative Periodogram of Pond 1 Effluent Ammonia

HARMONIC ANALYSIS OF POND 1 DATA

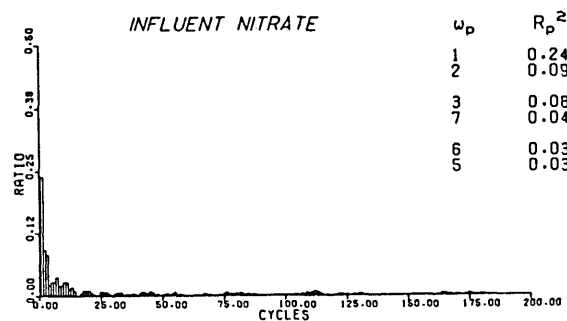


Fig 3.69a Spectrum of Pond 1 Influent Nitrate

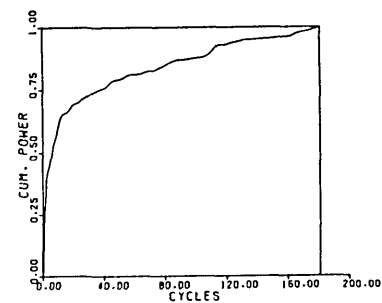


Fig 3.69b Cumulative Periodogram of Pond 1 Influent Nitrate

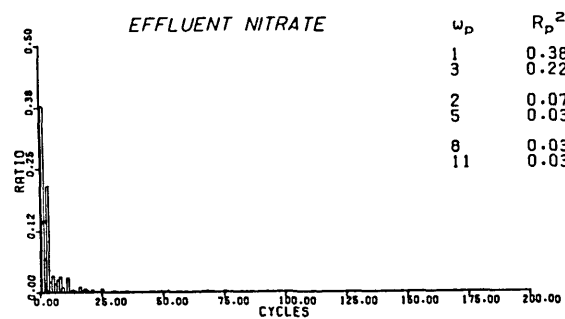


Fig 3.70a Spectrum of Pond 1 Effluent Nitrate

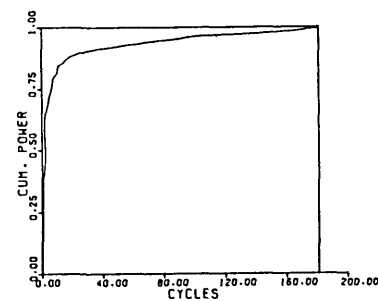


Fig 3.70b Cumulative Periodogram of Pond 1 Effluent Nitrate

HARMONIC ANALYSIS OF POND 1 DATA

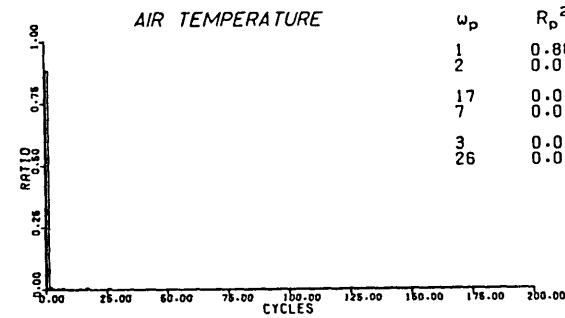


Fig 3.71a Spectrum of Pond 1 Air Temperature

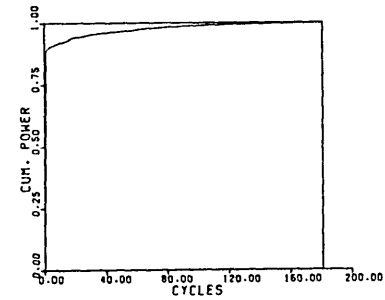


Fig 3.71b Cumulative Periodogram of Pond 1 Air Temperature

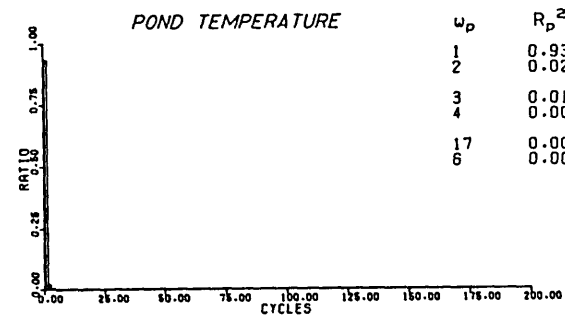


Fig 3.72a Spectrum of Pond 1 Pond Temperature

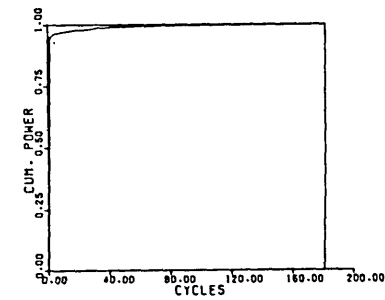


Fig 3.72b Cumulative Periodogram of Pond 1 Pond Temperature

HARMONIC ANALYSIS OF POND 1 DATA

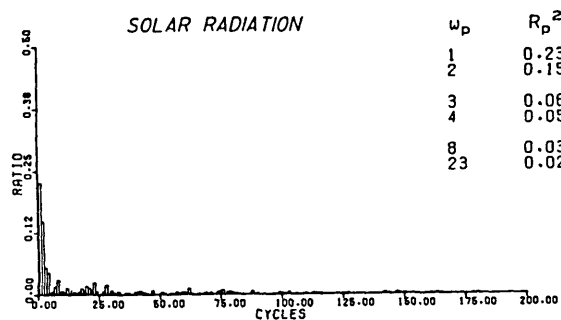


Fig 3.73a Spectrum of Pond 1 Solar Radiation

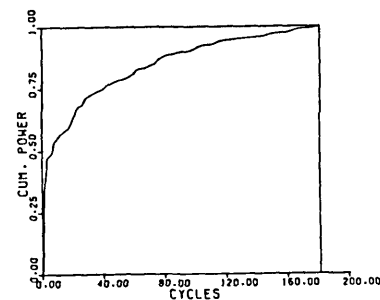


Fig 3.73b Cumulative Periodogram of Pond 1 Solar Radiation

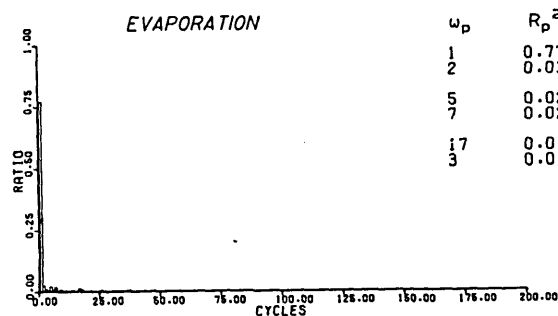


Fig 3.74a Spectrum of Pond 1 Evaporation

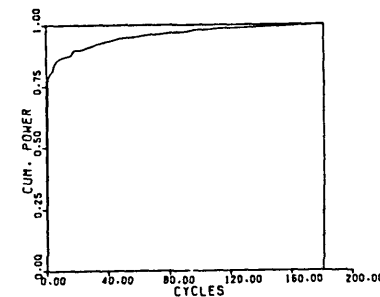


Fig 3.74b Cumulative Periodogram of Pond 1 Evaporation

HARMONIC ANALYSIS OF POND 1 DATA

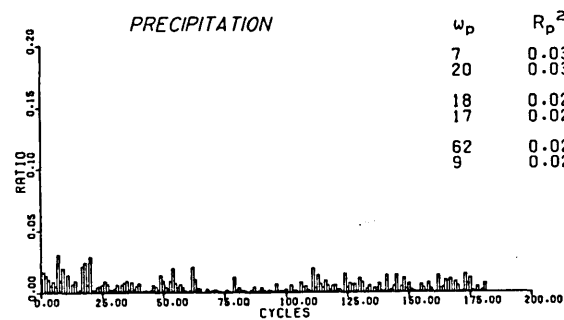


Fig 3.75a Spectrum of Pond 1 Precipitation

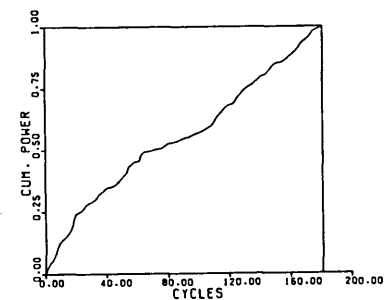


Fig 3.75b Cumulative Periodogram of Pond 1 Precipitation

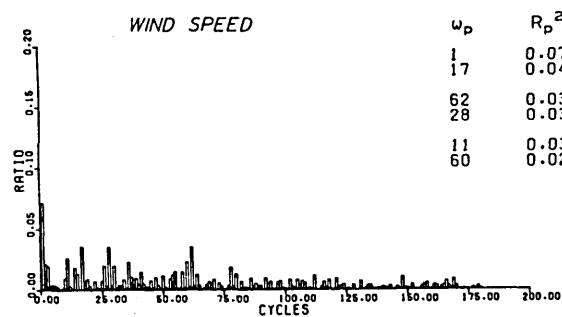


Fig 3.76a Spectrum of Pond 1 Wind Speed

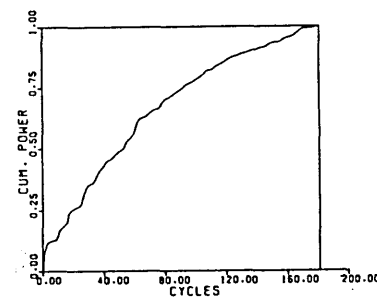


Fig 3.76b Cumulative Periodogram of Pond 1 Wind Speed

HARMONIC ANALYSIS OF POND 1 DATA

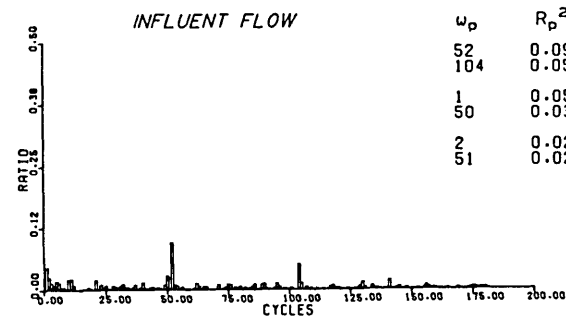


Fig 3.77a Spectrum of Pond 1 Influent Flow

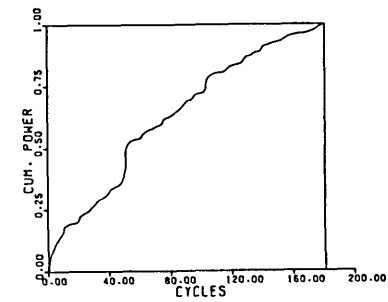


Fig 3.77b Cumulative Periodogram of Pond 1 Influent Flow

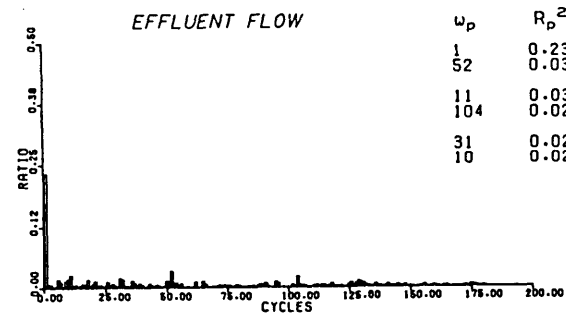


Fig 3.78a Spectrum of Pond 1 Effluent Flow

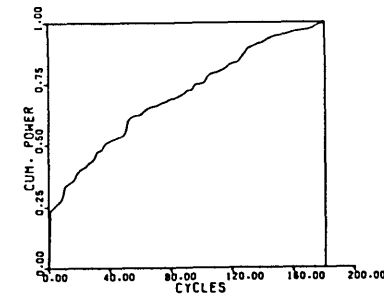


Fig 3.78b Cumulative Periodogram of Pond 1 Effluent Flow

HARMONIC ANALYSIS OF POND 2 DATA

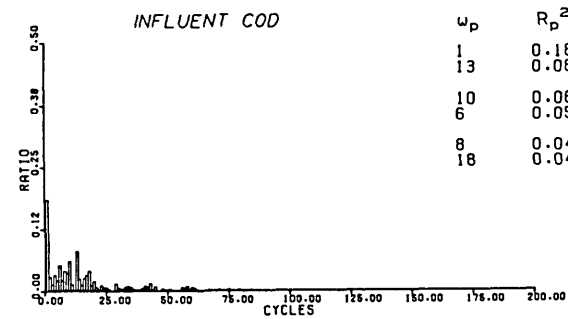


Fig 3.79a Spectrum of Pond 2 Influent Total COD

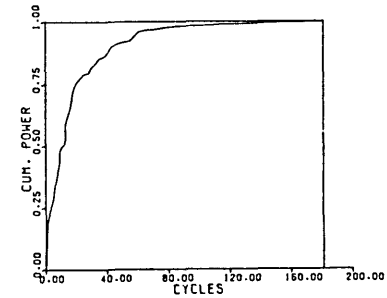


Fig 3.79b Cumulative Periodogram of Pond 2 Influent Total COD

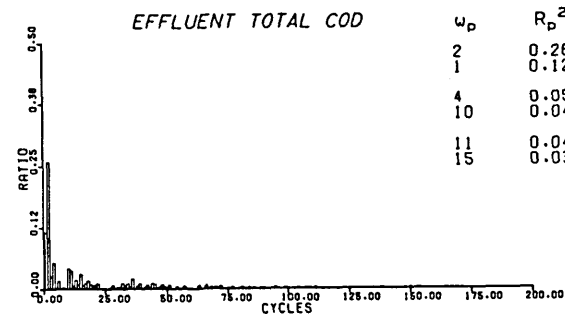


Fig 3.80a Spectrum of Pond 2 Effluent Total COD

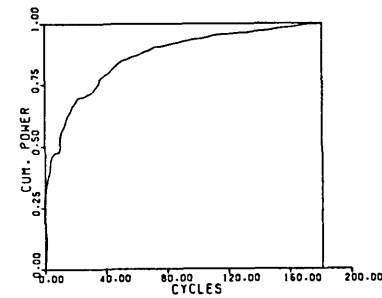


Fig 3.80b Cumulative Periodogram of Pond 2 Effluent Total COD

HARMONIC ANALYSIS OF POND 2 DATA

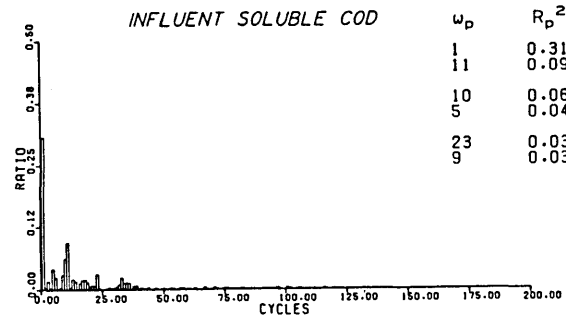


Fig 3.81a Spectrum of Pond 2 Influent Soluble COD

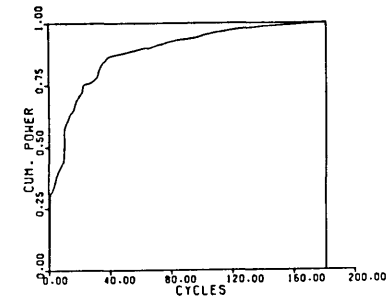


Fig 3.81b Cumulative Periodogram of Pond 2 Influent Soluble COD

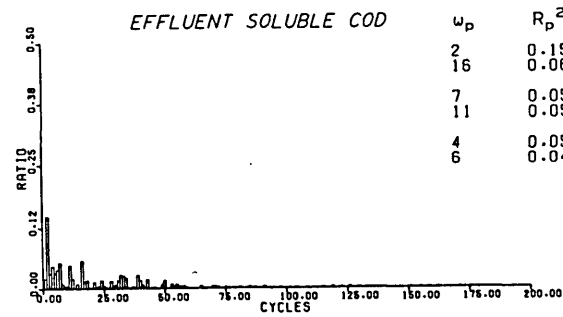


Fig 3.82a Spectrum of Pond 2 Effluent Soluble COD

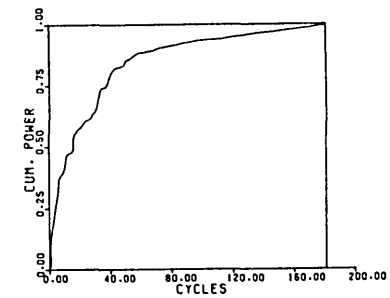


Fig 3.82b Cumulative Periodogram of Pond 2 Effluent Soluble COD

HARMONIC ANALYSIS OF POND 2 DATA

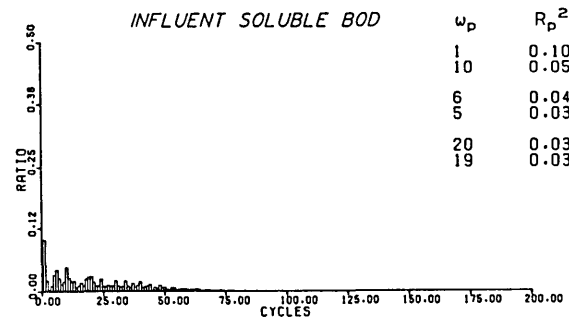


Fig 3.83a Spectrum of Pond 2 Influent Soluble BOD

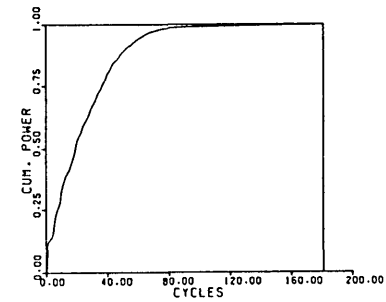


Fig 3.83b Cumulative Periodogram of Pond 2 Influent Soluble BOD

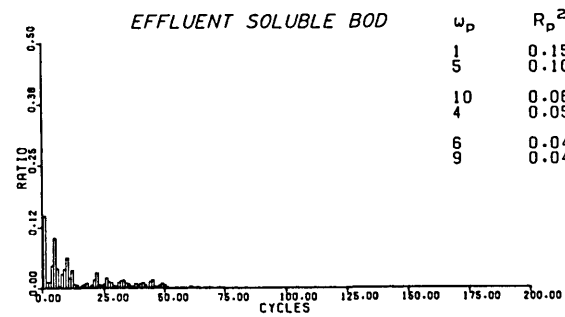


Fig 3.84a Spectrum of Pond 2 Effluent Soluble BOD

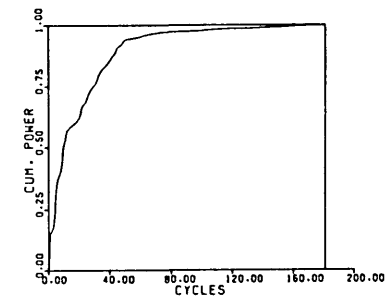


Fig 3.84b Cumulative Periodogram of Pond 2 Effluent Soluble BOD

HARMONIC ANALYSIS OF POND 2 DATA

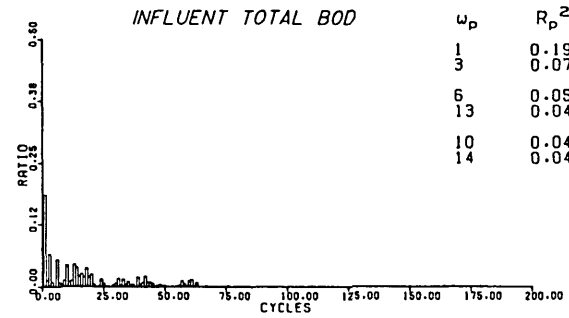


Fig 3.85a Spectrum of Pond 2 Influent Total BOD

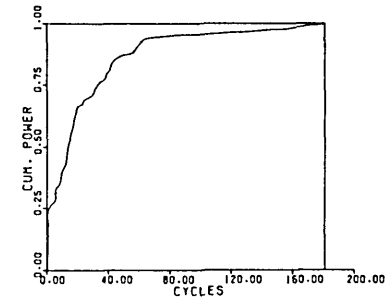


Fig 3.85b Cumulative Periodogram of Pond 2 Influent Total BOD

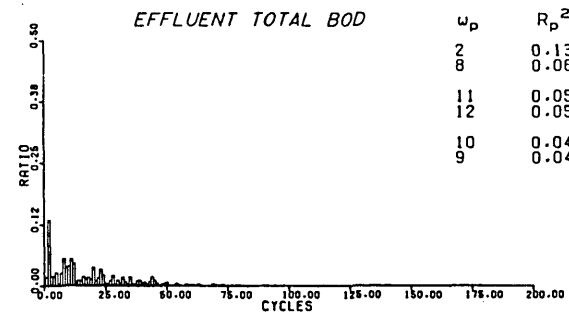


Fig 3.86a Spectrum of Pond 2 Effluent Total BOD

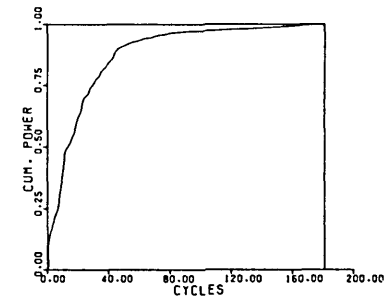


Fig 3.86b Cumulative Periodogram of Pond 2 Effluent Total BOD

HARMONIC ANALYSIS OF POND 2 DATA

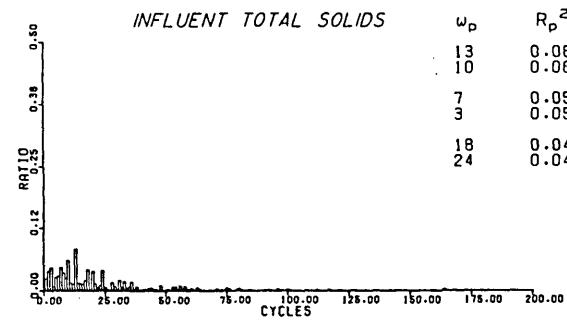


Fig 3.87a Spectrum of Pond 2 Influent Total Solids

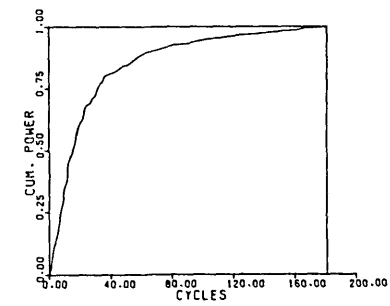


Fig 3.87b Cumulative Periodogram of pond 2 Influent Total Solids

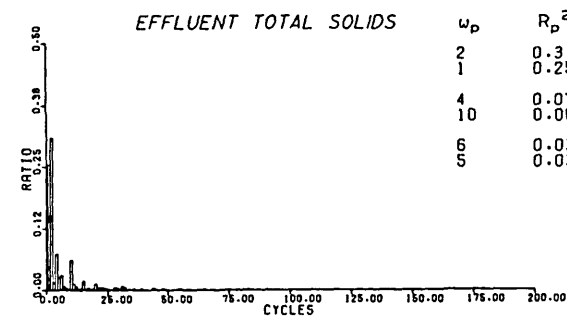


Fig 3.88a Spectrum of Pond 2 Effluent Total Solids

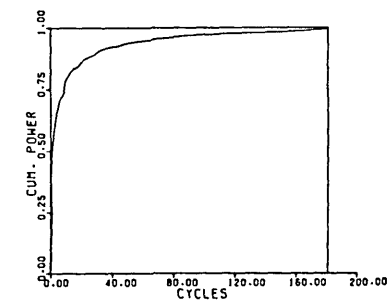


Fig 3.88b Cumulative Periodogram of Pond 2 Effluent Total Solids

HARMONIC ANALYSIS OF POND 2 DATA

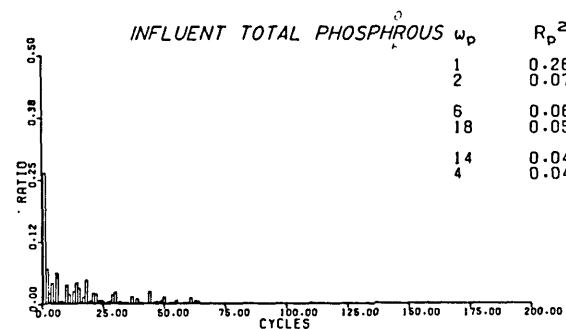


Fig 3.89a Spectrum of Pond 2 Influent Total Phosphorous

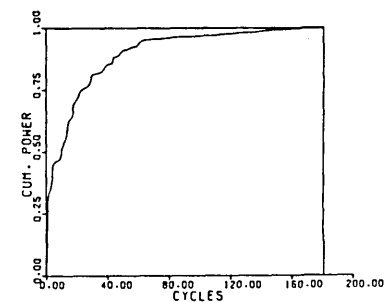


Fig 3.89b Cumulative Periodogram of Pond 2 Influent Total Phosphorous

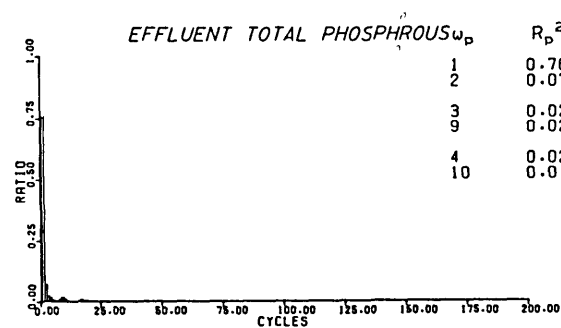


Fig 3.90a Spectrum of Pond 2 Effluent Total Phosphorous

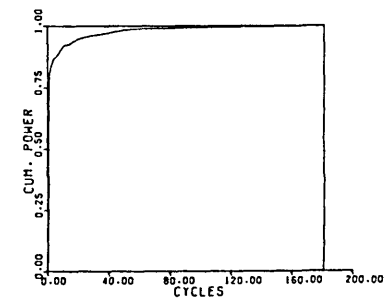


Fig 3.90b Cumulative Periodogram of Pond 2 Effluent Total Phosphorous

HARMONIC ANALYSIS OF POND 2 DATA

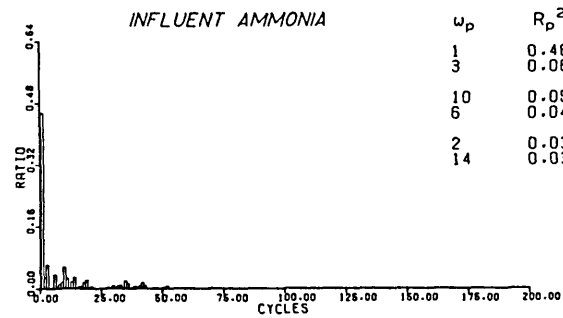


Fig 3.91a Spectrum of Pond 2 Influent Ammonia

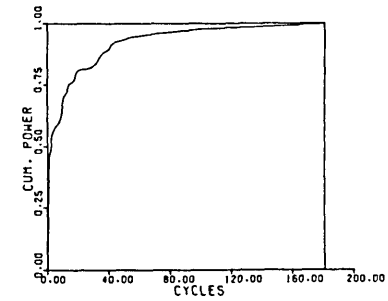


Fig 3.91b Cumulative Periodogram of Pond 2 Influent Ammonia

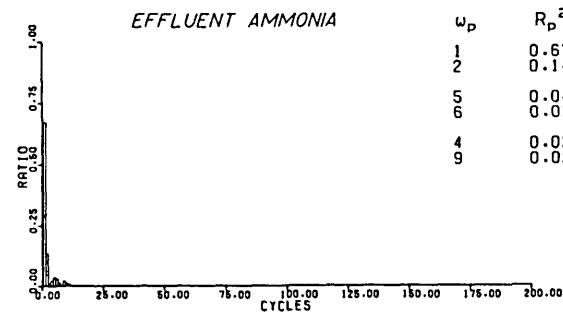


Fig 3.92a Spectrum of Pond 2 Effluent Ammonia

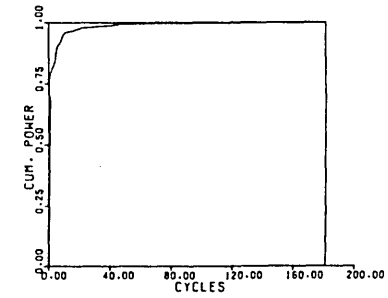


Fig 3.92b Cumulative Periodogram of Pond 2 Effluent Ammonia

HARMONIC ANALYSIS OF POND 2 DATA

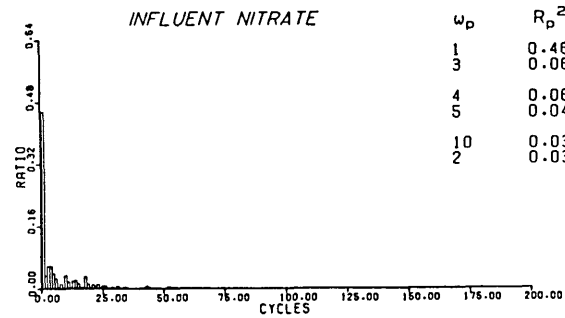


Fig 3.93a Spectrum of Pond 2 Influent Nitrate

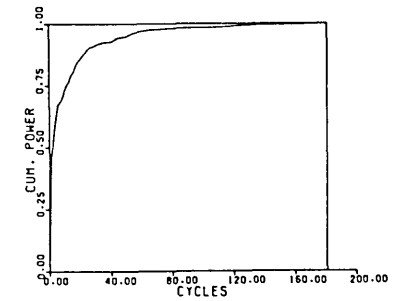


Fig 3.93b Cumulative Periodogram of Pond 2 Influent Nitrate

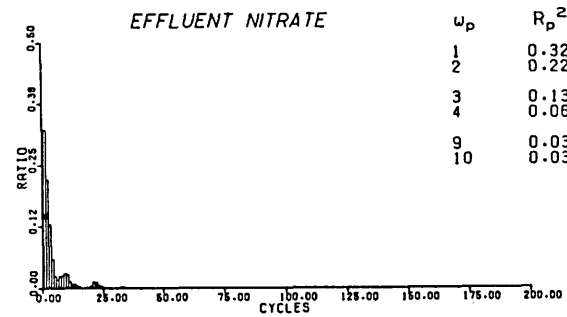


Fig 3.94a Spectrum of Pond 2 Effluent Nitrate

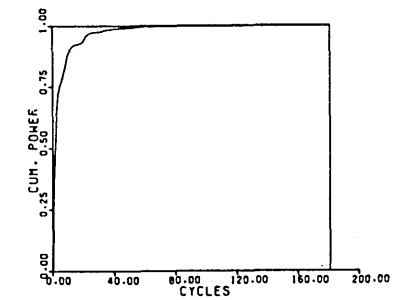


Fig 3.94b Cumulative Periodogram of Pond 2 Effluent Nitrate

HARMONIC ANALYSIS OF POND 2 DATA

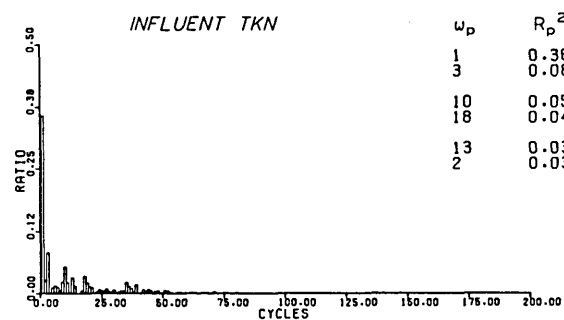


Fig 3.95a Spectrum of Pond 2 Influent Total Kjeldahl Nitrogen

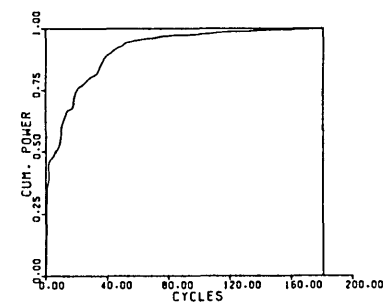


Fig 3.95b Cumulative Periodogram of Pond 2 Influent Total Kjeldahl Nitrogen

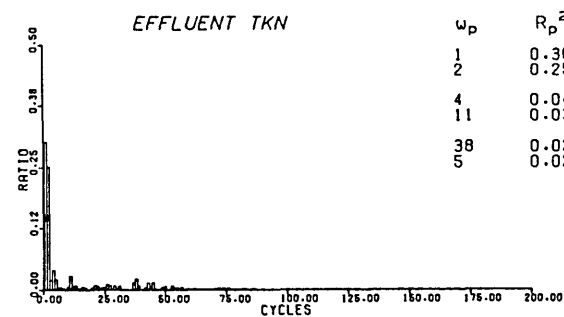


Fig 3.96a Spectrum of Pond 2 Effluent Total Kjeldahl Nitrogen

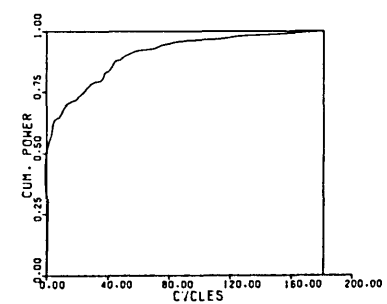


Fig 3.96b Cumulative Periodogram of Pond 2 Effluent Total Kjeldahl Nitrogen

HARMONIC ANALYSIS OF POND 2 DATA

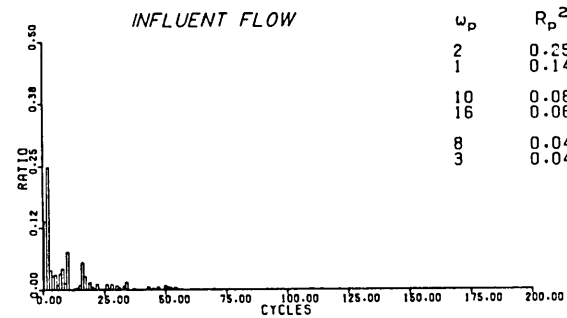


Fig 3.97a Spectrum of Pond 2 Influent Flow

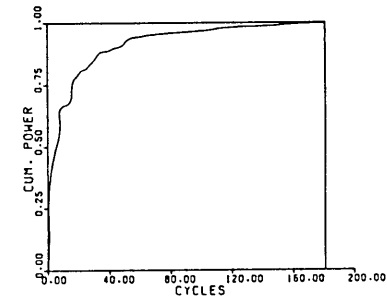


Fig 3.97b Cumulative Periodogram of Pond 2 Influent Flow

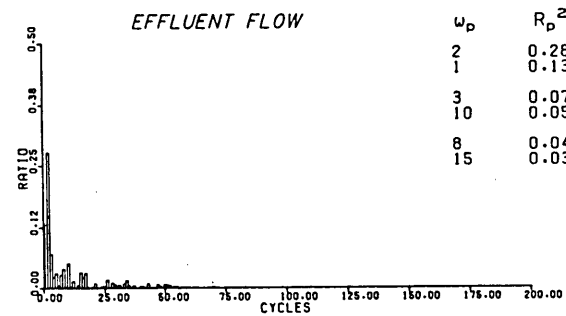


Fig 3.98a Spectrum of Pond 2 Effluent Flow

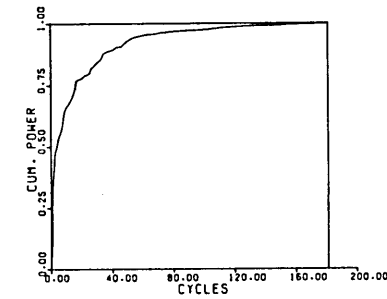


Fig 3.98b Cumulative Periodogram of Pond 2 Effluent Flow

HARMONIC ANALYSIS OF POND 2 DATA

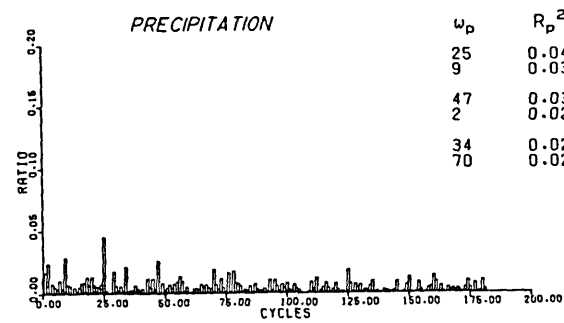


Fig 3.99a Spectrum of Pond 2 Precipitation

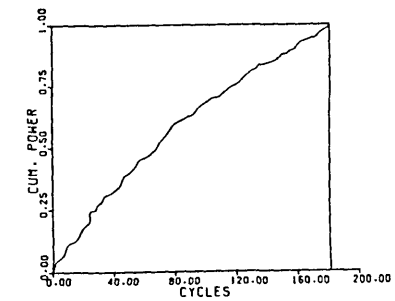


Fig 3.99b Cumulative Periodogram of Pond 2 Precipitation

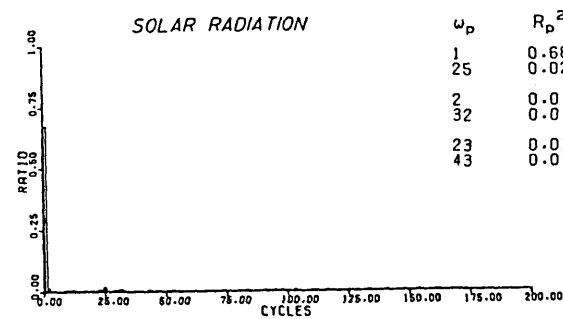


Fig 3.100a Spectrum of Pond 2 Solar Radiation

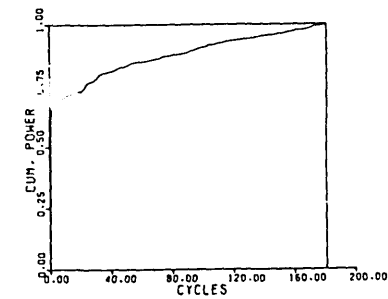


Fig 3.100b Cumulative Periodogram of Pond 2 Solar Radiation

HARMONIC ANALYSIS OF POND 2 DATA

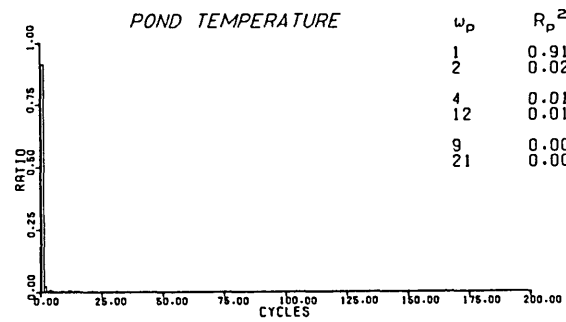


Fig 3.101a Spectrum of Pond 2 Pond Temperature

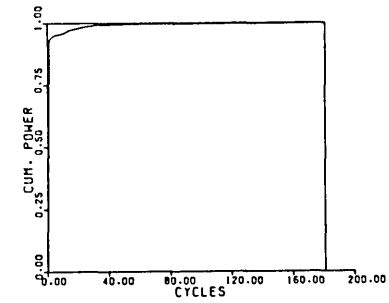


Fig 3.101b Cumulative Periodogram of Pond 2 Pond Temperature

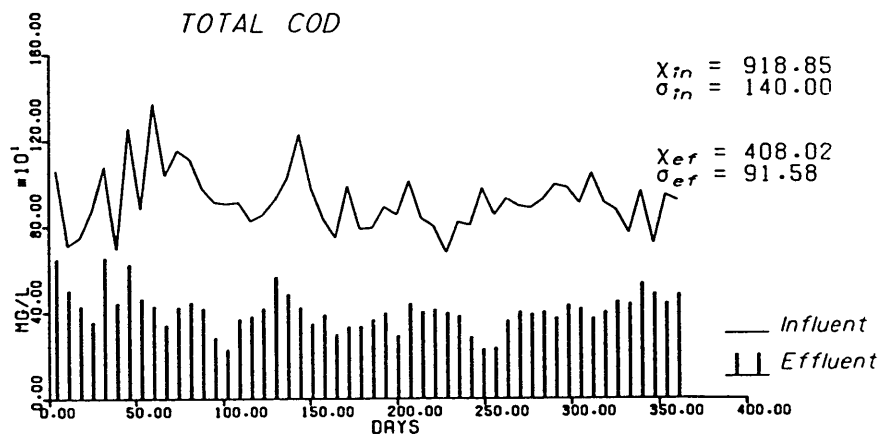


Fig 3.102 7-days Moving Average of Influent & Effluent
Total COD (Pond 1)

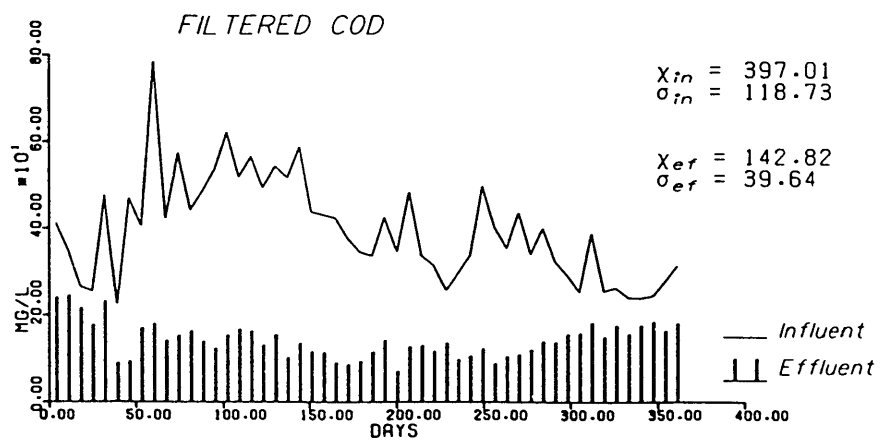


Fig 3.103 7-days Moving Average of influent & Effluent
Filtered COD (Pond 1)

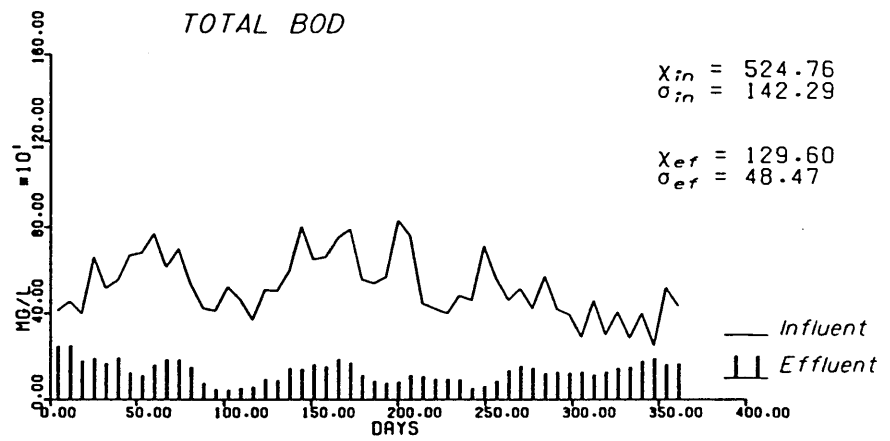


Fig 3.104 7-days Moving Average of Influent & Effluent
Total BOD (Pond 1)

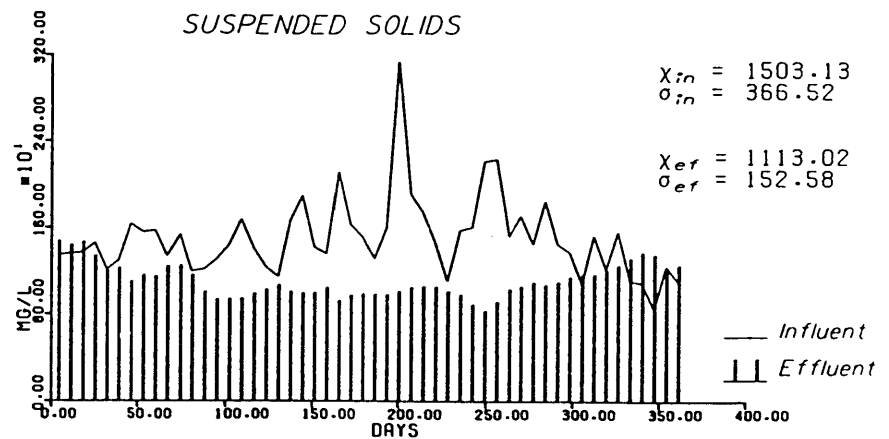


fig 3.105 7-days Moving Average of Influent & Effluent
Suspended Solids (Pond 1)

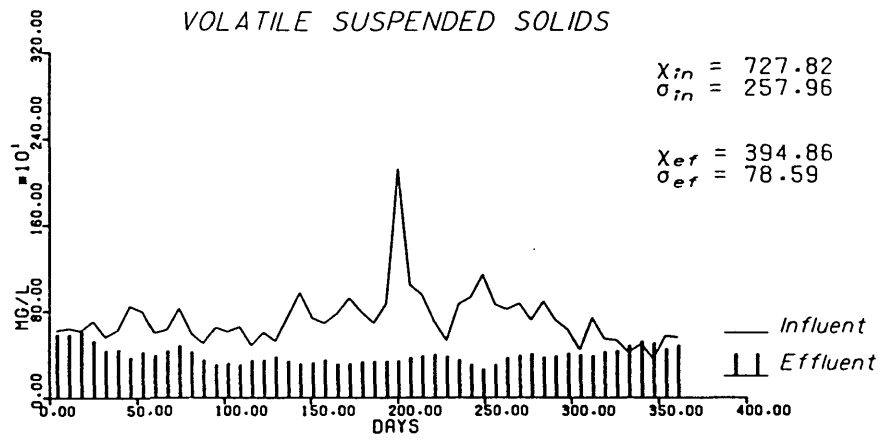


Fig 3.106 7-days Moving Average of Influent & Effluent Volatile Suspended Solids (Pond 1)

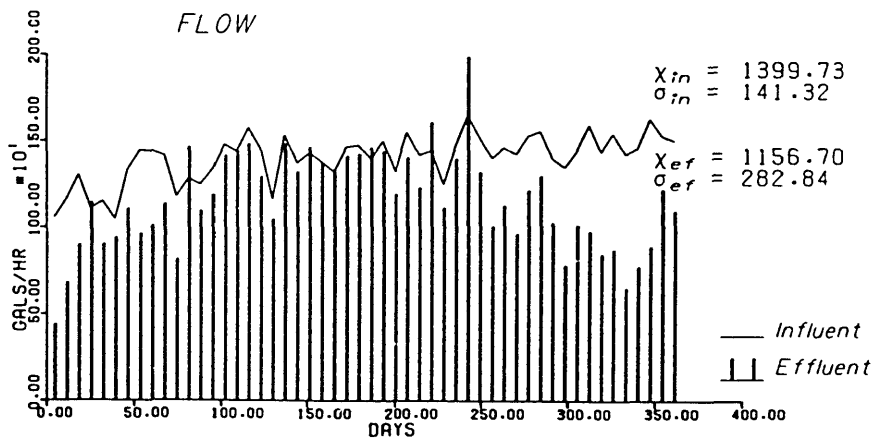


Fig 3.107 7-days Moving Average of Influent & Effluent Flow (Pond 1)

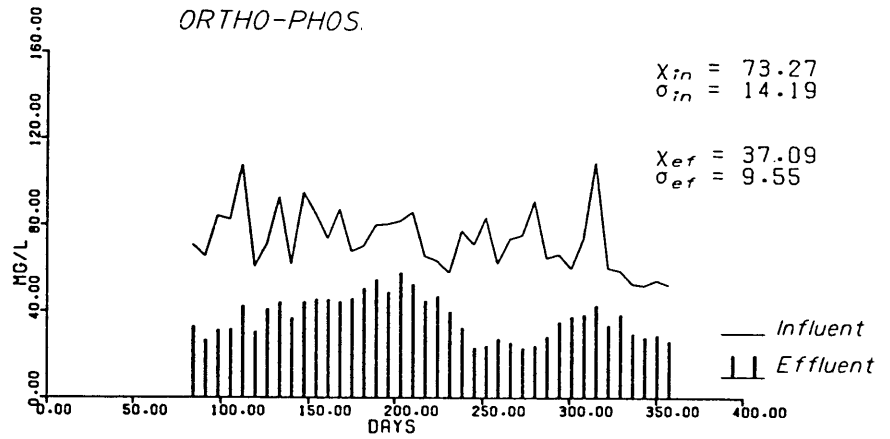


Fig 3.108 7-days Moving Average of Influent & Effluent Ortho-phosphorus (Pond 1)

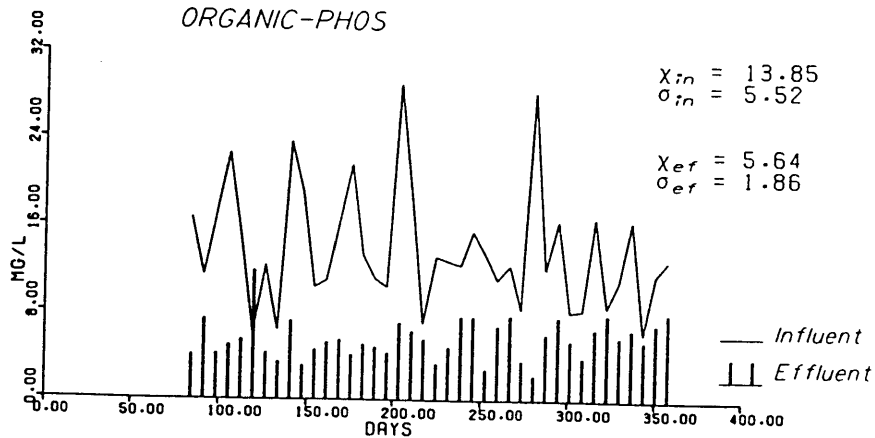


Fig 3.109 7-days Moving Average of Influent & Effluent Organic Phosphorus (Pond 1)

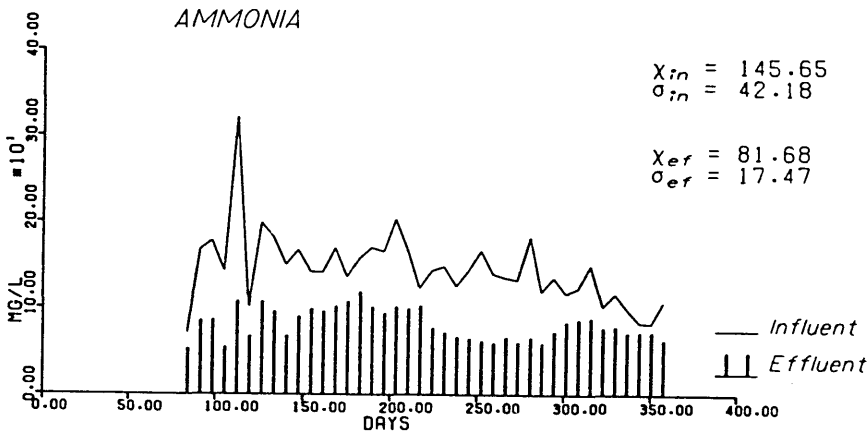


Fig 3.110 7-days Moving Average of Influent & Effluent Ammonia (Pond 1)

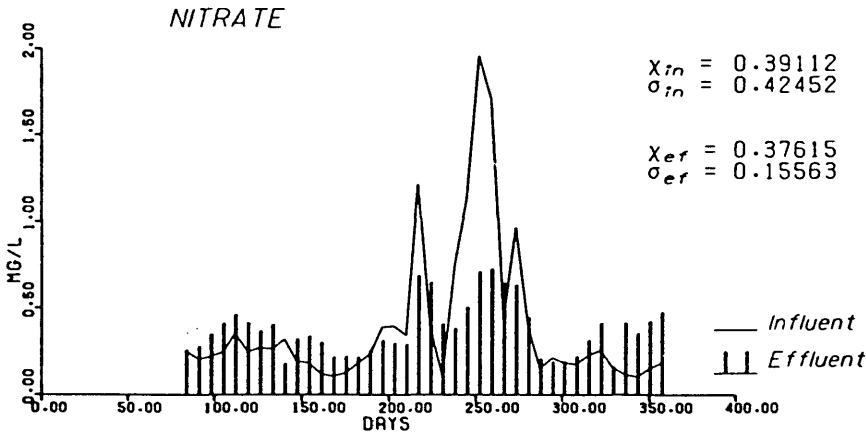


Fig 3.111 7-days Moving Average of Influent & Effluent Nitrate (Pond 1)

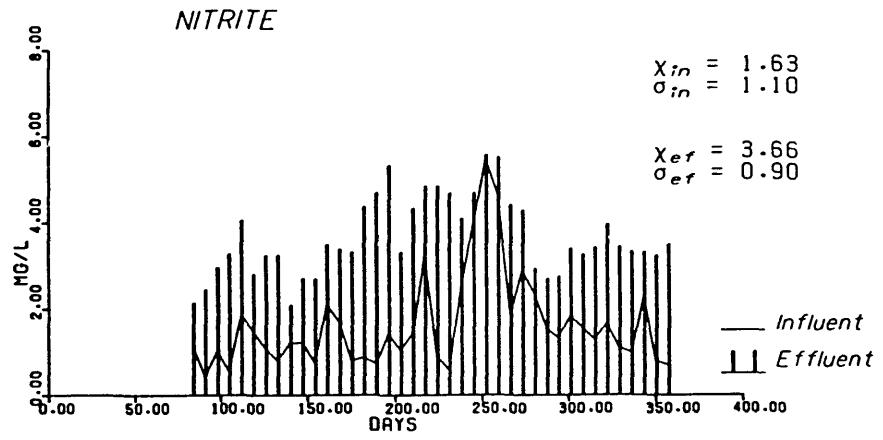


Fig 3.112 7-days Moving Average of Influent & Effluent Nitrite (Pond 1)

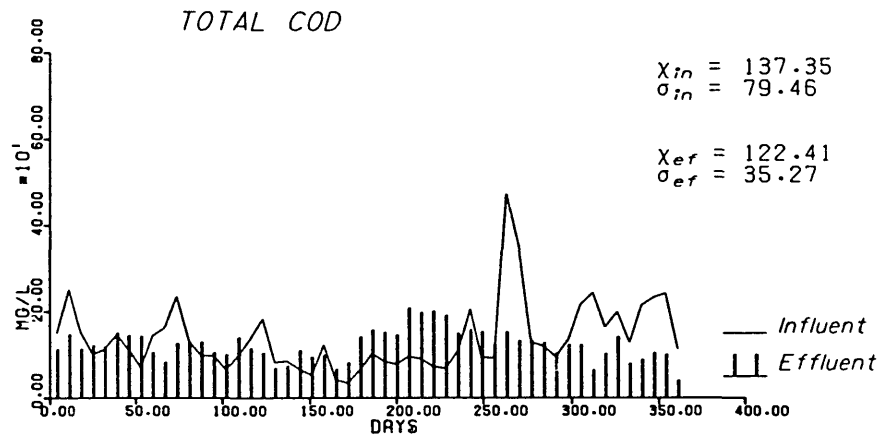


Fig 3.113 7-days Moving Average of Influent & Effluent Total COD (Pond 2)

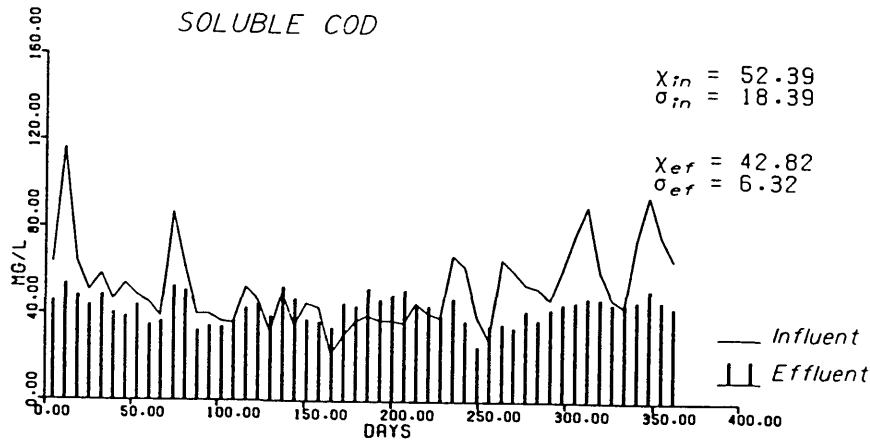


Fig 3.114 7-days Moving Average of Influent & Effluent Soluble COD (Pond 2)

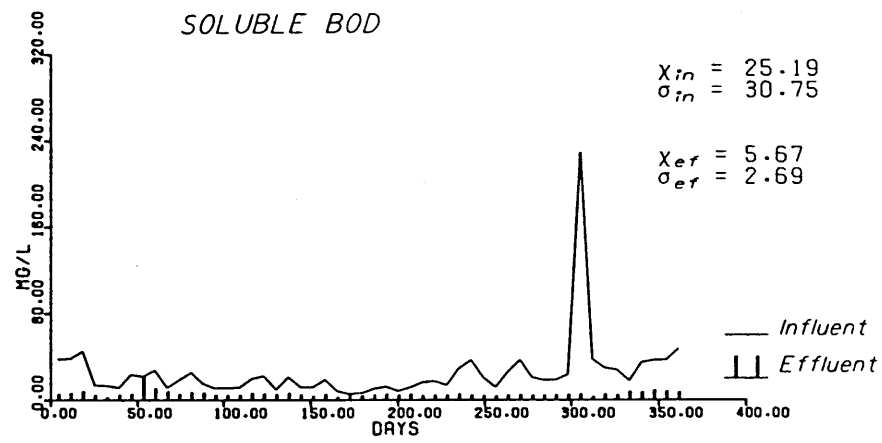


Fig 3.115 7-days Moving Average of Influent & Effluent Soluble BOD (Pond 2)

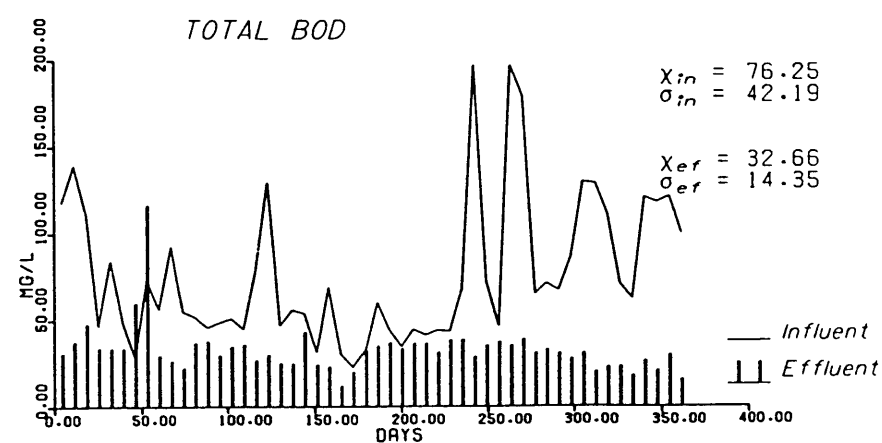


Fig 3.116 7-days Moving Average of Influent & Effluent Total BOD (Pond 2)

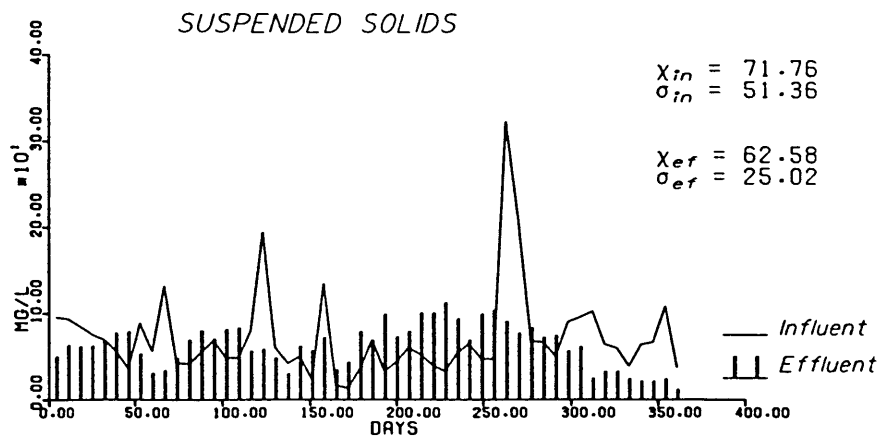


Fig 3.117 7-days Moving Average of Influent & Effluent
Suspended Solids (Pond 2)

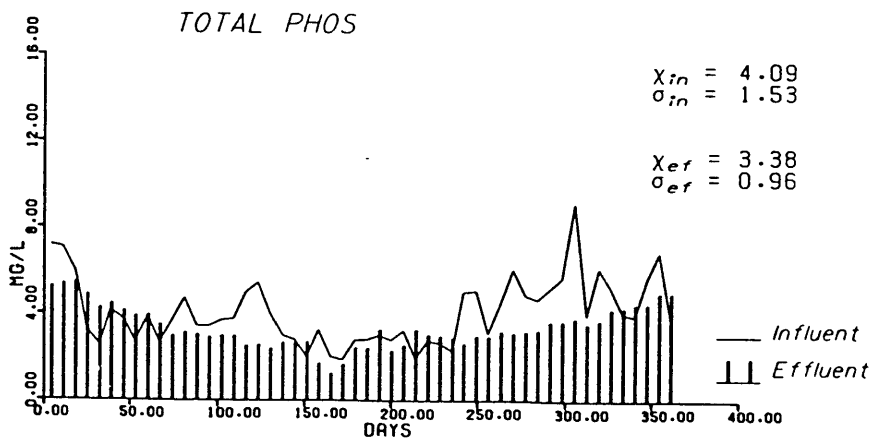


Fig 3.118 7-days Moving Average of Influent & Effluent
Total Phosphorus (Pond 2)

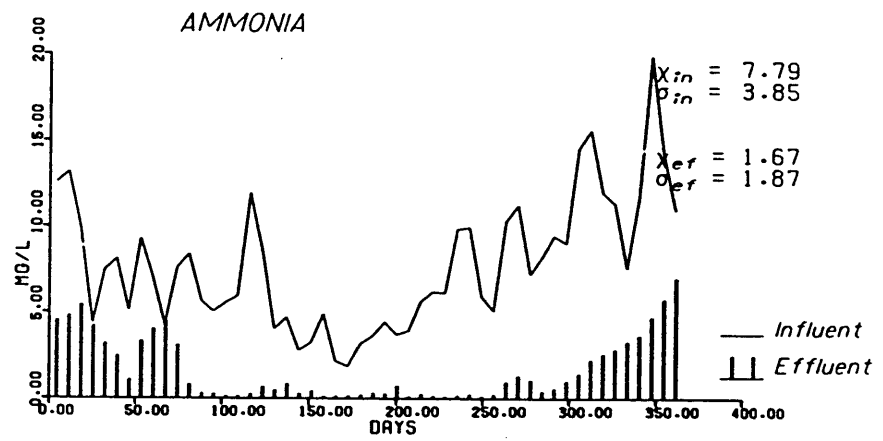


Fig 3.119 7-days Moving Average of Influent & Effluent Ammonia (Pond 2)

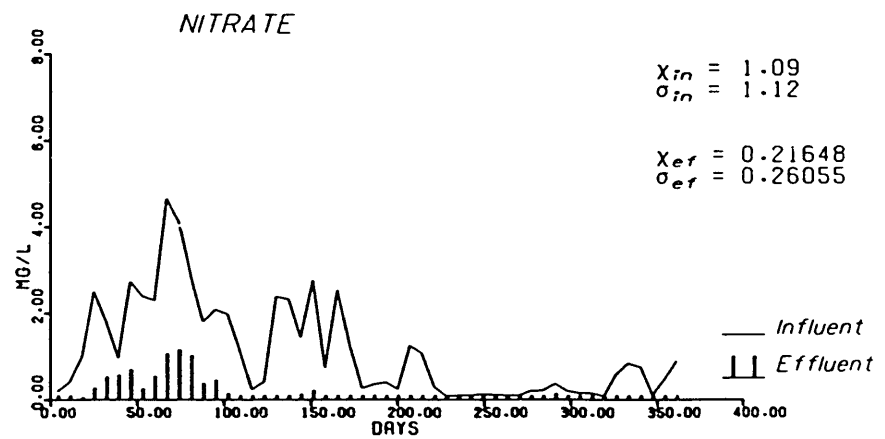


Fig 3.120 7-days Moving Average of Influent & Effluent Nitrate (Pond 2)

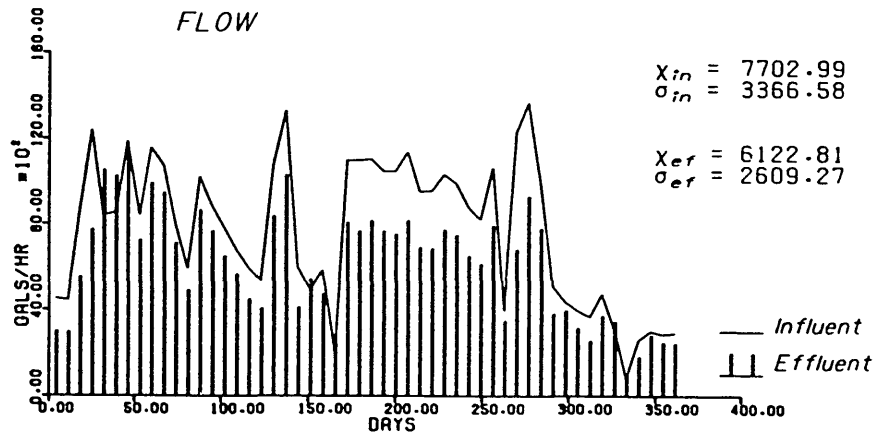


Fig 3.121 7-days Moving Average of Influent & Effluent Flow (Pond 2)

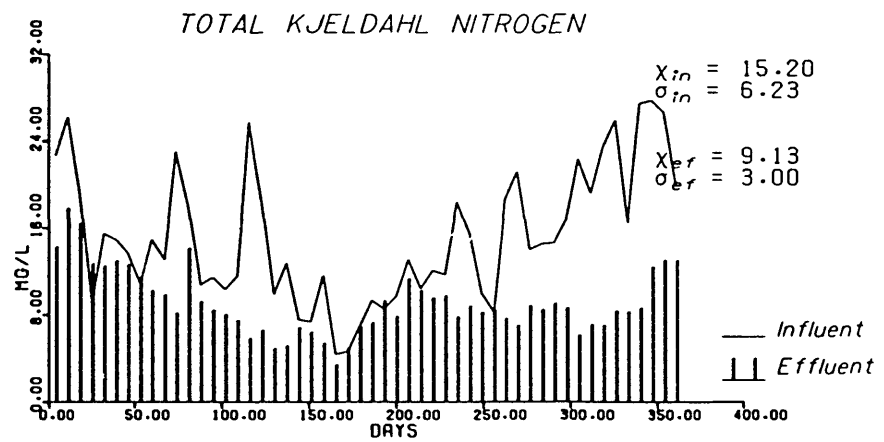


Fig 3.122 7-days Moving Average of Influent & Effluent Total Kjeldahl Nitrogen (Pond 2)

AUTOCORRELOGRAM OF RANDOM NUMBERS

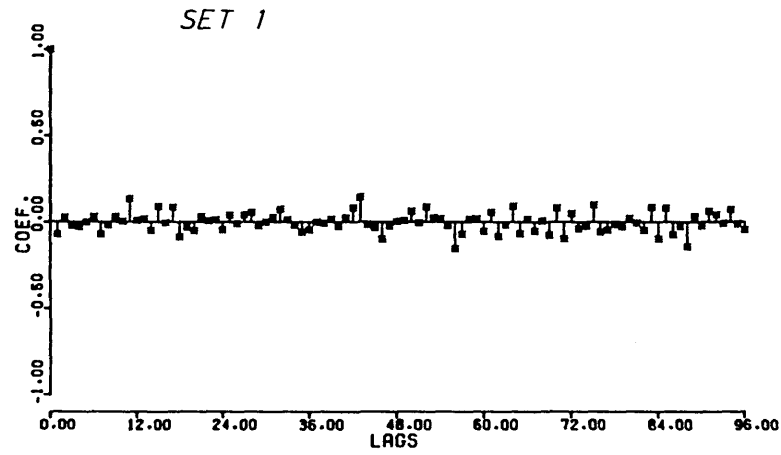


Fig 3.123 Autocorrelogram of Pseudo-Random Numbers
(Set 1)

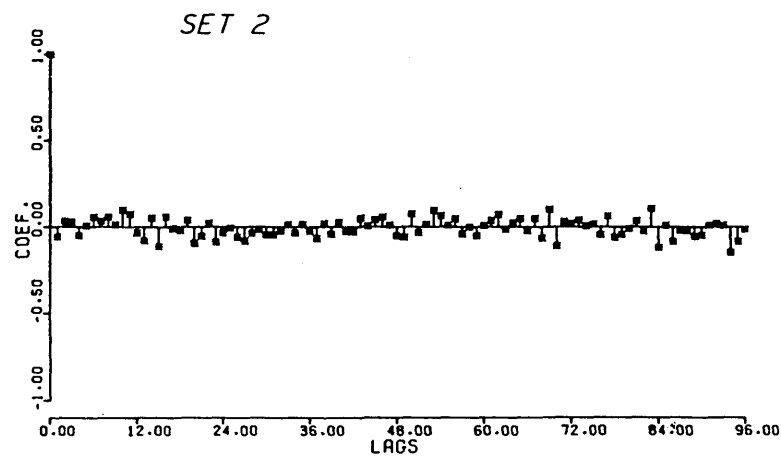


Fig 3.124 Autocorrelogram of Pseudo-Random Numbers
(Set 2)

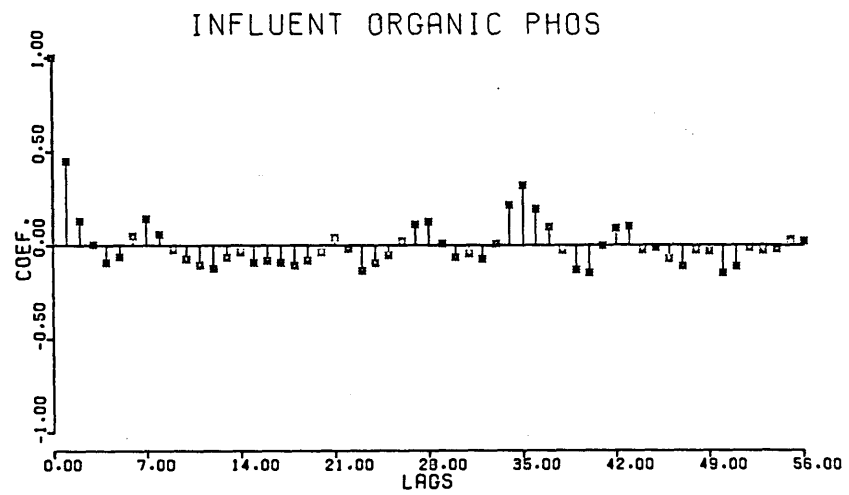


Fig 3.125 Autocorrelogram of Pond 1 Influent Organic Phosphorus

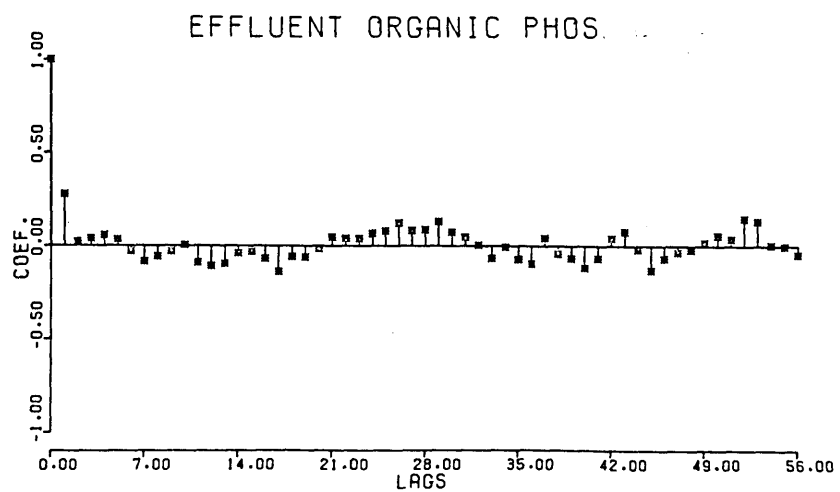


Fig 3.126 Autocorrelogram of Pond 1 Effluent Organic Phosphorus

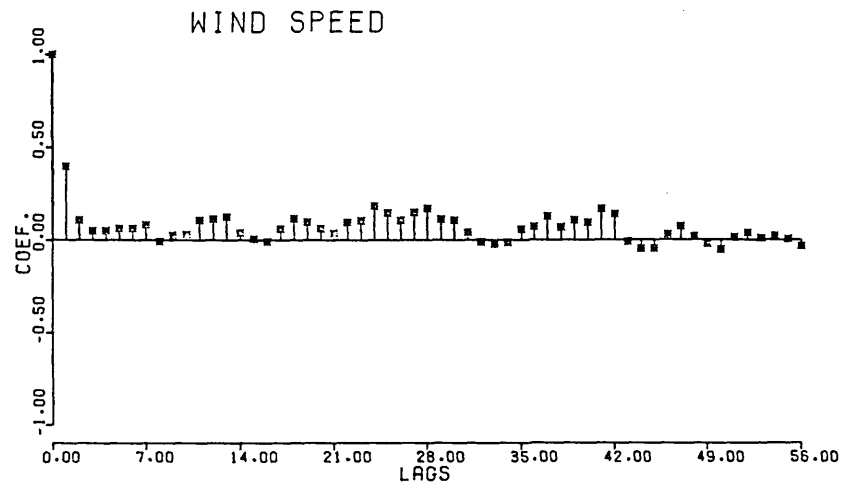


Fig 3.127 Autocorrelogram of Pond 1 Wind Speed

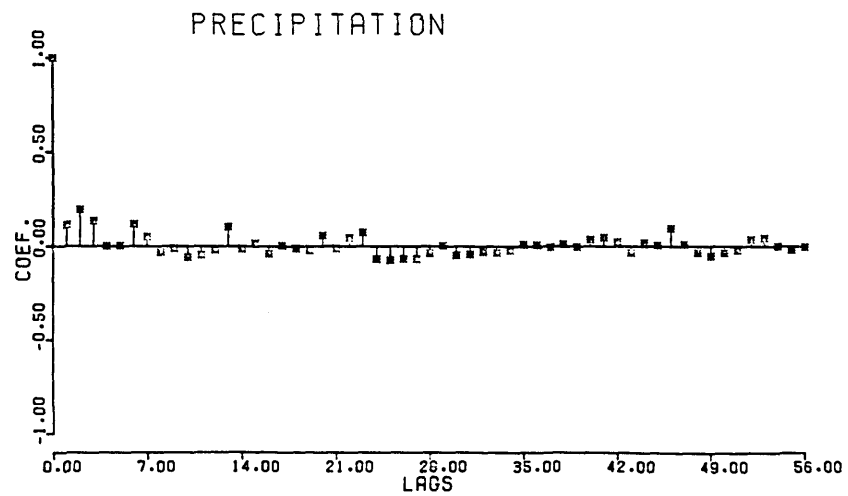


Fig 3.128 Autocorrelogram of Pond 1 Precipitation

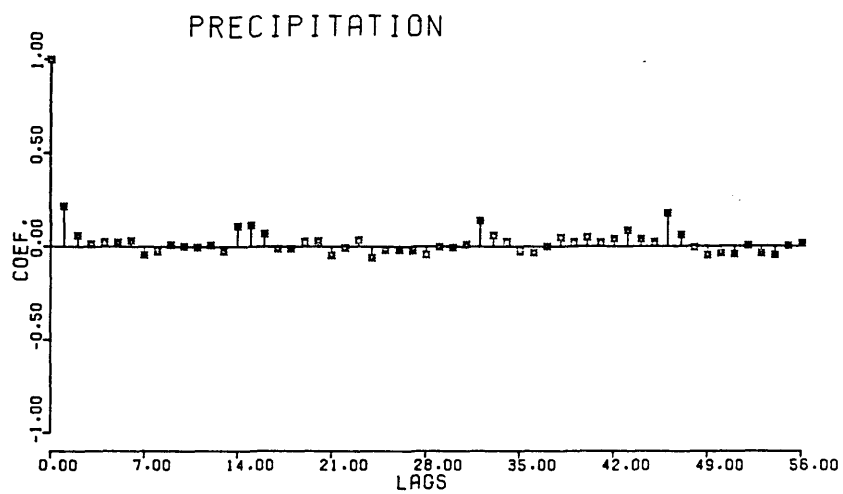


Fig 3.129 Autocorrelogram of Pond 2 Precipitation

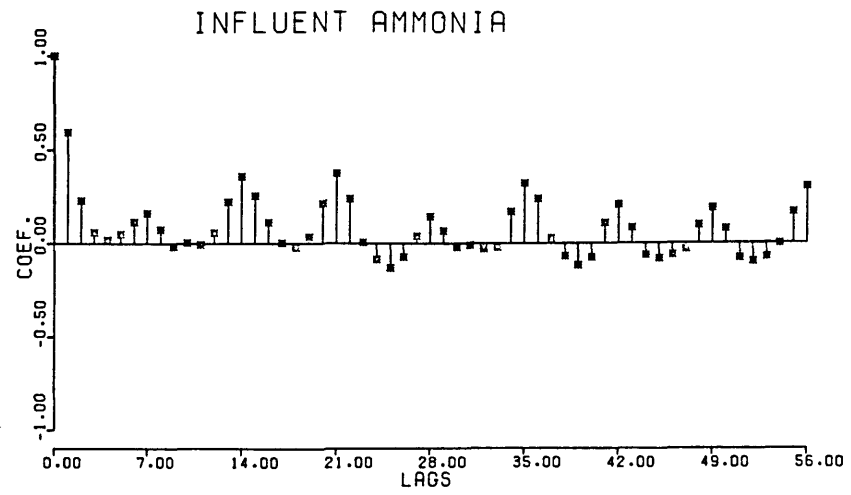


Fig 3.130 Autocorrelogram of Pond 1 Influent Ammonia

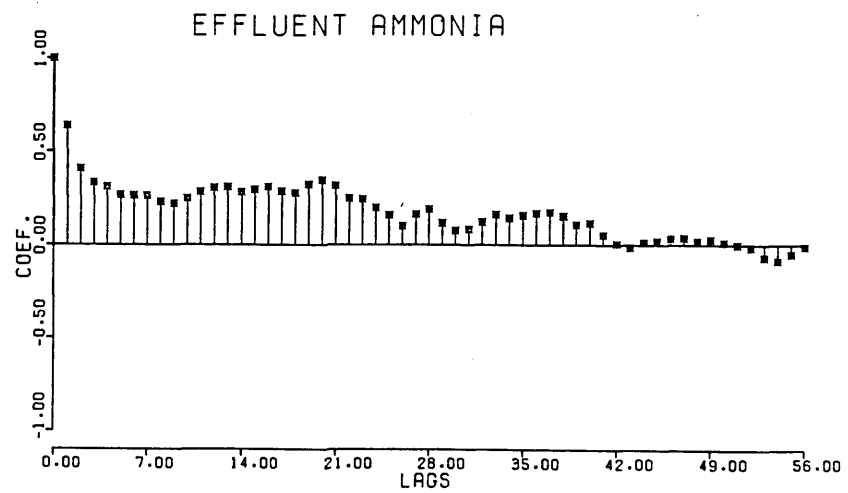


Fig 3.131 Autocorrelogram of Pond 1 Effluent Ammonia

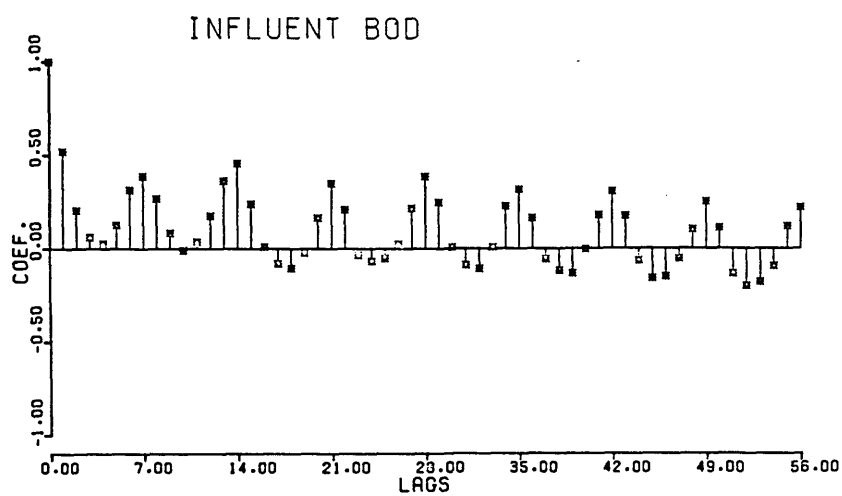


Fig 3.132 Autocorrelogram of Pond 1 Influent BOD

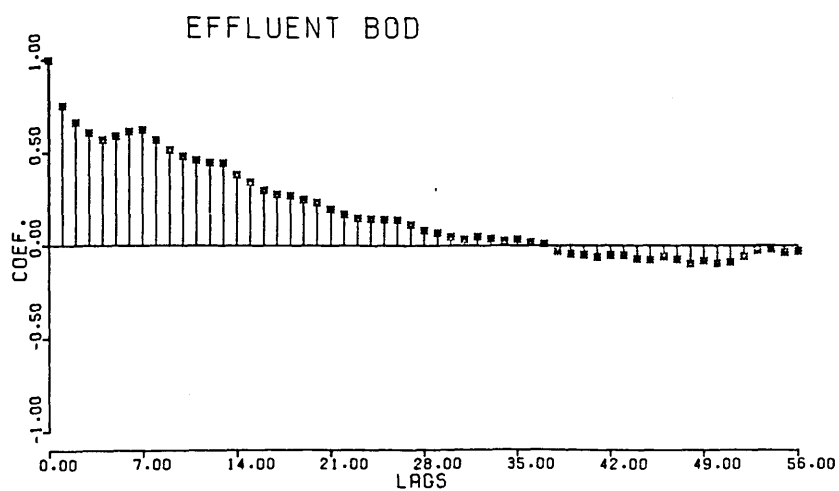


Fig 3.133 Autocorrelogram of Pond 1 Effluent BOD

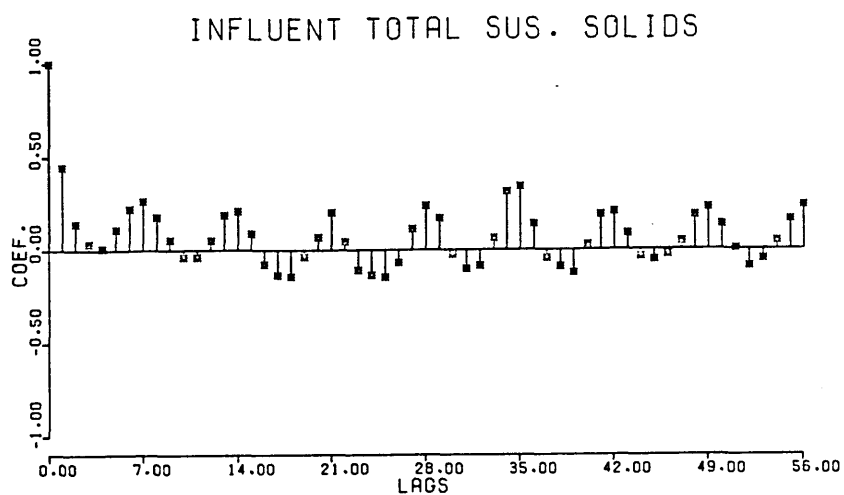


Fig 3.134 Autocorrelogram of Pond 1 Influent Total Suspended Solids

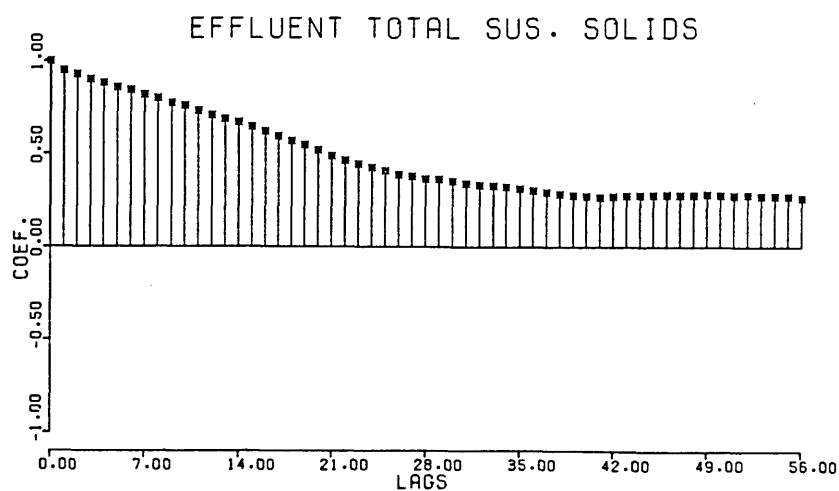


Fig 3.135 Autocorrelogram of Pond 1 Effluent Total Suspended Solids

CROSS-CORRELATIONS FOR POND 1 DATA

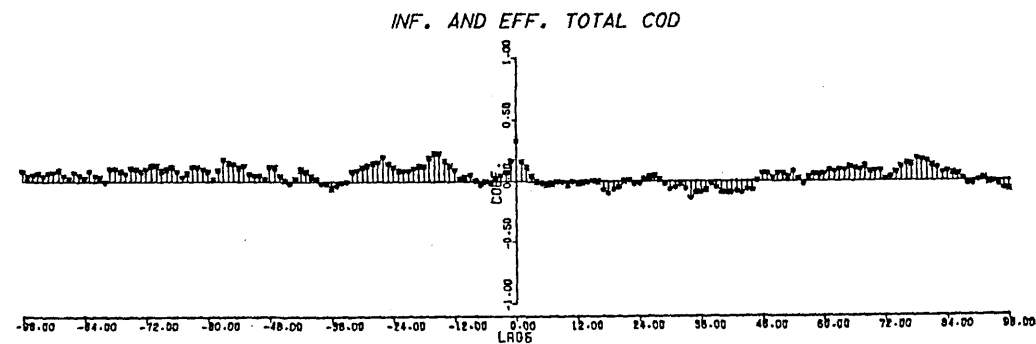


Fig 3.136 Cross-Correlogram for Pond 1 Influent & Effluent Total COD

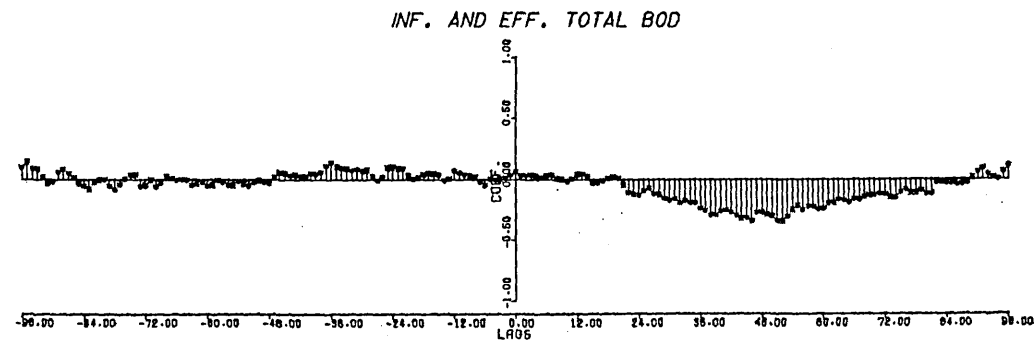


Fig 3.137 Cross-Correlogram for Pond 1 Influent & Effluent Total BOD

CROSS-CORRELATIONS FOR POND 1 DATA

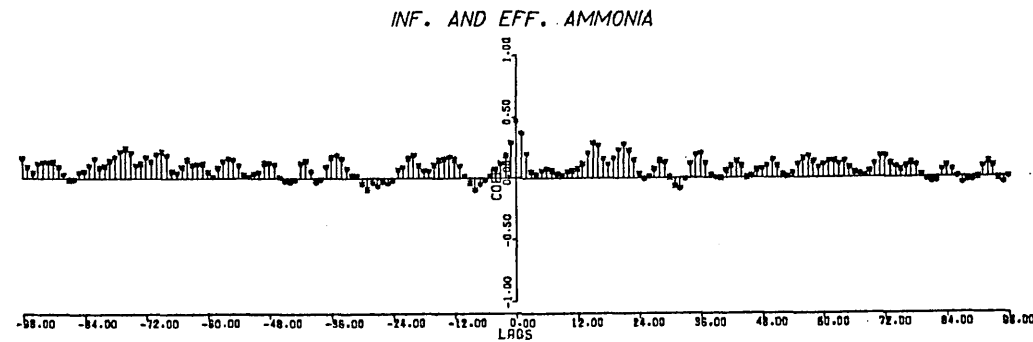


Fig 3.138 Cross-Correlogram for Pond 1 Influent & Effluent Ammonia

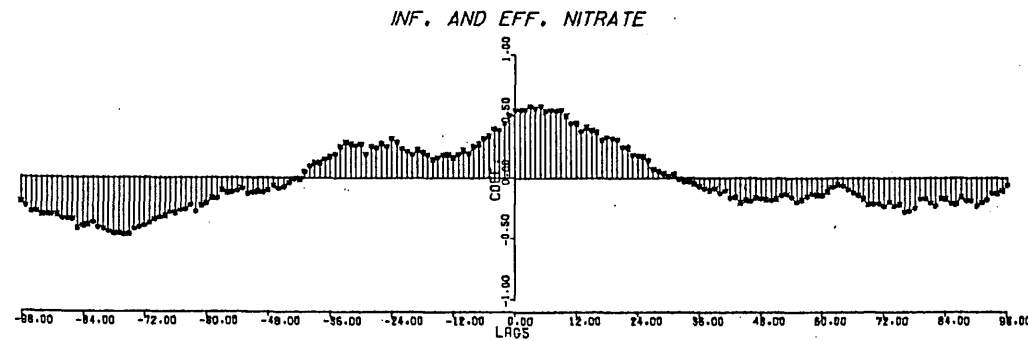


Fig 3.139 Cross-Correlogram for Pond 1 Influent & Effluent Nitrate

CROSS-CORRELATIONS FOR POND 1 DATA

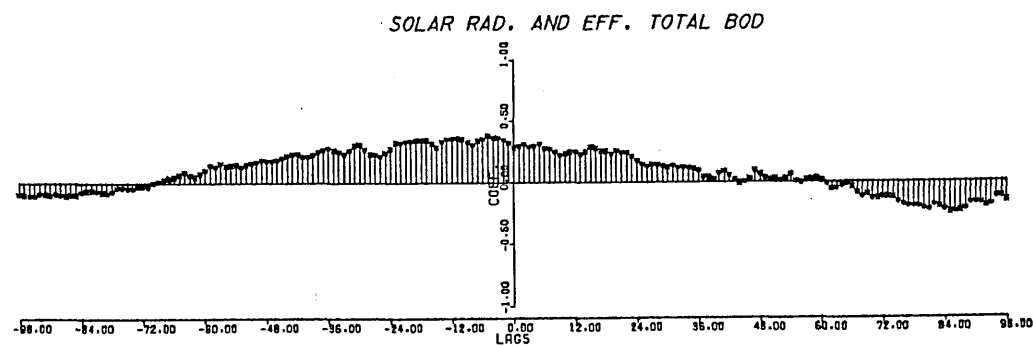


Fig 3.140 Cross-Correlogram for Pond 1 Solar Radiation & Effluent Total BOD

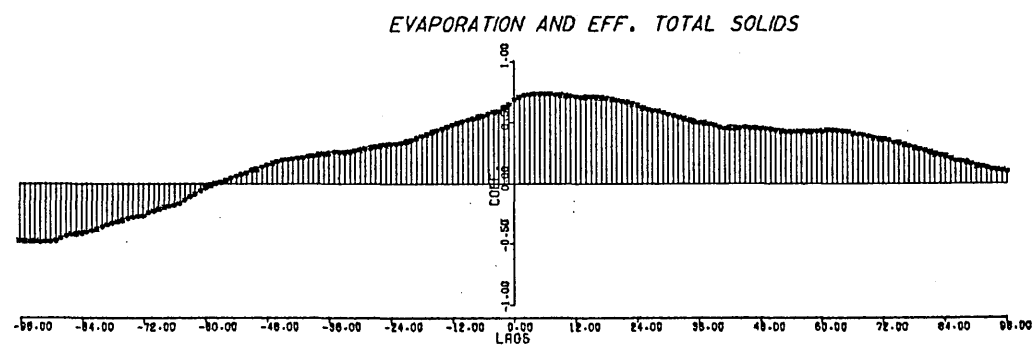


Fig 3.141 Cross-Correlogram for Pond 1 Evaporation & Effluent Total Solids

CROSS-CORRELATIONS FOR POND 1 DATA

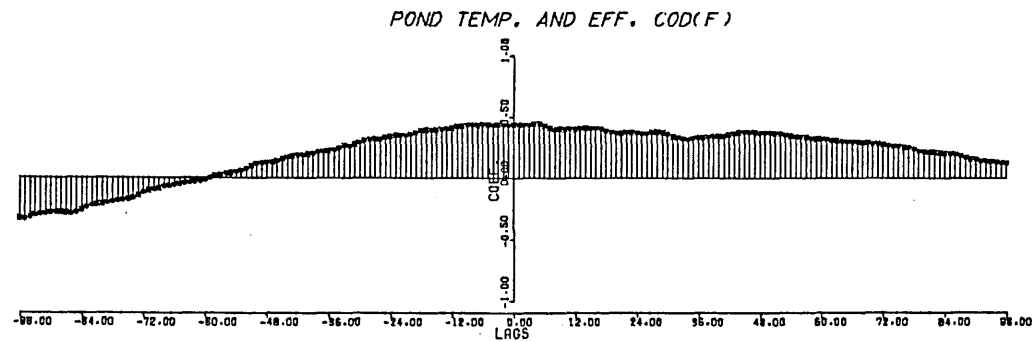


Fig 3.142 Cross-Correlogram for Pond 1 Pond Temperature & Effluent Filtered COD

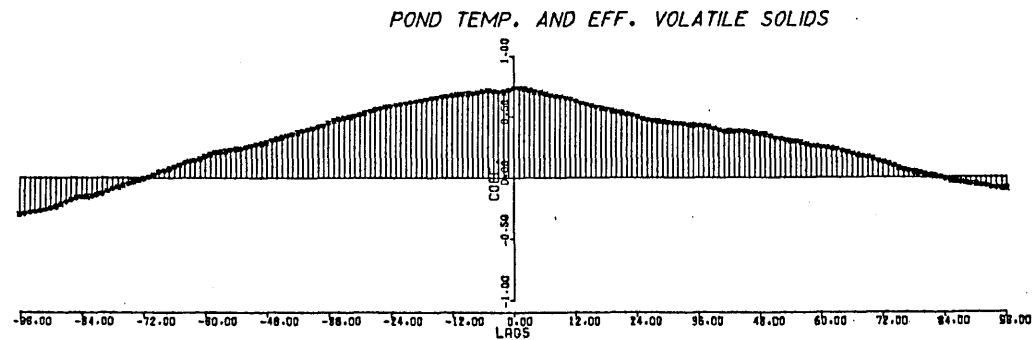


Fig 3.143 Cross-Correlogram for Pond 1 Pond Temperature & Effluent Volatile Suspended Solids

CROSS-CORRELATIONS FOR POND 1 DATA

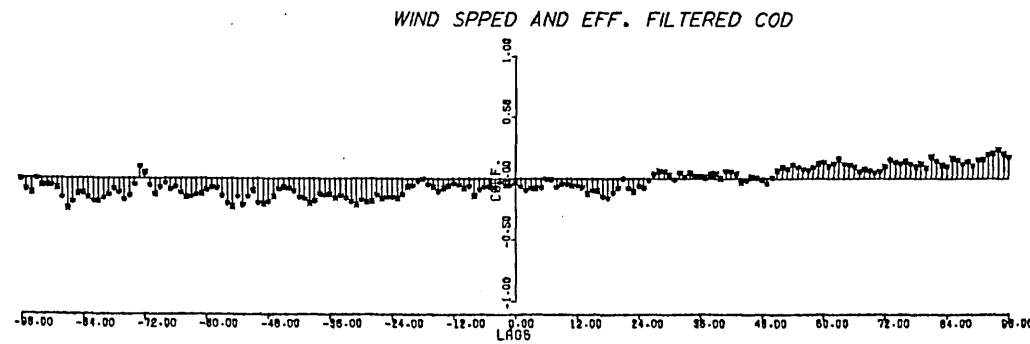


Fig 3.144 Cross-Correlogram for Pond 1 Wind Speed & Effluent Filtered COD

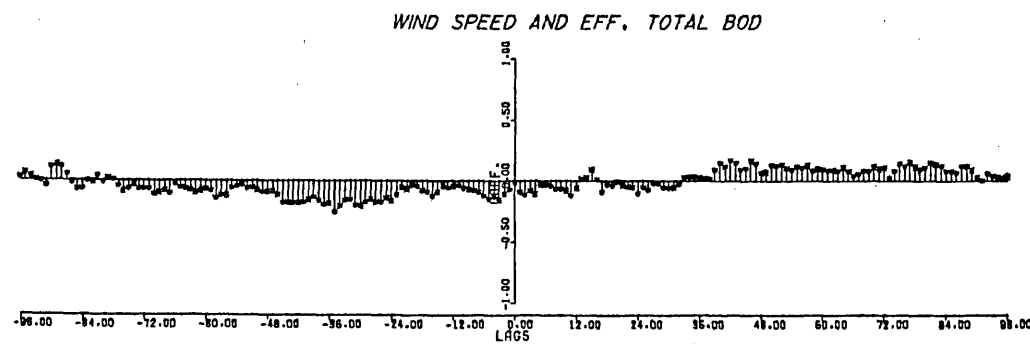


Fig 3.145 Cross-Correlogram for Pond 1 Wind Speed & Effluent Total BOD

CROSS-CORRELATIONS FOR POND 1 DATA

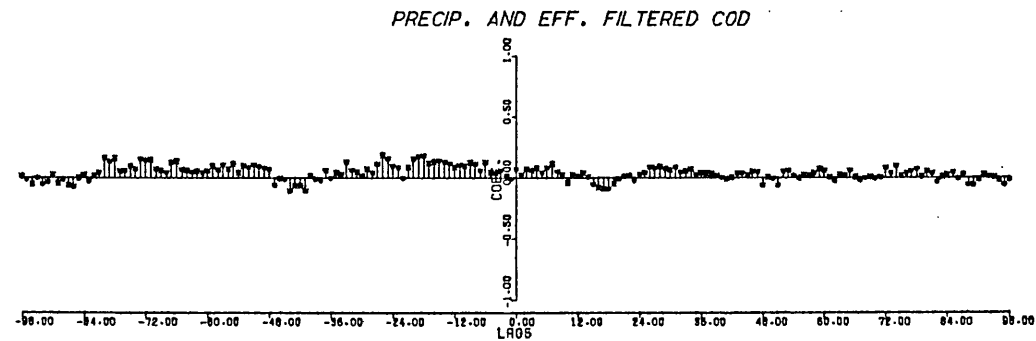


Fig 3.146 Cross-Correlogram for Pond 1 Precipitation & Effluent Filtered COD

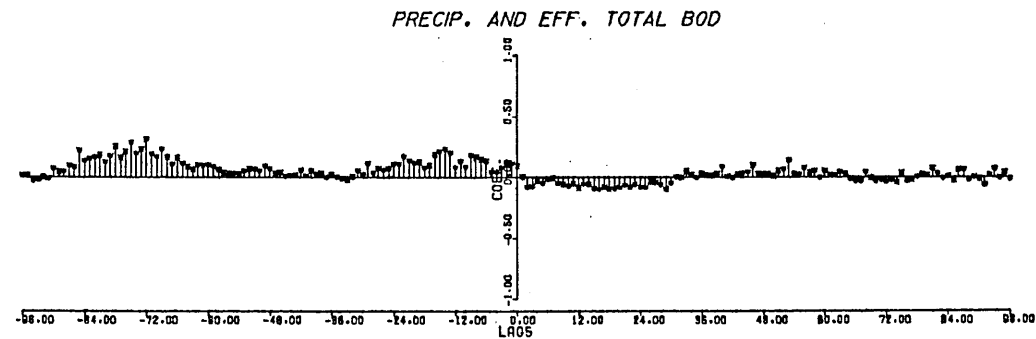


Fig 3.147 Cross-Correlogram for Pond 1 Precipitation & Effluent Total BOD

CROSS-CORRELATIONS FOR POND 2 DATA

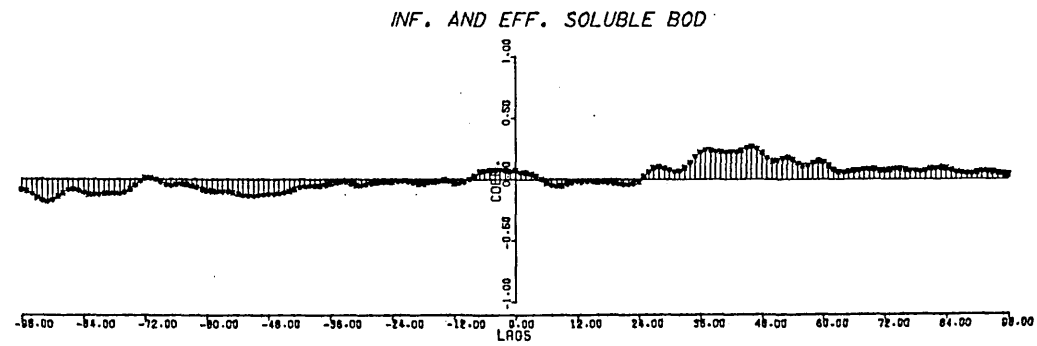


Fig 3.148 Cross-Correlogram for Pond 2 Influent & Effluent Soluble BOD

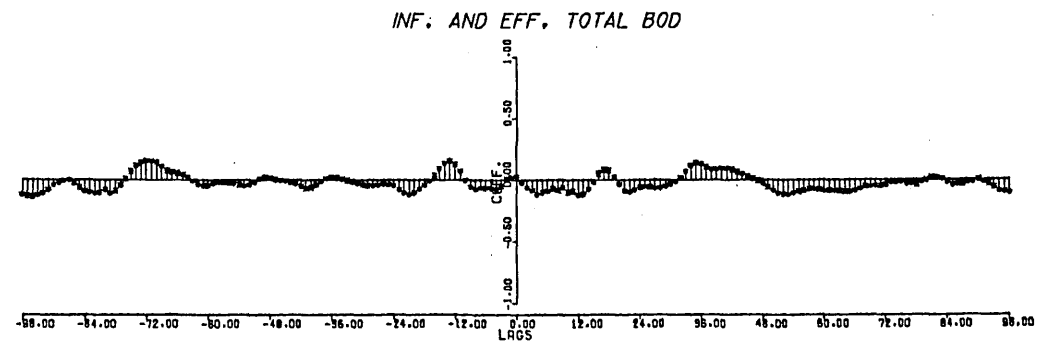


Fig 3.149 Cross-Correlogram for Pond 2 Influent & Effluent Total BOD

CROSS-CORRELATIONS FOR POND 2 DATA

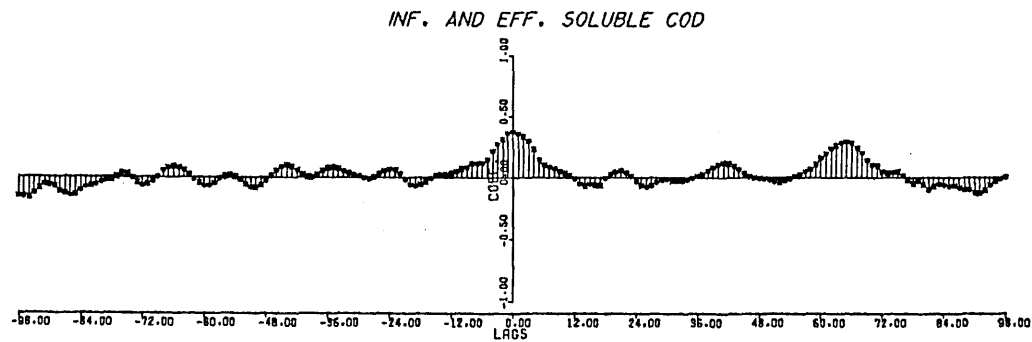


Fig 3.150 Cross-Correlogram for Pond 2 Influent & Effluent Soluble COD

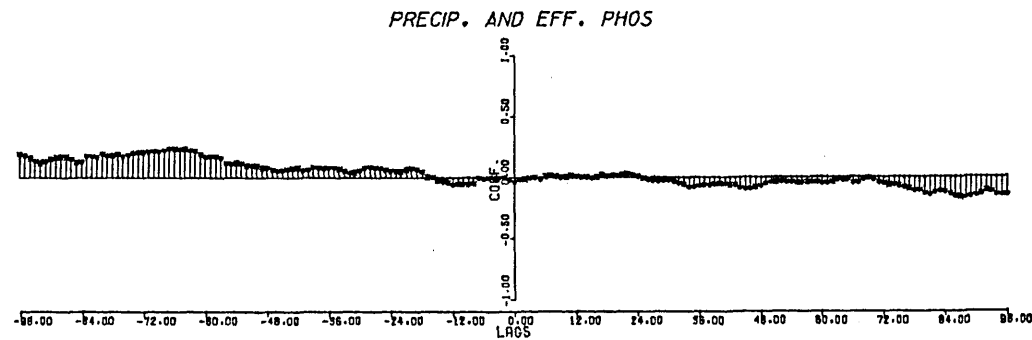


Fig 3.151 Cross-Correlogram for Pond 2 Precipitation & Effluent Total Phosphorous

CROSS-CORRELATIONS FOR POND 2 DATA

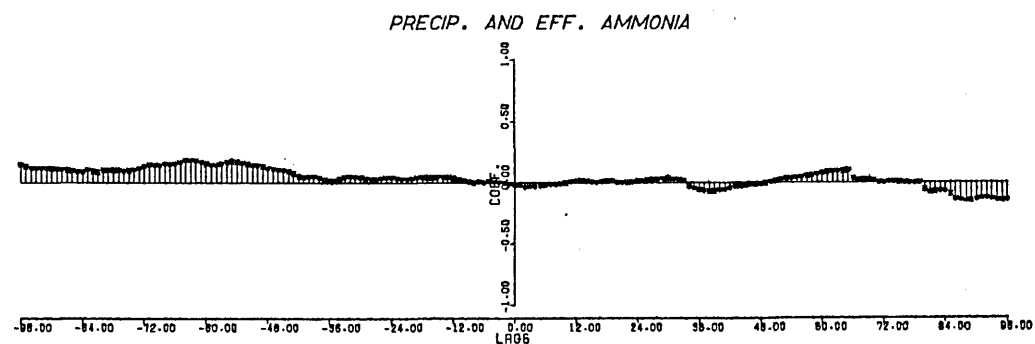


Fig 3.152 Cross-Correlogram for Pond 2 Precipitation & Effluent Ammonia

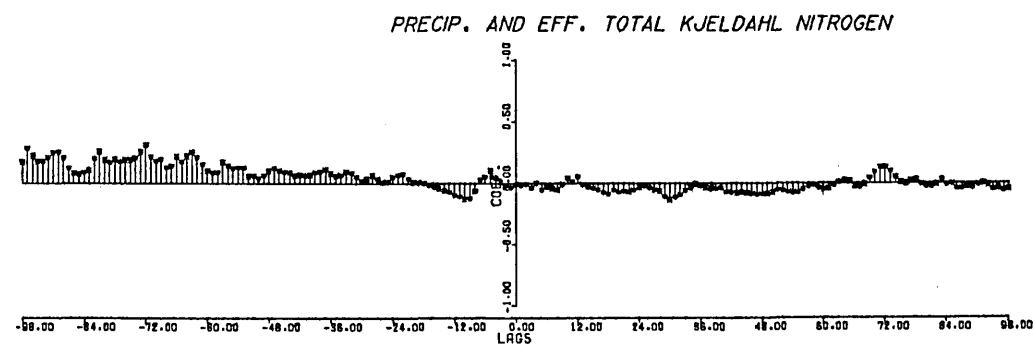


Fig 3.153 Cross-Correlogram for Pond 2 Precipitation & Effluent Total Kjeldahl Nitrogen

CROSS-CORRELATIONS FOR POND 2 DATA

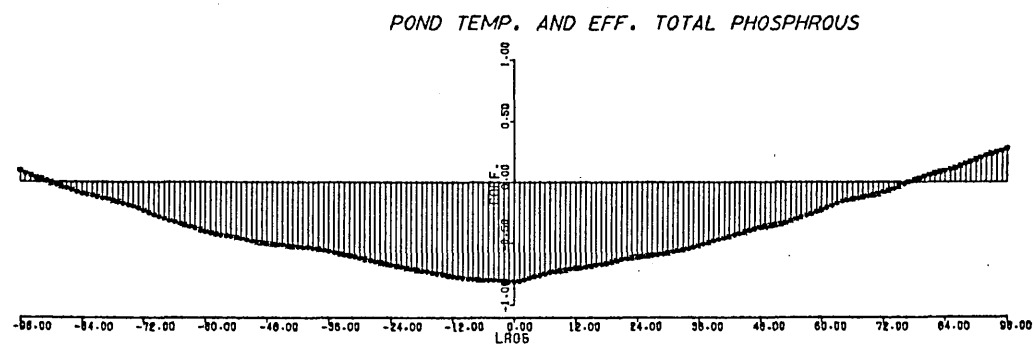


Fig 3.154 Cross-Correlogram for Pond 2 Pond Temperature & Effluent Total Phosphorous

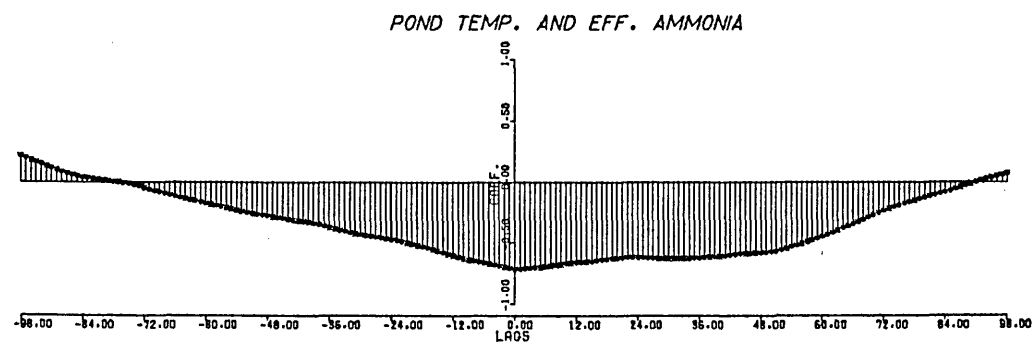


Fig 3.155 Cross-Correlogram for Pond 2 Pond temperature & Effluent Ammonia

CROSS-CORRELATIONS FOR POND 2 DATA

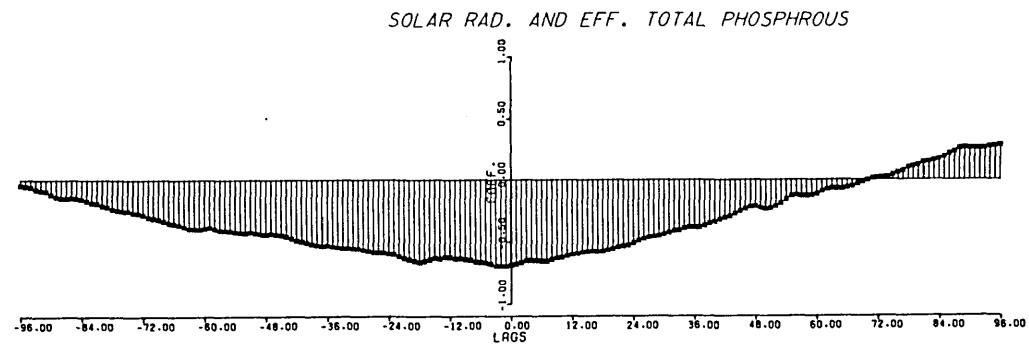


Fig 3.156 Cross-Correlogram for Pond 2 Solar Radiation & Effluent Total Phosphorous

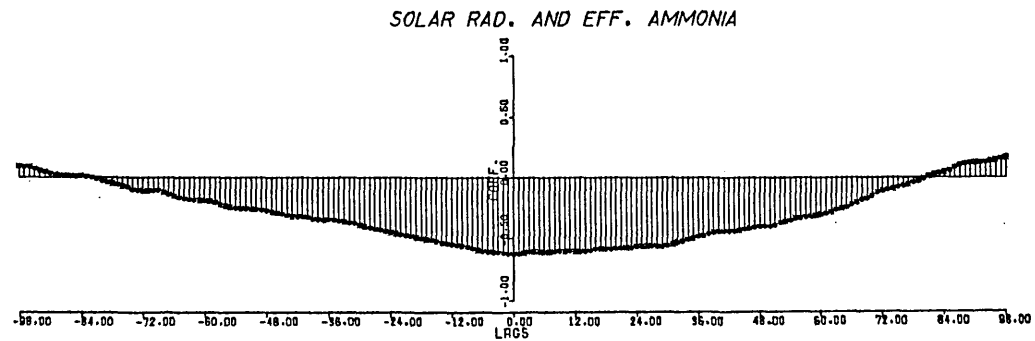


Fig 3.157 Cross-Correlogram for Pond 2 Solar Radiation & Effluent Ammonia

CROSS-CORRELATIONS FOR POND 2 DATA

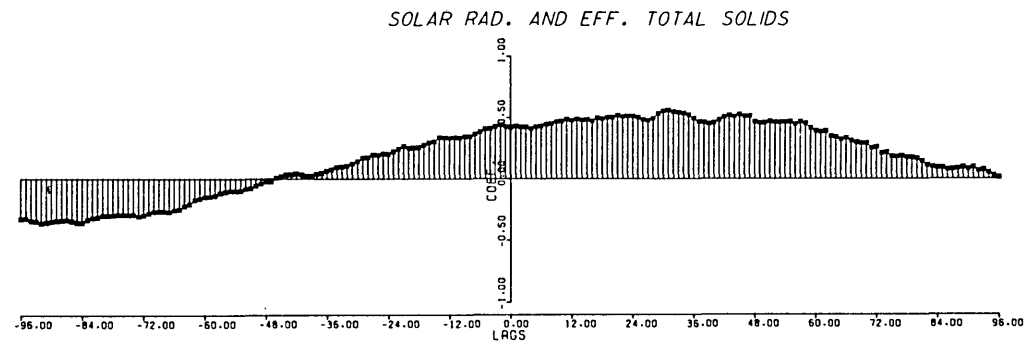


Fig 3.158 Cross-Correlogram for Pond 2 Solar Radiation & Effluent Total Solids

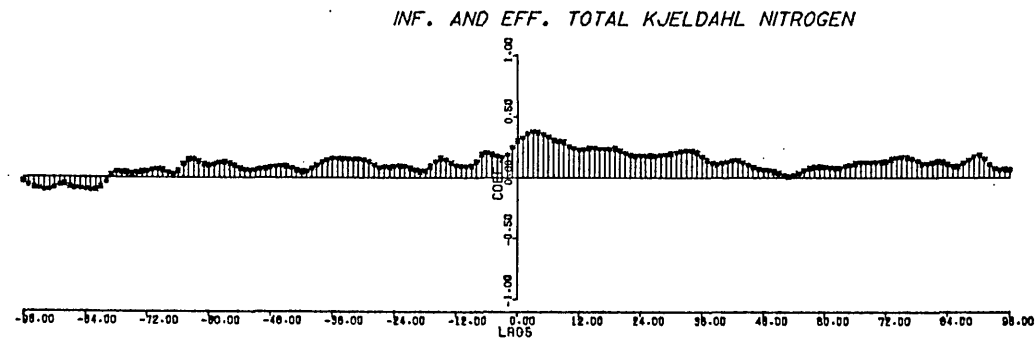


Fig 3.159 Cross-Correlogram for Pond 2 Influent & Effluent total Kjeldahl Nitrogen

TABLE 3.4: Abbreviations for Pond 1 Measured Variables

ABBREVIATION	MEASURED VARIABLE
INCOD	Influent COD
EFCOD	Effluent COD
INCODEF	Influent Filtered COD
EFCODEF	Effluent Filtered COD
INBOD	Influent BOD
EFBOD	Effluent BOD
INTOTS	Influent Total Suspended Solids
EFTOTS	Effluent Total Suspended Solids
INVOLS	Influent Volatile Suspended Solids
EFVOLS	Effluent Volatile Suspended Solids
INFLOW	Influent Flow
EFFLOW	Effluent Flow
AIRTMP	Air Temperature
LAGTMP	Pond Temperature
SOLAR	Solar Radiation
EVAPO	Evaporation
PRECIP	Precipitation
WIND	Wind Speed
INOTOP	Influent Ortho-phosphorous
EFOTOP	Effluent Ortho-phosphorous
INMETP	Influent Meta-phosphorous
EFMETP	Effluent Meta-phosphorous
INORGP	Influent Organic-phosphorous
EFORGP	Effluent Organic-phosphorous
INN3	Influent Ammonia
EFN3	Effluent Ammonia
INNO3	Influent Nitrate
EFNO3	Effluent Nitrate

TABLE 3.5: Correlations For Set 1 Pond 1 Data

	INOD	EFOOD	INCOF	EFCOF	INBOD	EFBOD	INTOTS	EFTOTS	INVOLS	EFVOLS	INFLOW	EFFLOW	AIRIMP	LAGIMP	SOLAR	EVAPO	PRECIP	WIND
INOD	1.00																	
EFOOD	.33	1.00																
INCOF	.62	.12	1.00															
EFCOF	.24	.56	.14	1.00														
INBOD	.53	-.10	.55	-.17	1.00													
EFBOD	.06	.33	-.20	.25	.06	1.00												
INTOTS	.31	-.23	.38	-.21	.64	-.12	1.00											
EFTOTS	.00	.49	-.27	.53	-.20	.65	-.28	1.00										
INVOLS	.42	-.14	.45	-.17	.73	-.09	.65	-.25	1.00									
EFVOLS	-.06	.46	-.28	.51	-.19	.61	-.28	.94	-.22	1.00								
INFLOW	-.19	-.15	-.18	-.12	0.28	-.19	-.13	-.18	-.19	-.20	1.00							
EFFLOW	-.16	-.30	-.03	-.27	-.05	-.34	-.04	-.49	-.03	-.46	.66	1.00						
AIRIMP	.06	.38	-.25	.44	-.21	.51	-.20	.79	-.20	.72	-.10	-.53	1.00					
LAGIMP	.04	.38	-.28	.44	-.23	.50	-.22	.80	-.21	.73	-.11	-.49	.96	1.00				
SOLAR	.04	.21	-.05	.27	.06	.28	.08	.47	-.01	.47	-.30	-.33	.49	.50	1.00			
EVAPO	-.02	.30	-.28	.37	-.29	.36	-.18	.68	-.21	.60	.02	-.54	.85	.83	.29	1.00		
PRECIP	.00	.06	-.12	.06	-.11	.09	-.16	.08	-.09	.11	.09	.35	.04	.10	-.01	-.10	1.00	
WIND	-.20	-.10	-.17	-.04	-.08	-.02	-.04	-.04	.00	.04	.02	.04	-.11	-.06	-.05	.08	.07	1.00

TABLE 3.6

FACTOR LOADING MATRIX FOR SET 1 POND 1 DATA

	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	18
INOD	-.02	.73	.45	.02	-.13	-.11	.07	-.18	-.26	.06	-.05	-.22	-.24	-.11	-.01	-.01	.01	.00
EFOD	.54	.21	.55	-.19	.13	-.10	.09	.13	-.14	-.48	.04	.09	.08	.07	-.01	.00	.00	.00
INODF	-.32	.69	.38	-.17	-.06	-.11	-.11	.01	-.04	-.27	.04	.35	.12	-.07	-.03	-.01	.00	.00
EFODF	.58	.16	.50	-.16	.05	-.24	-.20	.18	.36	.15	-.15	-.21	.08	.06	.02	.01	.00	.00
INBOD	-.32	.80	-.10	.27	.04	.04	.15	.08	-.04	.10	.11	-.01	-.04	.33	.08	.01	.00	.00
EFBOD	.63	.15	-.04	.27	.22	.18	.51	.20	-.12	.13	-.21	-.04	.18	.08	.00	-.02	.01	-.01
INITOTS	-.36	.64	-.30	.27	-.19	-.04	-.10	.09	.17	-.22	-.37	.12	-.10	-.02	-.06	.01	.00	.00
EFTOTS	.93	.06	.02	.12	.05	.04	.10	.13	.11	.04	.09	.09	-.17	-.03	-.01	-.05	-.16	.05
INVOLS	-.35	.71	-.16	.31	-.06	-.13	.04	-.02	.20	-.15	.29	-.14	.18	-.16	.00	-.01	-.01	.00
EFVOLS	.88	.03	-.01	.15	.15	.02	.06	.17	.17	.02	.17	.12	-.21	-.07	.00	.02	.13	-.03
INFLOW	-.21	-.52	.27	.34	-.59	-.19	.09	.26	-.06	.00	.01	.04	.00	-.04	.15	.11	-.01	.00
EFFLOW	-.59	-.43	.36	.44	-.10	.04	-.07	.23	-.03	.03	.06	-.04	-.01	.05	-.20	-.14	.02	.00
AIRIMP	.90	.08	-.08	.12	-.29	.01	-.06	-.16	-.05	.03	.01	.00	.08	.05	-.11	.06	.05	.13
LAGIMP	.90	.05	-.07	.17	-.22	-.01	-.09	-.17	-.05	.03	.04	-.01	.05	.06	-.16	.08	-.04	-.12
SOLAR	.52	.27	-.29	.13	.09	.23	-.59	.31	-.27	.00	.01	.06	.05	-.05	.88	-.02	.00	.00
EVAPO	.80	-.02	-.19	.01	-.38	-.28	.02	-.20	-.01	-.01	-.03	.05	.05	.04	.12	-.19	.03	.02
PRECIP	.06	-.22	.41	.63	.35	.14	-.17	-.42	.07	-.03	-.07	.08	.03	.00	.10	.01	-.01	.00
WIND	-.03	-.20	-.28	.19	.43	-.79	-.05	.05	-.15	.04	-.02	.01	-.01	.00	.04	.03	-.01	.01

TABLE 3.7

CORRELATIONS FOR SET 2 POND 1 DATA

	INTOP	EFOTOP	INMEIP	EFMEIP	INORGP	EFORGP	INN3	EFN3	INN3	EFN3	INFLOW	EFFLOW	LAGIMP	SOLAR	EVAPO	PRECIP	WIND
INTOP	1.00																
EFOTOP	.21	1.00															
INMEIP	-.08	-.17	1.00														
EFMEIP	-.10	-.12	.15	1.00													
INORGP	.38	.08	-.09	-.06	1.00												
EFORGP	-.16	-.10	-.01	-.13	-.09	1.00											
INN3	.55	.21	-.20	-.14	.26	-.13	1.00										
EFN3	.19	.58	-.11	-.12	-.01	-.08	.46	1.00									
INN3	.02	-.24	-.18	.00	-.09	-.07	.05	-.15	1.00								
EFN3	-.01	-.33	-.03	.13	-.13	-.07	.00	-.24	.54	1.00							
INFLOW	-.22	-.08	.11	.03	-.15	.04	-.30	-.13	.09	.02	1.00						
EFFLOW	-.02	.16	-.06	-.09	.01	-.02	-.04	.06	.12	-.01	.65	1.00					
LAGIMP	-.19	-.46	.29	.22	-.13	.10	-.33	-.35	-.06	.01	.10	-.44	1.00				
SOLAR	.03	-.12	.14	.08	.17	-.05	.09	-.13	.08	.15	-.17	-.29	.31	1.00			
EVAPO	-.18	-.40	.24	.22	-.14	.08	-.28	-.28	-.02	.06	.09	-.53	.91	.32	1.00		
PRECIP	.04	-.07	-.01	.03	.04	-.02	-.08	-.02	.06	.02	.21	.41	.08	-.08	-.08	1.00	
WIND	-.04	-.05	-.15	.03	-.13	-.03	-.07	-.03	.18	.18	.02	.01	-.02	-.02	.08	.03	1.00

TABLE 3.8

FACTOR LOADING MATRIX FOR SET 2 POND 1 DATA

	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17
INOTOP	.45	-.40	.28	-.41	.03	.17	.05	.15	-.32	.18	-.18	-.22	-.32	-.05	.16	.00	.00
EFOTOP	.68	-.10	-.31	.12	-.29	.11	.05	-.20	.22	.01	-.01	-.42	-.11	.05	-.23	-.01	-.01
INMEIP	-.36	-.03	-.31	-.33	-.22	-.33	.36	-.01	-.28	.42	.28	-.08	.16	-.08	-.06	.01	.00
EFMEIP	-.33	-.02	.04	-.16	-.59	-.18	-.28	.37	.37	.22	-.28	.04	-.02	-.09	.01	.00	.00
INORCP	.28	-.32	.12	-.57	.32	-.02	-.35	-.21	.12	.13	-.14	-.09	.38	.11	-.01	.00	.00
EFORCP	-.15	.10	-.24	.25	.61	.18	.25	.29	.32	.41	-.11	-.07	.00	-.01	.00	.00	.00
INNHB	.57	-.45	.31	-.14	-.02	.12	.29	.16	-.05	.03	-.14	.38	.01	-.02	-.26	.01	-.01
EFNHB	.62	-.20	-.20	.14	-.36	.27	.37	.07	.11	-.06	.00	.03	.29	.08	.24	-.02	-.01
INNO3	-.10	.25	.77	.10	.02	-.02	.23	-.01	.05	-.14	-.09	-.24	.20	-.38	-.02	.00	-.01
EFNO3	-.25	.15	.76	.09	-.08	-.18	.19	.10	.05	.07	.06	-.12	-.01	.46	-.02	-.02	-.01
INFLOW	-.14	.73	-.14	-.28	-.08	.11	.23	-.28	-.03	.01	-.39	.06	-.04	.07	.01	.17	-.05
EFFLOW	.40	.80	.00	-.31	.00	.00	.08	-.11	.06	.08	-.05	.11	-.06	-.02	.01	-.23	.07
LAGTMP	-.84	-.21	-.16	-.19	-.03	.31	.08	.04	-.06	-.15	-.09	-.06	.01	.03	-.04	-.16	-.14
SOLAR	-.32	-.45	.23	-.25	-.01	.01	.21	-.42	.47	.05	.24	.12	-.21	-.07	.06	.01	.00
EVAPO	-.82	-.29	-.10	-.05	-.09	.33	.12	-.05	-.07	-.11	-.20	-.07	.06	.06	-.05	.00	.17
PRECIP	.03	.42	.04	-.50	-.01	.44	-.08	.36	.12	-.16	.44	-.04	.01	.01	-.05	.08	.01
WIND	-.08	.15	.34	.35	-.23	.53	-.29	-.24	-.16	.46	.12	.09	.05	-.03	-.02	.00	-.02

TABLE 3.9: Abbreviations for Pond 2 Measured Variables

ABBREVIATION	MEASURED VARIABLE
INCOD	Influent Total COD
EFCOD	Effluent Total COD
INSCOD	Influent Soluble COD
EFSCOD	Effluent Soluble COD
INSBOD	Influent Soluble BOD
EFSBOD	Effluent Soluble BOD
INBOD	Influent Total BOD
EFBOD	Effluent Total BOD
INTOTS	Influent Total Solids
EFTOTS	Effluent Total Solids
INTOTP	Influent Total Phosphorous
EFTOTP	Effluent Total Phosphorous
INN ₃	Influent Ammonia
EFN ₃	Effluent Ammonia
INNO ₃	Influent Nitrate
EFNO ₃	Effluent Nitrate
INTKN	Influent Total Kjeldahl Nitrogen
EFTKN	Effluent Total Kjeldahl Nitrogen
PRECIP	Precipitation
SOLAR	Solar Radiation
INFLOW	Influent Flow
EFFLOW	Effluent Flow
LAGTMP	Pond Temperature

TABLE 3.10

CORRELATIONS FOR SET 1 POND 2 DATA

	INCOO	EFCOO	INSCOO	EFSCOO	INSBOD	EFBOD	INBOD	EFBOD	INTOIS	EFTOIS	INTOIP	EFTOIP	INN13	EFN13	INN03	EFN03	INTKN	EFTKN
INCOO	1.00																	
EFCOO	.09	1.00																
INSCOO	.57	-.09	1.00															
EFSCOO	.02	.00	.37	1.00														
INSBOD	.26	-.05	.32	.07	1.00													
EFBOD	.03	-.03	.12	.13	.07	1.00												
INBOD	.81	-.01	.53	.03	.31	.13	1.00											
EFBOD	.08	.32	-.07	.14	.03	.62	.03	1.00										
INTOIS	.76	.07	.24	-.09	.14	.09	.69	.10	1.00									
EFTOIS	-.14	.61	-.27	-.23	-.09	-.26	-.16	.31	.04	1.00								
INTOIP	.45	-.02	.45	.10	.44	.04	.48	-.10	.24	-.17	1.00							
EFTOIP	.30	-.08	.48	.21	.22	.26	.30	.09	.08	-.29	.43	1.00						
INN13	.57	-.07	.74	.24	.39	.25	.62	.00	.27	-.30	.59	.53	1.00					
EFN13	.30	-.32	.50	.25	.16	.38	.31	-.03	.09	-.60	.30	.82	.49	1.00				
INN03	-.23	-.27	-.16	-.08	-.17	.15	-.34	.08	-.22	-.22	-.34	-.07	-.41	.12	1.00			
EFN03	.04	-.03	.06	.00	-.08	.13	-.13	.09	-.03	-.06	-.13	.07	-.12	.16	.69	1.00		
INTKN	.64	-.04	.73	.27	.32	.17	.61	-.08	.37	-.34	.57	.52	.84	.51	-.26	.04	1.00	
EFTKN	.12	.18	.36	.22	.02	.28	.14	.31	.02	.04	.22	.67	.30	.55	.01	.24	.30	1.00

TABLE 3.11

FACTOR LOADING MATRIX FOR SET 1 POND 2 DATA

	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	18
INOD	.74	.36	.00	-.46	.13	-.11	-.03	.06	.04	-.11	-.07	.04	-.04	-.05	-.10	.15	.16	.07
EFOD	-.10	.39	.69	.13	.30	-.07	.10	-.09	.20	-.41	-.12	-.01	.00	.03	.00	-.02	-.07	-.03
INSCOD	.80	-.04	-.06	.06	.20	-.17	.22	-.12	-.30	.09	-.26	-.02	-.18	-.04	-.06	-.10	-.02	-.04
EFSCOD	.30	-.24	.01	.41	.06	-.61	.40	.32	.36	.07	.05	.03	.04	-.04	.04	.03	.01	.00
INSEOD	.44	.16	-.06	.11	-.19	.54	.53	.32	-.14	-.16	.05	-.07	-.01	.03	.01	.00	.01	.00
EFSEOD	.29	-.44	.45	-.10	-.63	-.06	.04	-.17	.08	-.07	-.01	-.13	-.14	-.12	.13	.01	.06	-.03
INBOD	.76	.38	.02	-.32	-.10	-.06	-.09	.08	-.04	.05	-.14	-.08	.31	-.06	.12	-.07	-.01	-.03
EFBOD	.00	-.15	.83	-.06	-.41	.00	.10	.01	-.06	.15	-.03	.17	.10	.09	-.16	-.04	.00	.05
INIOIS	.47	.44	.14	-.59	-.09	-.15	-.16	.27	.06	.07	.15	.00	-.18	.07	.00	-.03	-.11	-.06
EFIOIS	-.39	.50	.59	.07	.30	.12	.04	.03	-.18	.19	.06	.09	-.07	-.13	.17	.03	.01	.08
INIUIP	.67	.21	-.09	.17	.04	.36	.10	-.10	.48	.28	-.11	.01	-.05	-.02	-.01	-.01	-.02	.02
EFIUIP	.70	-.39	.15	.24	.15	.21	-.31	.14	.00	-.07	.05	.24	.00	-.10	.10	-.02	.06	-.14
INNHB	.87	.09	-.03	.16	-.08	-.02	.09	-.30	-.14	.01	.13	.02	.08	-.05	-.03	.19	-.14	-.03
EFNHB	.69	-.58	-.08	.05	.00	.05	-.28	.13	.00	-.16	-.02	.03	-.03	-.06	.02	-.05	-.10	.19
INNO3	-.29	-.70	-.05	-.50	.13	.09	.20	.03	.02	.02	-.19	.15	.00	.12	.13	.11	-.04	-.03
EFNO3	-.02	-.55	.16	-.53	.43	.08	.31	-.12	.07	.03	.19	-.09	.07	-.15	-.08	-.08	-.01	.00
INIKN	.87	.05	-.07	-.01	.10	-.09	.12	-.27	.00	-.05	.18	.10	.00	.22	.12	-.09	.08	.04
EFIKN	.05	-.40	.52	.21	.33	.08	-.25	.12	-.06	.14	.00	-.27	.02	.16	.00	.07	.03	-.01

TABLE 3.12

CORRELATIONS FOR SET 2 POND 2 DATA

	INCOD	EFCOD	INSCOD	EFSCOD	INSBOD	EFBOD	INBOD	EFBOD	INTOTS	EFTOTS	INTKN	EFTKN	PRECIP	SOLAR	INFLOW	EFFLOW	LAGIMP
INCOD	1.00																
EFCOD	.09	1.00															
INSCOD	.57	-.09	1.00														
EFSCOD	.02	.00	.37	1.00													
INSBOD	.26	-.04	.32	.07	1.00												
EFBOD	.03	-.03	.12	.13	.07	1.00											
INBOD	.81	-.01	.53	.03	.31	.13	1.00										
EFBOD	-.08	.32	-.07	.04	.03	.62	.03	1.00									
INTOTS	.76	.07	.24	-.09	.14	.09	.69	.10	1.00								
EFTOTS	-.14	.61	-.27	-.23	-.09	-.26	-.16	.31	.04	1.00							
INTKN	.64	-.04	.73	.27	.32	.17	.61	-.08	.37	-.34	1.00						
EFTKN	.12	.18	.36	.22	.02	.28	.14	.31	.02	.04	.30	1.00					
PRECIP	-.01	-.09	.01	.04	-.05	.08	-.02	-.02	.04	-.05	.04	-.01	1.00				
SOLAR	-.26	.25	-.42	-.17	-.18	-.23	-.22	.00	-.05	.42	-.50	-.42	-.31	1.00			
INFLOW	-.27	.28	-.33	-.09	-.23	-.03	-.29	.20	-.12	.42	-.41	-.02	-.08	.34	1.00		
EFFLOW	-.33	.25	-.33	-.12	-.25	-.03	-.35	.21	-.17	.37	-.40	.02	-.05	.27	.89	1.00	
LAGIMP	-.39	.21	-.52	-.10	-.29	-.25	-.36	-.03	-.18	.39	-.57	-.50	-.06	.79	.33	.24	1.00

TABLE 3.13

FACTOR LOADING MATRIX FOR SET 2 POND 2 DATA

	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17
INCOD	.73	.32	.48	-.01	-.15	.08	.08	-.06	-.05	-.08	.03	-.02	-.14	.00	.12	.21	.08
IFCOD	-.23	.69	.12	.26	.16	-.39	.00	-.04	-.36	-.23	.13	.03	-.09	-.04	-.02	-.07	-.03
INSCOD	.77	.12	-.03	.36	.07	.08	.12	.07	-.07	.37	-.14	-.05	-.23	-.02	.03	-.09	-.03
EFSCOD	.26	-.02	-.34	.43	.36	.04	.62	.11	.21	-.23	-.05	.02	.01	.03	-.05	.03	.02
INSBOD	.42	.05	.10	.01	.29	.01	-.36	.75	.09	-.04	.13	-.04	.01	.00	.03	-.01	.01
EFBOD	.25	.32	-.54	-.57	.23	.18	.14	.00	-.20	.07	.10	-.09	-.05	.19	-.08	.01	.04
INBOD	.72	.32	.42	-.12	-.04	.12	.07	-.01	.06	.08	.07	.36	.06	.05	-.09	-.05	.01
EFBOD	-.07	.67	-.41	-.42	.27	-.06	.00	.02	.10	.02	-.22	.09	.02	-.19	.10	.03	-.02
INTOTS	.47	.43	.53	-.30	-.19	.08	.16	-.08	.24	-.16	.01	-.22	.02	-.01	.00	-.14	-.05
EFTOTS	-.50	.58	.23	.17	.00	-.39	-.15	.05	.16	.13	-.23	-.05	.05	.18	-.08	.05	.04
INTKN	.84	.12	.04	.20	.00	.03	.07	.02	-.28	.08	-.08	-.10	.35	-.04	.00	.03	-.01
EFKN	.35	.43	-.53	.28	-.04	-.12	-.19	-.29	.26	.14	.33	-.03	.06	-.01	.05	.01	.01
PRECIP	.10	-.12	-.19	-.23	-.53	-.55	.41	.32	.00	.12	.09	.00	-.01	-.06	-.04	.03	-.01
SOLAR	-.66	.11	.49	-.03	.36	.14	.16	-.02	.04	.21	.16	-.08	-.01	-.15	-.18	.08	-.02
INFLOW	-.62	.46	-.10	.18	-.33	.40	.10	.19	-.03	.01	.03	.03	.02	.08	.06	.06	-.19
EFFLOW	-.61	.42	-.20	.17	-.39	.38	.04	.17	-.04	-.01	-.01	-.01	.01	-.08	-.04	-.07	.19
LAGIMP	-.74	-.03	.37	-.07	.26	-.06	.34	.03	-.03	.16	.11	.03	.09	.07	.25	-.06	.05

TABLE 3.14

CORRELATIONS FOR SET 3 POND 2 DATA

	INSCOD	EFSCOD	INTOTS	EFTOTS	INTOTIP	EFTOTIP	INN3	EFN3	INN3	EFN3	INIKN	EFTKN	PRECIP	SOLAR	INFLOW	EFFLOW	LAGIMP
INSCOD	1.00																
EFSCOD	.37	1.00															
INTOTS	.24	-.09	1.00														
EFTOTS	-.27	-.23	.04	1.00													
INTOTIP	.45	.10	.24	-.17	1.00												
EFTOTIP	.48	.21	.08	-.29	.47	1.00											
INN3	.74	.24	.27	-.30	.59	.53	1.00										
EFN3	.50	.25	.09	-.60	.30	.82	.49	1.00									
INN3	-.16	-.08	-.22	-.22	-.34	-.07	-.41	.12	1.00								
EFN3	.06	.00	-.03	-.06	-.13	.07	-.12	.16	.69	1.00							
INIKN	.73	.27	.37	-.34	.57	.52	.84	.51	-.26	.04	1.00						
EFTKN	.36	.22	.02	.04	.22	.67	.30	.55	.01	.24	.30	1.00					
PRECIP	.01	.04	.04	-.05	.05	-.04	.04	-.02	.03	.10	.04	-.01	1.00				
SOLAR	-.42	-.17	-.05	.42	-.38	-.68	-.52	-.64	-.04	-.23	-.50	-.42	-.31	1.00			
INFLOW	-.33	-.09	-.12	.42	-.29	-.25	-.47	-.31	.17	.18	-.41	-.02	-.08	.34	1.00		
EFFLOW	-.33	-.12	-.17	.37	-.36	-.22	-.49	-.27	.31	.36	-.40	.02	-.05	.27	.89	1.00	
LAGIMP	-.52	-.10	-.18	.39	-.51	-.82	-.60	-.72	-.03	-.25	-.57	-.50	-.06	.79	.33	.24	1.00

TABLE 3.15

FACTOR LOADING MATRIX FOR SET 3 POND 2 DATA

	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17
INSCOD	.75	-.06	.18	-.03	-.33	-.25	.05	-.16	-.22	.16	-.32	.13	.00	-.07	.02	.00	-.02
EFSCOD	.32	.04	.00	-.50	-.71	.02	.12	.14	.26	-.19	-.01	-.04	-.01	.03	.00	.00	.01
INTOTS	.26	-.27	.28	.55	.03	-.40	.22	.49	.14	-.06	-.03	.03	.02	.03	-.02	.01	-.02
EFTOTS	-.50	-.12	.65	.14	.05	.18	.34	-.27	.00	-.13	-.13	-.16	.02	-.03	-.08	.04	.07
INTOTP	.63	-.26	.23	.16	.05	.05	-.35	-.24	.49	.16	-.02	.02	.01	-.02	-.03	.03	-.01
EFTOTP	.80	.30	.19	-.19	.27	.17	.02	.11	.01	-.04	-.08	-.19	-.05	-.06	.04	-.04	-.16
INN3	.84	-.28	.15	.05	-.12	-.08	-.09	-.15	-.22	-.03	.11	-.04	-.07	.22	-.15	.01	-.04
EFN3	.79	.36	-.11	-.20	.16	-.04	-.02	.27	-.05	.16	.00	-.08	-.07	-.10	-.15	.07	.11
INN3	-.19	.76	-.41	.12	-.02	-.27	.07	-.10	.10	.11	-.15	-.15	.16	.15	-.01	.01	.00
EFN3	.04	.78	-.06	.32	-.16	-.27	.14	-.27	.07	-.10	.16	.06	-.16	-.11	-.06	-.03	-.03
INTKN	.82	-.16	.15	.14	-.20	-.24	-.08	-.09	-.13	.02	.21	-.19	.14	-.08	.13	-.02	.05
EFTKN	.51	.42	.42	-.21	.11	.20	.40	-.01	.06	.23	.16	.14	.08	.08	.05	-.01	.01
PRECIP	.09	.07	-.21	.56	-.42	.63	.04	.12	-.05	.18	-.02	-.06	-.07	.01	.02	-.01	.00
SOLAR	-.76	-.31	.12	-.17	-.04	-.32	.09	.00	.07	.29	.01	-.13	-.24	.05	.08	-.03	.01
INFLOW	-.56	.41	.55	-.02	-.14	.00	-.34	.17	-.04	.06	-.02	.01	.06	.00	-.09	-.17	.02
EFFLOW	-.54	.57	.47	.03	-.13	-.02	-.29	.09	-.10	-.02	.02	.01	-.03	.04	.09	.18	-.03
LAGTIP	-.83	-.31	-.05	-.12	-.22	-.06	.10	.02	-.03	.20	.12	-.03	.14	-.11	-.14	.06	-.11

TABLE 4.1: Global Recommended Loadings
(After Gloyna, 1971)

ENVIRONMENTAL CONDITIONS	SURFACE LOADING Kg BOD ₅ /ha.d	DETENTION PERIOD Days
FRIGID ZONE, with seasonal ice cover, uniformly low water temperature and variable cloud cover.	10	200
COLD SEASONAL CLIMATE, with seasonal ice cover and temperate summer temperatures for short seasons.	10-15	200-100
TEMPERATE TO SEMI-TROPICAL, occasional ice cover, no prolonged cloud cover.	50-150	100-33
TROPICAL, uniformly distributed sunshine and temperature, and no seasonal cloud cover.	150-350	33-17

TABLE 4.2: Recommended Loading for Regions in the USA
(After Canter and Englande, 1970)

REGION	SURFACE LOADING Kg BOD ₅ /ha/day	DETENTION PERIOD Days
Northern	21.2	117
Central	37.0	82
Southern	49.4	31

TABLE 4.3: Concentrations of Toxicant affecting a 50% Chlorophyll Reduction
(After Gloyna 1977)

ORGANIC CHEMICAL	TOXIC CONCENTRATION (mg/l)	ORGANIC CHEMICAL	TOXIC CONCENTRATION (mg/l)
Methanoic Acid	220	3-Butanol	24,200
Ethanoic Acid	350	1-Hexanol	1,275
Propanoic Acid	250	2-Hexanol	3,100
Butanoic Acid	340	Heptanol	525
2-Methyl Propanoic Acid	345	Octanol	250
Pentanoic Acid	280	Propenoic Acid	120
3-Methyl Butanoic Acid	400	Butenoic Acid	280
Hexanoic Acid	320	2,3 Dihydroxy Butanedioic Acid	480
Heptanoic Acid	180	Hydroxyethanoic Acid	2,700
Octanoic Acid	220	Sulfanilic Acid	970
Ethanedioic Acid	290	Methoxyethanoic Acid	580
Propanedioic Acid	460	2-Oxo Propanoic Acid	880
Butanedioic Acid	2,200	Ethanoic Andrydride	360
Pentanedioic Acid	1,200	Propanal	3,450
Hexanedioic Acid	900	1-Butanal	2,500
Heptanedioic Acid	700	2-Methyl Propanal	3,450
Methanol	31,000	1-Heptanal	240
Ethanol	27,200	1,2-Ethanediol	180,000
1-Propanol	11,200	1,2-Propanediol	92,000
2-Propanol	17,400	Phenol	1,060
1-Butanol	8,500	Cresol	800
2-Butanol	8,900	Malthane	160
		Ortho (pesticide)	320
		DDT in Xylene	120

TABLE 4.4: BOD Removal Efficiencies and Pond Areas
(Based on Gloyna's formulation)

POND SYSTEM	ACTUAL SURFACE AREA m ²	COMPUTED SURFACE AREA m ²	BOD REMOVAL Efficiencies (%)
Corrine	14,900	51,474	83-96
Eudora	31,600	143,703	93-97
Peterborough	34,000	103,550	59-98
Kilmichael	21,000	19,883	98-99

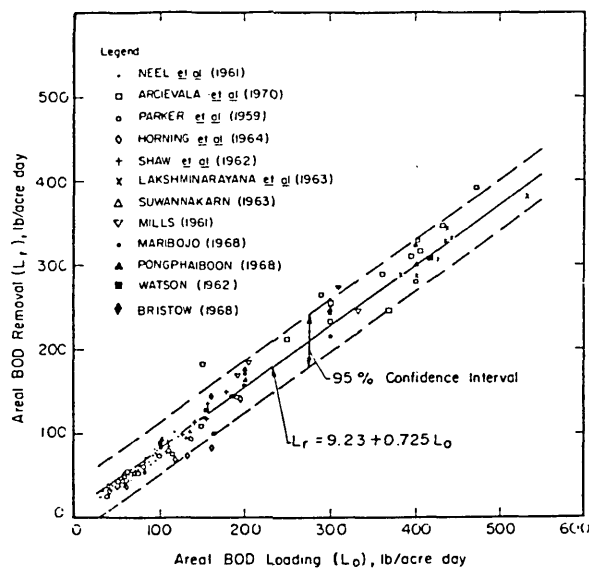


Fig.4.1... Areal BOD removal as a function of BOD loading
(After McGarry & Pescod, 1970)

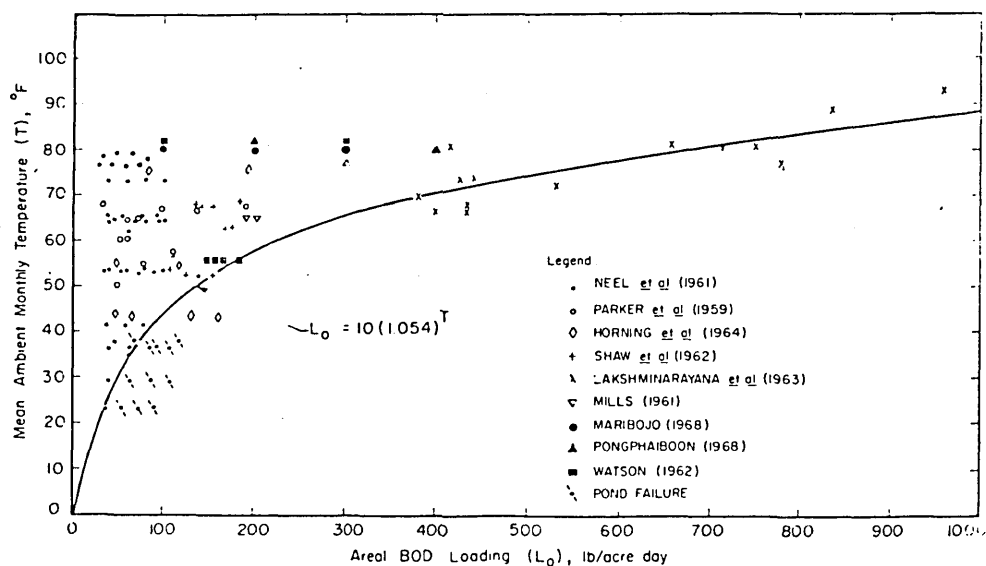


Fig 4.2 ... Frangible facultative pond loading as a function of temperature
(After McGarry & Pescod, 1979)

TABLE 4.5: Regression Equations for Pond Systems

EQUATION	UNITS	REGRESSION COEFFICIENT	SOURCE
$L_R = 0.725L_O + 10.37$	Kg/ha/day	0.995	McGarry and Pescod (1970)
$L_R = 0.730L_O + 2$	Kg/ha/day		Mara and Silva (1979)
$L_R = 0.730L_O + 25$	Kg/ha/day	0.965	Arthur (1981)
$L_R = 0.810L_O + 7.67$	Kg/ha/day	0.996	Hideo Kawai (1982)
$L_R = 0.940L_O - 1.04$	lb/ac/day	0.965	Middlebrooks et al. (1982)
$C_R = 0.610C_O + 13$	mg/l/day	0.890	Bradley (1983)

TABLE 4.6: Units of the Parameters in Larsen's Equation

PARAMETER	UNITS
Wind speed	Miles/Hr
Solar Radiation	Btu/day sq. ft
Relative humidity	%
Temperature	°F
Flow rate	gals/day
BOD	mg/l
Pond Area	sq. ft

TABLE 4.7: Dimensionless Products and Unit Conversion Factors

DIMENSIONLESS PRODUCTS	UNIT CONVERSION FACTOR
MOT	1.0783×10^7
RED	1.0
TTC	0.0879
TEMPR	1.0
DRY	1.0

TABLE 4.8: Actual and Computed Areas
(Based on Larsen's Formulation)

POND SYSTEM	ACTUAL AREA (m ²)	DESIGN AREA (m ²)	AREAS (m ²) FOR RED NUMBERS (%)		
			0	50	100
Corrine	14,900	35,343	34,456	35,146	36,230
Eudora	31,600	32,114	30,712	31,384	32,439
Peterborough	34,000	70,841	67,694	69,216	72,608
Kilmichael	21,000	14,731	14,195	14,512	15,010

TABLE 4.9: First Order Decay Coefficients for Stabilization Ponds

DECAY RATE (day ⁻¹)	SOURCE
0.110-0.125	Greenfield (1926)
0.172-0.241	Gotaas (1948)
0.165-0.259	Moore et al. (1950)
0.114-0.148	Thomas (1950)
0.170-0.600	Marais & Shaw (1961)
0.100-0.270	Ballinger et al. (1962)
0.029-0.075	Thirumurthi (1974)

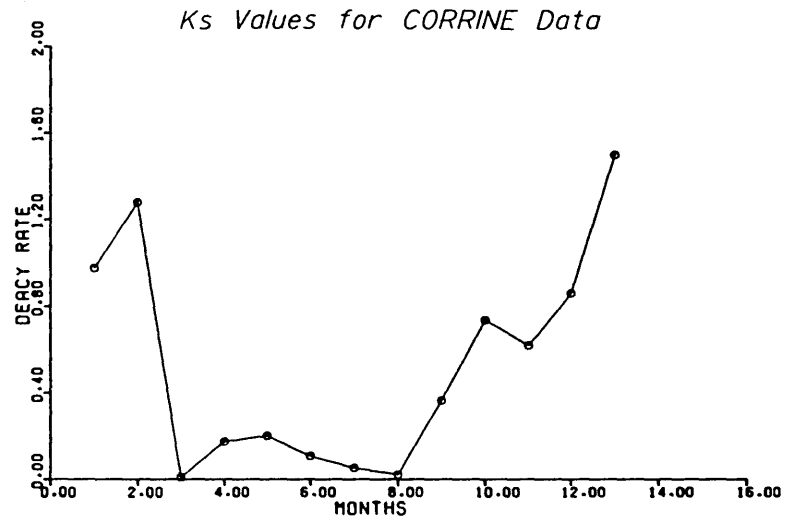


Fig 4.3 Computed Monthly 1st Order Decay Rate Constant for Corrine Data

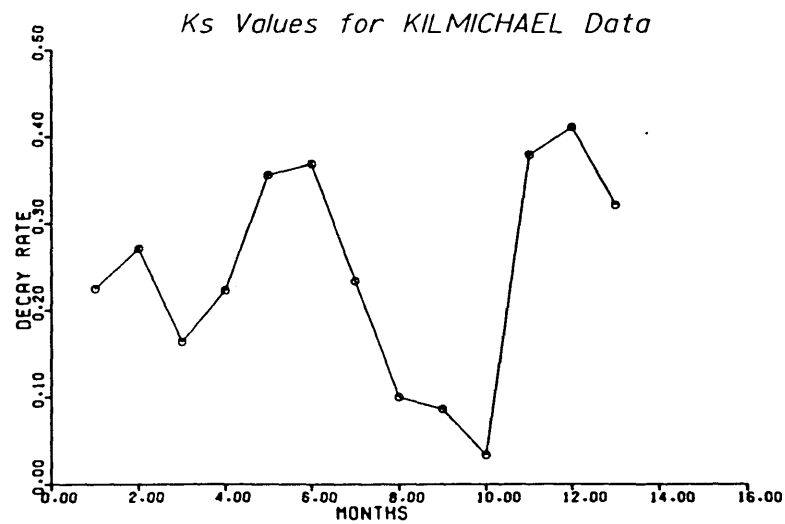


Fig 4.4 Computed Monthly 1st Order Decay Rate Constant for Kilmichael Data

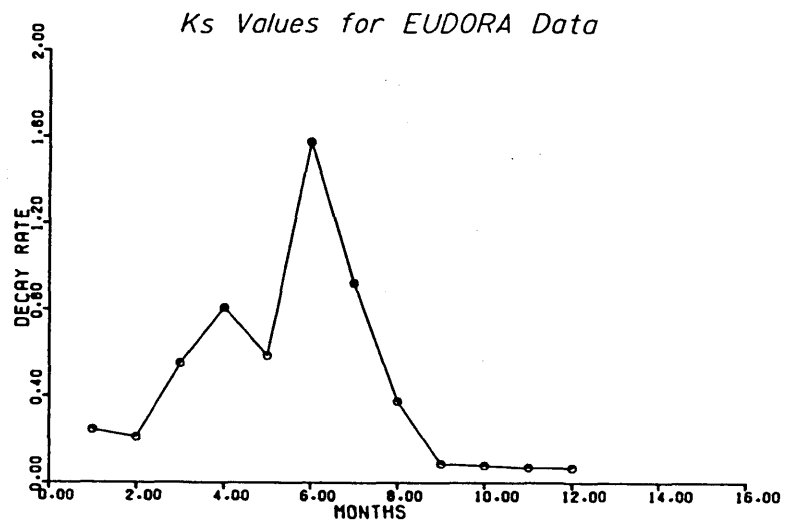


Fig 4.5 Computed Monthly 1st Order Decay Rate Constant for Eudora Data

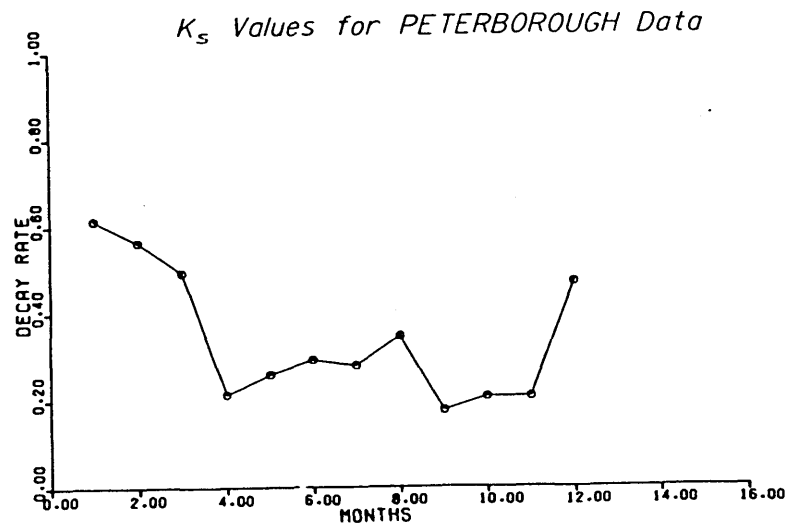


Fig 4.6 Computed Monthly 1st Order Decay Rate Constant for Peterborough Data

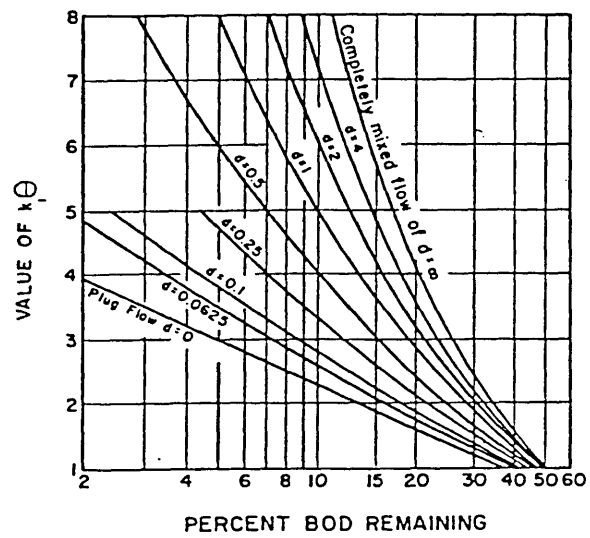
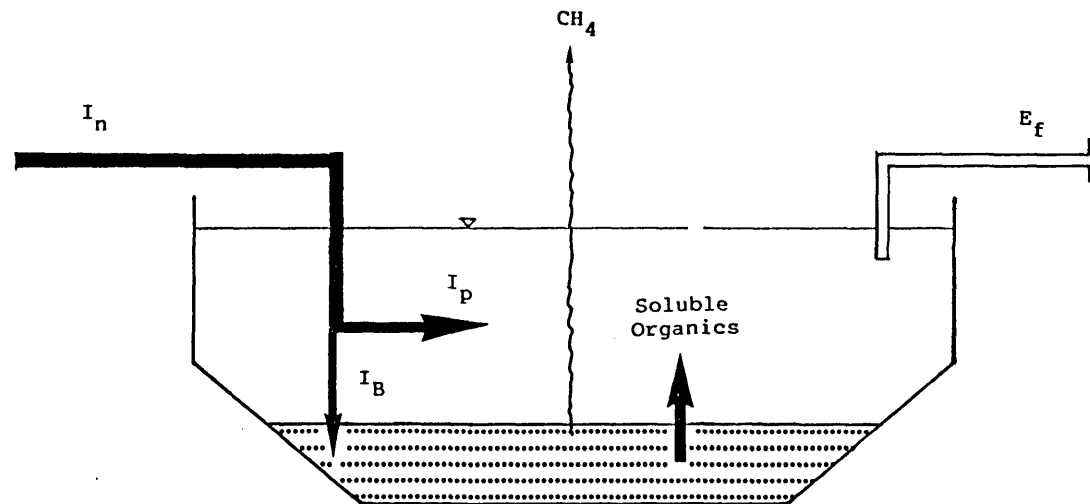


Fig 4.7. Non-Ideal Reactor Design Formula Chart
(After Thirumurthi, 1974)

TABLE 4.10: Toxicity Factor for Various Organic Substances
(after Thirumurthi (1974))

ORGANIC SUBSTANCE	CONCENTRATION IN POND INFLUENT (mg/l)	VALUE OF C_{TOX} FACTOR
Mathanoic Acid	180	2.0
Mathanoic Acid	360	16.0
Ethanoic Acid	270	1.6
Propanoic Acid	180	2.65
Hexanoic Acid	200	1.3
Hexanoic Acid	300	5.0
1-Butanol	4,000	2.0
Octanol	150	2.0
Octanol	200	4.0
Malthane	70	2.5
Malthane	140	8.0



I_n :- Influent Stream

E_f :- Effluent Stream

I_p :- Portion of Influent remaining in pond liquid

I_B :- Portion of Influent settling to the benthos

Fig 4.8 Marais' Conceptual Description of Pond Systems

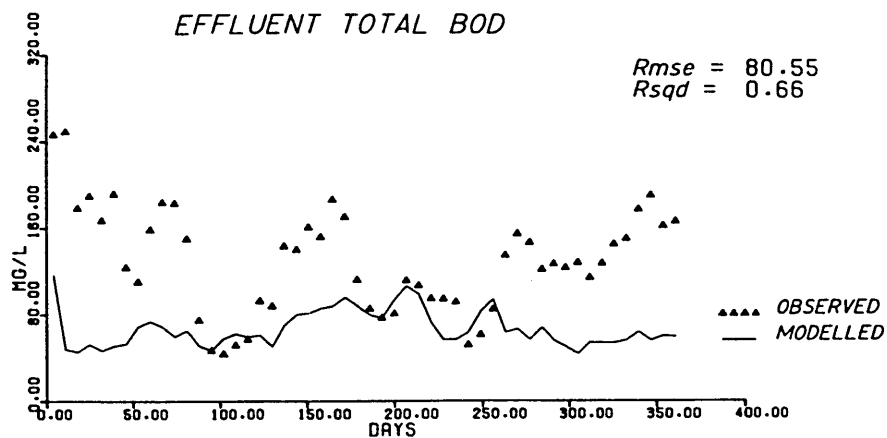


Fig 4.9 Simulation Results of Marais' Model for Pond 1 Total BOD

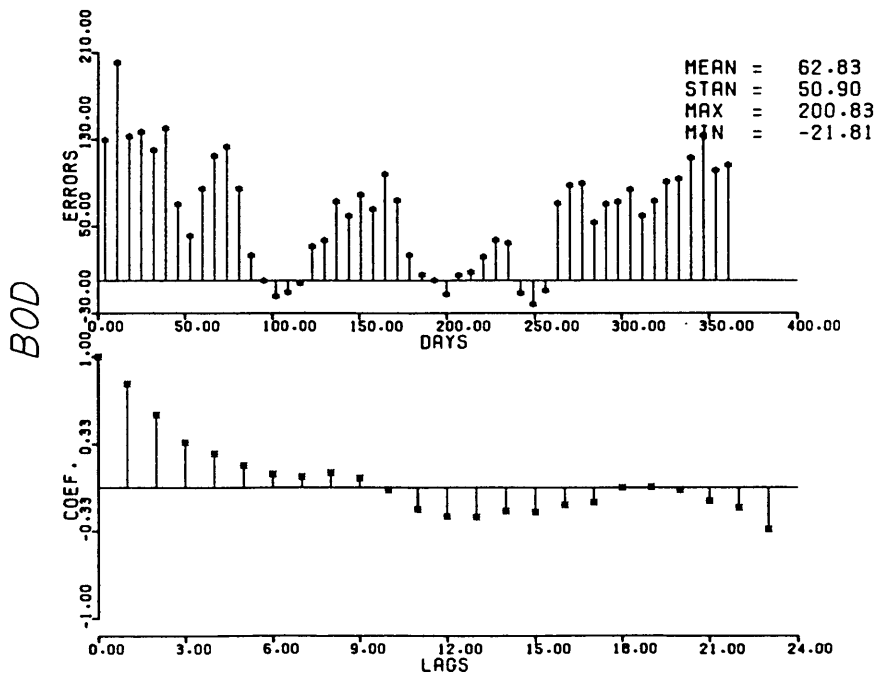


Fig 4.10 Errors Plot of Marais' Model Simulation for Pond 1 Total BOD

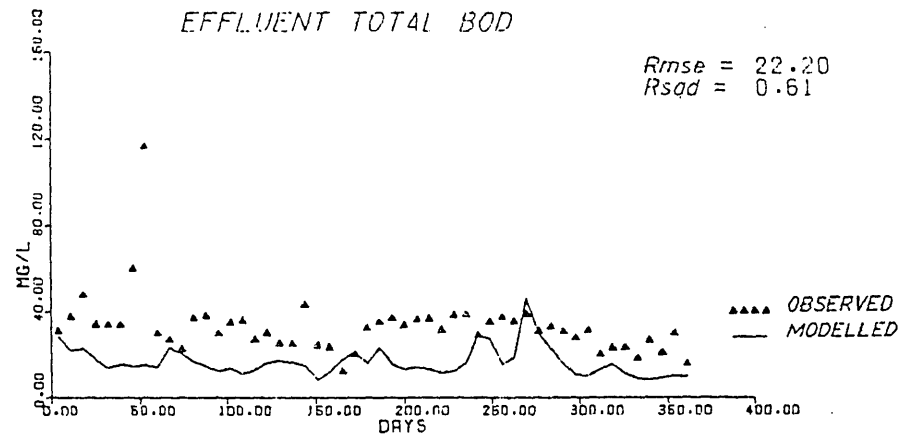


Fig 4.11 Simulation Results of Marais' Model for Pond 2 Total BOD

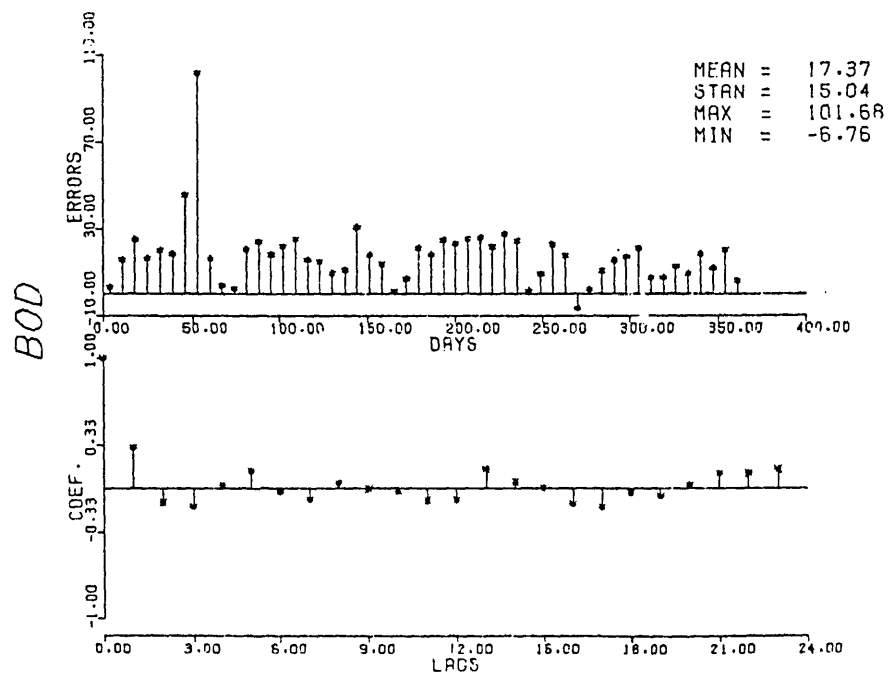


Fig 4.12 Errors Plot of Marais' Model Simulation for Pond 2 Total BOD

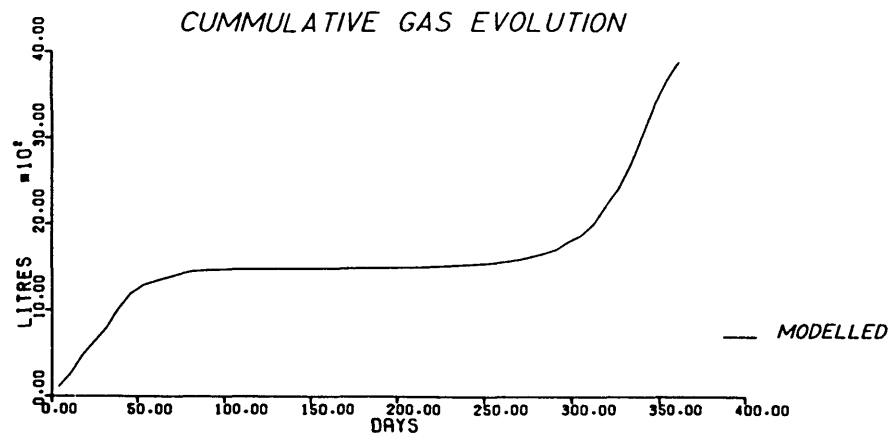


Fig 4.13 Computed Cumulative Gas Evolution
(Marais' Model for Pond 1 Data)

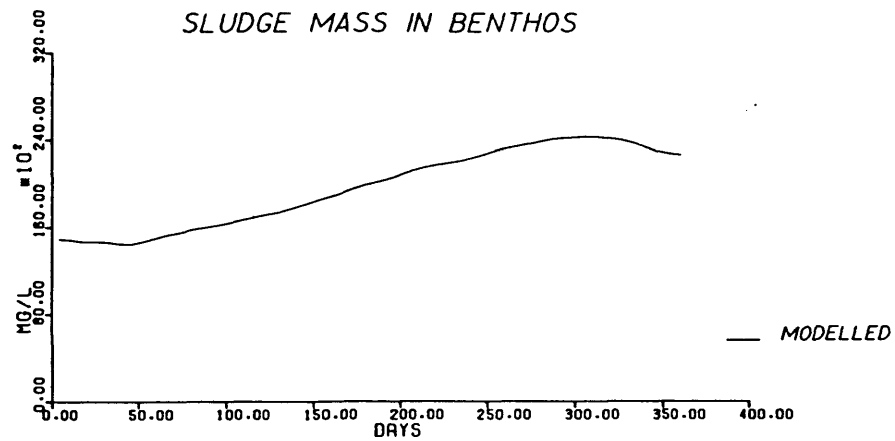


Fig 4.14 Computed Sludge Mass in Benthos
(Marais' Model for Pond 1 Data)

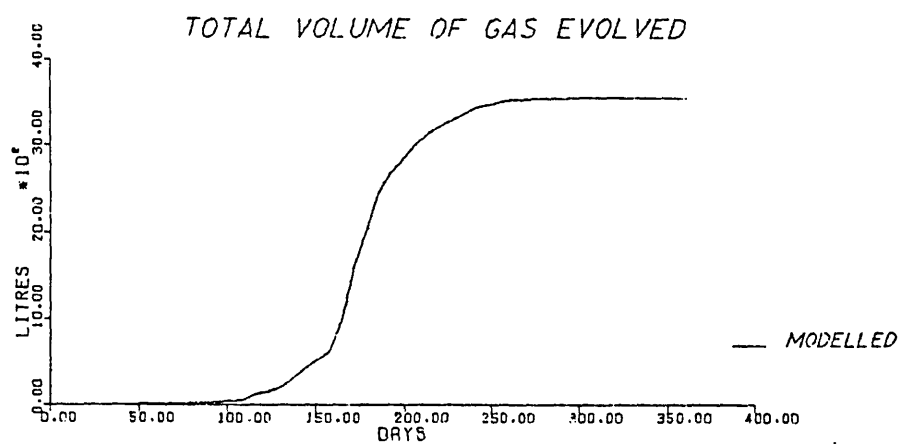


Fig 4.15 Computed Cumulative Gas Evolution
(Marais' Model for Pond 2 Data)

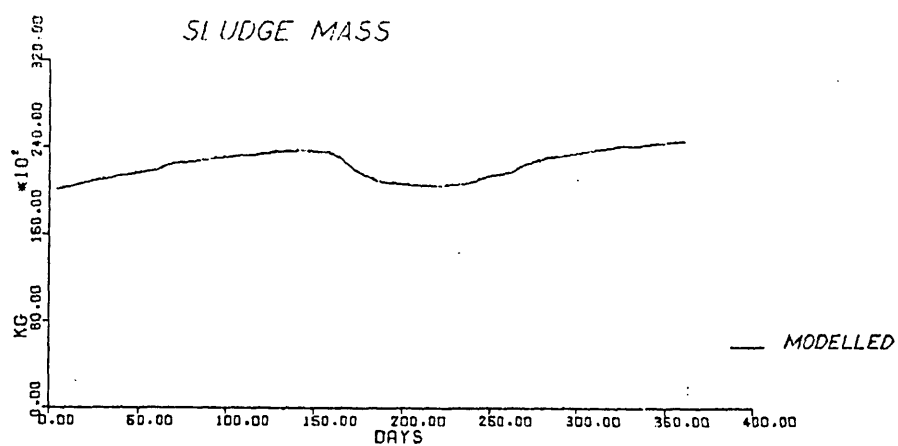
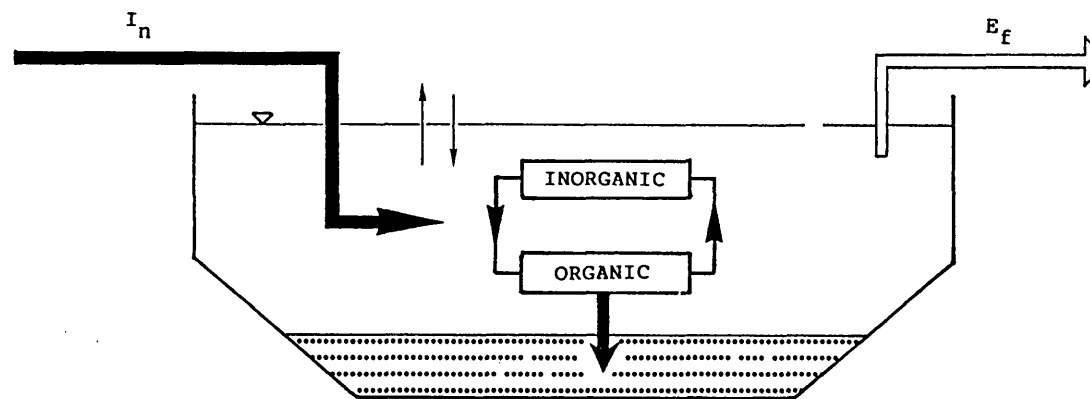


Fig 4.16 Computed Sludge Mass in Benthos
(Marais Model for Pond 2 Data)



I_n :- Influent Stream

E_f :- Effluent Stream

Fig 4.17 Ferrara's (1978) Conceptual Description of Facultative Pond Systems

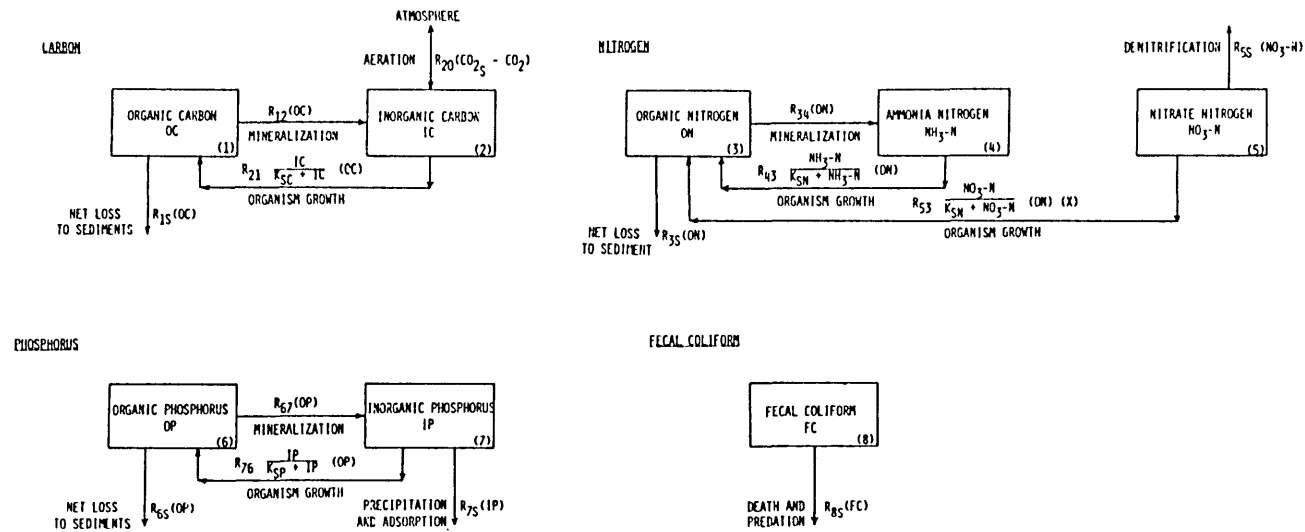


Fig 4.18 Ferrara's (1978) Description of Transformations in Facultative Ponds

TABLE 4.11
Suggested values for model coefficients (Ferrara, 1978)

ELEMENT	RATE	VALUE (day^{-1})
CARBON	K_{12}^*	0.050
	K_{1s}	0.020
	K_{21}^*	0.085
	K_{20}^*	8.64/depth
NITROGEN	K_{34}^*	0.060
	K_{35}	0.030
	K_{43}^*	0.100
	K_{53}^*	0.100
	K_{5s}	0.070
PHOSPHORUS	K_{67}^*	0.050
	K_{6s}	0.020
	K_{76}^*	0.035
	K_{7s}	0.002
FECAL COLIFORMS	K_{8s}^*	4.000

* Temperature Dependent Rates

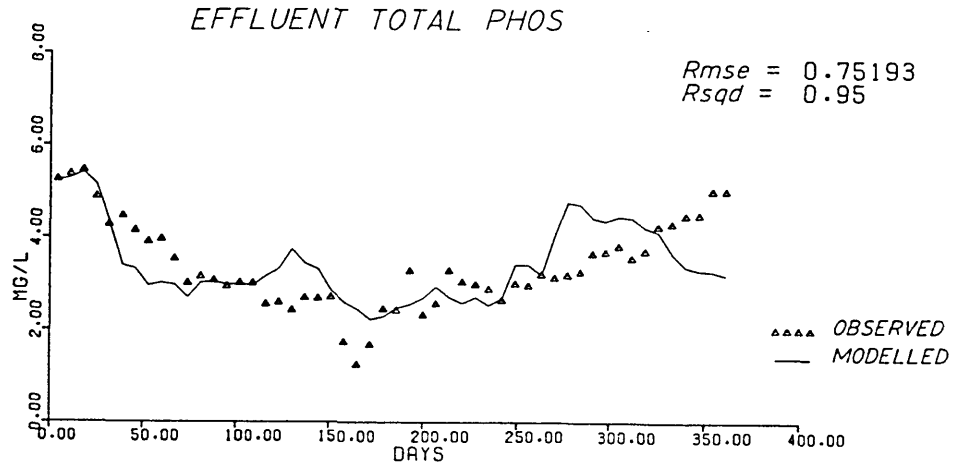


Fig 4.19 Simulation Results of Ferrara's (1978) Model for Pond 2 Total Phosphorus

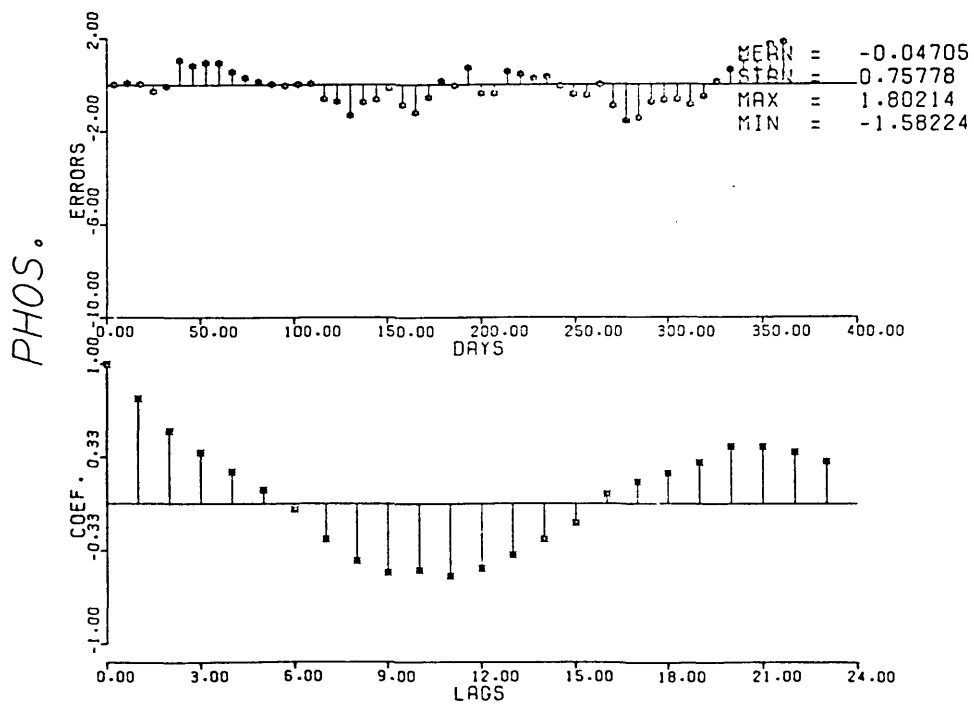


Fig 4.20 Errors Plot of Ferrara's (1978) Model Simulation for Pond 2 Total Phosphorus

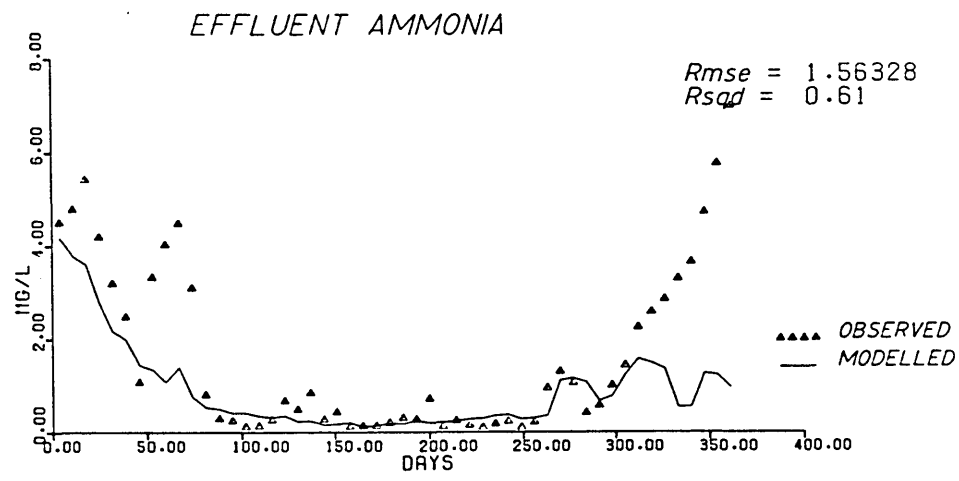


Fig 4.21 Simulation Results of Ferrara's (1978) Model for Pond 2 Ammonia

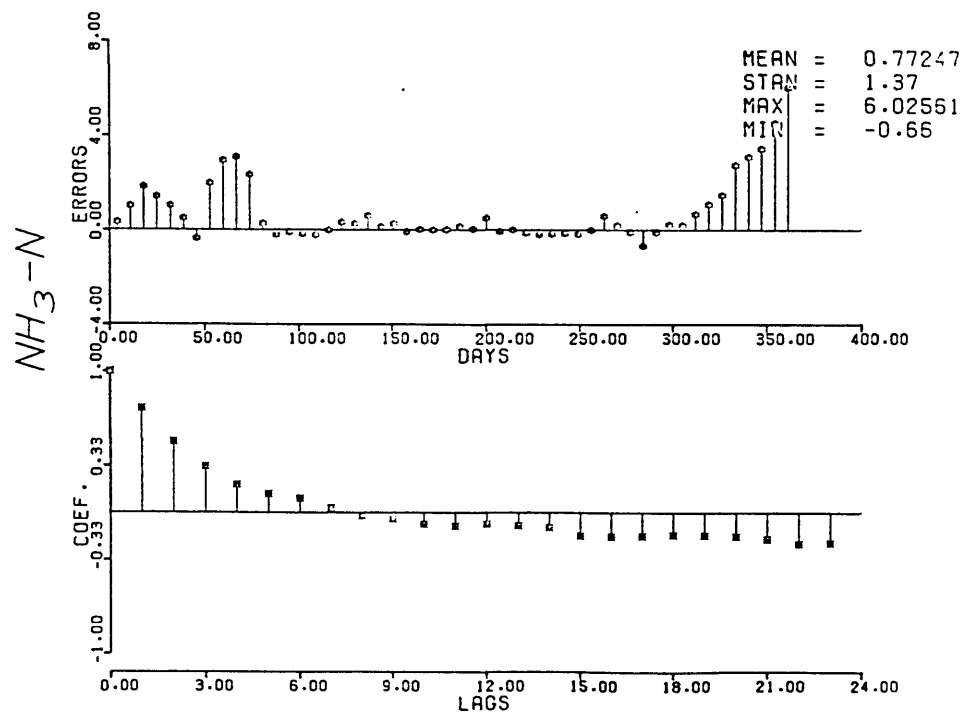


Fig 4.22 Errors Plot of Ferrara's (1978) Model Simulation for Pond 2 Ammonia

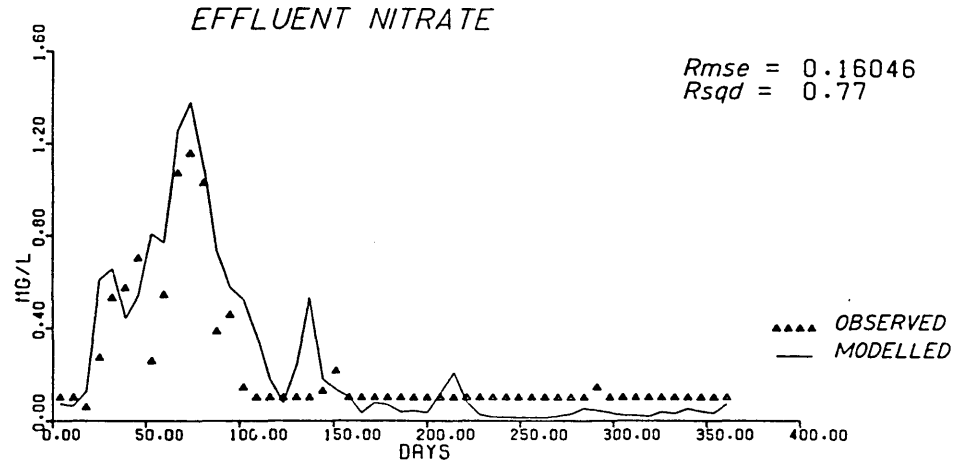


Fig 4.23 Simulation Results of Ferrara's (1978) Model for Pond 2 Nitrate

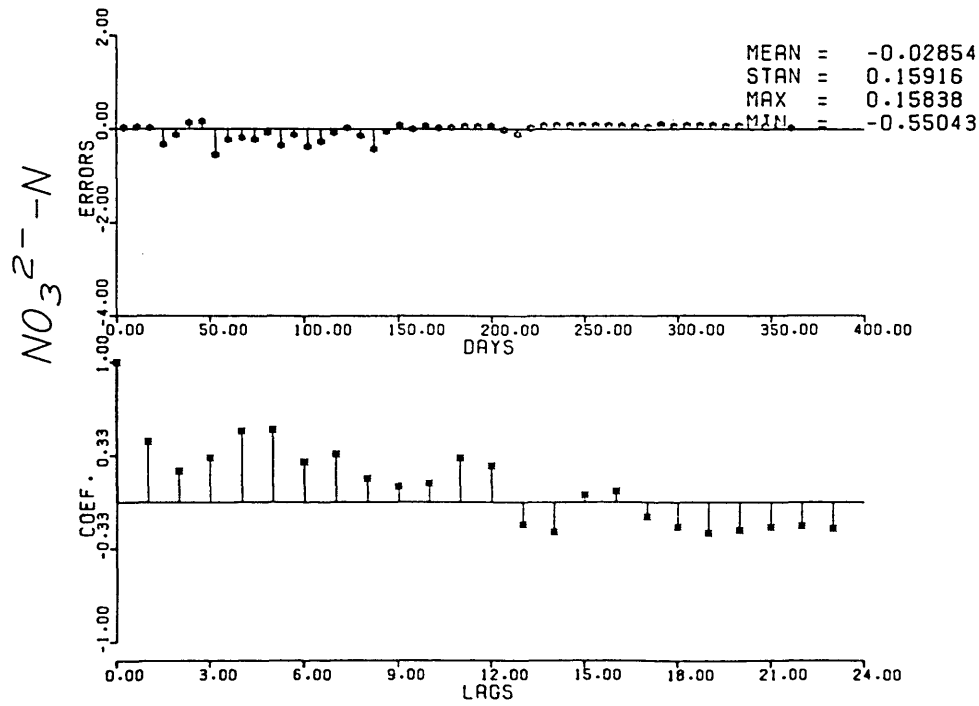


Fig 4.24 Errors Plot of Ferrara's (1978) Model Simulation for Pond 2 Nitrate

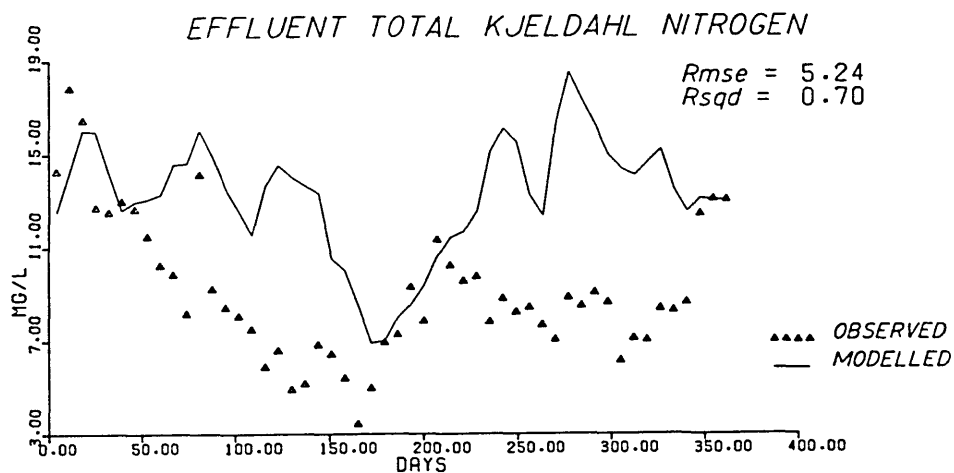


Fig 4.25 Simulation Results of Ferrara's (1978) Model for Pond 2 Total Kjeldahl Nitrogen

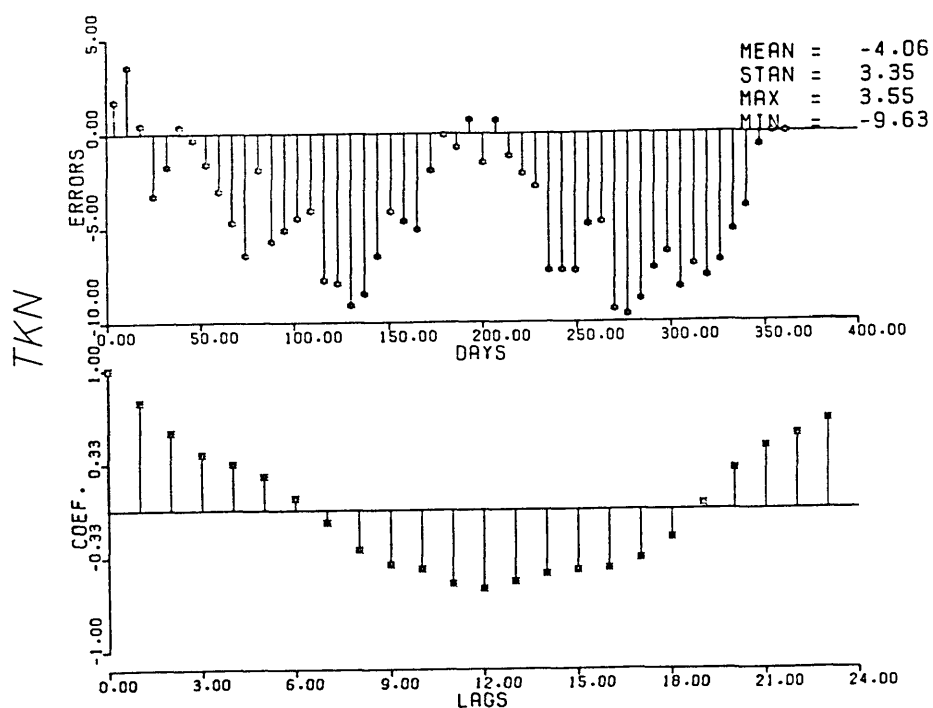


Fig 4.26 Errors Plot of Ferrara's (1978) Model Simulation for Pond 2 Total Kjeldahl Nitrogen

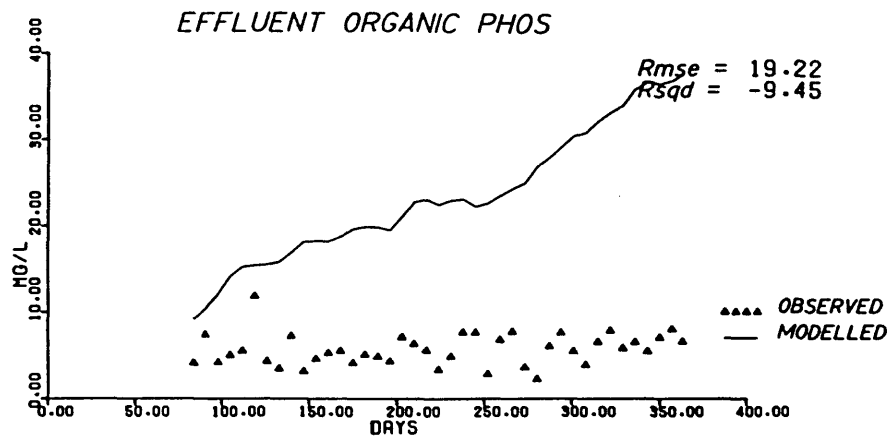


Fig 4.27 Simulation Results of Ferrara's (1978) Model for Pond 1 Organic Phosphorus

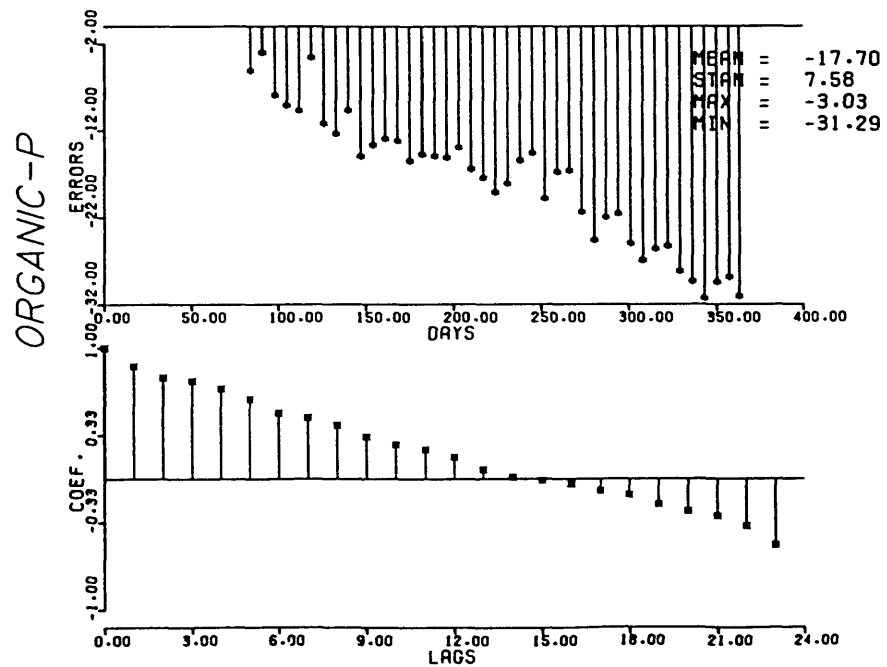


Fig 4.28 Errors Plot of Ferrara's (1978) Model Simulation for Pond 1 Organic Phosphorus

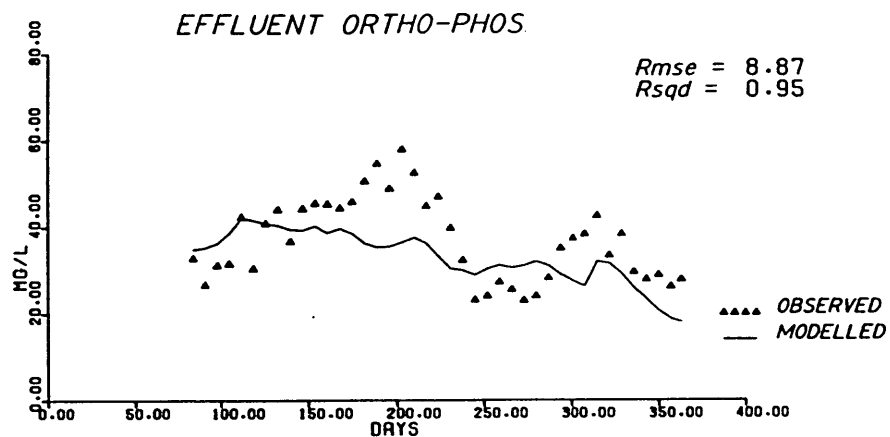


Fig 4.29 Simulation Results of Ferrara's (1978) Model for Pond 1 Ortho-phosphorus

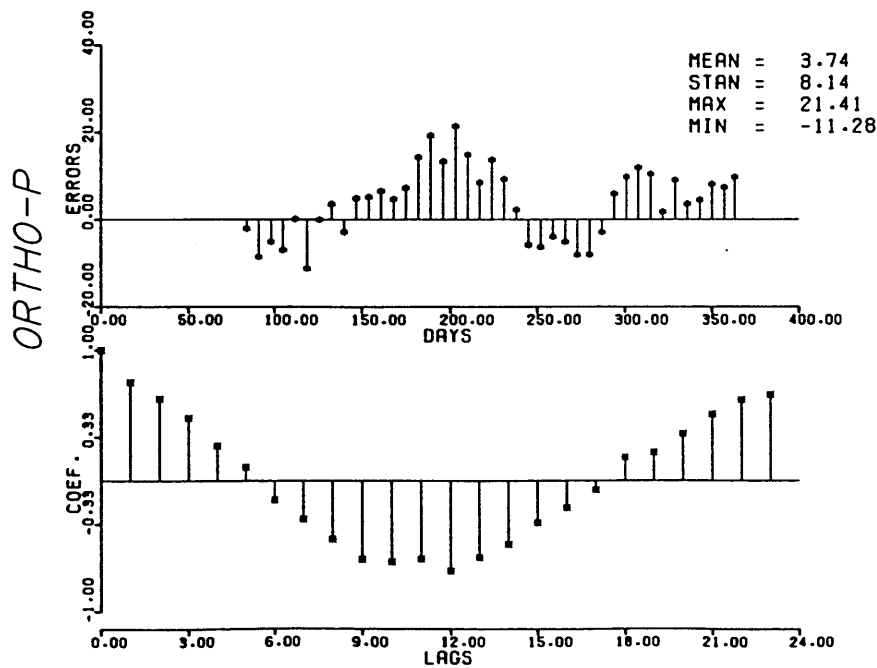
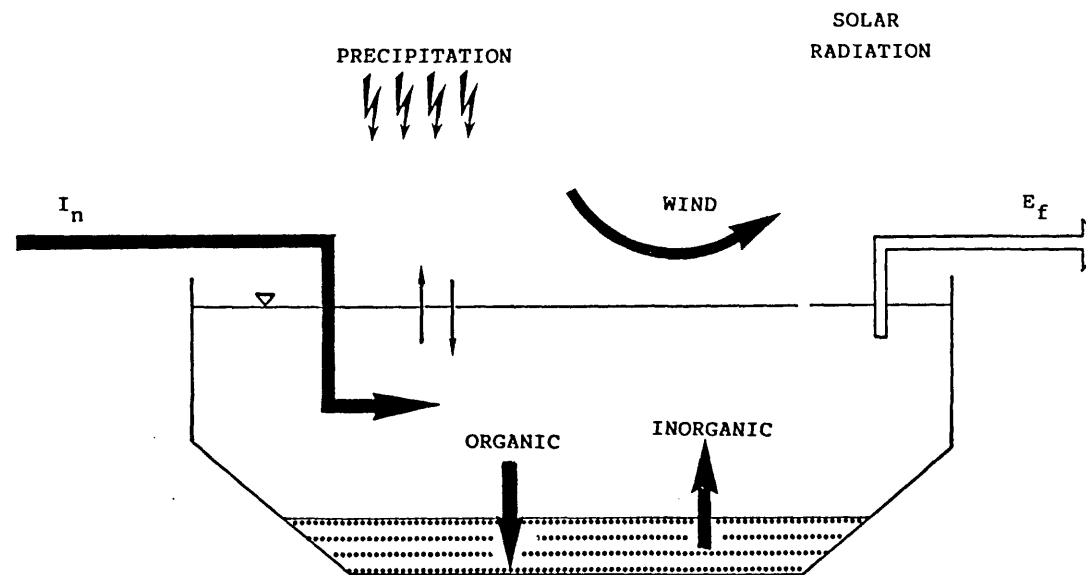


Fig 4.30 Errors Plot of Ferrara's (1978) Model Simulation for Pond 1 Ortho-phosphorus



I_n :- Influent Stream
 E_f :- Effluent Stream

Fig 4.31 Fritz's (1978) Conceptual Description of Facultative Pond Systems

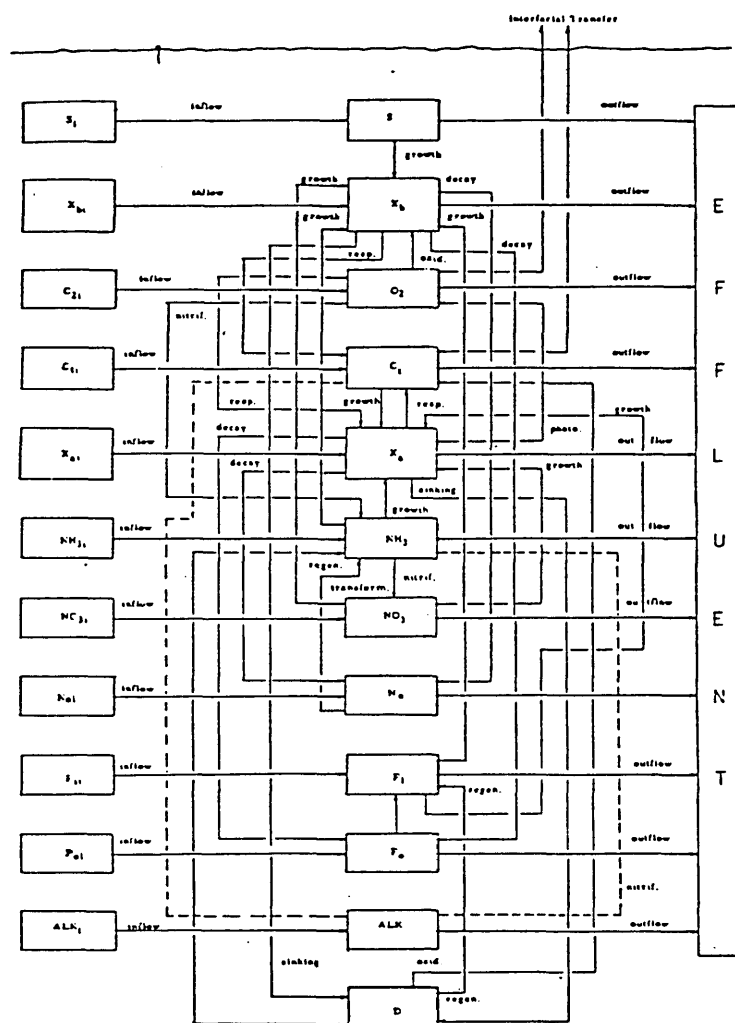


Fig 4.32 Fritz's (1978) Schematic of Facultative Pond Interactions

Table 4.12 Mass Balance Reaction Rates for Fritz's (1978) Model

Reaction	Volumetric Reaction Rate
Bacterial Growth:	
Degradable COD Utilization	$r_s = -k_x X_b \left[\frac{S}{K_s + S} \right] \left[\frac{O_2}{K_{O_2} + O_2} \right] \left[\frac{N_i^*}{K_{bn} + N_i} \right] \left[\frac{P_i}{K_{bp} + P_i} \right]$
Bacterial Growth	$r_{xb1} = -Y r_s - k_b X_b$
Sedimentation	$r_{xb2} = -Y r_s - k_b X_b$
Algal Growth:	
Algal Growth	$r_{xa1} = X_a \left\{ \mu_a f(L) f(T) \left[\frac{CO_2}{K_{CO_2} + CO_2} \right] \left[\frac{N_i}{K_{an} + N_i} \right] \left[\frac{P_i}{K_{ap} + P_i} \right] - k_a \right\}$
Sedimentation	$r_{xa2} = -s_a X_a$
Oxygen Generation and Utilization:	
Interfacial Transfer	$r_{ldo} = \frac{A}{V} K_{lO_2} (O_s - O_2)$
Photosynthesis	$r_{2do} = 1.244 r_{xa1}$
Bacterial Decay	$r_{3do} = -[(1-1.42Y)(-r_s) + 1.42 k_b X_b]$
Nitrification	$r_{4do} = -4.57 r_{3am}$
Organic Nitrogen:	
Bacterial & Algal Decay	$r_{lno} = 0.124 k_b X_b + 0.063 k_a X_a$
Organic N $\rightarrow$ NH ₃	$r_{2no} = -q_N N_o$
Ammonia:	
Organic N $\rightarrow$ NH ₃	$r_{lam} = -r_{2no}$
Algal & Bacterial Growth	$r_{2am} = 0.063(r_{xa1} + k_a X_a) - 0.124(-Y r_s)$
Nitrification	$r_{3am} = -\frac{\hat{\mu}_N}{Y_N} \left[\frac{NH_4}{K_N + NH_4} \right] \left[\frac{O_2}{K_{NO_2} + O_2} \right] [C_{pH}] [C_T]$
Benthic Regeneration	$r_{4am} = R_N/d$
Nitrate:	
Nitrification	$r_{1na} = r_{3am}$
Algal & Bacterial Growth	$r_{2na} = r_{2am}$
Organic Phosphorus:	
Bacterial & Algal Decay	$r_{lop} = 0.009 k_a X_a + 0.024 k_b X_b$
Organic P $\rightarrow$ Inorganic P	$r_{2op} = -q_p P_o$
Inorganic Phosphorus:	
Algal & Bacterial Growth	$r_{lip} = 0.009 (r_{xa1} + k_a X_a) - 0.024(-Y r_s)$
Organic P Inorganic P	$r_{2ip} = r_{2op}$
Benthic Regeneration	$r_{3ip} = R_p/d$
Total Inorganic Carbon Generation and Utilization:	
Interfacial Transfer	$r_{lct} = \frac{12}{44} \frac{A}{V} K_{lCO_2} (CO_{2s} - CO_2)$
Bacterial Decay	$r_{2ct} = \frac{12}{32} r_{3do}$
Photosynthesis	$r_{3ct} = -1.314 r_{xa1}$
Benthic Decay	$r_{4ct} = R_c C_m/d$
Alkalinity:	
Nitrification	$r_{la} = -7.14 r_{3am}$
Detritus:	
Sedimentation	$r_{ld} = (1/d) (s_a X_b + s_b X_b)$
Upwelling	$r_{2d} = U_r D$
$N_1^{\pm}$ = total nitrogen concentration (ammonia and nitrate)	

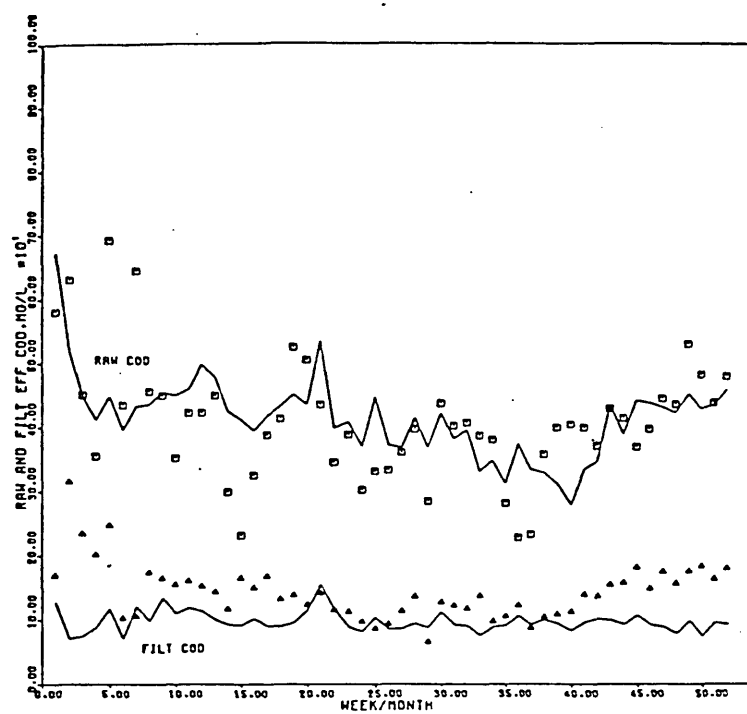


Fig 4.33 Fritz's Model Simulation for Pond 1 Filtered & Total COD

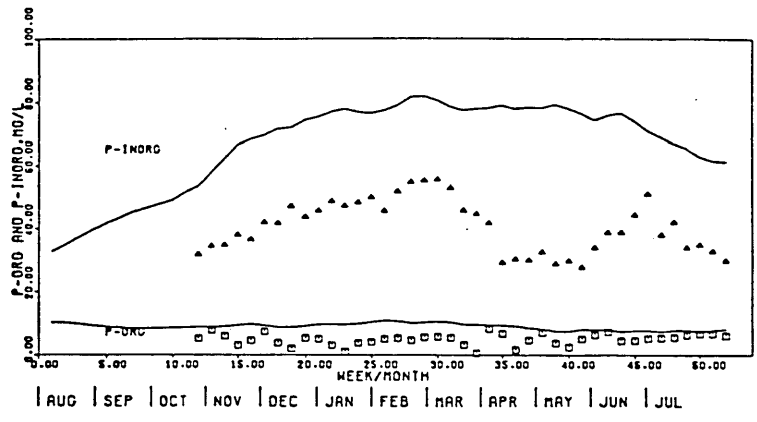
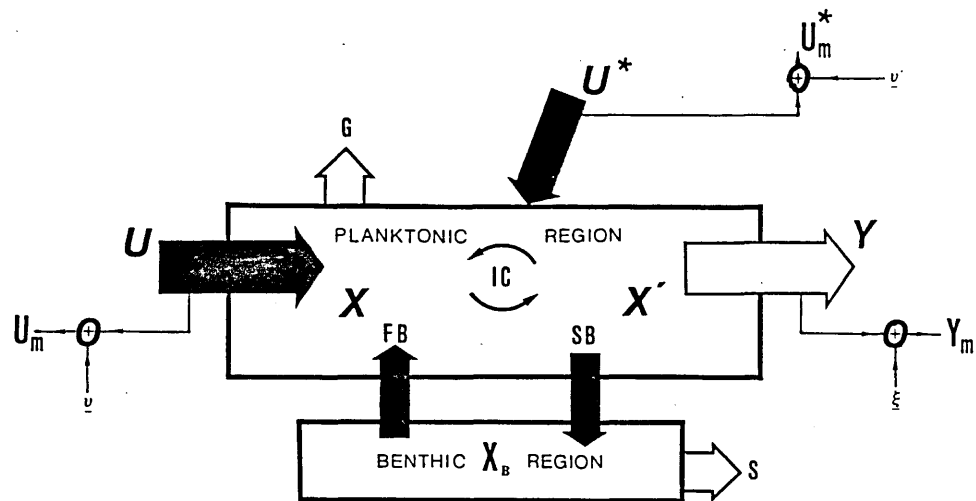
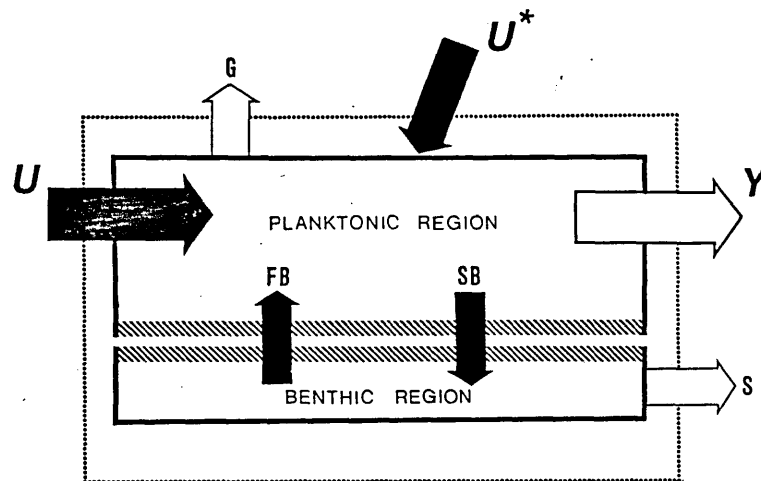


Fig 4.34 Fritz's Model Simulation for Pond 1 Inorganic & Organic Phosphorous



- U - Influent Stream
- U_m - Measured characteristics of Influent Stream
- U^* - Environmental factors influencing Pond behaviour (eg Solar Radiation, Wind)
- U_m^* - Measured Environmental factors
- G - Gases evolving from pond system (eg CH_4 , CO_2)
- FB - Internal loading to the planktonic region resulting from the feedback of organic and inorganic material from the benthos
- SB - Internal loading to the benthic subregion resulting from the transport of material, mainly particulate organics from the planktonic region
- IC - Internal cycling processes (eg activity of producers, decomposers and consumers)
- Y - Effluent Stream
- Y_m - Measured characteristics of Effluent Stream
- v - Errors in measuring the characteristics of Influent Stream
- v^* - Errors in measuring the characteristics of external Environmental factors
- ξ - Errors in measuring the characteristics of Effluent Stream
- X - State of the pond system in the Influent region
- X' - State of the pond system in the Effluent region
- X_B - State of the pond system in the benthos
- S - Loss of organics through Seepage

Fig 5.1 Input - State - Output diagram for facultative Pond Systems



- U - Influent Stream
- U* - Environmental factors influencing Pond Behaviour (eg Solar Radiation, Wind)
- FB - Internal loading to the planktonic region resulting from the feedback of organic and inorganic material from the benthos
- SB - Internal loading to the benthic subregion resulting from the transport of material, mainly particulate organics from the planktonic region
- G - Gases evolving from pond systems (eg CH_4 , CO_2)
- Y - Effluent Stream
- S - Loss of organics through Seepage

Fig 5.2 Boundaries for the Planktonic and Benthic coupled sub-systems

TABLE 5.1: Measured Output Descriptive Variables

		POND 1	POND 2
CARBONACEOUS ORGANICS	SOLUBLE	Filtered COD	Soluble COD Soluble BOD
	TOTAL	Total COD Total BOD	Total COD Total BOD
	SUSPENDED	Volatile Solids Total Solids	Total Solids
NUTRIENTS (N & P)	INORGANIC	Ammonia Nitrate Phosphorous	Ammonia Nitrate
	TOTAL		Phosphorous Nitrogen
	ORGANIC	Phosphorous	

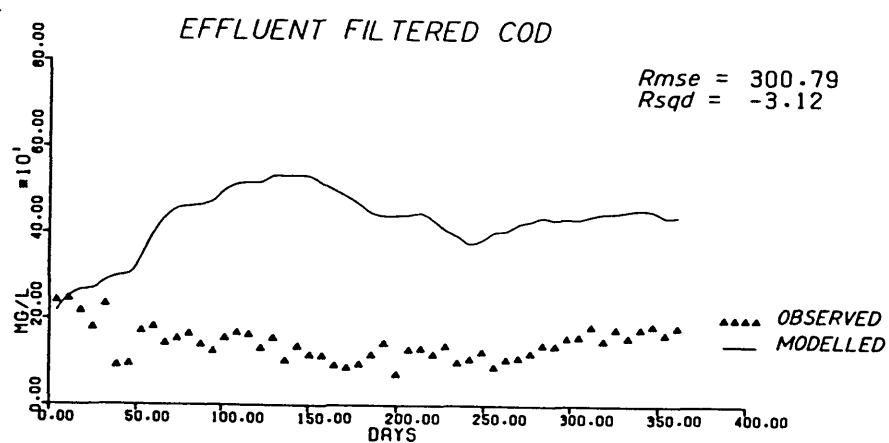


Fig 5.3 Simulation Plot for Pond 1 Effluent Filtered COD (Zero Rates Model)

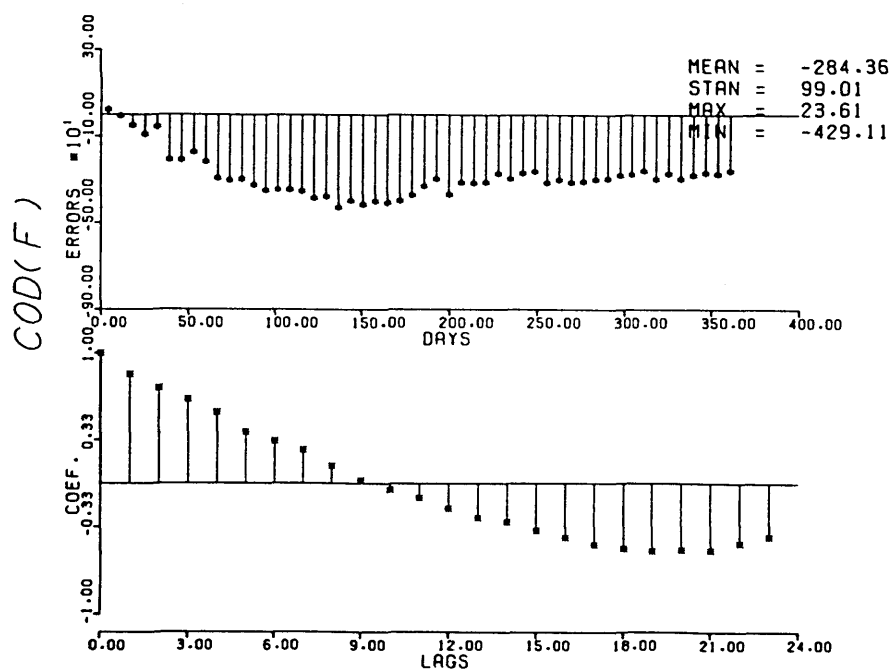


Fig 5.4 Errors Plot for Zero Rates Model Simulation of Pond 1 Effluent Filtered COD

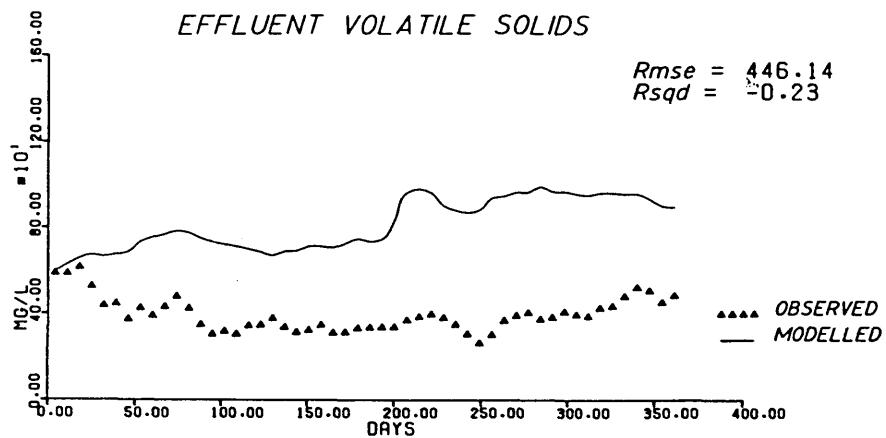


Fig 5.5 Simulation Plot for Pond 1 Effluent Volatile Suspended Solids (Zero Rates Model)

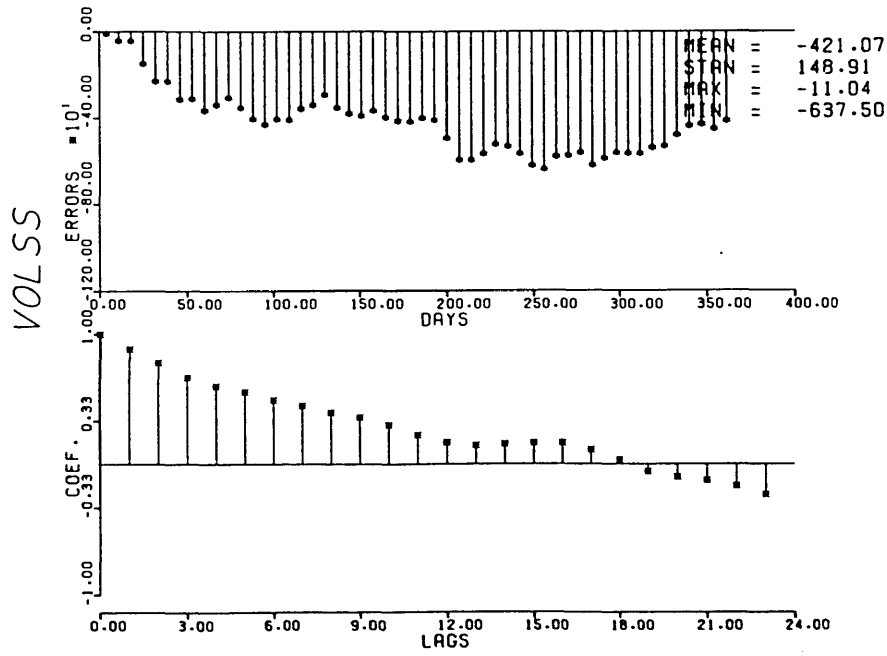


Fig 5.6 Errors Plot for Zero Rates Model Simulation of Pond 1 Effluent Volatile Suspended Solids

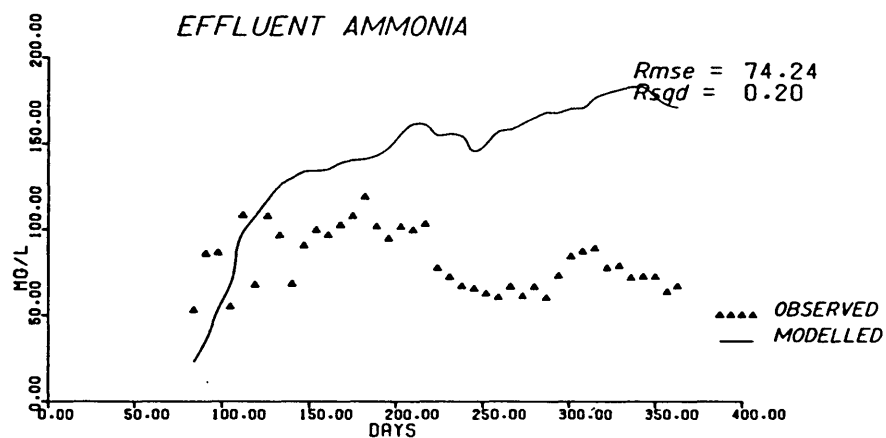


Fig 5.7 Simulation Plot for Pond 1 Effluent Ammonia (Zero Rates Model)

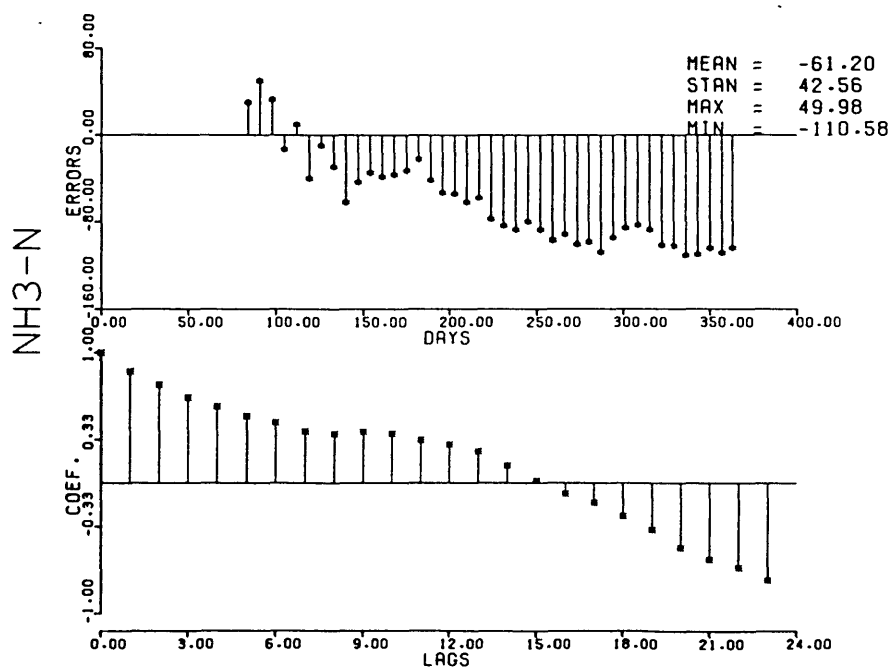


Fig 5.8 Errors Plot for Zero Rates Model Simulation of Pond 1 Effluent Ammonia

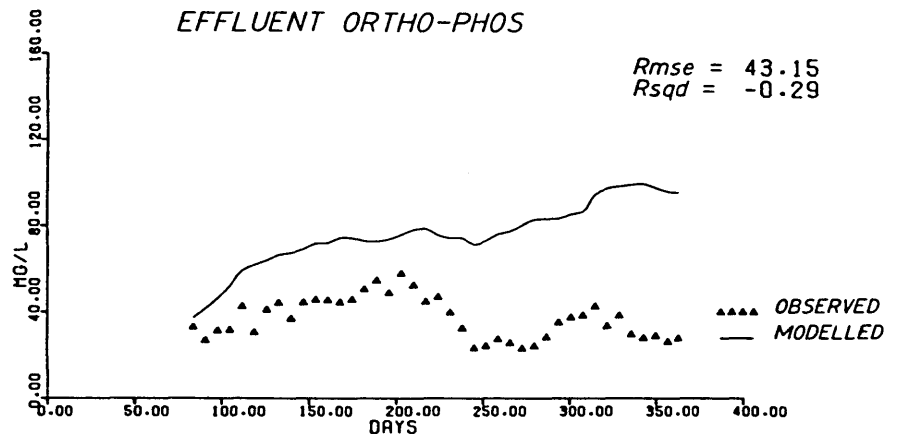


Fig 5.9 Simulation Plot for Pond 1 Effluent Ortho-Phosphorus (Zero Rates Model)

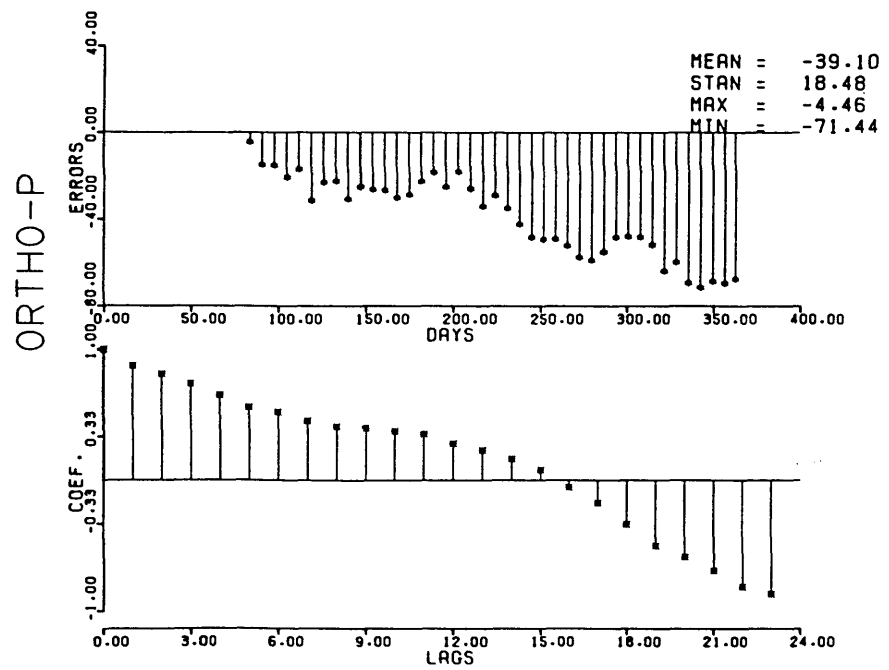


Fig 5.10 Errors Plot for Zero Rates Model Simulation of Pond 1 Effluent Ortho-Phosphorus

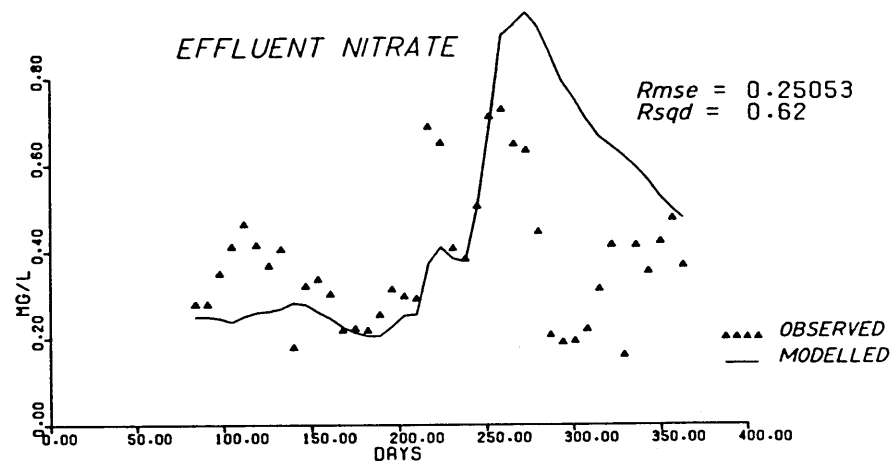


Fig 5.11 Simulation Plot for Pond 1 Effluent Nitrate (Zero Rates Model)

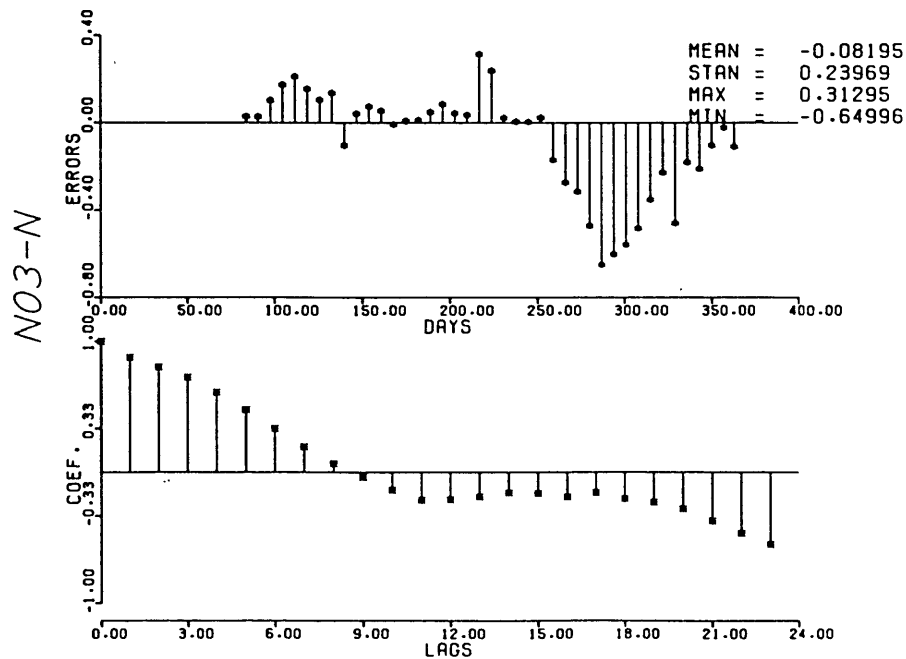


Fig 5.12 Errors Plot for Zero Rates Model Simulation of Pond 1 Effluent Nitrate

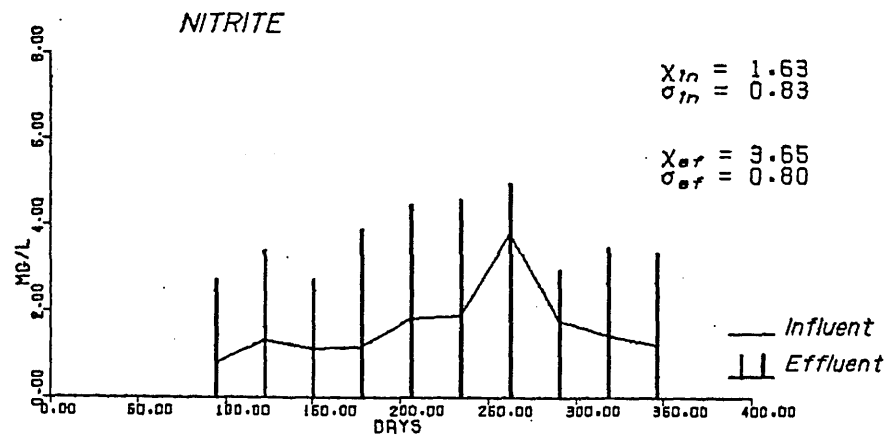


Fig 5.13 Monthly Averaged Influent & Effluent Nitrite for Pond 1 Data

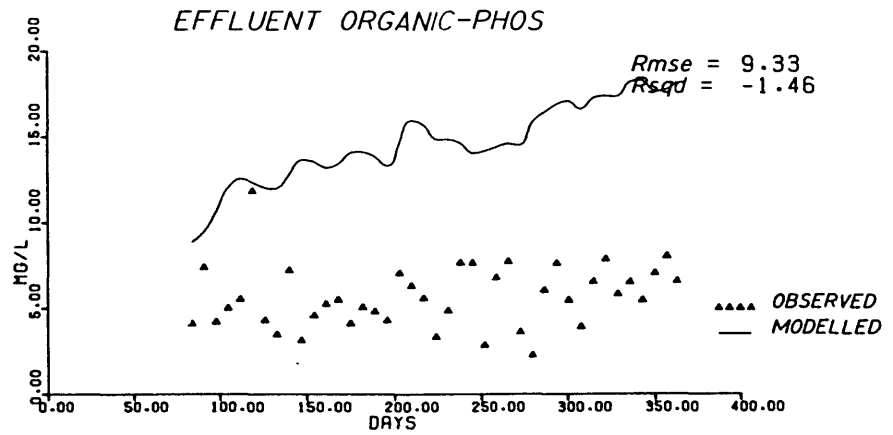


Fig 5.14 Simulation Plot for Pond 1 Effluent Organic-Phosphorus (Zero Rates Model)

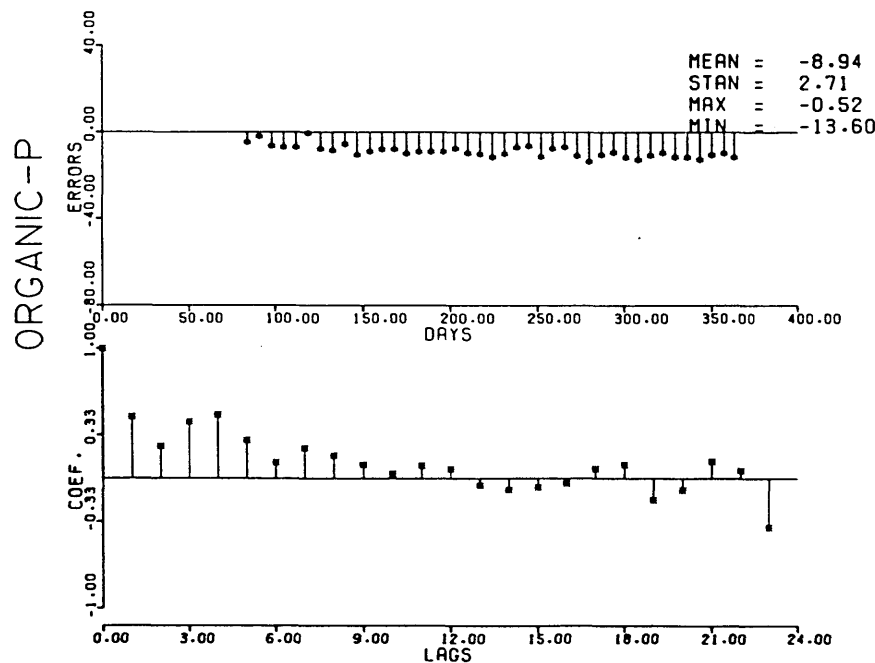


Fig 5.15 Errors Plot for Zero Rates Model Simulation of Pond 1 Effluent Organic-Phosphorus

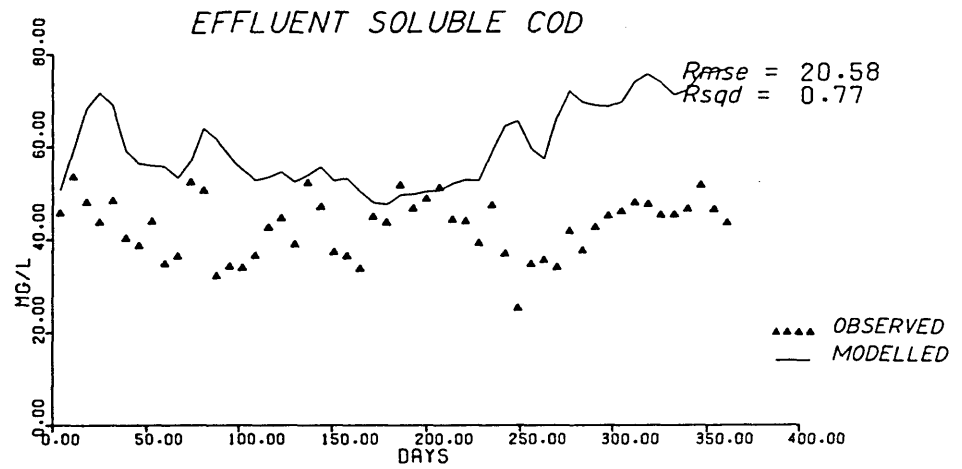


Fig 5.16 Simulation Plot for Pond 2 Effluent Soluble COD (Zero Rates Model)

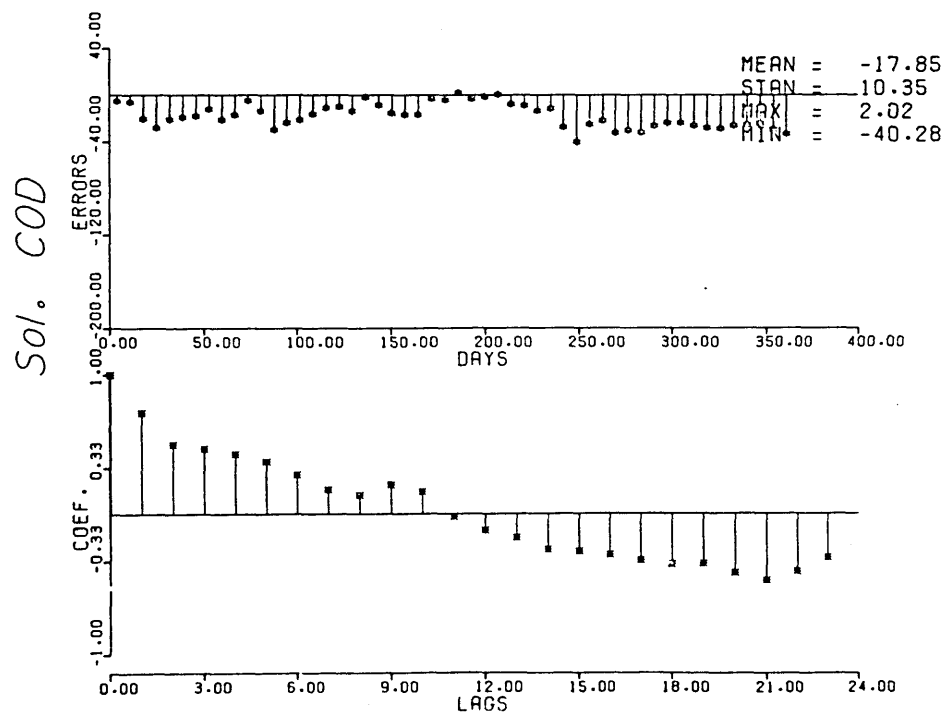


Fig 5.17 Errors Plot for Zero Rates Model Simulation of Pond 2 Effluent Soluble COD

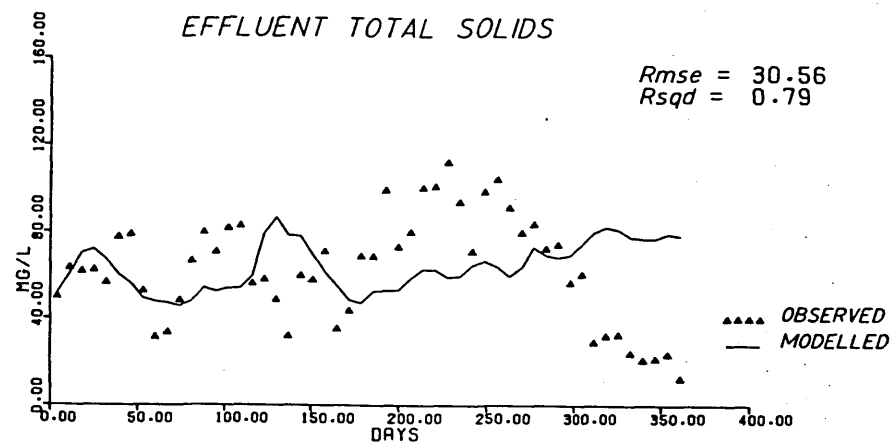


Fig 5.18 Simulation Plot for Pond 2 Effluent Total Solids (Zero Rates Model)

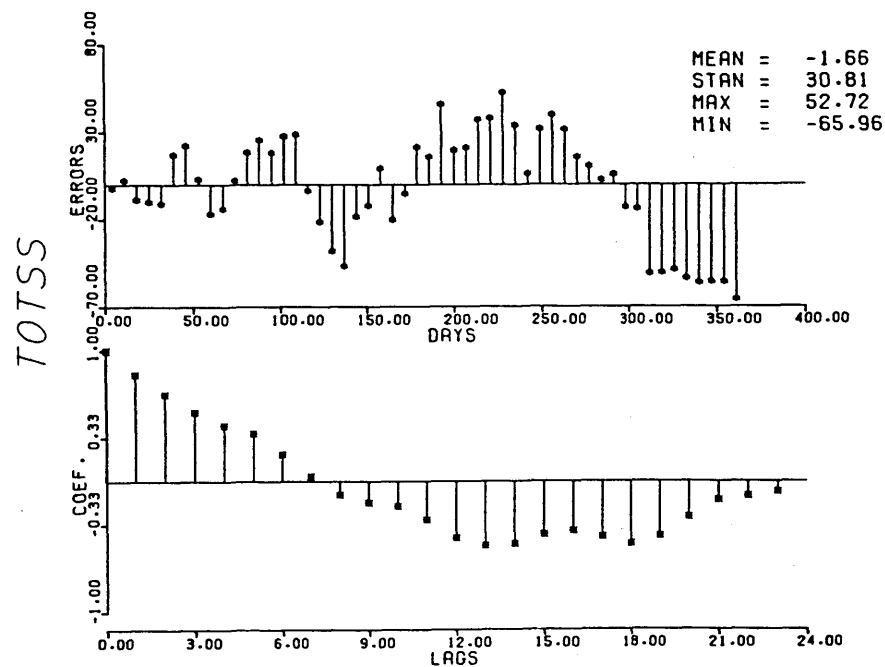


Fig 5.19 Errors Plot for Zero Rates Model Simulation of Pond 2 Effluent Total Solids

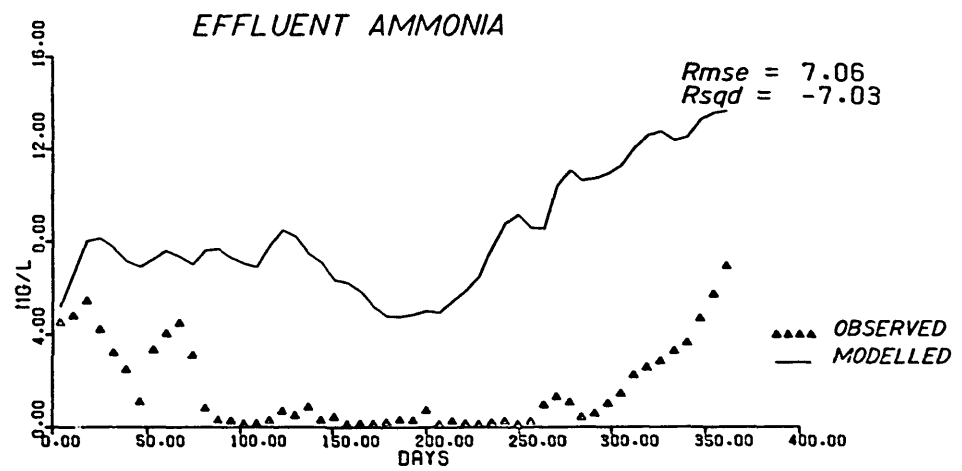


Fig 5.20 Simulation Plot for Pond 2 Effluent Ammonia (Zero Rates Model)

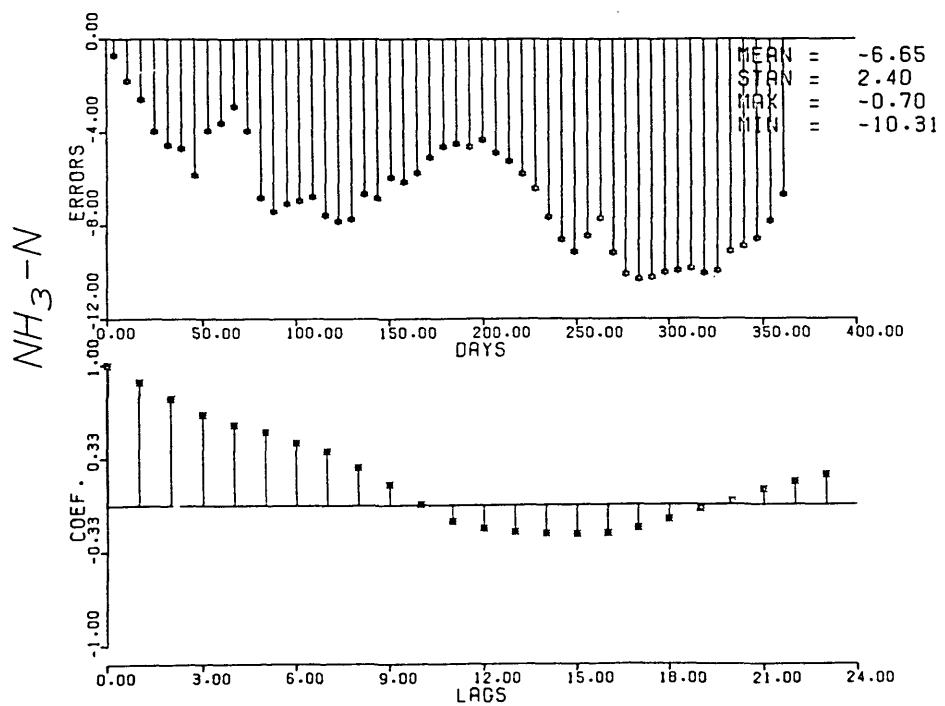


Fig 5.21 Errors Plot for Zero Rates Model Simulation of Pond 2 Effluent Ammonia

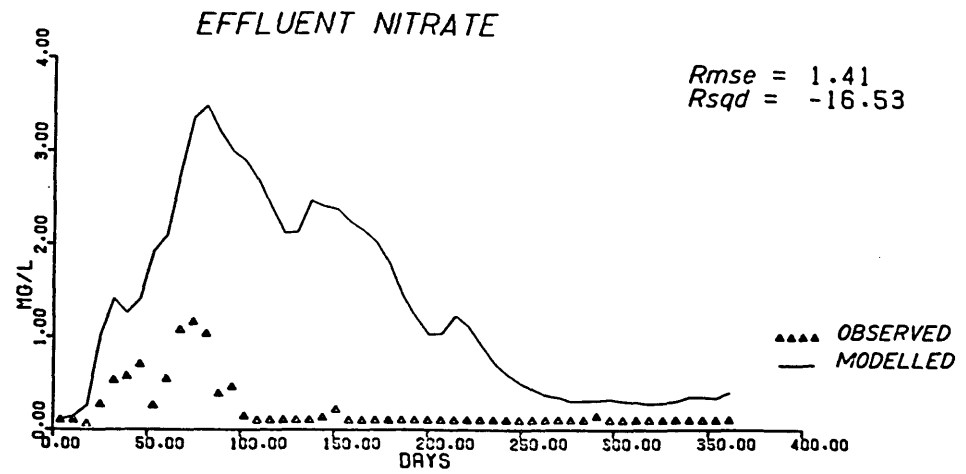


Fig 5.22 Simulation Plot for Pond 2 Effluent Nitrate
(Zero Rates Model)

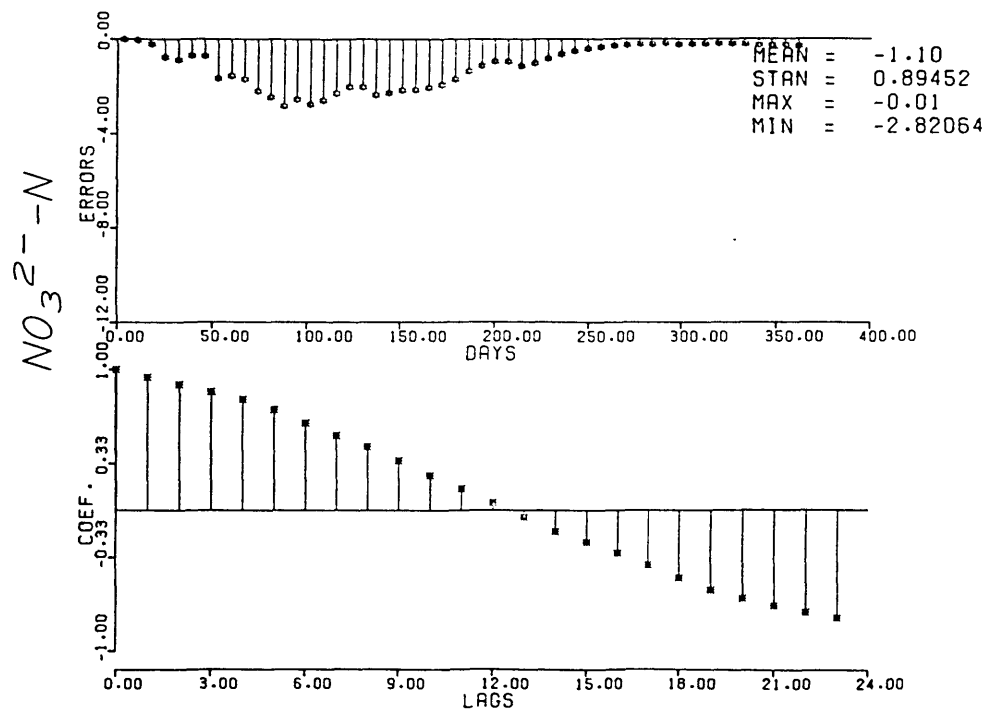


Fig 5.23 Errors Plot for Zero Rates Model Simulation of Pond 2
Effluent Nitrate

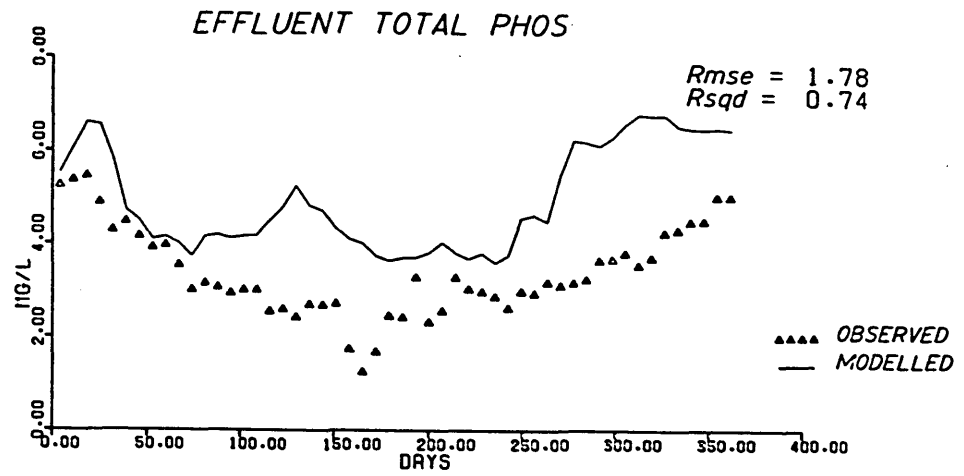


Fig 5.24 Simulation Plot for Pond 2 Effluent Total Phosphorus (Zero Rates Model)

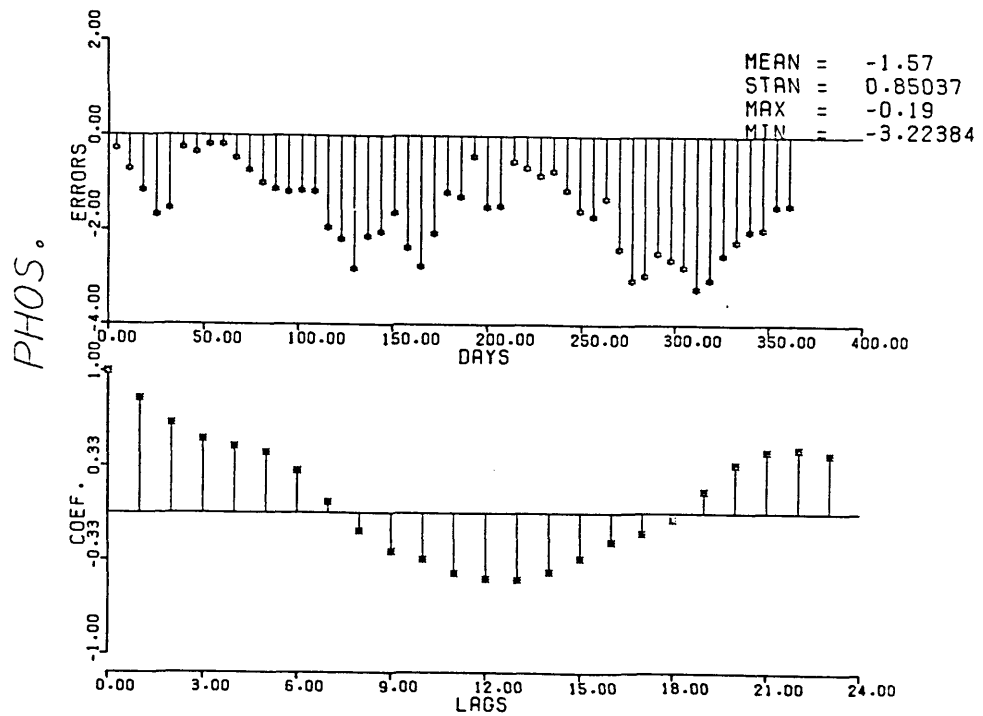


Fig 5.25 Errors Plot for Zero Rates Model Simulation of Pond 2 Effluent Total Phosphorus

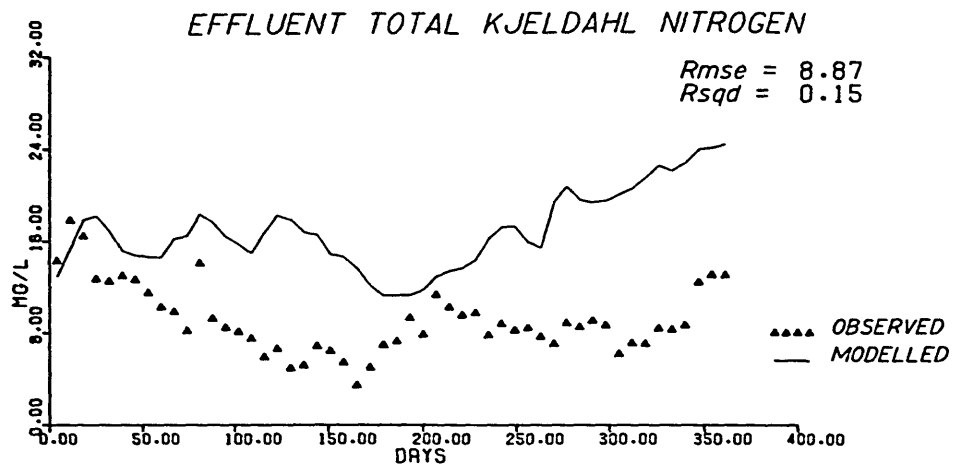


Fig 5.26 Simulation Plot for Pond 2 Effluent Total Kjeldahl Nitrogen (Zero Rates Model)

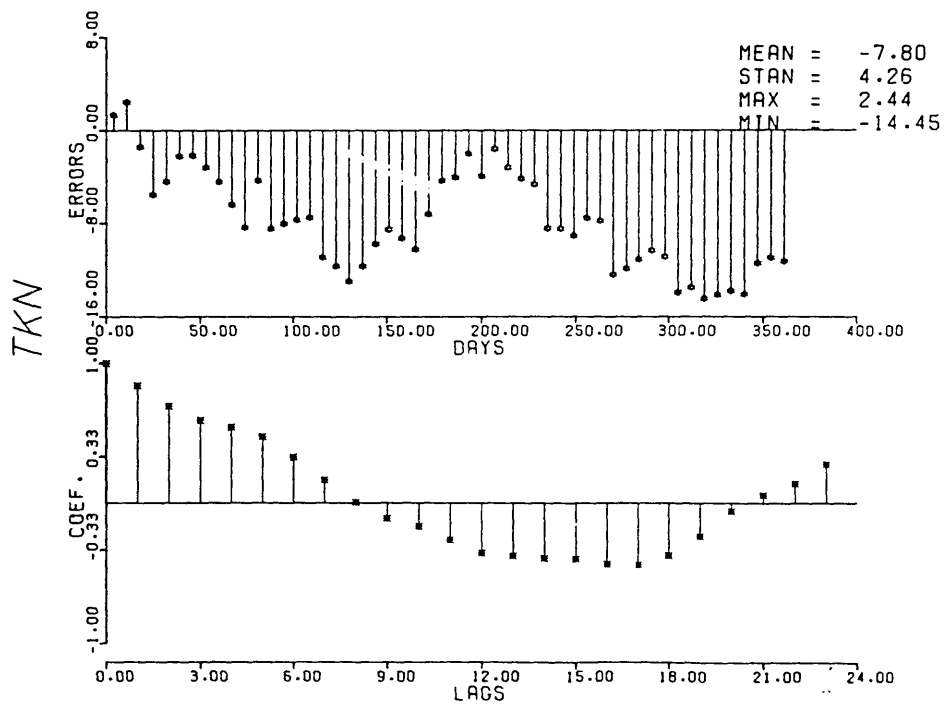


Fig 5.27 Errors Plot for Zero Rates Model Simulation of Pond 2 Effluent Total Kjeldahl Nitrogen

TABLE 5.2:
Estimates of Sink Rate in Source/Sink Model for Pond 1 Data

MODELLED STATES	PARAMETER VALUES ($\text{mg l}^{-1} \text{d}^{-1}$)			
	INSTRUMENTAL VARIABLE			ROSENBROCK (SLS)
	Q = 0	Q = .001	Q = 1	
Filtered COD	5.870	3.970	3.53	5.477
Volatile Solids	7.910	5.670	3.63	7.688
Organic Phosphorous	0.194	0.189*	0.140	0.191
Ortho - Phosphorous	0.885	0.945*	1.105	0.926
Ammonia	1.580	1.530*	1.190	1.577
Nitrate	0.002	0.002	-0.003	0.001

* Q = 0.0001

TABLE 5.3:
Estimates of Sink Rate in Source/Sink Model for Pond 2 Data

MODELLED STATES	PARAMETER VALUES ($\text{mg l}^{-1} \text{d}^{-1}$)			
	INSTRUMENTAL VARIABLE			ROSENBROCK (SLS)
	Q = 0	Q = .001	Q = 1	
Soluble COD	0.601	0.538	0.677	0.506
Total Solids	-0.056	0.570	0.719	0.364
Ammonia	0.223	0.165	1.190	0.174
Nitrate	0.038	0.006	-0.003	0.018
Total Phosphorous	0.048	0.031	1.105	0.039
Total Nitrogen	0.255	0.237	0.140	0.234

TABLE 5.4:
Estimates of First Order Decay Rate Constant for Pond 1 Data

MODELLED STATES	PARAMETER VALUES (d^{-1})	
	EKF	ROSENBROCK (SLS)
	Q = 0 R = 1	
Filtered COD	0.082	0.043
Volatile Solids	0.024	0.020
Organic Phosphorous	0.271	0.038
Ortho-Phosphorous	0.033	0.026
Ammonia	0.028	0.021
Nitrate	0.018	0.018

TABLE 5.5:
Estimates of First Order Decay Rate Constant for Pond 2 Data

MODELLED STATES	PARAMETER VALUES (d^{-1})	
	EKF	ROSENBROCK (SLS)
	Q = 0 R = 1	
Soluble COD	0.021	0.0163
Total Solids	0.021	0.0404
Ammonia	0.054	0.1458
Nitrate	0.162	0.2329
Total Phosphorous	0.015	0.0163
Total Nitrogen	0.038	0.0314

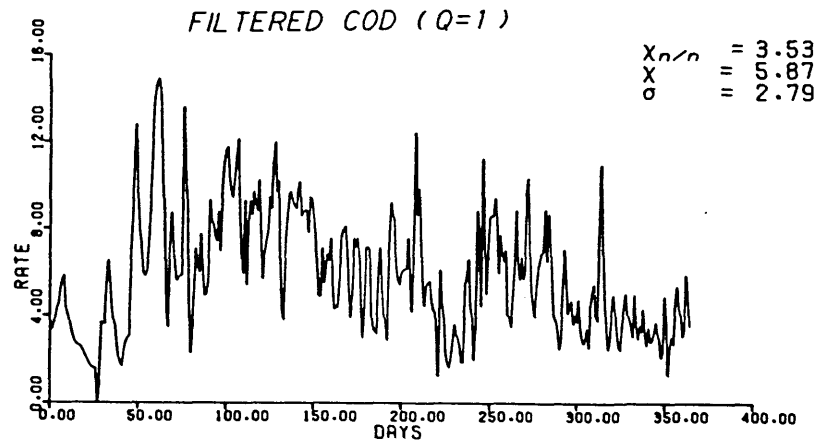


Fig 5.28 Trajectory of IV estimate of Pond 1 Filtered COD S/Sink Rate (Q = 1)

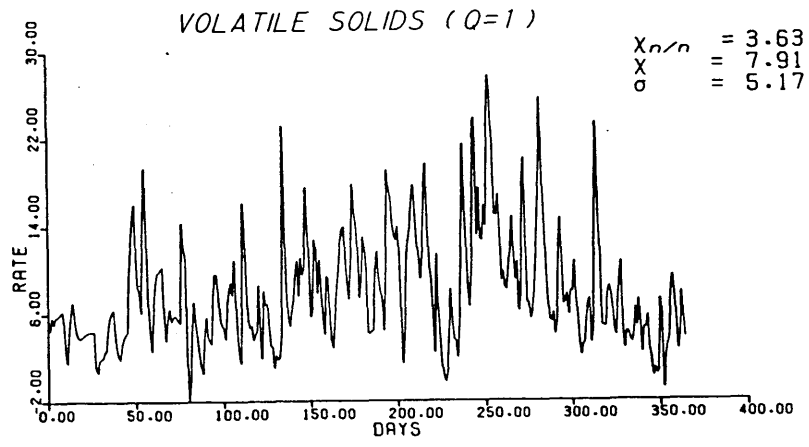


Fig 5.29 Trajectory of IV estimate of Pond 1 Volatile Suspended Solids S/Sink Rate (Q = 1)

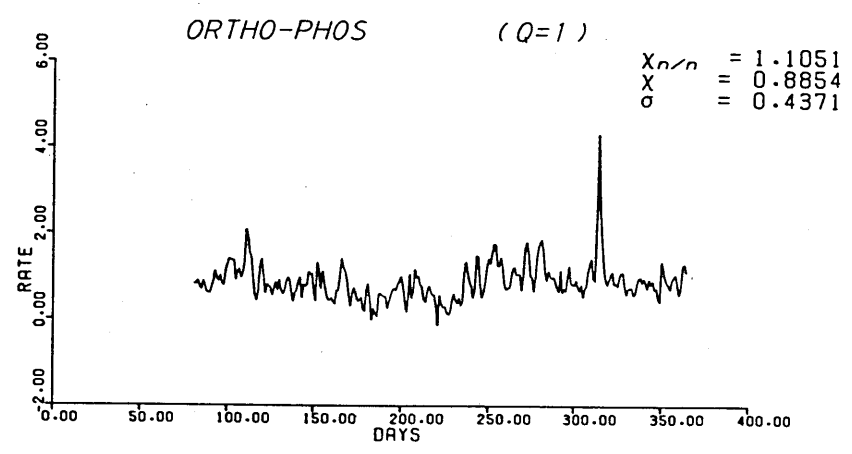


Fig 5.30 Trajectory of IV estimate of Pond 1 Ortho-phosphorus S/Sink Rate (Q = 1)

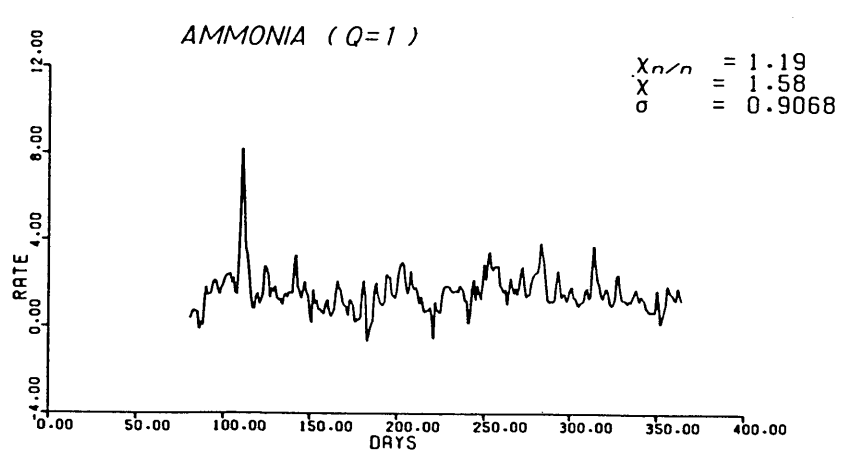


Fig 5.31 Trajectory of IV estimate of Pond 1 Ammonia S/Sink Rate (Q = 1)

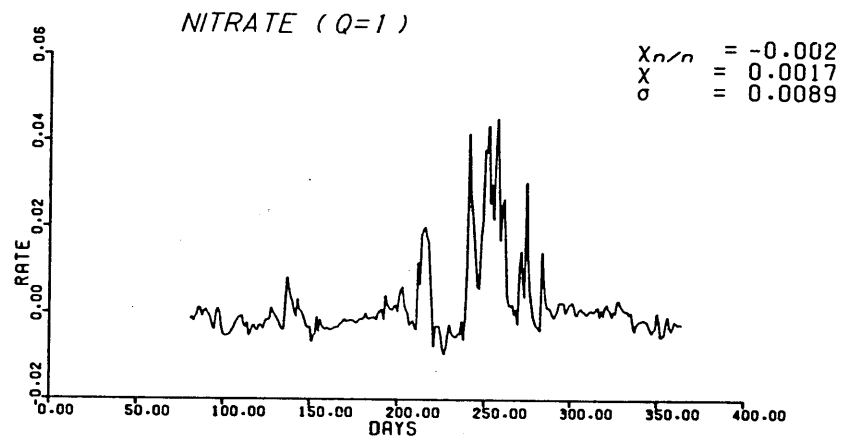


Fig 5.32 Trajectory of IV estimate of Pond 1 Nitrate S/Sink Rate (Q = 1)

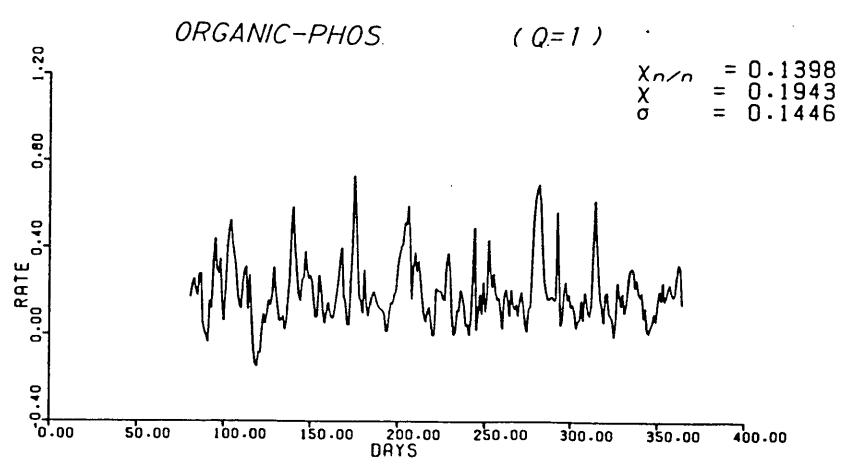


Fig 5.33 Trajectory of IV estimate of Pond 1 Organic-Phosphorus S/Sink Rate (Q = 1)

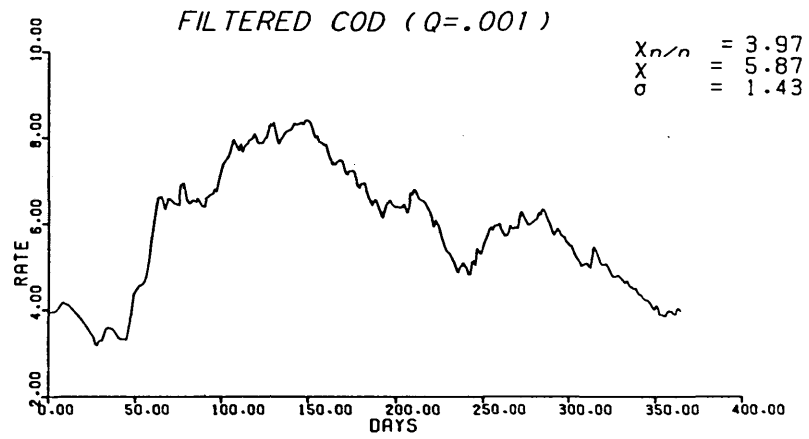


Fig 5.34 Trajectory of IV estimate of Pond 1 Filtered COD S/Sink Rate (Q = .001)

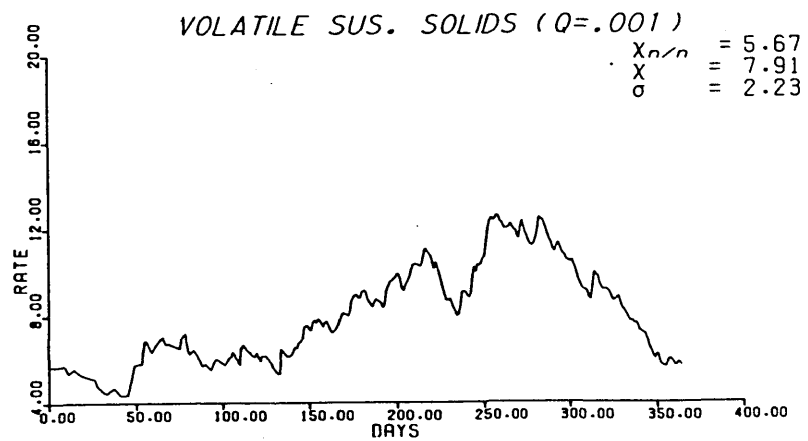


Fig 5.35 Trajectory of IV estimate of Pond 1 Volatile Suspended Solids S/Sink Rate (Q = .001)

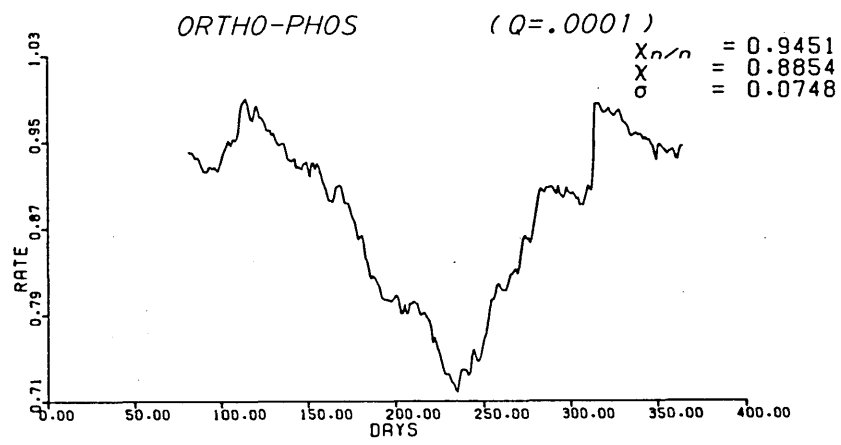


Fig 5.36 Trajectory of IV estimate of Pond 1 Ortho-Phosphorus S/Sink Rate (Q = .0001)

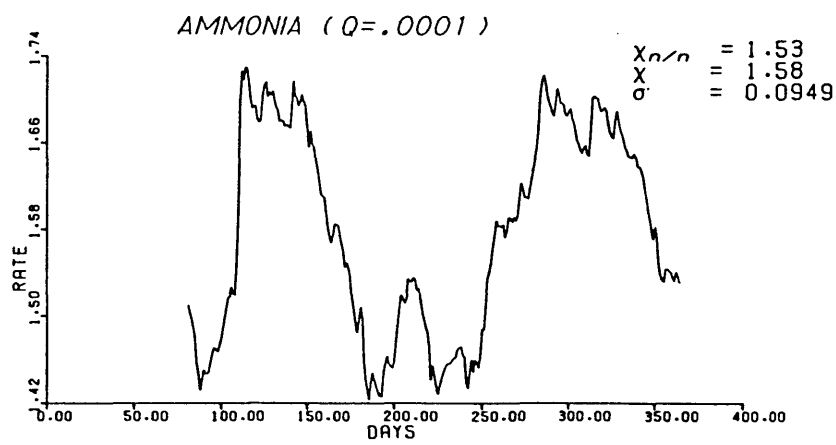


Fig 5.37 Trajectory of IV estimate of Pond 1 Ammonia S/Sink Rate (Q = .0001)

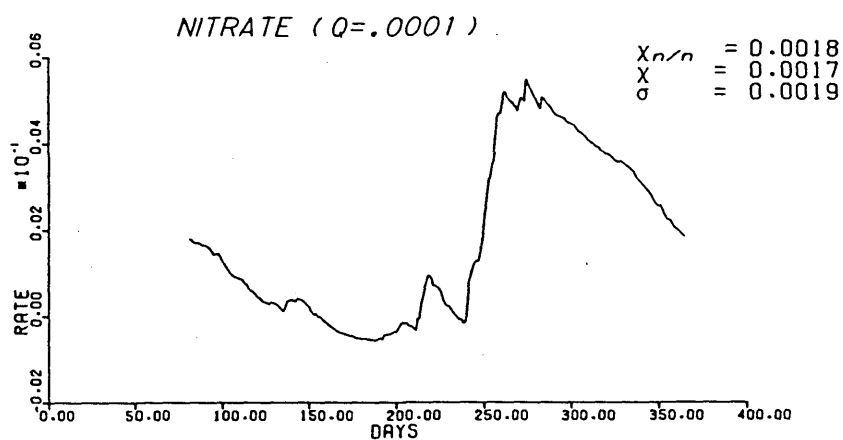


Fig 5.38 Trajectory of IV estimate of Pond 1 Nitrate
S/Sink Rate (Q = .0001)

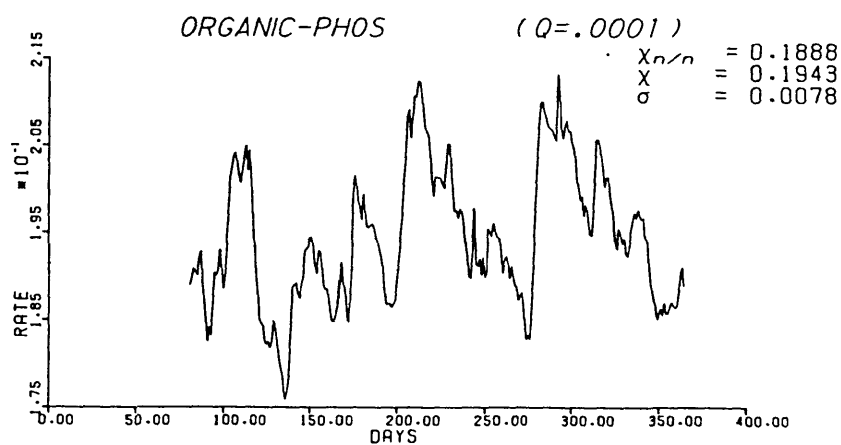


Fig 5.39 Trajectory of IV estimate of Pond 1 Organic-Phosphorus
S/Sink Rate (Q = .0001)

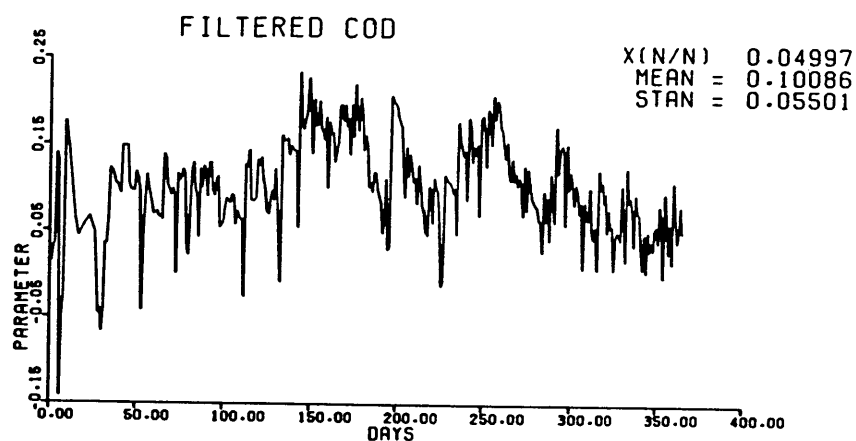


Fig 5.40 Trajectory of EKF estimate of Pond 1 Filtered COD First Order Decay Rate

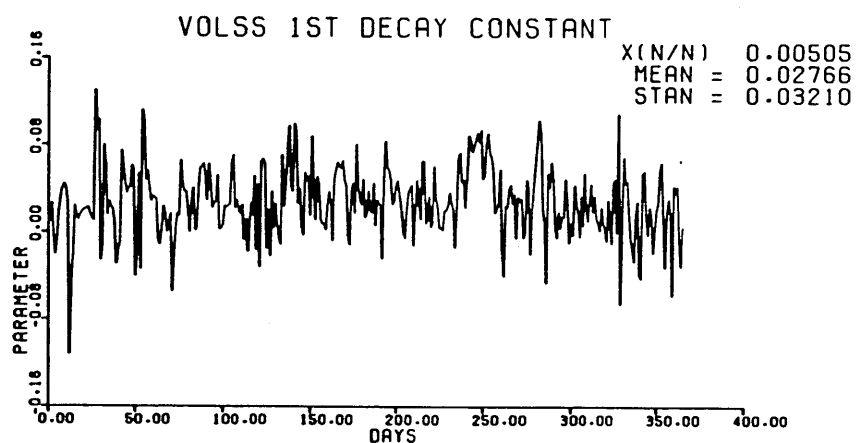


Fig 5.41 Trajectory of EKF estimate of Pond 1 Volatile Suspended Solids First Order Decay Rate

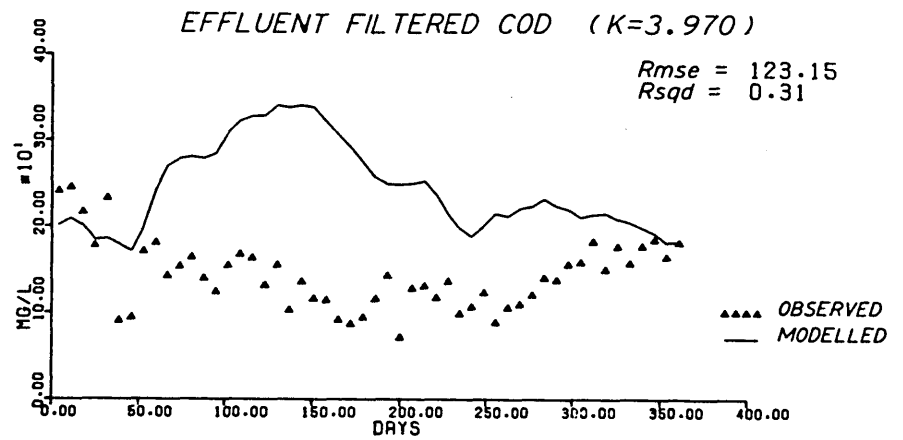


Fig 5.42 Simulation Plot for Pond 1 Effluent Filtered COD:
S/Sink Model (IV estimate of Sink Rate for $Q = .001$)

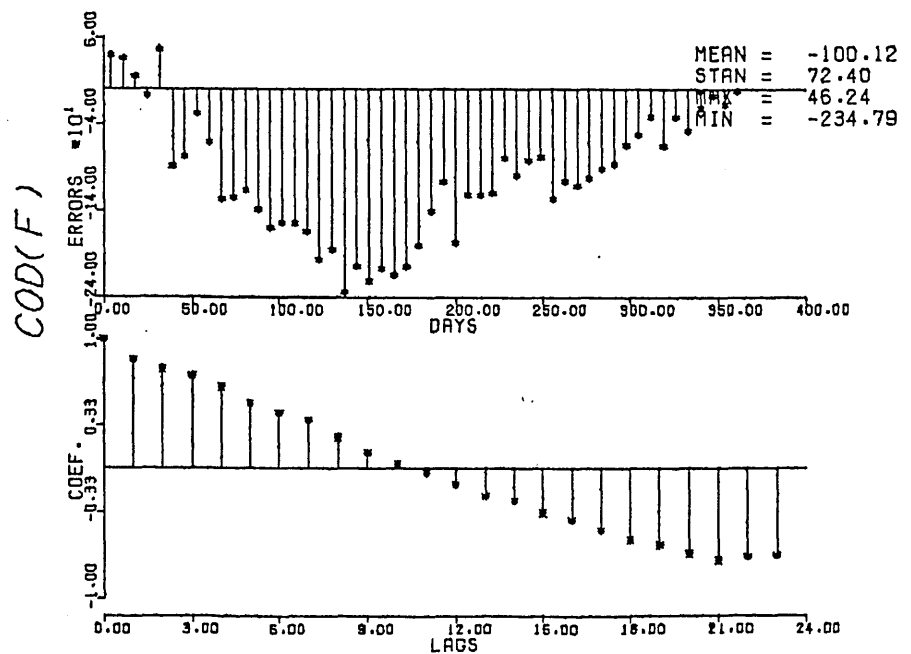


Fig 5.43 Errors Plot for S/Sink Model Simulation of Pond 1
Effluent Filtered COD (IV estimate of Sink Rate for
 $Q = .001$)

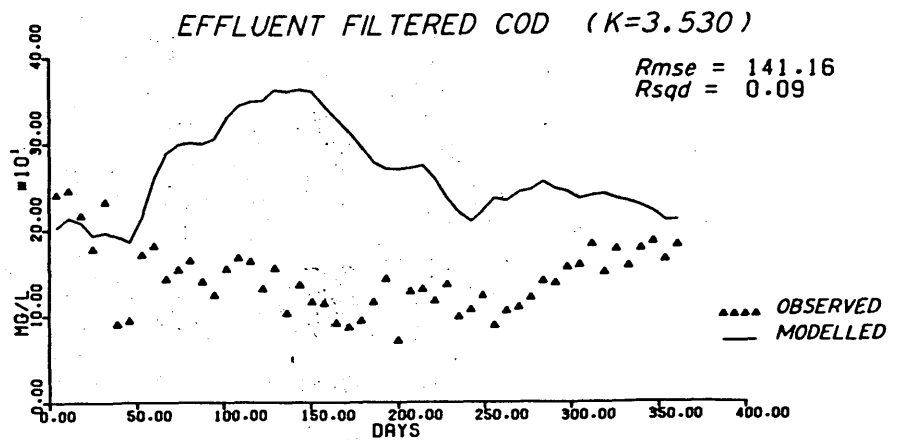


Fig 5.44 Simulation Plot for Pond 1 Effluent filtered COD:
S/Sink Model (IV estimate of Sink Rate for Q = 1)

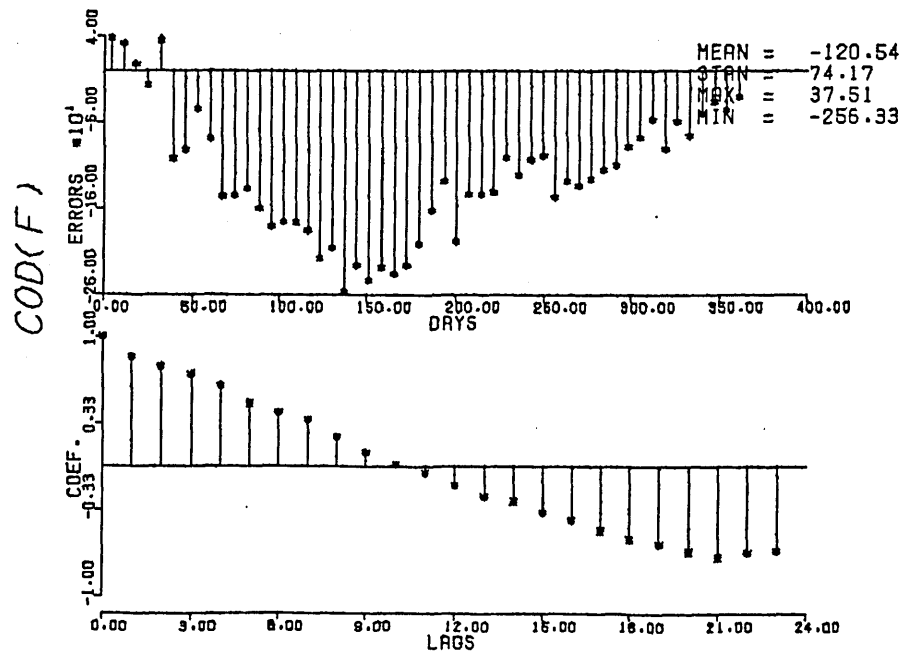


Fig 5.45 Errors Plot for S/Sink Model Simulation of Pond 1
Effluent filtered COD (IV estimate of Sink Rate for
Q = 1)

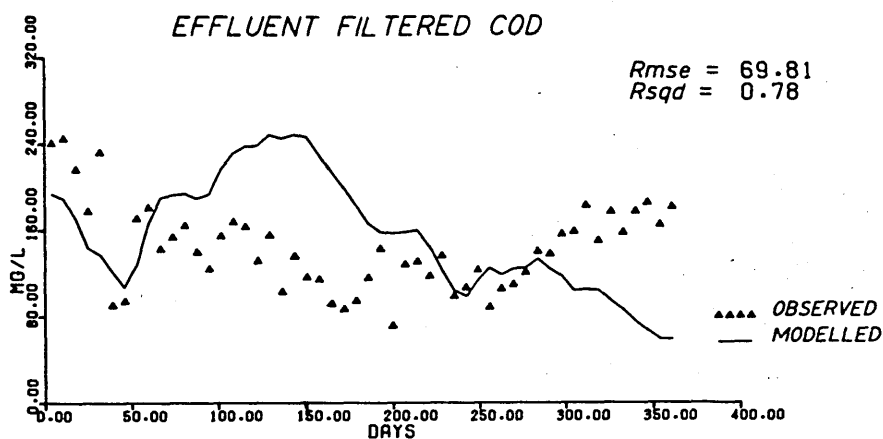


Fig 5.46 Simulation Plot for Pond 1 Effluent filtered COD:
S/Sink Model (IV estimate of Sink Rate for $Q = 0$)

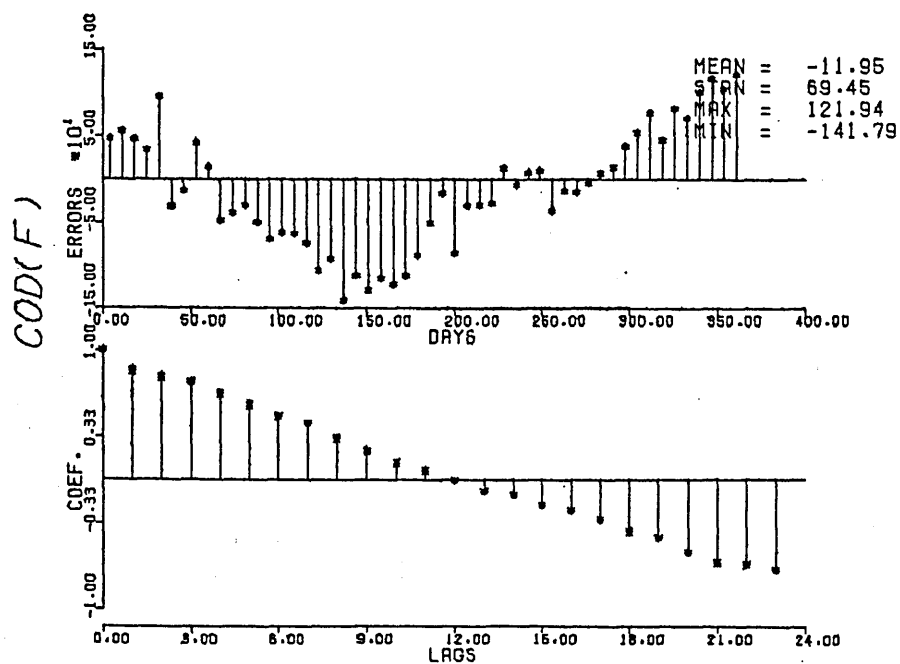


Fig 5.47 Errors Plot for S/Sink Model Simulation of Pond 1
Effluent filtered Cod (IV estimate of Sink Rate for $Q = 0$)

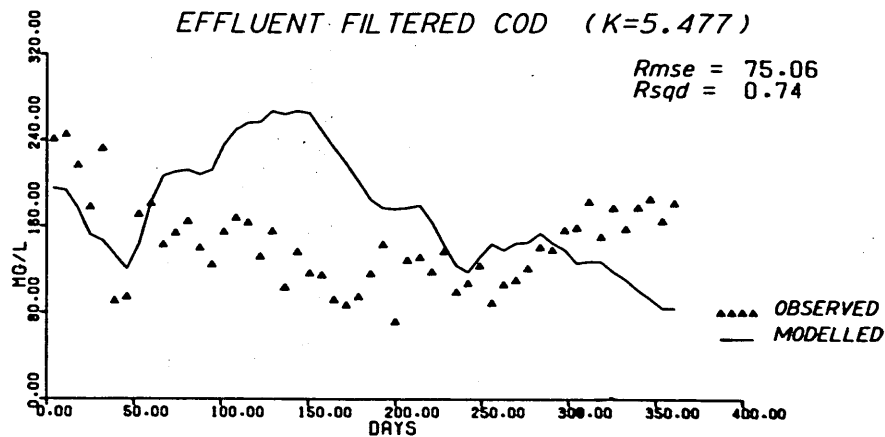


Fig 5.48 Simulation Plot for Pond 1 Effluent filtered COD:
S/Sink Model (Rosenbrock estimate of Sink Rate)

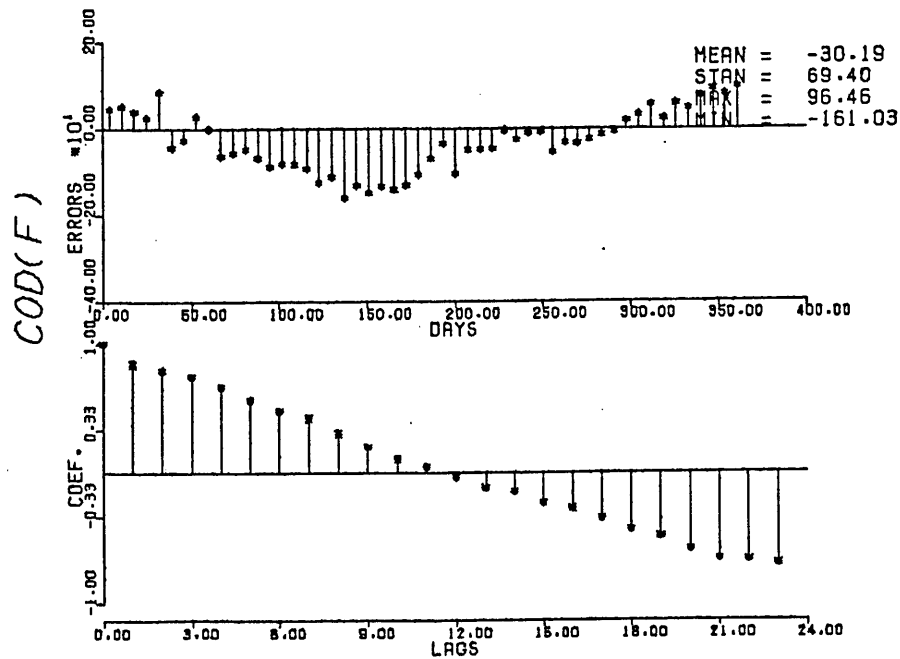


Fig 5.49 Errors Plot for Sink Model Simulation of Pond 1
Effluent filtered COD (Rosenbrock estimate of Sink Rate)

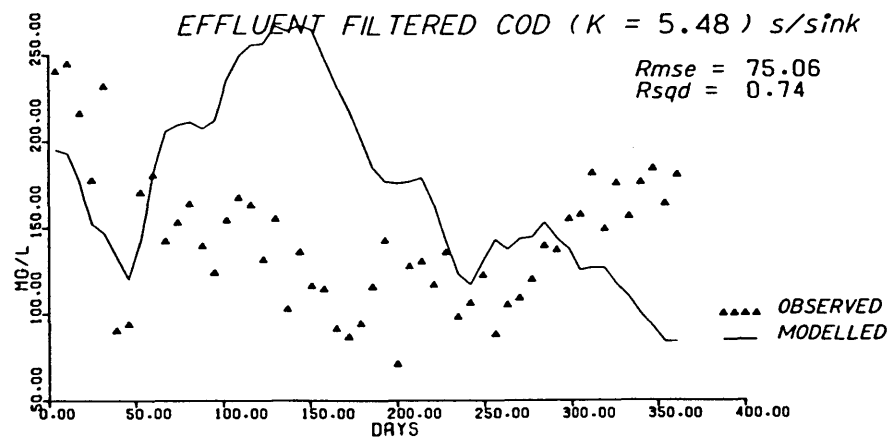


Fig 5.50 Simulation Plot for Pond 1 Effluent Filtered COD (S/Sink Model)

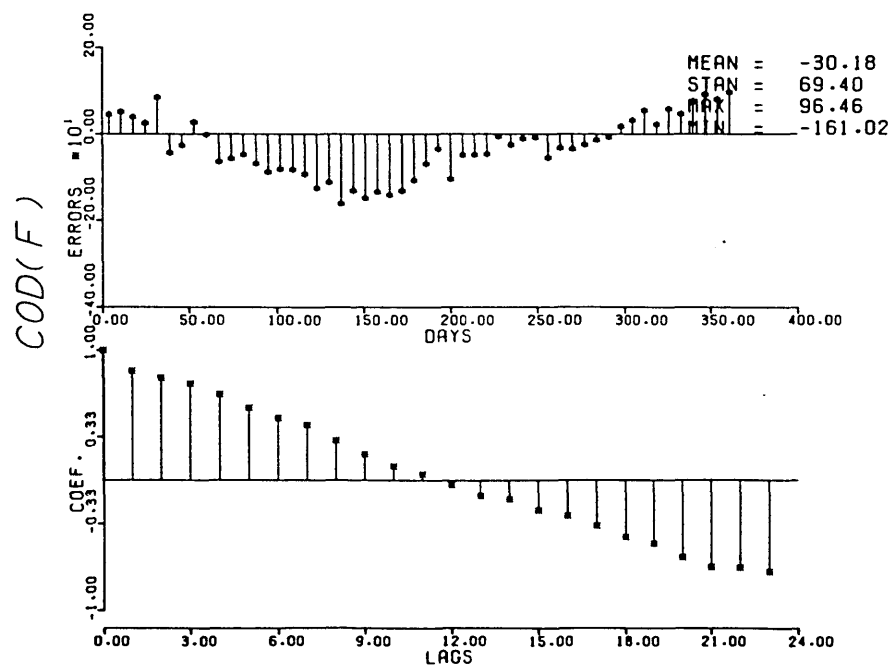


Fig 5.51 Errors Plot for S/Sink Model Simulation of Pond 1 Effluent Filtered COD

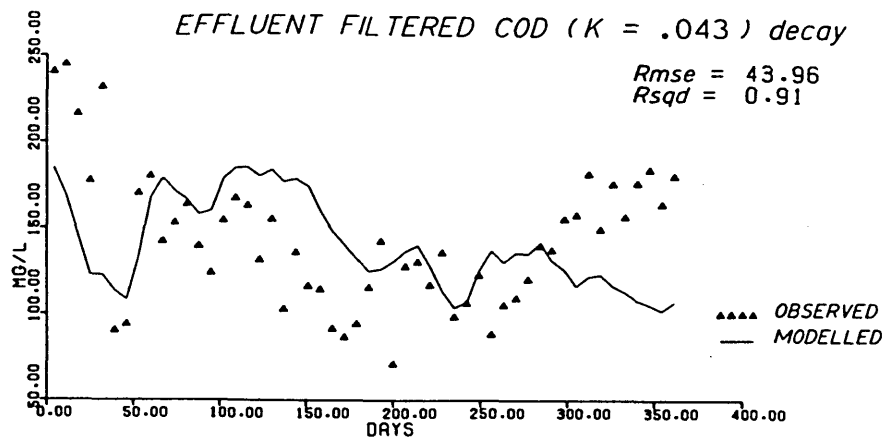


Fig 5.52 Simulation Plot for Pond 1 Effluent Filtered COD (First Order Decay Rate Model)

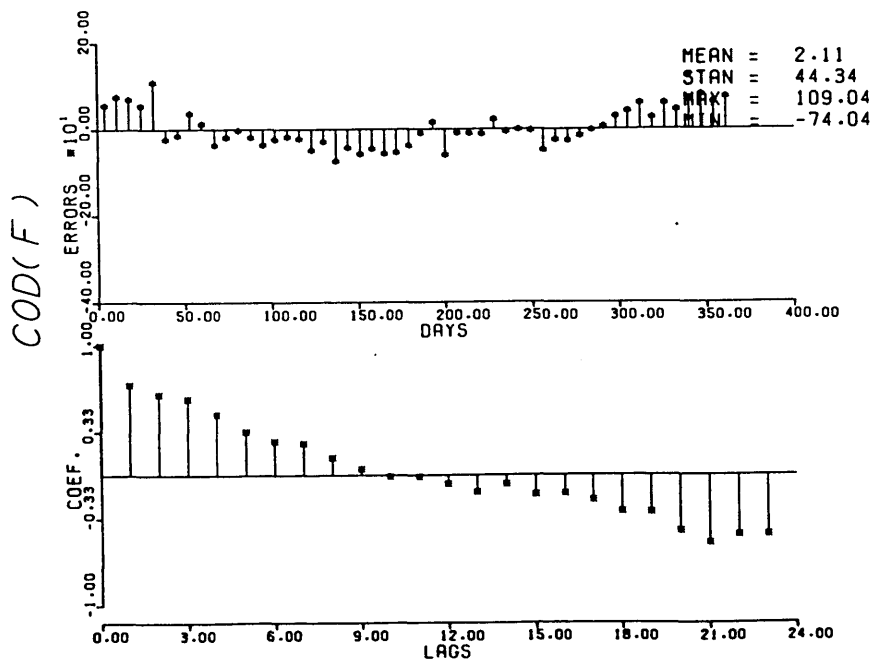


Fig 5.53 Errors Plot for First Order Decay Rate Model Simulation of Pond 1 Effluent Filtered COD

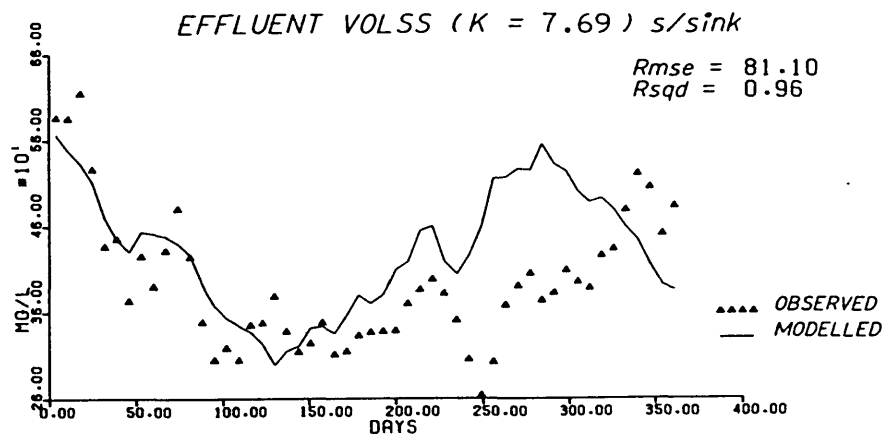


Fig 5.54 Simulation Plot for Pond 1 Effluent Volatile Suspended Solids (S/Sink Model)

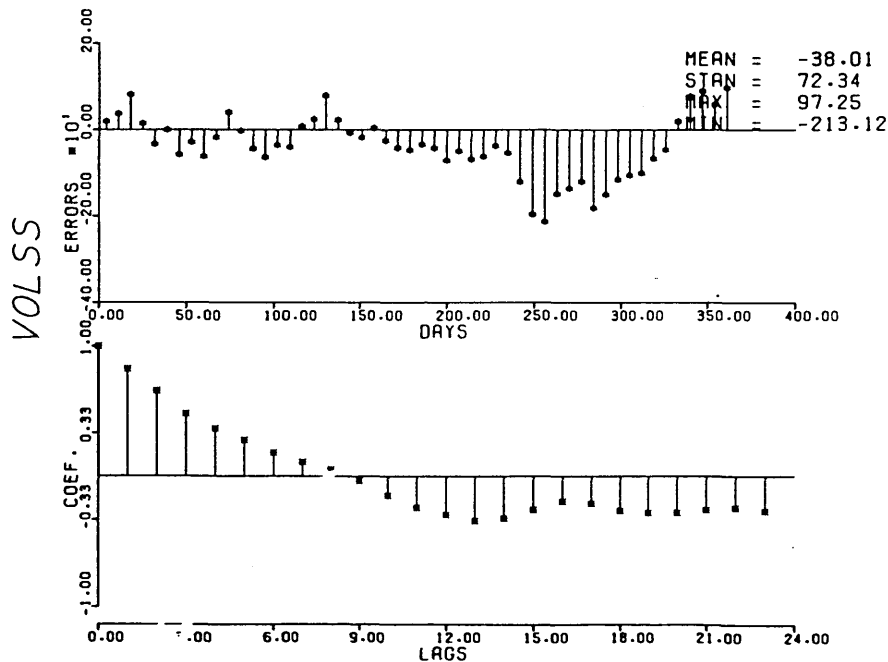


Fig 5.55 Errors Plot for S/Sink Model Simulation of Pond 1 Effluent Volatile Suspended Solids

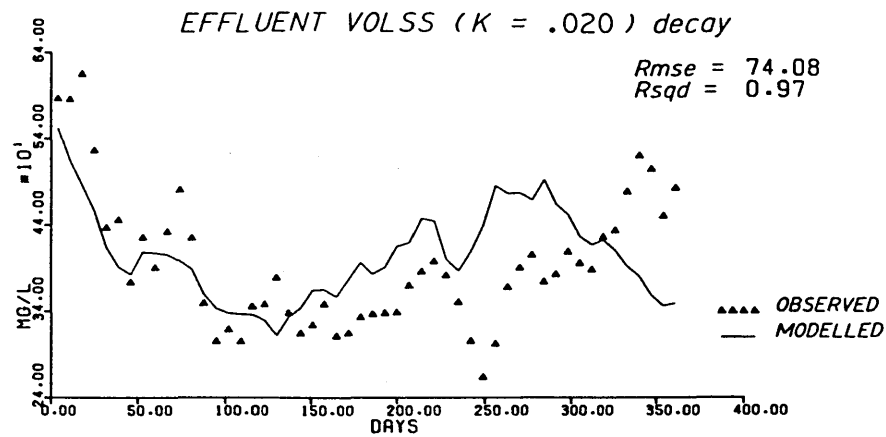


Fig 5.56 Simulation Plot for Pond 1 Effluent Volatile Suspended Solids (First Order Decay Rate Model)

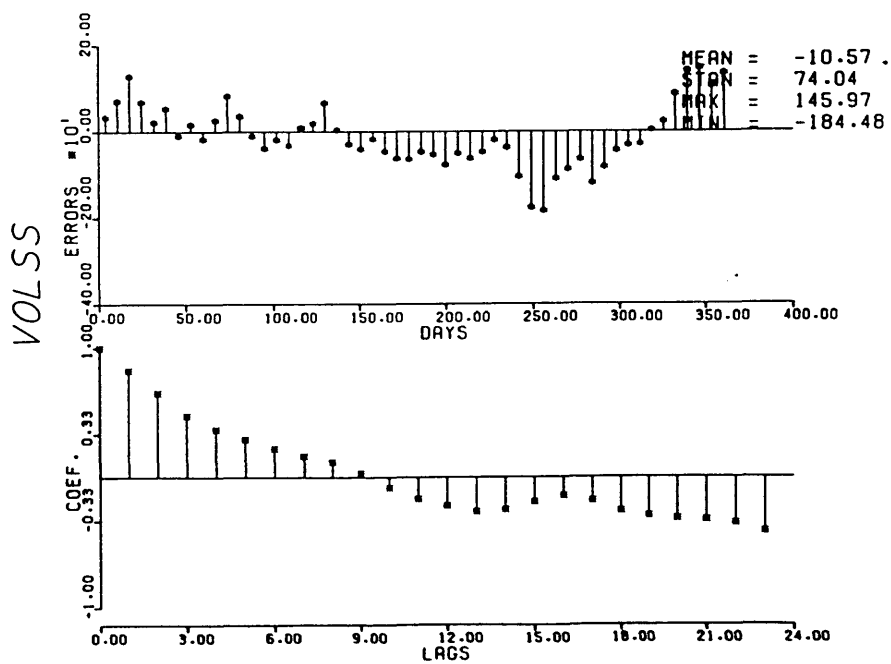


Fig 5.57 Errors Plot for First Order Decay Rate Model Simulation of Pond 1 Effluent Volatile Suspended Solids

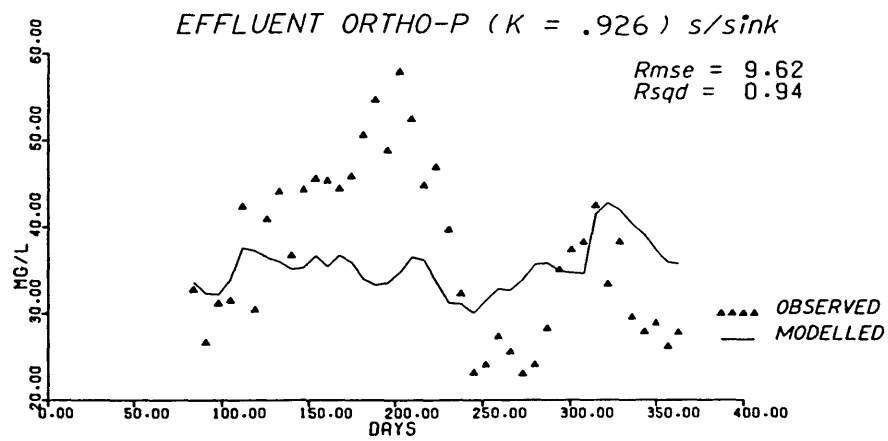


Fig 5.58 Simulation Plot for Pond 1 Effluent Ortho-Phosphorus (S/Sink Model)

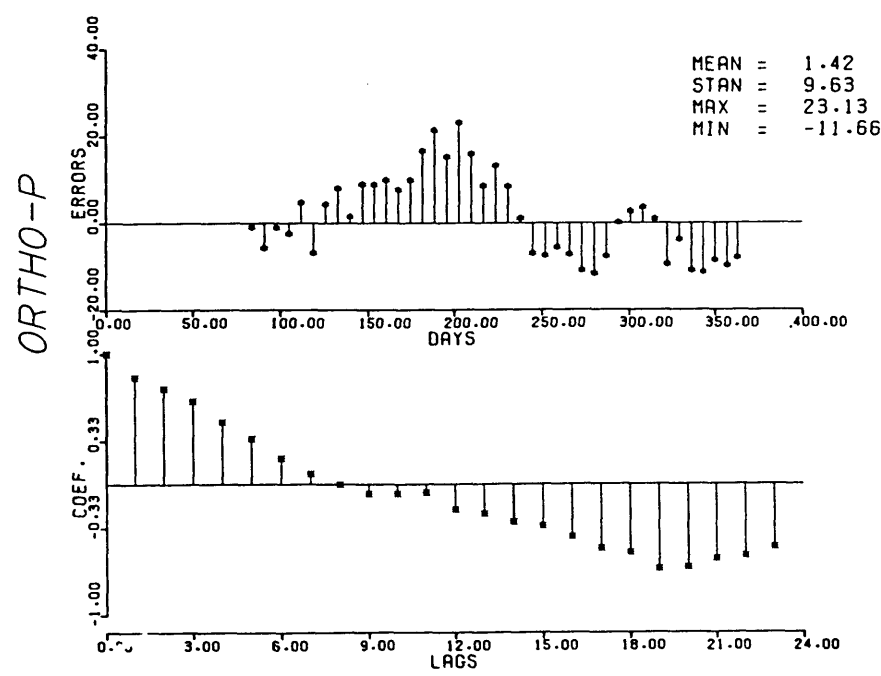


Fig 5.59 Errors Plot for S/Sink Model Simulation of Pond 1 Effluent Ortho-Phosphorus

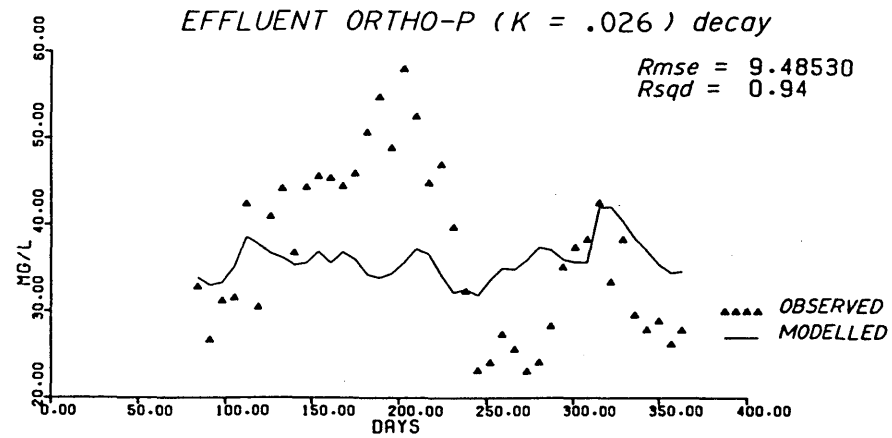


Fig 5.60 Simulation Plot of Pond 1 Effluent Ortho-Phosphorus (First Order Decay Model)

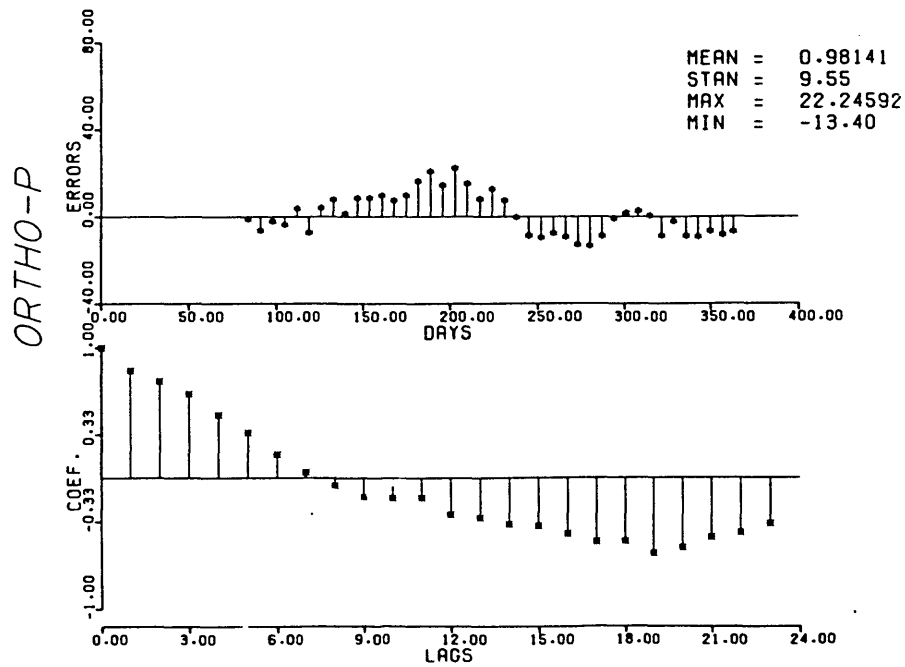


Fig 5.61 Errors Plot for First Order Decay Rate Model Simulation of Pond 1 Effluent Ortho-Phosphorus

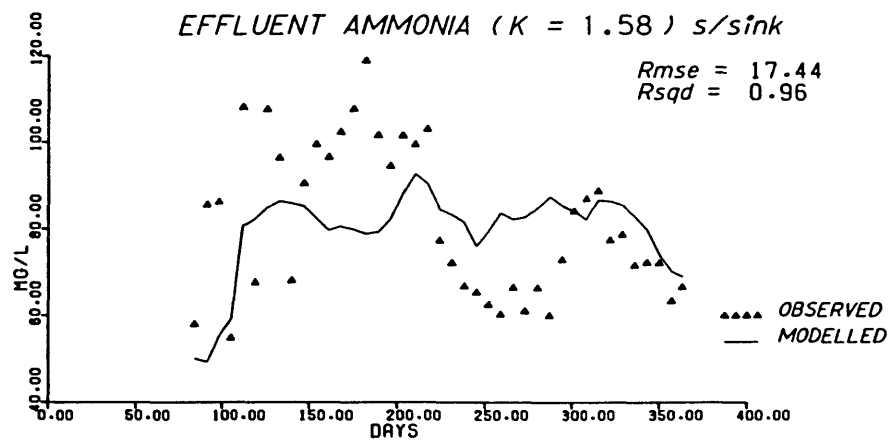


Fig 5.62 Simulation Plot for Pond 1 Effluent Ammonia (S/Sink Model)

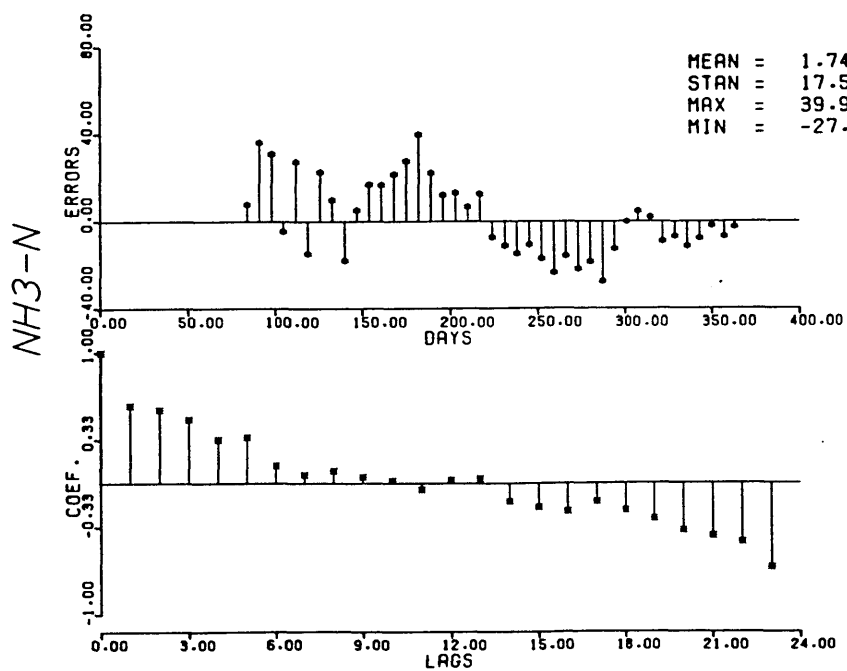


Fig 5.63 Errors Plot for S/Sink Model Simulation of Pond 1 Effluent Ammonia

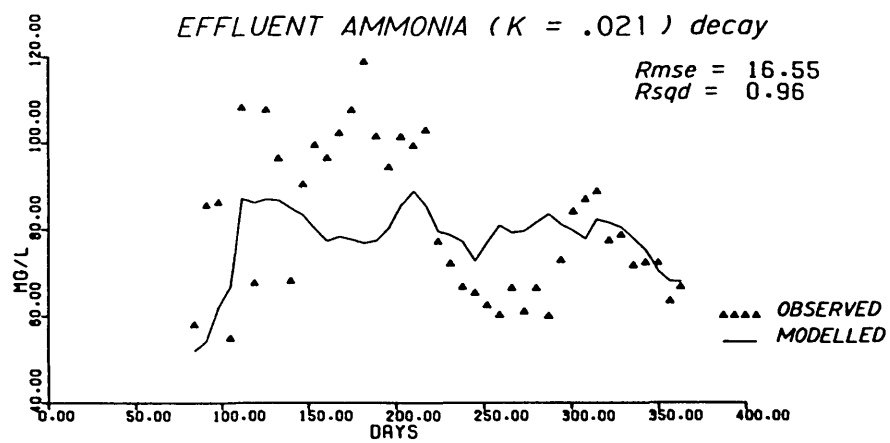


Fig 5.64 Simulation Plot of Pond 1 Effluent Ammonia (First Order Decay Model)

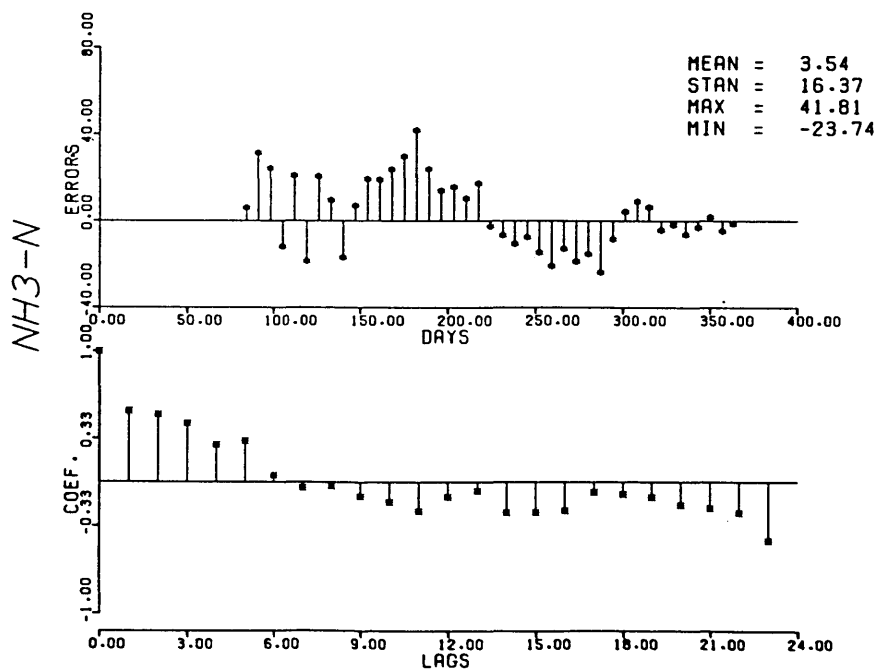


Fig 5.65 Errors Plot for First Order Decay Rate Model Simulation of Pond 1 Effluent Ammonia

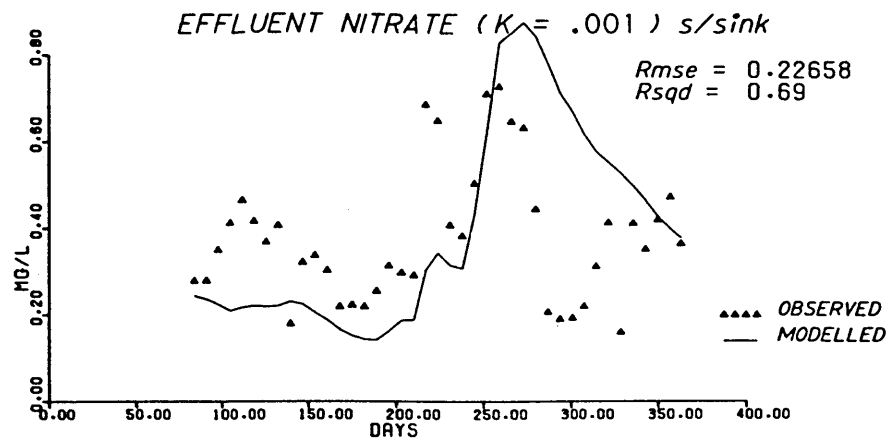


Fig 5.66 Simulation Plot for Pond 1 Effluent Nitrate (S/Sink Model)

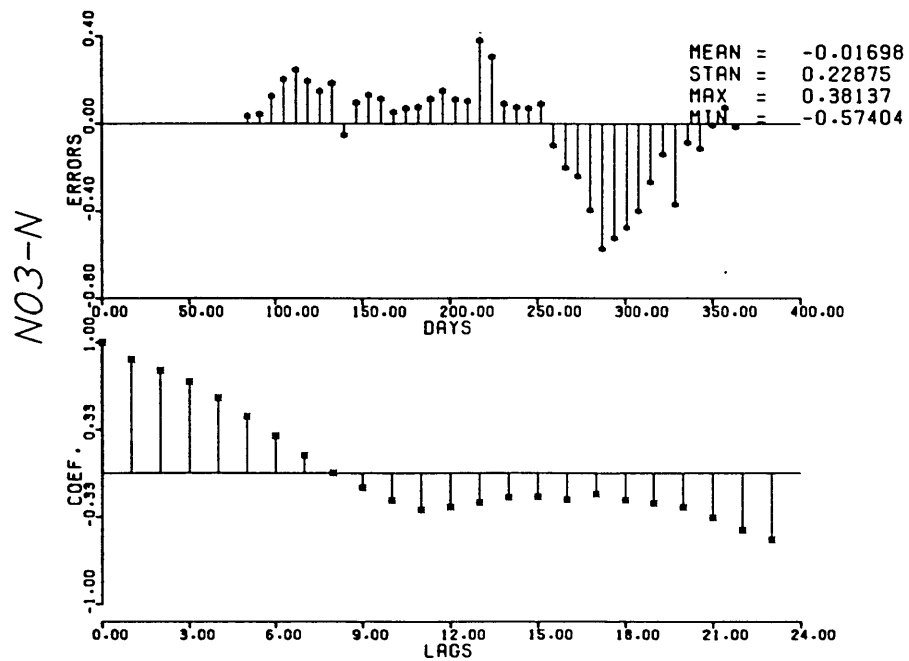


Fig 5.67 Errors Plot for S/Sink Model Simulation of Pond 1 Effluent Nitrate

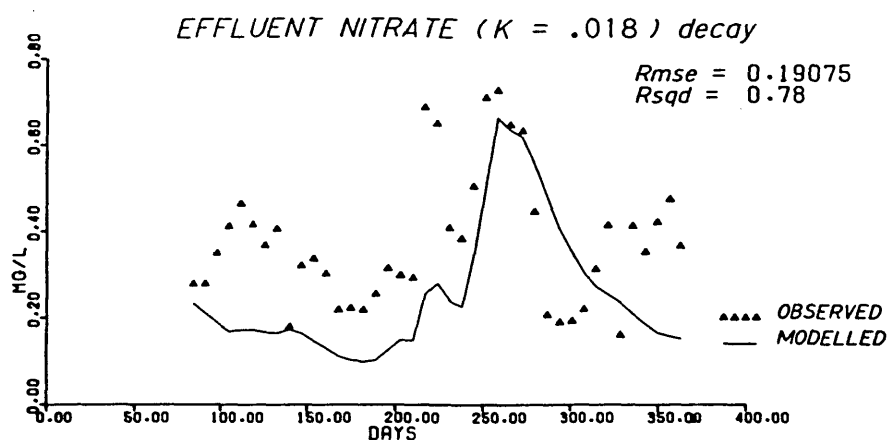


Fig 5.68 Simulation Plot of Pond 1 Effluent Nitrate
(First Order Decay Model)

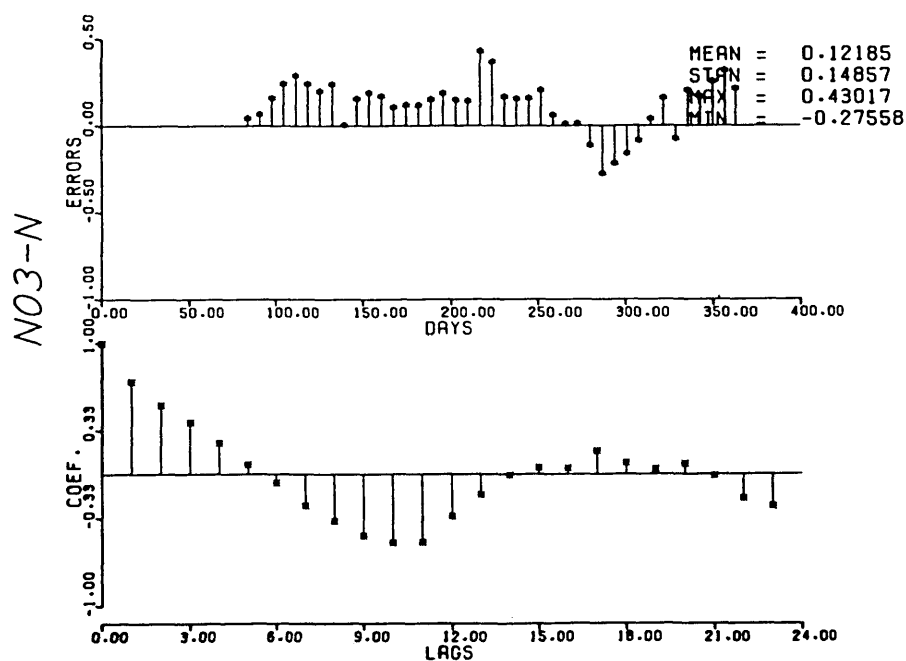


Fig 5.69 Errors Plot for First Order Decay Rate Model Simulation
of Pond 1 Effluent Nitrate

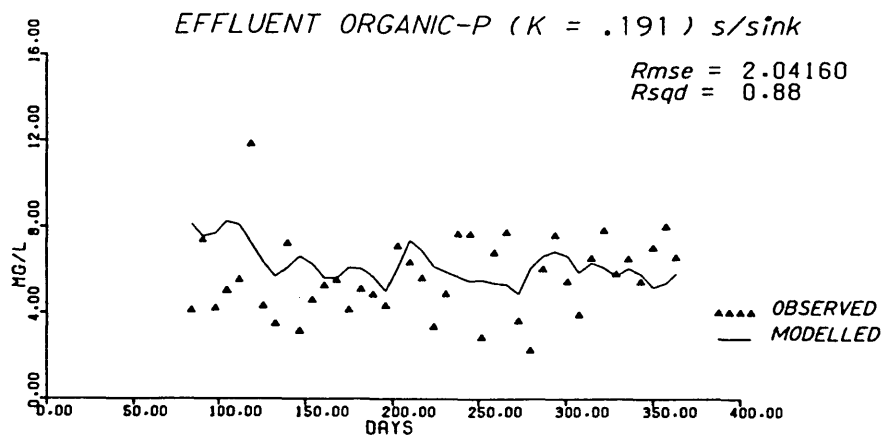


Fig 5.70 Simulation Plot for Pond 1 Effluent Organic-Phosphorus (S/Sink Model)

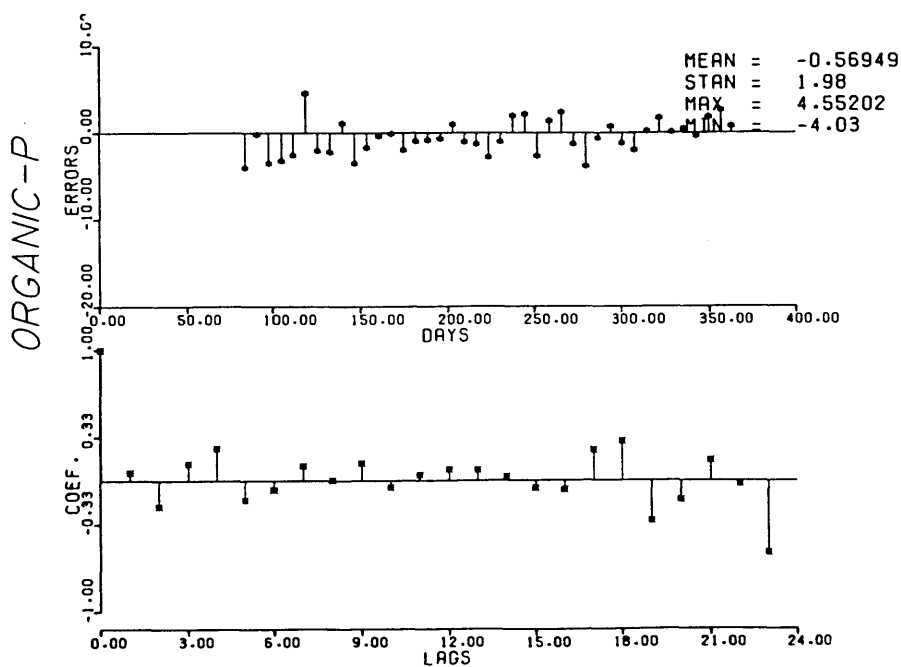


Fig 5.71 Errors Plot for S/Sink Model Simulation of Pond 1 Effluent Organic-Phosphorus

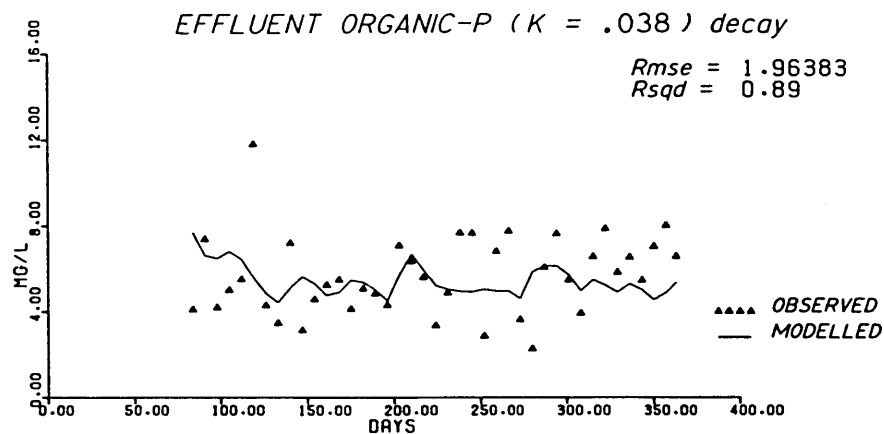


Fig 5.72 Simulation Plot of Pond 1 Effluent Organic-Phosphorus (First Order Decay Model)

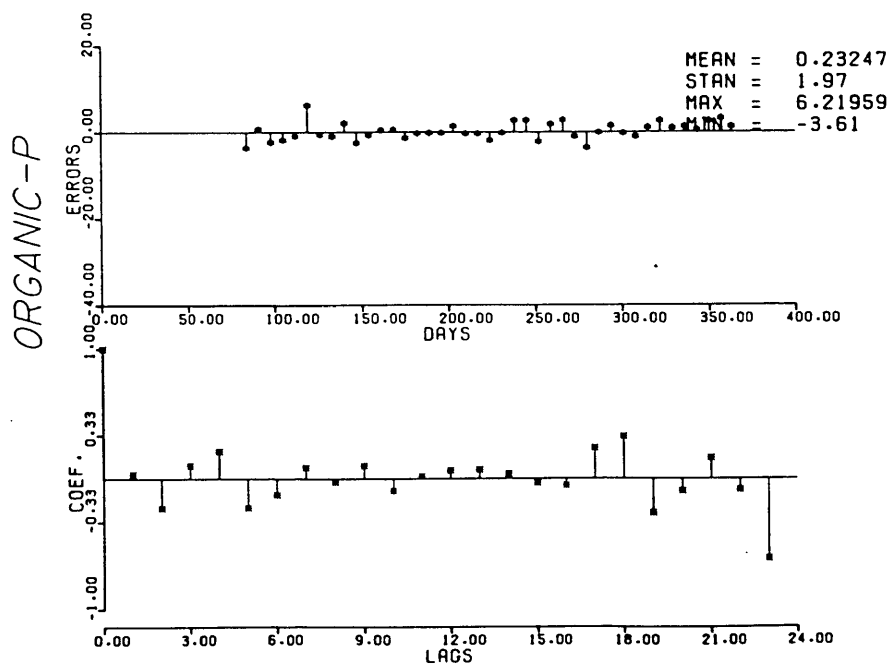


Fig 5.73 Errors Plot for First Order Decay Rate Model Simulation of Pond 1 Effluent Organic-Phosphorus

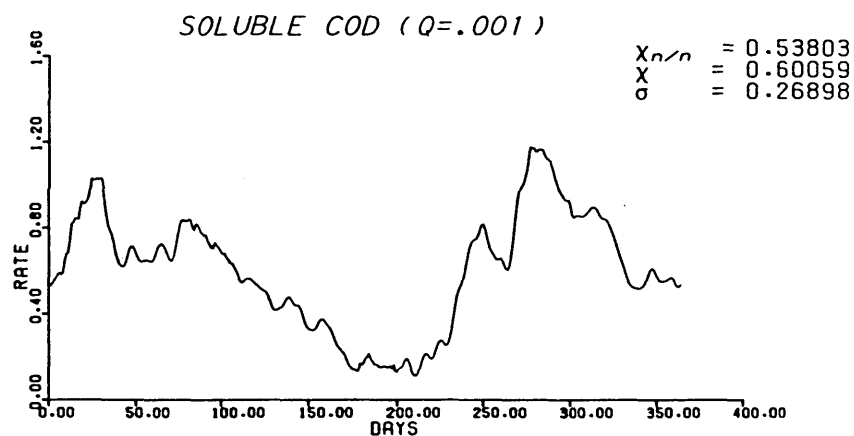


Fig 5.74 Trajectory of IV estimate of Pond 2 Soluble COD
S/Sink Rate (Q = .001)

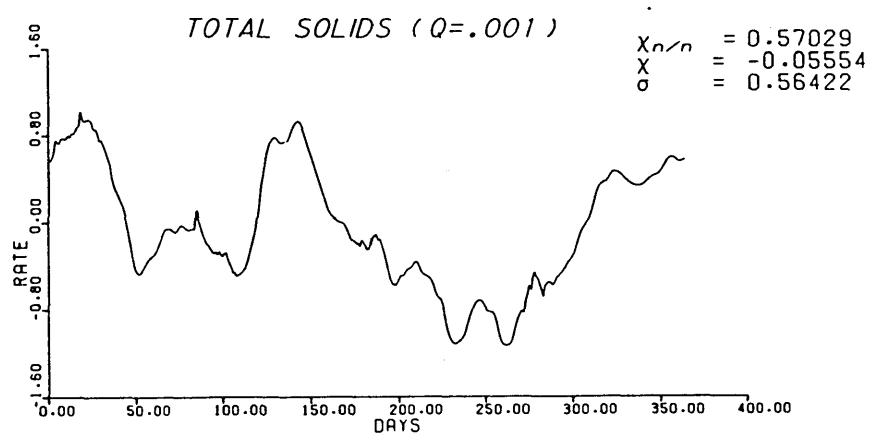


Fig 5.75 Trajectory of IV estimate of Pond 2 Total Solids
S/Sink Rate (Q = .001)

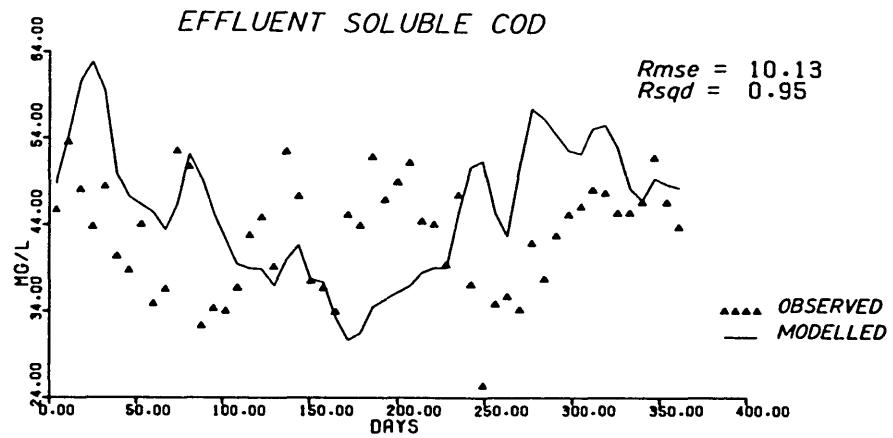


Fig 5.76 Simulation Plot for Pond 2 Effluent Soluble COD (S/Sink Model)

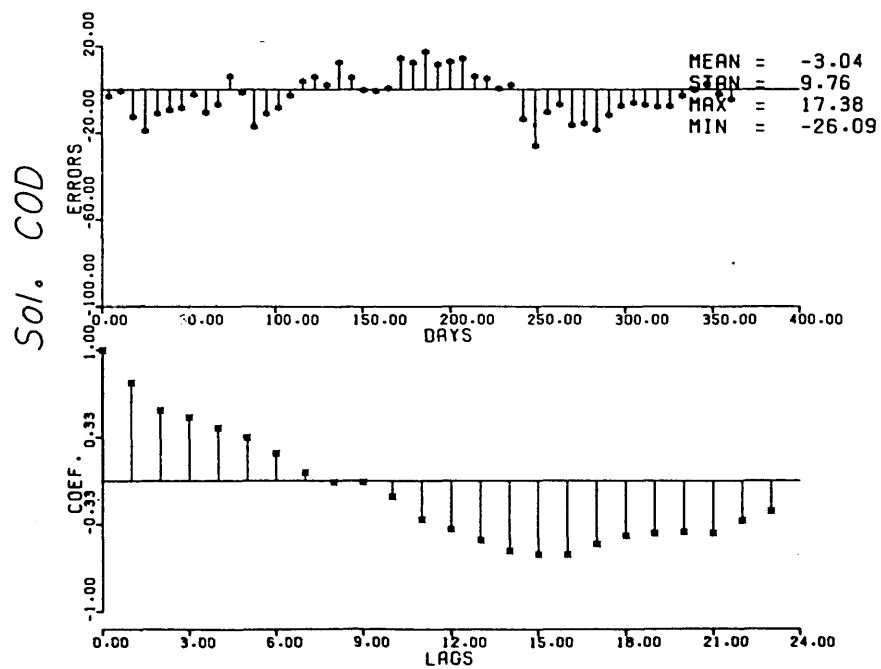


Fig 5.77 Errors Plot for S/Sink Model Simulation of Pond 2 Effluent Soluble COD

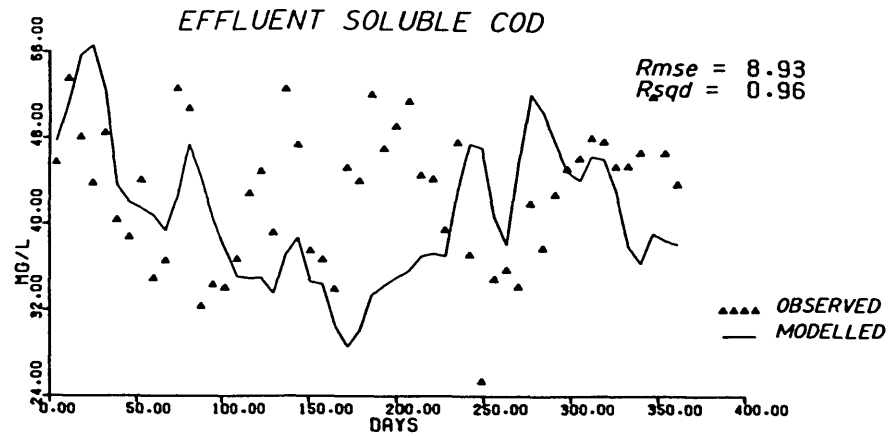


Fig 5.78 Simulation Plot for Pond 2 Effluent Soluble COD (First Order Decay Model)

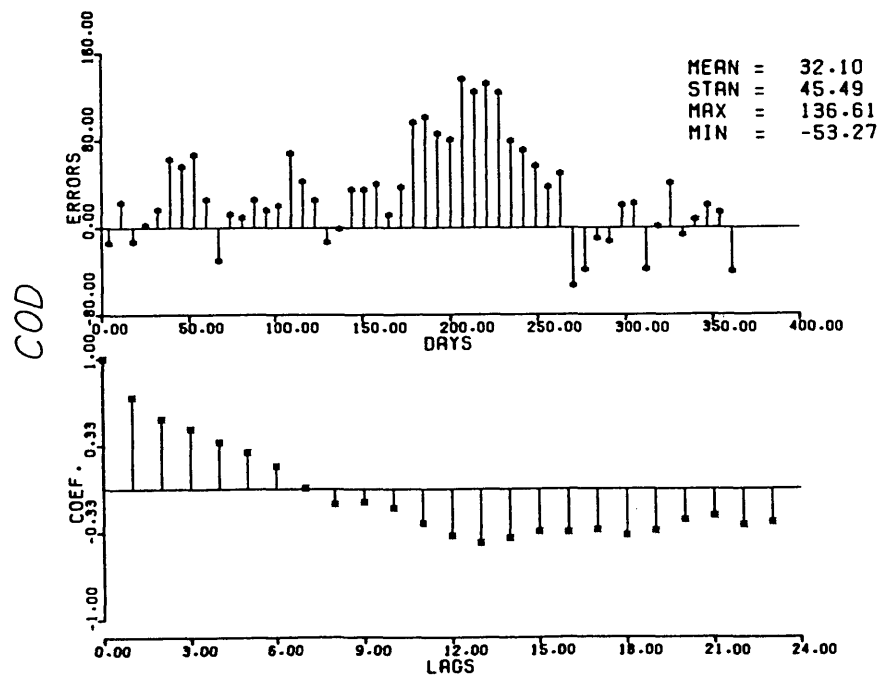


Fig 5.79 Errors Plot for First Order Decay Model Simulation of Pond 2 Effluent Soluble COD

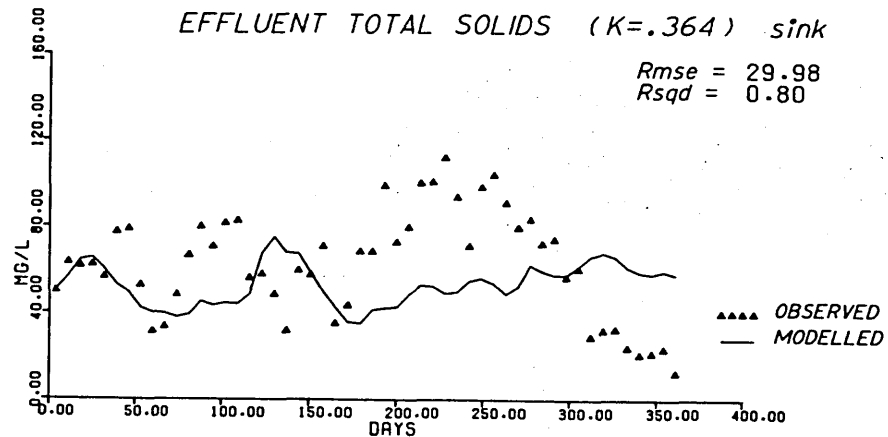


Fig 5.80 Simulation Plot for Pond 2 Effluent Total Solids (S/Sink Model)

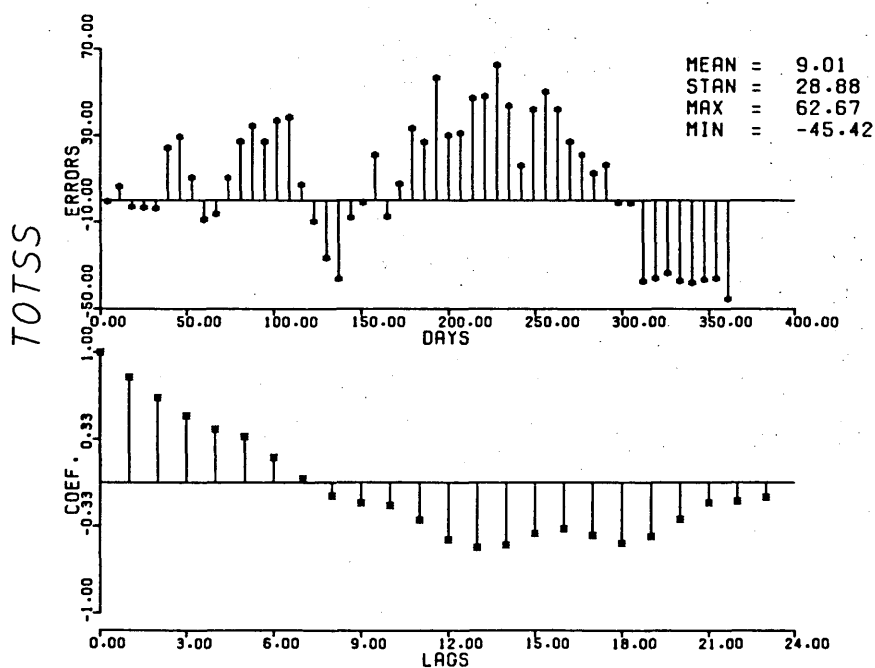


Fig 5.81 Errors Plot for S/Sink Model Simulation of Pond 2 Effluent Total Solids

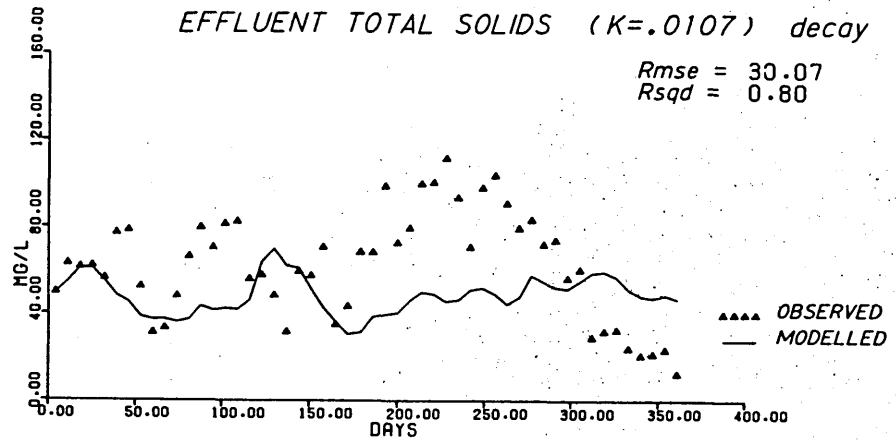


Fig 5.82 Simulation Plot for Pond 2 Effluent Total Solids (First Order Decay Model)

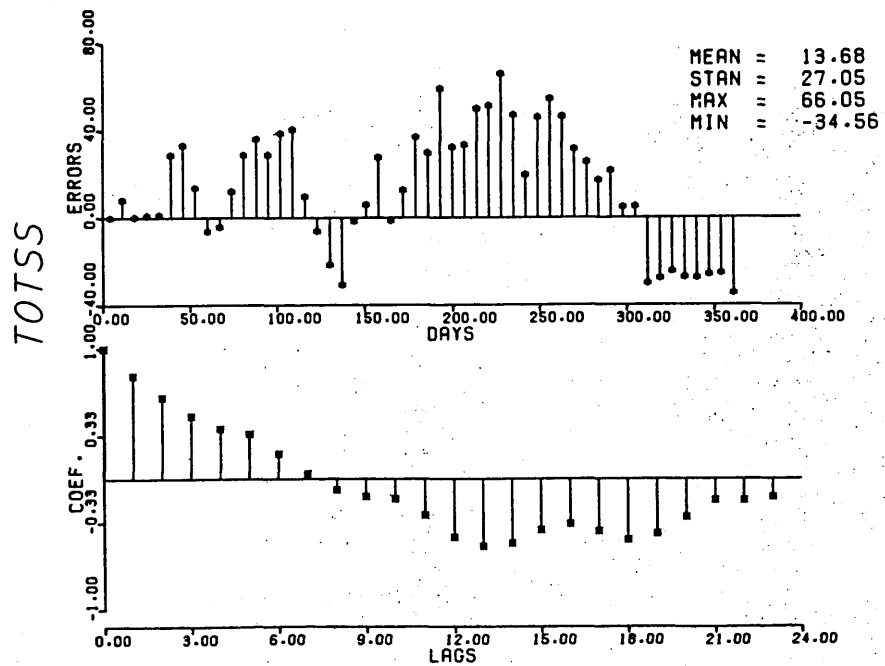


Fig 5.83 Errors Plot for First Order Decay Model Simulation of Pond 2 Effluent Total Solids

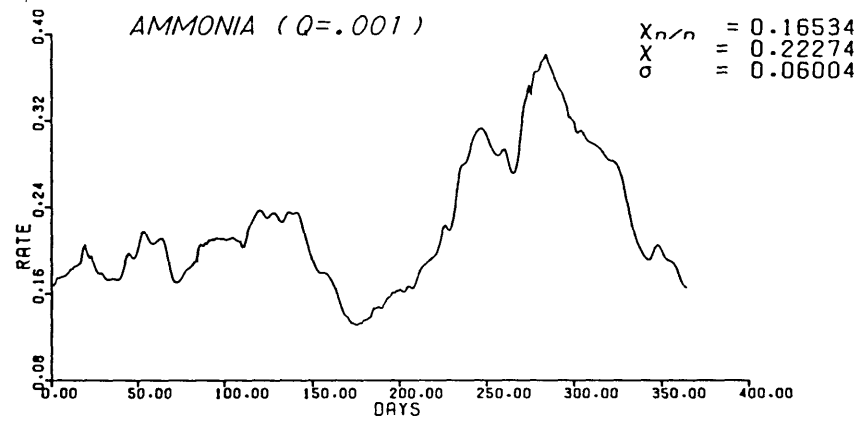


Fig 5.84 Trajectory of IV estimate of Pond 2 Ammonia S/Sink Rate (Q = 0.001)

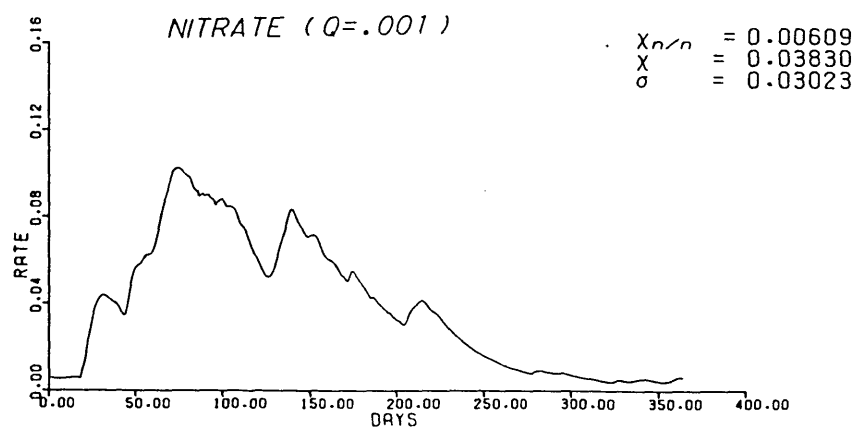


Fig 5.85 Trajectory of IV estimate of Pond 2 Nitrate S/Sink Rate (Q = 0.001)

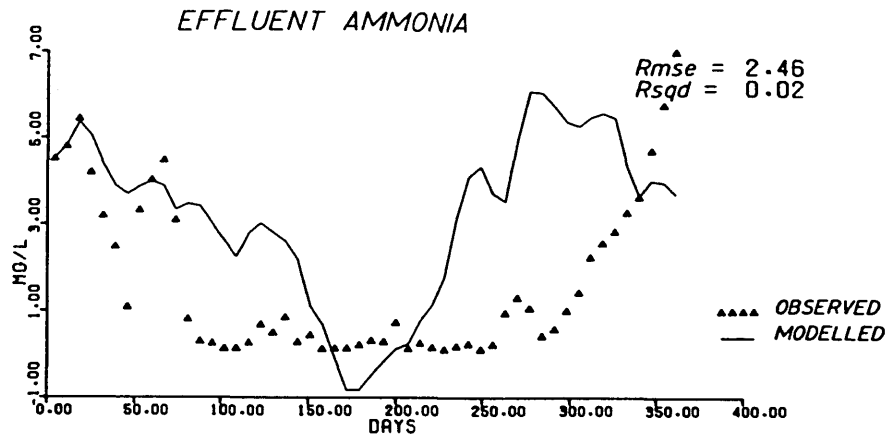


Fig 5.86 Simulation Plot for Pond 2 Effluent Ammonia (S/Sink Model)

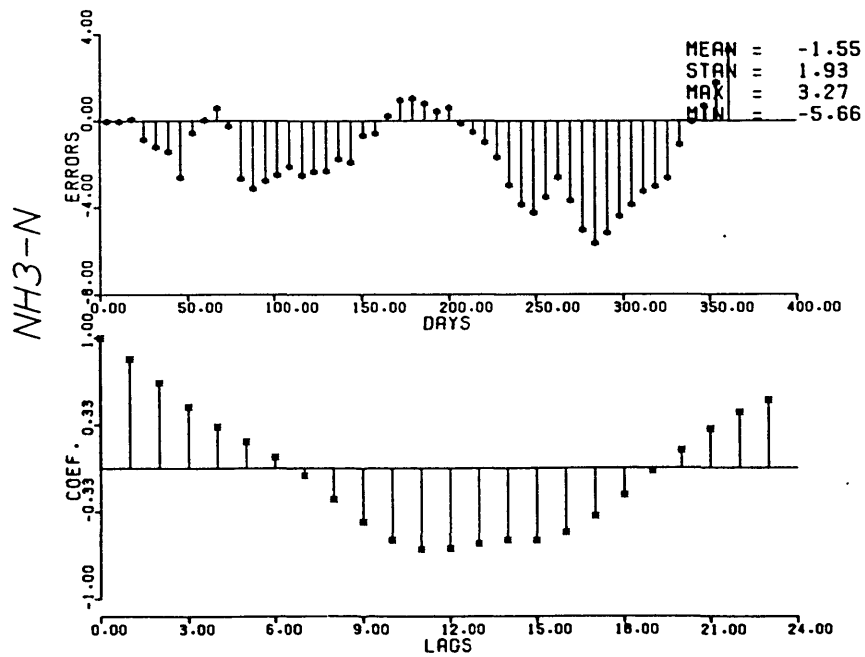


Fig 5.87 Errors Plot for S/Sink Model Simulation of Pond 2 Effluent Ammonia

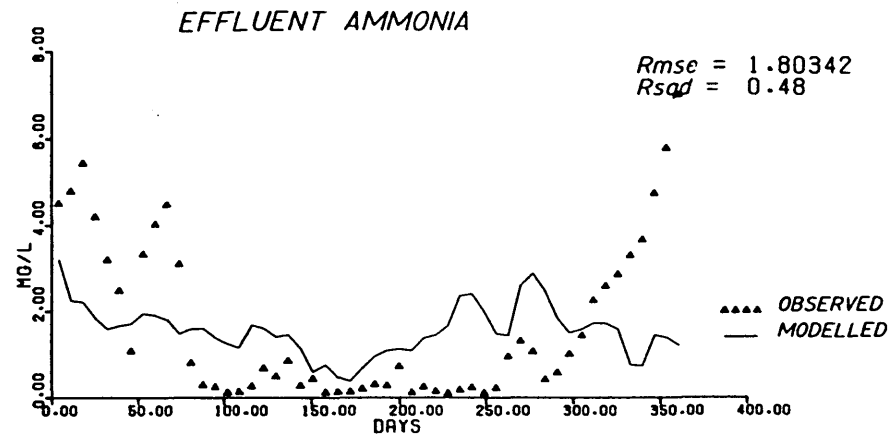


Fig 5.88 Simulation Plot for Pond 2 Effluent Ammonia
 (First Order Decay Model)

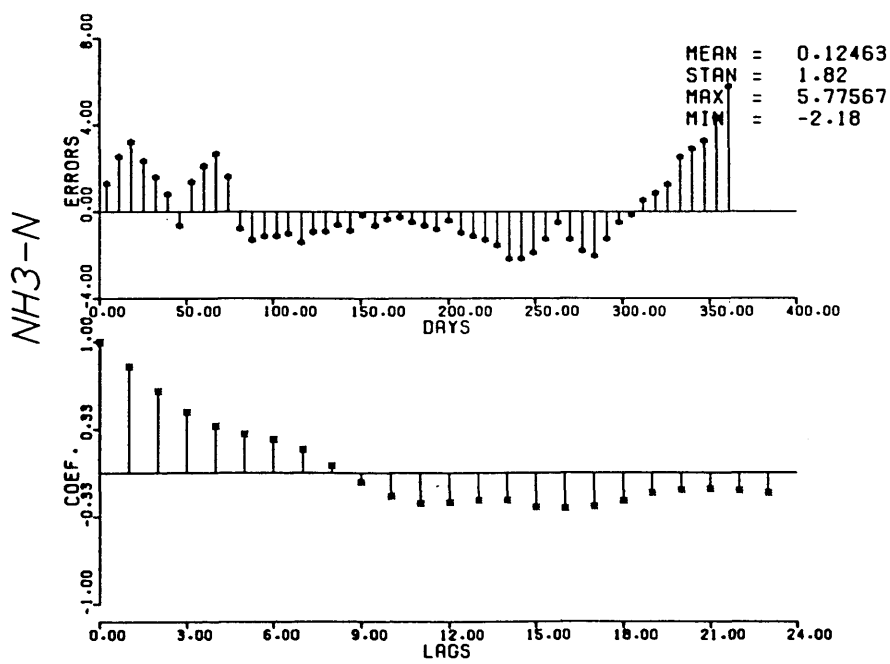


Fig 5.89 Errors Plot for First Order Decay Model Simulation of
 Pond 2 Effluent Ammonia

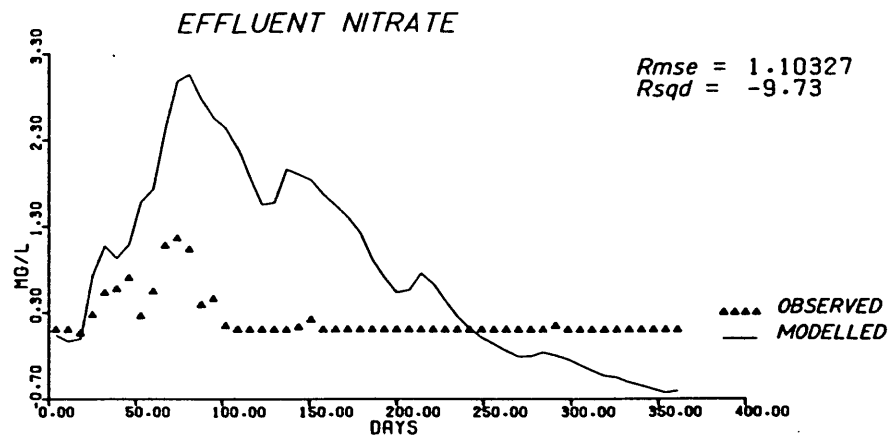


Fig 5.90 Simulation Plot for Pond 2 Effluent Nitrate (S/Sink Model)

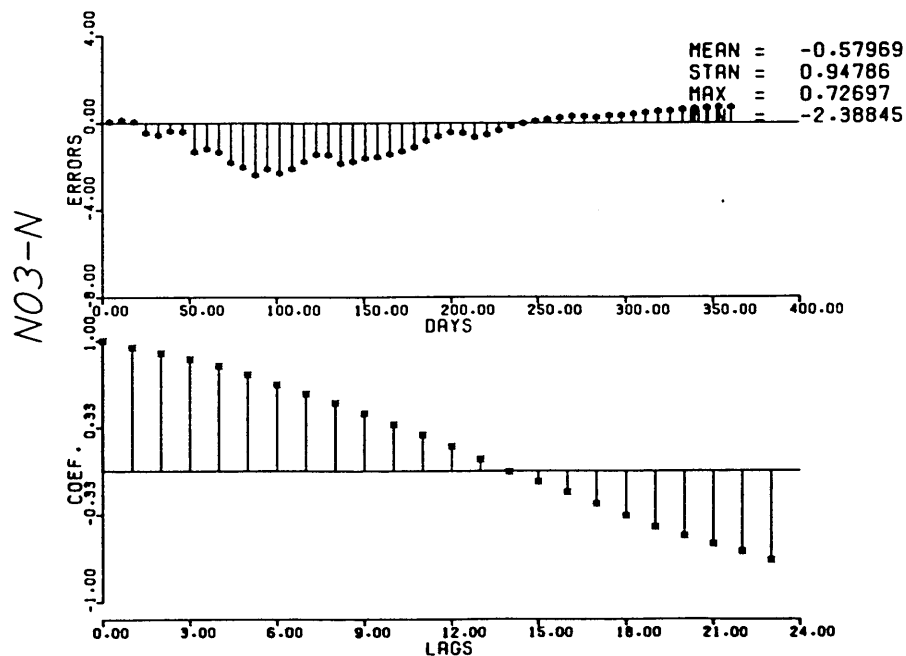


Fig 5.91 Errors Plot for S/Sink Model Simulation of Pond 2 Effluent Nitrate

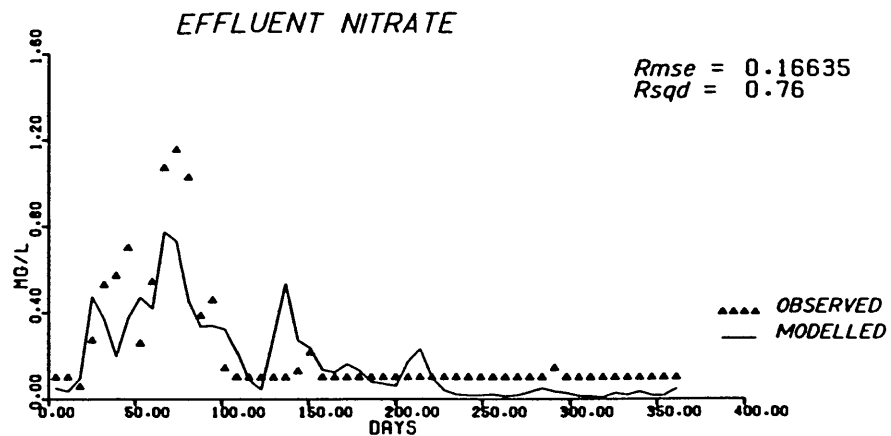


Fig 5.92 Simulation Plot for Pond 2 Effluent Nitrate
(First Order Decay Model)

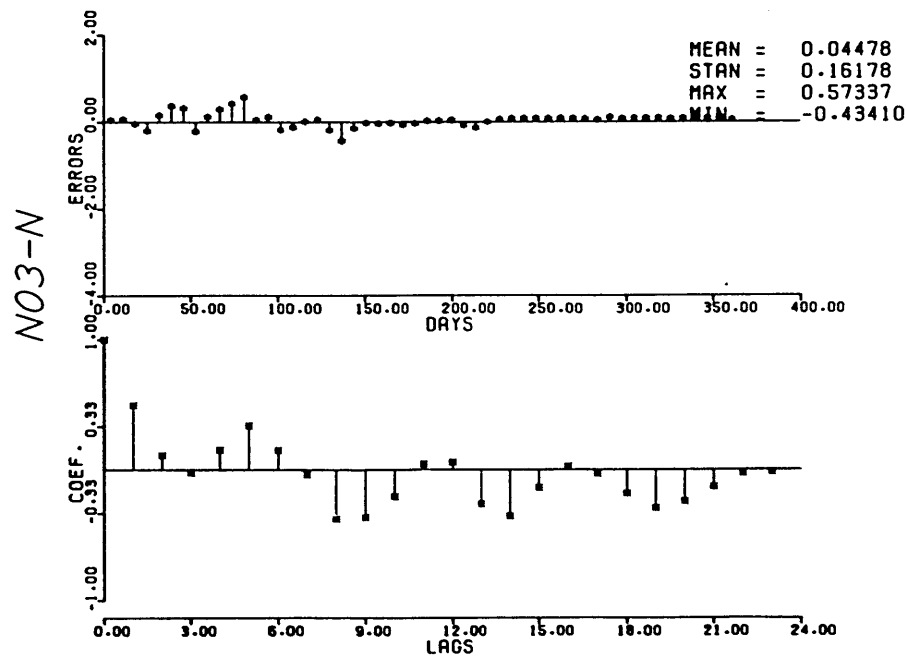


Fig 5.93 Errors Plot for First Order Decay Model Simulation of
Pond 2 Effluent Nitrate

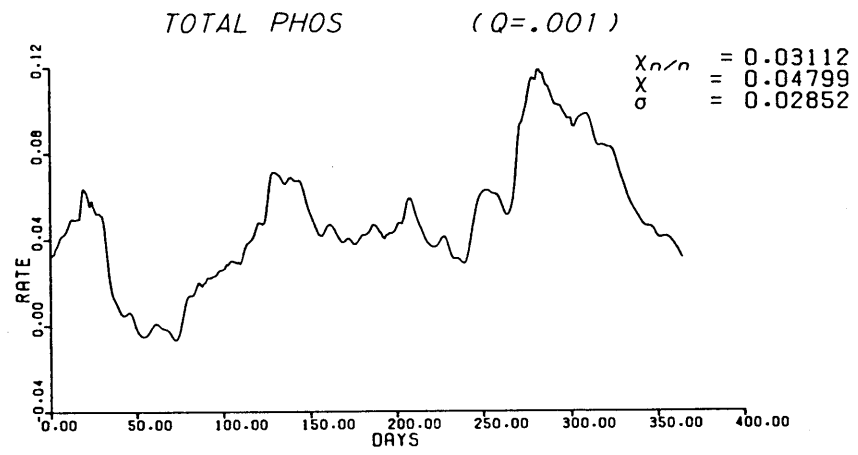


Fig 5.94 Trajectory of IV estimate for Pond 2 Total Phosphorus S/Sink Rate (Q = .001)

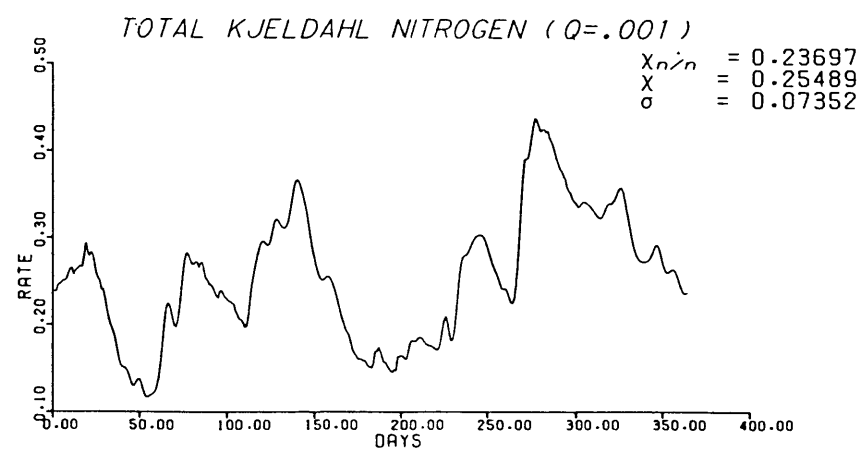


Fig 5.95 Trajectory of IV estimate for Pond 2 Total Kjeldahl Nitrogen S/Sink Rate (Q = .001)

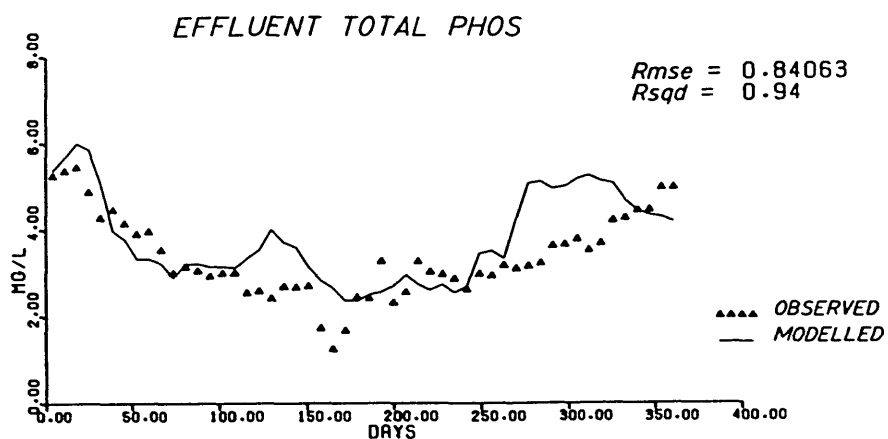


Fig 5.96 Simulation Plot for Pond 2 Effluent Total Phosphorus (S/Sink Model)

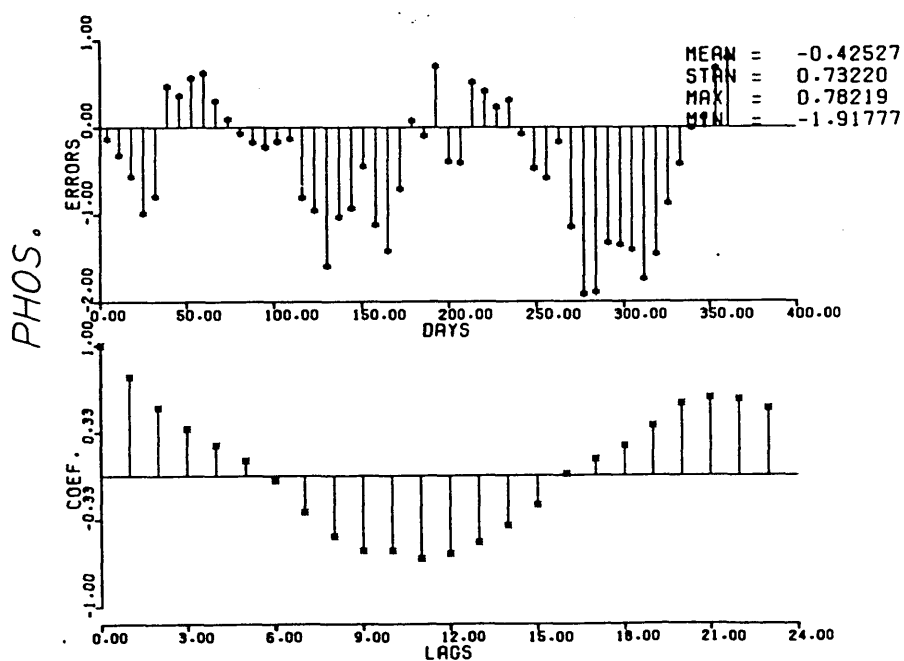


Fig 5.97 Errors Plot for S/Sink Model Simulation of Pond 2 Effluent Total Phosphorus

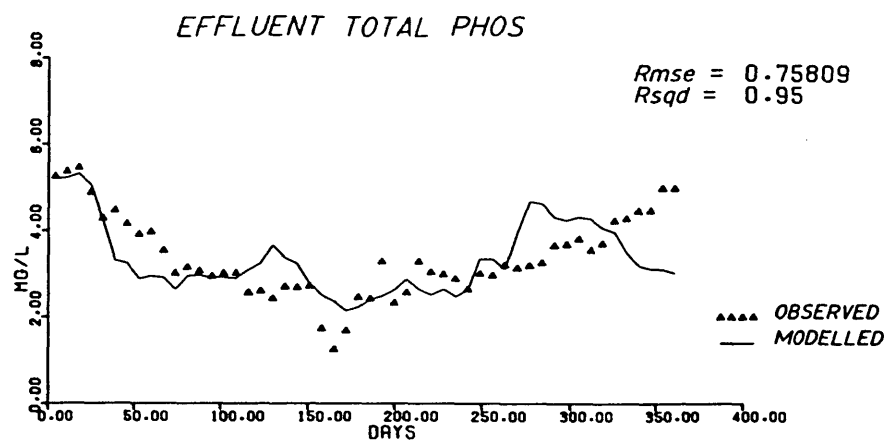


Fig 5.98 Simulation Plot for Pond 2 Effluent Total Phosphorus (First Order Decay Model)

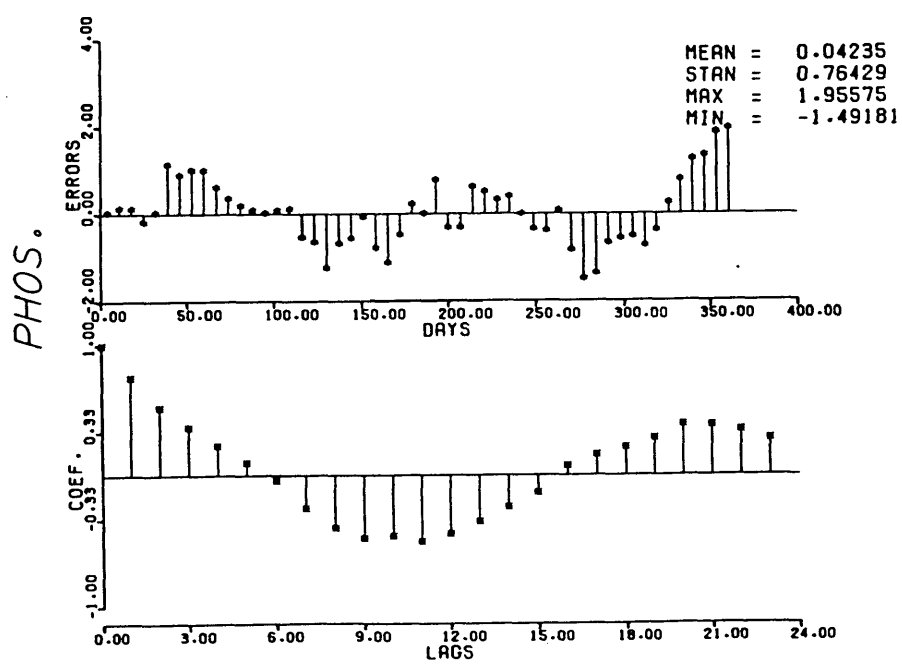


Fig 5.99 Errors Plot for First Order Decay Model Simulation of Pond 2 Effluent Total Phosphorus

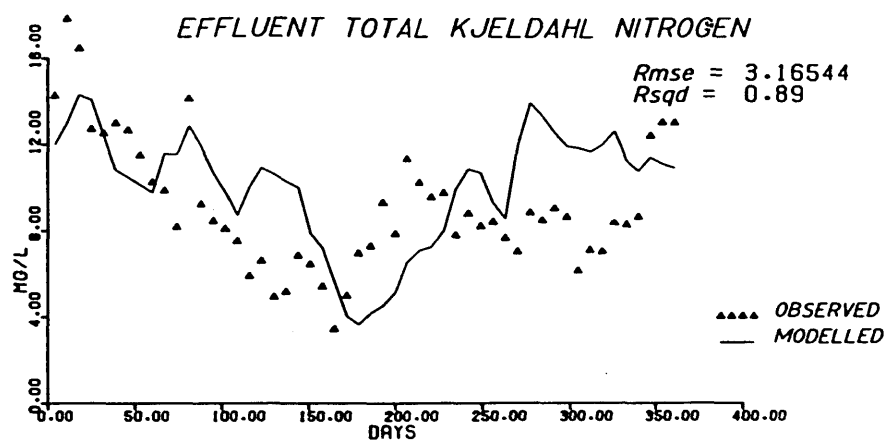


Fig 5.100 Simulation Plot for Pond 2 Effluent Total Kjeldahl Nitrogen (S/Sink Model)

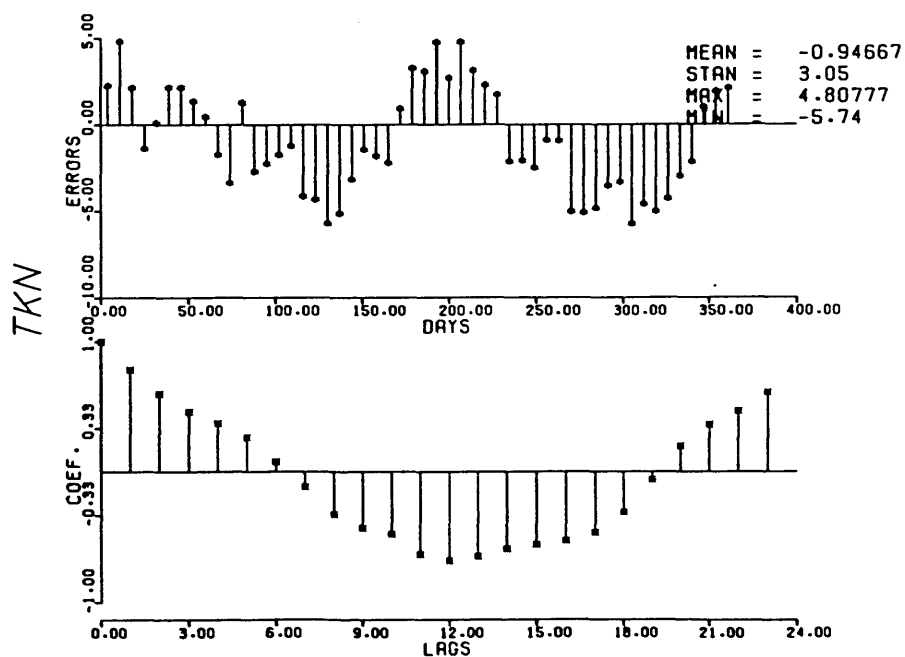


Fig 5.101 Errors Plot for S/Sink Model Simulation of Pond 2 Effluent Total Kjeldahl Nitrogen

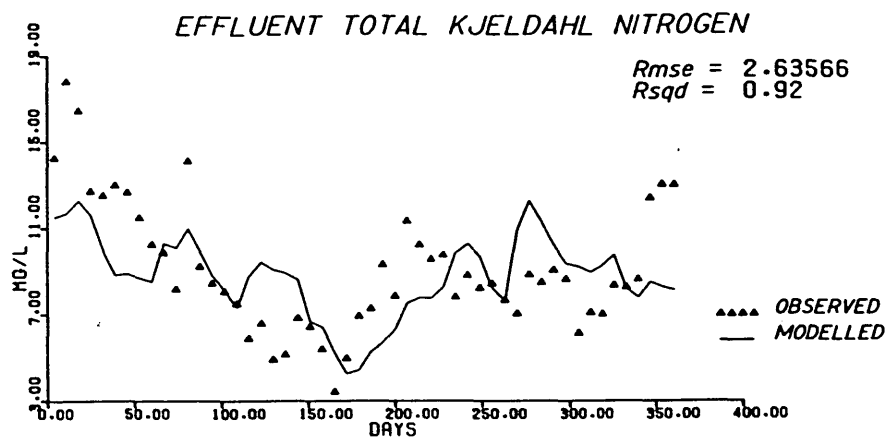


Fig 5.102 Simulation Plot for Pond 2 Effluent Total Kjeldahl Nitrogen (First Order Decay)

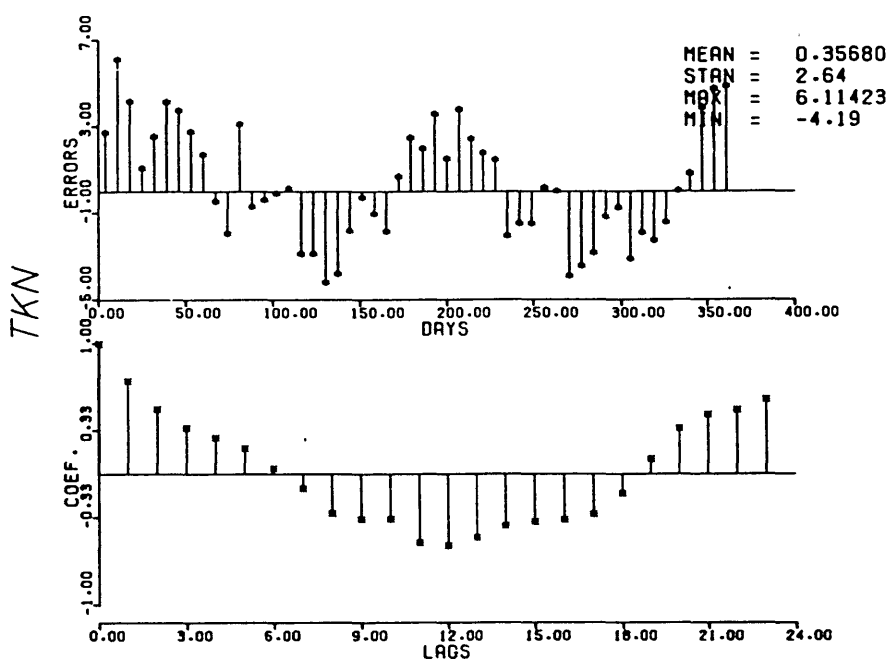


Fig 5.103 Errors Plot for First Order Decay Model Simulation of Pond 2 Effluent Total Kjeldahl Nitrogen

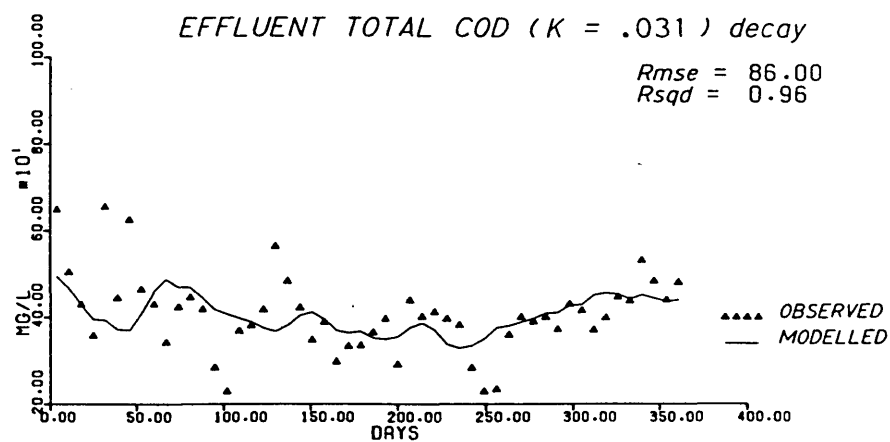


Fig 5.104 Simulation Plot for Pond 1 Effluent Total COD (First Order Decay)

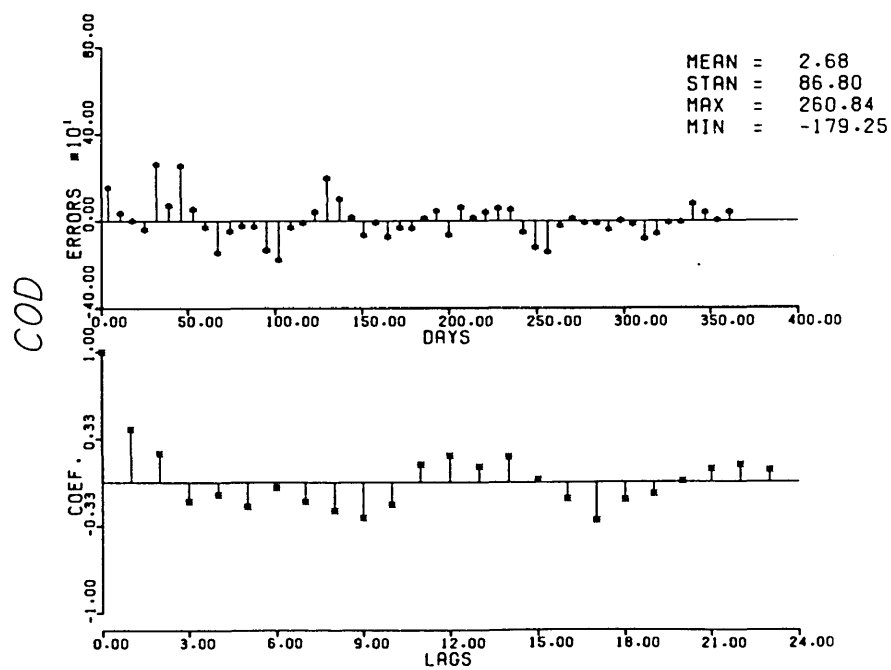


Fig 5.105 Errors Plot for First Order Decay Model Simulation of Pond 1 Effluent Total COD

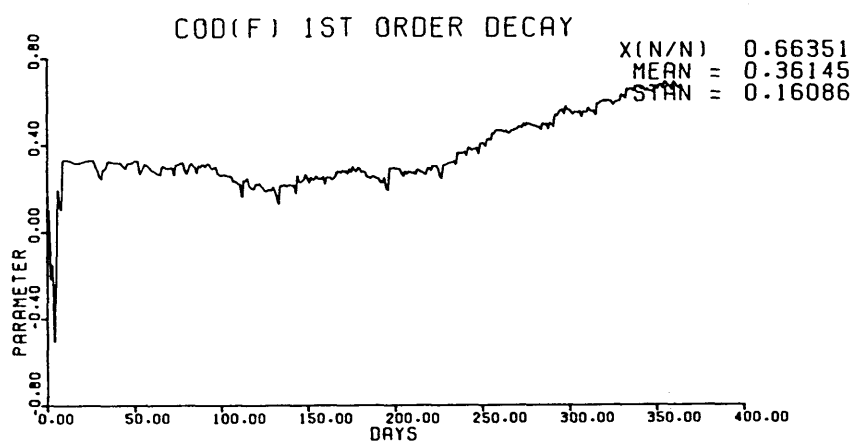


Fig 5.106 Trajectory of EKF estimate of First Order Decay Rate:
Filtered COD for Carbonaceous Organics Working Model 1

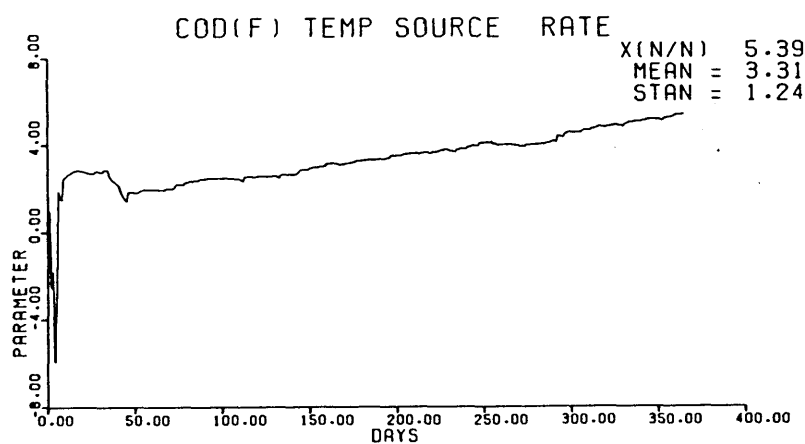


Fig 5.107 Trajectory of EKF estimate of coefficient for Temperature
dependent source rate: Filtered COD for Carbonaceous
Organics Working Model 1

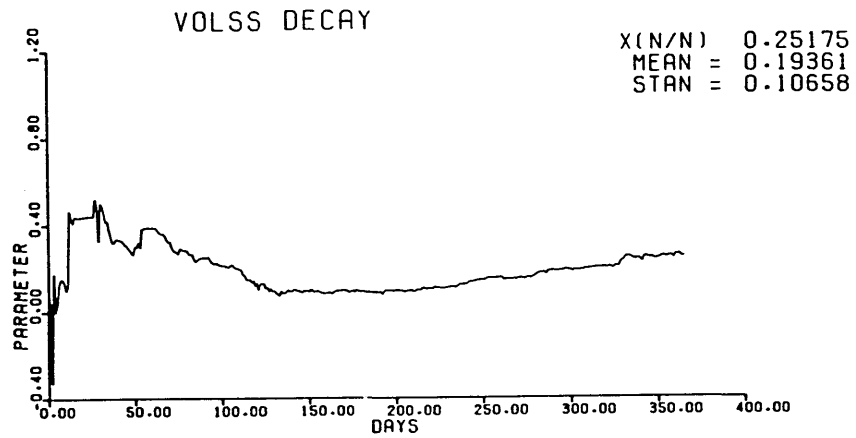


Fig 5.108 Trajectory of EKF estimate of First Order Decay Rate: Volatile Suspended Solids for Carbonaceous Organics Working Model 1

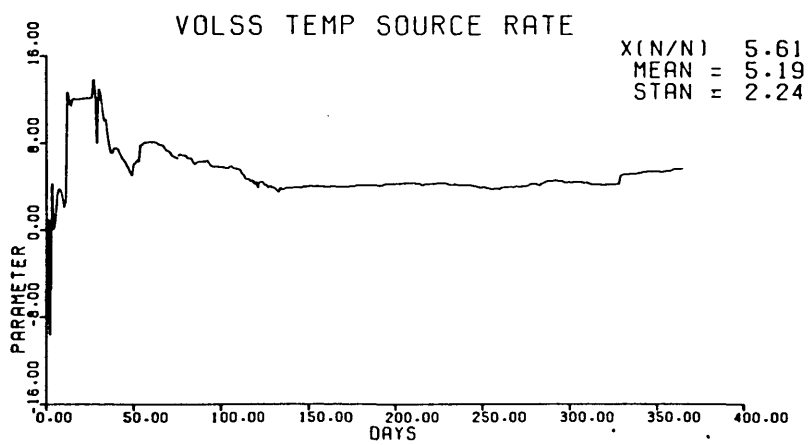


Fig 5.109 Trajectory of EKF estimate of Coefficient for Temperature dependent source rate: Volatile Suspended Solids for Carbonaceous Organics Working Model 1

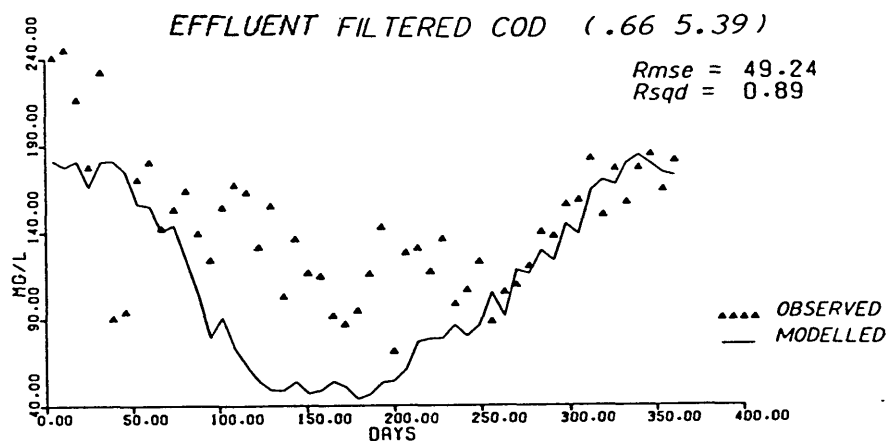


Fig 5.110 Simulation Plot for Pond 1 Effluent Filtered COD
(Carbonaceous Organics Working Model 1)

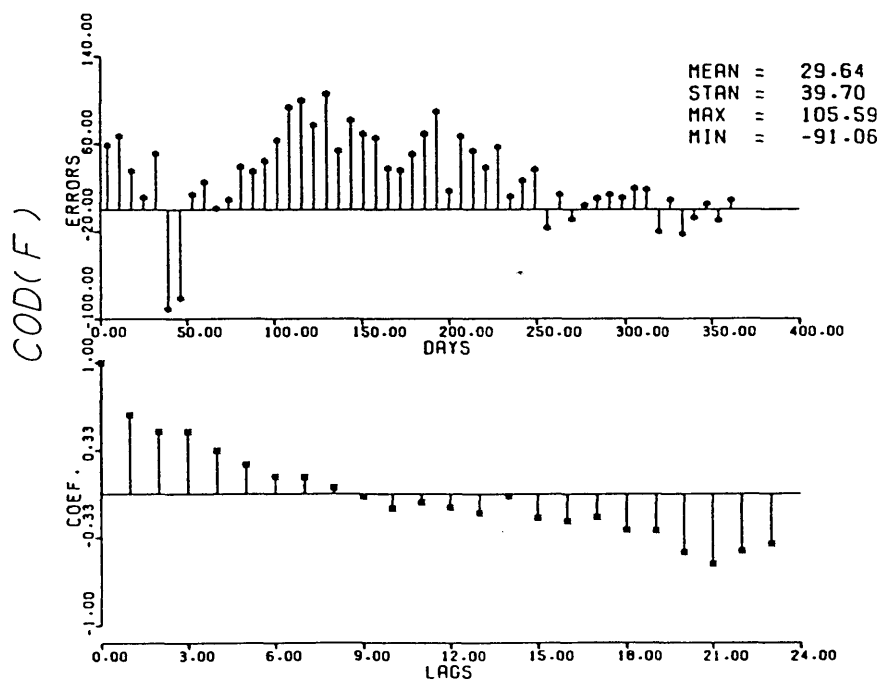


Fig 5.111 Errors Plot for Carbonaceous Organics Working Model 1
Simulation of Pond 1 Effluent Filtered COD

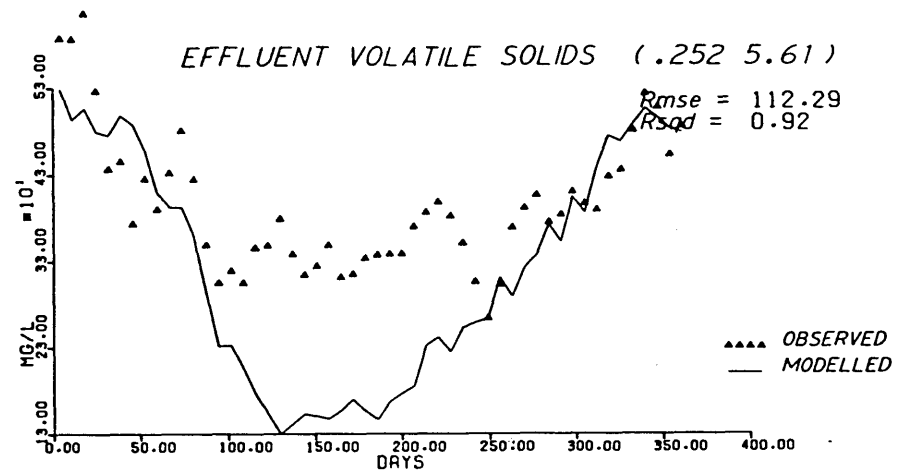


Fig 5.112 Simulation Plot for Pond 1 Effluent Volatile Suspended Solids (Carbonaceous Organics Working Model 1)

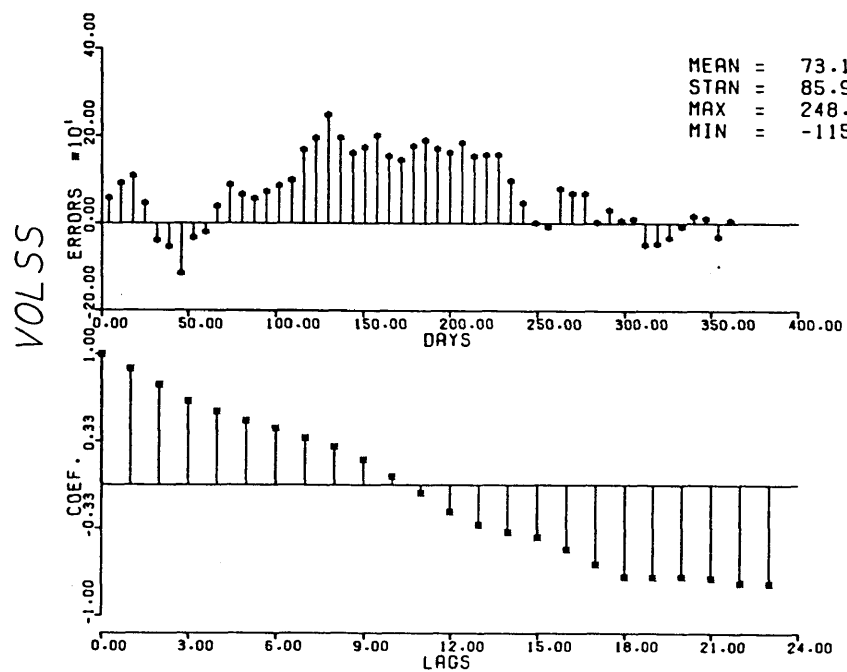


Fig 5.113 Errors Plot for Carbonaceous Organics Working Model 1 Simulation of Pond 1 Effluent Volatile Suspended Solids

TABLE 5.6:
Block Estimates of Carbonaceous Organic Working Model 2 Parameters

	PARAMETER VALUES			
MODELLED STATE	" θ_1 " (d ⁻¹)	" θ_2 "	" θ_3 " (d ⁻¹)	" θ_4 "
Pond 1 Soluble Organics (COD(F))	0.086	1.014	0.057	1.123
Pond 1 Particulate Organics (VOLSS)	0.082	1.037	0.068	1.093
Pond 2 Soluble Organics (SCOD)	0.053	1.057	0.045	1.140
Pond 2 Particulate Organics (TOTSS)	0.100	1.013	0.104	1.029

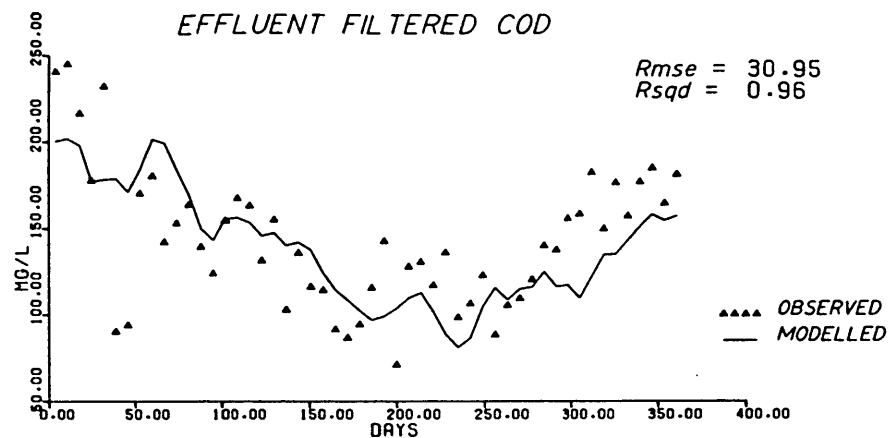


Fig 5.114 Simulation Plot for Pond 1 Effluent Filtered COD (Carbonaceous Organics Working Model 2)

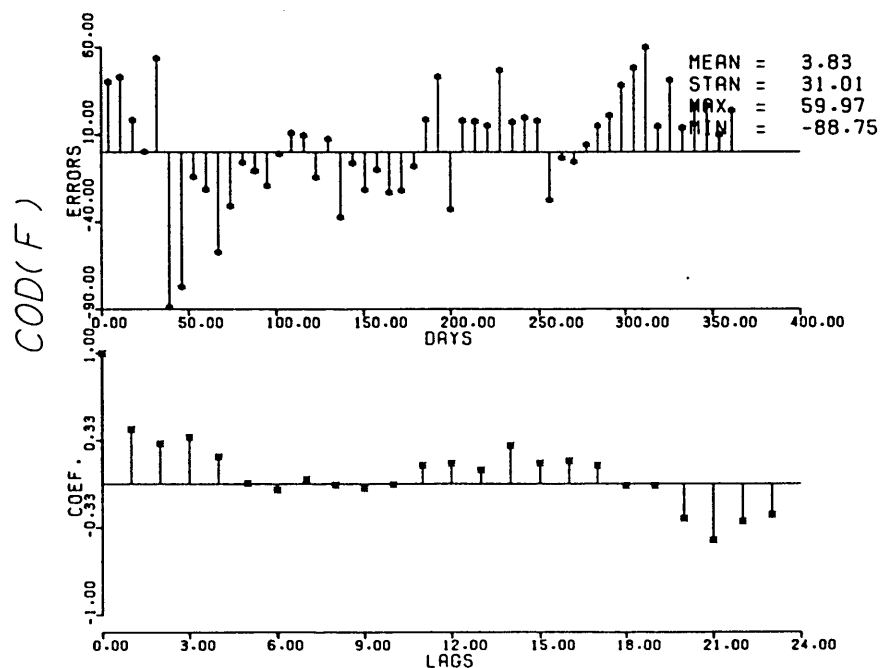


Fig 5.115 Errors Plot for Carbonaceous Organics Working Model 2 Simulation of Pond 1 Effluent Filtered COD

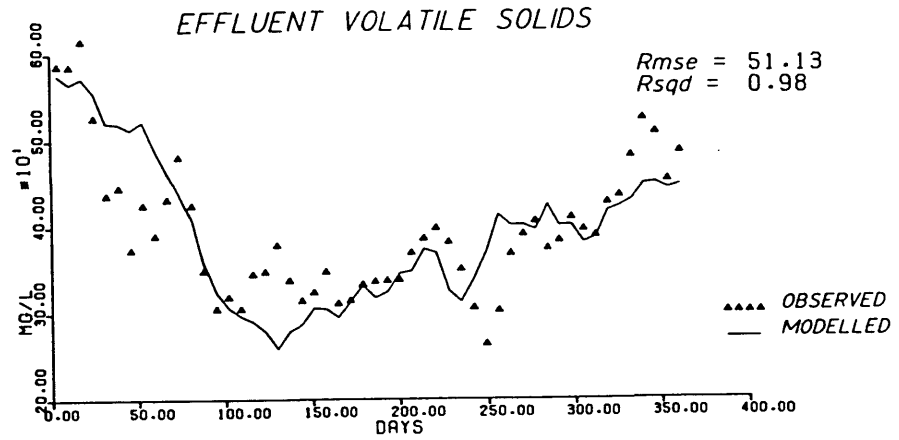


Fig 5.116 Simulation Plot for Pond 1 Effluent Volatile Sus. Solids (Carbonaceous Organics Working Model 2)

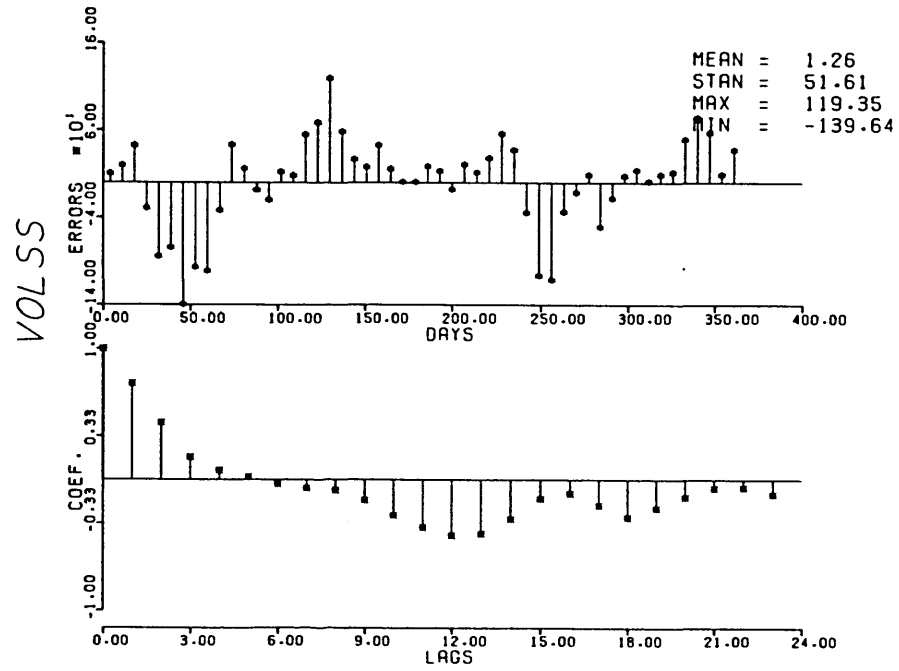


Fig 5.117 Errors Plot for Carbonaceous Organics Working Model 2 Simulation of Pond 1 Effluent Volatile Sus. Solids

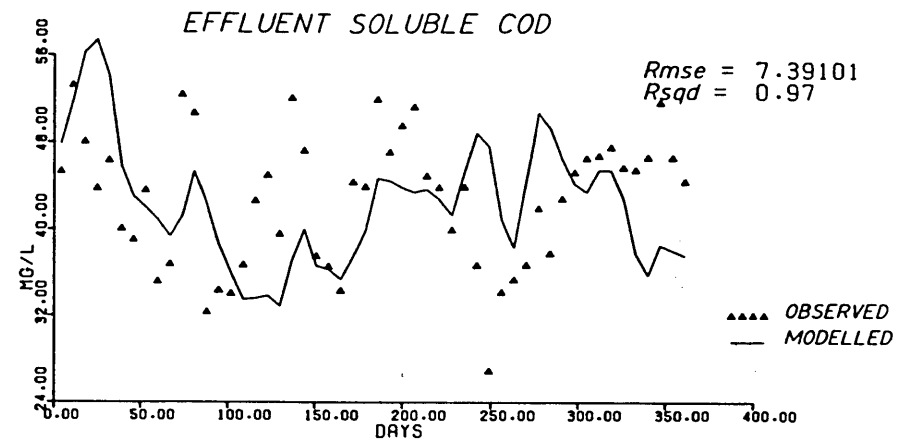


Fig 5.118 Simulation Plot for Pond 2 Effluent Soluble COD (Carbonaceous Organics Working Model 2)

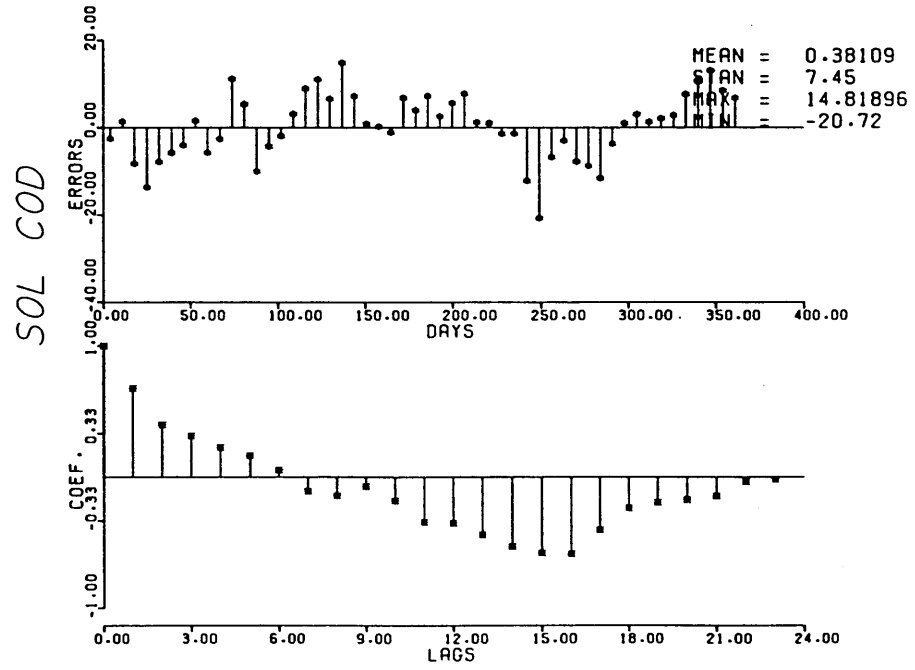


Fig 5.119 Errors Plot for Carbonaceous Organics Working Model 2 Simulation of Pond 2 Effluent Soluble COD

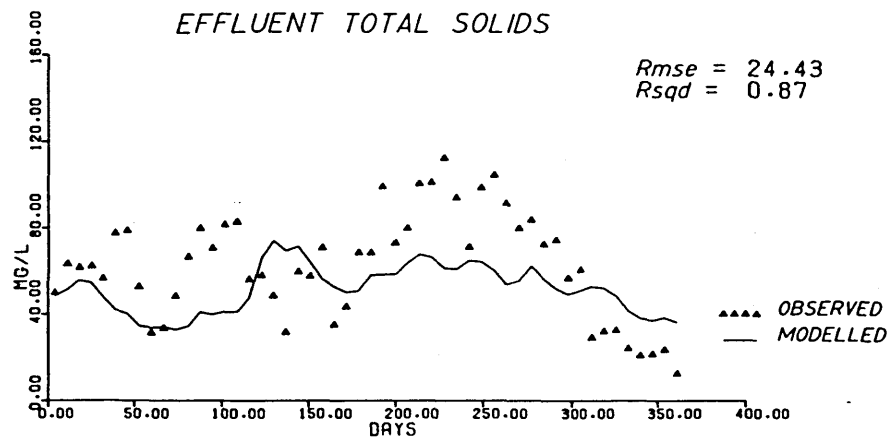


Fig 5.120 Simulation Plot for Pond 2 Effluent Total Solids (Carbonaceous Organics Working Model 2)

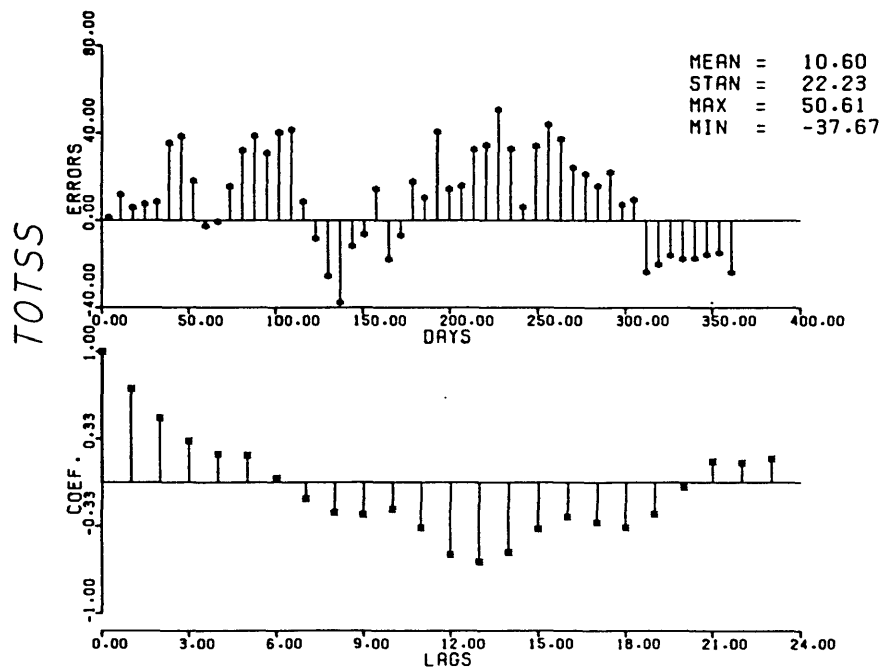


Fig 5.121 Errors Plot for Carbonaceous Organics Working Model 2 Simulation of Pond 2 Effluent Total Solids

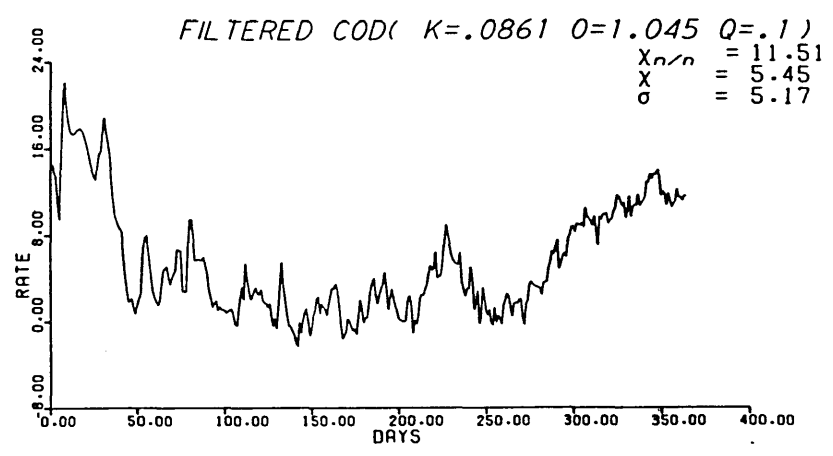


Fig 5.122 Trajectory of IV estimate of S/Sink Rate for Pond 1
Filtered COD (Carbonaceous Organics Working Model 3)

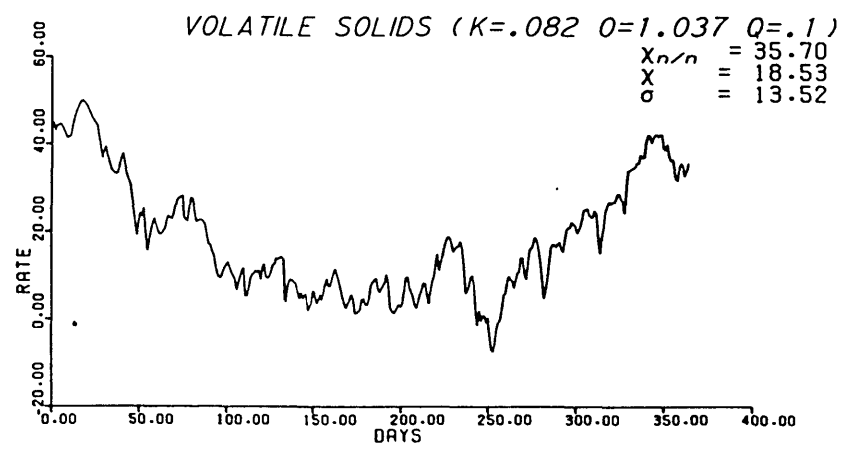


Fig 5.123 Trajectory of IV estimate of S/Sink Rate for Pond 1
Volatile Suspended Solids (Carbonaceous Organics Working
Model 3)

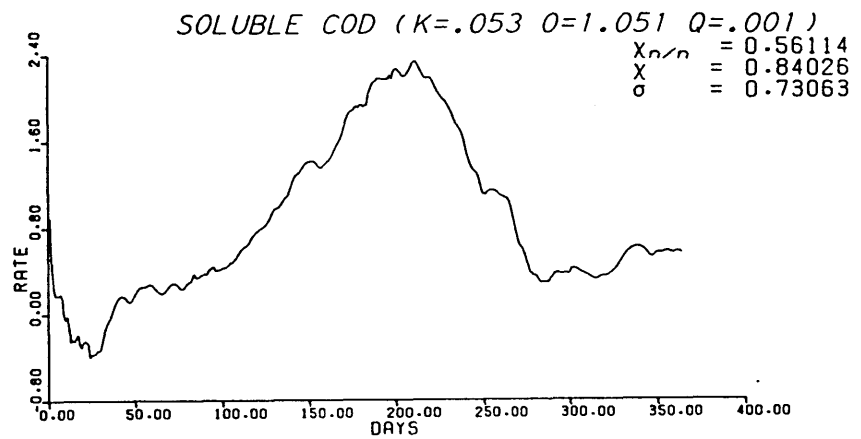


Fig 5.124 Trajectory of IV estimate of S/Sink Rate for Pond 2
Soluble COD (Carbonaceous Organics Working Model 3)

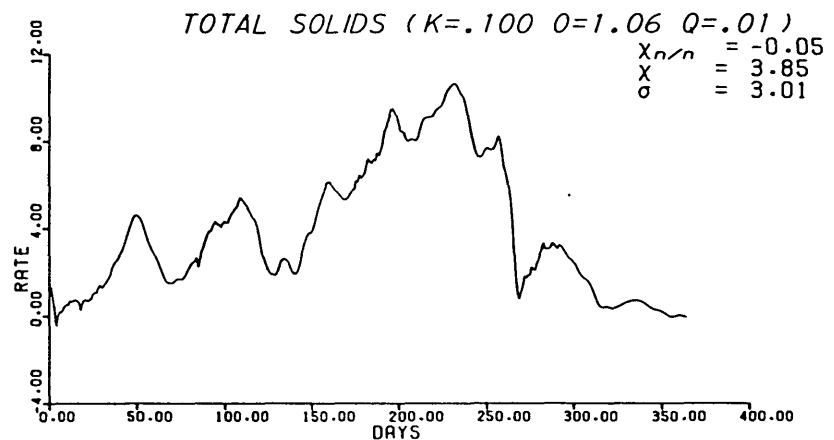


Fig 5.125 Trajectory of IV estimate of S/Sink Rate for Pond 2
Total Solids (Carbonaceous Organics Working Model 3)

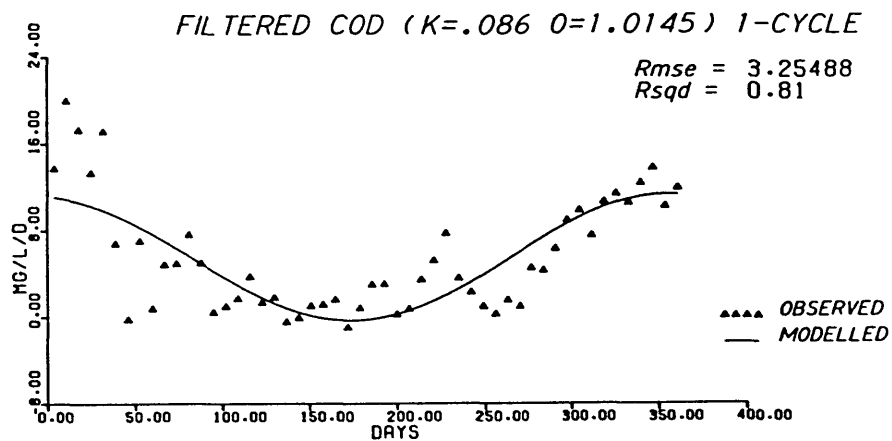


Fig 5.126 1-Cycle fit to Pond 1 Filtered COD lumped S/Sink term in Carbonaceous Organics Working Model 3

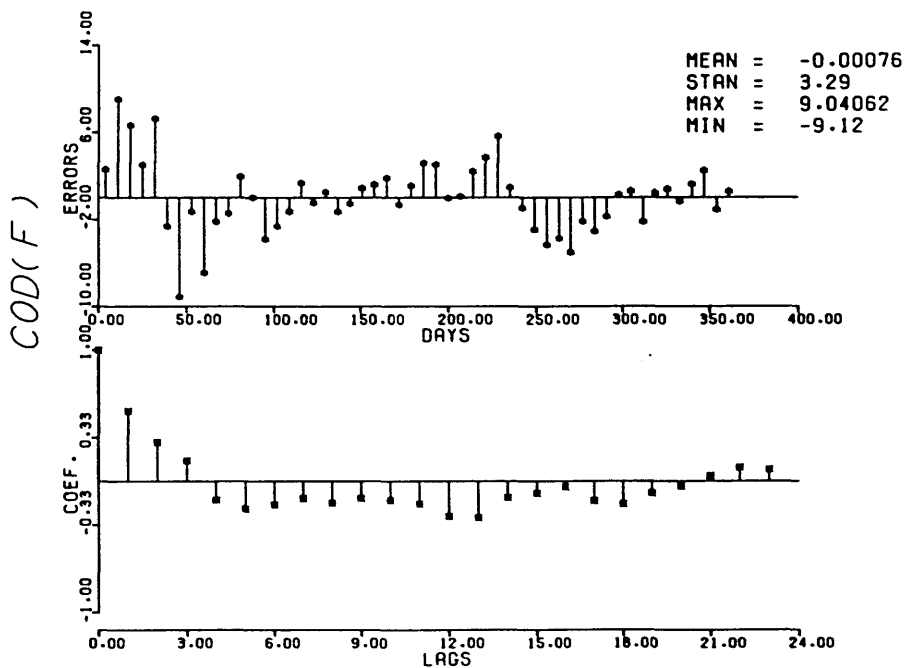


Fig 5.127 Errors Plot for 1-Cycle fit to Pond 1 Filtered COD lumped S/Sink term in Carbonaceous Organics Working Model 3

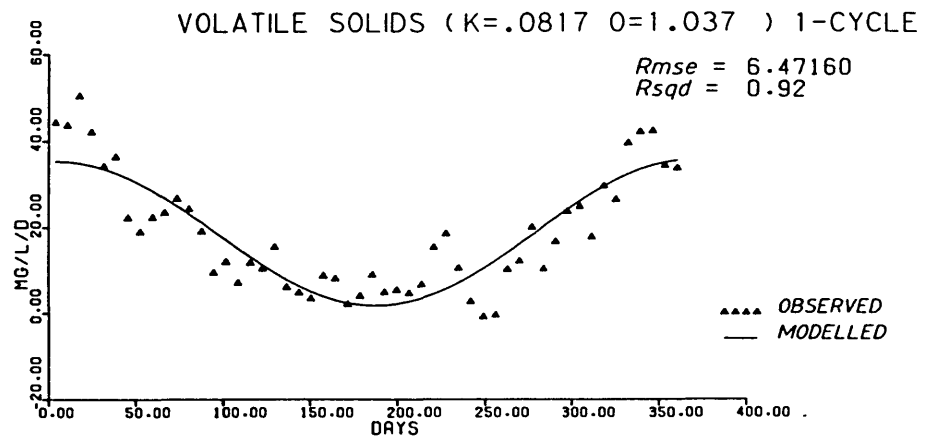


Fig 5.128 1-Cycle fit to Pond 1 Volatile Suspended Solids lumped S/Sink term in Carbonaceous Organics Working Model 3

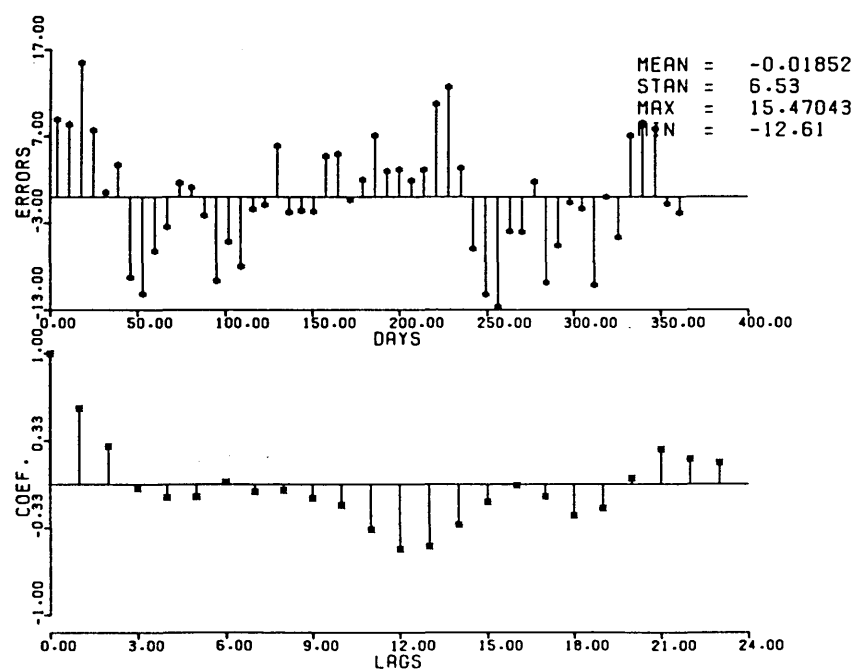


Fig 5.129 Errors Plot for 1-Cycle fit to Pond 1 Volatile Suspended Solids lumped S/Sink term in Carbonaceous Organics Working Model 3

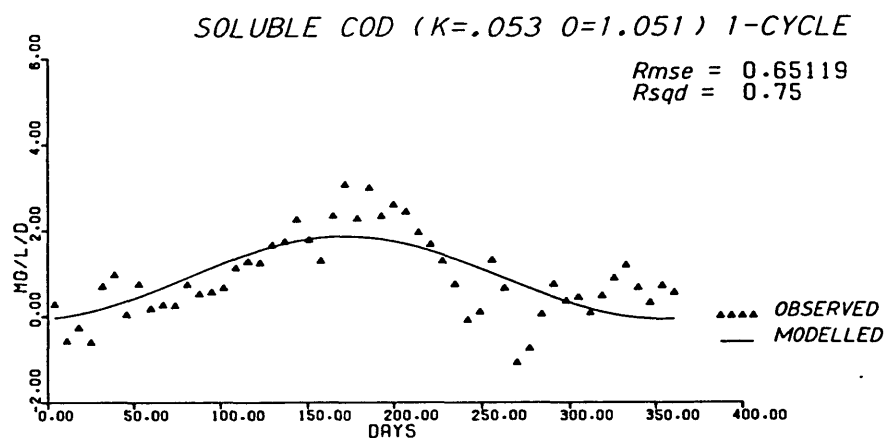


Fig 5.130 1-Cycle fit to Pond 2 Soluble COD lumped S/Sink term in Carbonaceous Organics Working Model 3

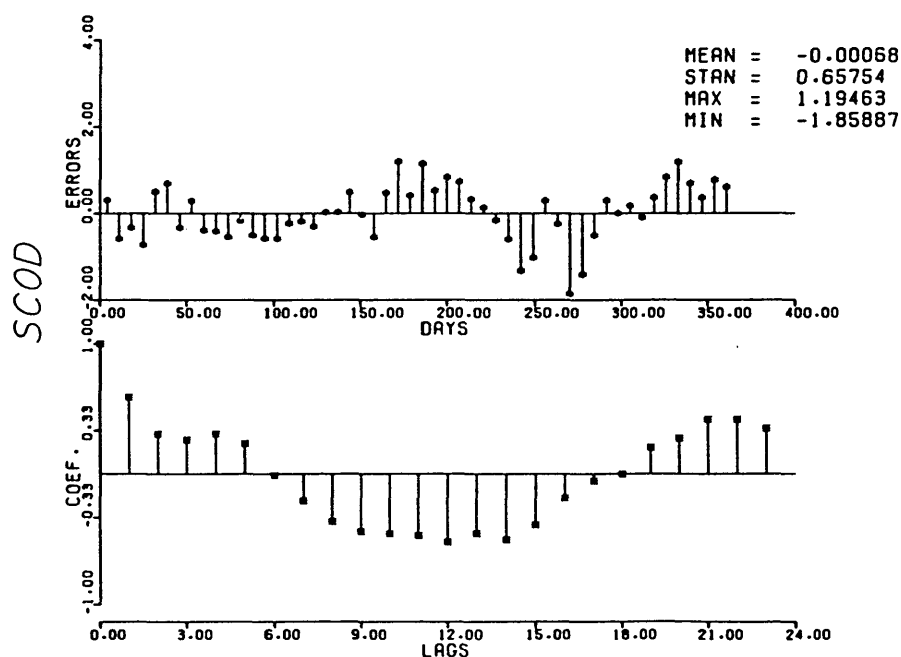


Fig 5.131 Errors Plot for 1-Cycle fit to Pond 2 Soluble COD lumped S/Sink term in Carbonaceous Organics Working Model 3

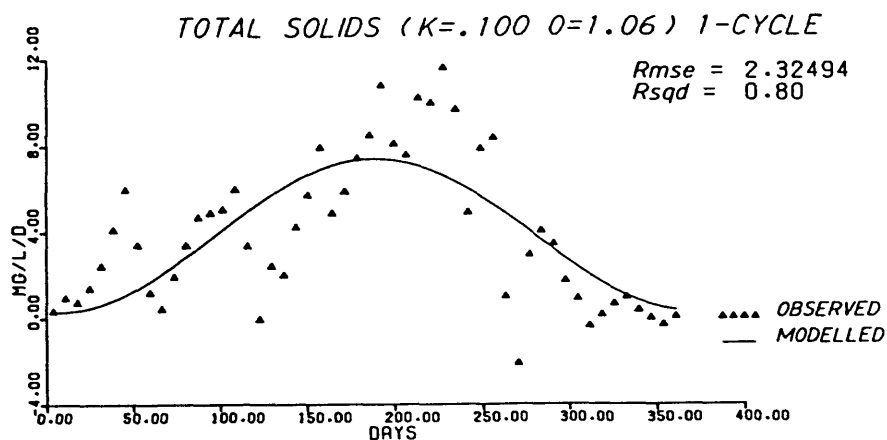


Fig 5.132 1-Cycle fit to Pond 2 Total Solids lumped S/Sink term in Carbonaceous Organics Working Model 3

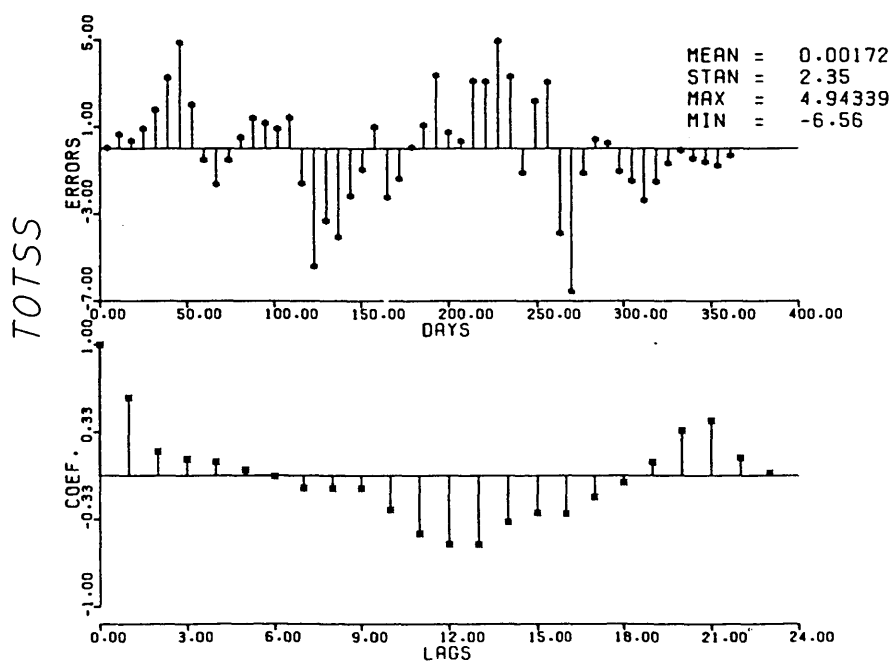


Fig 5.133 Errors plot for 1-Cycle fit to Pond 2 Total Solids lumped S/Sink term in Carbonaceous Organics Working Model 3

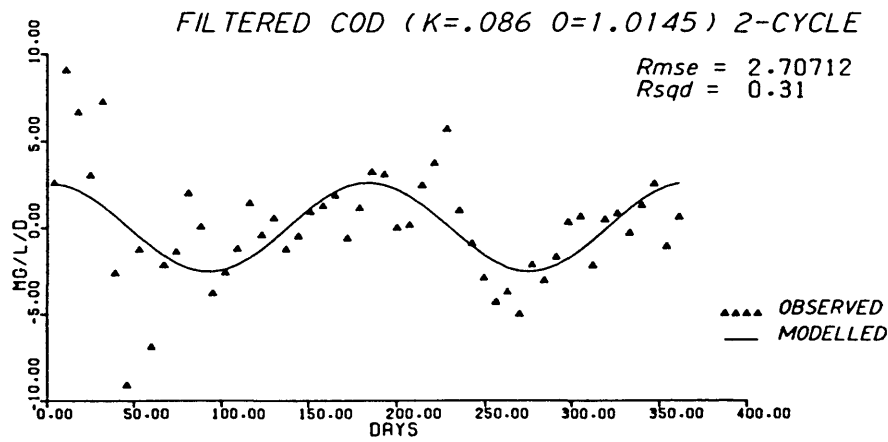


Fig 5.134 2-Cycle fit to Residuals of 1-Cycle fit to Pond 1 Filtered COD lumped S/Sink term in Carbonaceous Organics Working Model 3

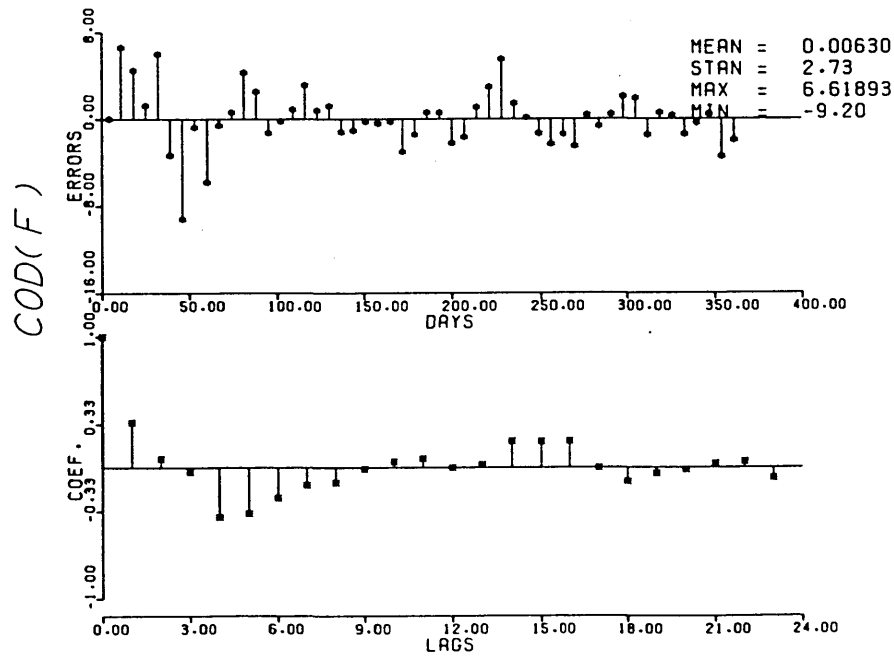


Fig 5.135 Errors Plot for 2-Cycle fit to Residuals of 1-Cycle fit to Pond 1 Filtered COD lumped S/Sink term in Carbonaceous Organics Working Model 3

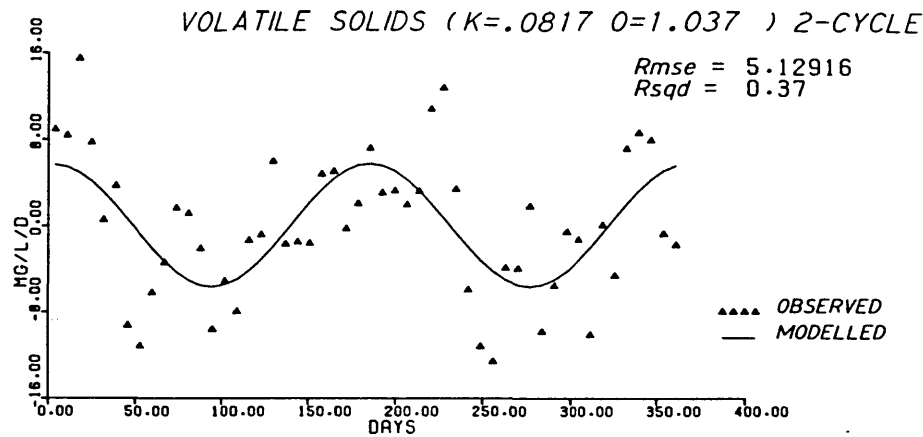


Fig 5.136 2-Cycle fit to Residuals of 1-Cycle fit to Pond 1 Volatile Solids lumped S/Sink term in Carbonaceous Organics Working Model 3

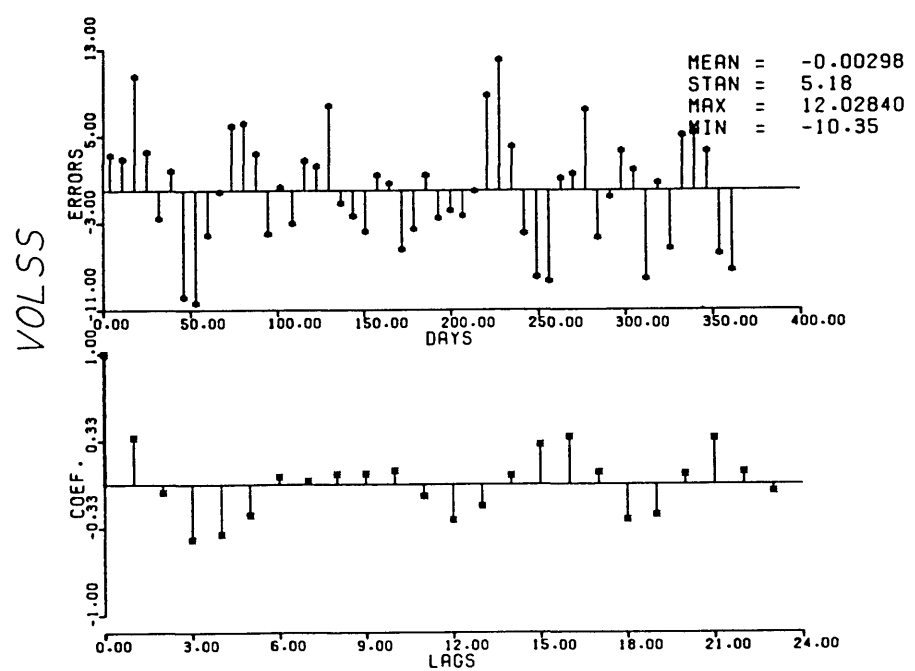


Fig 5.137 Errors Plot for 2-Cycle fit to Residuals of 1-Cycle fit to Pond 1 Volatile Solids lumped S/Sink term in Carbonaceous Organics Working Model 3

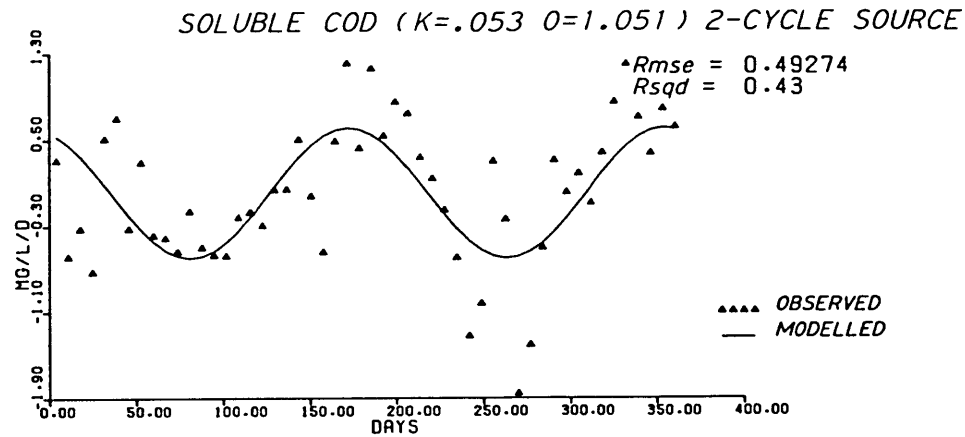


Fig 5.138 2-Cycle fit to Residuals of 1-Cycle fit to Pond 2
Soluble COD lumped S/Sink term in Carbonaceous Organics
Working Model 3

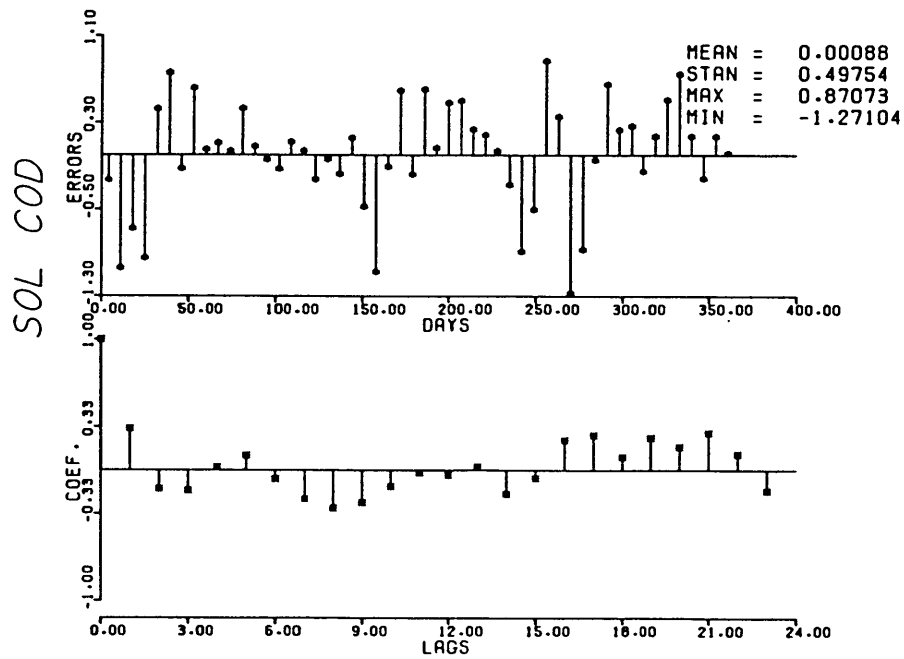


Fig 5.139 Errors Plot for 2-Cycle fit to Residuals of 1-Cycle fit
to Pond 2 Soluble COD lumped S/Sink term in Carbonaceous
Organics Working Model 3

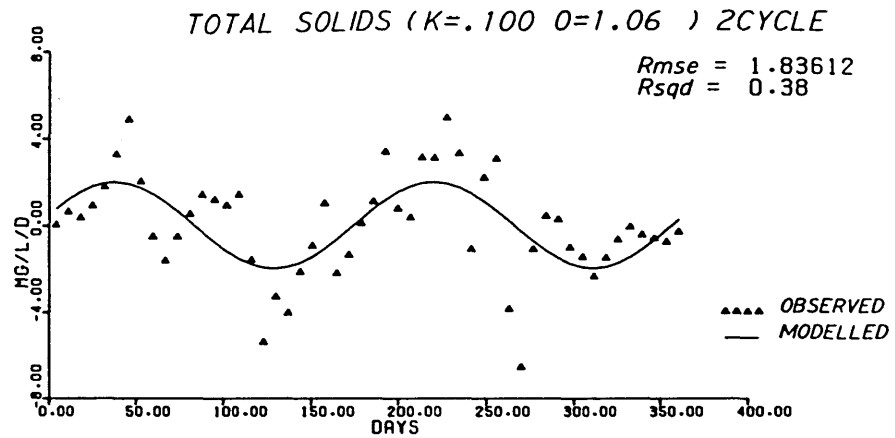


Fig 5.140 2-Cycle fit to Residuals of 1-Cycle fit to Pond 2 Total Solids lumped S/Sink term in Carbonaceous Organics Working Model 3

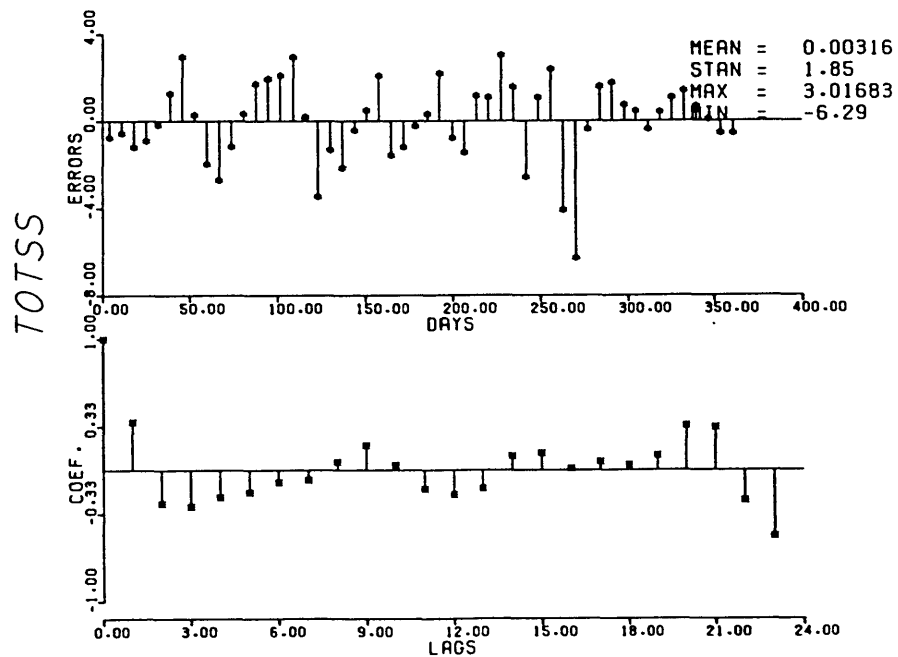


Fig 5.141 Errors Plot for 2-Cycle fit to Residuals of 1-Cycle fit to Pond 2 Total Solids lumped S/Sink term in Carbonaceous Organics Working Model 3

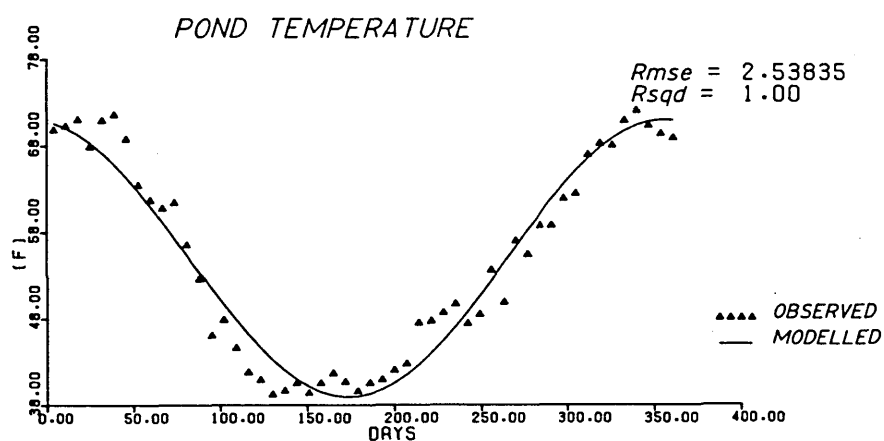


Fig 5.142 1-Cycle fit to Pond 1 Pond Temperature

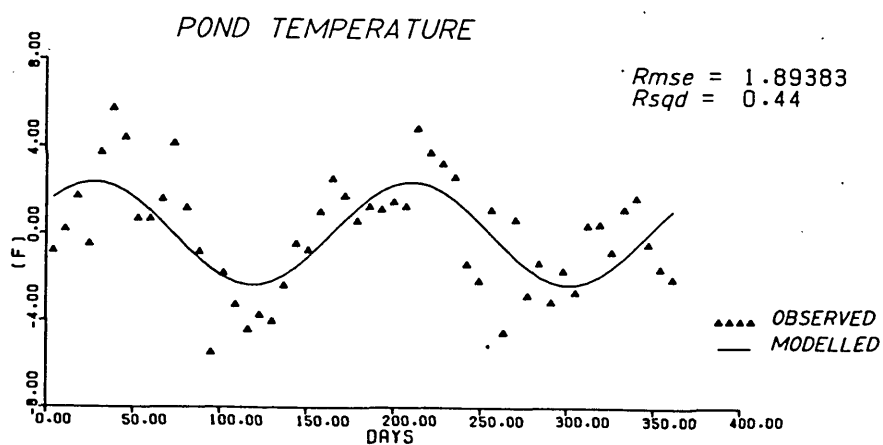


Fig 5.143 2-Cycle fit to Residuals of 1-Cycle fit to Pond 1 Pond Temperature

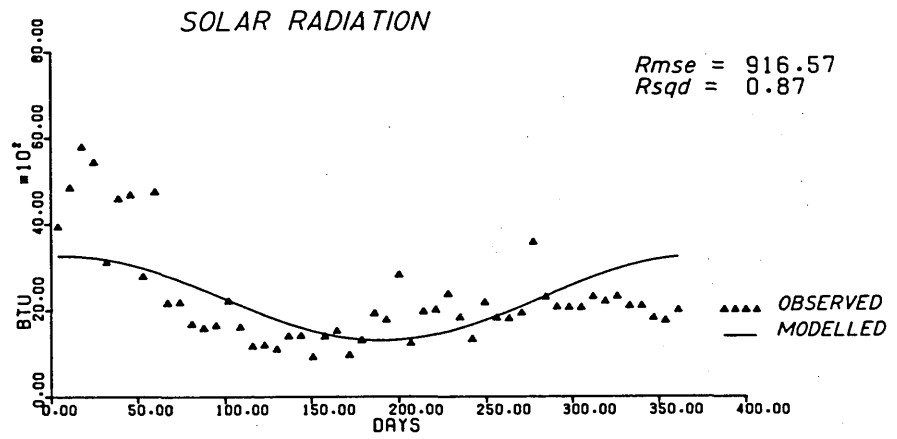


Fig 5.144 1-Cycle fit to Pond 1 Solar Radiation

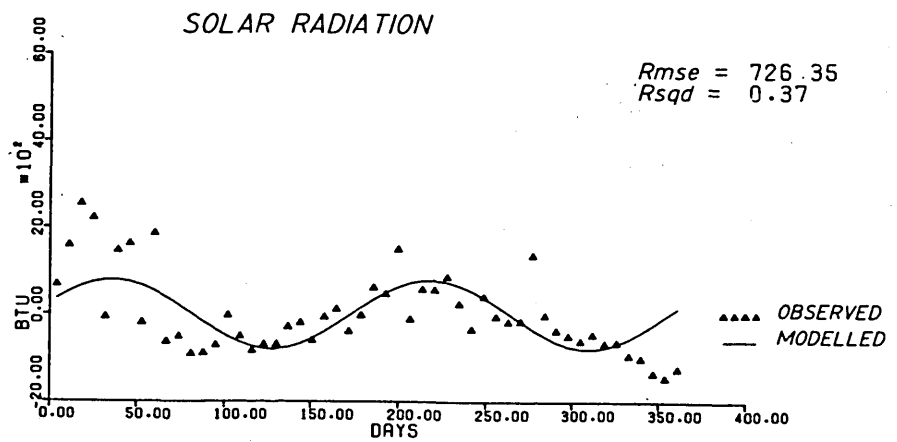


Fig 5.145 2-Cycle fit to Residuals of 1-Cycle fit to Pond 1 Solar Radiation

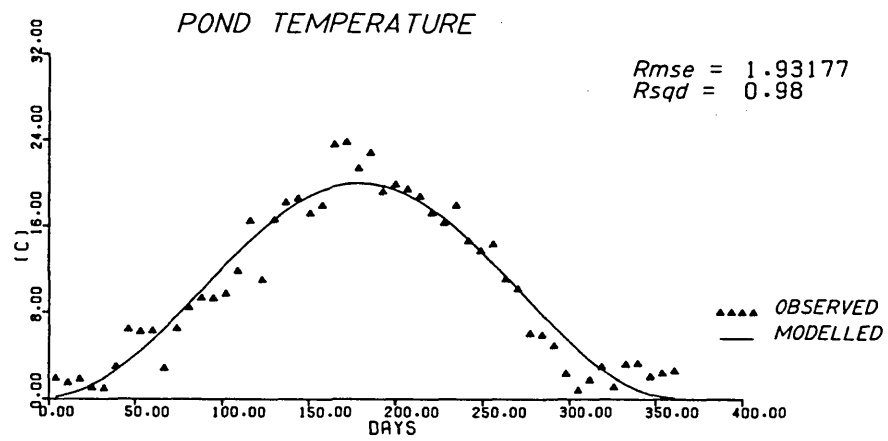


Fig 5.146 1-Cycle fit to Pond 2 Pond Temperature

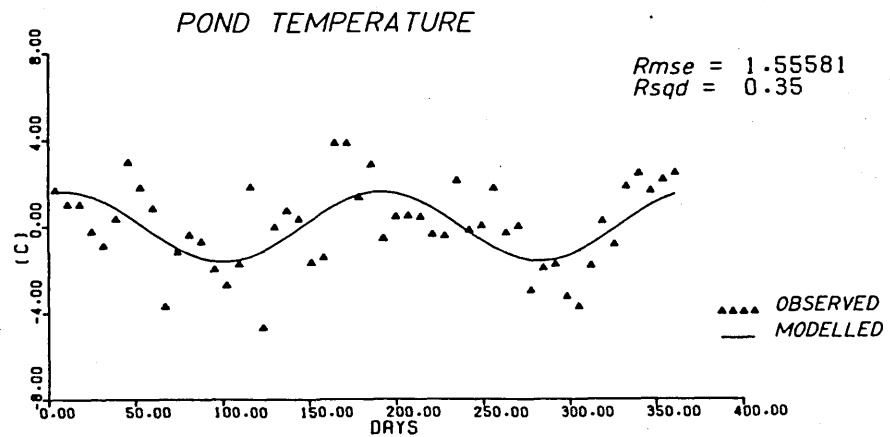


Fig 5.147 2-Cycle fit to Residuals of 1-Cycle fit to Pond 2 Pond Temperature

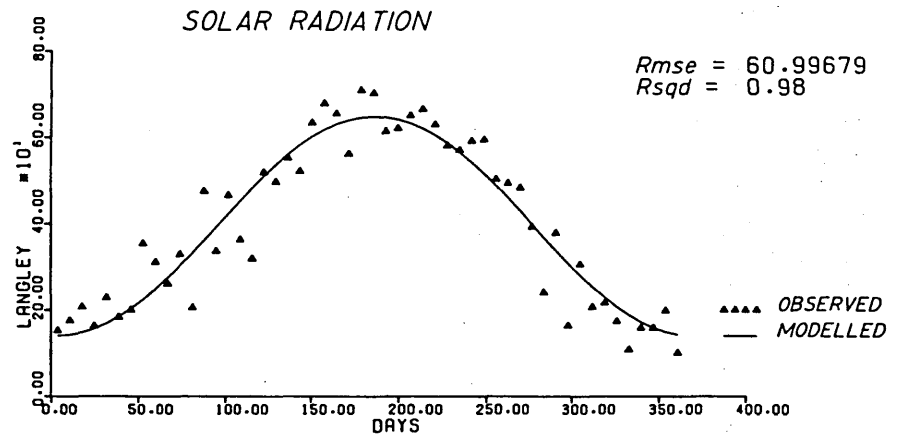


Fig 5.148 1-Cycle fit to Pond 2 Solar Radiation

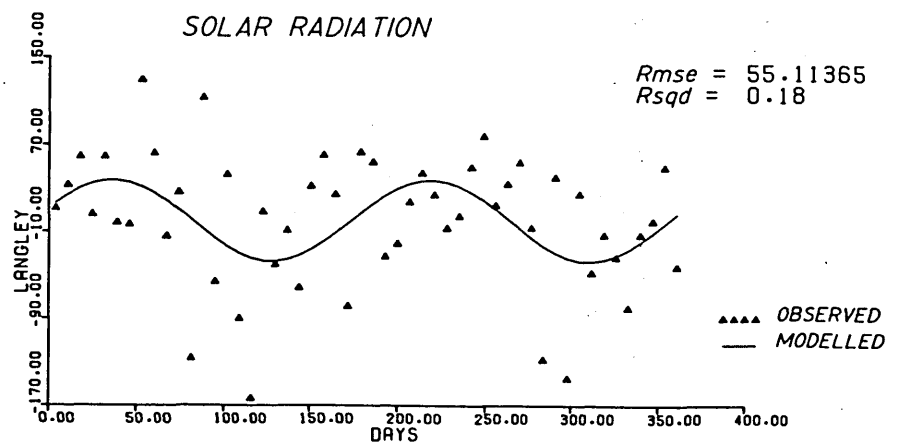


Fig 5.149 2-Cycle fit to Residuals of 1-Cycle fit to Pond 2 Solar Radiation

TABLE 5.7 [‡] Coefficients for Sine and Cosine Relations for the 1 and 2-Cycles fit to the lumped S/Sink term in Carbonaceous Working Model 3, Pond Temperature and Solar Radiation for Pond 1 and Pond 2 Data

CHARACTERISTICS	UNITS	μ	A	B	C	D
Filtered COD (Pond 1)	$\text{mg l}^{-1} \text{d}^{-1}$	5.455	5.731	-0.875	2.570	0.107
Volatile Sus. Solids (Pond 1)	$\text{mg l}^{-1} \text{d}^{-1}$	18.527	16.780	1.491	5.656	0.611
Soluble COD (Pond 2)	$\text{mg l}^{-1} \text{d}^{-1}$	0.890	-0.940	0.180	0.568	-0.199
Total Solids (Pond 2)	$\text{mg l}^{-1} \text{d}^{-1}$	3.829	-3.520	-0.384	0.527	1.929
Pond Temperature (Pond 1)	$^{\circ}\text{F}$	54.943	15.828	-2.422	1.382	1.934
Solar Radiation (Pond 1)	BTU	2282.363	970.950	129.462	255.922	751.297
Pond Temperature (Pond 2)	$^{\circ}\text{C}$	10.061	-9.918	0.519	1.543	0.495
Solar Radiation (Pond 2)	$\text{gcal cm}^{-2} \text{d}^{-1}$	395.070	-253.628	18.327	11.522	35.719

[‡] Typical Equation:- $y(t) = \mu + A \cos(\omega t) + B \sin(\omega t) + C \cos(2\omega t) + D \sin(2\omega t)$

Where $\omega = \frac{2\pi}{365}$; $t = \text{Time in Days } \{0 \leq t \leq 365\}$

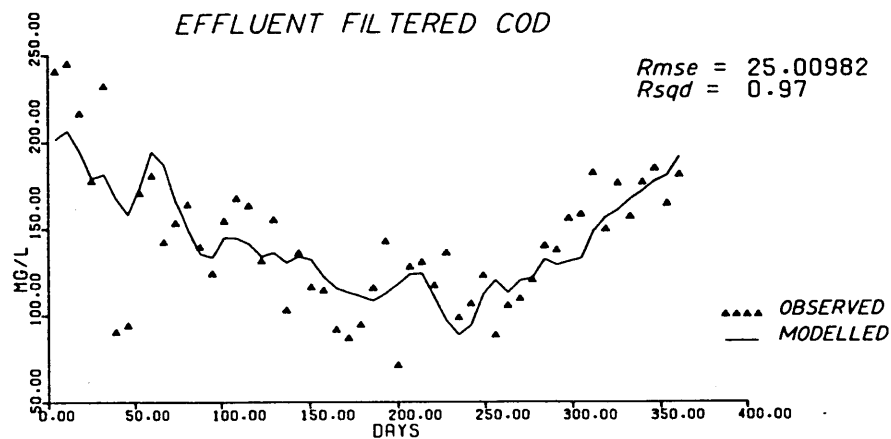


Fig 5.150 Simulation Plot for Pond 1 Effluent Filtered COD
(Carbonaceous Organics Working Model 3)

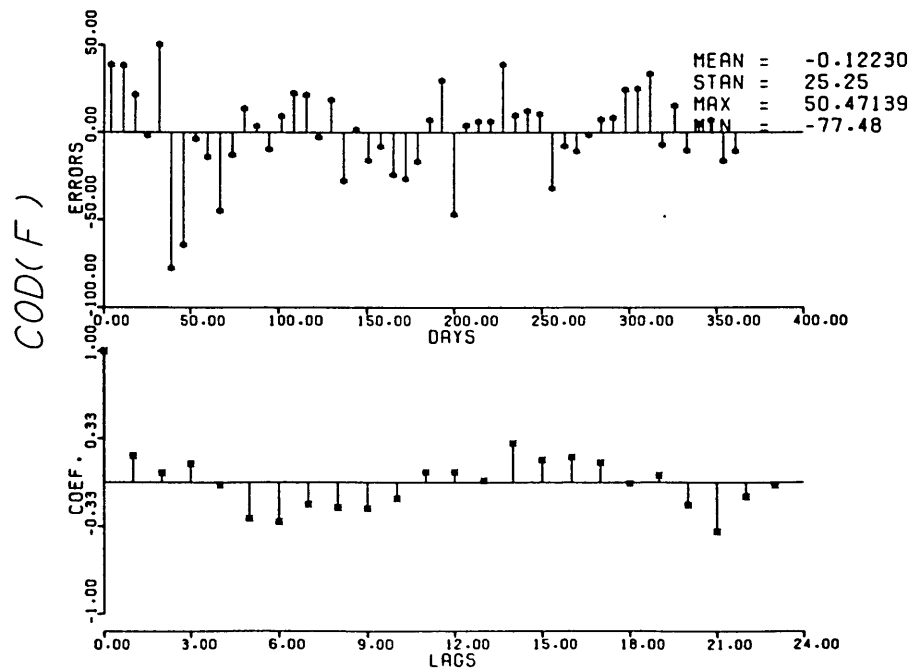


Fig 5.151 Errors Plot for Carbonaceous Organics Working Model 3
fit to Pond 1 Effluent Filtered COD

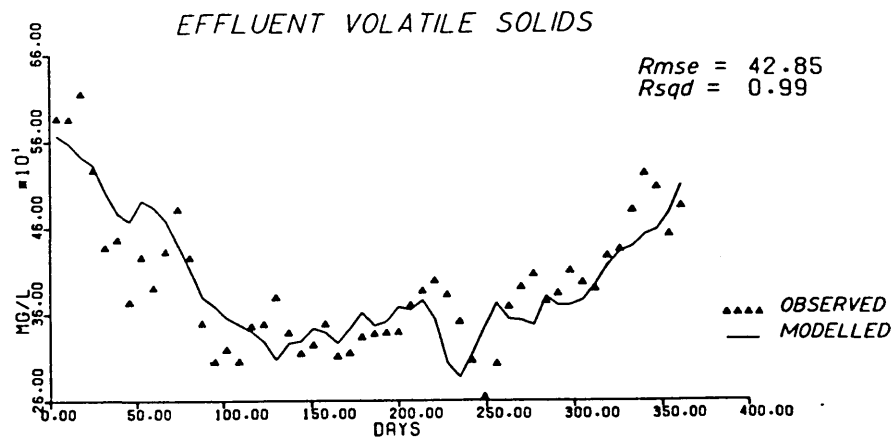


Fig 5.152 Simulation Plot for Pond 1 Effluent Volatile Suspended Solids (Carbonaceous Organics Working Model 3)

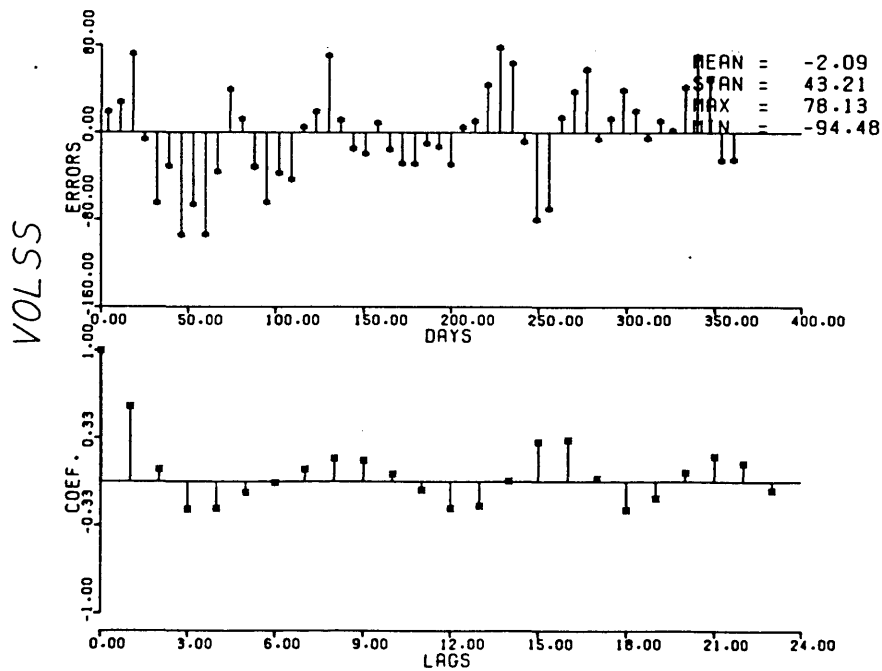


Fig 5.153 Errors Plot for Carbonaceous Organics Working Model 3
fit to Pond 1 Effluent Volatile Suspended Solids

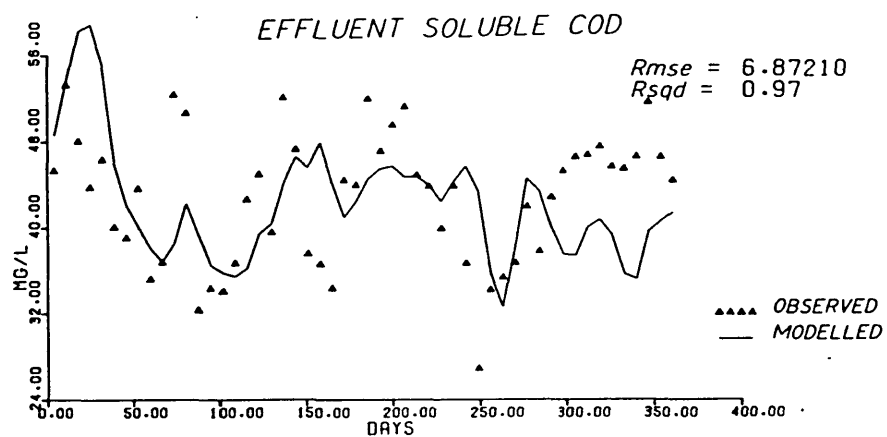


Fig 5.154 Simulation Plot for Pond 2 Effluent Soluble COD
(Carbonaceous Organics Working Model 3)

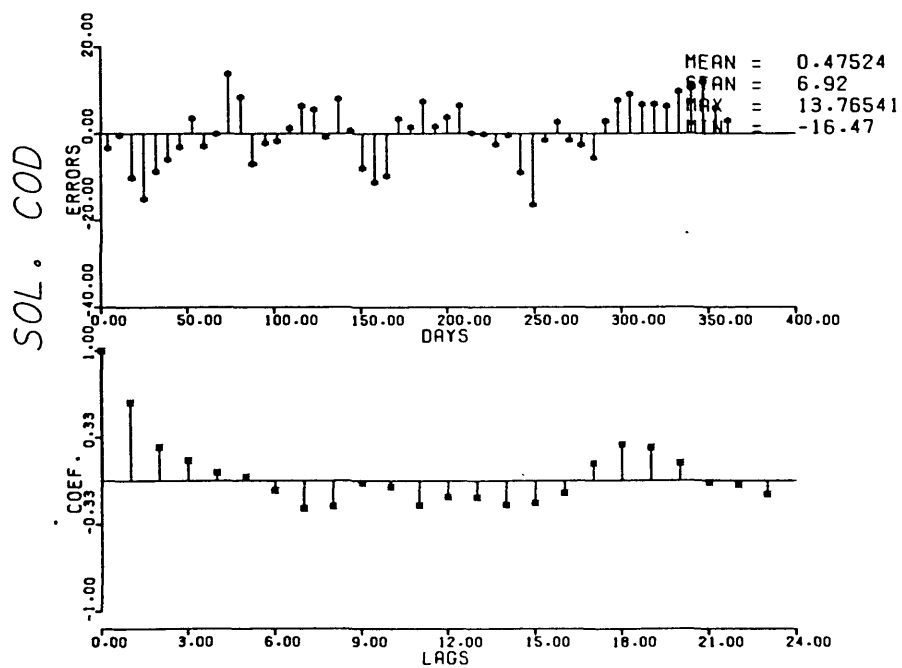


Fig 5.155 Errors plot for Carbonaceous Organics Working Model 3
fit to Pond 2 Effluent Soluble COD

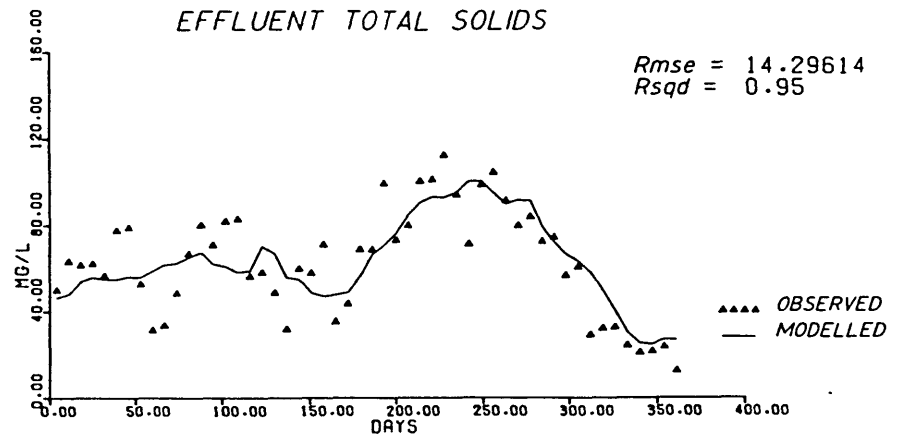


Fig 5.156 Simulation Plot for Pond 2 Effluent Total Solids (Carbonaceous Organics Working Model 3)

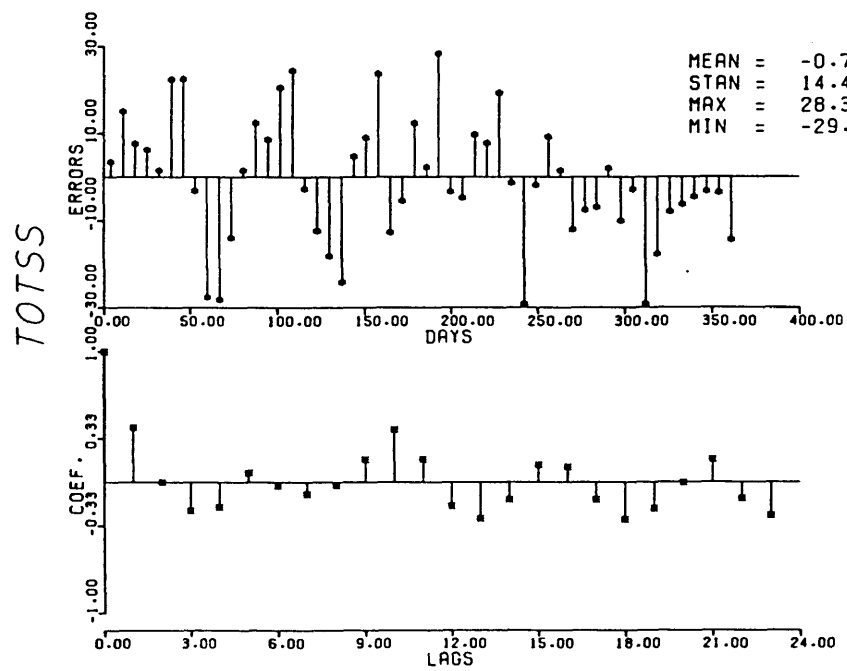


Fig 5.157 Errors Plot for Carbonaceous Organics Working Model 3 fit to Pond 2 Effluent Total Solids

TABLE 5.8: Mean Influent and Internal Loadings

	MEAN LOADING LEVELS ($\text{mg l}^{-1} \text{d}^{-1}$)			
	SOLUBLE ORGANICS		PARTICULATE ORGANICS	
	Influent	Internal	Influent	Internal
NEW MEXICO (POND 1)	8.406	5.455	15.410	18.527
CORRINE (POND 2)	2.050	0.890	2.808	3.829

TABLE 5.9 Estimated Values for Parameters in Nutrients Working Model 1 for Pond 1 Data

MODELLED STATED	PARAMETER VALUES	
	" θ_1 " (d^{-1})	" θ_2 "
Ortho-Phosphorous	0.037	1.046
Ammonia	0.023	1.013

TABLE 5.10^I Coefficients in Sine and Cosine Relation for 2-Cycle fit to lumped S/Sink term in Nutrients Working Model 2 for Pond 1 Data

MODELLED STATES	COEFFICIENTS ($\text{mg l}^{-1} \text{d}^{-1}$)		
	μ	A	B
Ortho-Phosphorous	0.031	0.383	0.011
Ammonia	0.071	0.674	-0.084

^I Typical Equation:- $y(t) = \mu + A \cos \frac{4\pi \cdot t}{365} + B \sin \frac{4\pi \cdot t}{365}$

Where t = Time in Days $\{0 \leq t \leq 365\}$

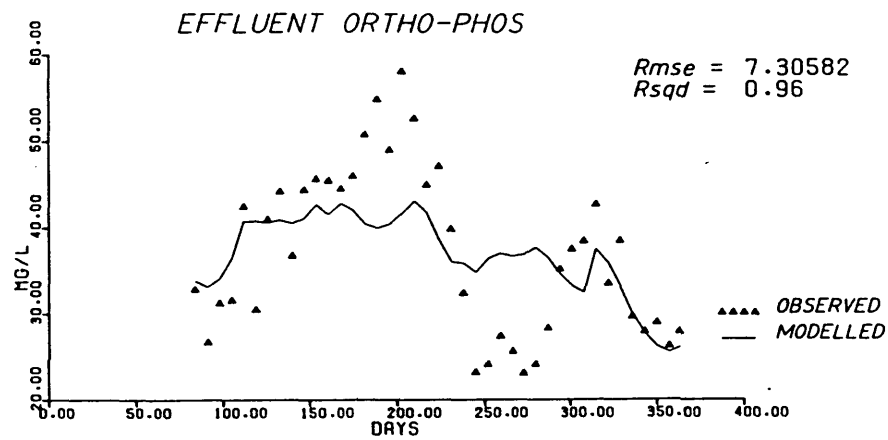


Fig 5.158 Simulation Plot for Pond 1 Effluent Ortho-Phosphorus (Nutrients Working Model 1)

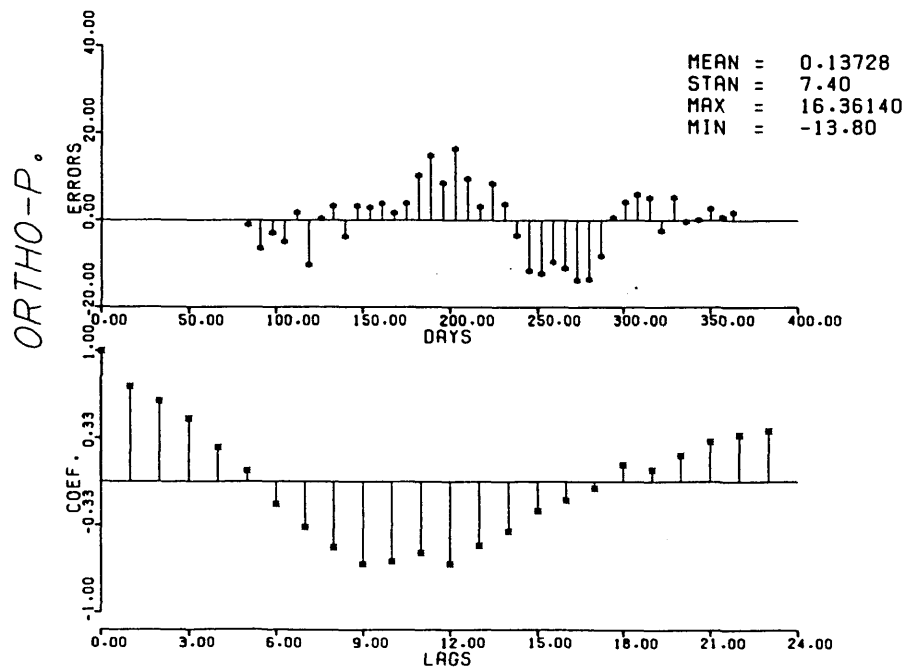


Fig 5.159 Errors Plot for Nutrients Working Model 1 Simulation of Pond 1 Effluent Ortho-Phosphorus

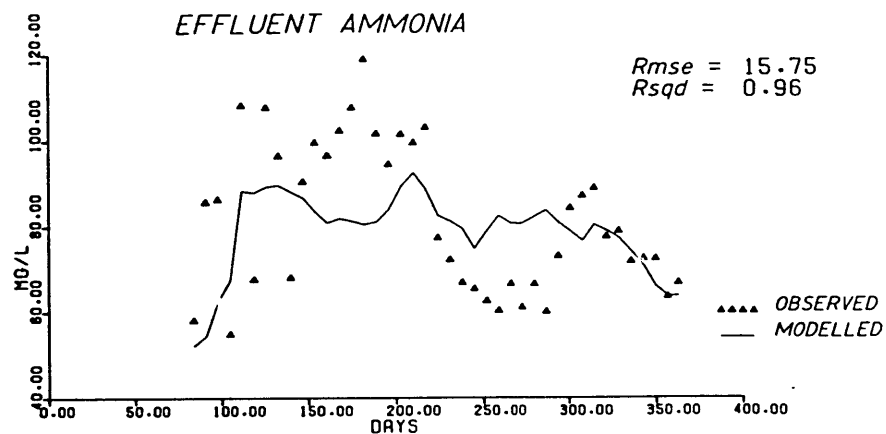


Fig 5.160 Simulation Plot for Pond 1 Effluent Ammonia (Nutrients Working Model 1)

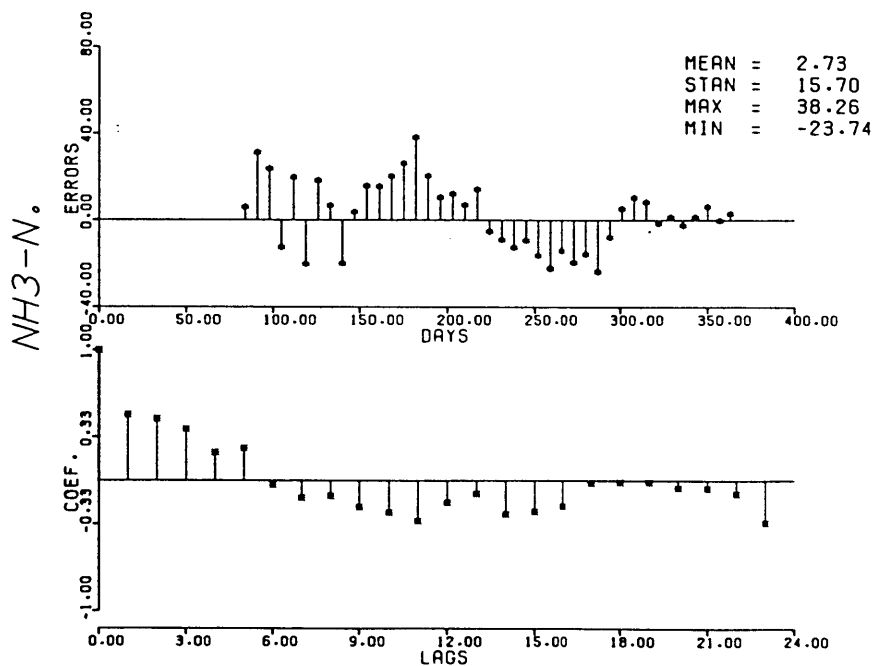


Fig 5.161 Errors Plot for Nutrients Working Model 1 Simulation of Pond 1 Effluent Ammonia

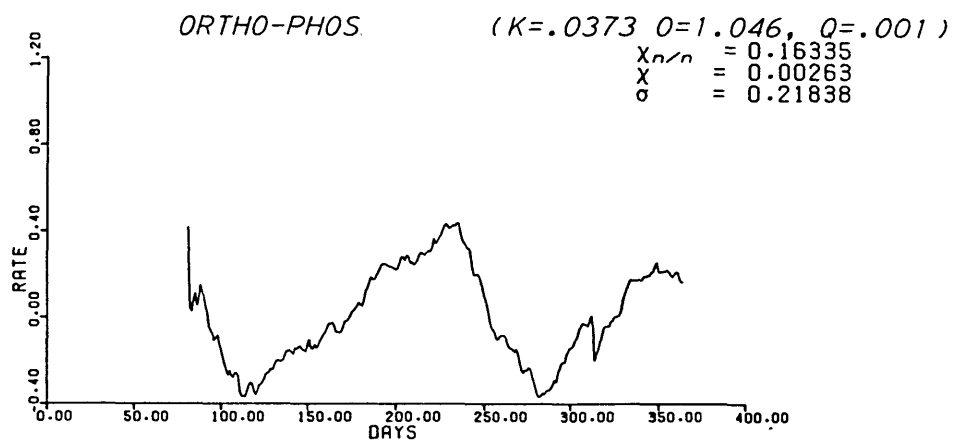


Fig 5.162 Trajectory of IV estimate of lumped S/Sink term for Pond 1
Ortho-Phosphorus (Nutrients Working Model 2)

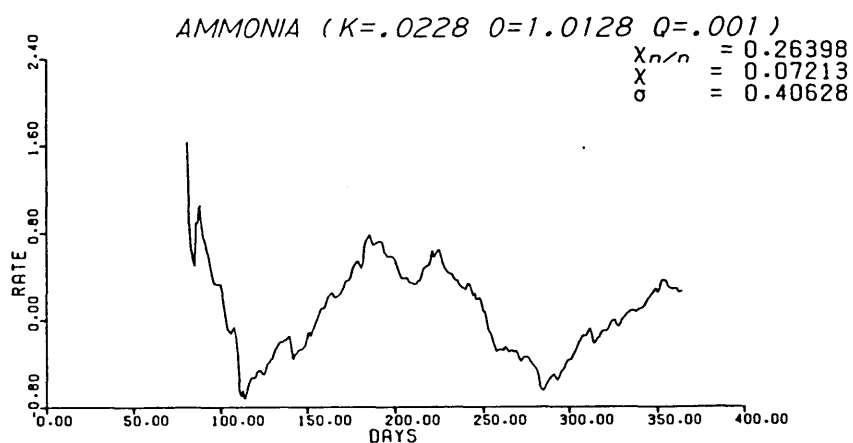


Fig 5.163 Trajectory of IV estimate of lumped S/Sink term for Pond 1
Ammonia (Nutrients Working Model 2)

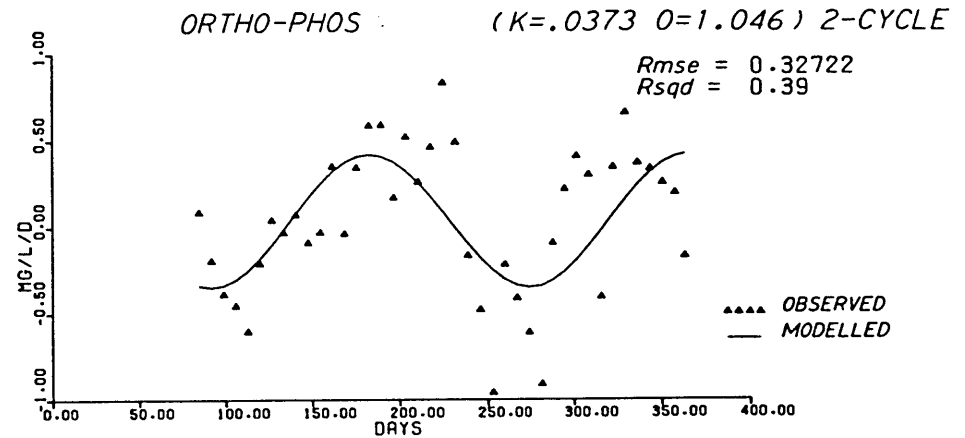


Fig 5.164 2-Cycles fit to lumped S/Sink term in Nutrients Working Model 2 for Pond 1 Ortho-Phosphorus

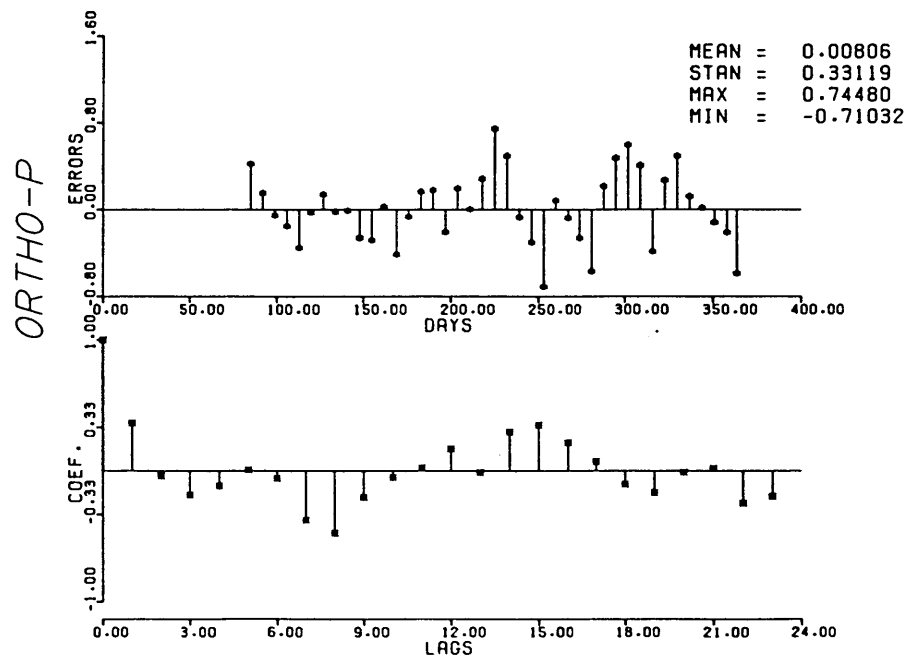


Fig 5.165 Errors Plot for 2-Cycles fit to lumped S/Sink term in Nutrients Working Model 2 for Pond 1 Ortho-Phosphorus

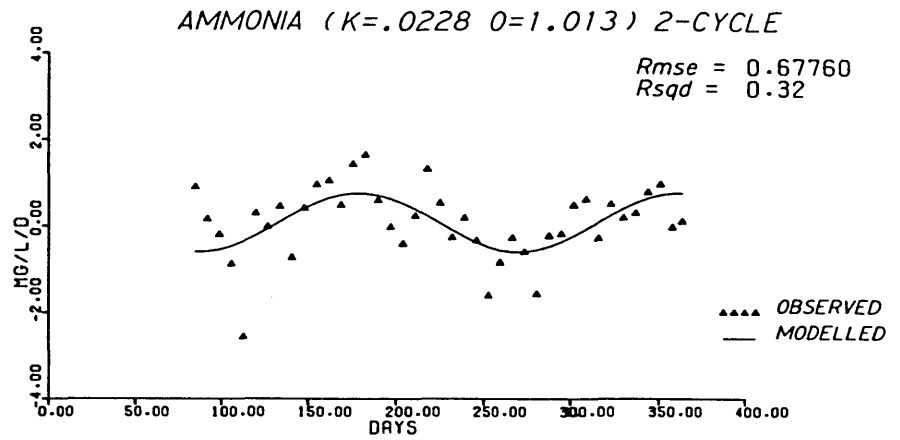


Fig 5.166 2-Cycles fit to lumped S/Sink term in Nutrients Working Model 2 for Pond 1 Ammonia

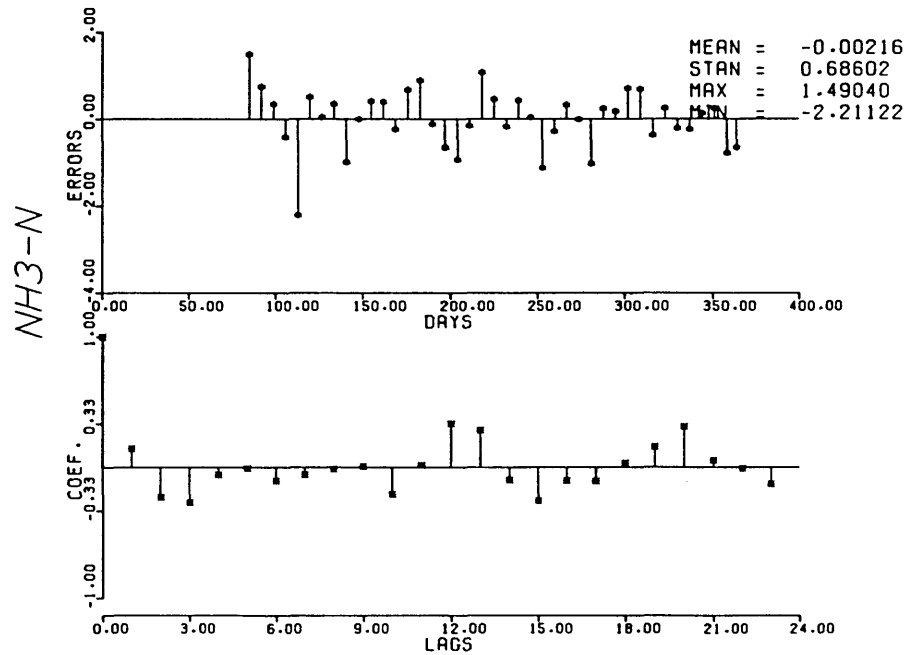


Fig 5.167 Errors Plot for 2-Cycles fit to lumped S/Sink term in Nutrients Working Model 2 for Pond 1 Ammonia

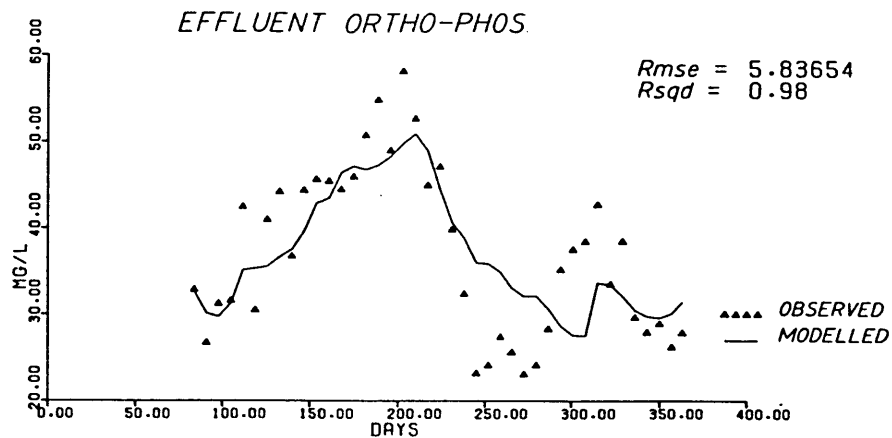


Fig 5.168 Simulation Plot for Pond 1 Effluent Ortho-Phosphorus (Nutrients Working Model 2)

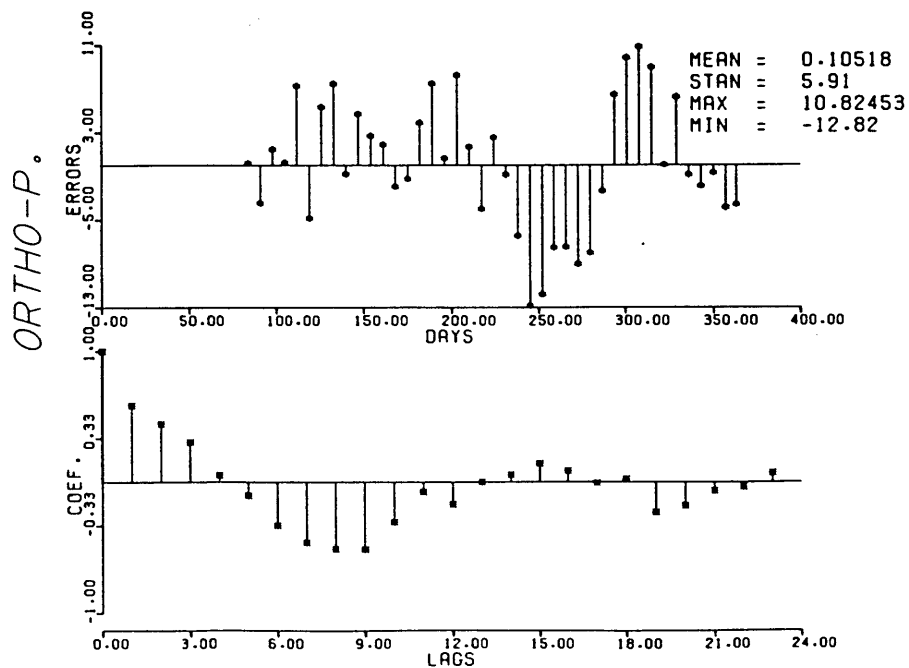


Fig 5.169 Errors Plot for Nutrients Working Model 2 Simulation of Pond 1 Effluent Ortho-Phosphorus

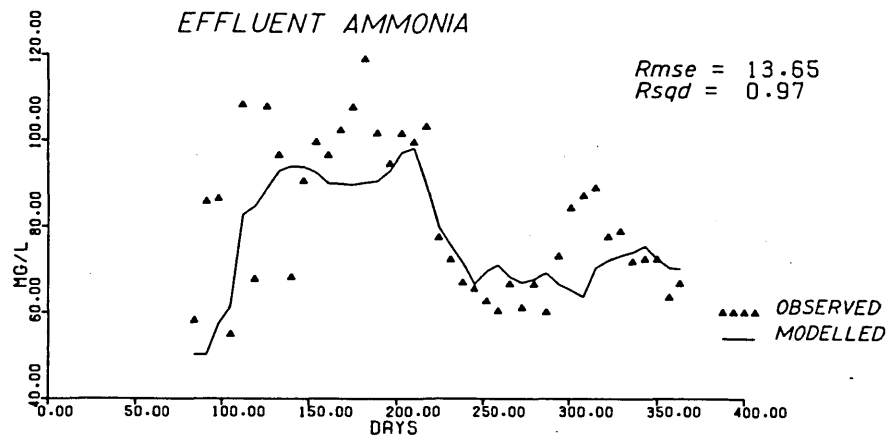


Fig 5.170 Simulation Plot for Pond 1 Effluent Ammonia
(Nutrients Working Model 2)

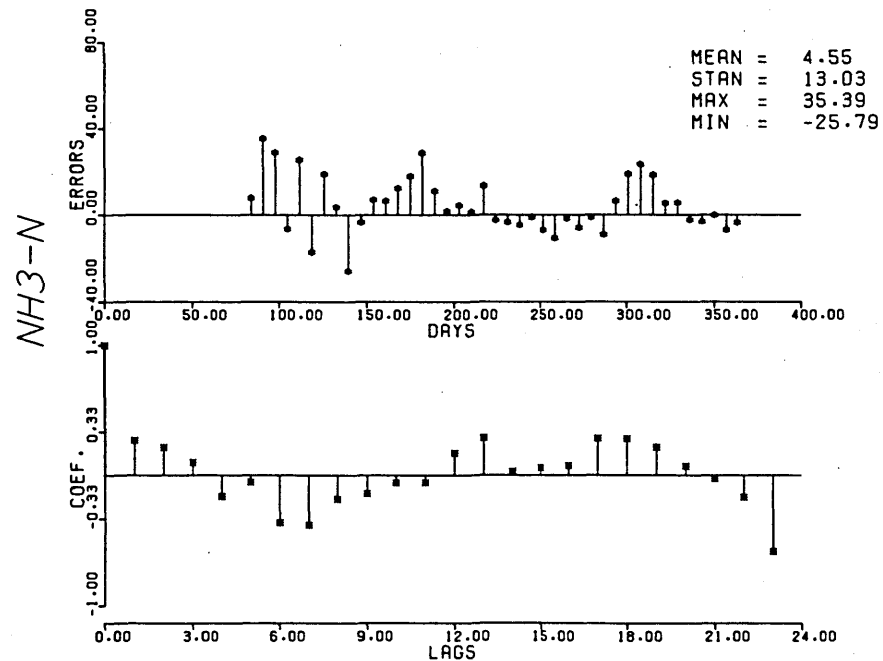


Fig 5.171 Errors Plot for Nutrients Working Model 2 Simulation
of Pond 1 Effluent Ammonia

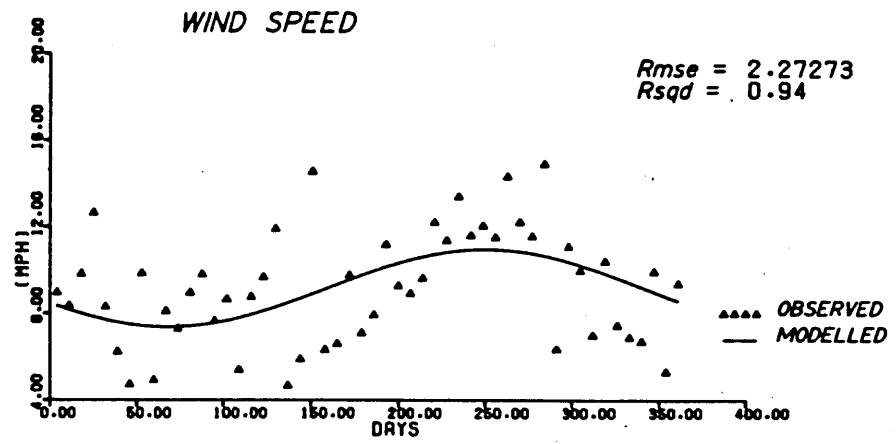


Fig 5.172 1-Cycle fit to Weekly Averaged Wind Speed for Pond 1 Data

TABLE 5.11 Estimated Values of Parameters in Nutrients Working Model 3 for Pond 1 Data

MODELLED STATES	PARAMETER VALUES					
	θ_1	θ_2	θ_3	θ_4	θ_5	θ_6
Ortho-Phosphorous	0.047	1.087	1.350	1.500	0.014	3.382
Ammonia	0.032	1.052	2.600	1.500	0.027	0.156

TABLE 5.12 Estimated Values of Parameter in Nutrients Working Model for Pond 2 Data

MODELLED STATES	PARAMETER VALUES	
	θ_1 (d ⁻¹)	θ_2
Nitrate	0.434	1.09
Ammonia	0.615	1.08

TABLE 5.13^I Coefficients for Sine and Cosine Relations for the 1 and 2-Cycles fit to the lumped S/Sink term in Nutrients Working Model 4 for Pond 2 Ammonia

	μ	A	B	C	D
Ammonia	0.1059	0.2919	0.1233	0.2193	-0.0004

^I Typical Equation:-

$$y(t) = \mu + A \cos(\omega t) + B \sin(\omega t) + C \cos(2\omega t) + D \sin(2\omega t)$$

Where $\omega = \frac{2\pi}{365}$; t = Time in Days $\{0 \leq t \leq 365\}$

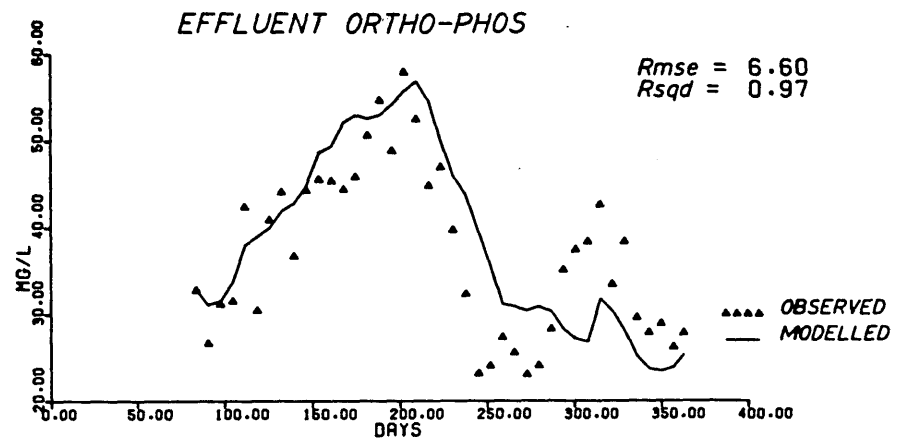


Fig 5.173 Simulation Plot for Pond 1 Effluent Ortho-Phosphorus (Nutrients Working Model 3)

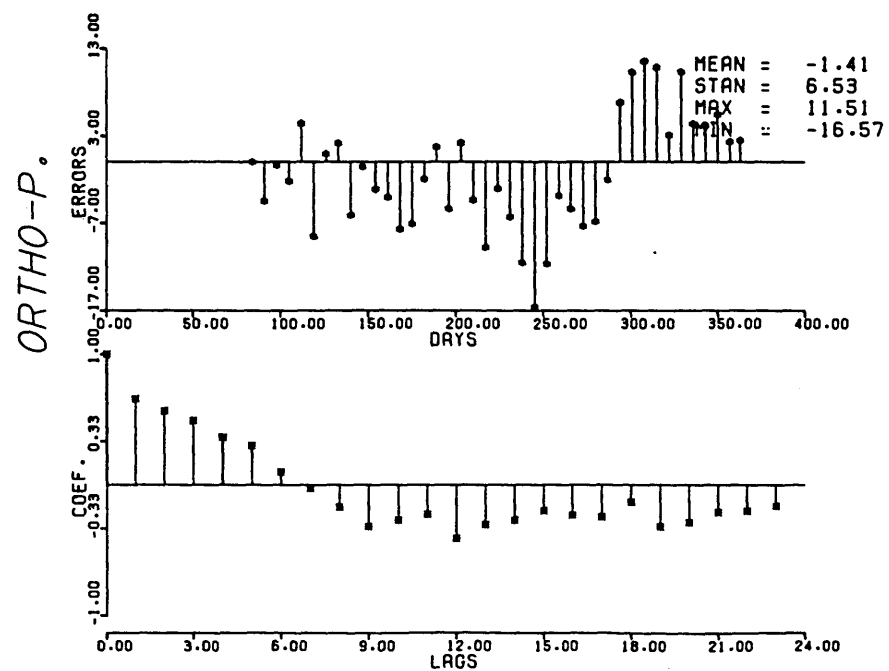


Fig 5.174 Errors Plot for Nutrients Working Model 3 simulation of Pond 1 Effluent Ortho-Phosphorus

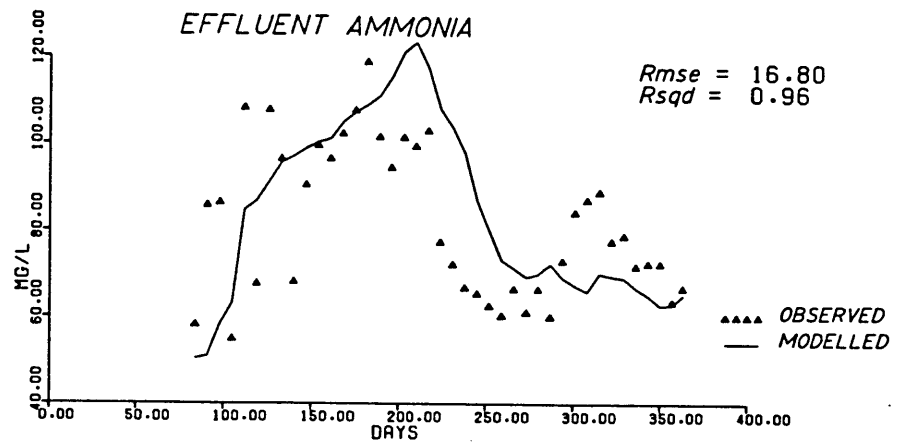


Fig 5.175 Simulation Plot for Pond 1 Effluent Ammonia (Nutrients Working Model 3)

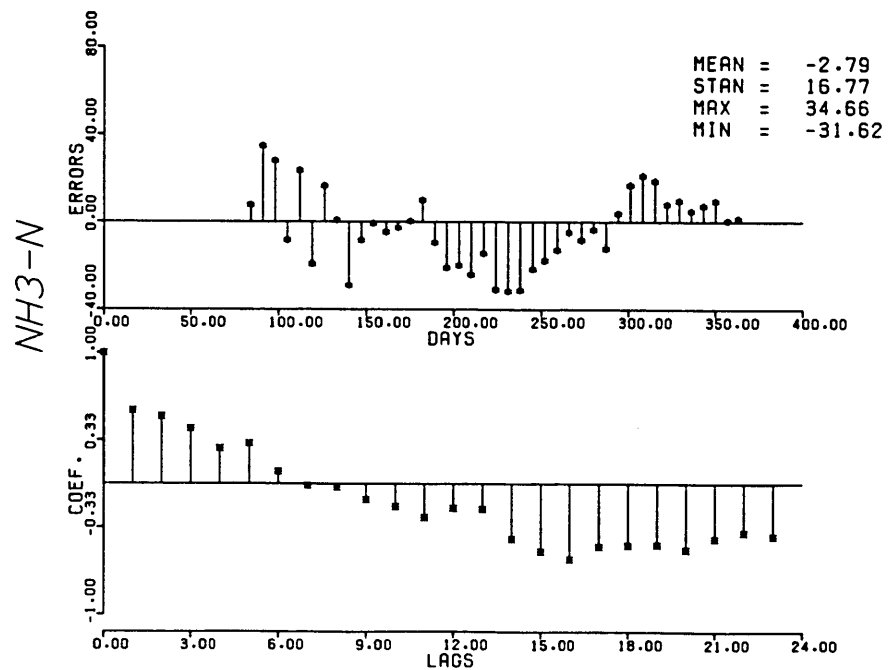


Fig 5.176 Errors Plot for Nutrients Working Model 3 Simulation of Pond 1 Effluent Ammonia

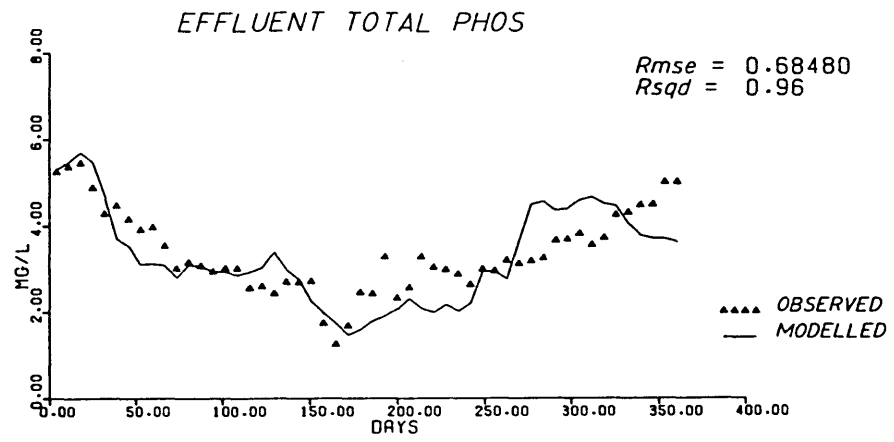


Fig 5.177 Simulation Plot for Pond 2 Effluent Total Phosphorus (Nutrients Working Model 1)

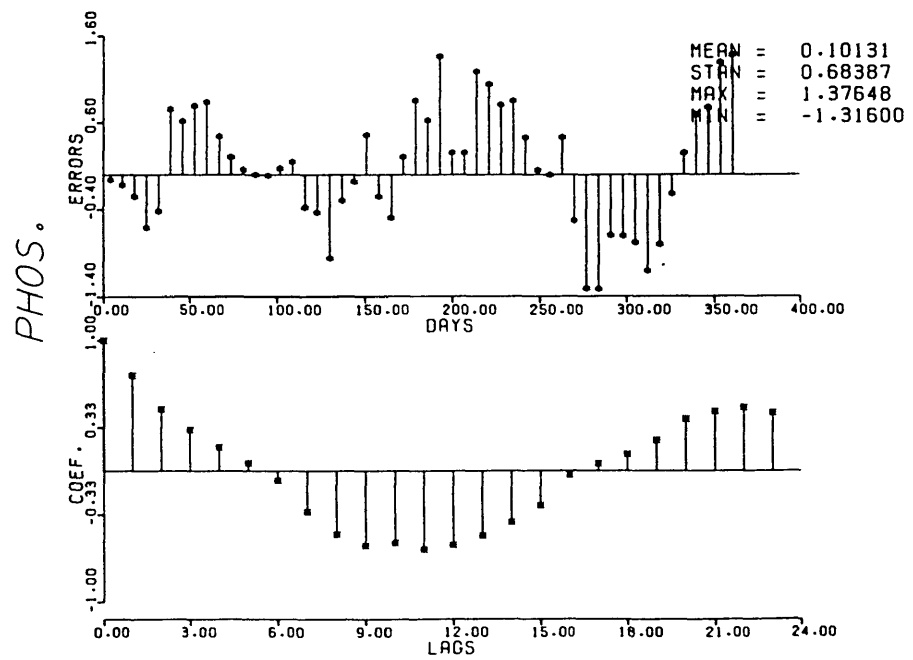


Fig 5.178 Errors Plot for Nutrients Working Model 1 Simulation of Pond 2 Effluent Total Phosphorus

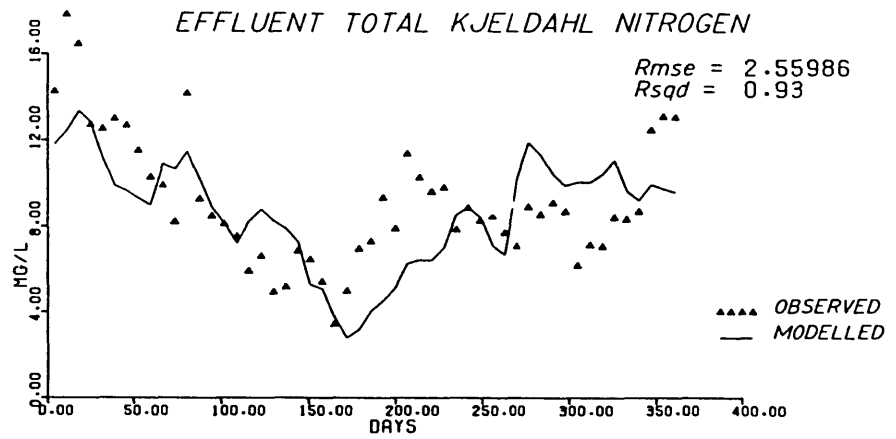


Fig 5.179 Simulation Plot for Pond 2 Effluent Total Kjeldahl Nitrogen (Nutrients Working Model 1)

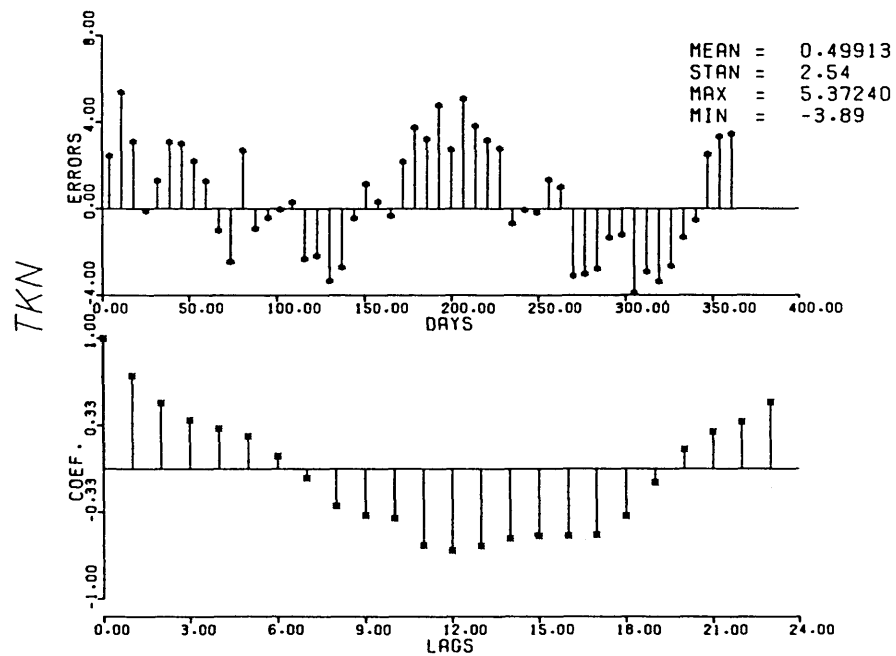


Fig 5.180 Errors Plot for Nutrients Working Model 1 Simulation of Pond 2 Effluent Total Kjeldahl Nitrogen

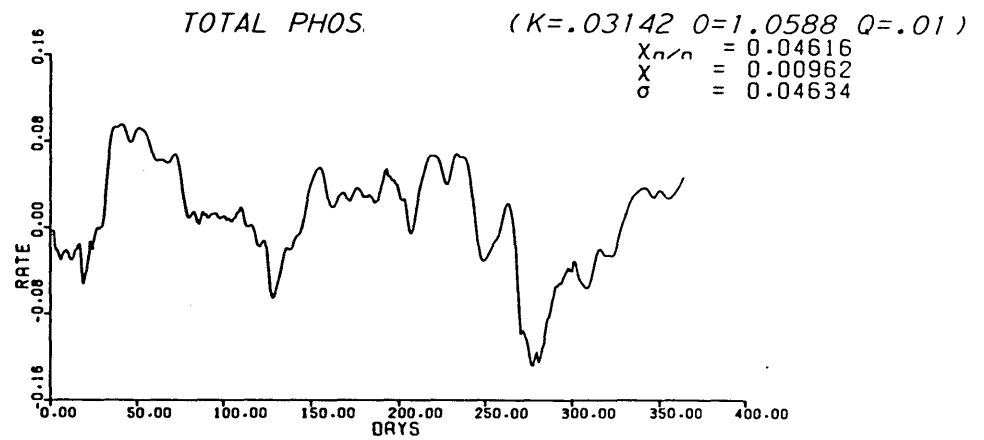


Fig 5.181 Trajectory of IV estimate of Pond 2 Total Phosphorus
lumped S/Sink term (Nutrients Working Model 2; Q = .01)

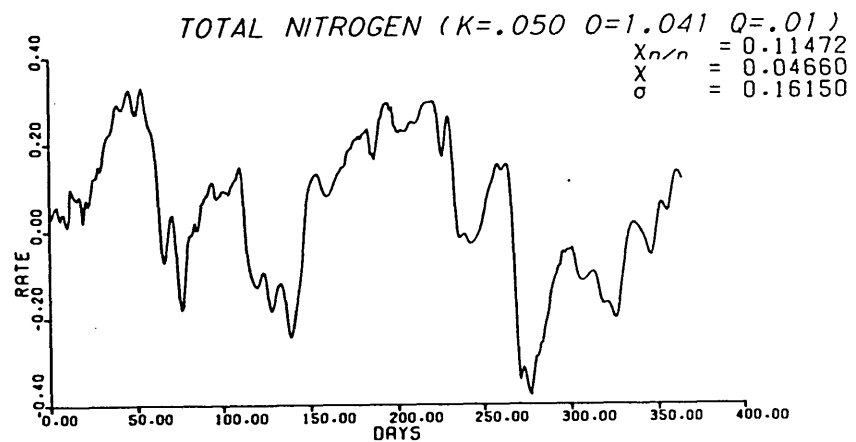


Fig 5.182 Trajectory of IV estimate of Pond 2 Total Kjeldahl
lumped S/Sink term (Nutrients Working Model 2 ; Q = .01)

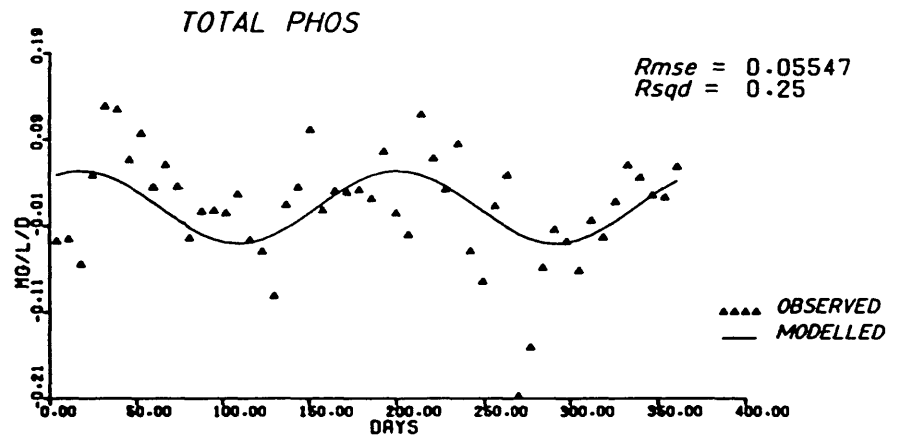


Fig 5.183 2-Cycle fit to lumped S/Sink term in Nutrients Working Model 2 for Pond 2 Total Phosphorus

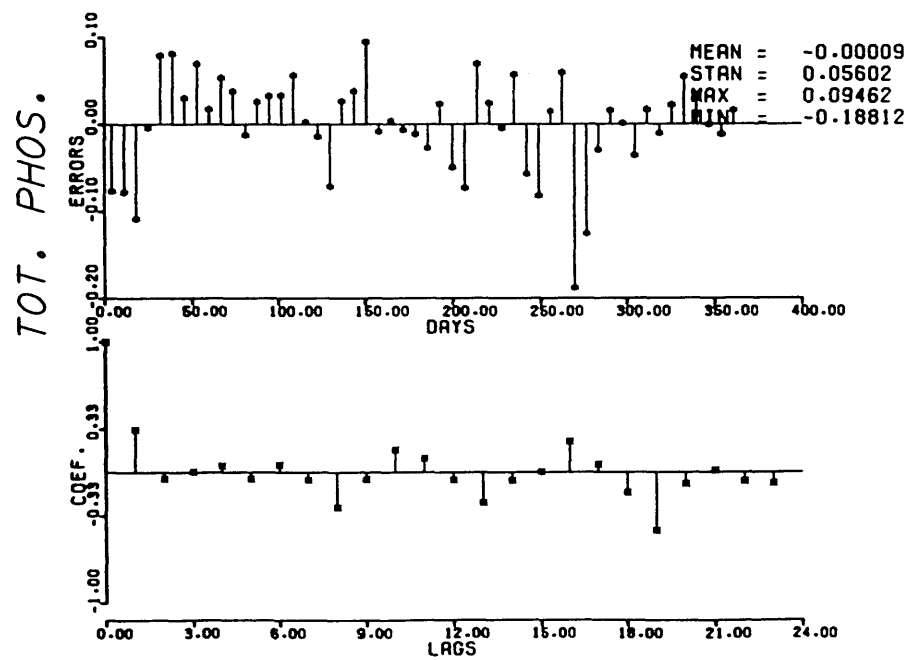


Fig 5.184 Errors Plot for 2-Cycle fit to lumped S/Sink term in Nutrients Working Model 2 for Pond 2 Total Phosphorus

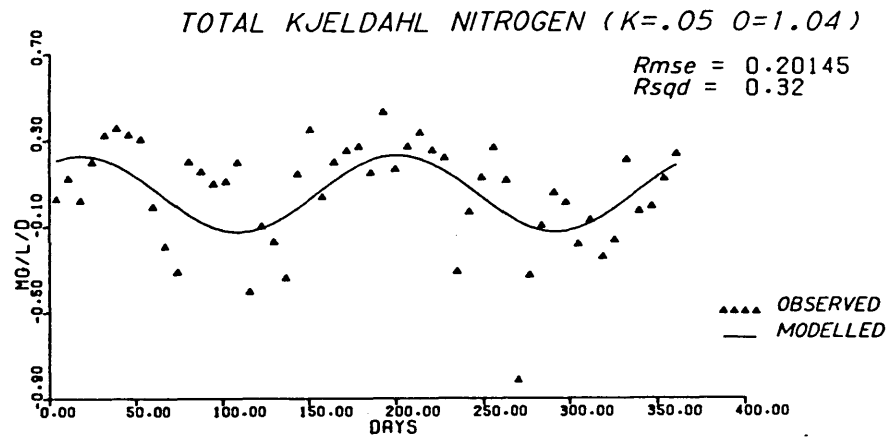


Fig 5.185 2-Cycle fit to lumped S/Sink term in Nutrients Working Model 2 for Pond 2 Total Kjeldahl Nitrogen

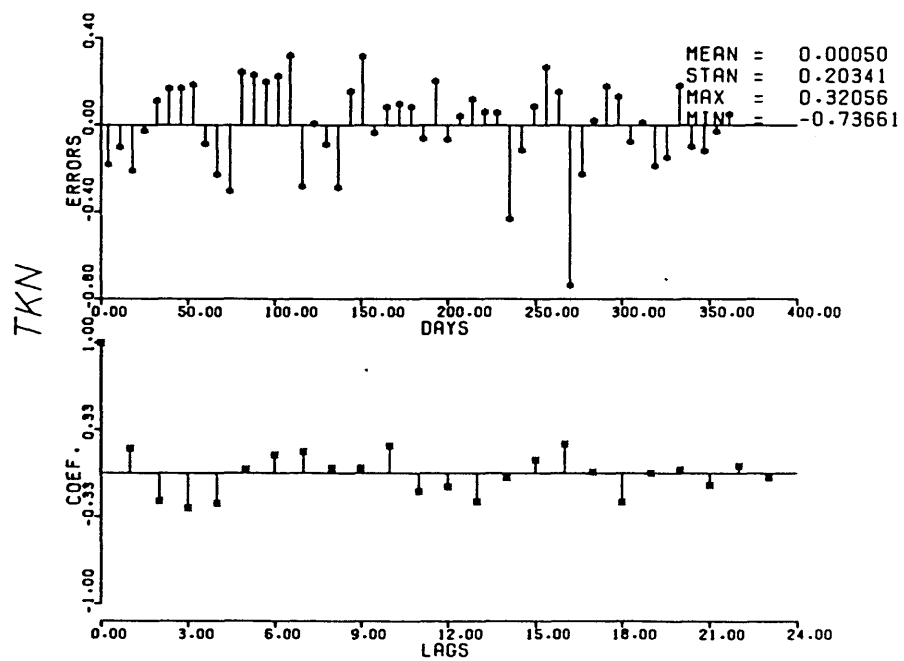


Fig 5.186 Errors Plot for 2-Cycle fit to lumped S/Sink term in Nutrients Working Model 2 for Pond 2 Total kjeldahl Nitrogen

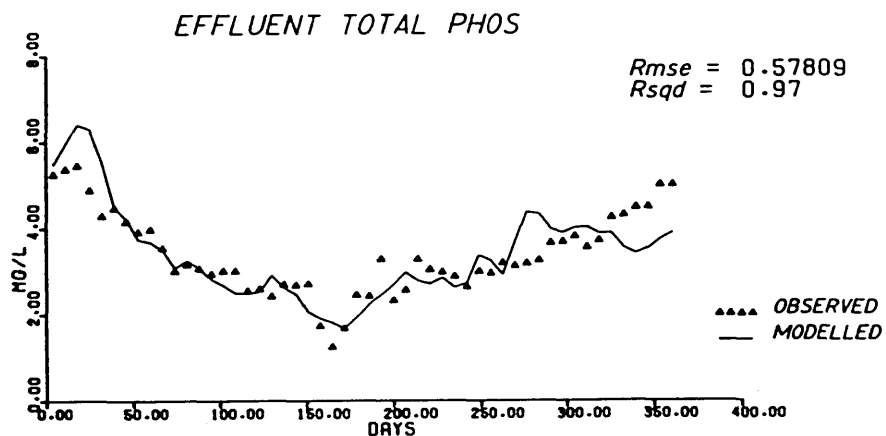


Fig 5.187 Simulation Plot for Pond 2 Effluent Total Phosphorus (Nutrients Working Model 2)

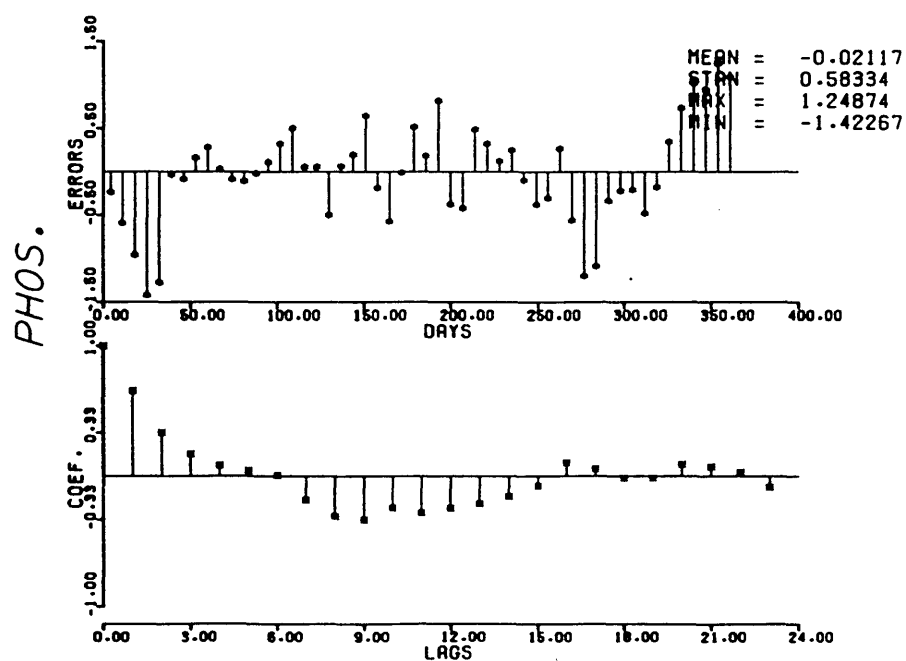


Fig 5.188 Errors Plot for Nutrients Working Model 2 Simulation for Pond 2 Effluent Total Phosphorus

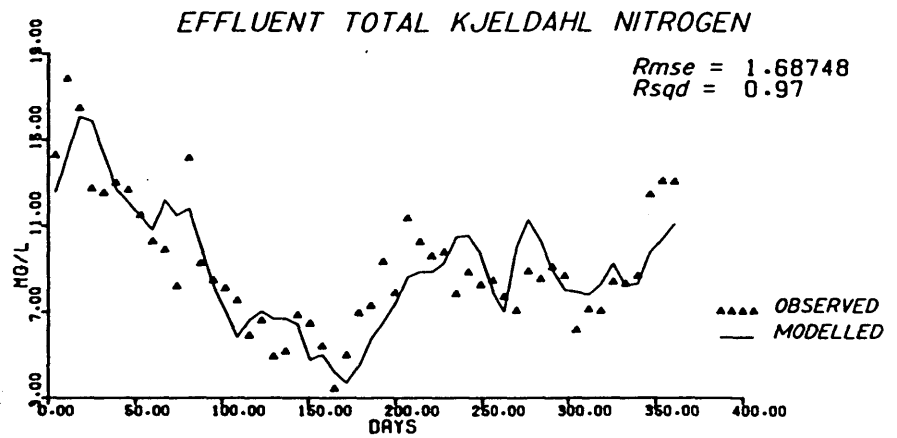


Fig 5.189 Simulation Plot for Pond 2 Effluent Total Kjeldahl Nitrogen (Nutrients Working Model 2)

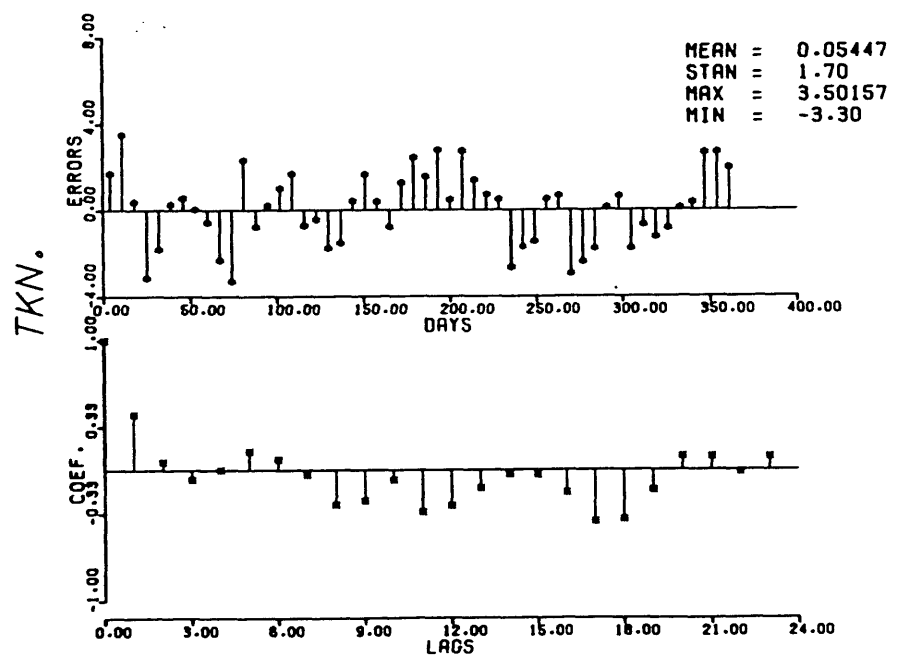


Fig 5.190 Errors Plot for Nutrients Working Model 2 Simulation for Pond 2 Effluent Total kjeldahl Nitrogen

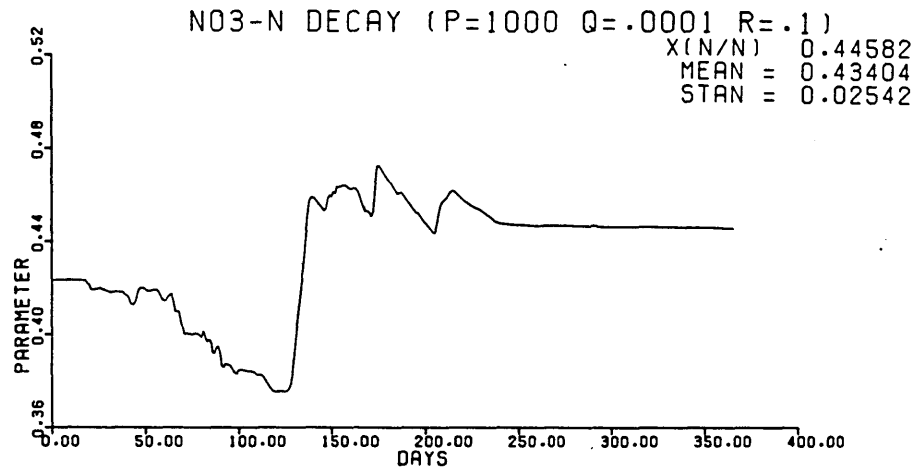


Fig 5.191 Trajectory of EKF estimate of First Order Decay
 Rate: Nitrate for Nutrients Working Model 1 (Pond 2)

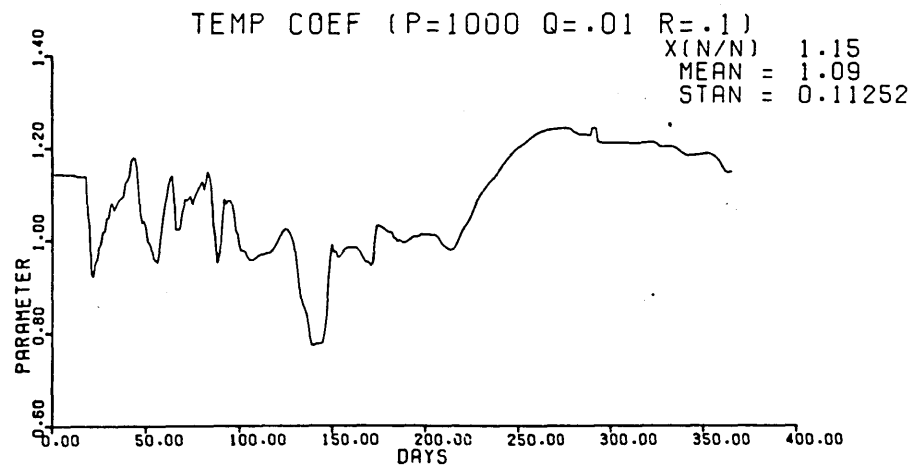


Fig 5.192 Trajectory of EKF estimate of Temperature Coefficient
 in First Order Decay Rate: Nitrate for Nutrients
 Working Model 1 (Pond 2)

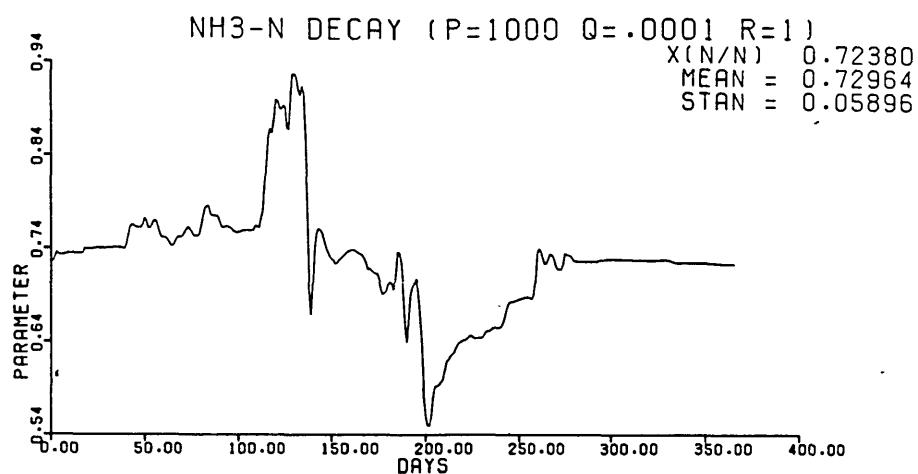


Fig 5.193 Trajectory of EKF estimate of First Order Decay Rate: Ammonia for Nutrients Working Model 1 (Pond 2)

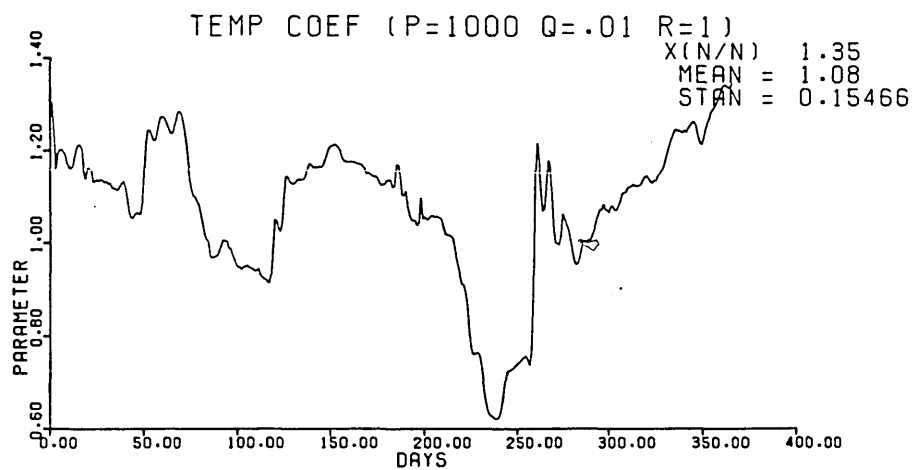


Fig 5.194 Trajectory of EKF estimate of Temperature Coefficient in First Order Decay Rate: Ammonia for Nutrients Working Model 1 (Pond 2)

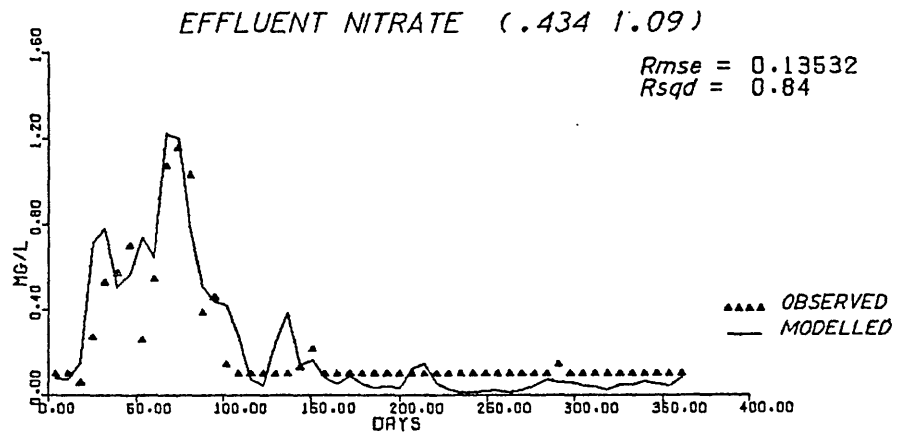


Fig 5.195 Simulation Plot for Pond 2 Effluent Nitrate (Nutrients Working Model 1)

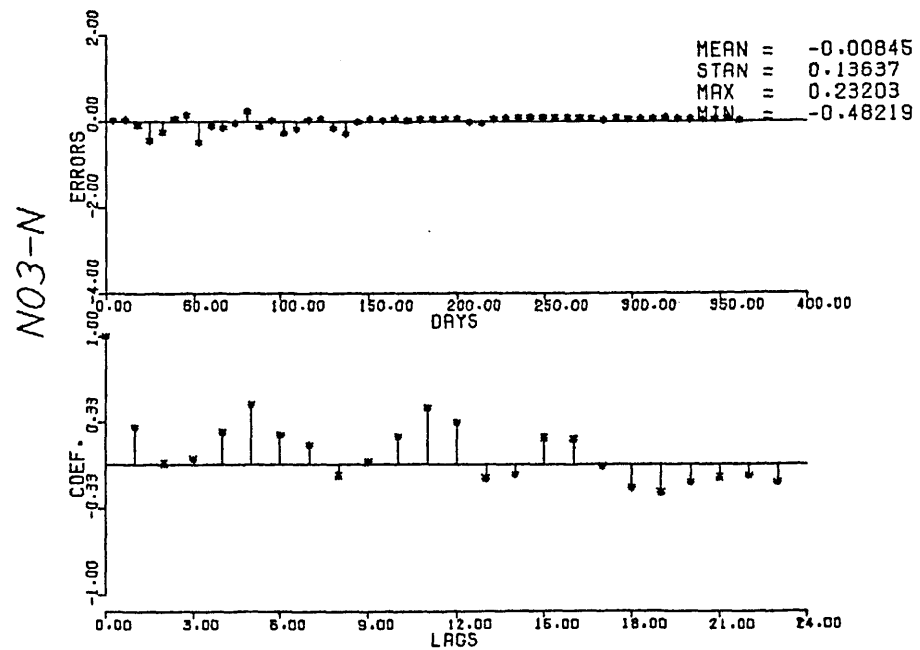


Fig 5.196 Errors Plot for Nutrients Working Model 1 Simulation of Pond 2 Effluent Nitrate

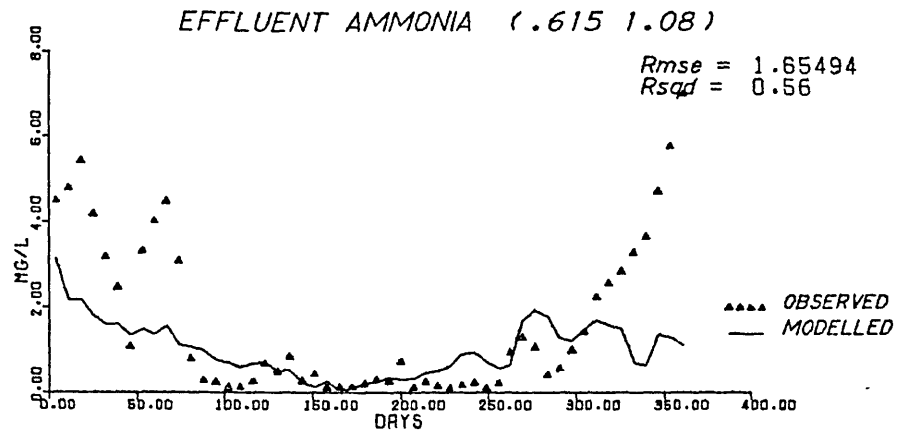


Fig 5.197 Simulation Plot for Pond 2 Effluent Ammonia
(Nutrients Working Model 1)

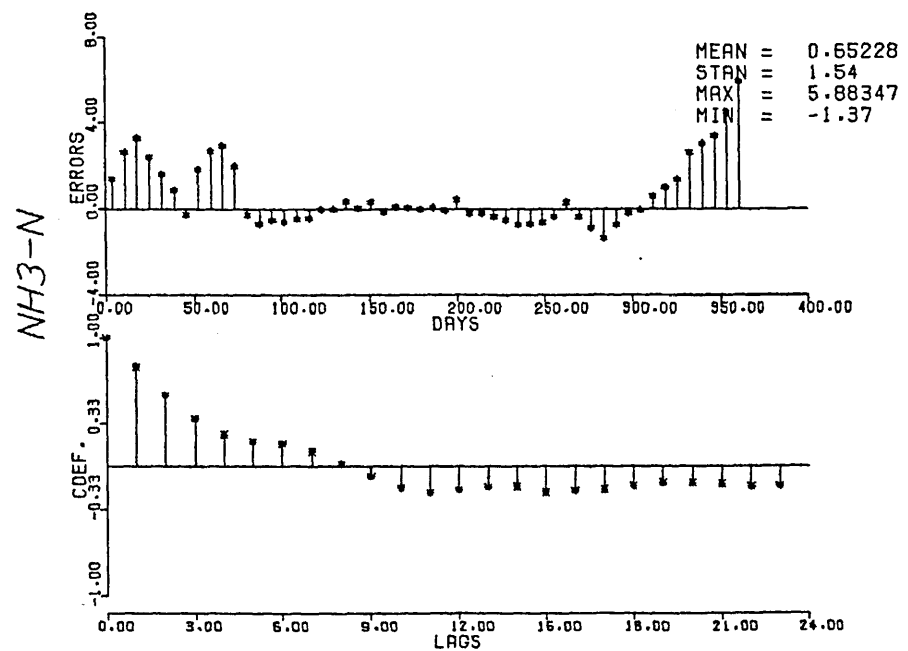


Fig 5.198 Errors Plot for Nutrients Working Model 1 Simulation
of Pond 2 Effluent Ammonia

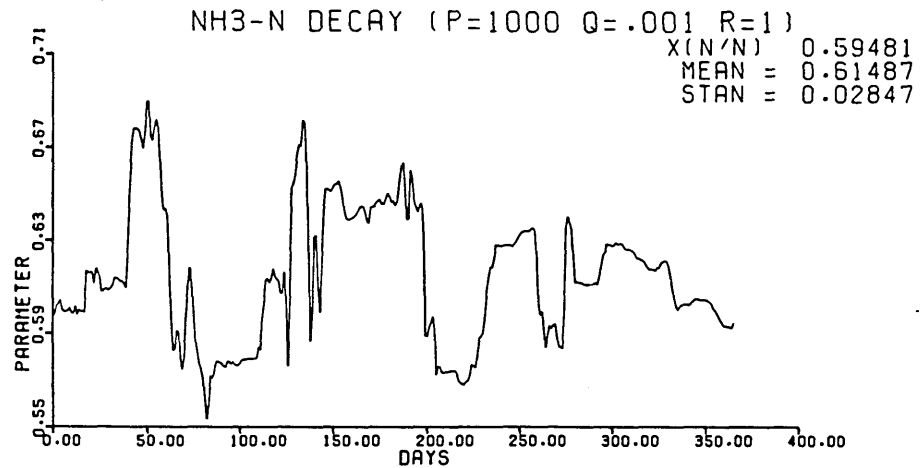


Fig 5.199 Trajectory of EKF estimate of First Order Decay Rate:
Ammonia for Nutrients Working Model 4 (Pond 2)

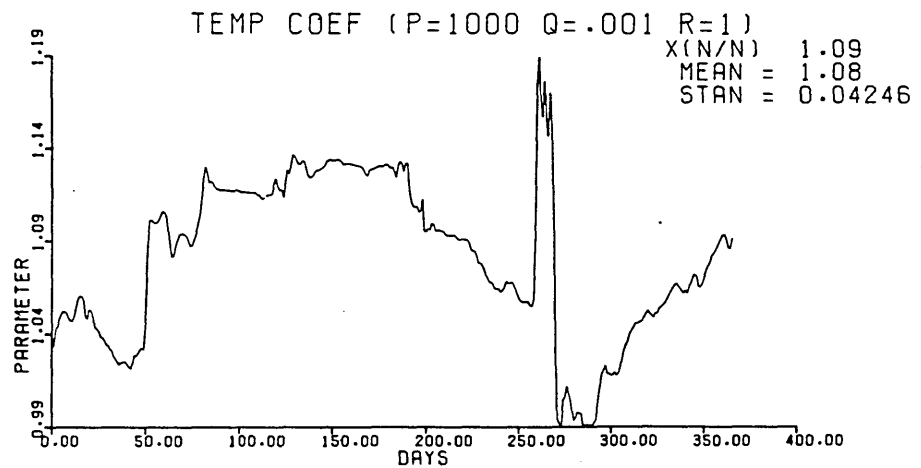


Fig 5.200 Trajectory of EKF estimate of Temperature Coefficient
for First Order Decay Rate: Ammonia for Nutrients
Working Model 4 (Pond 2)

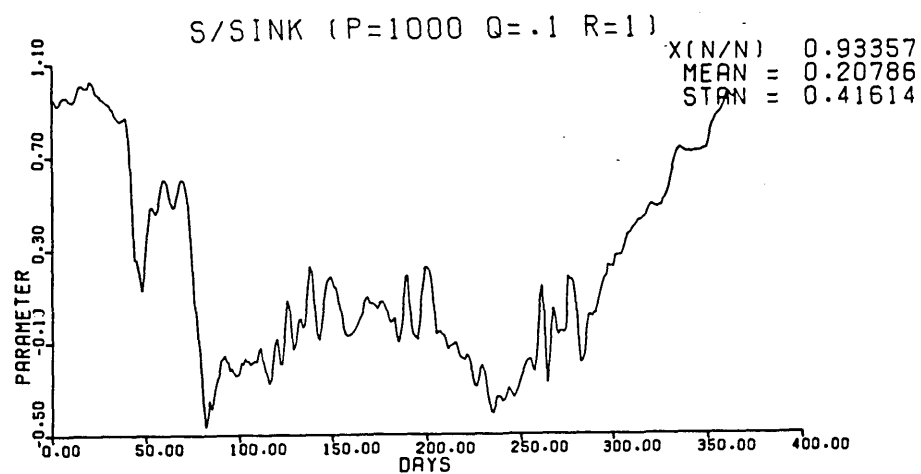


Fig 5.201 Trajectory of EKF estimate of Lumped S/Sink term in Nutrients Working Model 4 for Pond 2 Ammonia

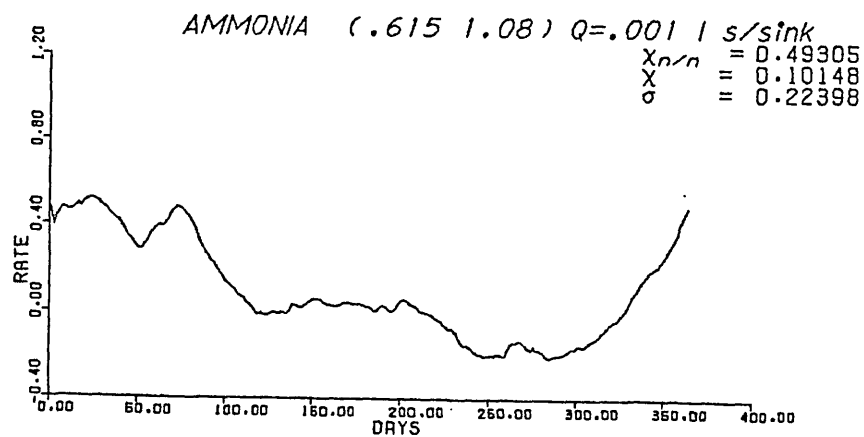


Fig 5.202 Trajectory of IV estimate of Lumped S/Sink term in Nutrients Working Model 4 for Pond 2 Ammonia

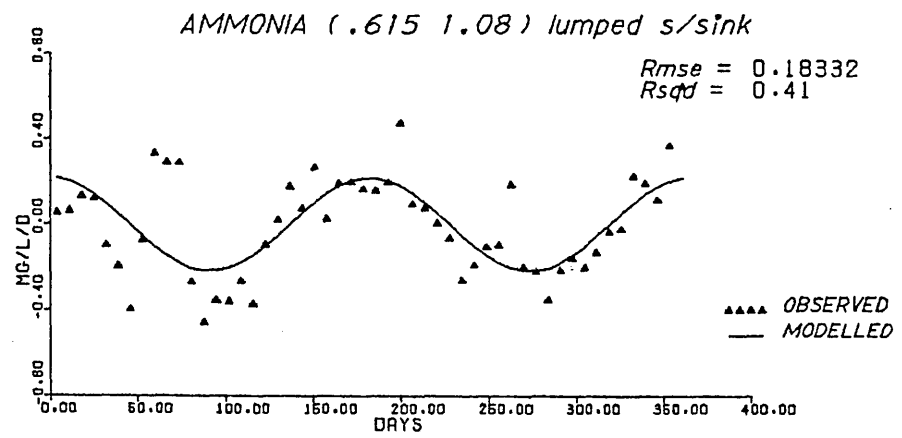


Fig 5.203 2-Cycles fit to Residuals of 1-Cycle fit to Lumped S/Sink term in Nutrients Working Model 4 for Pond 2 Ammonia

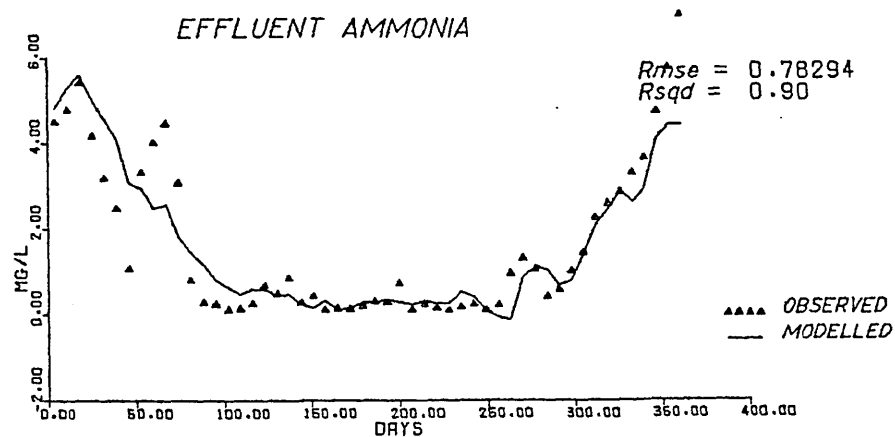


Fig 5.204 Simulation Plot for Pond 2 Effluent Ammonia (Nutrients Working Model 4)

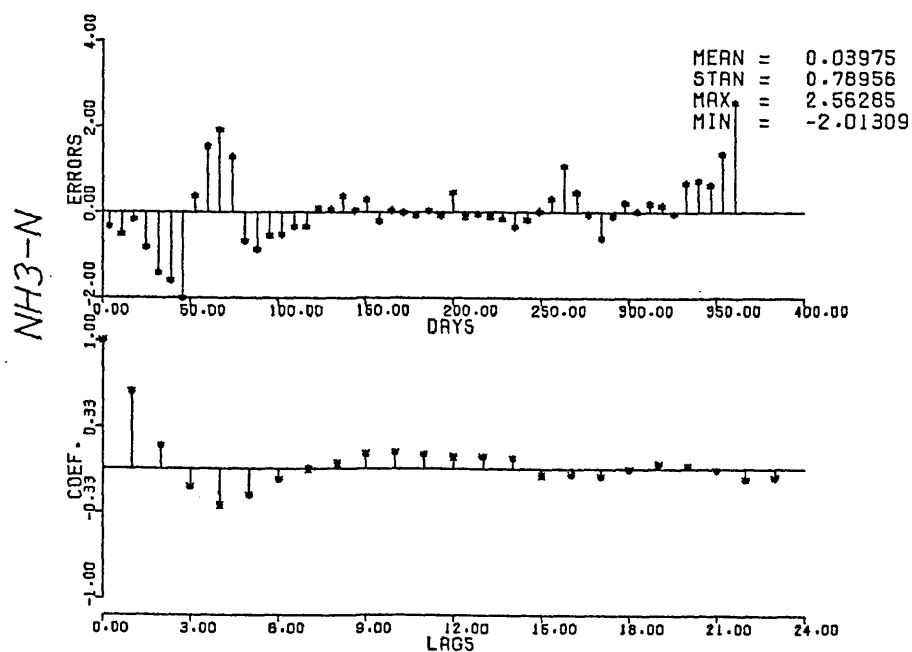


Fig 5.205 Errors Plot for Nutrients Working Model 4 Simulation of Pond 2 Effluent Ammonia

APPENDIX A

The numerical integration scheme and model assessment procedure used in this work are described in this appendix.

A1 SIMULATION PROCEDURE

The representation of the time rate of change of n system states, for mathematical models based on spatial aggregation and on a conservation of mass principle, can be represented by the following general non-linear differential equation format:

$$dx_i/dt = f(U_i, X_i, \theta_i, t) \quad \dots A.1$$

for $i = 1, 2, \dots, n$

$$U_i \subset U$$

$$X_i \subset X$$

$$\theta_i \subset \theta$$

$$x_i \in X_i$$

Where; $f(\dots)$ is the a general non-linear vector valued function of the terms in brackets; U_i is a p -dimensional vector of the input factors deemed to influence the the i^{th} state x_i (note, U_i is a subset of the set of all system inputs U); X_i is a 1-dimensional vector made up of the states of the system which interact in some way to influence the dynamics of the i^{th} state; θ_i represents the r -dimensional vector of parameters (rate coefficients etc.) in the functional representation for the components taken into account in the derivation of time rate of change equation for the i^{th} state; and t is the independent variable time.

The above representation comprises of a completely deterministic mathematical model representation in which it is deemed that all

relevant system mechanisms have been accounted for and the resulting equation is a true description of the system.

To obtain the profile of each state x_i over the simulation period, the corresponding Equation (A.1) has to be integrated over the simulation period.

Because of the non-linear functionals and the coupling that results for interacting states, a closed form solution, cannot generally be obtained. However, because the resulting time rate equation is based on a conservation of mass principle, it is formulated on an accounting scheme basis; involving loss and gain terms. This results in a stable form, for most model representations, which can easily be integrated using numerical integration techniques.

Numerical integration techniques range from the simple fixed step Euler scheme to more accurate procedures with variable step length and error tolerance checking (Burden et al., 1981) . Trial simulation runs showed that a simple fourth order Runge Kutta procedure with a fixed step length of 2.4 hours for a one year simulation period produced results comparable in accuracy with a variable step integrating routine, for the number of significant figures of interest in the measurement data. The integration routines are presented in the microfiche attached.

The state(s) profiles which result from integrating Eq. A.1 represents systems states which may have corresponding measures in the measurement data available. The suitability of the resulting model as a description of pond process has often been evaluated

based on a visual comparison between the resulting model output profile and the profile of the measured data sequence. In this work, a more formal procedure was adopted for the assessment of simulation results.

A2 MODEL ASSESSMENT PROCEDURE

In this work, the simulation results were subjected to a formal model assessment procedure. This centred around the basic premise that an adequate process model, calibrated for a particular data set, should at least be able to reproduce the subset of pond behavioural characteristics described by the data set. This means that the resulting discrepancies between model output and the observed process behaviour (ie. the errors or residual sequence) should not contain any information relating to dominant pond process mechanisms.

Where some information in the residual sequence was indicated, this was explained in terms of the system dynamics. It is deemed that only in this respect can the maximum possible information on system dynamics be identified from the data.

If the information contained in the data (ie. the measured sequences) are assumed to be wide sense stationary, then ideally, the errors should be characteristically indistinguishable from white noise.

The formal assessment procedure was as follows:

1) SIMULATION PLOTS:

The simulation plots comprise of plots of model output and observed pond behaviour on the same set of axes to enable an initial visual assessment to be made of how close the simulated system behaviour was to the observed.

This criterion has been the only one applied to most modelling exercises on pond systems. However, without a formal statistical analysis of the resulting errors for the modelled states, scaling effects can result in erroneous conclusions being drawn on the representativeness of the adopted model structure.

For modelled states where no measurement information was available, the resulting model output was assessed in relation to the expected behaviour based on the process description given.

The simulation plots for measured states show two statistics. These describe, the extent to which the model accounts for the observed variations in the measured states. These are the root mean squared errors (R_{mse}) and the coefficient of determination (R_{sqd}) respectively.

The root mean squared errors, represents the square root of the mean of the squared errors. This is given by:

$$R_{mse} = (\sum e_i^2 / n)^{1/2} \quad \dots A.2$$

Where; e_i represents the discrepancy between model output and observed behaviour at the i^{th} sampling instant; and n represents the total number of data points in the simulation period.

For a perfect model fit, there will be no errors (ie. $e_i = 0$; $i=1, \dots, n$) resulting in a R_{mse} of zero. Because the measurement data

itself is noisy (and could possibly be structured to some extent), a R_{mse} of zero would not be expected. In general, however, a model producing a better fit to the data would be expected to have a lower R_{mse} .

The coefficient of determination R_{sqd} is a measure of the "goodness of fit" and it describes the extent to which the observed variation in the measured sequence is accounted for by the model output profile. This is given by:

$$R_{sqd} = 1 - \sum e_i^2 / Y_i^2 \quad \dots A.3$$

Where, Y_i represents the measured state value at the i^{th} sampling instant. A perfect model fit gives an R_{sqd} value of 1. The possible range of values for R_{sqd} is $-\infty$ to $+1$, and the closer this is to $+1$, the better the model fit to the data.

2) ERRORS PLOTS

The errors plots comprises of plots of the error sequences and their autocorrelograms. The basic statistics of the error sequence namely; the mean, standard deviation, maximum and minimum values are presented in the top right hand corner of the errors plots.

A good model fit to the data would be expected to produce an error sequence, evenly distributed about the "zero mean" line with an autocorrelogram similar to that for a white noise sequence (insignificant serial correlation coefficients at all lags eg. see Fig. 3.123).

Trends or seasonal patterns in the plot of the error sequence

indicate an inadequate system representation by the model.

The autocorrelogram of the errors gives an indication of the signals content of the errors sequence and hence, to an extent its information content. Inadequate system description would be represented by persistent significant correlations.

The model with a better fit to the data should have a mean error close to zero, a smaller standard deviation for the errors and a smaller range for the errors (as depicted by the maximum and minimum error values).

APPENDIX B

The recursive estimation algorithms used in this work are presented in this appendix.

These are:

- o The Instrumental Variable algorithm

- o The Extended Kalman filter

The instrumental variable algorithm is developed as an extension of the recursive least squares algorithm and the Extended Kalman filter as an extension to the linear Kalman filter.

B1 THE RECURSIVE INSTRUMENTAL VARIABLE ALGORITHM

The discrete-time multiple input single output system model (MISO) can be represented by:

$$A(q^{-1})y(t_k) = \sum B_j(q^{-1}) \cdot U_j(t_k) + v(t_k) \quad \dots B.1$$

Where; $U_j(t_k)$, $j = 1, 2, \dots, m$ are observations of the multiple system inputs at the k^{th} sampling instant, $y(t_k)$ is the observation of the system output at the k^{th} sampling instant, $v(t_k)$ is a sequence of zero-mean, random errors or disturbances, q^{-1} is the backward shift operator, $A(q^{-1})$ and $B_j(q^{-1})$ are polynomials of q^{-1} , of orders 1 and p_j respectively, with parameters a_j and b_{ji} to be estimated.

If the following vectors are defined:

$$\begin{aligned} \underline{z}^T(t_k) = & (y(t_{k-1}), \dots, y(t_{k-1}); U_1(t_{k-1}), \dots, U_1(t_{k-p_1}), \\ & \dots, U_m(t_{k-1}), \dots, U_m(t_{k-p_m})) \end{aligned} \quad \dots B.2$$

$$\underline{a} = (a_1, \dots, a_1, b_{11}, \dots, b_{1p_1}, \dots, b_{m1}, \dots, b_{mp_m})^T \quad \dots B.3$$

Then Equation B.1 can be represented as:

$$y(t_k) = \underline{z}^T(t_k) \cdot \underline{a} + v(t_k) \quad \dots B.4$$

The recursive least squares algorithm for the estimates $\hat{\underline{a}}$ of the parameter $\underline{a}$ is (Young et al., 1971):

$$\hat{\underline{a}}(t_k) = \hat{\underline{a}}(t_{k-1}) + \underline{K}(t_k) \cdot (y(t_k) - \underline{z}^T(t_k) \hat{\underline{a}}(t_{k-1})) \quad \dots B.5$$

$$\underline{K}(t_k) = P(t_{k-1}) \cdot \underline{z}(t_k) \cdot (1 + \underline{z}^T(t_k) P(t_{k-1}) \underline{z}(t_k))^{-1} \quad \dots B.6$$

$$P(t_k) = P(t_{k-1}) - K(t_k) \cdot \underline{z}^T(t_k) \cdot P(t_{k-1}) \quad \dots B.7$$

Where;

$$P(t_k) = \left[\sum_{j=1}^k \underline{z}(t_j) \cdot \underline{z}^T(t_j) \right]^{-1} \quad \dots B.8$$

The recursive least squares algorithm suffers from the major restriction that there can arise problems of bias in the parameter estimates because Equation B.1 is a "structural model" (Kendall and Stuart, 1961). The recursive instrumental variable was developed to overcome this problem (Young et al., 1971).

Given suitable estimate of the parameters in the $A(q^{-1})$ and $B_j(q^{-1})$ polynomials, denoted by $\hat{A}(q^{-1})$ and $\hat{B}_j(q^{-1})$ respectively (eg. those derived from a least squares estimation), a time series $\hat{x}(t_k)$ can be computed by:

$$\hat{x}(t_k) = (1 - \hat{A}(q^{-1})) \cdot \hat{x}(t_k) + \sum_{j=1}^m \hat{B}_j(q^{-1}) \cdot u_j(t_k) \quad \dots B.9$$

$\hat{x}(t_k)$ corresponds to an estimate of the hypothetical noise-free output response of the system chosen to be totally uncorrelated with the noise $v(t_k)$. This is customarily called the sequence of "Instrumental Variables" (IV) and Equation B.9, the "Auxiliary Model".

The IV vector is then defined as:

$$\begin{aligned}\hat{x}^T(t_k) = & (\hat{x}(t_{k-1}), \dots, \hat{x}(t_{k-1}); U_1(t_{k-1}), \dots, U_1(t_{k-p_1}), \\ & \dots, U_m(t_{k-1}), \dots, U_m(t_{k-p_m}))\end{aligned}\quad \dots B.10$$

and the recursive instrumental variable algorithm is given by:

$$\hat{a}(t_k) = \hat{a}(t_{k-1}) + K(t_k) \cdot (y(t_k) - z^T(t_k) \hat{a}(t_{k-1})) \quad \dots B.11$$

$$K(t_k) = \hat{P}(t_{k-1}) \cdot \hat{x}(t_k) \cdot (1 + z^T(t_k) \hat{P}(t_{k-1}) \hat{x}(t_k))^{-1} \quad \dots B.12$$

$$\hat{P}(t_k) = \hat{P}(t_{k-1}) - K(t_k) \cdot z^T(t_k) \cdot \hat{P}(t_{k-1}) \quad \dots B.13$$

Where;

$$\hat{P}(t_k) = \left[\sum_{j=1}^k \hat{x}(t_j) \cdot z^T(t_j) \right]^{-1} \quad \dots B.14$$

As the estimation proceeds, the estimates of the parameter vector improves and the auxiliary model is updated in order to enhance the quality of the instrumental variables $x(t_k)$ and improve the statistical efficiency of the estimates.

This updating can be carried out either:

- o On-line with the recursive estimates used for continuous update of the auxiliary model parameters or,
- o Off-line with repeated runs through the block of data with the auxiliary model estimates updated only after each run is finished.

The iterative procedure is considered complete when further iterations yield negligible change in the estimates.

The recursive least squares and IV algorithms were developed

along the basic assumption that the parameters a_i and b_{ji} in the $A(q^{-1})$ and $B_j(q^{-1})$ polynomials of Equation B.1 are constant. In order to investigate the trajectories of possible time varying parameters, the following random walk model is usually assumed for the parameter vector $\underline{a}$:

$$\underline{a}(t_k) = \underline{a}(t_{k-1}) + \underline{\varepsilon}(t_{k-1}) \quad \dots B.15$$

Where $\underline{\varepsilon}(t_k)$ is a vector of zero-mean white noise sequences with covariance matrix;

$$Q = E(\underline{\varepsilon}(t_k) \cdot \underline{\varepsilon}^T(t_k)) \quad \dots B.16$$

$E(.)$ is the expectation operator.

The IV algorithms for the recursive estimation of the parameters is then given by:

PREDICTION

$$\hat{\underline{a}}(t_k/t_{k-1}) = \hat{\underline{a}}(t_{k-1}) \quad \dots B.17$$

$$P(t_k/t_{k-1}) = P(t_{k-1}) + Q \quad \dots B.18$$

CORRECTION

$$\hat{\underline{a}}(t_k/t_k) = \hat{\underline{a}}(t_k/t_{k-1}) + \underline{K}(t_k) \cdot (y(t_k) - \underline{z}^T(t_k) \hat{\underline{a}}(t_k/t_{k-1})) \quad \dots B.19$$

$$\underline{K}(t_k) = P(t_k/t_{k-1}) \hat{\underline{x}}(t_k) \cdot (1 + \underline{z}^T(t_k) P(t_k/t_{k-1}) \hat{\underline{x}}(t_k))^{-1} \quad \dots B.20$$

$$P(t_k/t_k) = P(t_k/t_{k-1}) - K(t_k) \cdot z^T(t_k) \cdot P(t_k/t_{k-1})$$

.. 8.21

Here, t_k/t_{k-1} denotes the prediction at time t_k on the basis of all measured information up to and including that available at the last sampling instant, t_{k-1} and t_k/t_k represents the updated estimate on the basis of measurement information acquired at time t_k .

B2 THE KALMAN FILTER AND EXTENDED KALMAN FILTER

Assuming the state $\underline{x}(\cdot)$ of a system being forced by the control variable $U(\cdot)$, can be represented by the following discrete-discrete model;

SYSTEMS EQUATION

$$\underline{x}(t_{k+1}) = A(t_k) \cdot \underline{x}(t_k) + B(t_k) \cdot U(t_k) + \underline{w}(t_k) \quad \dots R.22$$

MEASUREMENT EQUATION

$$\underline{y}(t_k) = H(t_k) \cdot \underline{x}(t_k) + \underline{v}(t_k) \quad \dots R.23$$

where $\underline{w}(t_k)$ and $\underline{v}(t_k)$ are white noise sequences with covariance matrices $Q(t_k)$ and $R(t_k)$ respectively; $A(\cdot)$ is the state transition matrix; $H(\cdot)$ is the measurement matrix; $B(\cdot)$ is the control matrix and $\underline{y}$ is the vector of measurements made at a given time instant.

If the initial conditions are given by:

$$E(\underline{x}(t_0)) = \hat{\underline{x}}_0 \text{ and } \text{Var}(\underline{x}(t_0)) = P_0 \quad \dots R.24$$

and the processes $\underline{v}(\cdot)$ and $\underline{w}(\cdot)$ are assumed to be independent, (ie. $E(\underline{w}(t_k) \cdot \underline{v}^T(t_j)) = 0$ for all j and k , then the optimal one-step-ahead prediction $\hat{\underline{x}}(t_{k+1}/t_k)$ of $\underline{x}(t_{k+1})$, given the measurements $\{y(t_1), y(t_2) \dots y(t_j)\}$, equals (Maybeck, 1979):

PREDICTION

$$\hat{\underline{x}}(t_{k+1}/t_k) = A(t_k)\hat{\underline{x}}(t_k/t_k) + B(t_k).U(t_k) \quad \dots B.25$$

with covariance matrix

$$P(t_{k+1}/t_k) = A(t_k).P(t_k/t_k).A^T(t_k) + Q(t_k) \quad \dots B.26$$

At time t_{k+1} when the observation $y(t_{k+1})$ becomes available, the one-step-ahead prediction can be updated:

UPDATE

$$\begin{aligned} \hat{\underline{x}}(t_{k+1}/t_{k+1}) &= \hat{\underline{x}}(t_{k+1}/t_k) + K(t_{k+1}).(y(t_{k+1}) - \\ &\quad H(t_{k+1}).\hat{\underline{x}}(t_{k+1}/t_k)) \end{aligned} \quad \dots B.27$$

with covariance matrix

$$P(t_{k+1}/t_{k+1}) = (I - K(t_{k+1}).H(t_{k+1})).P(t_{k+1}/t_k) \quad \dots B.28$$

The Kalman gain is given by:

$$\begin{aligned} K(t_{k+1}) &= \\ P(t_{k+1}/t_k)H^T(t_{k+1}).(H(t_{k+1}).P(t_{k+1}/t_k)H^T(t_{k+1}) + R(t_{k+1}))^{-1} \end{aligned} \quad \dots B.29$$

Given the initial conditions, (B.24), Equations B.25 to B.29 can be solved for each time step recursively.

THE EKF ALGORITHMS

In the non-linear case where the state $\underline{x}(t)$ of the system at any time t , is well modelled by the stochastic differential equation:

$$d\underline{x}(t) = f(\underline{x}(t), \underline{u}(t), t).dt + d\underline{B}(t) \quad \dots B.30$$

where $f(\dots)$ is a known n -vector of functions of three arguments, $\underline{u}(\cdot)$ is a p -vector of deterministic input/control functions, and $d\underline{B}(\cdot)$ is a Brownian motion with diffusion $\underline{Q}(t)$ for all $t \in T$.

Written in white noise notation:

$$\dot{\underline{x}}(t) = f(\underline{x}(t), \underline{u}(t), t) + \underline{w}(t) \quad \dots B.31$$

where $\underline{w}(\cdot)$ is a zero-mean white Gaussian noise process with covariance kernel:

$$E(\underline{w}(t), \underline{w}^T(t+\tau)) = \underline{Q}(t)d(\tau) \quad \dots B.32$$

The dynamic driving noise $\underline{B}(\cdot)$ or $\underline{w}(\cdot)$ is assumed to be additive and has no structural components.

In order to exploit linear estimation theory, the initial condition $\underline{x}(t_0)$ is assumed to be a Gaussian random n -vector with mean $\underline{x}_0$ and covariance $\underline{P}_0$.

If the discrete-time measurements can be modelled in general as a known nonlinear function of the state plus linearly additive noise corruption for all $t_k \in T$ as:

$$y(t_k) = h(\underline{x}(t_k), t_k) + \underline{v}(t_k) \quad \dots B.33$$

where $h(\dots)$ is a known m -vector of functions of the state and time, and $v(\dots)$ is a white Gaussian noise sequence of mean zero and covariance kernel:

$$E(\underline{v}(t_k), \underline{v}^T(t_j)) = \begin{cases} R(t_k) & t_k = t_j \\ 0 & t_k \neq t_j \end{cases} \quad \dots B.34$$

Linearising Eqs. B.31 and B.33 about a nominal reference trajectory $x_n(\cdot)$ by means of a Taylor series expansion and assuming that the perturbations about the reference trajectories are small results in:

$$d\dot{x}(t) = F(t; \underline{x}_n(t)) \cdot dx(t) + \underline{u}(t) \quad \dots B.35$$

where

$$F(t; \underline{x}_n(t)) \triangleq \left. \frac{df}{dx}(x, u(t), t) \right|_{x=x_n(t)} \quad \dots B.36$$

and;

$$dz(t_k) = H(t_k; x_n(t_k)) \cdot dx(t_k) + \underline{v}(t_k) \quad \dots B.37$$

where

$$H(t_k; x_n(t_k)) \triangleq \left. \frac{dh}{dx}(x, t_k) \right|_{x=x_n(t_k)} \quad \dots B.38$$

The output of such a filter would be the optimal estimate of $dx(t)$ for all $t \in T$ denoted as $d\hat{x}(t)$ thus:

$$\hat{x}(t) \doteq x_n(t) + d\hat{x}(t) \quad \dots B.39$$

The extended Kalman filter measurement update incorporates the measurement $y(t_k)$ by means of:

$$K(t_k) = P(t_k/t_{k-1})H^T(.) \cdot (H(.)P(t_k/t_{k-1})H^T(.) + R(t_k))^{-1} \quad \dots B.40$$

$$\hat{x}(t_k) = \hat{x}(t_k/t_{k-1}) + K(t_k) \cdot (y(t_k) - h(\hat{x}(t_k/t_{k-1}), t_k)) \quad \dots B.41$$

$$P(t_k) = P(t_k/t_{k-1}) - K(t_k) \cdot H(.) \cdot P(t_k/t_{k-1}) \quad \dots B.42$$

$$H(.) = H(t_k; \hat{x}(t_k/t_{k-1})) = \left. \frac{dh}{dx}(x, t) \right|_{x=\hat{x}(t_k/t_{k-1})} \quad \dots B.43$$

The estimate is propagated forward to the next sampling time t_{k+1} by integrating:

$$x(T/t_k) = \int f(x(T/t_k), U(T), T) \quad \dots B.44$$

$$P(T/t_k) = \int F(T, x(T/t_k)) \cdot P(T/t_k) + P(T/t_k) \cdot F(T, x(T/t_k)) + Q(T) \quad \dots B.45$$

where;

$$F(\tau, \hat{x}(\tau/t_k)) = \left. \frac{df}{dx}(x, u(\tau), \tau) \right|_{x=\hat{x}(\tau/t_k)} \quad \dots B.46$$