FISH FARM EFFLUENT CONTROL AND

THE DEVELOPMENT OF AN EXPERT SYSTEM

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Fish Farm Effluent Control and the Development of an Expert System

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SYNOPSIS

In recent years, freshwater fish farmers have come under increasing pressure from the Water Authorities to control the quality of their farm effluents. This project aimed to investigate methods of treating aquacultural effluent in an efficient and cost-effective manner, and to incorporate the knowledge gained into an Expert System which could then be used in an advice service to farmers.

From the results of this research it was established that sedimentation and the use of low pollution diets are the only cost effective methods of controlling the quality of fish farm effluents. Settlement has been extensively investigated and it was found that the removal of suspended solids in a settlement pond is only likely to be effective if the inlet solids concentration is in excess of 8 mg/litre. The probability of good settlement can be enhanced by keeping the ratio of length/retention time (a form of mean fluid velocity) below 4.0 metres/minute. The removal of BOD requires inlet solids concentrations in excess of 20 mg/litre to be effective, and this is seldom attained on commercial fish farms. Settlement, generally, does not remove appreciable quantities of ammonia from effluents, but algae can absorb ammonia by nutrient uptake under certain conditions. The use of low pollution, high performance diets gives pollutant yields which are low when compared with published figures obtained by many previous workers.

Two Expert Systems were constructed, both of which diagnose possible causes of poor effluent quality on fish farms and suggest solutions. The first system uses knowledge gained from a literature review and the second employs the knowledge obtained from this project's experimental work.

Consent details for over 100 fish farms were obtained from the public registers kept by the Water Authorities. Large variations in policy from one Authority to the next were found. These data have been compiled in a computer file for ease of comparison.

Keywords: Fish farm, effluent, Expert System.

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CHAPTER 1

INTRODUCTION

1.1 Project Definition

The aim of this project was to provide a service for fish farmers which would enable them to minimise the effect of effluent quality constraints on the overall optimisation of farm production. This objective was pursued by:

(i) Obtaining as much information as possible (by experimental investigations and from other literature) regarding the performance and cost-effectiveness of currently available pollution abatement technology in application to aquaculture.

(ii) Using the above information to construct a "knowledge base" for an Expert System. It is hoped that this may be used by farmers via British Telecom's "Videotex" network in the future.

The project involved the collaboration of several departments at Aston University, and also BP Nutrition (UK) Ltd, who are a major supplier of commercial fish diets in the UK, and are associated with companies supplying fish diets throughout much of Europe. This liaison between Aston and BP was organised by Aston's Interdisciplinary Higher Degrees Scheme. Following the successful completion of this project, BP Nutrition (UK) Ltd should benefit through the alleviation of a major constraint on their customers' productivity, and fish farmers will have access to a technology which can prescribe for individual farms the most cost-effective way of maintaining satisfactory effluent quality.

1.2 Project Context

As mentioned earlier, BP Nutrition (UK) Ltd are a major manufacturer of commercial fish diets in the UK. Together with its sister companies in continental Europe it forms BP Nutrition which is Europe's largest supplier of such feeds. In addition, BP Nutrition manufactures other speciality animal feeds like pet food and zoo animal diets. The turnover of this company in 1987 was approximately £1840m.

As well as selling fish feeds, BP Nutrition (UK) Ltd also assists its customers with management problems on their farms. This activity is closely linked with the company's sales function. The principal advantages accruing to the company in providing such a service are:

(i) The more successful the company's customers are, the more fish they can rear and hence the more of the company's feed they will purchase.

(ii) Solving a customer's problems raises the company's esteem, and thus further encourages the customer to buy the company's products.

A decade ago, fish farming was a very small industry. Farms were small and few in number and so collectively did not pose much of a threat to the quality of our inland waterways. Aquaculture, however, has undergone a rapid increase in scale of operations in the last few years; the attendant pollution hazards are now considerable. All economic factors favour this increase to continue, but Water Authorities can penalise fish farms or any sources of industrial discharge that contravene pollution legislation, and if river quality deteriorates, new farms may not be permitted to discharge their effluents into inland waterways. This would effectively prevent new trout and salmon smolt farms from starting production.

The treatment of fish farm effluent had been studied relatively little at the time of conception of this project. This meant that BP Nutrition (UK) Ltd could not adequately advise their customers regarding effluent control. This collaborative project was a proposed solution to this problem. The project was started in October 1984 and was completed by September 1987.

In the following chapter, the published literature concerning the effects of aquacultural discharges on the aquatic environment is reviewed. Chapter 3 reviews the UK's water pollution control legislation and procedures and, in particular, includes a survey of fish farm discharge consents in the 10 Water Authority areas in England and Wales and the 7 Scottish River Purification Board regions. Chapter 4 reviews possible treatment strategies for fish farm effluents. These include sedimentation and biofiltration. Chapter 5 documents this project's experimental work on effluent treatment whilst Chapter 6 describes the experimental work on the use of specially formulated diets to reduce pollutant output by fish. Chapter 7 documents the development of an Expert System which can advise farmers on an individual basis concerning the control of their effluents. Finally, Chapter 8 summarises this project's contribution to knowledge regarding the cost-effective control of fish farm effluents.

CHAPTER 2

THE ENVIRONMENTAL IMPACT OF INTENSIVE FRESHWATER FISH FARMING

2.1 Pollutant output from freshwater fish farms

In this section, the term "pollutant" refers to any substance or effect whose presence in aquacultural effluents is of concern to the Water Authorities. The three principal water quality parameters to be considered, therefore, are suspended solids, BOD, and ammoniacal nitrogen (NH_3-N). The pH of fish farm effluents is also monitored by the Water Authorities, but is difficult to predict from input parameters, as it is influenced by a number of factors including the concentrations of carbon dioxide and nitrogenous metabolites. Phosphorus will be considered separately because it is not regularly monitored by any Water Authority although it is of increasing concern.

Several authors have published pollutant output figures for freshwater fish culture systems (Table 2.1.1). These data were derived either from surveys of farms or from laboratory experiments. The majority of the published figures express output in terms of mass of pollutant discharged per mass of food fed. The main drawback to this approach is that the output of pollutants varies with the proportion of food fed which has been digested by the fish. The absorption of nutrients on passage through the alimentary canal of a fish will reduce a food's pollution potential. From and Rasmussen (1984) have shown that assimilation of diet varies inversely with feeding rate. Hence, pollutant output figures will be influenced by feeding rate, a factor which is not always mentioned in the literature. This, and variations in diet composition, account for much of the scatter in the summarised data.

Some data, though, are quoted in the form of output rates based on either farm production rates or stock levels. For the purpose of comparison, the last two categories of data were converted to units of mass output per mass of feed using either an assumed food conversion ratio of 1.5 or a feeding rate of 1.5% of body weight per day respectively.

In 1982, Alabaster circulated a questionnaire to all members of the European Inland Fisheries Advisory Commission (EIFAC) requesting information regarding water flows and pollutant concentrations upstream and downstream of fish farms. It also considered dissolved oxygen consumption and the use of chemicals on farms. From the published summary of the data derived from the correspondents in 15 countries, pollutant output figures per 100 kg food fed were calculated using an assumed Food Conversion Ratio (FCR) of 1.5. Great variations were found in the output figures for the different European countries, with Italy providing figures of doubtful accuracy, with suspended solid outputs well in excess of food inputs. The mean of all of the observations was a very high suspended solids output figure of 3.4 kg/day per tonne/annum production which corresponds with 82.7 kg/100 kg feed at an assumed FCR value of 1.5. The mean BOD output figure was also fairly high at 41.4 kg/100 kg feed, whilst ammoniacal nitrogen output was very high at 7.3 kg/100 kg feed. No reason for these high pollutant output figures was given. The data covered all types of freshwater farmed species, although rainbow trout farms constituted the majority of the systems included in this study.

Solbé (1982) circulated a similar questionnaire to the 10 Water

Authorities in England and Wales, the 7 Scottish River Purification Boards, and the Department of the Environment for Northern Ireland. It requested water quality information on UK freshwater fish farms. Data were obtained for 141 of the 470 freshwater fish farms in operation in 1980. As with the Alabaster survey, very high solids outputs were recorded, with an average figure of 1350 kg/tonne of fish produced. This corresponds to a percentage figure of 90% of food fed at an assumed FCR of 1.5. Again, no reason for this very high figure was apparent or suggested. Solbe's BOD and ammonia output figures, however, were much more in line with other literature at 19.0 and 3.7 kg/100 kg feed respectively (see Table 2.1.1).

Warrer-Hansen (1982a) quoted figures based on several investigations carried out by the Danish Water Quality Institute during the period 1976-78. These data were presented in two sections. The first described pollutant output as a proportion of rainbow trout biomass in Danish earth pond systems. This may be converted to a percentage of food fed using an assumed feeding rate of 1.5% of body mass per day. Such derivation resulted in a moderate suspended solids production of 33.3 kg/100 kg feed, a low BOD output of 12.5 kg/100 kg feed, and an extremely low ammonia output of 0.83% of food fed. Again, no reason was provided for these exceptional results. Data for trout fed on minced trash fish as opposed to dry pelleted diets were also included in this study. These showed a relatively high pollutant output, with suspended solids, BOD and ammonia outputs being 170%, 260% and 430% higher than the corresponding values obtained from fish fed on dry diets. Greater phosphate outputs were also recorded.

The second section of Warrer-Hansen's (1982a) data gave annual data discharge outputs for a production of 100 tonnes of rainbow trout fed with dry feed. Conversion of these values, again using an assumed FCR of 1.5, yielded a similar solids output to those obtained in the first section of this author's work, however the BOD and ammoniacal nitrogen outputs were much higher at 23.3 and 3.0 kg/100 kg feed respectively. No explanation was offered for these discrepancies.

Bergheim and Selmer-Olsen (1982) conducted a detailed study of the output of BOD from a large trout farm in Norway and found that these ranged from 22.9 to 28.8 kg/100 kg feed. Liao (1970) assessed the pollutant production of a US trout hatchery and reported high solids and BOD output values of 52.0 and 60.0 kg/100 kg feed respectively. The ammonia output value of 2.9 kg/100 kg feed was more consistent with other worker's results. Willoughby et al. (1972) quotes much lower values of 30.0, 34.0 and 3.2 kg/100 kg feed for suspended solids, BOD, and ammonia output respectively from US trout hatcheries. They did not, however, provide any source for these data.

Fauré (1977) reported high solids and BOD output levels of 45.0 and 52.0 kg/100 kg feed respectively. These data were obtained from a pilot study carried out at the Centre Technique du Genie Rural des Eaux et Fôrets (C.T.G.R.E.F.) in Bordeaux. Fauré noted that these values lay between those of Liao (1970) and Willoughby et al. (1972) and attributed the differences to experimental error.

Klontz et al. (1978) obtained the following expression relating suspended solids output to FCR from their work on a US trout hatchery:

$$\text{SS output (kg/100 kg feed)} = 95\ (1 - \frac{1}{FCR})$$

Assuming an FCR value of 1.5, this yields a solids output of 31.7 kg/100 kg feed; a value which is similar to much of the other published data.

Quellerou et al. (1982) studied the effluents from three rainbow trout farms in Brittany and obtained mean outputs of 38.2, 12.1 and 3.1 kg/100 kg feed for suspended solids, BOD, and ammoniacal nitrogen respectively. They also modified the suspended solids production equation of Klontz et al. to fit their data as their low FCR values yielded aberrant solids production values when applied to this equation. Their equation is as follows:

$$\text{SS production (kg)} = (1 - Kd)\ (33\ FCR - 20)\ \frac{f}{100}$$

where Kd = settlement factor (Dimensionless)

FCR = Food Conversion Ratio

f = Food fed (kg)

The settlement factor, Kd, corresponds to the proportion of solids produced which settles in the rearing tanks.

Harman (1978) investigated the output of polluting metabolites on an experimental rainbow trout farm which used a very low water throughput with pure oxygen aeration. He obtained reasonable BOD and ammonia outputs of 14.0 and 3.35 kg/100 kg feed respectively, but an inexplicably high solids output of 65.0 kg/100 kg feed.

Brett and Zala (1975) studied the diurnal patterns of nitrogen excretion (as ammonia and urea) and oxygen consumption in fingerling sockeye salmon (Onchorynchus nerka) fed at a ration of 3% of body mass per day. Ammonia output was found to peak 4-4½ hrs after feeding and averaged 1.16 kg/100 kg food fed. Urea output was found to be much lower and also fairly constant at 0.17 kg/100 kg feed. If parallels are drawn with juvenile rainbow trout, then the relatively low value for ammonia output suggests that the faster growth rates of juvenile fish result in a greater proportion of the digested protein being used for growth purposes and a consequently smaller proportion being used as an energy source with ammonia being the principal excretory by-product.

The 1987 meeting of the EIFAC Working Party on Fish Farm Effluents concentrated on suspended solids output (EIFAC, 1987). At this meeting, the results of a recent survey of suspended solids output from fish farms in EIFAC countries were presented; they had a mean output of 36.5 kg/100 kg food fed. It was noted that this value was much lower than those obtained by Alabaster (1982) and Solbé (1982). This was attributed to improvements in the quality of pelleted fish diets. It would seem unlikely, however, that advancements in diet manufacture alone, could account for the 59% drop from Solbé's 1982 figure to the 1987 EIFAC figure. No other explanation was provided, however, Black (Bromage - personal communication, 1986) measured the suspended solids output from rainbow trout reared on a UK broodstock trout farm. He studied four groups of trout with mean weights of 4.5, 15.0, 270.0 and 1100.0 grammes, and found solids output to be 23.0 kg/100 kg feed for all four groups. This is low in comparison with other reported data but again no explanation was offered. The level reported by Black represents the mean of 32 observations with solids output values ranging from 7.1 to 37.4 kg/100 kg feed. The fact that such variation exists begs the question as to the reliability of the assay procedures which yielded these and other data. One of the problems with this study was that no allowance was made for the diurnal variability of suspended solids output. Black's results were based on effluent samples taken from trout rearing tanks at unspecified times relative to feeding. The brief spells of vigorous fish activity during feeding were associated with much higher effluent solids output than at other times (this is established in Chapter 6 of this thesis). Unless these solids output peaks around feeding times can be quantified in some way, overall mean output figures are of limited value.

In conclusion, a large degree of unaccountable variation exists in the reported output values for suspended solids, BOD, and ammoniacal nitrogen (Table 2.1.1). Much of this is probably due to inaccuracies in the methods of effluent sampling. It is difficult to obtain manageably small yet representative samples of daily effluent flows of millions of gallons when fish metabolite outputs vary diurnally. Fairly narrow modal output ranges, however, can be derived from the data obtained. Suspended solid outputs are typically between 30 and 40 kg/100 kg food fed for dry diets, whilst the equivalent ranges for BOD and ammonia output are 15-30 and 3-4 kg/100 kg food fed respectively.

The data are summarised below. All figures are in kg of pollutant output/100 kg feed.

Table 2.1.1 Pollutant Output in Intensive Fish Culture

(Kg output per 100 Kg feed)

Reference	S Solids	BOD	NH_3-N	Notes
Alabaster (1982)	82.73	41.37	7.3	Assumed FCR=1.5 European freshwater
Solbé (1982)	90.00	19.00	3.7	Assumed FCR=1.5 All UK freshwater
Warrer-Hansen (1982)	36.67	23.33	3.0	Assumed FCR=1.5 Danish freshwater
" (")	33.33	12.50	0.83	Assumed 1.5% feeding Danish earth ponds
Bergheim and Selmer-Olsen (1982)	-	22.9-28.8	-	Earth ponds Trout, Norway
Liao (1970)	52.00	60.00	2.9	US Hatchery, Trout
Willoughby *et al* (1972)	30.00	34.00	3.2	US Hatchery, Trout
Fauré (1977)	45.00	52.00	-	
Klontz *et al* (1978)	31.67	-	-	Assumed FCR=1.5 US Hatchery
Quellerou *et al* (1982)	38.20	12.10	3.1	French raceways Trout
Harman (1978)	65.00	14.00	3.35	UK Intensive Trout Tanks
Brett and Zala (1975)	-	-	1.16	Juvenile Sockeye salmon fed at 3% body wt/day
EIFAC (1987)	36.5	-	-	UK assuming FCR=1.5
Black (1986)	23.0	-	-	UK trout farm

In addition to suspended solids, BOD and ammonia, phosphorus pollution is an area of increasing importance in Europe and in Finland it is considered to be of greater environmental signficance than BOD. It is estimated (Sumari, 1982) that about 6 tonnes/annum of phosphorus is discharged, in the form of organic and inorganic compounds, by Finland's fish farms. This is out of all proportion with the economic significance of the industry. In the UK, Solbé (1982) estimated that 15.7 kg total phosphorus was discharged by fish farms per tonne of fish produced. Phosphorus discharges in aquaculture, however, are not yet controlled by the Water Authorities.

As would be expected, levels of phosphorus compounds in aquacultural effluents show a positive correlation with dietary phosphorus, the principal source of which is bone meal (Ketola, 1982). In one study (Quellerou et al., 1982) total phosphorus excretion was found to equal 7.5% of the mass of food fed, with orthophosphate phosphorus (PO_4-P) excretion equalling 1.4% of food mass. pH change is not considered to be a major effect of aquacultural effluents on rivers (Alabaster, 1982).

2.2 Environmental consequences of pollutant output

Much work has been done concerning the effects of strongly adverse water quality conditions on fish and inland fisheries (EIFAC 1964, 1968, 1970, 1973). It is in the commercial interests of fish farmers to prevent such conditions from developing in their rearing tanks and so the water quality of aquacultural effluents rarely ever approaches the limits suggested in the above papers (except during cleaning operations). The long-term effects of such dilute effluents on the ecosystems of receiving waters are less well documented and the subject of much controversy.

At the 1982 workshop of the European Inland Fisheries Advisory Commission (EIFAC) in Silkeborg, Denmark, several papers were presented which together contained the results of extensive research into the environmental effects of freshwater fish farming in the UK, and in several other European countries (see Alabaster, 1982; Mantle, 1982; Markmann, 1982; Solbé, 1982; Sumari, 1982). The main conclusions regarding the issue of the environmental effect of aquacultural discharges were:

(i) In the UK, damage to fisheries by aquacultural discharges is rare.

(ii) Biotic Index data indicate little general environmental damage caused by fish farming in the UK.

(iii) Major pollution incidents involving UK fish farms were usually the result of temporarily very high suspended solids output during cleaning operations.

(iv) Significant damage to downstream fisheries is possible if receiving water flowrate is less than about 5 litres/second (95040 gallons per day) per tonne of annual farm production.

(v) Minor problems of 'sewage fungus', low dissolved oxygen levels and deposition of solids on plants and the stream bed tend to arise where receiving water flowrate is less than about 8 litres/second per tonne of annual farm production. (Dilution would appear to be an inappropriate choice of parameter in considering sewage fungus as an increase in dilution at constant cross-sectional area would give rise to a proportional increase in velocity and so the nutrient supply to an area of sewage fungus growth fixed to the stream bed would remain unchanged. A more appropriate parameter would be the ratio of effluent outlet cross-sectional area to receiving water flow cross-sectional area).

(vi) Increased eutrophication of receiving waters by fish farm effluent is common in Europe generally, and particularly in Finland. This is attributed to discharges of inorganic phosphorus.

The term "sewage fungus" is a misleading term for the attached macroscopic growths of certain saprobic micro-organisms, principally the sheathed filamentous bacterium Sphaerotilus natans. Hawkes (1962) stated that its growth requires sources of nitrogen and phosphorus and also certain soluble organic compounds which are virtually absent in clean rivers. Webb (1985) found that cationic starches strongly promoted the growth of sewage fungus in rivers.

A major area of concern is the eutrophication of oligotrophic freshwater lakes by cage culture. Korzeniewski et al. (1982) found that a trout cage farm, whose production rate in 1980/81 was 28.9

tonnes/annum, was causing a considerable steady increase in the degree of eutrophication of a Polish lake of 4 km^2 area. Furthermore, the authors advised that such cage culture in lakes should be discontinued in Poland. Trojanowski et al. (1982) analysed the bottom sediments of the same lake and found a similar steady increase in eutrophication.

The most ecologically dangerous consequence of eutrophication is the development of colonies of toxic cyanobacteria associated with blooms of blue-green algae. These form heterocystous nitrogen-fixing groups on the algae which have been implicated in many mammal, bird, and fish poisoning cases. The most common bacterium is Anabaena flos-aquae which excretes a range of six poisons known as anatoxins. Blue-green algae frequently dominate the phytoplankton in eutrophic aquatic systems because of their lower specific maintenance energy rates allowing them to continue growth when available light is reduced by excessive growth of phytoplankton (Codd, 1985).

Aside from the specific effects of blue-green algae, all species of aquatic plant will undergo photosynthesis, producing oxygen, during daylight hours and subsequently undergo respiration, with the absorption of oxygen and production of carbon dioxide, at night. The high concentrations of algae in spring and summer in eutrophic water masses, therefore, can cause large diurnal fluctuations in dissolved oxygen and carbon dioxide concentrations.

The effluent from fish farms is, therefore, not generally harmful to most freshwater ecosystems, but it has the potential to cause considerable environmental damage if it is badly managed.

CHAPTER 3

UK WATER POLLUTION CONTROL PROCEDURE

3.1 UK water pollution control legislation

The Control of Pollution Act 1974 (Part II) empowers the Water Authorities in England and Wales and the River Purification Boards in Scotland to control discharges into rivers, lakes and coastal waters by means of licensing. Under Section 32, Subsection (1) of this Act, the discharge of any trade or sewage effluent into any of the types of "relevant water" mentioned above is illegal unless authorised by a licence, and is in accordance with any conditions to which the licence is subject.

Section 34, Subsection (4) allows the Water Authorities, in granting discharge consent licences, to impose conditions regarding any or all of the following.

(i) Design and location of effluent outlets.

(ii) The "nature, composition, temperature, volume and rate" of discharges, and periods during which they may be made.

(iii) Provision of sampling facilities.

(iv) Provision and maintenance of measurement and analysis apparatus relevant to (ii) above.

(v) Keeping of records of measurements of any parameters controlled in (ii) above.

(vi) Informing the Water Authority regarding the levels of any controlled parameter.

(vii) Taking steps to prevent discharges from coming into contact with any specified underground water.

Although this Act was passed by Parliament in 1974, most of the substantive provisions of Section II which deal with water pollution were not implemented until August 1985. This delay was largely due to concern by the Department of the Environment that the implementation of this Act, unless conducted very gradually and carefully, could lead to great increases in costs for both the Water Authorities and industry (Matthews, 1987).

Section II of the Control of Pollution Act 1974 supersedes the Rivers (Prevention of Pollution) Acts 1951 and 1961 and the Clean Rivers (Estuaries and Tidal Waters) Act 1960 in England and Wales, and the Rivers (Prevention of Pollution) (Scotland) Acts 1951 and 1965 in Scotland. The main aims of the new Act were to generally increase the scope of, and to increase public involvement in, water pollution control. The latter aim is pursued by introducing requirements for more extensive advertising of discharge proposals and increasing the powers of local authorities with respect to water pollution control. Also, any member of the public can conduct a private prosecution under the 1974 Act whereas the previous legislation in the 1961 Act required all prosecutions to be referred to the Attorney General.

The initial fears surrounding the Act's implementation have to some extent been borne out. Increasing the powers of the Water Authorities has brought new types of discharge (e.g. discharges at sea) under their jurisdiction. These require new control practices and in many cases the result has been an increase in Authority workload with minimal environmental benefit (Matthews, 1987). The proposed privatisation of the Water Authorities in 1989 will result in the responsibility for water pollution control being taken over by a National Rivers Authority.

Much disagreement has existed for some time between the UK and EEC Parliaments concerning environmental policy. This is particularly true of water pollution control legislation. At the heart of the dispute is EC Directive 76/464. One of the main aims of this legislation is to set, for the whole community, limit values for recognised pollutants, above which no discharge consent limit must be set. The UK alone has opposed this policy since it was first proposed in 1975, but has gradually complied with most other parts of the Directive. This has been accomplished mainly through the implementation of the Control of Pollution Act 1974. Most significantly, river quality objectives were introduced in 1978. (Haigh, 1984). It is also proposed that the current National Water Council/Department of the Environment classification be modified to take account of EEC directives (Mance, 1986).

3.2 Water Authority Policy

The way in which a Water Authority interprets Parliamentary Legislation when dealing with, for example, a fish farm depends on a number of factors specific to that case.

The principal factors are:

(i) The NWC/DoE water quality class of the receiving water (see table 3.2.1).

(ii) The objective water quality class of the receiving water.

(iii) The mean, and likely minimum, receiving water flowrate.

(iv) Pressure from influential groups of people who use the receiving water for recreational purposes (eg anglers).

The objective water quality class (ii) is set after consideration of the watercourse's social, recreational and industrial value, and the potential for its causing a nuisance if its water quality were poor (STWA, 1984).

The above factors determine:

(i) The severity of the licensing conditions imposed.

(ii) The extent to which the Authority will, in practice, tolerate minor or temporary breaches of the licensing conditions.

In past experience (McLoughlin and Forster, 1982a), the decision to prosecute has usually been taken as a result of one or more of the following:

Table 3.2.1 NWC/DoE Water quality classification

The following water quality classification scheme was originally proposed by the NWC and was subsequently adopted by the DOE.

Limiting Criteria	Class				
	1A	1B	2	3	4
Dissolved Oxygen (%sat^n)	>80	>60	>40	>10	<10
BOD_5 (mg/l)	<3.0	<5.0	<9.0	<17.0	>17.0
Total NH_3-N (mg/l)	<0.4	<0.9	-	-	-

All limiting criteria are 95 percentile.

Class	Description
1A	High quality water suitable for potable supply abstractions and capable of supporting game or other high class fisheries.
1B	Water of less high quality than class 1A, but usable for substantially the same purposes.
2	Water suitable for potable supply after advanced treatment, and capable of supporting reasonably good coarse fisheries.
3	Polluted to such an extent that fish are either absent or only sporadically present. May be used for low grade industrial abstraction.
4	Grossly polluted and likely to cause a nuisance.

(i) Persistent breaches of licensing conditions despite warnings.

(ii) Negligence or indifference despite warnings.

(iii) Deliberate acts causing breaches of licensing conditions.

(iv) Breaches of licensing conditions having serious consequences in terms of pollution.

Recent Government spending cutbacks have made it more difficult for the Water Authorities to follow up incidents of breach of licensing conditions than in the past. Informal comment suggests that the Authorities now concentrate mainly on the more severe cases of contravention of water pollution law, and most dischargers are unlikely even to receive a warning unless their consent conditions are consistently being breached to a considerable degree.

3.3 Fish Farm Discharge Consents

In August 1985, Section 41 of the Control of Pollution Act 1974 was implemented. This allows any member of the public access to details of all discharge consents issued by any Water Authority or River Purification Board (RPB). In order to study the way in which the authorities control aquacultural discharges, a list of representative freshwater fish farms was prepared for every Water Authority in England and Wales, and every River Purification Board in Scotland. For most Authorities, the number of farms listed was at least ten, but in some cases the number of sizeable freshwater fish farms which could be located was considerably less than this. These lists were sent to the offices of the appropriate authorities, and consent details were obtained from every Water Authority in England and Wales and every River Purification Board in Scotland. Of the Scottish RPB's, however, only four out of the seven set limits to fish metabolite concentrations for the farms listed, and the Highland RPB did not impose any consent conditions on the farms listed as they were all found to be sea cage sites for which the Board's consent was not required.

Policy regarding the control of aquacultural effluents was found to vary greatly from one Authority to the next. The principal water quality parameters controlled are:

(i) Ammoniacal Nitrogen (NH_3-N).

(ii) 5 day Biochemical Oxygen Demand (BOD) with nitrification suppressed using allylthiourea (BOD_5ATU).

(iii) Total Suspended Matter dried at 105,C (SS 105,C).

These methods are described in Chapter 5.

In addition, a limit is usually imposed on the volumetric flowrate of an effluent.

Table 3.3.1 - Water Quality Parameters controlled in Fish Farm Discharge Consents

	Authority											RPB		
Parameter	ST	W	SW	S	T	A	Y	N	WL	NW	SY	TW	TY	NE
Ammoniacal-N	*	*	*	*	*	*	*	*	*	*	*		*	*
BOD_5(ATU)	*	*		*	*	*	*	*	*		*			
Suspended Solids	*	*	*	*	*	*	*	*	*	*	*	*	*	*
Volume Flow	*	*	*	*	*	*	*	*	*	*	*	*	*	*
BOD_5(NON-ATU)	*		*				*		*	*			*	*
Dissolved O_2		*	*	*		*	*				*	*	*	
pH		*	*	*	*			*	*	*	*			*
Formaldehyde	*	*							*		*			
Malachite Green		*		*					*	*	*			
Free Chlorine		*								*	*			
Total Copper		*		*							*			
Total Phenols		*		*							*			
Colour		*												
Turbidity		*		*										
Cationic Detergents		*									*			
Antibiotics		*							*					
Oils (Visible)		*		*		*					*			*
Promotion of Sewage Fungus downstream		*									*			
Any chemical in a "toxic" concentration				*	*			*	*			*	*	*
Temperature									*					
Total Number	6	17	6	12	6	6	6	6	11	7	14	4	6	6

N.B. Forth and Clyde River Purification Boards only set limits on volumetric flowrate of effluent.

Key to abbreviations on Table 3.3.1 and numbers on Figs. 3.3.2 - 3.3.5

(1)	ST	Severn-Trent	(8)	N	Northumbrian
(2)	W	Wessex	(9)	WL	Welsh
(3)	SW	South West	(10)	NW	North West
(4)	S	Southern	(11)	SY	Solway
(5)	T	Thames	(12)	TW	Tweed
(6)	A	Anglian	(13)	TY	Tay
(7)	Y	Yorkshire	(14)	NE	North East

Consent conditions are also imposed concerning a wide range of other parameters, with great variations in policy among the different authorities. This is illustrated in Table 3.3.1. From this table it can be seen that the Wessex Water Authority is concerned with more water quality parameters than any other authority. The Tweed River Purification Board, on the other hand, only considers four.

The severity of the consent conditions imposed regarding these parameters also varies greatly among the different Water Authorities and River Purification Boards. This is illustrated by Figs. 3.3.2, 3.3.3 and 3.3.4, and more comprehensively in Appendix A. These show the mean consent maximum concentrations for suspended solids, BOD, and ammoniacal nitrogen respectively for the 14 authorities listed above. From these histograms it is obvious that the Anglian Water Authority is the most lenient with respect to aquacultural discharges in England and Wales, whilst the Yorkshire Water Authority is the strictest in terms of these three pollutants. In Scotland, the Solway River Purification Board appears to be the strictest, whilst the North East RPB is the most lenient of the RPB's which impose quantitative conditions regarding these three parameters.

The percentage of the farms listed in each area which were operating without consents is shown in Fig. 3.3.5. From this histogram it appears that the urgency with which the different Authorities strive to control aquacultural discharges is highly variable. The Anglian and South West Water Authorities seem particularly relaxed in their attitude.

Although it is likely that a considerable proportion of this policy variation among the Authorities will be attributable to differences in "corporate personality", it can still be rationalised to a limited extent. The area controlled by Anglian Water Authority, for example, contains very few fish farms. Crop farms in this area collectively have a much greater environmental impact than do fish farms and hence the control of fish farm discharges is only a minor objective for the Anglian Water Authority. The South West Water Authority area contains more fish farms, but is relatively sparsely populated and has an abundance of freshwater rivers and streams and hence again may not be very concerned about fish farm effluents.

Wessex Water Authority area, on the other hand, has one of the highest concentrations of trout farms in the U.K., many of which are situated on high quality rivers which also support game fisheries. A similar situation exists in Yorkshire. Much of the variation in policy among the different Authorities can be similarly accounted for in terms of the concentration of fish farming activity, the total environmental effect of different areas' fish farms in relation to their other industries, and the recreational value of their watercourses.

Fig. 3.3.2

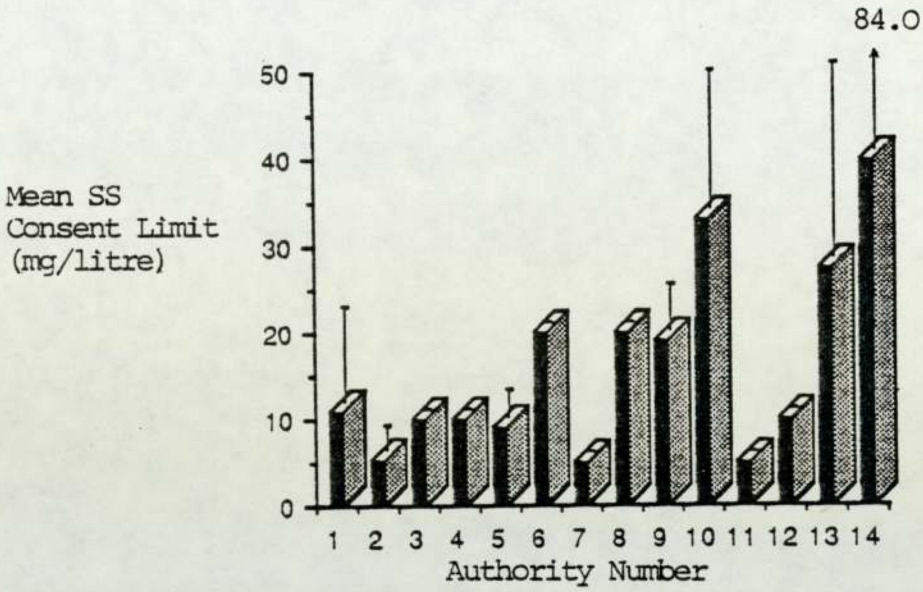
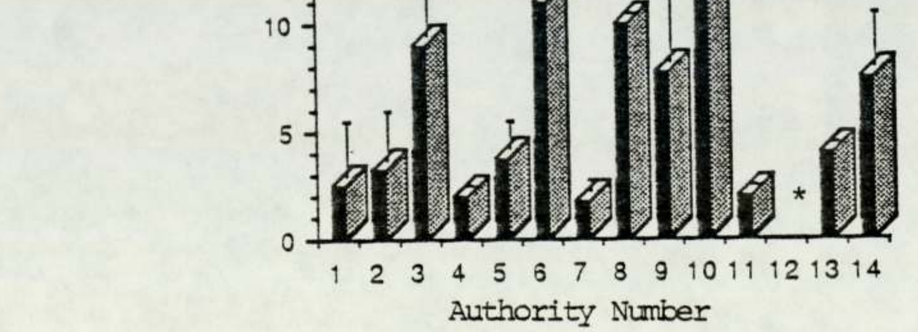


Fig. 3.3.3



* NO CONSENT LIMIT APPLIED

All figures are means +/- standard deviation

Fig. 3.3.4

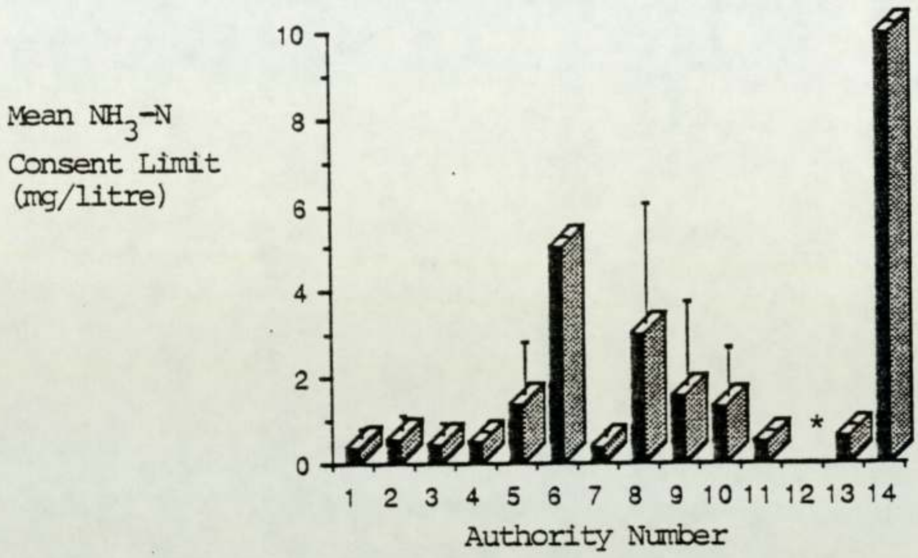
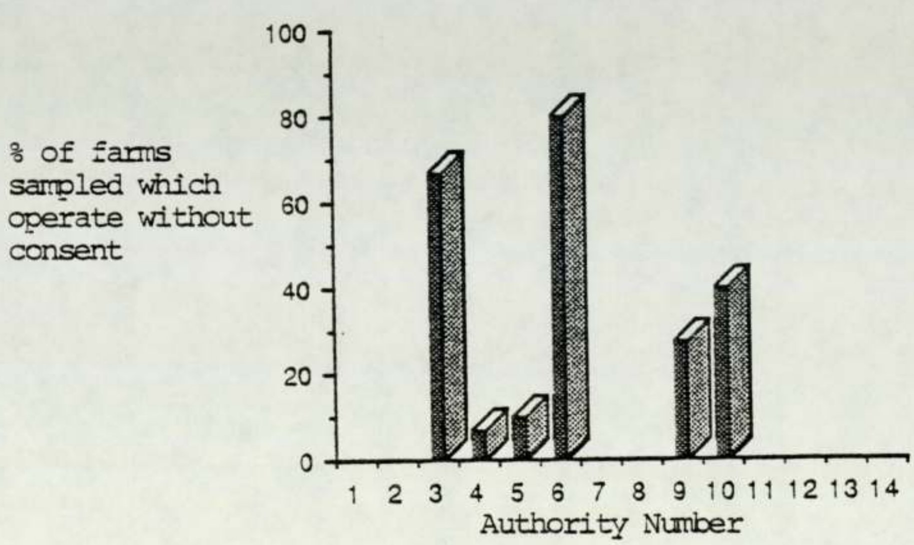


Fig. 3.3.5



* NO CONSENT LIMIT APPLIED

All figures are means +/- standard deviation

3.4 Water pollution control in other European Countries

(from McLoughlin and Forster, 1982b)

The mechanisms of water pollution control are broadly similar throughout Europe, although there are some notable exceptions. The EC directive 76/464 calls for the setting up of a community-wide river quality objectives which take account of the intended uses of different rivers, e.g. all rivers intended for use as game fisheries would have similar objective water quality in terms of major pollutants throughout the community (Haigh, 1984). Comparison of water pollution control procedures among the community member states must therefore include an assessment of the roles played by national water quality objectives in determining consent limits, as these are a necessary precursor to the setting up of community-wide objectives by means of a consensus of the member states.

In France, the water pollution control system bears close similarity to that of the UK. Discharge authorisation is managed at river catchment area level as in the UK. Authorisation may be granted subject to conditions regarding flowrate and levels of suspended solids, BOD, ammonia, nitrate and phosphate. Water quality objectives are not set nationally, but are set for individual rivers and these influence the conditions imposed on the discharger.

In West Germany a unique system of financial charges is operated. The Wastewater Charges Act (1978) empowers Germany's Water Authorities to impose a financial levy on any business concern which discharges a polluting effluent into a waterway. The charges are formulated on the basis of estimates of the amounts of suspended solids and chemical oxygen demand (COD) discharged annually. No evidence was found of any national programme of water quality objectives as recommended in EC 76/464. An obvious drawback to this system of charges is that large prosperous business concerns can afford to cause more environmental damage than others (Bohl, 1982).

In the Netherlands, the Surface Waters Pollution Act is the principal legislation for protection of inland waterways. Control is by means of a system of permits to discharge to which conditions may be attached concerning flowrate and concentrations of BOD, ammonia, COD and suspended solids. As in West Germany, no reference to water quality objectives was found.

The Belgian Water Purification Corporations authorise discharge of effluents into surface waters. Four Corporations were established in 1971, the first has jurisdiction over all coastal waters, a second covers the Scheldt basin, a third controls discharges to the Meuse, Seine and Rhine basins whilst the fourth Corporation covers the Flemish region excluding its coasts. Water Quality Objectives were established by a Royal Decree in 1976 which also legalised the imposition of conditions concerning discharge rates, sampling facilities and accidental discharge safeguards, but no reference to specific pollutant concentrations was found.

In Luxembourg a ministerial decree specifies the consent conditions for individual dischargers. The small size of the country allows the Ministry of Environment to handle the whole pollution control system down to the issuing and enforcement of consent conditions. As in most other European countries, consent conditions principally concern levels of suspended solids, ammonia and BOD. No reference to a national programme of water quality objectives was found.

Discharge of effluent into inland waterways in Italy is authorised by local communes. Precise limits on water quality parameters are specified in Italy's water pollution legislation. These limits are varied according to the intended uses of different watercourses. A discharger is not permitted to achieve compliance with conditions imposed by means of dilution with water abstracted "specifically for that purpose". Italy is therefore closer than most other EEC countries to compliance with EC 76/464 which calls for a consistent policy on consents throughout the community.

In Greece the local authorities issue permits to discharge into inland waters, but only after a study has been conducted by the applicant, the results of which must indicate the estimated quantity of discharge and the likely environmental effects. It must also suggest measures for monitoring and control of the effluent. No national programme of water quality objectives exists, but local authorities must not allow water quality in any river or stream to fall below objective limits set by regional "prefects".

The Danish Environmental Act (1974) regulates discharges to watercourses through a system of approvals, prescriptions and bans. Consent for discharges must be obtained from the regional authorities. The authorities may impose consents regarding the output of BOD, suspended solids, nitrogen and phosphorus in effluents. They also may impose limits on farm production and food used. This is currently a source of much controversy in Denmark as its fish farming community regards this as unfair (Jørgensen, 1982). No reference to national water quality objectives was found.

In Finland, EC directives are, of course, of no consequence as it is not a member of the European Community. According to Finland's "Water Law", a concession of the Water Court is required for all discharges to watercourses. Concessions are granted by this court subject to conditions concerning production levels, water flowrate, removal of dust from feed and prohibition of overfeeding, the building of settlement ponds and the removal of accumulated sludge. These conditions are enforced by the National Board of Waters. This board has published a directive on the control of fish farming activities (Sumari, 1982).

In Sweden, the Environmental Preservation Act provides for the control of industrial discharges by the regional County Administration Boards. These are empowered to issue consents to dischargers and attach conditions concerning flowrate and pollutant concentrations as in most of Europe (Lysén, 1984).

In Norway, until recently, surface water pollution was only considered a problem in a few localities. This was due to its sparse population, high rainfall and the small catchment areas of its rivers. The increase in industrial and agricultural activities, including salmon farming, and the growing awareness of the possible effects of environmental pollution over the past two decades has, however, led to the passing of "The Act on Protection against Water Pollution" in 1970, followed by "The Act Against Pollution" in 1981. This Act is administered by the Ministry of Environment but handled locally by the municipal and county authorities who operate a consents system with the imposition of flowrate and water quality conditions (Hareide, 1984).

In the Republic of Ireland, County Councils and Borough Corporations control water pollution through legislation under the Local Government (Water Pollution) Act 1977. The relevant sections of this Act bear close resemblence to those of the UK's Control of Pollution Act 1974. The Act allows local authorities to issue licences which may include conditions regarding similar criteria to those imposed by UK Water Authorities. Progress has been slow, however, and by the end of 1980 fewer than 300 licences had been issued. The Act also makes provision for the establishment of EEC style water quality objectives. By 1982, though, no water management plan had been established nationally (McLoughlin and Forster, 1982b).

The European Community, in its aim of establishing a community-wide pollution control policy, must therefore overcome considerable policy difference among its member states if it is to succeed. These differences are the inevitable result of decades of separate development, and their resolution can only be achieved through lengthy negotiation and the exchange of ideas throughout the European Community.

CHAPTER 4

EFFLUENT TREATMENT TECHNOLOGY IN AQUACULTURE

4.1 Treatment of aquacultural effluent by sedimentation

Solbé (1982) found that, in 1980, only about 17% of UK freshwater fish farms treated their effluent prior to discharge, and in almost every case in which effluent treatment took place it consisted of a simple lagoon. Harman (1978) found that, in comparison with sand filtration, microstraining and air flotation, simple sedimentation was the most cost effective way of reducing suspended solids to below 10 mg/l when treatment costs were compared with costs of increased abstraction which would have been incurred were this method used to bring about the same reduction in suspended solids concentration.

The above information strongly suggests that simple lagooning may be financially the most attractive solution to problems of high suspended solids levels in aquacultural effluents. Furthermore, several sources (Liao, 1970; Liao and Mayo, 1974; Harman, 1978; STWA, 1983) indicate that lagooning can also be a fairly efficient method for the reduction of BOD in effluents, and attribute this to the fact that much of the solid material present in most wastewaters is aerobically biodegradable. Finally, Harman (1978), Liao (1970), and Liao and Mayo (1974) have found that, under certain conditions, lagooning can significantly reduce levels of ammoniacal nitrogen in aquacultural effluents.

4.1.1 The effect of retention time on performance

The efficiency with which a simple lagoon may reduce the concentrations of these pollutants, however, is heavily dependent on several factors. Retention time has been claimed to influence lagoon performance considerably. Liao (1970) found that settlement ponds with retention times greater than about 3 days tended to remove about 60% of both suspended solids and BOD from aquacultural effluent if BOD loading rate is about 8 grammes/m^2day, and shorter retention times gave considerably poorer results. Also, a pond with a retention time of only 2-3 hours could remove about 80% of both BOD and suspended solids from raceway cleaning wastes. Liao attributed the relative ease of treatment of this type of effluent to the biodegradation of accumulated material on raceway beds during the one week periods between cleaning operations causing binding among solid particles.

Harman (1978), however, found that a lagoon with a retention time of 50.2 hours developed algal blooms in spring and summer. This resulted in net increases in BOD and suspended solids levels in the effluent passing through the pond. Toms <u>et al</u>. (1975) claimed that lagoons with retention times in excess of about 50 hours are liable to develop algal blooms. Harman found that algae also had a beneficial effect on effluent quality as they absorbed ammonia, the levels of which were otherwise largely unaffected by passage through the lagoon. Toms <u>et al</u>. did not observe any ammonia removal in ponds containing algae, however. Harman attributed this to preferential uptake of nitrate by algae, as this was present in high concentrations (>10 mg/l) in the experiments of Toms <u>et al</u>. Harman also found that pH increased markedly in the presence of algae, and values as high as 9.0 had been recorded. Algal uptake of CO_2 was suggested as the major cause of this.

4.1.2 The formation of ammonia by solids decomposition

The decomposition of settled solids in lagoons has been widely suspected as a cause of high ammonia levels in effluents, as much of the solid matter present in aquacultural and sewage effluents is nitrogenous. Continuous sludge removal is advisable, provided it is economically feasible (H. A. Hawkes - personal communication). Harman (1978) and STWA (1983) have observed net increases in ammonia levels in effluent passing through ponds, not containing significant amounts of algae or other aquatic plants, in which settled solids have been allowed to accumulate. Liao (1970), however, claimed to have removed in excess of 70% of the ammonia present in salmonid hatchery wastewater using simple lagooning with retention times of between 2.3 and 6.0 days, but did not mention any facility for the continuous removal of settled solids in his lagoons. His experiments were carried out in late spring and summer, though, and so direct ammonia uptake by algae and other aquatic plants is possible. Also, nitrification may have been considerable at such high retention times.

4.1.3 The effects of pond geometry on performance

Pond geometry can also influence performance. Deep lagoons often suffer from oxygen depletion near the bottom due to poor oxygen transfer from the surface, and insufficient light intensity to support photosynthesis. This problem is particularly common in summer, when thermal "stratification" further hinders oxygen transfer. When oxygen depletion occurs, anaerobic biodegradation of organic material at the lagoon bed generates toxic gases, the principal compounds being ammonia, methane, hydrogen sulphide, and phosphine (Wheaton, 1977).

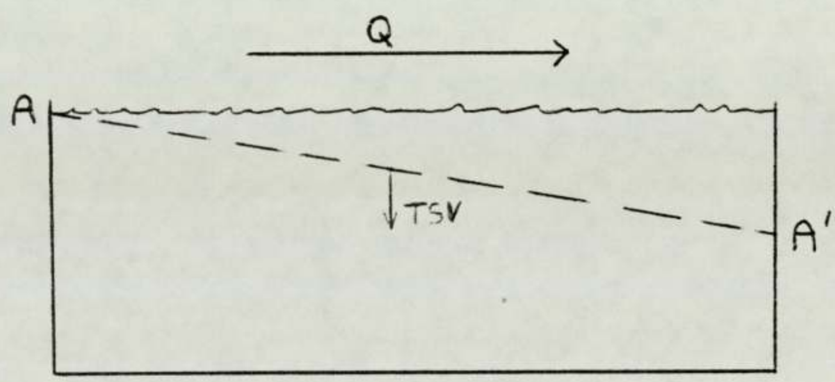
Another important aspect of lagoon geometry is that of length/width ratio. Arceivala (1983), claimed that if this ratio is greater than about 8, then a good approximation of "plug flow" is likely, with similar linear velocities prevailing throughout the pond. If, however, length/width is less than about 4, then "dead" areas of very little fluid movement can occur. This decreases the effective retention time or mean residence time of the lagoon. Mangelson and Watters (1972) performed tracer studies with a large number of lagoon designs and found that the mean residence time of a pond could be increased by as much as 75% by the installation of baffles to increase the length/width ratio of the flow path. From fluid mechanics theory (Douglas et al., 1979) it is apparent that the formation of "dead" areas, known as boundary layer separation, is often associated with a sudden increase in flow cross-sectional area as in the case where aquacultural effluent flows into a settlement lagoon from a relatively narrow conduit. Even if length/width ratio is large, great depth could allow boundary layer separation to occur near the inlet, as the flow cross sectional area of the pond would be large. Pond inlets should, therefore, be as broad as possible, and excessive pond depth should be avoided.

4.1.4 The effect of surface hydraulic loading on performance

Warrer-Hansen (1982b) gives an outline of a design procedure for sedimentation ponds which uses surface area as the critical design parameter. The main underlying premise is that the smallest particle size for which exclusion from the effluent is desirable corresponds to a particular settling velocity, and that all larger particles will settle faster. As particles of this critical size settle, "clean" effluent is left behind. The rate of generation of this is proportional to lagoon surface area, and must at least be equal to the throughput rate if satisfactory effluent clarification is to be achieved. This hypothesis is illustrated in Fig. 4.1.4.1.

The plane AA' links all the uppermost particles of the smallest size to be considered. This could be chosen, for example, as the size which is smaller than the particles which form the largest 99 % by

Fig. 4.1.4.1



weight of the expected size distribution. This plane descends with a velocity TSV, the terminal settling velocity of these "critical" particles. As it does so, "clean" effluent is generated at a rate equivalent to the product of TSV and the pond surface area. For 99% settlement to be possible, the volumetric flowrate of water through the pond must not exceed the generation rate. This theory appears reasonable provided that the outlet weir is sufficiently broad and shallow to preferentially derive its effluent from the uppermost reaches of the pond. A deep, narrow weir would delay the outflow of much of the "clean" effluent generated at the surface whilst allowing some deeper water containing partially settled solids to exit more rapidly. Clearly, if a pond was designed so as to derive its effluent evenly from all its depth (i.e. perfect horizontal plug flow), then retention time would be a critical factor in determining performance. Common sense suggests, though, that the depth of most outlet weirs on fish farm settlement ponds is small compared to pond depth.

4.1.5 Other factors influencing pond performance

As mentioned earlier, algae and other aquatic plants can remove ammonia (and other nutrients) from water by direct uptake. Corpron and Armstrong (1983) introduced 60 Elodea densa plants to a 37 litre tank in which 40 Macrobrachium rosenbergii shrimps were being reared and observed a reduction in ammonia concentration of about 95%. The application of this method to European fish farms, however, may be completely impracticable. One major disadvantage is that, if aquatic plants are grown in a settlement lagoon, removal of accumulated solids is difficult. Another is that in temperate climates, many species of aquatic plant die in autumn, and their decomposition releases much of the nutrients which they have absorbed, and also gives rise to elevated suspended solids and BOD levels. The shrimp culture tank mentioned above was kept at a temperature of $28^{o}C$.

Another factor affecting lagoon performance is the size distribution of the particles to be settled. Warrer-Hansen (1982a) claimed that intact rainbow trout excreta settled much more rapidly than did fragmented ones. This is in accordance with classical particle mechanics theory. Certain designs of circular tank allow excreta to leave in relatively good condition, soon after "production". These types of tank design were found to be associated with efficient sedimentation in effluent treatment lagoons. Burley and Klapsis (1985) have designed a square tank with very good fluid mixing and "self-cleaning" efficiency.

Wind blowing over the surface of a lagoon can impair its efficiency. Price and Clements (1974) found that quite considerable deterioration in sedimentation performance of sewage settlement tanks resulted from strong winds blowing perpendicular to the direction of water flow as this tended to force the flow towards the far side of the tanks thus causing hydraulic short-circuiting. Winds blowing parallel to the direction of flow, either co-currently or counter-currently, had little effect on lagoon performance. Lagoons should therefore be sheltered from crosswinds if possible.

Mudrak (1981) studied settlement lagoons at four salmonid hatcheries and found the performance of concrete ponds to be far superior to that of earthen ponds of similar retention time. He did not, however, offer any explanation as to why this should be. No reason is apparent, and this claim has not been supported by any other author.

The performance of lagoons can be enhanced by introducing various modifications of design and operation. Liao (1970) found that when sedimentation was preceded by aeration, lagoon retention times of 4-10 hours were sufficient to remove 50-90% of the pollution load from salmonid hatchery effluent. This was attributed to the agitating effect of the air bubbles inducing flocculation of the particles. Liao and Mayo (1974) confirm this as being an efficient treatment method. This does have an economic advantage over simple settlement in that it requires a much smaller land area, but it also incurs pumping costs. Another rather expensive but effective modification is the use of inclined plates or tubes to increase settlement capacity (White et al. 1976). These are aligned in the direction of flow throughout the depth of a settlement pond and effectively convert it to a vertical stack of shallow ponds each having the surface area of the original pond. This approximately doubles pond capital cost though (Muir, 1982).

Harman (1978) studied the enhancement of sedimentation by processes of flocculation using polymers and coagulation using lime. He found flocculation to be a fairly efficient and cost-effective modification, but found lime coagulation to be rather expensive in his trials.

Lime dosing is also widely used outside aquaculture to precipitate inorganic phosphates. Application techniques for this, and the use of transition metal salts as an alternative, are fully described by Porcella and Bishop (1975) and Gloyna and Eckenfelder (1970). Even without the addition of precipitating reagents, some reduction in phosphorus levels during settlement is expected because 20-30% of the orthophosphate in most aquacultural effluents is associated with suspended solids (Muir, 1982). In addition to forming insoluble calcium phosphates, soluble phosphates have been found to adsorb to suspended matter in rivers up to a maximum loading of 2.8 micromoles per gramme (Stabel and Geiger, 1985).

4.1.6 The Swirl Concentrator

Another type of solids removal device is described by Warrer-Hansen (1982b). This is the swirl-concentrator, which consists of a circular tank with a central standpipe water outlet and a tangentially directed inlet stream. The effect of this design is to concentrate the settled solids near the centre of the tank floor, where they are continuously removed via a small outlet pipe. Warrer-Hansen (1982b) claimed that the rotational flow pattern induced by this design caused suspended solids to be drawn to the tank floor and hence effective settlement could be attained using much smaller surface areas than those of conventional settlement ponds. This type of device, more commonly called a hydrocyclone, is widely used in the brewing industry and elswhere. In the UK at present, the use of swirl-concentrators is almost unheard of, but they are very commonly used in Denmark, usually as a precursor to settlement in a conventional lagoon, as their small surface areas do not facilitate the removal of the majority of effluent solids. It is possible that the low head loss tolerance on most fish farms might prelucde the use of high velocity and therefore high head loss cyclones. A well-designed conventional settlement pond could, therefore, eliminate any need for swirl concentrators. A company in Denmark manufactures swirl concentrators for fish farming out of stainless steel.

Table 4.1.2 summarises all factors purported to influence settlement pond efficiency.

4.2 Treatment of aquacultural effluent by biofiltration

Biofiltration is the process of removal of nutrients, principally ammonia, from effluents by means of bacterial degradation. The bacterial population is situated in a film coating the surface of a packing of either small rocks or plastic rings, over which the effluent is passed. In contrast to lagooning, biofiltration is not widely used in the European aquaculture industry. Warrer-Hansen (1982b) attributes this to the combination of the relatively high cost of biofiltration and the very large volumetric flowrates of effluent discharged, relative to production levels, in fish farming.

The performance of biofilters is influenced by several factors. Wild et al. (1971) developed charts which show the rate of ammonia biodegradation, or nitrification, as a function of temperature and pH. Nitrification was found to be highly favoured by high temperatures (up to 30^{o}C), and severely impaired below about 10^{o}C. Optimum pH is about 8.4, with values of below about 6.5 being highly unfavourable. Harman (1978) found the performance of two different types of biofilter to be very poor at a temperature of 8.5^{o}C and a pH of 6.3. Liao and Mayo (1974) developed an equation relating nitrification rate to temperature at a pH varying between 7.5 and 8.0. In the development of this equation, however, it appears that zero order reaction kinetics and negligible mass transfer control of overall reaction rate were tacitly assumed.

Table 4.1.2

Factors affecting sedimentation efficiency - a literature summary

Factor	Desirable Condition	Reason(s)	References
Lagoon retention time	Less than 50 hours	Prevents formation of algal blooms	Toms et al. (1975) Harman (1978)
"	Greater than 3 days	Ensures thorough removal of suspended solids and associated pollutants	Liao (1970)
Removal of accumulated solids	Continuous, if economically practicable	Prevents NH_3 formation by decomposition of nitrogenous wastes	Harman (1978) STWA (1983) H A Hawkes (personal communication)
Lagoon depth	"Shallow"	Prevents anaerobic conditions from developing near bottom	Wheaton (1977)
"	As above	Minimises formation of "dead" areas near inlet	Douglas et al. (1979) (General fluid mechanics theory)
Length/Width ratio of lagoon	Greater than 8.0	Minimises formation of "dead" areas near inlet	Arceivala (1983)
Installation of baffles	Advisable if economically practicable	Can prevent formation of "dead" areas near inlet	Mangelson and Watters (1972) Arceivala (1983)
Lagoon surface area	Greater than: Throughput rate/smallest particle settling velocity	Desired effluent quality unattainable by simple sedimentation otherwise	Warrer-Hansen (1982b)
Aquatic plant growth in lagoon	Advisable only in high temperature applications	Absorbs ammonia, but dies in autumn in temperate climates	Corpron and Armstrong (1983)
Rearing tanks preceding lagoon	"Self-cleaning" hydraulics	Allows excreta to enter lagoon in more settleable condition	Warrer-Hansen (1982a) Burley and Klapsis (1985)

Table 4.1.2 cont.

Factor	Desirable Condition	Reason(s)	References
Surface winds over lagoon	Minimise in direction perpendicular to flow	Considerable deterioration in sedimentation efficiency otherwise	Price and Clements (1974)
Aeration preceding sedimentation	Advisable if landspace is at a premium	Greatly increases performance at low retention times	Liao (1970) Liao & Mayo (1974)
Flocculation using polymers	As above	As above	Harman (1978)
Coagulation using lime	Not advisable generally	Improves performance, but is expensive	Harman (1978)
Use of inclined plates or tubes	Advisable if landspace is at a premium	More intensive use of available landspace	White *et al.* (1976) Muir (1982)
Use of Danish "swirl concentrator" design	Advisable subject to size/capital cost constraints	Facilitates removal of accumulated solids	Warrer-Hansen (1982b)
Pond material of construction	Concrete in preference to earth	Greater efficiency for a given retention time	Mudrak (1981)

Rogers and Klemetson (1985) conducted experiments with fixed film biofilters and found that at temperatures ranging between 22.5 and 30.8°C and hydraulic loadings of 0.3-1.2 gpd/ft^2, the filters removed 50-90% of their ammonia loading. Ignoring the detrimental effect of lower temperatures, for similar ammonia removal from 1 million gallons/day of aquacultural effluent to take place at the maximum hydraulic loading of 1.2 gpd/ft^2, the required filter surface area would exceed 800,000 ft^2. With a typical filter medium specific surface area of 60 ft^2/ft^3, the required filter volume would be approximately 14000 ft^3.

The presence of high concentrations of organic matter in effluent can inhibit nitrification by promoting the growth of heterotrophic bacteria (Liao and Mayo, 1974; Harman, 1978). Also, high suspended solids concentrations can result in rapid plugging of the filter medium. This leads to high pumping costs and frequent periods of maintenance "downtime".

In view of the low water temperatures and high flow rates involved in fish farming, and the relatively high cost of biofiltration, it appears unlikely that this method could ever be widely used for the removal of ammonia from European aquacultural effluents. Weatherley (1983), though, has successfully applied biofiltration to a laboratory scale recirculating aquaculture system. In such a system, temperature control is relatively easy. Similar work has been carried out at Aston University since 1973.

4.3 Other methods for treatment of aquacultural effluent

Various alternative methods for the treatment of fish farm effluent have been considered. None of these techniques, however, have found wide application in the industry.

Wheaton (1977) gives extensive design information about sand filtration, but no data concerning the cost-effectiveness of this process. Harman (1978) tested a pressure sand filter and found it to have high capital and operating costs. The filter was also very prone to plugging as fibrous components of fish faeces tended to stick to the anthracite which comprised one layer of filter medium. Wheaton (1977) also outlined the theory behind other types of mechanical screening process, including microscreening. As with sand filtration, though, Harman (1978) found microscreening too expensive to be economically feasible in application to aquaculture. No data have been found to date regarding the performance of or cost effectiveness of other types of mechanical screening process (eg rotary screens) in an aquacultural context.

In Sweden, the manufacturers of the Triangelfilter® claim that it will cost-effectively remove suspended solids from fish farm effluent. It is, however, very similar in design to a microscreen and is only claimed to treat waterflows which are rather small in an aquacultural context.

A continuous centrifugation process was also described by Wheaton (1977). This was claimed to be applicable to aquaculture, but no cost data were supplied, and no other work regarding this technique was found.

In the search for a cost-effective technique for the removal of ammonia from fish farm effluents, ion-exchange in beds of clinoptilolite was considered. The process is discussed by Wheaton (1977), and analysed in detail by Dryden and Weatherley (1987). This technique has an advantage over biofiltration in that it is efficient even at low temperatures. It is, however, an expensive process and would be especially so if used to treat the high volume-flow effluents generated by fish farms. Also, the regeneration of clinoptilolite beds requires backwashing with considerable volumes of saline solution. The disposal of the spent backwash solution, which would contain ammonia, would present problems.

A more recently developed effluent treatment technique is that of "bioaugmentation". This refers to the introduction of a commercially available culture of eight different types of water purifying bacteria which degrade organic material and oxidise ammonia (Ehrlich et al., 1987). Larsen (1986) reports that introducing this reagent to a Danish trout pond on a flow-through basis at a rate of 1 p.p.m. reduced its effluent ammonia concentration by 90%. To achieve such an effect on a commercial scale, however, would require one gallon of this culture for every million gallons of farm effluent. One litre of the reagent costs about 6.5 Canadian dollars, and so the annual treatment cost for a farm with a water consumption of 1 million g.p.d. would be nearly £5000. Also, the rapid decomposition of organic material would cause serious oxygen depletion in the effluent stream.

In conclusion, it would appear that sedimentation may be the only economically feasible way of treating fish farm effluents. The evidence suggests that the mechanical filtration of such high volume flows is prohibitively expensive. This was explicitly stated by Warrer-Hansen (1982b). It has been claimed that well designed lagoons can remove most of the BOD and suspended solids from effluents, and may also reduce ammonia levels to an extent. There is no economically feasible process, though, which would remove the majority of the ammonia present in aquacultural effluents. The cheapest ammonia removal device - the nitrifying biofilter - could not treat fish farm effluents effectively because of their low temperatures. Even if this were not the case, the high volume flows of most effluents would require an impracticably large biofilter.

This project has therefore concentrated on sedimentation and the use of low pollution diets (see Chapter 6) in the development of an effective strategy for effluent control.

CHAPTER 5

PERFORMANCE CHARACTERISTICS OF A VARIETY OF SEDIMENTATION PONDS

5.1 Sedimentation in settlement lagoons - a general survey

This work was carried out between April and August 1986 in order to test and quantify the influence of the parameters identified in Chapter 4 on settlement pond performance. A total of 17 ponds (numbered 1-17) at 12 different farms were studied. The experimental procedure was as follows:

5.1.1 Materials and Methods

A day was devoted to the study of each pond so that water samples could be fully analysed within 24 hours of sampling. The following measurements were taken:

(i) Pond length.

(ii) Pond breadth (mean estimated for irregular ponds).

(iii) Width of inlet(s) to pond.

(iv) Water temperature.

(v) Air temperature.

(vi) Pond retention time (by observing the passage of a tracer of "malachite green" (C.I. 42000) introduced to the pond inlet at about 1g per 10^6 gallons per day water flow).

(vii) Dissolved oxygen levels at the inlet and outlet.

In addition, the following were noted:

(i) Volumetric flowrate through pond (rough estimate obtained from farmer).

(ii) Lateral positioning of pond inlet(s) relative to the sides of the pond.

(iii) Type of diet fed to the fish.

(iv) The time since the last complete removal of the accumulated solids from the pond bed (obtained from the farmer).

(v) Dimensions and positioning of baffles (if any).

(vi) Forecasted wind speed and direction.

(vii) Direction of alignment of pond (where applicable).

(viii) Type of rearing tanks used.

(ix) Other factors, e.g. presence of fish, algae, etc, unusual odours, weather conditions.

Four replicate water samples were collected at the inlet and outlet to the pond and taken back to the laboratory for analysis. This replication helped to take account of intra-assay variance in the analyses. The sampling from the inlet and outlet was separated by the measured pond retention time to ensure comparability in input conditions which may be liable to rapid change (e.g. during feeding).

In the laboratory, the following tests were performed on all samples:

(i) Ammoniacal nitrogen (Thomas and Chamberlain, 1980). In this test, 50 ml of sample was placed in a Nessler® (The Tintometer Ltd., Salisbury) tube and 2 ml of Nessler's reagent were added. The resultant colour of the sample was then compared with tinted ground glass standards using Nessler's comparator.

(ii) 5-day, 20°C Biochemical Oxygen Demand (BOD) with allylthiourea (ATU) modification (DoE, 1981). Samples were diluted with an equal volume of dilution water which was prepared by aerating 2 litres distilled water for 1 hour and adding 1 ml of the following reagents in the order in which they are listed:

1. 0.0125% w/v $FeCl_3$
2. 2.75% w/v $CaCl_2$
3. 2.5% w/v $MgSO_4$
4. Phosphate buffer (42.5 g KH_2PO_4, 8.8 g NaOH, 2.0 g $(NH_4)_2SO_4$ in 1 litre of distilled water)
5. 0.05% w/v allylthiourea.

The samples were then brought to a temperature of 20°C and aerated by vigorous shaking. Dissolved oxygen levels were measured using a meter (Phox Instruments Ltd., Shefford, Beds.) to ensure that the saturation value of 9.26 mg/litre had been attained and the samples were then placed in 250 ml ground glass stoppered bottles. Before placing the stoppers on the bottles, the samples were allowed to stand for 10 minutes and the bottles were tapped lightly to dislodge small air bubbles. The stoppered bottles were then incubated at 20°C for 5 days after which dissolved oxygen levels were again measured. BOD was calculated by multiplying the measured oxygen deficit by 2, as the samples were diluted 1:1.

(iii) Total suspended matter dried at 103-105°C (APHA, 1975). In this test, Whatman® GF/C 5.5 cm glass fibre filter discs were placed in a vacuum filtration system and washed with 60 cm^3 distilled water. They were then removed and dried in an oven at 103-105°C for 1 hour. These prepared discs were weighed and then used to filter 500 cm^3 of effluent samples. They were then placed in the oven at 103-105°C for 1 hour before reweighing. The suspended solids concentration was calculated by doubling the disc weight increase to give a figure in mg/litre.

(iv) pH (using a meter: Phox Instruments Ltd, Shefford, Beds).

Means and sample standard deviations were calculated for all groups of replicate tests, and all regressions were performed using the analysis of variance technique incorporated in the MINITAB® statistics computer package (Ryan et al., 1981). Significance of correlations was tested using the Student one-tailed t-test (Snedecor and Cochran, 1980).

Tables 5.1.1.1, 5.1.1.2 and 5.1.1.3 (overleaf) give the values obtained for the design and operating parameters listed earlier and Table 5.1.1.4 records any additional notes made. Figs. 5.1.1.5., 5.1.1.6 and 5.1.1.7 show the inlet and outlet values and calculated percentage removals for suspended solids, BOD and ammonia respectively, whilst Figs. 5.1.1.8 and 5.1.1.9 show values and inlet/outlet changes for pH and dissolved oxygen.

Table 5.1.1.1 - Pond design and operating parameters (i)

Pond	Length (m)	Width (m)	Flowrate (gpdx10^6)	Surface Area (m^2)	Retention Time (min)	Inlet spec.
1	1250	8	18	10,000	90	2x36" pipes
2	195	15	25	2,900	35	4.5m channel
3	500	22	8	11,000	90	2m channel
4	80	9	8	720	8	3m channel
5	43.5	10.3	8	450	5	2.2m channel
6	22	15	8	330	3	2.2m channel
7	229	59	15	13,500	80	9m channel
8	66.5	18.5	8	1,220	20	3m channel
9	2.5 diameter		0.038	4.9	10	6" pipe
10	2.5 diameter		0.038	4.9	20	6" pipe
11	10.3	5.9	1	61	2	1.5m channel
12	52	3	0.75	156	150	1m channel
13	18.5 diameter		0.5	27	10	1.5 channel
14	8.9	3.7	0.7	33	10	0.4m channel
15	32	19	3.5	610	15	7x6"+3x9" pipes
16	14 diameter		3	156	30	2.2m channel
17	18.5	6	2	110	5	4" pipe +2.2m channel

Table 5.1.1.2 - Pond design and operating parameters (ii)

Pond	Diet	Material of Construction	Rearing Tanks	Direction of Alignment	Baffles	Time since sludge removal (weeks)
1	B	Earth	Earth	-	1 island	130
2	C	Earth	Earth	ENE/SSW	1 island	235
3	B	Earth	Earth	N/S	3-island chain	130
4	B	Earth	Earth	NNE/SSW	-	210
5	B	Earth	Earth	NNE/SSW	-	210
6	B	Earth	Earth	NNE/SSW	-	40
7	C	Earth	Earth	-	small island	520
8	C	Earth	Earth	NE/SW	-	260
9	E	Iron	Fibreglass	-	3 filter screens	0.6
10	E	Iron	Fibreglass	-	-	0.9
11	D	Concrete	Fibreglass	WNW/ESE	1 longitudinal	7
12	E	Earth	Concrete +Fibreglass	E/W	-	35
13	E	Earth	Fibreglass	-	-	50
14	E+F	Concrete	Concrete	N/S	-	40
15	F	Earth	Concrete	E/W	-	115
16	A	Earth	Earth	-	-	3
17	A	Earth	Earth+Iron	E/W	-	45

Table 5.1.1.3 - Pond design and operating parameters (iii)

Pond	Air Temp. (°C)	Water Temp. (°C)	Wind Direction and Speed (mph)	Ratio Length/width	Surface* loading (m^3/m^2day)	Inlet+ Ratio
1	4	6.5	NE 30	155	8.2	0.87
2	10	9	S 7	13	39.2	0.9
3	3.5	7	W 10	22	3.3	0.4
4	10	9	SW 8	9	51.6	0.29
5	10	8	SW 8	4.2	81.2	0.21
6	13	9	SW 5	1.5	110.2	0.15
7	14	10.5	S 10	3.9	5.1	0.23
8	14	10.5	S 10	3.6	29.8	0.16
9	13	13.5	SW 20	1	35.3	0.12_r
10	11.5	10.5	W 20	1	35.3	0.12_r
11	16	12	SW 10	1.7	74.5	0.25
12	14	10	SW 10	17.2	22.0	1.0
13	21	12	Light Var.	1	83.9	0.52_r
14	15	9	W 15	2.4	97.6	0.1
15	16	15	W 3	1.6	26.2	0.1
16	17	14	Light Var.	1	87.4	0.32_r
17	15	11	Light Var.	3.1	83.7	0.39

Key to symbols

* i.e. $\dfrac{\text{volumetric flowrate } (m^3/day)}{\text{surface area } (m^2)}$ Warrer-Hansen (1982) criterion for effective settlement (see Chapter 4)

\+ $\dfrac{\text{breadth of inlet conduit}}{\text{pond breadth at inlet}}$ (Chapter 4 - criterion for boundary layer separation leading to hydraulic short-circuiting)

for multiple inlets, breadths are summed

subsript r on inlet ratio: based on radius (circular lagoons)

Table 5.1.1.4 - Additional notes on settlement ponds

Pond	Notes
1	Very irregular in shape. Some sudden expansions.
2	
3	Stocked with fish. Continuous heavy rain during sampling.
4	Steep, unstable earth banks.
5	
6	Designed to promote "swirling" by inlet/outlet positioning.
7	Pumping out a rearing pond during sampling.
8	
9	
10	Measurement of retention time difficult - poor plug flow.
11	
12	Bloom of brown algae retained by a wooden barrier.
13	Designed to promote swirling.
14	Fish present in pond.
15	Pond contains a large amount of decomposing leaves.
16	Designed to promote swirling.
17	Fish present in pond.

Fig. 5.1.1.5(a): Inlet Suspended Solids Concentration (mean +/- S.D.)

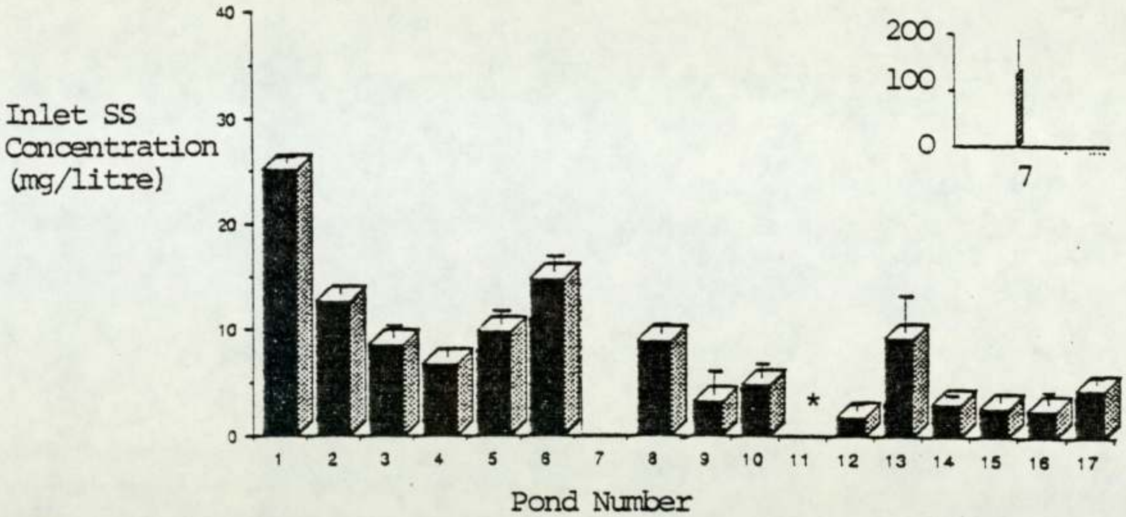


Fig. 5.1.1.5(b): Outlet Suspended Solids Concentration (mean +/- S.D.)

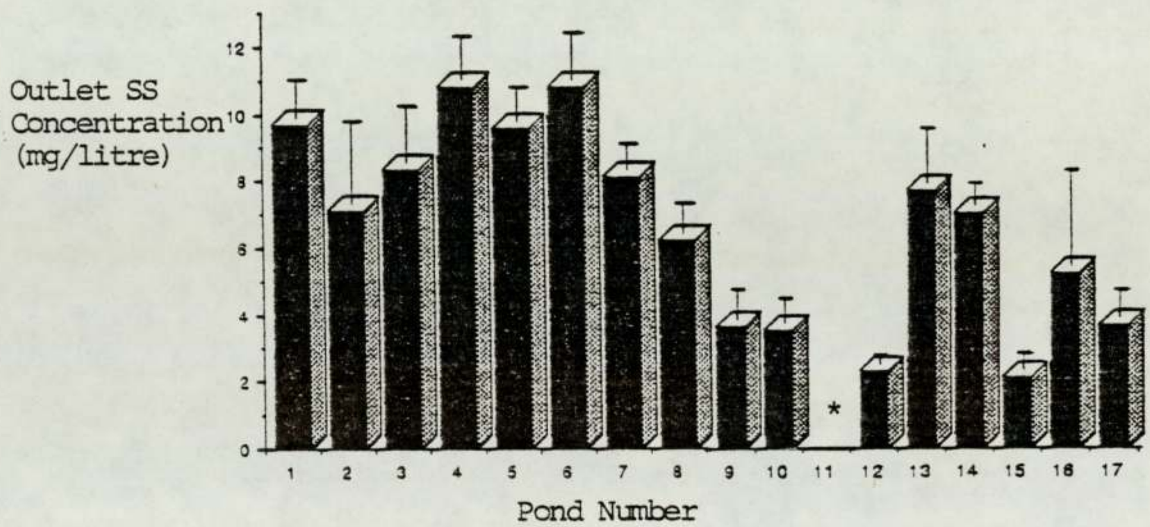
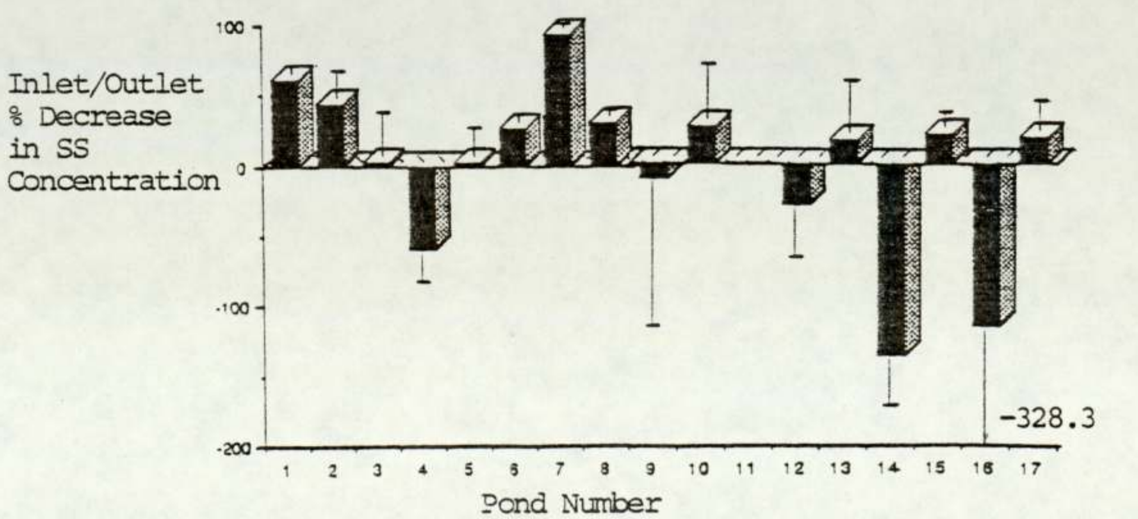


Fig. 5.1.1.5(c): Inlet/Outlet Percentage Decrease in SS Concentration



* Analysis failed due to laboratory accident

Fig. 5.1.1.6(a): Inlet BOD Concentration (mean +/- S.D.)

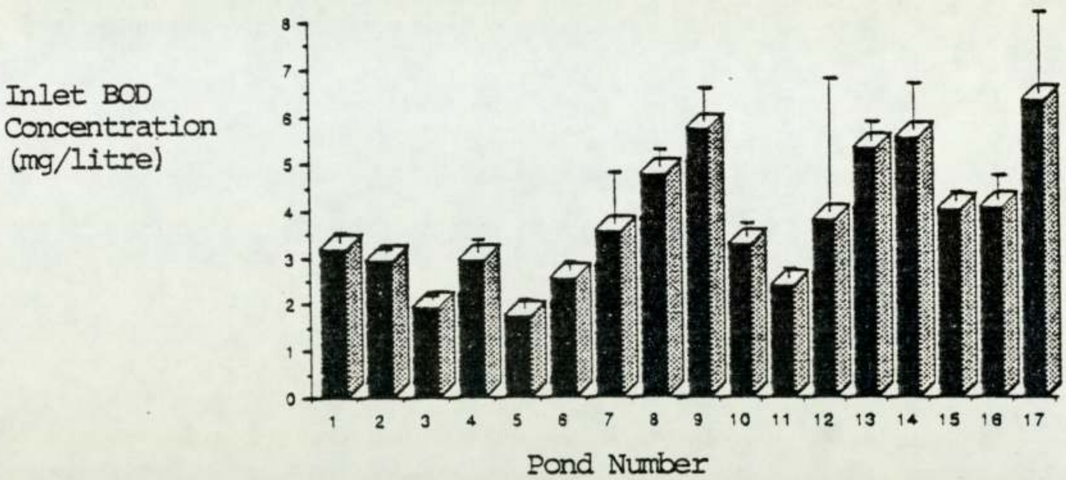


Fig. 5.1.1.6(b): Outlet BOD Concentration (mean +/- S.D.)

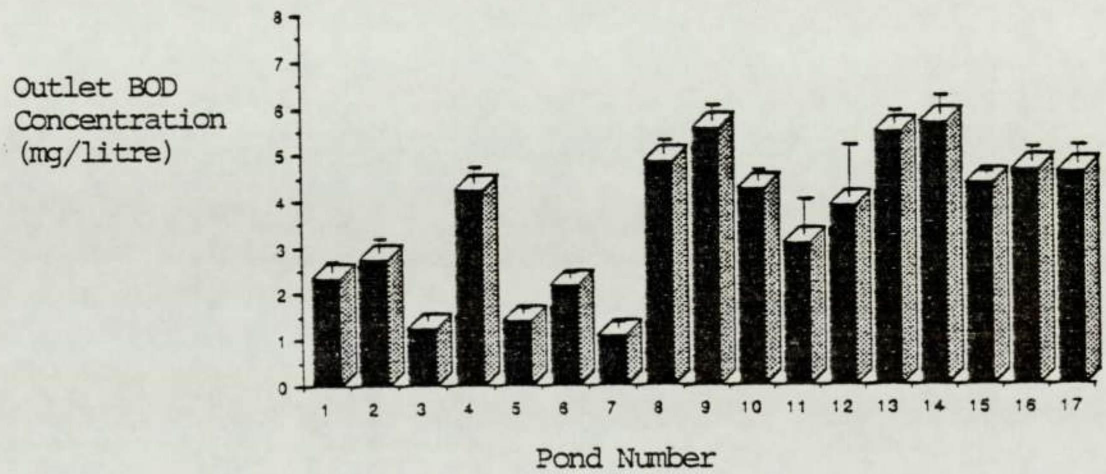


Fig. 5.1.1.6(c): Inlet/Outlet Percentage Decrease in BOD Concentration

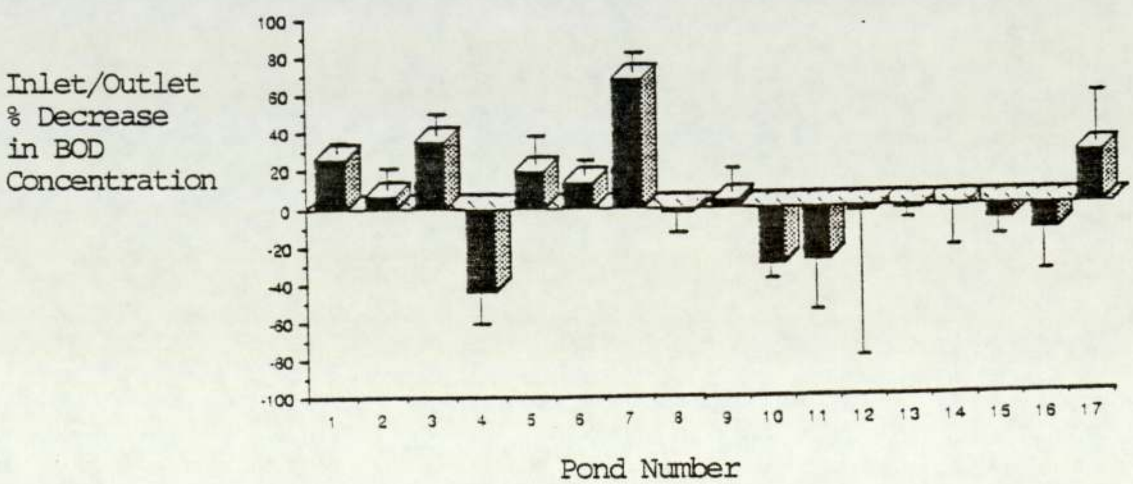


Fig. 5.1.1.7(a): Inlet NH_3-N Concentration (mean +/- S.D.)

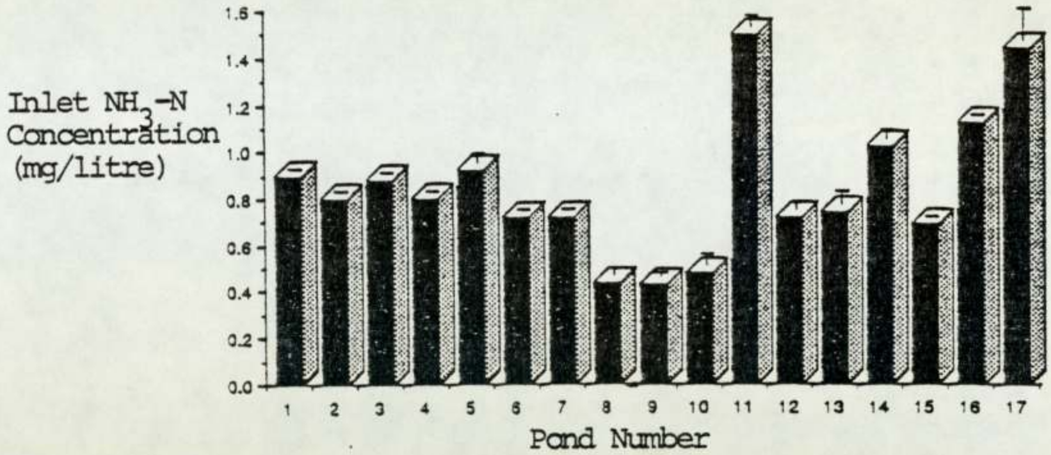


Fig. 5.1.1.7(b): Outlet NH_3-N Concentration (mean +/- S.D.)

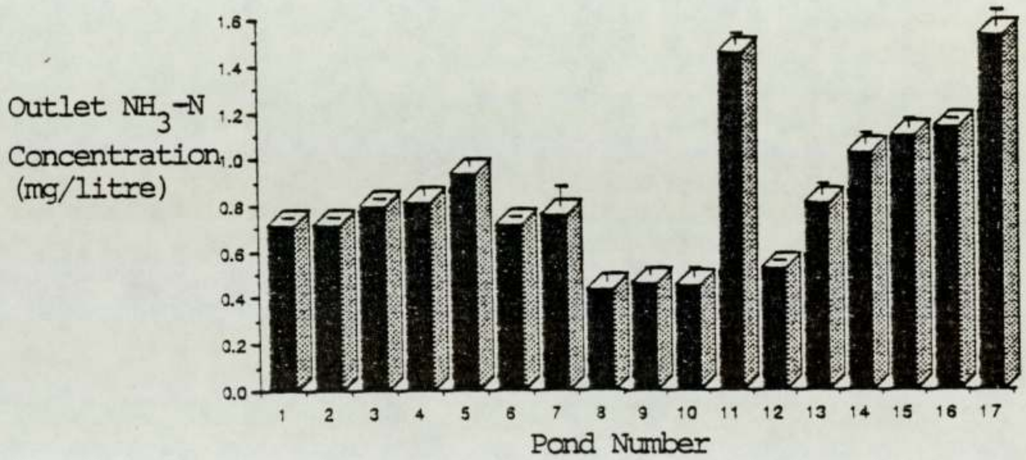


Fig. 5.1.1.7(c): Inlet/Outlet Percentage Decrease in NH_3-N Conc.

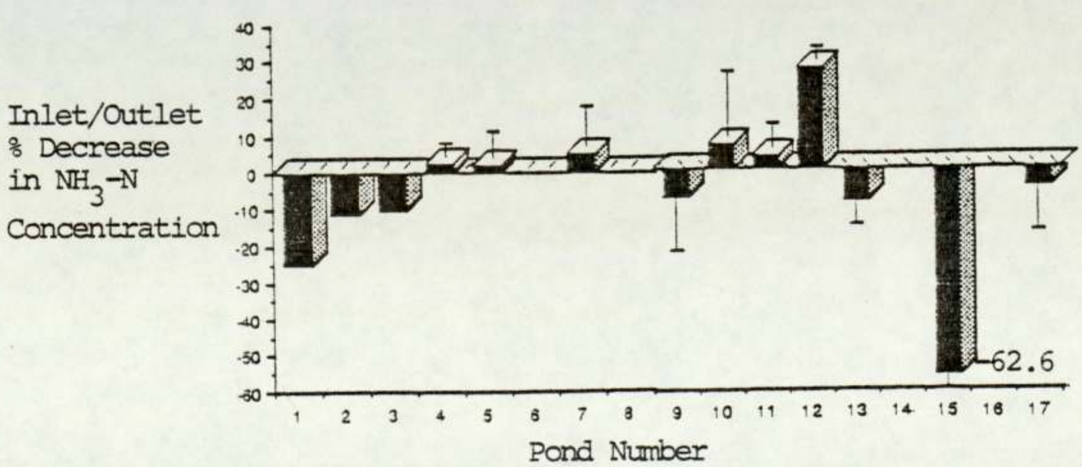


Fig. 5.1.1.8(a): Inlet pH Value (mean +/- S.D.)

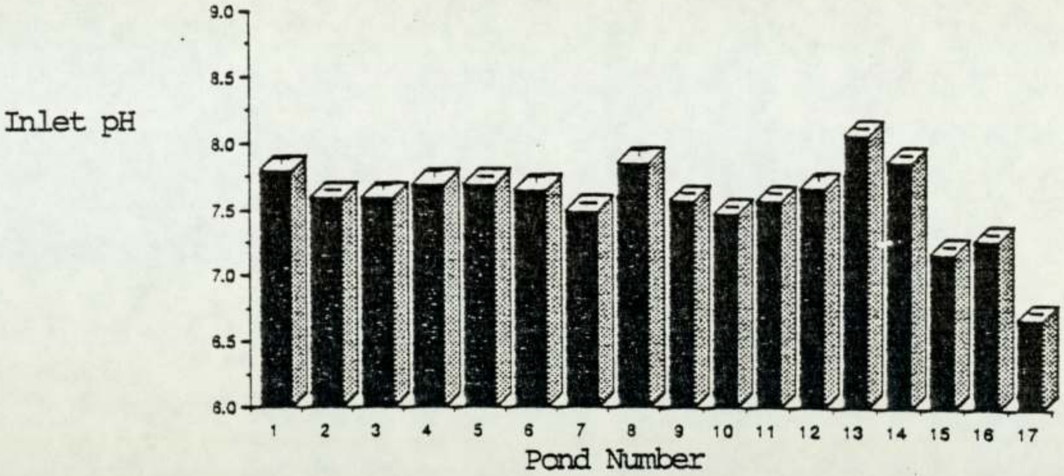


Fig. 5.1.1.8(b): Outlet pH Value (mean +/- S.D.)

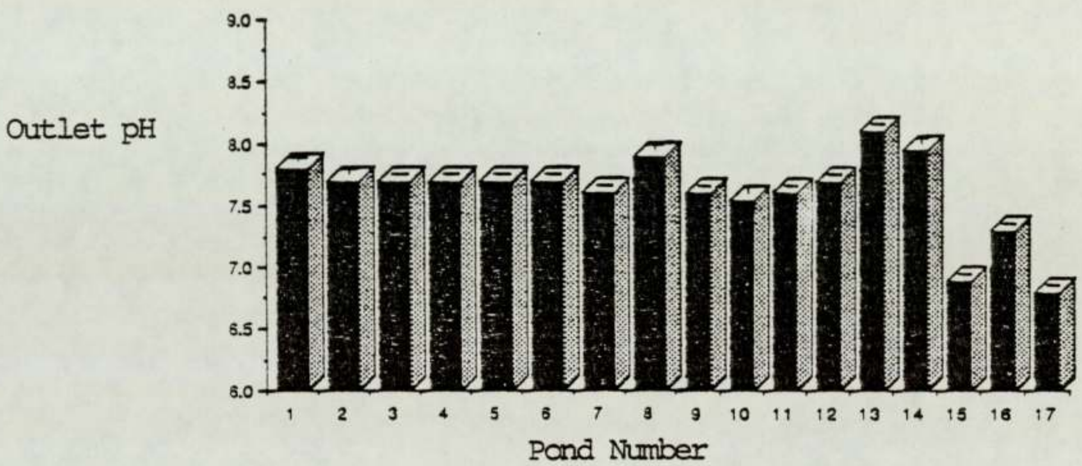


Fig. 5.1.1.8(c): Change in pH Value (mean +/- S.D.)

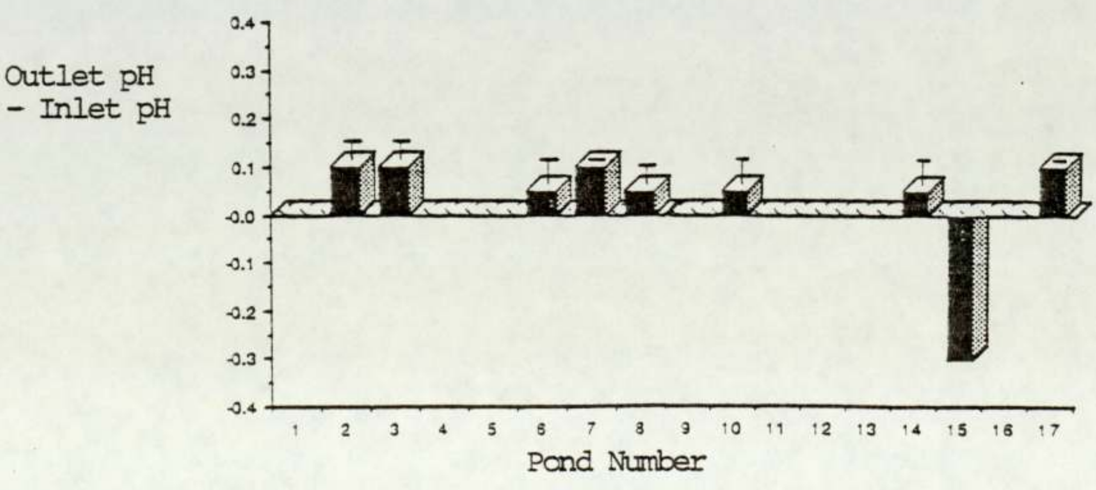


Fig. 5.1.1.9(a): Inlet D.O. Concentration

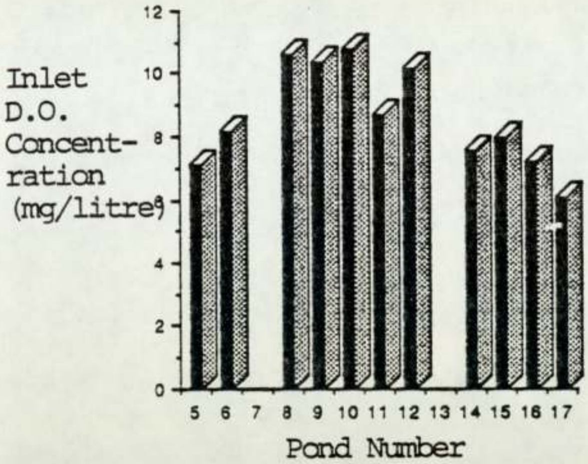


Fig. 5.1.1.9(d): Inlet D.O. - Saturation D.O.

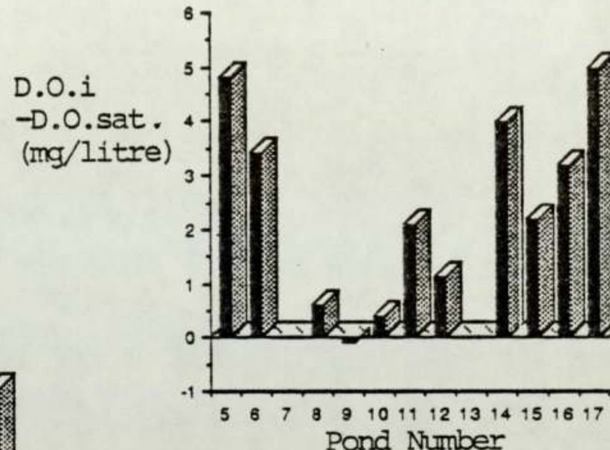


Fig. 5.1.1.9(b): Outlet D.O. Conc.

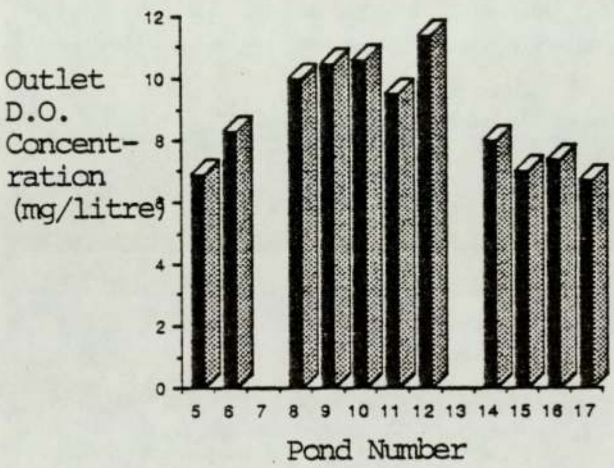


Fig. 5.1.1.9(e): % change in D.O. vs D.O.i - D.O.sat

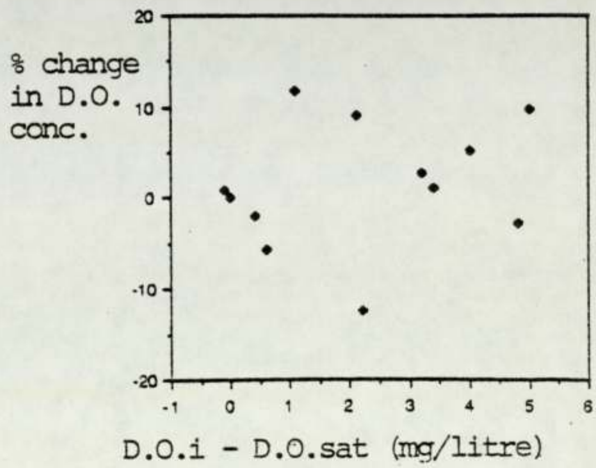
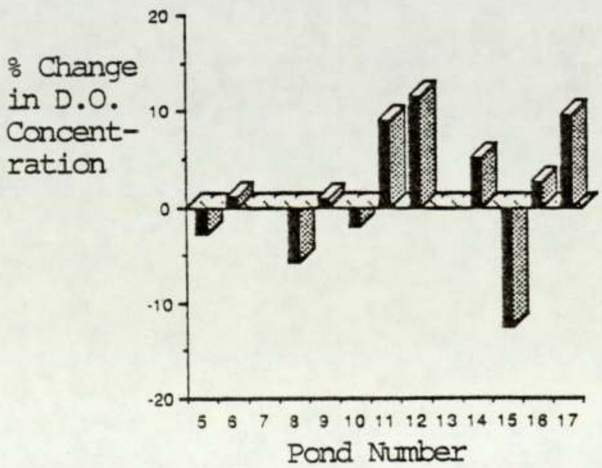


Fig. 5.1.1.9(c): % Change in D.O. Concentration



where: D.O.i = inlet dissolved oxygen concentration

D.O.sat = saturation dissolved oxygen concentration at pond temperature

5.1.2 Results and Discussion

5.1.2.1 Removal of suspended solids

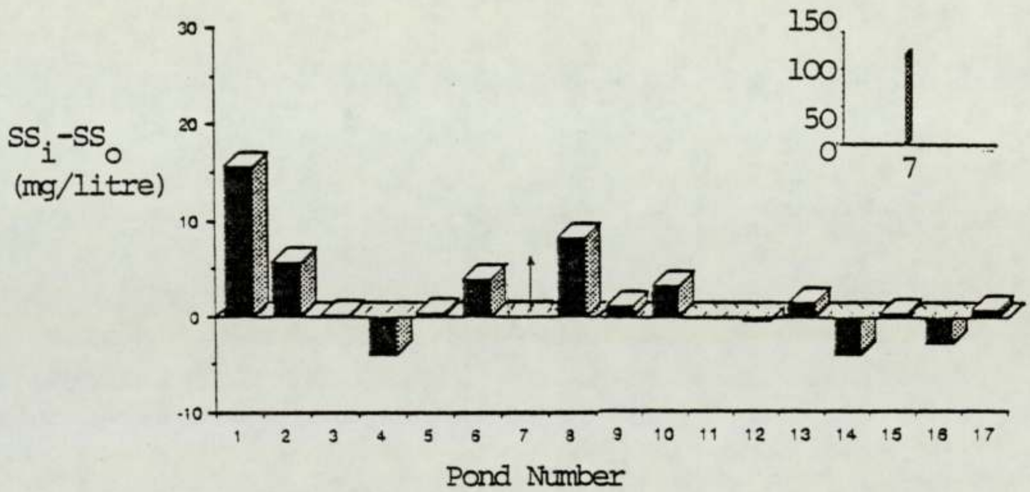
This correlated most strongly with inlet solids concentration. Fig. 5.1.2.1.1 shows the inlet/outlet concentration difference for all ponds studied and Fig. 5.1.2.1.2 shows this to relate to inlet concentration according to the function:

$$SS_i - SS_o = 0.975\ SS_i - 6.26 \qquad (p < 0.001) \qquad (1)$$

The fact that net increases in suspended solids concentration can occur indicates that resuspension of settled solids takes place in the ponds. The extent of this is such that, if the initial suspended solids levels in all of the ponds had been zero, we would expect the mean of all the outlet concentrations to be 6.26 mg/litre. This resuspension could only be due to the scouring of pond beds by fluid turbulence. The value of the coefficient of SS_i in equation (1), being so close to unity, indicates that hindering of settlement is negligible, and if no resuspension took place, the ponds would, on average, remove 97.5% of their solids loading. This relationship also implies that it is very difficult to achieve any net reduction solids concentration once the critical value of 6.26 mg/litre has been reached. Pond 7, with its very high SS_i result has a strong influence on the correlation, but without this data point it is still significant at $p<0.01$.

The decrease in suspended solids concentration expressed as a percentage of the inlet value is a more meaningful measure of

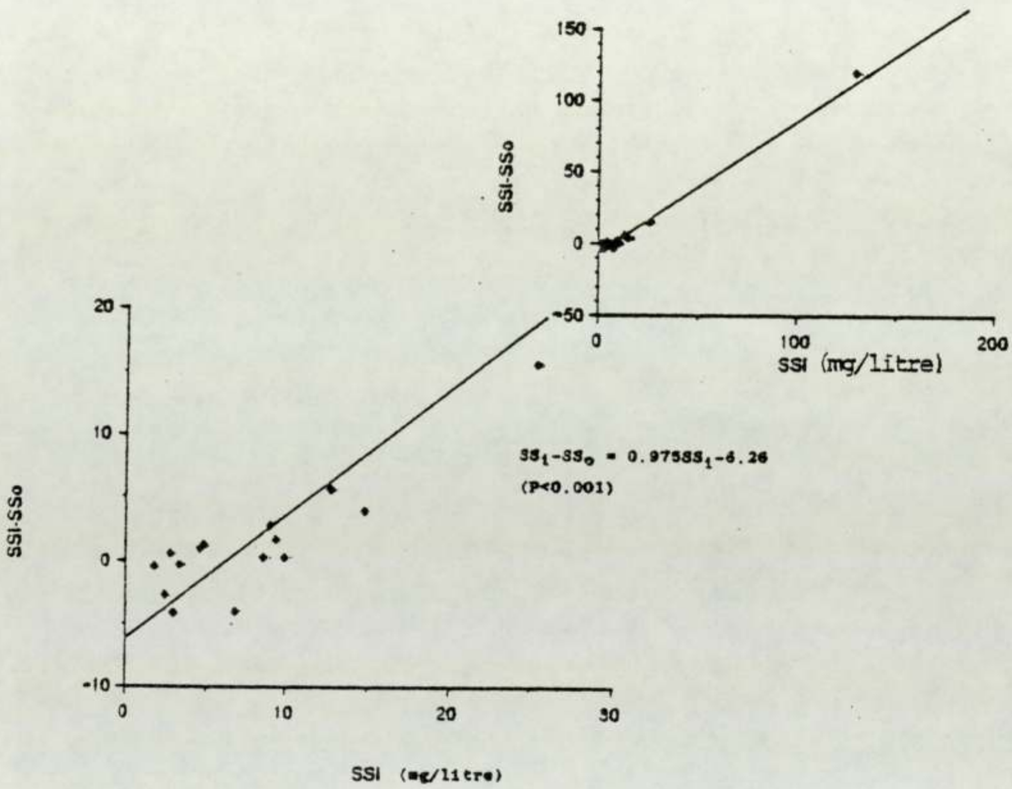
Fig. 5.1.2.1.1: Inlet-Outlet SS Concentration Difference



where: SS_i = Inlet suspended solids concentration

SS_o = Outlet suspended solids concentration

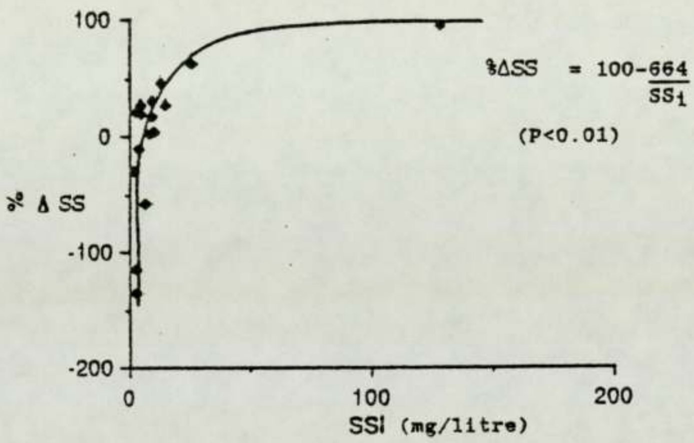
Fig. 5.1.2.1.2: Inlet/Outlet SS Difference versus Inlet SS Concentration



where: SS_i = Inlet suspended solids concentration

SS_o = Outlet suspended solids concentration

Fig. 5.1.2.1.3: Percentage Decrease in SS Concentration versus Inlet Concentration



where: SS_i = Inlet suspended solids concentration

%ΔSS = Percentage decrease in solids concentration from inlet to outlet efficiency, but suffers from great scattering of the data points where SS_i is low. This is shown on Fig. 5.1.2.1.3. From equation (1) the following expression can be derived:

$$\frac{SS_i - SS_o}{SS_i} \circ \times 100 = \%\Delta SS = 97.5 - \frac{626}{SS_i} \qquad (2)$$

An independent regression, plotting $\%\Delta$ against $1/SS_i$, assuming that 100% removal is possible at infinite SS_i, gives a similar equation:

$$\%\Delta SS = 100 - \frac{664}{SS_i} \qquad (p < 0.01) \qquad (3)$$

$(p < 0.05$, neglecting pond 7)

Although the removal of solids appears to correlate well with initial solids concentration, considerable discrepancies nevertheless exist, suggesting that some other factor or factors have significant influence on the performance of settlement ponds. As mentioned earlier, resuspension could only be caused by scouring of the pond bed by fluid turbulence. It would be expected, then, that high values of Reynolds Number would promote resuspension and cause settlement to be relatively poor for a given value of SS_i. Reynolds Number (Re) was calculated for all ponds except numbers 11, 13, 15 and 16 which are circular and therefore have non-linear flow geometry for which no Re could be defined which is comparable to that of the linear ponds. Re was calculated using the hydraulic mean diameter, d*, with the equation:

$$Re = \frac{\rho d^* \bar{U}}{\mu} \qquad (4)$$

where $$d^* = \frac{4\ (\text{flow area})}{\text{wetted perimeter}} \qquad (5)$$

and $$\bar{U} = L/\tau \qquad (6)$$

This was compared with the parameter ΔSS^* which expresses the difference between actual pond performance and that predicted using the correlation with SS_i established earlier and is defined as follows:

$$\Delta SS^* = 0.01\ ((\%\Delta SS\ \text{actual}) - (\%\Delta SS\ \text{equation (4)}))\ .\ SS_i \qquad (7)$$

This expresses deviation of the resuspension factor from the mean value according to equation (3) of 6.64 mg/litre. It did not, however, show any significant correlation with Re (Figs. 5.1.2.1.5 and 5.1.2.1.6).

This can be explained by resorting to classical fluid mechanics theory (Foust et al. 1960). In any system where fluid flows bounded by a solid wall, increasing Re influences momentum transfer at the fluid/solid interface in turbulent flow regimes by decreasing the thickness of the laminar sub-layer thus exposing more solid irregularities to turbulent fluid flow. Once Re has reached a sufficiently high value, however, the laminar sub-layer thickness is so small compared to the size of the irregularities that further Re increase does not per se influence momentum transfer. In this situation, momentum transfer from the fluid to unfixed particles on the solid surface becomes a function of the square of the mean fluid velocity. Mass transfer of these particles from the bed into the main fluid flow, although analogous to momentum transfer, correlates with a

Fig. 5 .1.2.1.4: SS Removal relative to Inlet SS Concentration (ΔSS^*)

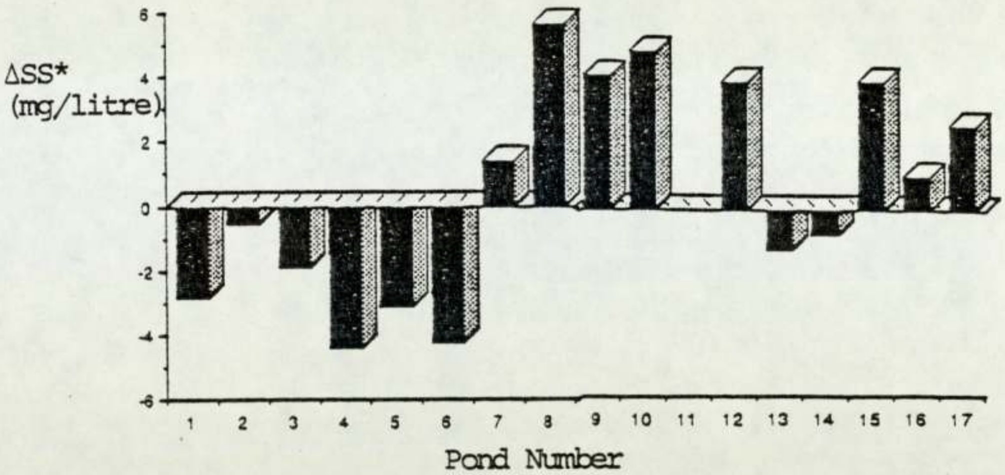


Fig. 5 .1.2.1.5: Reynolds Number for Rectangular Ponds (Re)

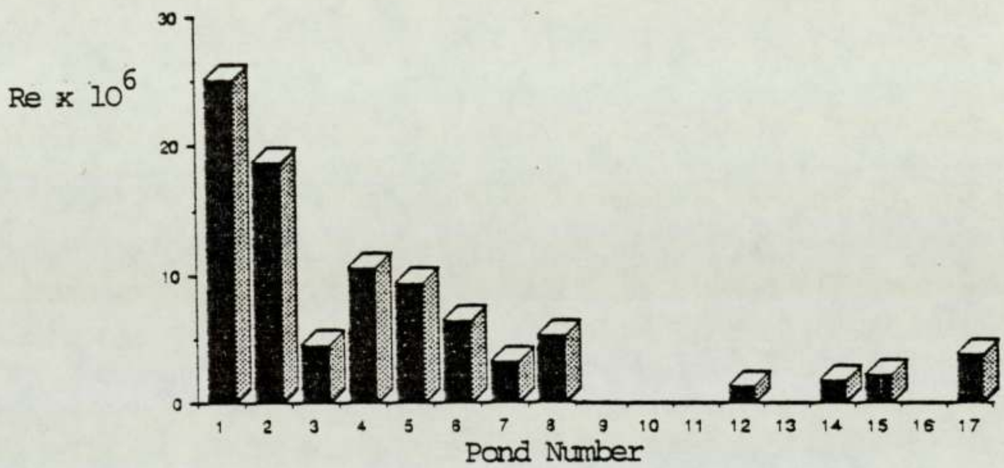
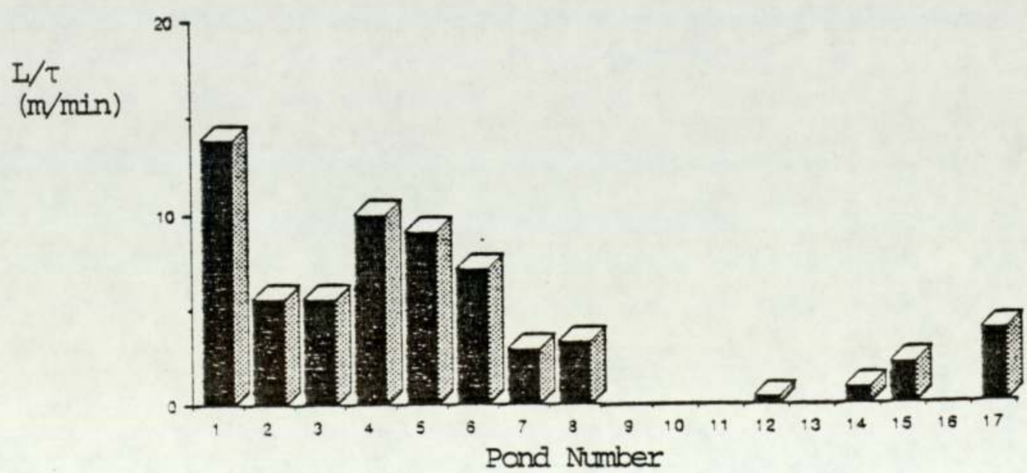


Fig. 5.1.2.1.6: Length/Retention Time Ratio for Rectangular Ponds (L/τ)



much smaller exponent of fluid velocity because much of the momentum transferred is used to promote the eddying of fluid in the wakes of particles rather than the movement of the particles themselves. In later work (see Bergé et al., 1984), attempts were made to develop mathematical models to predict turbulent flow behaviour. Such a rigorous deterministic approach to the complex problem of fluid turbulence, however, could only be applied to a few geometrically simple flow systems. An empirical approach is therefore necessary in the case of aquacultural settlement ponds. Mean fluid velocity for all rectangular ponds is given in Fig. 5.1.2.1.6.

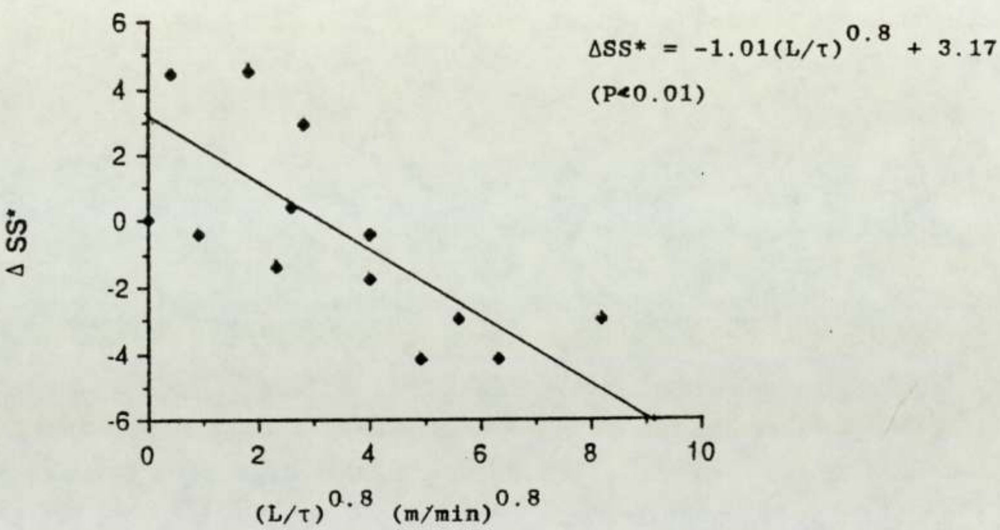
In this experiment, ΔSS^* was found to correlate with mean fluid velocity (L/τ) raised to the power of 0.8 according to the equation:

$$\Delta SS^* = -1.01\ (L/\tau)^{0.8} + 3.17 \qquad (p < 0.01) \qquad (8)$$

This is shown in Fig. 5.1.2.1.7. As with the Reynolds Number approach, this analysis was not applied to the circular ponds because of their complicated flow geometry. Flowpath length could not be measured for these ponds. Figs. 5.1.2.1.5 and 5.1.2.1.4 show values of Re for the 12 "linear" ponds, and ΔSS^* for all ponds, respectively. Note that all Re values exceed 10^6. The critical value for transition from laminar to turbulent regime in open channel flow is about 600 (Douglas et al., 1979).

From equation (8) it can be concluded that aquacultural settlement ponds should be designed so that the mean fluid velocity, L/τ, does not exceed 4.0 metres/minute and if possible should be less than 1 metre/min.

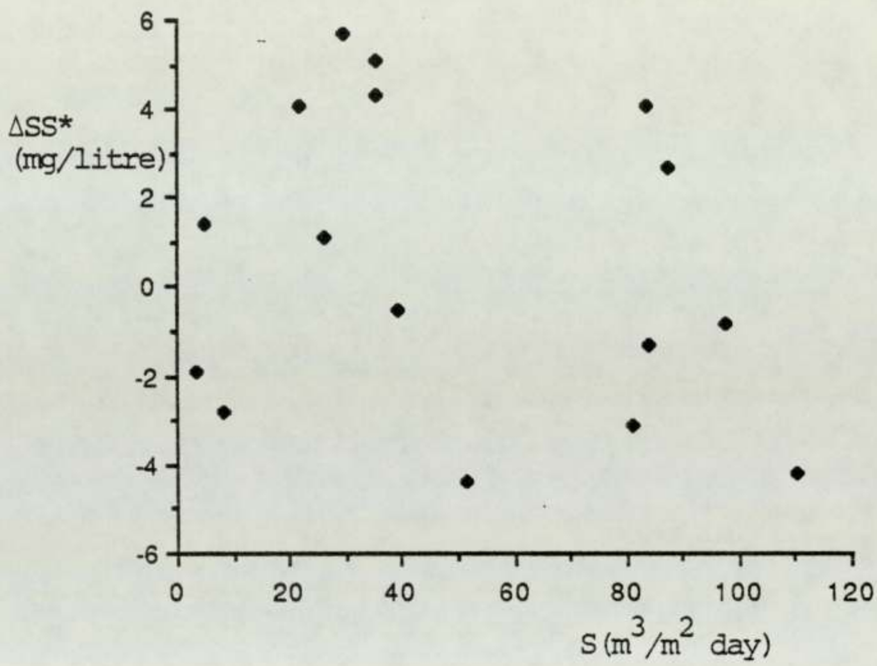
Fig. 5.1.2.1.7: Relative SS Removal (ΔSS^*) vs length/Retention Time Ratio



where: L/τ = Length /retention time ratio

ΔSS^* = Departure of suspended solids removal ($SS_i - SS_o$) from that predicted using the equation $\%\Delta SS = 100 - 664/SS_i$

Fig. 5.1.2.1.8: Relative SS Removal (ΔSS*) vs Surface Loading



where: S = Surface hydraulic loading

ΔSS^* = Departure of suspended solids removal ($SS_i - SS_o$) from that predicted using the equation $\%\Delta SS = 100 - 664/SS_i$

The surface hydraulic loading parameter defined by Warrer-Hansen (1982) did not appear to significantly influence settlement pond performance relative to inlet suspended solids concentration. This is shown in Fig. 5.1.2.1.8. The lack of correlation can be explained if we consider the case of pond 6. Dividing its throughput flowrate by its surface area (data in Table 5.1.1.1) gives a surface loading of 110 m^3/m^2 day - the highest of all the ponds studied. Despite this, the pond removed more than 25% of its suspended solids load. Ignoring resuspension, this means that the largest 25% of the solid particles in the influent must have had terminal settling velocities in excess of 110 m/day or 1.3 mm/sec. It has been found (Warrer-Hansen, 1982b) that the mean settling velocity of faeces from 5g trout fry is more than 15 mm/sec and the corresponding figure for adult fish is well in excess of 50 mm/sec. It is likely, therefore, that throughout the range of settlement ponds studied, surface loading was sufficiently low to allow a high proportion of the influent solids to settle, provided the degree of break up of faecal particles prior to settlement was not too great. This assertion is supported by the fact that the coefficient of SS_i in equation (1) is close to unity. Surface hydraulic loading is likely to become a limiting factor at much higher values than were encountered in this study. The spurious association in Fig. 5.1.2.1.8 suggests an approximate recommended design limit of 40 m^3/m^2 for surface hydraulic loading.

Length/width ratio and total breadth of inlet(s) relative to pond width did not significantly influence pond performance. As mentioned earlier, unfavourable values of these parameters would be expected to promote the formation of "dead" areas. This would reduce the effective surface area of a pond, but this would appear to be nevertheless adequate for nearly 100% solids removal to take place in all 16 ponds were no resuspension possible. All attempts to modify surface loading data using length/width and inlet ratios proved unsuccessful as it was not possible to formulate a hypothesis which combines the parameters of inlet ratio, length/width ratio, and nominal surface hydraulic loading to form a variable whose physical significance could be clearly understood.

The effect of diet on the ease of settlement of the suspended solids produced could not be adequately investigated in this study, as the 12 farms involved used 6 different diets in total, and so insufficient data were available to enable a fair comparison to be made.

The effluent from table and broodstock farms did not, as would be expected, settle any more readily than the effluent from farms principally producing juvenile fish. Very little data, though, were obtained for the latter category as only 3 of the 17 ponds studied were settling effluent from farms exclusively rearing juvenile fish.

An important conclusion of this research is that the large retention times advocated by Liao(1970) are not necessary for the efficient removal of solids. This means that high volume flow effluents may be treated cost-effectively.

Finally, materials of construction and surface crosswinds did not significantly influence the removal of suspended solids, although it would seem likely that the latter could become important at extreme values.

5.1.2.2 Removal of BOD

Unlike suspended solids, the removal of BOD did not correlate with its input concentration. Fig. 5.1.2.2.1 shows that it does, however, correlate to an extent with inlet suspended solids concentration. The log-linear correlation wrongly implies that more than 100% BOD removal is possible, but in this case 100% BOD removal is not attained until the projected function reaches an initial solids concentration of about 2000 mg/litre so it probably gives a reasonable approximation within the range of the data. As would be expected, the removal of BOD also shows some correlation with the removal of suspended solids (Fig. 5.1.2.2.2). This suggests that the principal mechanism of BOD removal in the ponds is the deposition of aerobically biodegradable solids.

The shallow gradient of the regression line in Fig. 5.1.2.2.1 indicates, however, that this is not a very efficient process. Note, for example, that with an inlet solids concentration as high as 20 mg/litre (ln SS_i=3.0), expected percentage removal of BOD is only 24%. Fig. 5.1.2.2.3 shows a remarkable lack of correlation between the inlet concentrations of BOD and suspended solids. It is evident, therefore, that a large variable component of soluble BOD is present in fish farm effluents. The proportion of total BOD which is soluble may depend on diet composition and the retention time of excreta and uneaten food in the rearing tanks where leaching and decomposition could form soluble BOD.

Fig. 5.1.2.2.1: BOD Removal vs Inlet SS Concentration (SS_i)

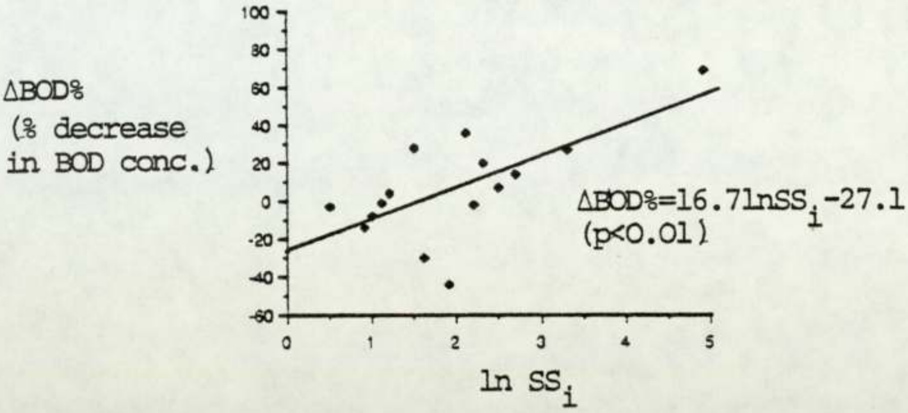


Fig. 5.1.2.2.2: BOD Removal vs SS Removal (ΔSS%)

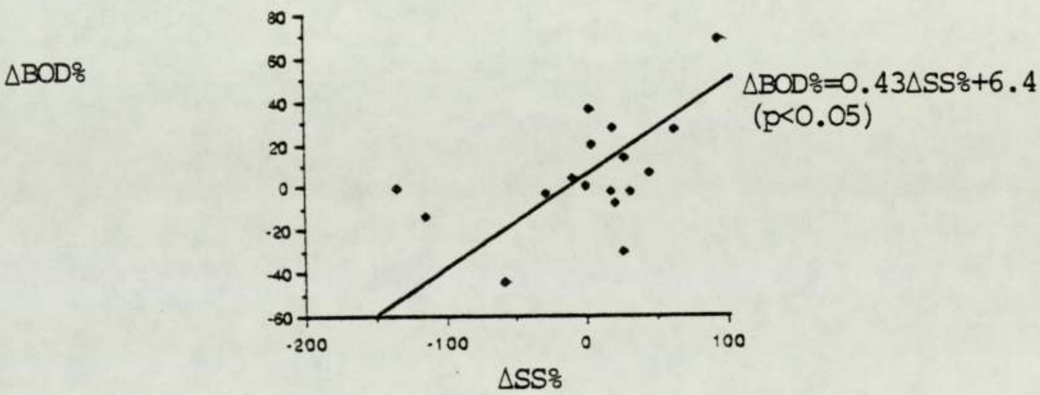
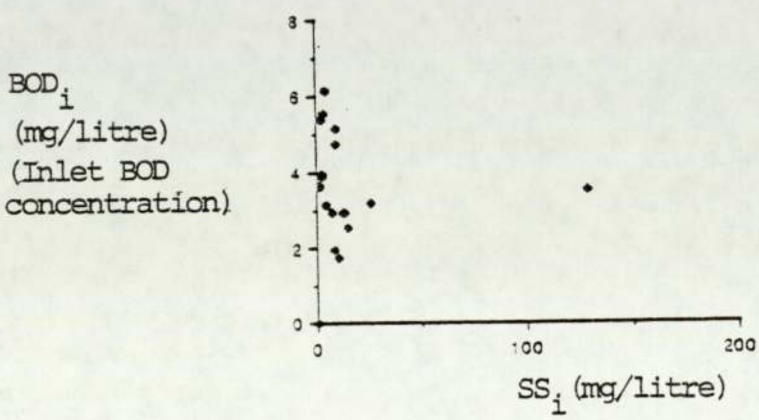


Fig. 5.1.2.2.3: Inlet BOD vs Inlet SS



ln denotes natural logarithm

It is highly unlikely that appreciable quantities of soluble BOD could be removed in a settlement pond. In a BOD test at 20°C, the extent of biodegradation of the substrate is negligible until several hours have elapsed (DoE, 1981). In this study, the highest water temperature encountered was 15°C and the largest retention time was 2.5 hours.

In 8 cases, BOD was found to increase on passage through the settlement pond. Where this was accompanied by an increase in suspended solids, it could be supposed that resuspension of biodegradable solids would account for this. Increases in suspended solids concentration on passage through ponds, though, have only been found at low inlet solids concentrations ($<$ 6 mg/litre). Bearing in mind the poor correlation between inlet solids and BOD, at such low concentrations, the contribution of solid BOD to total BOD would be very small, and so increases in solids concentration, unless proportionately very large, would be unlikely to give rise to considerable increases in total BOD.

Furthermore, in 4 cases BOD increased while suspended solids concentration decreased. One or more processes must exist, therefore, which contribute BOD to an effluent on passage through a settlement pond without causing a significant increase in suspended solids concentration. The only possible source of BOD is the layer of accumulated solids on the pond bed and so soluble BOD must leach from this layer into the water above. The extent to which this takes place will depend on the thickness and age of the solids layer as well as water velocity near the pond bed. Shortly after feeding, for instance, a relatively thick layer of fresh solids will be deposited and so the rate of leaching of soluble BOD will be high for some time. Also, uneaten food is likely to have a greater biodegradation potential than faeces, and so overfeeding is likely to promote the leaching of BOD. Also, high fluid velocities will assist the mass transfer of soluble BOD producing substances from the solids to the water.

The removal of BOD for a given inlet solids concentration correlated even more poorly with surface hydraulic loading than did suspended solids removal. This was probably due to differences in soluble BOD fraction and leaching rate having a more important influence than surface loading which, as mentioned earlier, was low enough in every case to allow good settlement, other factors permitting.

Retention time for given inlet solids concentration also correlated poorly, apart from the fact that the two worst cases of BOD increase (ponds 4 and 10) had retention times of 8 and 20 minutes respectively. This was probably sheer coincidence as it was established earlier that BOD assimilation (which would of course be time dependent) could not be significant. The length/retention time velocity factor mentioned in the suspended solids discussion did not appear to correlate. This confirms the relative unimportance of resuspension as regards BOD removal, although it could have been expected that high fluid velocities would promote the leaching of soluble BOD.

As with solids, the effect of diet on BOD removal could not be adequately assessed due to lack of sufficient data. Finally, length/width ratio, inlet specification, materials of construction, water temperature, and weather conditions, did not appear to have any significant influence on performance.

5.1.2.3 Removal of ammonia

In general, settlement did not remove significant quantities of ammonia from the effluents, and in many cases small increases in concentration were noted. The only instance in which a considerable amount of ammonia was removed was in the case of pond 12 where a large bloom of brown algae was retained near the exit by a plank of wood across the water surface. The removal of ammonia in this case was probably largely due to nutrient uptake by the algae (Harman, 1978). The main problem with using aquatic plants to remove ammonia from fish farm effluents is that most only live in spring and summer. When death and decomposition occurs in the autumn, all ammonia (and other nutrients) previously absorbed will be released. Some plants are available which survive throughout the year but growth is slow, and nutrient uptake minimal, in the winter months. Also, nutrient uptake in spring and summer is lower than that of deciduous plants because they do not undergo a seasonal rapid increase in biomass or "bloom".

The farm on which pond 15 was sited used a river supply which contained large quantities of leaves in autumn. These became waterlogged and submerged and the filter screen did not prevent many of these from entering the farm. The settlement pond at the time of this study contained about 2 years' accumulation of leaves together with farm effluent solids. No doubt, both of these contributed to a large increase in ammonia concentration as the effluent passed through this pond. It is likely that the decomposition of these leaves accounts for much of this increase.

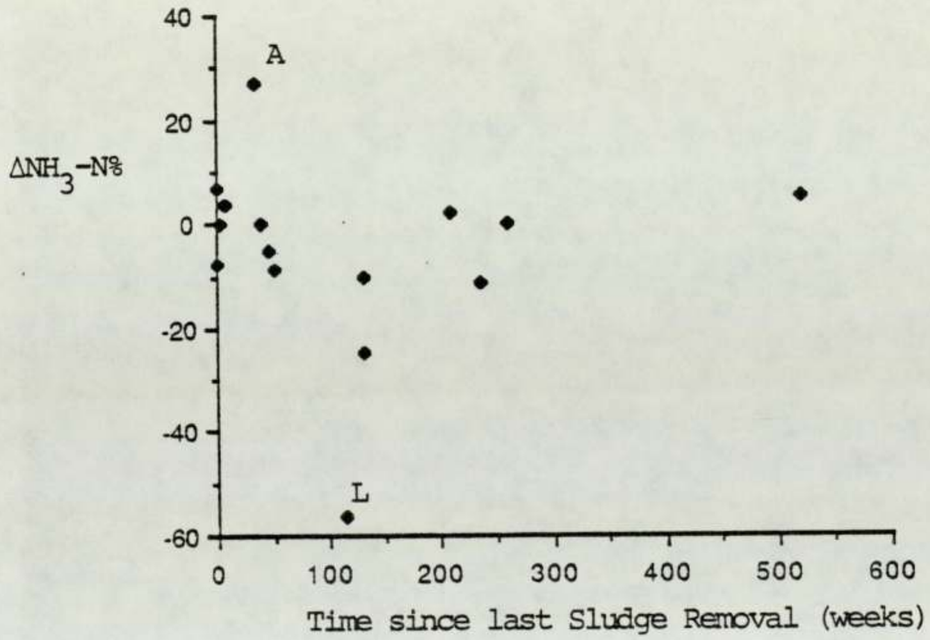
In the other cases where concentration was observed to increase, the only likely source of ammonia would appear to be the decomposition of accumulated nitrogenous solids on the pond bed. It would be expected then that ammonia increase should correlate with the length of time for which settled solids had been allowed to accumulate. Fig. 5.1.2.3.1 shows that this is not the case, and also illustrates the profound influence of algal blooms and decomposing leaves on the performance of settlement ponds. It is possible that most of the decomposition of a given element of nitrogenous sludge takes place in a matter of a few weeks and so the limiting factor would be the mean rate of deposition of fresh solids (most of which will take place during feeding and cleaning operations) and the nitrogen content of the solids. Uneaten food will almost certainly contain more nitrogen than will fish faeces and so overfeeding may contribute significantly to ammonia formation in settlement lagoons.

No other parameter measured showed any appreciable correlation with ammonia removal and it must be concluded that sedimentation is generally ineffective as a means of removing ammonia from aquacultural effluents.

5.1.2.4 Change in pH

It can be seen from Fig. 5.1.1.8 that settlement has very little effect on effluent pH. The only instance in which a significant change occurred was in the case of pond 15 (containing decomposing leaves); here a fall of 0.3 units was observed. An acidic decomposition product

Fig. 5.1.2.3.1: Ammonia Removal vs Time since last Sludge Removal



where: ΔNH_3-N% = Percentage decrease in ammoniacal nitrogen concentration from inlet to outlet

A: Pond containing algal bloom

L: Pond containing decomposing leaves

(possibly carbon dioxide) must have been responsible for this.

Other workers have not found pH to be significantly changed during settlement, but carbon dioxide uptake by algae can cause large increases (Toms et al. 1975, Harman 1978).

5.1.2.5 Change in dissolved oxygen concentration

Fig. 5.1.1.9 shows that dissolved oxygen concentration was not greatly affected by passage through the settlement ponds. This is surprising, as it is well established that oxygenation takes place by an equilibrium process at the air/water interface (Wheaton, 1977). In several cases, the effluent stream entered the pond with a D.O. concentration several p.p.m. below the saturation value. A probable explanation is that the surface areas of the ponds studied were small compared to the large and highly agitated areas of air/water interface which are generated by oxygenation apparatus in the form of small bubbles.

In the case of pond 15, with the decomposing leaves, a substantial fall in D.O. concentration was observed. This was most probably due to biodegradation of the leaves.

5.1.2.6 Sedimentation in settlement lagoons - conclusions

1. The effective settlement of suspended solids becomes progressively easier and more reliable as inlet concentration increases. It is difficult to attain at loadings below 8 mg/litre.

2. For a given inlet concentration, the settlement of suspended solids is influenced by the ratio of length/retention time (a form of mean fluid velocity). For best results this should not exceed 4 metres/minute.

3. The removal of solids is also influenced, to a limited extent, by surface hydraulic loading (i.e. volumetric flowrate/surface area). It is advisable to keep this below 40 m^3/m^2 day, although good results have in two cases been attained (for a given inlet solids concentration) at more than twice this value.

4. The removal of BOD also correlates with inlet suspended solids concentration, but the removal process implied is only effective at inlet solids concentrations in excess of 20 mg/litre.

5. A large and variable component of the BOD of fish farm effluents is soluble. This type of BOD is difficult, if not impossible, to remove within the constraints of commercial aquaculture.

6. Settlement, in general, does not remove significant quantities of ammonia from fish farm effluent. Blooms of algae in a settlement pond, however, can absorb ammonia as a nutrient.

7. pH is largely unaffected by settlement, but can be increased by the presence of decomposing vegetation.

8. Oxygenation does not take place to a significant extent in settlement ponds, although it would be expected that aquatic plants

would increase and decrease D.O. levels in daylight and at night respectively. Decomposing vegetation can also cause decreases in D.O. levels.

9. Contrary to the findings of Liao(1970), and widely held belief, large retention times (>1 hour) are not an essential prerequisite for the effective settlement of suspended solids.

5.2 The removal of ammonia using aquatic plants

Following the observation that pond 12, which contained a bloom of brown algae, removed over 27% of its ammonia loading, it was deemed important to determine whether or not another settlement pond, containing profuse growths of non-deciduous aquatic plants, could remove a substantial proportion of its ammonia loading over much of the year, thus providing an inexpensive effective solution to the problem of effluent ammonia control.

A suitable pond was found on a commercial trout farm and its ammonia removal performance was assessed periodically over the period from winter 1986/87 to late spring 1987. In addition, levels of suspended solids and BOD were monitored in order to establish whether or not any improvement in effluent quality in terms of ammonia levels was offset by deterioration in terms of suspended solids and BOD concentrations. Sampling times were chosen to provide as wide a range of environmental conditions (i.e. water temperature and light intensity) as possible. The pond surface area was about 120 m^2, and approximately 30% of this was covered with watercress throughout the period of this study. In addition, about 30% of the pond bed was covered with a thick growth of Elodea canadiensis pondweed. Finally, the pond surface contained small amounts of Lemna minor (duckweed) and a type of ryegrass. The conditions under which this study took place are described in Table 5.2.1 below.

Table 5.2.1

Date	Air Temp (°C)	Water Temp (°C)	Sky Conditions	Remarks
04.12.86	13	12	OVERCAST	
05.03.87	7	9	CLOUDY, BRIGHT	Some dead vegetation visible
25.03.87	10	10	CLOUDY, BRIGHT	
13.04.87	11	10	SUNNY	
28.04.87	19	11	SUNNY	Surface cress decreased to 20%

Notes additional to Table 5.2.1:

(i) All samples were taken between 11 a.m. and 12 noon.

(ii) The water supply was from a borehole and flowrate through the settlement pond was approximately 1 million gallons per day throughout the period of this study.

(iii) Pond retention time was approximately 20 minutes and its length was 33 metres.

The results of this experiment are illustrated by Figs. 5.2.2, 5.2.3 and 5.2.4. As in the previous experiment, the errors listed are sample standard deviations on 4 replicate samples.

Fig. 5.2.2: Inlet (in black) and Outlet NH_3-N Concentration

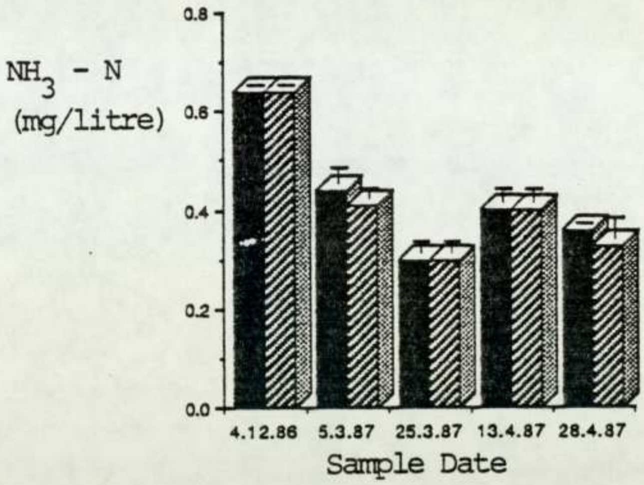


Fig. 5.2.3: Inlet and Outlet BOD Concentration

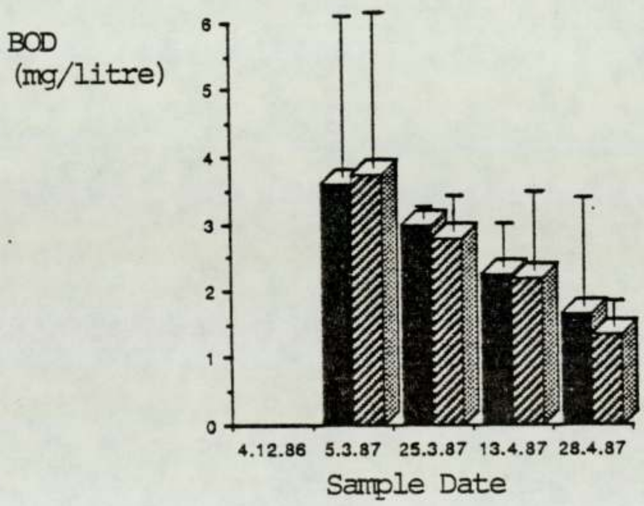
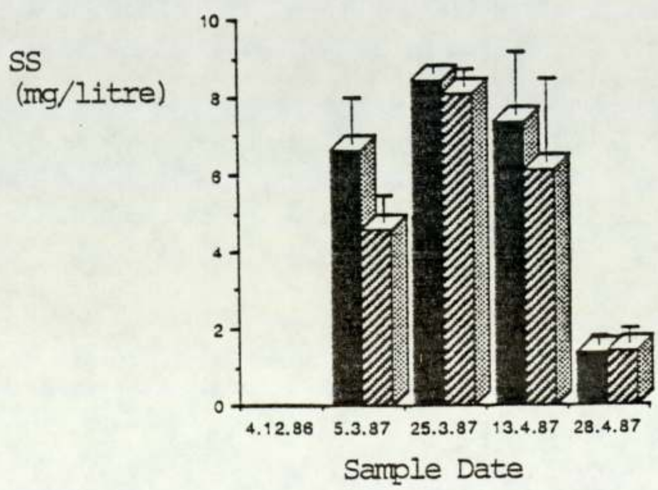


Fig. 5.2.4: Inlet and Outlet SS Concentration



All figures are means +/- sample standard deviations on 4 replicates

The results show that the pond does not remove significant quantities of ammonia from the effluent, even on a warm, sunny, late spring day. The slower growth of "evergreen" aquatic plants in spring and summer, therefore, results in much lower specific nutrient requirements than those of algae which undergo a very rapid growth at that time.

Generally, levels of solids and BOD in the effluent were not adversely affected by the presence of the plants. Removal efficiencies for these two parameters, and especially BOD, were not expected to be very high with such a dilute effluent.

It must be concluded, then, that the removal of ammonia as a nutrient by aquatic plants can only take place to a significant degree with deciduous plants like algae during their seasonal "blooming". The subsequent death of algae in autumn would be expected to have a detrimental effect on effluent quality.

5.3 Sedimentation in settlement lagoons - implications for fish farmers

For effective settlement, the inlet suspended solids concentration should be greater than 8 mg/litre. In this study, 50% of the settlement ponds satisfied this criterion, and generally performed satisfactorily. In cases where the dilute nature of an effluent prevents it from settling to the satisfaction of consent limits, the problem could be resolved by either reducing the water supply to the rearing tanks and negotiating with the relevant Water Authority for permission to discharge a more concentrated but less copious effluent, or diverting a proportion of the water supply to bypass the rearing tanks and using it to dilute the settled effluent just before discharge. The extent to which either of these solutions can be implemented will be limited by water quality criteria and the health of the fish in the rearing tanks. Extra aeration may be required. The probability of attaining good performance for a given inlet solids concentration can be maximised by adhering to the simple design principles which were established in the suspended solids discussion earlier.

BOD reduction correlates with suspended solids removal to a limited extent. The above recommendations, therefore, also apply to BOD removal. It should be borne in mind, however, that settlement does not remove BOD as effectively as it removes suspended solids, and that substantial reductions only occur where the initial solids concentration is high. In most cases, some manipulation of diet formulation and ration is necessary if effective control of effluent BOD levels is to be attained.

Settlement, generally, does not remove ammonia, but it may be removed to an extent in the summer months by algae. The best method for the control of effluent ammonia levels is likely to be the use of low pollution diets, the principles of which have been outlined earlier.

Similarly pH and dissolved oxygen levels are largely unaffected by settlement, but in this study these were never found to breach any consent limit imposed and so must be of minor importance as regards effluent quality.

The effluent treatment requirements of any fish farm will be greatly influenced by the severity of the discharge consent limits imposed by the relevant Water Authority. Where incremental consents have been applied to farms which receive their water supply from a river, the supply often contains a high enough concentration of solids for settlement in the rearing ponds to offset solids production sufficiently for the consent to be met without any settlement prior to discharge. Even relatively clean stretches of river can have suspended solids concentrations in excess of 10 mg/litre (STWA 1984), but borehole supplies usually contain little or no suspended solids. Strict consent limits applied to ammonia and BOD levels on the other hand, may be difficult to satisfy.

CHAPTER 6

THE EFFECTS OF FEEDING LOW POLLUTION DIETS

6.1 Low Pollution Diets - Principles and Commercial Status

As an alternative to treating fish farm effluent, its quality can be improved by manipulating the composition and ration of the diet fed to the fish. BP Nutrition and other feed companies are constantly involved in trying to improve the efficiency of their fish diets. This means increasing the proportion of the food fed to the fish which becomes incorporated in fish biomass, and reducing that proportion which goes to waste either through lack of digestion following consumption or simply through lack of consumption.

The three principal energy sources in animal diets are carbohydrates, fats, and proteins. Undigested proteins and carbohydrates give rise to suspended solids and BOD in the effluent, whilst undigested fats cause BOD and frequently give rise to a visible oily sheen on the surface of the effluent stream which some Water Authorities prohibit. Protein, ideally, should be used to build muscle tissue in the fish and not be used as an energy source. It is not an efficient source of energy, and in order to gain energy from protein the fish must deaminate it and excrete ammonia as a waste product. If sufficient high quality fat or carbohydrate sources are available, much of the protein in the diet will be spared for growth purposes unless the protein ration is excessive, or if the content and proportions of essential amino acids in the protein are poor (Phillips et al., 1964, 1965, 1966, 1967).

As would be expected, different sources of protein, fat and carbohydrate are digestible by fish to varying degrees and the ease with which protein can be assimilated into muscle tissue will vary with its source. For reasons of pollution control as well as conversion efficiency, it is desirable to use a diet which is carefully formulated using highly digestible fat and carbohydrate sources and high quality protein sources. The cost of the diet, however, must not be high enough to negate the advantages gained through its high quality.

Protein is usually the most expensive component of fish diets and therefore optimising protein ration is of great economic importance to feed manufacturers. The optimum level of dietary protein for rainbow trout usually lies in the range 35-50%, the exact figure being dependent on the quality of the protein sources used, diet composition, feeding rate, and the age and physical condition of the fish (Alexis et al., 1986).

The use of single cell proteins (SCP) as a cheap protein source is common practice. Murray and Marchant (1986) found that its use in rainbow trout fingerling diets led to poor nitrogen utilisation efficiency as well as poor growth and conversion. Supplementing the diet with the amino acid L-methionine improved its performance slightly. Anglesea (1982), on the other hand, found ICI's "Pruteen" SCP to be a fairly good protein source in terms of utilisation efficiency, but food conversion ratio (FCR) values were generally poorer than when fishmeal was used as the main protein source.

The principal source of protein in most fish diets is fishmeal. This can vary in quality considerably because the proportion and quality of protein present in fishmeal is influenced by the processing time and temperature and the type of drying technique employed as well as the type of fish used in its manufacture (Piggott, 1982). The amounts of other nutrients present in fishmeal can also vary considerably. Feed manufacturers must therefore spend much time and effort to ensure that the quality of their fishmeal supplies is adequate if effluent control through diet is to be a practical possibility.

A major problem with traditional pelleted dry diets is that they tend to sink rather quickly following feeding. Trout generally do not feed from the bottom of their rearing tanks and so any pellets which reach the bottom of the tanks are unlikely to be consumed. This has detrimental consequences in terms of food conversion efficiency and pollution control. Most feed manufacturers have developed low density pellets using a steam extrusion process. These pellets have much lower settling velocities in water than the traditional high density type and so the proportion of food wasted is lower.

Finally, any dust formed in pellet manufacture will contribute to the pollution load of a fish farm and will not be consumed by the fish. Diet formulation and manufacture must be carefully managed to minimise dust formation.

An important part of this project is to determine the pollutant output resulting from the use of low pollution diets as compared with ranges of published figures for trout farming in general.

Different strains of rainbow trout have been found to display varying efficiencies of dietary protein utilisation. Ming (1985) reported differences in peak ammonia excretion rates of up to 47% between two strains. Little research has been carried out in this area at present though.

6.2 First low pollution diet feeding trial (1985)

6.2.1 Materials and Methods

The first feeding trial from which effluent samples were taken was conducted between 8.3.85 and 7.6.85 at an intensive rainbow trout farm. Effluent samples were taken at the following times:

(i) 7-9th May

(ii) 6th June

In this trial, eight 1200 gallon tanks were stocked with rainbow trout in as similar a manner as was possible. All tanks were fed a fixed, identical ration of feed for the duration of the trial. Three different commercially available low pollution diets, two from BP Nutrition (UK) Ltd and a third from Fulmar, were used and compared for pollution abatement performance. In addition, the farmer calculated food conversion ratios for all tanks over the whole trial and these figures have been included in the results. Food conversion ratio will influence the cost-effectiveness of using a low pollution diet.

Samples of effluent were taken from each tank at different times of day and analysed, in the laboratory, for total ammonia, ATU suppressed BOD and total suspended matter using the tests defined earlier. At all sampling times, a sample of the inlet water to the system was taken. Also, the throughput rate of water for every tank was measured by allowing the inlet water stream to fill a 3 gallon bucket and measuring the time taken with a stopwatch. The following points are important:

(i) In the second sampling session (June) the effluent standpipes of all the tanks were cleaned thoroughly before sampling in an attempt to reduce scatter in suspended solids results.

(ii) Also in the second sampling session, a duplicate of every sample was taken in order to reduce intra-assay variance.

(iii) Some samples were lost due to breakage during refrigeration. In the first batch (May), many were accidentally frozen and their containers shattered.

(iv) Samples were cooled to between 2 and 6°C by placing in a refrigerator on the farm immediately after sampling. Transportation, at ambient temperature, from the farm to the laboratory took about 2.5 hours for the first batch, and about 5 hours for the second.

(v) Pollutant output figures were calculated by multiplying effluent concentrations by flowrates to obtain mass output rates which were then expressed as percentages of the feeding rate. It was assumed that the inlet water was pure since pollutant concentrations in the supply were measured four times daily and were found to be consistently far too low to be measured accurately. The pollutant output figures were compared with ranges of data listed in Chapter 2, Table 2.1.1.

Table 6.2.1.1 shows stocking and feeding data for the eight tanks.

Table 6.2.1.1: Stocking and Feeding Data

3rd May 1985

Tank	Fish No.	Total Mass (Kg)	Mean Fish Wt. (g)
1	1892	416	174
2	1885	427	178
3	1890	422	185
4	1869	424	185
5	2048	437	153
6	2015	410	160
7	2041	374	160
8	2002	394	155

7th June 1985

Tank	Fish No.	Total Mass	Mean Fish Wt.
1	1885	494	262
2	1882	482	256
3	1887	485	257
4	1867	493	264
5	2043	433	212
6	2014	461	229
7	2036	442	217
8	2002	436	218

Tank	Feed
1	Feed 1
2	Feed 1
3	Feed 3
4	Feed 3
5	Feed 1
6	Feed 1
7	Feed 2
8	Feed 2

Total amount fed/day: 3.5 Kg (All tanks)
1st Feed (2 pm) : 1.8 Kg
2nd Feed (4.30 pm) : 1.7 Kg

6.2.2 Results and Discussion

The results of this experiment for both batches are shown in Figs. 6.2.2.1., 6.2.2.2., 6.2.2.3., and 6.2.2.4. Analysis of variance using MINITAB® (Ryan et al., 1981) found no significant differences in pollutant output between any pair of diets in the trial. It is possible that little difference actually existed, but the issue was complicated by the very high degree of intra-assay variance encountered in this trial, particularly with the suspended solids test. It can be seen from the results (Fig. 6.2.2.1) that the scrubbing of the effluent pipes before the sampling of batch 2 reduced this variance to an extent. The solids output figures for batch 2 compare very favourably with the literature range. BOD output figures for both batches were also very low, whereas ammoniacal nitrogen output figures were relatively high, suggesting poor nitrogen utilisation efficiency for all three diets. This is borne out by the rather poor conversion efficiencies obtained for these diets. Because of the high intra-assay variances encountered in this trial, it was decided that in the second feeding trial, in the winter of 1986/87, all water quality tests would be duplicated.

Furthermore, obtaining 30 or so samples in one session necessitated the storage of some samples for over a week prior to analysis. It was decided that in the second feeding trial no more than 12 samples would be taken at any farm visit and all of these would be processed the following day, and several replicate visits would be made.

Fig. 6.2.2.1 Suspended Solids Output (mean+/-S.D.)

(a) Batch 1 (b) Batch 2

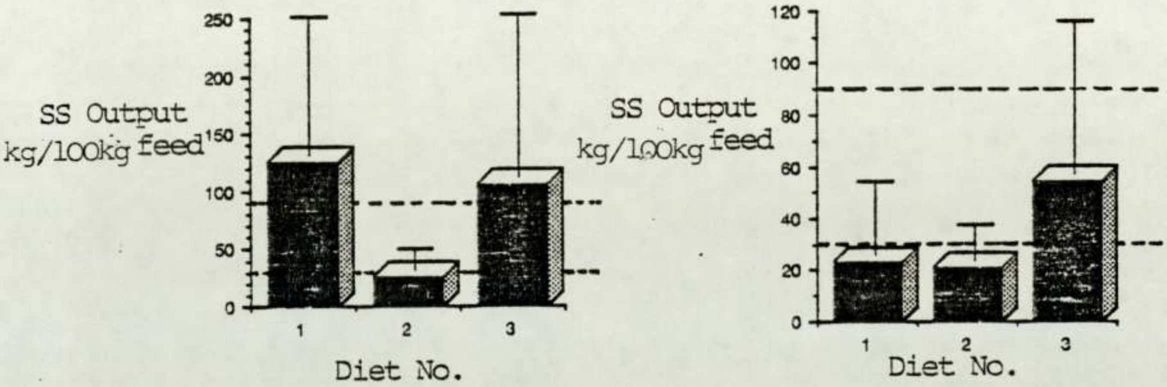
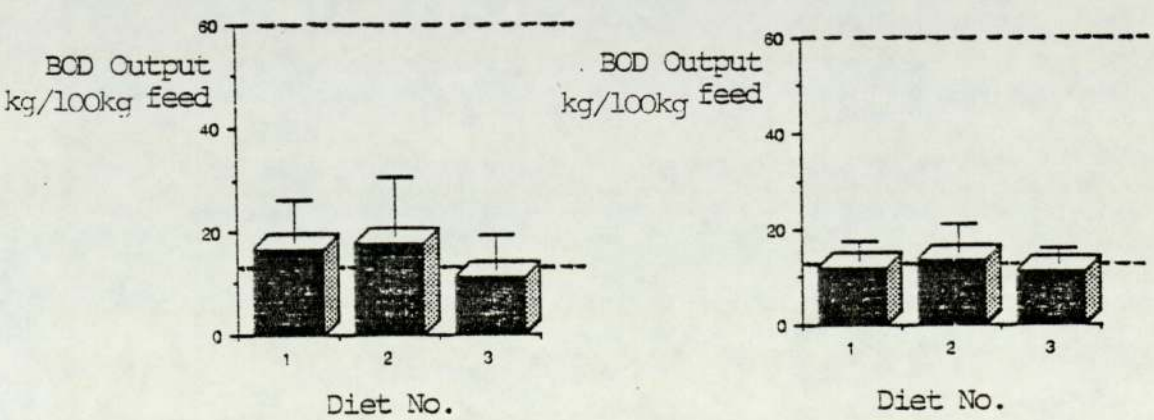


Fig 6.2.2.2 BOD Output (mean+/-S.D.)

(a) Batch 1 (b) Batch 2



Dotted lines denote ranges of literature values

Fig. 6.2.2.3 Ammoniacal Nitrogen Output (mean +/- S.D.)

(a) Batch 1 (b) Batch 2

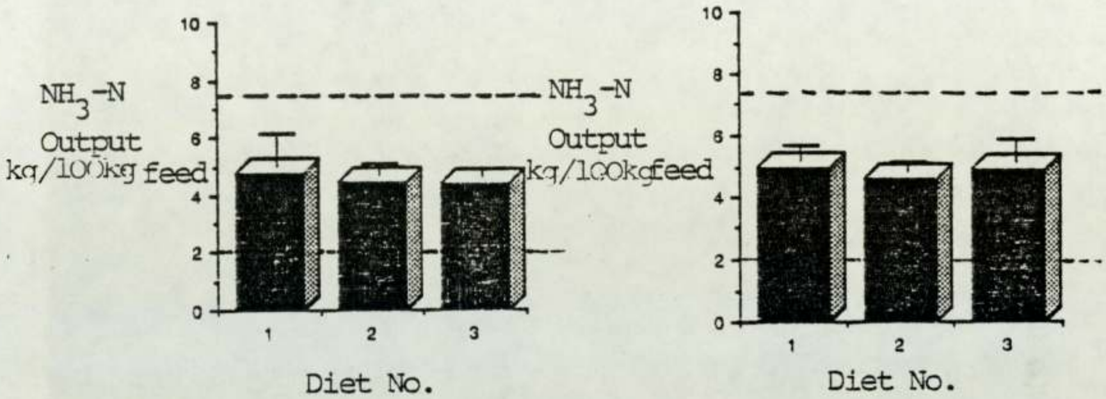
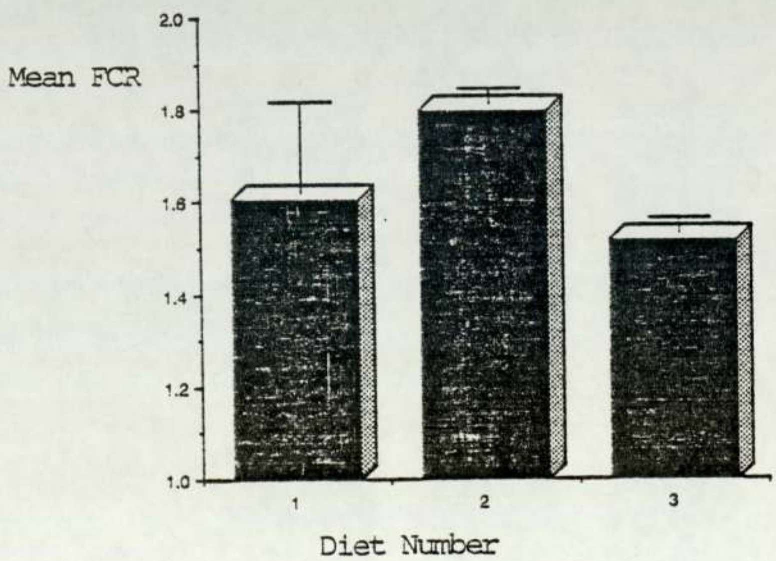


Fig. 6.2.2.4 Food Conversion Ratio (mean +/- S.D.)



Dotted lines denote ranges of literature values

6.3 Second low pollution diet feeding trial (1986/87)

6.3.1 Materials and Methods

In this trial, six tanks were stocked with rainbow trout of as similar weight and in as similar numbers as was possible, as before. The tanks were fed as follows:

(i) Two tanks were fed the first of two commercially available high performance diets supplied by BP Nutrition (UK) Ltd (Diet 1) according to the company's standard feeding guide.

(ii) Another two tanks were fed a second BP diet (Diet 2) according to the feeding guide.

(iii) The remaining two tanks were fed diet 2 with the rations reduced from the feeding guide values by 19%.

The composition of the two diets, as stated by their manufacturer, is shown in Table 6.3.1.1 below.

Table 6.3.1.1

		Diet 1	Diet 2
Crude Protein	, %	46.0	47.0
Oil	, %	12.0	15.0
Ash	, %	12.5	13.0
Crude Fibre	, %	2.0	2.5
Moisture	, %	10.0	8.0
Nitrogen Free Extract	, %	17.5	14.5
Digestible Energy	(Mcal/kg)	3.03	3.5

Diet 2 was an expanded diet manufactured using a steam extrusion process, whereas Diet 1 was pelleted in a conventional pellet mill.

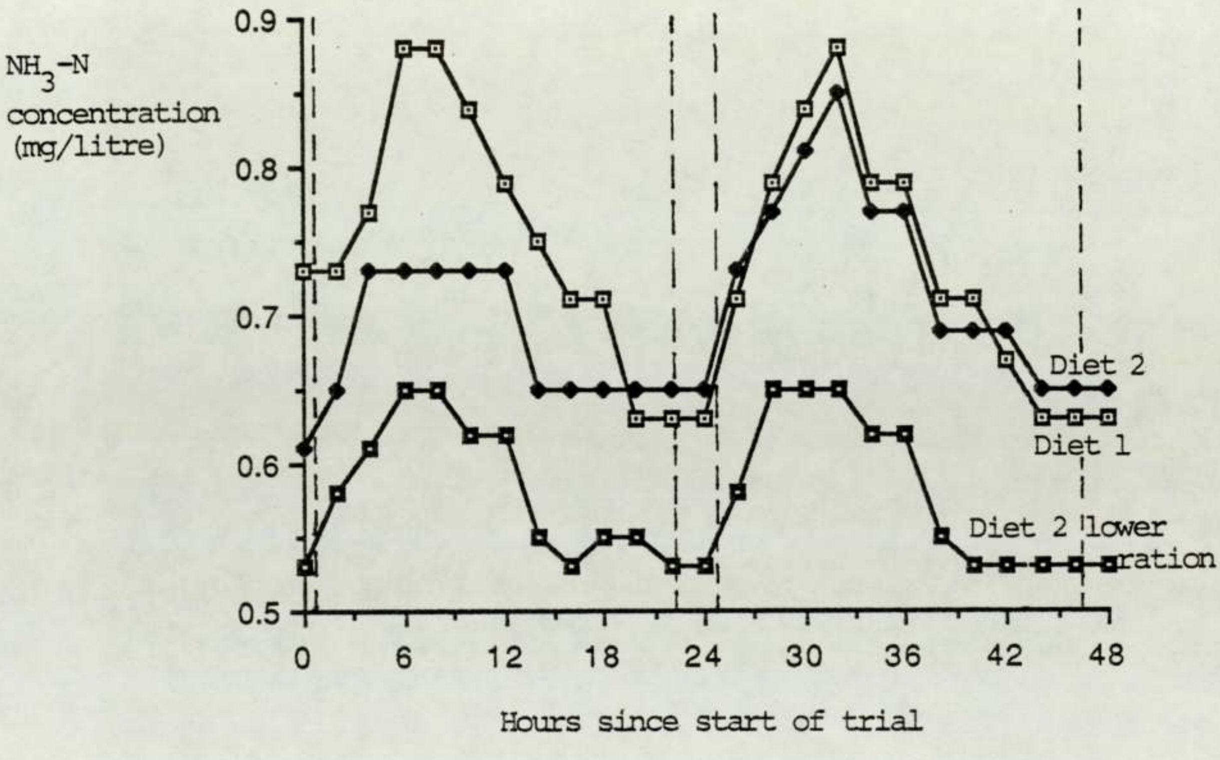
Ammonia levels were monitored every two hours over a 48 hour period after the system had been allowed two weeks to stabilise. This allowed the diurnal variations in ammonia output to be studied, whilst enabling the calculation of accurate mean ammoniacal nitrogen output figures. This approach could not be used in the measurement of suspended solids and BOD output as it was found in the previous feed trial that a massive peak in solids output occurred immediately after feeding. For the measurement of diurnal variations and calculation of mean output figures for suspended solids and BOD to be possible, the simultaneous sampling of the six tanks immediately after feeding with repetition at one minute intervals for the following ten minutes or thereabouts would be necessary.

BOD and suspended solids measurements, therefore, were always made about an hour before the first feed of the day. At this time, the levels of these parameters would be expected to be at a minimum in all six tanks, and not undergoing rapid change. Six visits were made during the 114 day trial period, two in December, a few weeks after the beginning of the trial, two in January, and a final two in February, near the end of the trial. During these visits, 12 samples were taken (i.e. 2 identical samples from each tank) and batches were analysed alternately for suspended solids and BOD, giving a total of 36 BOD measurements and the same number of suspended solids tests spread over the trial period. Output figures were calculated in the same way as in the previous feed trial. After this work, an additional visit was made and another batch of twelve samples was obtained. Each sample was divided into two and one group was tested for BOD in the normal way. The second group of 12 was filtered through Whatman® GF/C glass fibre filter discs before being tested for BOD. In this way, the hypothesis formulated in the previous chapter (i.e. that a large variable component of aquacultural effluent BOD is soluble) was tested.

6.3.2 Results and Discussion

The diurnal variations in ammonia concentration are shown in Fig. 6.3.2.1. Water flowrate was found to vary slightly from one tank to the next and so it was necessary to adjust all concentration figures to a standard flowrate of 1.6 gallons/sec. The inlet water was not found to contain significant amounts of ammonia, suspended solids or BOD. A pronounced diurnal cycle is shown, with ammonia output peaking 8-10 hours after the second feed and at a minimum just before the first feed. This is a much later NH_3 output peak than that observed by Brett and Zala(1975). They were working exclusively with salmon fry, though, which would be expected to have faster metabolic rates than mature rainbow trout. No significant difference in mean output concentration was found between the two diets when fed at similar ration, but as would be expected, cutting the ration of diet 2 by 19% resulted in a significant ($p < 0.01$) reduction by 17.4%. Fig. 6.3.2.4, however, shows that output of ammonia as a percentage of food fed was not significantly changed by reducing the ration. Furthermore, no significant difference was found between the two diets when fed at a similar ration. Table 6.3.1.1 shows them to be of broadly similar composition, although it would be expected that the higher oil content of diet 2 would improve

Fig. 6.3.2.1 Ammonia Concentration in Tank Effluents



Dotted lines denote feeding times

All concentrations have been adjusted to a flowrate of 1.6 gallons/sec

Fig. 6.3.2.2

Suspended Solids Output

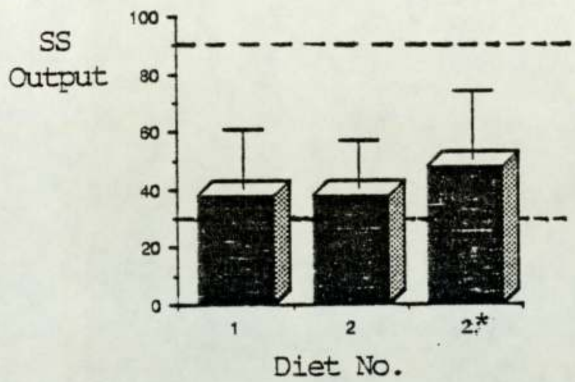


Fig. 6.3.2.3

BOD Output

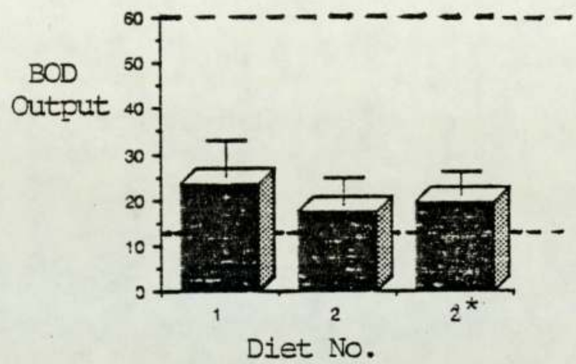


Fig. 6.3.2.4

Ammonia Output

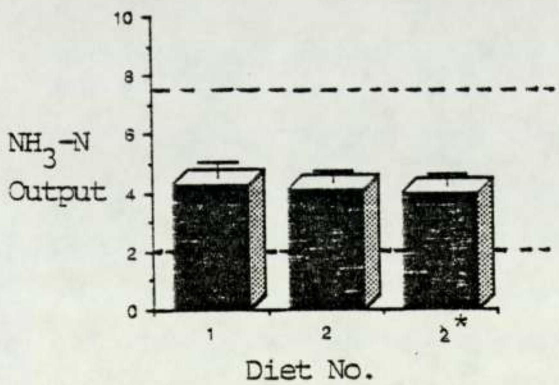
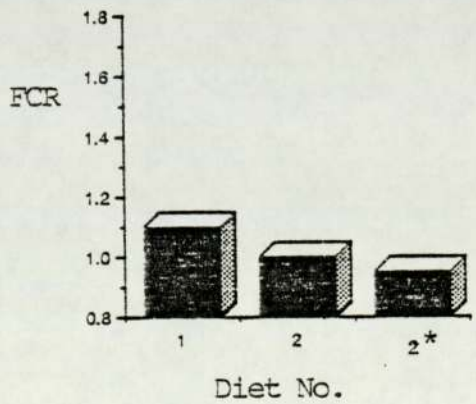


Fig. 6.3.2.5

Food Conversion Ratio



Dotted lines denote literature ranges

All output figures are in kg/100kg food fed(mean +/- S.D.)

its nitrogen utilisation efficiency as compared with diet 1. Table 6.3.2.6 gives stocking and feeding data for the ammonia testing period and the total mass of food fed over the whole trial period.

Table 6.3.2.6 Stock Weight and Feeding Rates during ammonia study

Tank No.	Diet	Mean Fish weight (g)	Total load (kg)	Mass of Food Fed per Day (kg)	Total Food Fed in Trial (kg)
1	1	110	224	2.39	322
2	1	100	202	2.39	347
3	2	94	192	2.39	329
4	2	102	206	2.39	337
5	2*	93	194	2.00	272
6	2*	94	192	2.00	266

* 19% reduced ration

Feeding times: 12 noon and 3 p.m. in two equal rations.

Figs. 6.3.2.2 and 6.3.2.3 show that the two diets do not significantly differ in terms of BOD and suspended solids output, but both compare very favourably with the literature data ranges. It should be borne in mind, however, that the BOD and solids output figures calculated in the experiment are diurnal minima and not averages, for reasons mentioned earlier. The literature figures are averages obtained from variable and often very large numbers of measurements taken in order to characterise fish farm effluent and not specifically to test low pollution diets. The output figures for ammonia, which were of course average figures, did not compare particularly well with the literature figures. Reducing the rations of Diet 2 by 19% did not reduce pollutant output as a proportion of food fed and in the case of suspended solids appeared to significantly increase output. This could only be attributed to experimental error, as the concentrations of solids in the tank effluents were all below 10 mg/litre and the accuracy of the test cannot be guaranteed at these low levels.

Fig. 6.3.2.5 shows FCR to be around 1.0 for all tanks with no great differences, although Diet 2 appeared to perform slightly better, possibly because of its greater oil content and it being an expanded diet. Informal evidence from the U.K. fish farming industry suggests that an FCR figure of 1.0 is very good indeed.

Table 6.3.2.7 below shows that about 60% of the BOD in the effluent from these tanks is soluble, confirming the hypothesis formulated in Chapter 7. The soluble fraction was also found to be very variable, although the limited accuracy of the BOD test may account for much of this variance.

Table 6.3.2.7 Solid and Soluble BOD (mean +/- S.D.)

	Unfiltered	+/-	Filtered	+/-	% Soluble	+/-
Mean BOD_5(ATU) on 12 samples adjusted to flowrate of 7.3 litres/second	2.5	1.04	1.5	0.4	60.0 (p < 0.02)	30.5

figures are in mg/litre.

6.4 Conclusions (from both trials)

(i) Pollutant output figures for low pollution diets compare favourably, although not outstandingly so, with published data. Careful choice of diet, therefore, may benefit the farmer by improving the quality of farm effluent.

(ii) Food Conversion Ratios obtained for the low pollution diets tested in the second trial compare very favourably with those obtained elsewhere in the industry. This will benefit the farmer financially.

(iii) Ammonia excretion peaks 8-10 hours after feeding. Afternoon feeding will therefore result in ammonia output peaking late at night, when likelihood of detection by the Water Authorities is small. This has been exploited by some of the less environment conscious fish farmers.

(iv) Approximately 60% of BOD in the effluent studied in this experiment is soluble. This limits the extent to which BOD can be removed from fish farm effluent within the economic constraints of the industry.

CHAPTER 7

THE USE OF EXPERT SYSTEMS

7.1 Expert Systems Technology

An Expert System is a computer program which explicitly represents human knowledge. It does this by incorporating rules which a human expert would use to draw conclusions from information. Interactive dialogue takes place between system and user, and the user's responses determine what questions the system subsequently asks and what conclusions it finally draws, thus reflecting the reasoning process of a human consultant. An Expert System can also, if required, explicate the chain of logic which led it to a conclusion (d'Agapeyeff, 1984).

The first attempts to simulate human reasoning in a computer took place in the mid 1960s as computer scientists were looking for ways in which to extend database theory beyond simple storage and retrieval of information. By the mid 1970s, the first commercial Expert Systems had been developed. The principles established in the construction of these early systems formed a basis for subsequent development (Addis, 1985).

The two most important components of an Expert System are the knowledge base and the inference engine. The knowledge base contains the actual expertise of the system, including all of the rules used, in the form of a software program. A person who uses the knowledge of human experts to construct knowledge bases is termed a knowledge engineer. The greatest fundamental difference between conventional computer programming and knowledge engineering is that verbal articulacy is of great importance in the construction of knowledge bases. An Expert System which asks poorly or ambiguously worded questions of the user would be of little value (Digital, 1985).

Rules are essentially equations which relate variables of importance in the Expert System's area of application, or domain. In most applications, two distinct types of variable are found. Firstly, there are numeric or object, variables. These take precise numerical values, e.g. a BOD consent limit of 5 mg/litre. Rules involving such variables are simple algebraic equations. The second type of variable is the assertion variable. This defines whether or not something is the case, e.g. whether or not a settlement pond contains algal blooms. These variables can have values of YES or NO, and can also take numerical probability values if an assertion cannot be proved or disproved with absolute certainty. Rules involving assertion variables can take the form of simple equivalence statements, but can involve probabilistic terms and may become quite complex if, for example, Bayesian cumulative probability updating theory (SPL, 1982) is used.

The inference engine executes the knowledge base, providing an interface between its specialised, high level language and the low level language of the computer's compiler. It also performs the "logical chaining" process mentioned earlier to justify conclusions drawn by the System. The construction of inference engines is usually carried out by computer scientists, as considerable specialist knowledge of computing is required (Digital, 1985).

Several Expert System "shells" are available which make the construction of systems relatively straightforward. A shell consists of an inference engine plus a programming language and compiler which are designed to simplify the construction of the knowledge base (Digital, 1985).

Expert Systems technology represents a very efficient method of knowledge dissemination and therefore the potential for its application to industry is considerable. It allows the bulk of an expert's advisory work to be carried out in his absence, leaving him free to concentrate on the more creative aspects of his work. This technology has been applied, with considerable reported success, to some areas of work, (d'Agapeyeff, 1984; Digital, 1985; Hayes-Roth et al., 1983). Much of UK industry, however, remains sceptical about the potential value of Expert Systems technology. d'Agapeyeff (1984) found that Expert Systems are often misconceived as being "inherently complex, risky and demanding".

Several of the more recently developed Expert System shells have a radically different approach to the building of knowledge bases. These establish the key parameters and their relationships of antecedence through interactive dialogue with the possessor of the expert knowledge. The human expert then takes the system through numerous "worked examples" in its domain and informs it of the outcomes. The system has a software routine which collates statistics for the worked examples and generates "rules" which relate the parameters and predict outcomes. With this type of shell, therefore, there is no need to write a computer program. The manufacturers of these shells claim that sizeable Expert Systems can be built in a matter of hours with very little prior knowledge of the shells.

No record was found of any previous application of Expert Systems technology to aquacultural problems. The development of an Expert System containing the knowledge of fish farm effluent control gained through research was, therefore, an important part of this project.

7.2 The First Expert System - MAX

The first Expert System devised within this project, called MAX, was developed on the SAGE shell (SPL, 1982) at the end of 1985. Its purpose was to diagnose likely causes of poor effluent quality following settlement in fish farms. This was constructed before the experimental work on settlement ponds had been carried out and so the knowledge base information was taken from the literature review.

In addition, the MICROEXPERT (ISIS, 1984) and TIMM (developed by Flow General Inc. in 1985) environments were tested by building small prototype effluent control Expert Systems. The MICROEXPERT language proved simple to use, but the presentation of questions and advice was limited to a format defined in the shell. This format involved the use of words and phrases of scientific language and some artificial intelligence jargon words which were judged to be unsuitable for an Expert System which was intended for consultation by fish farmers.

In contrast to the other shells studied, TIMM required no programming as the package contained an interactive program which elicits the key parameters and their relationships of antecedence from the human expert. Data from case histories, giving values of the key parameters and the subsequent outcomes of each case, are input to the system, and TIMM develops rules to predict outcomes using statistical methods. Using the TIMM shell, sizeable Expert Systems could be constructed in a matter of hours with minimal computing knowledge, but as with MICROEXPERT, lack of control over the consultation format made the construction of Expert Systems tailored to suit fish farmers difficult. The greatest and most fundamental problem with TIMM, however, was that the package generated rules by statistical methods with no understanding whatsoever of the physical context to which the data apply. The package thus frequently generated 'nonsense' rules which a human expert's insight would dismiss as simple statistical coincidence.

The SAGE shell is particularly suitable for the construction of "diagnosis" systems, having been developed from a previous shell called EMYCIN, which was in turn developed from the MYCIN Expert System. This was designed in its entirety to diagnose human blood disorders (Hayes-Roth et al., 1983). SAGE gave adequate flexibility of user interface format and the programming language was easy to use.

The SAGE manual was reviewed late in 1984 and a small prototype system was constructed early in 1985 to gain some proficiency in the use of the shell. In the autumn of 1985 the full scale system, MAX, was developed. This has a 740 line knowledge base program involving 57 variables related with 53 rules and took about 40 hours to construct. Where the quality of an effluent is unsatisfactory, it evaluates the likelihood of the reason being one or more of the following:

1. Insufficient water supply to the farm leading to a concentrated effluent.

2. Slack rationing of feed with a considerable fraction entering the effluent stream uneaten.

3. Diet being of poor quality with low digestibility of energy sources and poorly assimilated proteins.

4. Excessive hydraulic surface loading preventing effective settlement in the settling pond.

5. Bad pond inlet design leading to hydraulic short circuiting.

6. Pond shape being conducive to hydraulic short circuiting.

7. Infrequent cleaning of the pond resulting in a thick layer of decomposing nitrogenous sludge producing ammonia in the pond.

8. Blooms of algae giving rise to high levels of suspended solids and BOD in the effluent.

9. Surface crosswinds promoting short circuiting.

10. Poor quality of water supply due to leaching/runoff of inorganic fertilisers, etc.

As a last resort the system will recommend the use of a polymer flocculant to try to enhance settlement.

The above possibilities are considered in the order in which they are listed. This order represents a combination of how often each and every factor does constitute a major cause of poor effluent quality and how easily a remedy can be implemented. For example, slack rationing of feed often leads to uneaten food in the effluent stream. This is a common cause of effluent problems and one which is relatively easy to remedy. This must be considered before advocating any settlement pond design modifications, which may prove expensive. When the system is considering the possibilities in sequence and one possibility appears highly likely, all further possibilities are disregarded and the consultation with the user is terminated. This avoids tiresome evaluation of the more obscure possibilities when an obvious cause has been established. Following the research work carried out in 1986 and 1987 it became necessary to construct a new Expert System as the diagnosis and treatment strategy had changed considerably.

7.3 The Second Expert System System - MAX 2

MAX 2 was developed on the ENVISAGE shell (Systems Designers, 1985) which is basically an improved version of SAGE. Like MAX, MAX 2 diagnoses likely causes of poor effluent quality following settlement, but its knowledge base represents the state of knowledge after the experimental work of this project. It has a knowledge base of about 650 lines which involves 45 variables related with 63 rules. Where effluent quality is found to be unsatisfactory following settlement, MAX 2 evaluates the likelihood of this being attributable to one or more of the following:

1. (a) Slack rationing of feed leading to uneaten food in the effluent stream.

   (b) Feeding too infrequently with fish unable to consume large quantities of feed at feeding times leading to food wastage.

(c) Feeding concentrated in the morning with resultant ammonia output reaching high levels in the afternoon.

2. Unnecessarily small water supply to the farm leading to a concentrated effluent.

3. (a) Poor quality diet with low digestibility of energy sources and poorly assimilated proteins.

   (b) The system advises the use of a low pollution, high performance diet if the user has an ammonia problem and poor FCR.

4. Excessive hydraulic surface loading preventing effective settlement.

5. Poor design of the inlet(s) to the pond promoting hydraulic short circuiting.

6. Excessive fluid velocities in the settlement pond promoting the resuspension of accumulated solids.

7. Contamination of water supply with inorganic fertilisers applied to adjacent farmland.

8. The consents applied to the farm being unreasonable and impossible to satisfy within the economic constraints of commercial aquaculture.

9. The system advises the diversion of water flow from the farm, with possible extra oxygenation if the effluent is too dilute for effective settlement to be possible.

As with MAX, the above possibilities are considered in the order in which they are listed, that order representing a combination of how commonly each and every factor does cause effluent problems in practice and how easily a remedy can be implemented. MAX 2 also terminates the consultation when a possibility appears highly likely. Appendix B shows a log of a typical consultation with MAX 2 together with the hypothetical case history from which it is derived.

Program listings for both MAX and MAX 2 are contained in the floppy disc in the inside back cover of this thesis. Operating instructions for both programs are given in Appendix C.

MAX and MAX 2 are the first Expert Systems to have been devised to tackle an aquacultural problem. Neither of the two systems, however, are marketable as they stand, as fish farmers have numerous other management problems, aside from effluent. Considerable scope exists, therefore, for the development of Expert Systems dealing with disease diagnosis, husbandry, costing, marketing, and other areas of interest to fish farmers, and the combination of these with an effluent Expert System like MAX 2 to produce a large marketable system. Such an Expert System could be run on a mainframe computer at an office of a feed company. The company would probably have to supply the farmers with the necessary hardware, but such information links are becoming cheaper to organise as the current "revolution" in telecommunications progresses.

The system could also be used as a marketing tool for the feed company's products as in the case of ICI's "Wheat Counsellor" system (Digital, 1985).

CHAPTER 8

CONCLUSIONS AND SUMMARY

8. Conclusions and Summary

The main aim of this project has been satisfied in that a cost effective strategy has been developed for the control of fish farm effluent, and that this strategy has been incorporated into an Expert System. Also, Water Authority policy regarding fish farm discharge consents has been thoroughly researched.

The most significant progress has been made in the development of an effective mathematical model which provides a basis for the design of aquacultural settlement ponds. Prior knowledge in this area consisted largely of a number of unreliable, and frequently contradictory, heuristics with weak theoretical bases.

From the results of this research project, the following general conclusions may be drawn:

1. The dilute nature of most aquacultural effluents precludes the use of all but the most basic treatment technologies. The cost effective control of fish farm effluent is therefore achieved primarily through settlement and the use of low pollution diets.

2. The settlement of dilute effluents like aquacultural discharges is limited by the resuspension of accumulated solids by turbulent scouring of the pond bed. This means that effluent suspended solids levels below about 6.0 mg/litre are difficult to attain although the possibility of achieving such effluent clarity can be maximised by keeping the ratio of pond length to retention time (i.e. a form of

mean fluid velocity) as low as possible, preferably below 1.0 metre/minute. This critical factor of turbulent resuspension of accumulated solids was not mentioned in the literature, and so its description in this project represents a considerable contribution to knowledge in this area.

The inappropriateness of retention time as a design parameter for fish farm settlement ponds seriously questions the value of the work of Liao(1970) who suggested that settlement ponds should have retention times in excess of 3 days. The minimum surface area criterion proposed by Warrer-Hansen (1982b) appears more relevant.

3. The removal of BOD from fish farm effluent can only be partially accomplished by settlement. This is because approximately 60% of the BOD is soluble and both the retention times and the temperatures of aquacultural settlement ponds are insufficient to allow significant assimilation of soluble BOD to take place. If retention times of several days, as proposed by Liao(1970), were practicable, then appreciable assimilation of BOD could take place in aquacultural settlement. This would be influenced by retention time and temperature.

4. Settlement is generally ineffective as a means of ammonia removal. It is likely that algal blooms can remove significant quantities of ammonia during periods of rapid biomass gain, but slower growing perennial aquatic plants have insufficient specific rates of ammonia uptake to measurably reduce effluent ammonia levels. Corpron and Armstrong(1983) successfully removed much of the ammonia from their

prawn rearing tanks because growth rates of Elodea densa were clearly adequate at a temperature of 28°C.

5. The use of low pollution, high performance diets can result in relatively low output of suspended solids, BOD and ammonia as compared with published data. Diet control is the only efficient and cost effective means by which the output of ammonia and soluble BOD can be limited. The output rates for ammonia, however, were not particularly low in comparison with the published data.

6. The excretion of ammonia by adult rainbow trout reaches a maximum 8-10 hours after feeding. The restriction of feeding to the afternoon will therefore result in the ammonia peak occurring late at night. This diurnal pattern of ammonia excretion approximately agrees with that found by Brett and Zala(1975) for sockeye salmon. The diurnal profile will vary with fish size.

7. The severity of discharge consents imposed on fish farms can vary greatly from one Water Authority to the next and can also vary considerably within some Water Authority areas. The implementation of Section 41 of the Control of Pollution Act 1974 in August 1985 made possible the extensive review of fish farm discharge consents carried out in this project. This is clearly the first major review of such data.

The problem of fish farm effluent management, therefore, can be solved within the constraints of commercial aquaculture by a combination of optimising settlement pond design and operation and the use of low pollution, high performance diets. The control strategy developed in this project is incorporated in the second Expert System, MAX 2.

APPENDICES

Appendix A Discharge Consent Data (mean +/- S.D.)

Table A.1

No	Authority	Mean SS	+/-	n	Mean BOD	+/-	n	Mean NH_3	+/-	n
1	Severn-Trent	11.0	11.0	13	2.5	2.5	13	0.4	0.15	12
2	Wessex	5.2	2.8	33	3.2	2.3	33	0.54	0.36	33
3	South West	10.0	*	1	9.0	8.3	3	0.45	0.25	3
4	Southern	10.0	0	14	2.0	0	14	0.5	0	14
5	Thames	9.0	3.1	21	3.8	1.25	21	1.4	1.2	20
6	Anglian	20.0	*	1	11.0	*	1	5.0	*	1
7	Yorkshire	5.0	0	9	1.8	0.5	6	0.4	0.1	8
8	Northumbrian	20.0	0	2	10.0	*	1	3.0	2.8	2
9	Welsh	19.0	5.3	7	7.8	6.0	6	1.6	1.9	4
10	North-West	33.0	16.0	6	12.0	10.4	5	1.3	1.14	4
11	Solway	5.0	0	N.A.	2.0	0	N.A.	0.5	0	N.A.
12	Tweed	10.0	0	7	N.C.	-	-	N.C.	-	-
13	Tay	27.5	22.5	14	4.0	0	6	0.6	0	6
14	North East	40.0	44.0	3	7.5	2.5	2	10.0	0	2

Table A.2

No	Authority	No of farms sampled	% operating without consent	Notes
1	Severn-Trent	11	N.A.	Farm list obtained from W.A.
2	Wessex	19	0	
3	South West	9	67	
4	Southern	15	7	Uniform consents for all farms
5	Thames	11	10	
6	Anglian	5	80	
7	Yorkshire	8	0	
8	Northumbrian	4	N.A.	Farm list obtained from W.A.
9	Welsh	8	27.5	
10	North West	10	40	
11	Solway	N.A.	N.A.	Uniform consents for all farms
12	Tweed	7	N.A.	Farm list obtained from R.P.B.
13	Tay	14	N.A.	Farm list obtained from R.P.B.
14	North East	3	N.A.	Farm list obtained from R.P.B.
	TOTAL	114		

Key to symbols

N.A.: Not available
N.C.: No consent limits applied
*: Only one consent found

Appendix B A Consultation with MAX 2
Hypothetical Case Study

The trout farm in question has the following specifications:

Annual production	: 10 tonnes
Mean water throughput	: 1,000,000 gallons/day
Water supply	: river

It has a settlement pond with the following specifications:

Length	: 50 feet
Breadth	: 12 feet
Inlet Specification	: 1 foot width, positioned slightly off-centre
Retention time	: 3 minutes

The farm has the following discharge consent limits set by its Water Authority:

Ammonia	: 0.5 mg/litre absolute maximum
BOD_5 (ATU)	: 3.0 mg/litre incremental maximum
Suspended solids (dried at 105,C)	: 6.0 mg/litre incremental maximum

The fish are fed twice daily using a feeding guide supplied by the manufacturer. The diet fed is of fairly good quality. The water supply flowrate is limited to a maximum of 4500 m^3/day. The river supplying the farm has passed through land used extensively for crop farming. The settlement pond has not been cleaned out for about 18 months and contains a considerable amount of dead vegetation together with a heavy concentration of live pondweed.

The farm has persistently breached its suspended solids consent limit by a wide margin for several months and the Water Authority has threatened the farmer with prosecution. In addition, the ammonia consent limit is frequently breached, although to a lesser extent than the suspended solids consent limit.

On the following pages the farmer's consultation with MAX 2 is logged.

Appendix B

A Consultation with MAX 2

```
ENVISAGE Expert System Executive, copyright (C) Systems Designers 1984.
Version 3.0. Librarian version - release build; created by Librarian 24-JUL-1985
08:39:05
Licensed Site: Aston University

   If you do not know what to do at any time, type HELP

 Current model is __DUA1:[A7DTPJPH]MAX2.ENV;18 - Version 1
 Compiled at 25-Jun-87  14:40

Please type a command : BEGIN
MAX2 will ask you about your effluent problem and will try to find the most cost
effective strategy for its solution. Some of its questions will require you to
reply with a CERTAINTY FACTOR ranging between -5 and +5. Your answer must be an
integer. +5 means: TO A VERY GREAT EXTENT.  -5 means: TO LITTLE OR NO EXTENT.

How much of a problem do you have with effluent suspended solids levels?
(Range is -5 to 5): EXPLAIN

How much pressure are you under from your Water Authority to reduce your solids
output?

Please reply : 5

How much of a problem do you have with effluent BOD levels?
(Range is -5 to 5): -5

How much of a problem do you have with effluent ammonia levels?
(Range is -5 to 5): 3

Is your solids consent incremental?
(You may answer Yes or No): YES

Is your ammonia consent incremental?
(You may answer Yes or No): NO

What is your suspended solids consent limit in ppm ?
(Range is 0.00 to 1000.00): 6.0

What is your BOD consent limit in ppm ?
(Range is 0.00 to 500.00): 3.0

What is your ammonia consent limit in ppm ?
(Range is 0.00 to 100.00): 0.5

Do you wish MAX2 to consider your feeding strategy?
(You may answer Yes or No): YES

How possible is it that you could be wasting some food by slack rationing?
(Range is -5 to 5): EXPLAIN

Overfeeding is a very common cause of solids and BOD problems and it is easily
rectified. The possibility must be ruled out before continuing.

Please reply : 0
```

How many times a day do you feed your fish?
(Range is 0 to 10): 2

Do you use a feeding guide?
(You may answer Yes or No): YES

Do you do all of your feeding in the afternoon?
(You may answer Yes or No): NO

What is your abstraction limit in cubic metres/day ?
(Range is 0.00 to 100000.00): 4500

What is the average flowrate through your pond in gpd ?
(Range is 10.00 or more): 1000000

Do you wish MAX2 to consider the quality of the diet which you use?
(You may answer Yes or No): YES

What is the mean value of FCR on your farm ?
(Range is 0.50 to 3.00): EXPLAIN

Food Conversion Ratio equals mass of food fed divided by resultant gain in fish mass. If you have not assessed this, answer 1.0 and conduct a trial.

Please reply : 1.6

Do you use BP Mainstream High Performance expanded diet?
(You may answer Yes or No): EXPLAIN

BP Nutrition(UK) Ltd produce a very high quality expanded diet which has been shown to improve conversion and reduce pollutant output. Do you use this diet?

Please reply : NO

Do you wish MAX2 to consider the design and operation of your settlement pond?
(You may answer Yes or No): YES

What is the length of your settlement pond in feet ?
(Range is 0.01 to 5000.00): 50

What is the MEAN width of your settlement pond in feet ?
(Range is 0.01 to 3000.00): 12

Does your pond have more than one inlet?
(You may answer Yes or No): NO

What is the breadth of the inlet to your pond in feet ?
(Range is 0.01 to 3000.00): 1.0

How symmetrically positioned is the inlet (or inlets) to your pond?
(Range is -5 to 5): EXPLAIN

Look along the edge of your pond where the inlet(s) lead in. Do the inlet(s) and the pond sides form an even pattern or is there some irregularity in the spacing?

Please reply : 2

What is the retention time of your pond in minutes ?

(Range is 0.10 to 1000.00): 3

How long ago, in months, did you last clean out the accumulated solids from your pond?
(Range is 0.00 or more): 18

Does your pond contain a LARGE amount of aquatic plants?
(You may answer Yes or No): YES

How likely is it that your pond bed could contain a LARGE amount of decomposing vegetation?
(Range is -5 to 5): 1

In the vicinity of your water supply, how common are crop fields?
(Range is -5 to 5): EXPLAIN

Inorganic fertilisers applied to fields of crops can wash into rivers or leach into the water table during rainfall. Are there many well-used crop fields near your supply?

Please reply : 3

Does your water come from a river or stream?
(You may answer Yes or No): YES

CONCLUSIONS AND ADVICE

Try to ration your feeding more strictly as overfeeding is a very common cause of effluent problems and it is wasteful on food.

Your pollutant output will be reduced and you will obtain better conversion if you change to BP Mainstream High Performance expanded diet.

Your pond is too small for your water flowrate. It should have a surface area of at least 113.64 square feet at your current flowrate and should be proportionately greater if any flowrate increase is proposed.

The performance of your pond is being impaired by short circuiting. The total width of inlet(s) to your pond should be increased to at least 1.80 feet. Inlets should be as symmetrically and evenly distributed over the pond width as possible. Also, it is advisable to have as broad and shallow an OUTLET as possible.

The flow cross section area of your pond is too small. This is causing resuspension of accumulated solids. You should increase the mean depth of your p
ond to at least 1.75 feet.
This could probably be accomplished largely by the removal of accumulated solids from the pond bed.

Decomposing vegetation is a major cause of high effluent ammonia levels. Your ammonia problem cannot be solved unless this is removed.

It is likely that fertiliser runoff from nearby fields is contaminating your water supply.Your ammonia consent should be incremental. Ask your Water Authority to check supply ammonia levels and reconsider your consent.

Appendix C Operating Instructions for MAX and MAX 2

C.1 Running MAX

MAX will run on any computer which has the SAGE programming environment. The program listing contained on the floppy disc must be compiled by issuing the command:

SAGECOMP

When prompted for a source file, the filename

MAX.;26

must be entered. Other filenames requested are optional, but the filename stated for the input file must be remembered as this is to be used in the execution of the program. After compilation, MAX is run by issuing:

SAGEEXEC

followed by

MODEL (input filename)

Details of other running commands can be found in the SAGE manual (SPL, 1982).

C.2 Running MAX 2

MAX 2 uses the ENVISAGE shell which is accessed using the command:

ENVISAGESETUP

followed by:

ENVIMENU

The COMPILE option must be chosen and the appropriate filename is:

MAX2.ENV;19

After compilation, the EXECUTE option can be chosen using the filename:

MAX2.SIC

Details of other running commands can be found in the ENVISAGE manual (Systems Designers, 1985).

REFERENCES

References

Abeliovich (1985): Nitrification of Ammonia in Wastewater. Water Res. Vol. 19 (9), 1985, pp1097-1099.

Addis, T. R. (1985): Designing Knowledge Based Systems. Kogan Page Ltd, London 1985, 322pp.

Alabaster (1982): A Survey of Fish Farm Effluents in some EIFAC countries. EIFAC Tech. Paper No. 41, Silkeborg, Denmark 1982, pp5-20.

Alexis, M. N., Theochari, V., Paparaskeva-Papoutsoglou, E. (1986): Effect of Diet Composition and Protein Level on Growth, Body Composition, Haematological Characteristics and Cost of Production of Rainbow Trout. Aquaculture 58, 1986, pp75-83.

Anglesea, J. D. (1982): The Effects of a Dietary Bacterial Protein on Mineral Balance in Rainbow Trout (S. gairdneri Rich, 1836). Ph.D. thesis, Aston University, Birmingham, 1982.

APHA (1975): Standard methods for the examination of water and wastewater. American Public Health Association, Washington DC, 14th Edition, 1975, 1193pp.

Arceivala, S. J. (1983): Hydraulic Modelling for Waste Stabilisation Ponds. Journal of Environmental Engineering, A.I.C.E., 109 (5) 1983, pp265-268.

Bergé, P., Pomeau, Y., Vidal, C. (1984): Order within Chaos - Towards a Deterministic Approach to Turbulence. Hermann, 1984, 226pp.

Bergheim, A., Selmer-Olsen, A. R. (1978): River Pollution from a Large Trout Farm in Norway. Aquaculture, 14, 1978, pp267-270.

Black, P. A. (1986): Aquacultural Engineering - The Properties of Solid Waste Particles. Ph.D. Thesis, Aston University, 1986.

Bohl, M. (1982): Production of Freshwater Fish in the Federal Republic of Germany in relation to the Wastewater Charges Act. EIFAC Tech. Paper 41, 1982, pp.141-147.

Brett, J. R., Zala, C. A. (1975): Daily Pattern of Nitrogen Excretion and Oxygen Consumption of Sockeye Salmon (Oncorynchus nerka) under Controlled Conditions. J. Fish. Res. Board Can. 32, 1975, pp2479-2486.
Burley, R., Klapsis, A. (1985): Flow Distribution Studies in Fish Rearing Tanks. Part 2 - Analysis of Hydraulic Performance of 1m Square Tanks. Aquacultural Engineering 4, 1985, pp113-134.

Codd, G. A. (1985): Eutrophication and Toxic Cyanobacteria in Freshwaters. Water Pollution Control 84, 1985, pp.225-232.

Corpron, K. E., Armstrong, D. A. (1983): Removal of Nitrogen by an Aquatic Plant Elodea densa in Recirculating Macrobrachium Culture Systems. Aquaculture 32, 1983, pp347-360.

d'Agapeyeff, A. (1984): Report to the Alvey Directorate on a Short Survey of Expert Systems in UK Business. Alex d'Agapeyeff, Consultants in Information Technology, London, April 1984, 3pp.

Department of the Environment (1980): Methods for the Examination of Waters and Associated Materials - Suspended, Settleable, and Total Dissolved Solids in Waters and Effluents. HMSO, London, 1984, 27pp.

Department of the Environment (1981): Methods for the Examination of Waters and Associated Materials - Biochemical Oxygen Demand. HMSO, London, 1983, 26pp.

Digital (1985): "The Answer Lies In The Shell". Europa (Digital) Management Report, June 1985, pp2-5.

Douglas, J. F., Gasiorek, J. M., Swaffield, J. A. (1979): Fluid Mechanics, Pitman Publishing Ltd, London, 1979, pp104-106 and pp302-304.

Dryden, H. T., Weatherley, L. R. (1987): Aquaculture Water Treatment by Ion Exchange. Aquacultural Engineering 6, 1987, pp39-68.

Ehrlich, K. F., Cantin, M., Horsfall, F. L. (1987): Bioaugmentation - A New Technique to protect and restore waterways and improve aquacultural production. Paper prepared for presentation at annual meeting of Assoc. Quebec Tech. Eau., Montreal, Quebec, 4-7 March 1987, 14pp.

EIFAC (1964): Water Quality Criteria for European Freshwater Fish. Report on finely divided solids and inland fisheries. EIFAC Tech. Paper No. 1, 21pp.

EIFAC (1968): Water Quality Criteria for European Freshwater Fish. Report on extreme pH values and inland fisheries. EIFAC Tech. Paper No. 4, 24pp.

EIFAC (1970): Water Quality Criteria for European Freshwater Fish. Report on ammonia and inland fisheries. EIFAC Tech. Paper No. 11, 12pp.

EIFAC (1973): Water Quality Criteria for European Freshwater Fish. Report on dissolved oxygen and inland fisheries. EIFAC Tech. Paper No. 19, 10pp.

EIFAC (1987): European Inland Fisheries Advisory Commission Report on the Working Party on Fish Farm Effluents. The Hague, 29-30th May and 1st June 1987, 17pp.

Fauré, A. (1977): Mise au Point sur la Pollution engendree les Piscicultures. Piscic. Fr, (50), 1977, pp33-35.

From, J., Rasmussen, G. (1984): A Growth Model, Gastric Evacuation and Body Composition in Rainbow Trout, Salmo gairdneri, Richardson, 1836. Dana., Vol. 3, 1984, pp61-139.

Gloyna, E. F., Eckenfelder, W. W. Jr. (1970): Water Quality Improvement by Physical and Chemical Processes. University of Texas Press, Austin, Texas, 1970, 321pp.

Haigh, N. (1984): EEC Environmental Policy and Britain, An Essay and a Handbook. Environmental Data Services Ltd, 1984, 324pp.

Hareide, B. (1984): Protection of Surface Water Sources against Pollution - Norway. Water Supply 2, 1984, 35pp.

Harman, J. P. (1978): Characterisation, Treatment, and Utilisation of the Effluent from an Intensive Fish Farm. PhD Thesis, Aston University, 1978, 344pp.

Hawkes, H. A.: Bostock, Hill, Rigby - Consulting Scientist and Analyst, Birmingham B7 4DW

Hawkes, H. A. (1962): Biological Aspects of River Pollution. In River Pollution, Vol. 2, Causes and Effects, L. Klein, Butterworths 1962, pp374-382.

Hayes-Roth, F., Waterman, D. A., Lenat, D. B. (1983): Building Expert Systems. Addison-Wesley Inc., 1983, 248pp.

ISIS (1984): MICROEXPERT user's manual. ISIS Systems Ltd, Surrey, 1984, 129pp.

Ketola, H. G. (1982): Effect of Phosphorus in Trout Diets on Water Pollution. Salmonid 6(2), 1982, pp12,13,15.

Klontz, G. W., Brock, I. R., MacNair, J. A. (1978): Aquaculture Techniques - Water use and discharge quality. Moscow, Idaho, University of Idaho, Office of Water Research and Technology, Project A.054-IDA: 88pp.

Korzeniewski, K., Salata, W. (1982): Effect of Intensive Trout Culture on Chemistry of Lake Letowo Waters. Polskie Archiwum Hydrobiologii, 29 (3-4) 1982, pp633-657.

Larsen, J. L. (1986): Test Use of Bacterial Suspension as Anti-Pollution Agent in Freshwater Aquaculture. Rept. Royal Vet. and Agric., Univ. of Copenhagen.

Liao, P. B. (1970): Pollution Potential of Salmonid Fish Hatcheries. Water and Sewage Works 117 (12) 1970, pp439-443.

Liao, P. B., Mayo, R. D. (1974): Intensified Fish Culture combining Water Re-use with Pollution Abatement. Aquaculture 3, 1974, pp61-85.

Lysén, L. (1984): Protection of Surface Water Sources against Pollution - Sweden. Water Supply 2, 1984, pp40-42.

Mance, G. (1986): Water Quality Standards in Relation to the European Community. Water Pollution Control 85, 1986, pp25-31.

Mangelson, K. A., Watters, G. Z. (1972): Treatment Efficiency of Waste Stabilisation Ponds. J. Sanitary Engineering Division, Proc. ASCE, April 1972, pp407-425.

Mantle, G. J. (1982): Biological and Chemical Changes associated with The Discharge of Fish Farm Effluent. EIFAC Tech. Paper 41, pp103-112.

Markmann, P. N. (1982): Biological Effects of Effluents from Danish Fish Farms. EIFAC Tech. Paper 41, pp99-102.

Matthews, P. J. (1987): Part II of the Control of Pollution Act 1974: What it means. Water Pollution Control 86, 1987, pp140-153.

McLoughlin, J., Forster, M. J. (1982a): The Law and Practice relating to Pollution Control in the UK. Environmental Resources Ltd, 1982, 297pp.

McLoughlin, J., Forster, M. J. (1982b): The Law and Practice relating to Pollution Control in the Member States of the European Community - a Comparative Survey. 2nd Ed., Graham and Trotman, Environmental Resources Ltd., 1982, 326pp.

Ming, F. W. (1985): Ammonia Excretion Rate as an Index for comparing Efficiency of Dietary Protein Utilisation among Rainbow Trout (Salmo gairdneri) of different strains. Aquaculture 46, 1985, pp27-35.

Mudrak, V. A. (1981): Guidelines for Economical Commercial Fish Hatchery Wastewater Treatment. Proceedings of the Bio-Engineering Symposium for Fish Culture. American Fisheries Society, Bethesda, Traverse City, MI, Oct 1979, pp174-182, pub. 1981.

Muir, L. F. (1982): Economic Aspects of Waste Treatment in Fish Culture. EIFAC Tech. Paper 41, pp123-136.

Murphy, J. P., Lipper, R. I. (1970): BOD Production of Channel Catfish. Prog. Fish Cult., 32, 1970, pp4-9.

Murray, A. P., Marchant, R. (1986): Nitrogen Utilisation in Rainbow Trout Fingerling, Salmo gairdneri Richardson, 1836, fed Mixed Microbial Biomass. Aquaculture 54, 1986, pp263-275.

Phillips, A. M. Jr., Poldiolak, H. A., Poston, H. A., Livingston, D. L., Booke, H. E., Pyle, E. A., Hammer, G. L. (1964): Cortland Hatchery Report 32 for the year 1963. Fisheries Res. bulletin 27, pp23-32. State of New York Conservation Dept.

Phillips, A. M. Jr., Livingston, D. L., Poston, H. A. (1965): The Effect of Protein and Calorie Levels and Sources on the Growth of Brown Trout. Cortland Hatchery Report 33 for the year 1964. Fisheries Res. Bull. 28, pp11-19. State of New York Conservation Dept., Albany.

Phillips, A. M. Jr., Livingston, D. L., Poston, H. A. (1965): The Effect of Changes in Protein Quality, Calorie Sources and Calorie Levels upon the Growth and Chemical Composition of Brook Trout. Cortland Hatchery Report 34 for the year 1965. Fisheries Res. Bull. 29, pp6-7. State of New York Conservation Dept., Albany.

Phillips, A. M. Jr., Poston, H. A., Livingston, D. L., (1965): The Effects of Calorie Sources and Water Temperature upon Trout Growth and Body Chemistry. Cortland Hatchery Report 35 for the year 1966. Fisheries Res. Bull. 30, pp25-34. State of New York Conservation Dept., Albany.

Pigott, G. M. (1982): The Role of a Food Engineer in Fish Diet Research and Formula Development. Aquacultural Engineering 1, 1982, pp71-79.

Porcella, D. B., Bishop, A. B. (1975): Comprehensive Management of Phosphorus Water Pollution. Ann Arbor Science Publishers Inc., 1975, 295pp.

Price, G. A., Clements, M. S. (1974): Some Lessons from Model and Full-scale Tests in Rectangular Sedimentation Tanks. Water Pollution Control 73 (1), 1974, pp102-113.

Quellerou, J., Fauré, A., Fauré, C. (1982): Pollution Loads from a Rainbow Trout Farm in Brittany, France. EIFAC Tech. Paper 41 pp87-98.

Rogers, G. L., Klemetson, S. L. (1985): Ammonia Removal in Aquaculture Water Re-use Biofilters. Aquacultural Engineering 4, 1985, pp135-154.

Ryan, A. T., Joiner, B. L., Ryan, B. F. (1981): MINITAB Reference Manual. Duxbury Press, Boston, USA, 238pp.

Snedecor, G. W., Cochran, W. G. (1980): Statistical Methods, 8th Ed. The Iowa State Univ. Press, Ames, Iowa, USA, 433pp.

Solbé, J. F. de L. G. (1982): Fish Farm Effluents; A United Kingdom Survey. EIFAC Tech. Paper 41 pp29-56.

SPL (1982): An Introduction to the SAGE Expert System. Knowledge Engineering Group, Systems Programming Limited, April 1982, 331pp.

Stabel, H. H., Geiger, M. (1985): Phosphorus Adsorption to Riverine Suspended Matter. Water Res. 19 (11), 1985. pp1347-1352.

STWA (1983): Water Quality 1982/83. A report by the Water Quality Advisory Panel on the quality of water supplied, the quality of effluents discharged, and the quality of rivers within the Severn-Trent Water Authority's area. Severn-Trent Water Authority, Birmingham 1983, 176pp.

STWA (1984): Water Quality 1983/84. A report by the Water Quality Advisory Panel on the quality of water supplied, the quality of effluents discharged, and the quality of rivers within the Severn-Trent Water Authority's area. Severn-Trent Water Authority, Birmingham 1984., 183pp.

Sumari, O. (1982): A Report on Fish Farm Effluents in Finland. EIFAC Tech. Paper 41 pp21-28.

Systems Designers (1985): ENVISAGE Reference Manual. Knowledge Engineering Group, 1985, 236pp.

Thomas, L. C., Chamberlain, G. J. (1980): Colorimetric Chemical Analytical Methods. The Tintometer Ltd, Salisbury, 9th Ed 1980, 646pp.

Toms, I. P., Owens, M., Hall, J. A., Mindenhall, M. J. (1975): Observations on the Performance of Polishing Lagoons at a Large Regional Works. Water Pollution Control 74 (4), 1975, pp383-401.

Trojanowski, J., Trojanowska, C., Ratajczyk, H. (1982): Effect of Intensive Trout Culture in Lake Letowo on its Bottom Sediments. Polskie Archiwum Hydrobiologii 29 (3-4), 1982, pp659-670.

Warrer-Hansen, I. (1982a): Evaluation of Matter Discharged from Trout Farming in Denmark. EIFAC Tech. Paper 41, pp57-64.

Warrer-Hansen, I. (1982b): Methods of Treatment of Waste Water from Trout Farming. EIFAC Tech. Paper 41, p113-122.

Water Research Centre (1976): Emission Standards in relation to Water Quality Objectives. Water Research Centre Technical Report 17, June 1976, 36pp.

Weatherley, L. R. (1983): Protein via Fish Farming. Chemtech. October 1983, 7pp.

Webb, L. J. (1985): An Investigation into the Occurrence of Sewage Fungus in Rivers Containing Papermill Effluents. IV: Growth in Experimental Channels dosed with Effluents and Pure Compounds. Water Res. 19 (8), 1985, pp969-974.

Wheaton, F. W. (1977): Aquacultural Engineering. Wiley-Interscience, 1977, 708pp.

White, M. J., Baskerville, R. C., Day, M. C. (1976): Increasing the Capacity of Sedimentation Tanks by means of Sloping Plates. Water Pollution Control 75 (1), pp21-30.

Wild, H. E., Sawyer, C. N., McMahon, T. C. (1971): Factors affecting Nitrification Kinetics. J Wat. Pollution Control Federation. 43 (9), pp1845-1854.

Willoughby, H., Larsen, H. N., Bowen, J. T. (1972): The Pollutional Effects of Fish Hatcheries. Am. Fishes US Trout News 17 (3), pp6-7, 20-1.

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Optimising the Removal of Suspended Solids from Aquacultural Effluents in Settlement Lakes

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ABSTRACT

The effectiveness with which settlement ponds remove suspended solids from fish farm effluents can vary considerably from one pond to the next. A total of 16 settlement ponds on 11 UK commercial freshwater farms were studied in an attempt to relate performance in terms of solids removal to design and operating parameters

The results showed that it is difficult to achieve solids concentrations of less than 6 mg litre^{-1} through simple settlement although the probability of attaining such an effluent clarity can be optimised by keeping mean fluid velocity below 4 m min^{-1} and, if practicable, below 1 m min^{-1}.

NOMENCLATURE

A	Area of pond base (m^2)
d^*	Hydraulic mean diameter of pond (m)
D	Axial dispersion coefficient ($m^2\ s^{-1}$)
L	Pond length (m)
Q	Volumetric flowrate ($m^3\ min^{-1}$)
Re	Reynolds Number (dimensionless)
SS_i	Suspended solids concentration in influent (mg litre^{-1})
SS_o	Suspended solids concentration in effluent (mg litre^{-1})
ΔSS^*	Deviation of resuspension factor from mean (mg litre^{-1})
TSV	Terminal settling velocity of particle (m s^{-1})
$\bar{U}$	Mean fluid velocity (m min^{-1})
W	Mean pond width (m)

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%Δ	% decrease in solids concentration on passage through settlement pond (dimensionless)
μ	Dynamic viscosity of water (Ns m^{-2})
ρ	Density of water (kg m^{-3})
τ	Retention time of pond (min^{-1})

INTRODUCTION

The problem of controlling effluent quality is one of increasing importance to UK fish farmers, particularly those whose farms discharge into streams and rivers. The Control of Pollution Act, 1974, empowers the ten Water Authorities in England and Wales and the seven Scottish River Purification Boards to impose conditions regarding the quality and quantity of all industrial discharges, and prosecute those who breach their assigned limits.

The principal parameters controlled by the authorities are suspended solids (dried at 105°C), BOD (5 day) and ammoniacal nitrogen. This paper deals only with the treatment of solids. The other two factors are equally important but are not considered here.

The solids present in aquacultural effluents consist principally of faeces and waste food (Warrer-Hansen, 1982). Figures quoted for suspended solids output as a proportion of food fed range from 0·3 (Muir, 1982) to 0·65 (Harman, 1978) for intensively reared rainbow trout. Using the lower figures, this means that every tonne of trout fed at a ration of 1·5% body weight produces an average of 4·5 kg day^{-1} of suspended solids. The Water Authorities' consent limits on this parameter are typically of the order of 10 mg $litre^{-1}$ and some are much lower; hence the necessity to control the level of suspended solids in aquacultural effluent.

THE REMOVAL OF SOLIDS

In commercial aquaculture, effluent treatment almost always consists of settlement in a simple lagoon. Solbé (1982) found that, in 1980, approximately 62 out of a total of 367 UK freshwater farms surveyed treated their effluent prior to discharge, and in almost every case in which effluent treatment took place it was by means of a settlement pond. Other technologies are available for the removal of suspended solids from effluents but these have failed to find commercial application. The main reason for this appears to be one of costs.

Harman (1978) found that, in comparison with sand filtration, microstraining and air flotation, simple sedimentation was the most cost effective way of reducing suspended solids concentration to below 10 mg $litre^{-1}$ in the effluent from an intensive trout farm. One alternative solids removal device which has found widespread application in Denmark, though not in the UK, is the swirl-concentrator. It has been claimed (Warrer-Hansen, 1982) that this is a cost effective alternative to settlement where landspace is at a premium, but no performance data were given. In practice in Denmark, swirl-concentrators are usually used in conjunction with settlement ponds, often preceding them. A company in Jutland manufactures these devices in stainless steel for sale to fish farmers. Even in Denmark, therefore, the simple settlement pond is widely used to treat fish farm effluent.

THE DESIGN AND OPERATION OF SETTLEMENT PONDS

The efficiency with which settlement ponds remove suspended solids from aquacultural effluents can vary greatly from one pond to the next. Attempts to relate performance to design and operation parameters, however, have yielded widely differing, and often contradictory, results.

The majority of workers have concentrated on retention time as a determining factor in settlement pond performance. Liao (1970) found that settlement ponds with retention times in excess of 3 days could remove over 60% of the suspended solids from hatchery effluent and that shorter retention times gave considerably poorer results. Such high retention times would be very difficult to achieve within the constraints of commercial aquaculture and some workers (Toms *et al.*, 1975; Harman, 1978) have obtained good results using ponds with retention times of only a few hours.

Settlement pond geometry has also been cited as an important factor. Arceivala (1983) claimed that the ratio of length to width should be higher than 4, and ideally above 8, to optimise performance. Increasing this ratio decreases the value of dispersion number ($D/\bar{U}L$) thus promoting the approximation of 'plug flow' conditions in the pond. Mangelson and Watters (1972) performed tracer studies on a number of lagoon designs and found that the mean residence time of a pond could be increased by as much as 75% by the installation of baffles to increase the length–width ratio of the flowpath.

From basic fluid mechanics theory (Douglas *et al.*, 1979) it is apparent that the formation of 'dead' areas which are by-passed by the bulk of the fluid flow in a system is often associated with a sudden increase in flow cross-sectional area; this often occurs where aquacultural effluent flows into a settlement pond from a relatively narrow conduit. Pond inlets should, therefore, be as broad and as symmetrically located as possible relative to the pond's width to minimise this wastage of pond area.

Warrer-Hansen (1982) gives an outline of a design procedure for settlement ponds which uses surface area as the critical design parameter. The main underlying premise is that the smallest particle size for which exclusion from the final effluent is desirable corresponds to a particular settling velocity, and that all larger particles will settle faster (Fig. 1). An elevation of an idealised settlement pond is shown in Fig. 1. The plane *AA′* links all particles of the smallest size to be considered. This could be chosen, for example, as the size which is smaller than the particles which form the largest 99 wt% of the expected size distribution. This plane descends with a velocity, *TSV*, the terminal settling velocity of these particles. As it does so, 'clean' effluent is generated at a rate of X m^3 s^{-1} where

$$X = TSV.A\,(\mathrm{m^3\,s^{-1}}) \tag{1}$$

For removal of 99 wt% of the solids loading to be possible, the volumetric flowrate, Q, must not exceed X. This means that, for a given value of *TSV*, surface area A must exceed a critical value.

Price and Clements (1974) found that considerable deterioration in performance of sewage settlement ponds resulted from strong winds blowing *perpendicular* to the direction of water flow. Winds blowing parallel to the direction of flow, either co-currently or counter-currently, had little effect on performance. The authors established that crosswinds

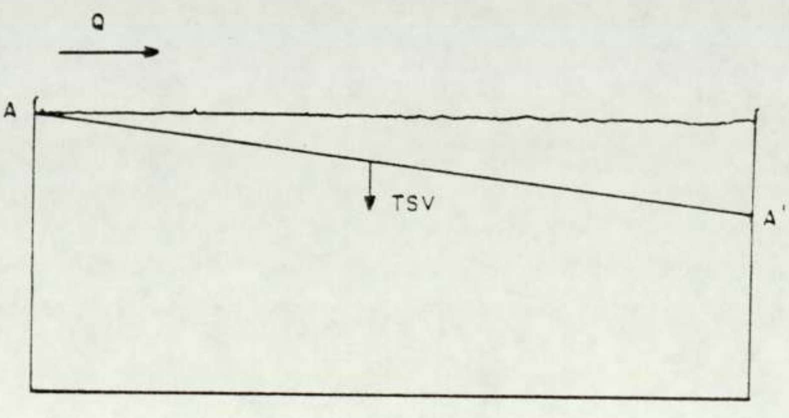

Fig. 1. Elevation of idealised settlement pond with a volumetric throughput rate of Q m^3 min^{-1}. The smallest particles considered form plane *AA′* which is descending with velocity *TSV* m s^{-1}.

promote the formation of a 'dead' area near the upwind side of settlement ponds.

Mudrak (1981) studied settlement ponds at four salmonid hatcheries in the US and found the performance of concrete ponds to be far superior to earthen ponds of similar retention time, but did not offer any explanation for this.

MATERIALS AND METHODS

In order to establish the effects, if any, of the aforementioned parameters on settlement pond performance, a total of 16 ponds (numbered 1–16) on 11 UK commercial freshwater fish farms were studied. A whole day was devoted to the study of each pond so that water samples could be fully analysed within 24 h of sampling. The following variables were measured:

- (i) pond length
- (ii) pond breadth (mean estimated for irregular ponds)
- (iii) width of inlet(s) to pond
- (iv) pond retention time (by observing the passage of a tracer of 'malachite green' (CI 42000) introduced to the pond inlet at about 1 g per 10^6 gpd flow).

In addition, the following were noted:

- (i) volumetric flowrate through pond (rough estimate obtained from farmer)
- (ii) direction of alignment of pond (where applicable)
- (iii) dimensions and positioning of baffles (if any)
- (iv) forecasted wind speed and direction
- (v) material(s) of construction of the pond
- (vi) other factors, e.g. presence of fish, algae, etc.; exceptional weather conditions

Four replicate water samples were collected at both the inlet and outlet of all the ponds and stored for later analysis. The sampling from the inlet and outlet were separated by the measured pond retention time to ensure comparability under input conditions which may be liable to rapid change (e.g. during feeding). All samples were analysed for total suspended matter dried at 103–105°C (APHA, 1975). This test consists of filtering the sample through a Whatman® GF/C glass fibre filter disc which is then dried for 1 h at 103–105°C and weighed.

TABLE 1
Pond Design and Operating Characteristics

Pond No.	*Length (L) (m)*	*Width (W) (m)*	*Flowrate (Q) ($m^3 min^{-1}$)*	*Retention time (τ) (min)*	*Wind direction and speed ($m s^{-1}$)*	*Pond alignment*	*Material of construction*	*Inlet(s) spec.*	*Baffle(s) spec.*	*Notes*
1	1250	8	56·8	90	NE 13·5	Irregular	Earth	2 × 36 in pipes	1 island	Very irregular. Some sudden expansion
2	500	22	25·3	90	W 4·5	N/S	Natural lake	2 m channel	3 island chain	Stocked with fish. Heavy rain
3	229	59	47·4	80	S 4·5	Irregular	Natural lake	9 m channel	Small island	Rearing pond being cleaned out
4	195	15	78·9	35	S 3	ENE/WSW	Earth	4·5 m channel	1 island	—
5	80	9	25·3	8	SW 3·5	NNE/SSW	Earth	3 m channel	—	Steep, unstable earth banks
6	66·5	18·5	25·3	20	S 4·5	NE/SW	Earth	3 m channel	—	—
7	52	3	2·4	150	SW 4·5	E/W	Earth	3 m channel	—	Wooden barrier retaining algal bloom
8	43·5	10·3	25·3	5	SW 3·5	NNE/SSW	Earth	2·2 m channel	—	—
9	32	19	11·0	15	W 1·5	E/W	Earth	7 × 6 in + 3 × 9 in pipes	—	Decomposing leaves on pond bed
10	22	15	25·3	3	SW 2	NNE/SSW	Earth	2·2 m channel	—	Designed to promote 'swirling' motion
11	18·5 diameter		1·6	10	Light var	Circular	Earth	1·5 m channel	—	Designed to promote swirling
12	18·5	6	6·3	5	Light var	E/W	Earth	4 in pipe + 2·2 m channel	—	Fish present in pond
13	14 diameter		9·5	30	Light var	Circular	Earth	2·2 m channel	—	Designed to promote swirling
14	8·9	3·7	2·2	10	W 6·5	N/S	Concrete	0·4 m channel	—	Fish present in pond
15	2·5 diameter		0·12	20	W 9	Circular	Iron	6 in pipe	—	Poor plug flow approximation
16	2·5 diameter		0·12	10	SW 9	Circular	Iron	6 in pipe	3 filter screens	—

DESCRIPTION OF SYSTEMS STUDIED

Details of the design and operating parameters for all 16 ponds are shown in Table 1.

POND PERFORMANCE

The performance of the 16 ponds in terms of solids removal is shown in Figs 2–4.

RESULTS AND DISCUSSION

The removal of suspended solids correlated most strongly with inlet solids concentration. Figure 5 shows the inlet/outlet concentration

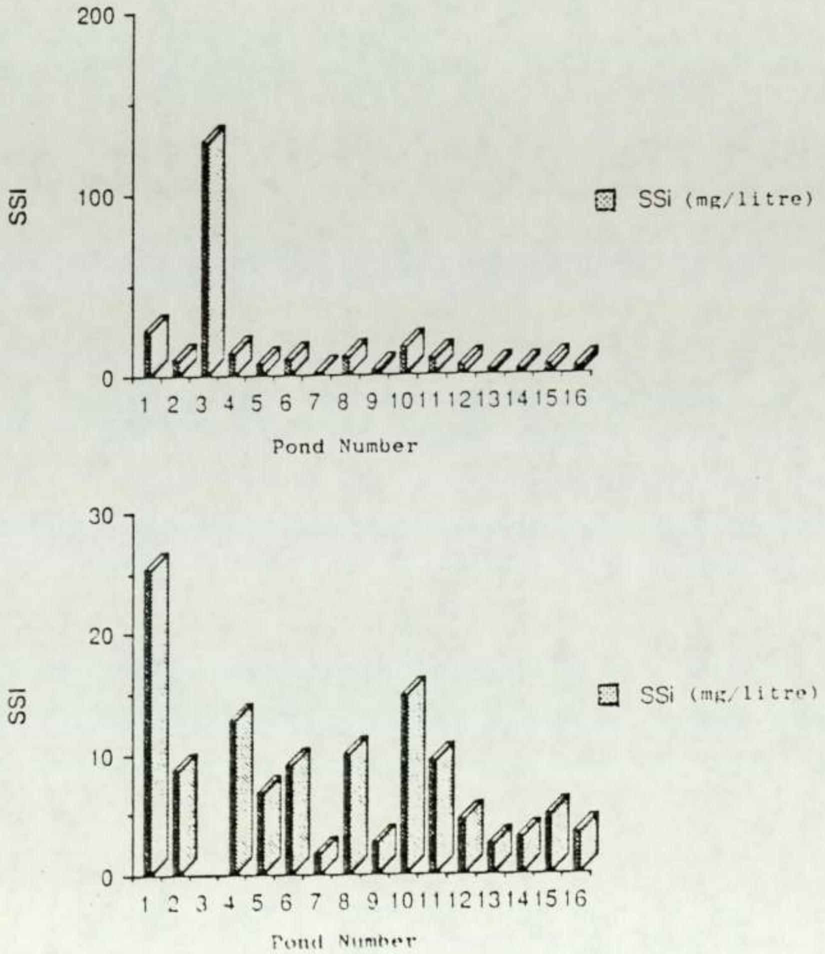

Fig. 2. Inlet suspended solids concentration (mg litre^{-1}) for all ponds studied, numbered as in Table 1.

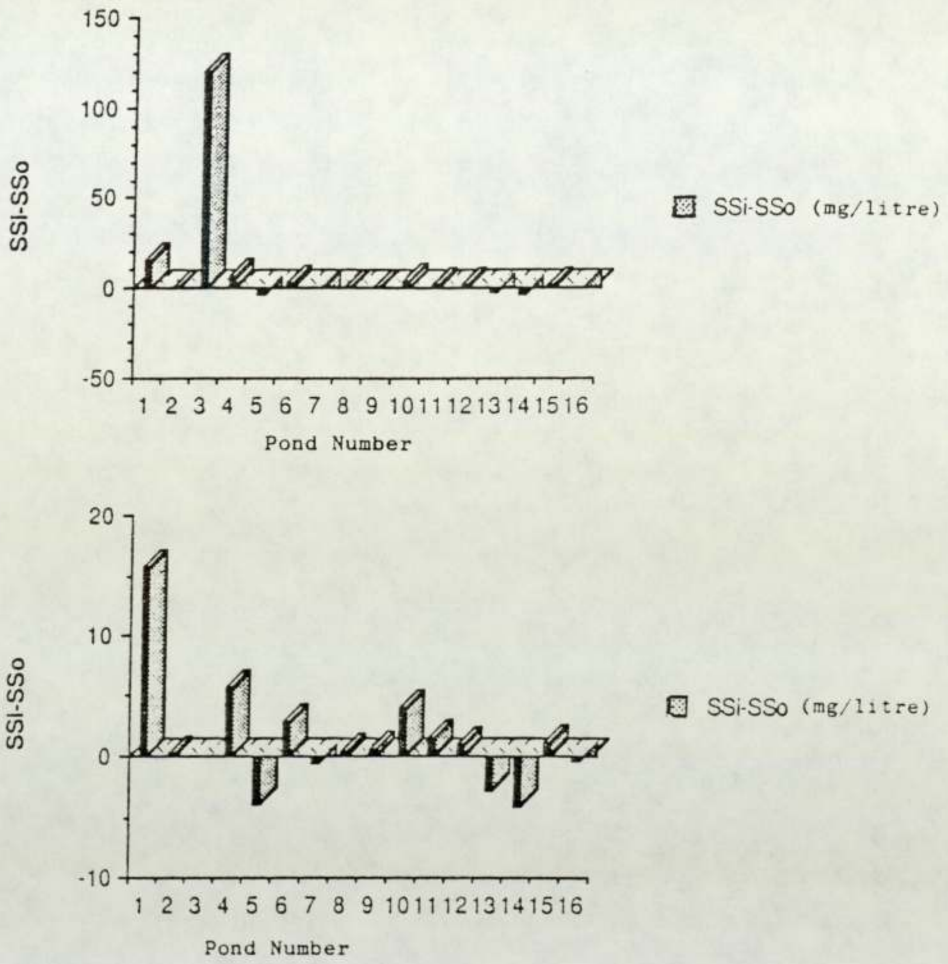

Fig. 3. Inlet solids concentration minus outlet solids concentration for all ponds in mg litre^{-1}.

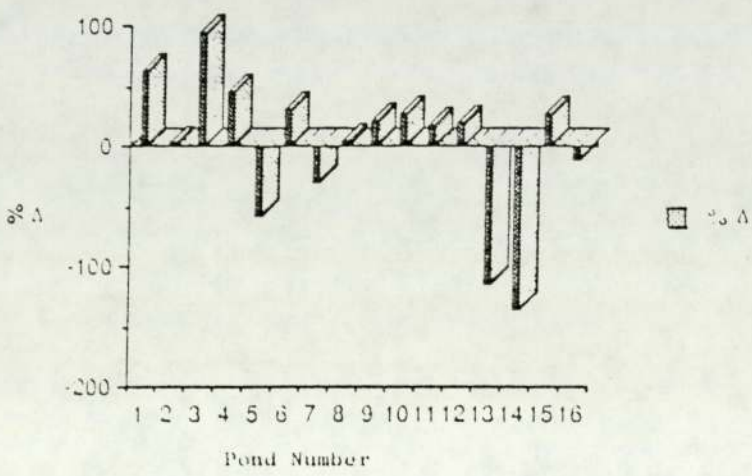

Fig. 4. Percentage removal of solids in all 16 ponds, defined as the inlet outlet solids difference (Fig. 3) expressed as a percentage of inlet solids concentration (Fig. 2).

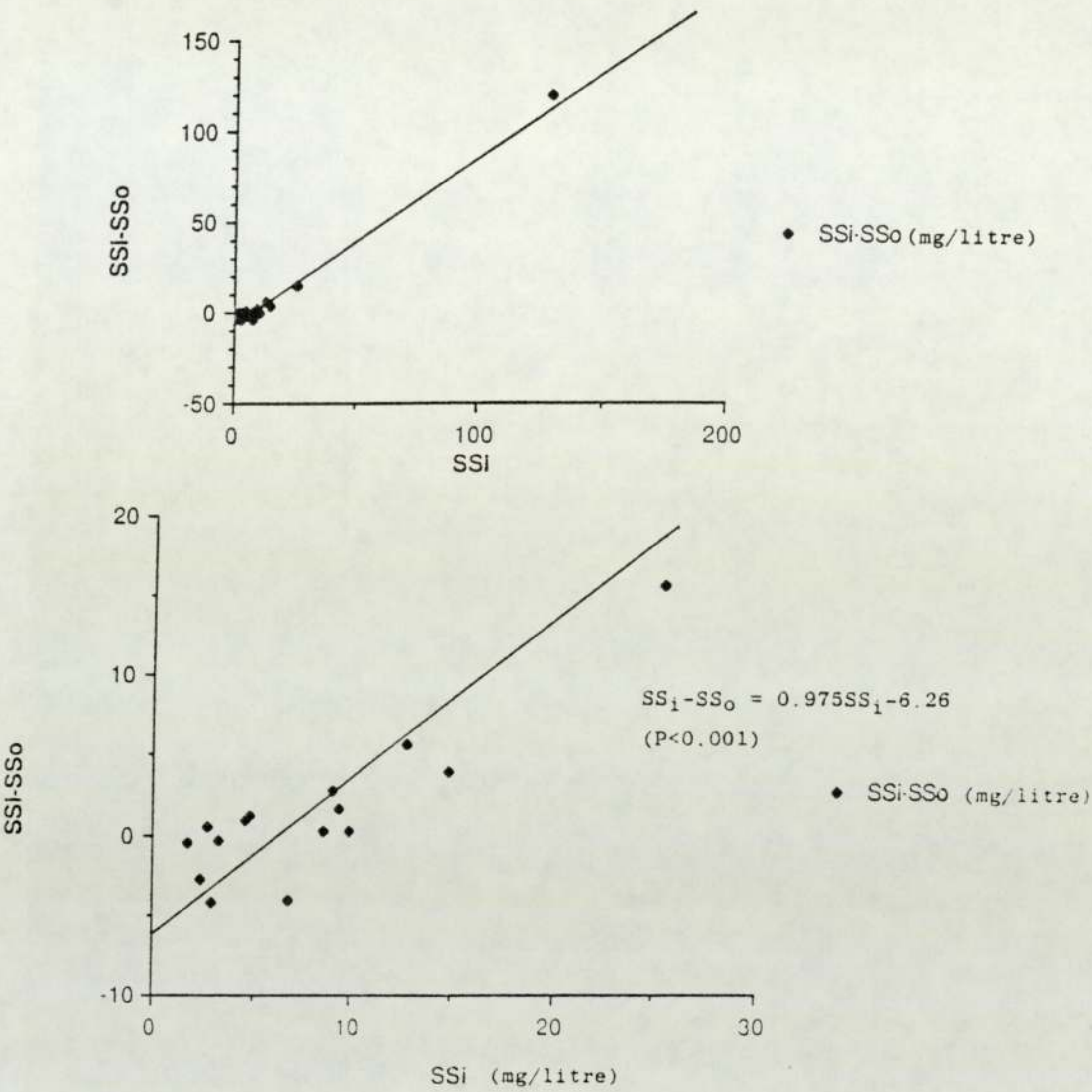

Fig. 5. Inlet/outlet difference (Fig. 3) versus inlet solids (Fig. 2) (both in mg litre^{-1}). The linear regression: $SS_i - SS_o = 0{\cdot}975\,SS_i - 6{\cdot}26$, is significant at $P < 0{\cdot}001$.

difference to relate to inlet concentration according to the function:

$$SS_i - SS_o = 0{\cdot}975\,SS_i - 6{\cdot}26 \qquad (2)$$

The fact that net increases in suspended solids concentration can occur indicates that resuspension of settled solids takes place in the ponds. The extent of this is such that, if the initial suspended solid levels in all of the ponds had been zero, we would expect the mean of all the outlet concentrations to be 6·26 mg litre^{-1}. This resuspension could only be due to the scouring of pond beds by fluid turbulence. The value of the coefficient of SS_i in eqn (2), being so close to unity, indicates that hindering of settlement is negligible, and if no resuspension took place, the ponds would, on average, remove 97·5% of their solids loading. This relationship also implies that it is very difficult to achieve any net reduction solids concentration once the critical value of 6·26 mg litre^{-1} has been reached.

The decrease in suspended solids concentration expressed as a percentage of the inlet value is a more meaningful measure of efficiency, but suffers from great scattering of the data points where SS_i is low. This is shown in Fig. 6. From eqn (2) the following expression can be derived:

$$\left(\frac{SS_i - SS_o}{SS_i}\right) \times 100 = \%\Delta = 97{\cdot}5 - \frac{626}{SS_i} \qquad (3)$$

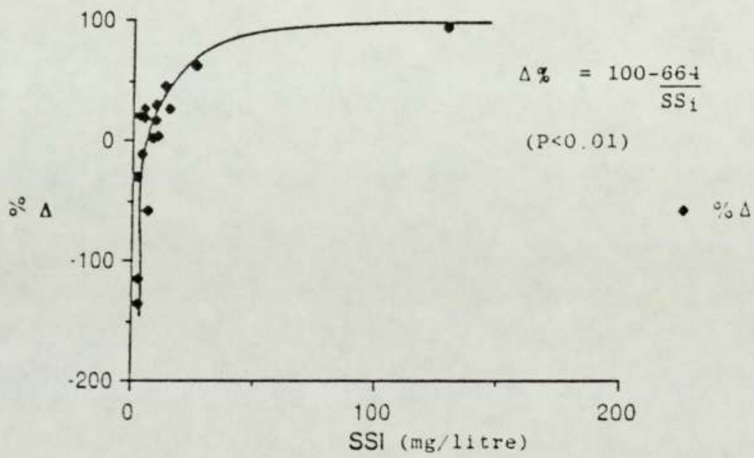

Fig. 6. Percentage removal of solids (Fig. 4) versus inlet solids (Fig. 2) in mg litre^{-1}. The asymptotic regression: $\%\Delta = 100 - 664/SS_i$, is significant at $P < 0{\cdot}01$.

An independent regression, assuming that 100% removal is possible at infinite SS_i, gives the similar equation:

$$\%\Delta = 100 - \frac{664}{SS_i} \qquad (4)$$

Although the removal of solids appears to correlate well with initial solids concentration, considerable discrepancies nevertheless exist, suggesting that some other factor or factors have significant influence on the performance of settlement ponds. As mentioned earlier, resuspension could only be caused by scouring of the pond bed by fluid turbulence. It would be expected, then, that high values of Reynolds Number would promote resuspension and cause settlement to be relatively poor for a given value of SS_i. Reynolds Number (Re) was calculated for all ponds except numbers 11, 13, 15 and 16 which are circular and therefore have non-linear flow geometry for which no Re could be defined which is comparable to that of the linear ponds. Re was calculated using the hydraulic mean diameter, d^*, with the equation:

$$\mathrm{Re} = \frac{\rho d^* \bar{U}}{\mu} \tag{5}$$

where:

$$d^* = \frac{4\,(\text{flow area})}{\text{wetted perimeter}} \tag{6}$$

and

$$\bar{U} = L/\tau \tag{7}$$

This was compared with the parameter ΔSS^* which is defined as follows:

$$\Delta SS^* = 0{\cdot}01\,(\%\Delta\ \text{actual} - \%\Delta\ \text{eqn}\ (4))\,SS_i \tag{8}$$

This expresses deviation of the resuspension factor from the mean value according to eqn (4) of 6·64 mg litre^{-1}. It did not, however, show any significant correlation with Re (Figs 7 and 8).

This can be explained by resorting to classical fluid mechanics theory (Foust *et al.*, 1960). In any system where fluid flow is bounded by a solid wall, increasing Re influences momentum transfer at the fluid–solid interface in turbulent flow regimes by decreasing the thickness of the laminar sub-layer, thus exposing more solid irregularities to turbulent

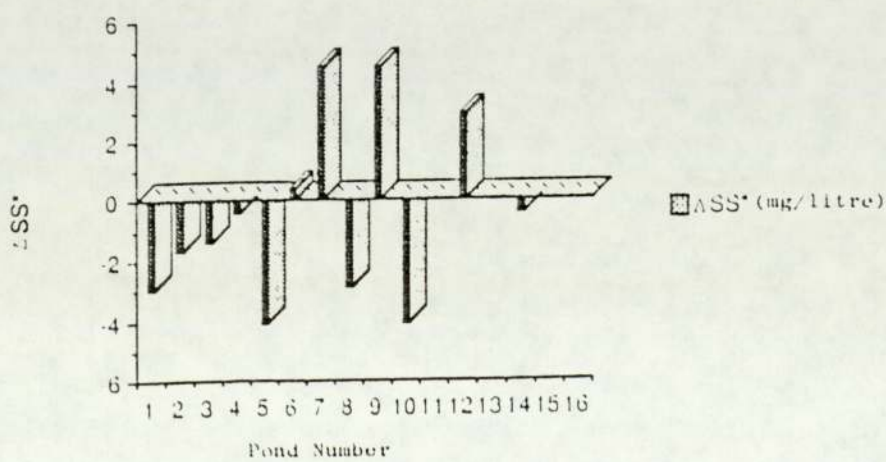

Fig. 7. Deviation of suspended solids removal ($SS_i - SS_o$) from that predicted using eqn (4), as used in Fig. 9.

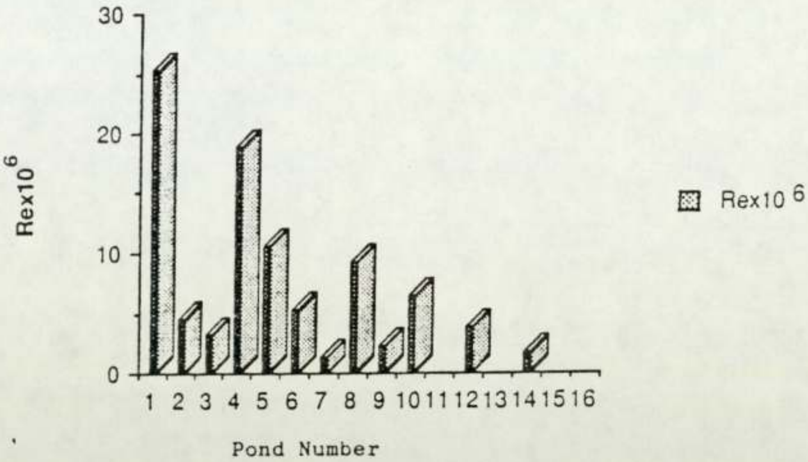

Fig. 8. Values of Reynolds Number for all ponds studied.

fluid flow. Once Re has reached a sufficiently high value, however, the laminar sub-layer thickness is so small compared to the size of the irregularities that further Re increase does not *per se* influence momentum transfer. In this situation, momentum transfer from the fluid to unfixed particles on the solid surface becomes a function of the square of the mean fluid velocity. Mass transfer of these particles from the bed into the main fluid flow, although analogous to momentum transfer, correlates with a much smaller exponent of fluid velocity because much of the momentum transferred is used to promote the eddying of fluid in the wakes of particles rather than the movement of the particles themselves.

In this experiment, ΔSS^* was found to correlate with mean fluid velocity (L/τ) raised to the power of 0·8 according to the equation:

$$\Delta SS^* = -1{\cdot}01(L/\tau)^{0\cdot 8} + 3{\cdot}17 \qquad (9)$$

This is shown in Fig. 9. Figures 7 and 8 show values of Re and ΔSS^* for the 12 'linear' ponds. Note that all Re values exceed 10^6. The critical value for transition from laminar to turbulent regime in open channel flow is about 600 (Douglas *et al.*, 1979).

From eqn (9) it can be concluded that aquacultural settlement ponds should be designed so that the mean fluid velocity, L/τ, does not exceed 4·0 m min^{-1} and, if possible, should be less than 1 m min^{-1}.

The surface hydraulic loading parameter defined by Warrer-Hansen (1982) did not appear to significantly influence settlement pond performance relative to inlet suspended solids concentration. This can be explained if we consider the case of pond 10. Dividing its throughput flowrate by its surface area (data in Table 1) gives a surface loading of

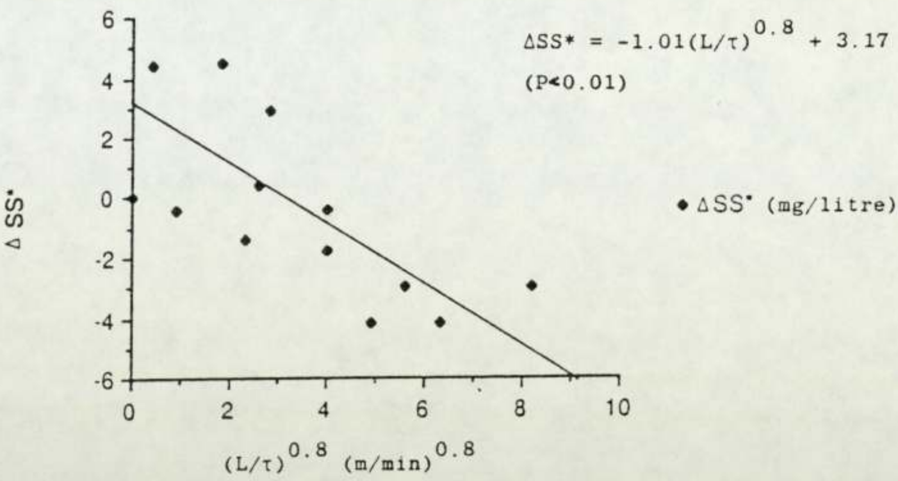

Fig. 9. Deviation of suspended solids removal ($SS_i - SS_o$) from that predicted using eqn (4) (Fig. 7) versus length–retention time ratio raised to the power of 0·8. The linear regression:

$\Delta SS^* = -1{\cdot}01(L/\tau)^{0{\cdot}8} + 3{\cdot}17$ is significant at $P < 0{\cdot}01$.

110 $m^3\ m^{-2}\ day^{-1}$ — the highest of all the ponds studied. Despite this, the pond removed more than 25% of its suspended solids load. Ignoring resuspension, this means that the largest 25% of the solid particles in the influent must have had terminal settling velocities in excess of 110 m day^{-1} or 1·3 mm s^{-1}. It has been found (Warrer-Hansen, 1982) that the mean settling velocity of faeces from 5 g trout fry is more than 15 mm s^{-1} and the corresponding figure for adult fish is well in excess of 50 mm s^{-1}. It is likely, therefore, that throughout the range of settlement ponds studied, surface loading was sufficiently low to allow a high proportion of the influent solids to settle, provided the degree of break up of faecal particles prior to settlement was not too great. This assertion is supported by the fact that the coefficient of SS_i in eqn (2) is close to unity. Surface hydraulic loading is likely to become a limiting factor at much higher values than were encountered in this study.

Length–width ratio and total breadth of inlet(s) relative to pond width did not significantly influence pond performance. As mentioned earlier, unfavourable values of these parameters would be expected to promote the formation of 'dead' areas. This would reduce the effective surface area of a pond, but this would appear to be nevertheless adequate for nearly 100% solids removal to take place in all 16 ponds were no resuspension possible.

Finally, materials of construction and surface crosswinds did not significantly influence the removal of suspended solids, although it would seem likely that the latter could become important at extreme values.

IMPLICATIONS FOR FISH FARMERS

For effective settlement, the inlet suspended solids concentration should not be much lower than 10 mg litre^{-1}. In cases where the dilute nature of an effluent prevents it from settling to the satisfaction of consent limits, the problem could be resolved by either reducing the water supply to the farm and negotiating with the relevant Water Authority for permission to discharge a more concentrated but less copious effluent, or diverting a proportion of the influent water supply to bypass the rearing tanks and using it to dilute the settled effluent just before discharge. The extent to which either of these solutions can be implemented will be limited by water quality criteria and the health of the fish in the rearing tanks.

The probability of attaining good performance for a given inlet solids concentration can be maximised by designing the pond so that the ratio of length-to-retention time is as low as possible.

The effluent treatment requirements of any fish farm will be greatly influenced by the severity of the discharge consent limits imposed by the relevant Water Authority. Where incremental consents have been applied to farms which receive their water supply from a river, the supply often contains a high enough concentration of solids for settlement in the rearing ponds to offset solids production sufficiently for the consent to be met without any settlement prior to discharge. Even relatively clean stretches of river can have suspended solids concentrations in excess of 10 mg litre^{-1} (STWA, 1984), but borehole supplies usually contain little or no suspended solids.

CONCLUSIONS

1. The effective settlement of suspended solids is limited at low inlet solids concentrations due to the resuspension of settled solids by turbulent scouring of the pond bed. It is difficult to achieve effective solids removal where inlet concentration is below 10 mg litre^{-1}, it is also difficult to attain effluent concentrations of less than 6 mg litre^{-1}.
2. For a given inlet solids concentration, settlement can be optimised by maintaining a low mean fluid velocity to minimise turbulent resuspension. This should be less than 4 m min^{-1}.
3. The surface hydraulic loading of all 16 fish farm settlement ponds studied is too low for its variation to have a significant influence on performance.
4. Length–width ratio, inlet specification, surface winds and materials of construction had no significant influence on settlement efficiency.

ACKNOWLEDGEMENTS

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REFERENCES

APHA (1975). *Standard Methods for the Examination of Water and Wastewater.* 14th edn, American Public Health Association, Washington DC, p. 1193.

Arceivala, S. J. (1983). Hydraulic modelling for waste stabilisation ponds. *Journal of Environmental Engineering, AICE*, **109** (5), 265–8.

Douglas, J. F., Gasiorek, J. M. & Swaffield, J. A. (1979). *Fluid Mechanics*. Pitman Publishing Ltd. London.

Foust, A. S., Wenzel, L. A., Clump, C. W., Maus, L. & Anderson, L. B. (1960). *Principles of Unit Operations*. John Wiley & Sons Inc, New York.

Harman, J. P. (1978). *Characterisation, treatment and utilisation of the effluent from an intensive fish farm*. PhD thesis, Aston University, 344 pp.

Liao, P. B. (1970). Pollution potential of salmonid fish hatcheries. *Water and Sewage Works*, **117** (12), 439–43.

Mangelson, K. A. & Watters, G. Z. (1972). Treatment efficiency of waste stabilisation ponds. *J Sanitary Engineering Divisions, Proc A.S.C.E.*, (April) 407–25.

Mudrak, V. A. (1981). Guidelines for economical commercial fish hatchery wastewater treatment. *Proc Bio-Engineering Symposium for Fish Culture*. Am. Fish Soc., Bethseda, Traverse City, MI, Oct 1979, 174–82.

Muir, J. F. (1982). Economic aspects of waste treatment in fish culture. *EIFAC Tech Paper*, **41**, 123–36.

Price, G. A. & Clements, M. S. (1974). Some lessons from model and full-scale tests in rectangular sedimentation tanks. *Water Pollution Control*, **73** (1), 102–13.

Solbe, J. F. (1982). The nature and effects of fish farm effluents. Water Research Centre. Paper submitted at IFM commercial trout farming symposium, Reading University, April 1982.

STWA (1984). *Water Quality 1983/84*. Severn-Trent Water Authority, Birmingham.

Toms, T. P., Owens, M., Hall, J. A. & Mindenhall, M. J. (1975). Observations on the performance of polishing lagoons at a large regional works. *Water Pollution Control*, **74** (4), 383–401.

Warrer-Hansen, I. (1982). Methods of treatment of waste water from trout farming. *EIFAC Tech Paper*, **41**, 113–22.