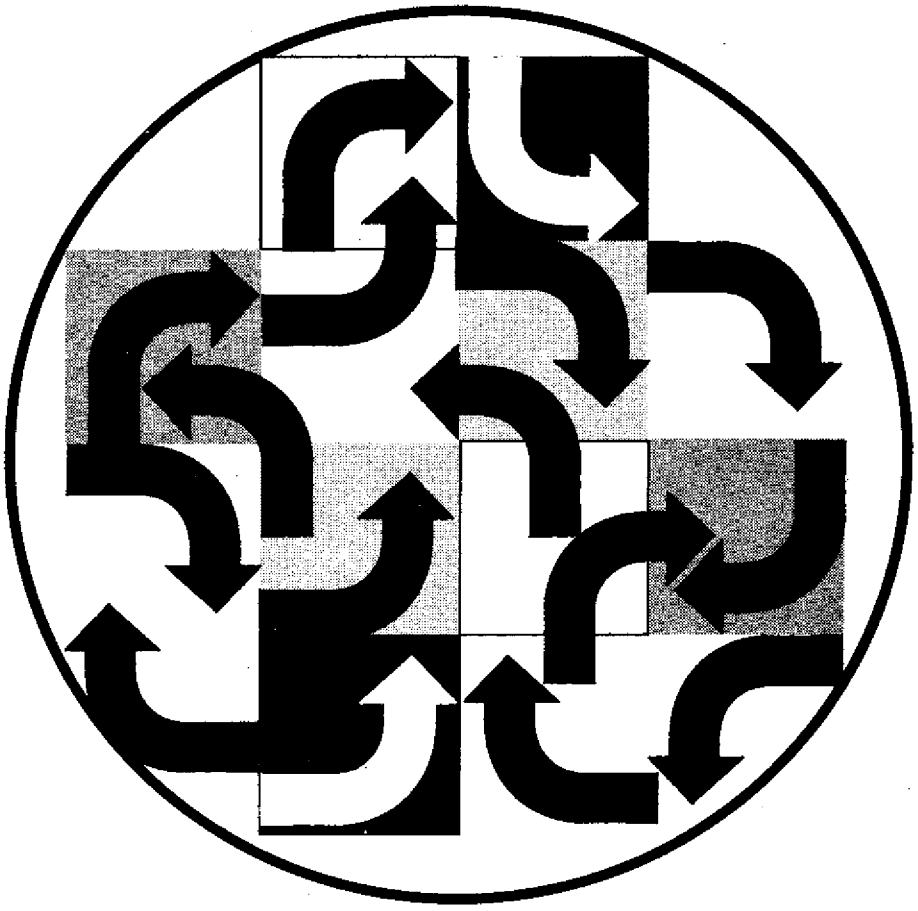
DESIGN OF HIGH-DENSITY RECIRCULATING AQUACULTURE SYSTEMS



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DESIGN OF HIGH-DENSITY RECIRCULATING AQUACULTURE SYSTEMS

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An Analysis of Biological, Economic, and Engineering Factors Affecting the Cost of Fish Production in Recirculating Aquaculture Systems

Thomas M. Losordo and Philip Westerman
North Carolina State University
Raleigh, North Carolina

Abstract

Aquaculture production in recirculating systems has been the focus of research and development efforts for decades. Although considerable resources have been expended on these systems in the private sector, there is a scarcity of data on the economic or engineering performance of commercial scale recirculating production systems. This paper presents the results of a computer simulation of tilapia production in a small recirculating production system. Much of the performance data has been developed at a demonstration facility at North Carolina State University. A model sensitivity analysis was performed to determined where changes in the production system could be made to realize the greatest production cost reductions.

Introduction

Recirculating aquaculture production systems have stirred a great deal of interest in the aquaculture community in the United States and worldwide. There is little doubt that most fish grown in ponds, floating net pens, or raceways can be reared in commercial scale recirculating systems, given enough resources. Unfortunately, the economic viability of growing commonly cultured species in recirculating systems is not as certain. The question of system economics has not always been adequately addressed prior to the development of an aquaculture business based on recirculating technology. Although there are numerous corporations and entrepreneurs selling "package turnkey" systems, there are relatively few reports of profitable commercial aquaculture recirculating production systems in operation (Losordo et al. 1989). (Mention of a specific product or tradename does not constitute an endorsement by North Carolina State University nor imply its approval to the exclusion of other suitable products.) Currently most commercial recirculating production systems are small (less than 45,000 kg / year), providing fresh high quality product at high prices to niche markets in the United States. Although there have recently been a number of large scale efforts in commercial system development, only one large scale system remains operating in the United States with long term production of wholesale quantities of fish.

Given the level of interest and activity in commercial recirculating production systems, there is a scarcity of data and systematic evaluations of the economic aspects of these systems in the literature. To reduce the number of future economic failures and aid in the design and development of successful production systems, non-biased and non-proprietary studies of the biological, economic and engineering aspects of recirculating systems must be completed.

As part of a study of the feasibility of recirculating aquaculture production systems, the authors developed a computer model to simulate the biological, engineering and economic performance of closed systems. The preliminary results were presented in Losordo et. al. (1989). The findings concluded that while catfish were not economical to grow in recirculating systems, striped bass hybrids showed promise. A sensitivity analysis of the simulation of the production cost of hybrid striped bass to changes in selected input costs was reported in Losordo (1991). In each simulation, only one selected input was adjusted by 10% and the resulting fish production cost was recorded. The selected input variables included the cost of bulk oxygen, cost of electricity, cost of feed, cost of labor, production mortality, feed conversion ratio, and system carrying capacity. Figure 1 displays the results of the seven simulations.

Interestingly enough, a 10% intensification of the production capacity of the system (without a corresponding increase in the fixed investment) produced the largest decline (5%) in production cost. These results suggested that future efforts in recirculating production system development should be in the intensification of systems without making them more expensive.

This paper will further investigate this conclusion and describe the results of simulations of the production of tilapia in a recirculating system. The computer model was upgraded to more realistically represent a functioning recirculating production system and where possible, the model utilized verified input data from a recirculating fishproduction demonstration system at North Carolina State University. The sensitivity of production costs to changes in the variable and fixed input costs and the biological and engineering performance of the system is reported.

Computer Simulation Methods

A general dynamic simulation model of recirculating aquaculture systems was developed, using a commercial software language called STELLA¬, as a

framework in which to determine the engineering, economic and biological performance of recirculating fish production systems. While the model will not be described in detail in this paper, a description of the input and output variables and general model structure follows.

STELLA Model Overview

The computer software package referred to as STELLA¬is a product of High Performance Systems, Inc. The computer language was developed for use only on the Apple Macintosh computer. For a description of the programming language, the reader is referred to "An Academic User's Guide to STELLA¬(B. Richmond, S. Peterson, and P. Vescuso, 1987, High Performance Systems Inc., Lyme, NH 03768).

The model was originally developed to simulate the growth and production of hybrid striped bass or channel catfish. The model was expanded for this study to include the production of tilapia. Many of the engineering simulation functions were also expanded and refined for this study. Fish growth rate algorithms were selected from the literature as listed in Table 1. To run the simulation, the user enters the appropriate data describing the production system as input variables listed in Table 1. The units of measure are described here in international system (SI) metric units; however, they are mixed between SI and English in the actual model to reflect those most commonly used by the aquaculture community for each variable. The model user can select the variables to be output by the computer in either table or graph format. Output variables are updated and listed after each simulation time step, and can include any calculated variable value within the model structure. These values, too numerous to list, include biological parameters such as individual fish weight, fish number, system fish biomass, oxygen consumed by fish respiration, and tank ammonia, nitrite, nitrate, and dissolved oxygen concentration. Engineering output variables can include data such as liquid oxygen consumption rate, new water addition rates (make-up water), recirculating power requirements, heating and cooling rates (BTU/hr), and the power usage rates of various system components. Economic variables that can be tracked can include total fish production cost and a production cost breakdown for each input variable such as feed cost or water cost per lb of fish produced. The output variables used for system analysis in this report are listed in Table 2.

Tilapia Production Simulation

The recirculating production system simulation model was configured to simulate the production of tilapia in a small (43,500 kg/year, 96,000 lbs/year) production system. The following are details of the production system being simulated and the simulation model assumptions.

Production System Description

The production system was modelled as a continuous production unit consisting of 3 juvenile "nursery" tanks and 8 growout tanks. Each nursery tank was capable of growing up to 6,560 - 83 g tilapia from 3 gram (50.8 cm, 2" long) animals in 90 days. With the harvest of one nursery tank each month, the 3 nursery tanks provide enough 83 g tilapia to restock 2 growout tanks per month. Each growout tank provides for the production of approximately 1,200 - 567 g (1.25 lb) tilapia in approximately 120 days. With 8 growout tanks, 2 harvests per month could be scheduled providing a production system with an overall yearly tilapia production yield of slightly over 43,500 kg (96,000 lbs).

Systems Design and Cost

The cost estimates in this section are derived from the development of a demonstration "Fish Barn" and recirculating production systems at North Carolina State University. The components described here are, for the most part, of the same scale and similar to those currently in use at the North Carolina Fish Barn project.

The simulated production system is housed within a 28 m x 11 m (92' x 36') metal clad, wooden post and beam barn with minimal insulation and an earthen floor. Each tank is configured with an individual water renovation system consisting of an in-line resistance type electric water heater, a 0.57 m (20 ft^3) floating bead filter as described by Losordo (1991) and a 1/4 hp rotating biological contactor (total surface area = 418 m^2) used in series (Figure 2). Dissolved oxygen additions to the culture tanks were provided by either in tank diffused aeration or whole recycle stream oxygenation with pureoxygen (depending on the simulation input selections). Total cost of each 4.25 m diameter, 1.45 m deep fiberglass tank growout system, with all associated support equipment installed, was modelled to be $16,300 (Table 3).

The nursery tank systems were modelled as 3 meter diameter, 1.25 m deep fiberglass tanks. Water and wastewater treatment for all three nursery tanks is provided by one bead filter and one RBC as described above. Total cost of the nursery production system is $18,490 (Table 4).

A depreciation rate for each system was estimated by the "straight line" method. The yearly rate of depreciation was estimated by calculating the dollar value per year of depreciation for each component (cost installed / component life), then summing the yearly depreciated value for each component and dividing by the total system cost (Yearly Depreciation Rate = [Σ (cost installed / component life) / total system cost]).

Building cost, including water and electrical service, was modelled at $129.10 per square meter ($12 / ft^2)

or $39,744. The total production system cost (excluding the building cost) was modelled at an additional $148,890 or approximately $3.42 of fixed investment per kg of annual production capacity ($1.55 / lb). The overall proposed system layout can be viewed in Figure 3.

Base Case Production Simulation Results

For this study, two base case tilapia production growth cycles were simulated with the model. One was set up to simulate production with atmospheric aeration utilizing low-pressure regenerative type blowers and "air-stone" diffusers. The second was set up to simulate the production utilizing pure gaseous oxygen and whole recycle stream injection. The reader should note that the production of tilapia at the simulated densities in recirculating systems with low pressure air aeration has not been attempted by the authors. The feasibility of in-tank aeration at the required rates is not certain. The agitation developed within the tank by such systems may be not be appropriate for proper fish growth. The simulation assumptions for both base cases are listed in Table 5.

The production cost results of the base case (with atmospheric aeration) can be viewed in Figure 4. The results of the simulation indicate thatunder the given conditions overall fish production cost will be approximately $2.79 / kg ($1.27 / lb). Given the same set of assumptions except with pure oxygen as the oxygen source, the cost of production was estimated to be $2.86/ kg ($1.30 / lb). Subsequent modelling runs determined that production costs with liquid oxygen would equal that for atmospheric aeration with a bulk oxygen cost of $ 0.18 /m^3 ($ 0.51 / 100 ft^2).

The simulation also provides us with some basic engineering information on which to base a rational systems design. Selected information included a peak recirculation rate of 568 lpm (150 gpm) for each growout tank, a peak flow rate of new water of 4.8 lpm per tank at peak feed rates of 27.5 kg per tank per day (61 lbs).

While the simulated production cost of fish in this system as described is probably too high to be economically viable in a wholesale market environment, there are areas where improvements could be made. The sensitivity analysis of the modelled system that follows provides invaluable information as to which avenues of improvement to pursue with future research and development activities.

System Sensitivity Analysis

A total of 12 model input variables was selected for study in the sensitivity analysis of the simulated system. The 12 variables are grouped and reported as biological variables, variable operating cost inputs, engineering performance variables, and system or fixed cost variables. Twelve modelling "runs" were executed; each time only one of the variables was changed by 10%. In most cases the variables were changed to provide a simulated improvement in the system performance. Some variable levels of the base case were so good (based on North Carolina Fish Barn performance experience) that a 10% change for the better was judged as not being realistic. In these cases the variables were changed for the worse. In all cases, changes in fish production costs were monitored. The overall results of the sensitivity analysis can be viewed in Figure 5. Changes in both total production cost and significant individual analysis of the production changes are listed and discussed below.

Biological Variables

The model input variables that directly affect the biological performance of the cultured product are feed conversion ratio (FCR) and fish mortality rate. When the FCR was reduced from 1.3 to 1.17 (10% change) the overall cost of producing tilapia changed by $0.088 / kg ($0.04 / lb.) or 3.2%. Analysis of the productioncost breakdown yielded interesting results. Although we would expect the change to cause a reduction in total feed consumed, the simulation results also indicated that changes in new water use, heating requirements, recirculation rate, aeration rate and electric demand charges also occurred. While the water recirculation rate, new water addition rate and aeration rate were directly affected by FCR, the change in heating requirement was indirectly caused by the reduced water exchange rate. The electric demand charge decreased as a result of decreased water pumping and heating rates.

The mortality rate was increased for this sensitivity analysis by 10%. The change in mortality rate yielded only a 2.3% change, or a $0.066 / kg ($0.03 / lb) difference in overall production cost. The detailed simulation results showed that although some changes occurred in almost all production cost categories, the major changes occurred in feed cost and fingerling cost per kg of production. The reader should note that mortality was simulated as a fractional constant loss over the entire growth cycle. In fact, losses later in the growth cycle would have a greater effect on production costs than losses occurring earlier in the cycle.

Variable Costs

The modelled variable costs changed (decreased 10%) in this sensitivity analysis were the costs of feed, electricity, liquid oxygen, and labor. As in the previous study (Losordo 1991), the greatest change in fish production cost was effected by a change in feed cost. This may not be totally unexpected as the reader should note that feed costs are the largest single production cost in the base case simulation (see Figure 4). The sensitivity analysis indicates that changes in the electricity rate, labor rate of pay, and oxygen cost rate caused changes in production costs of 1.5%, 1.4% and 0.1% respectively.

These results, in concert with the FCR sensitivity results, suggest that significant production cost savings could be realized if a feed that produced a better FCR were available at a reduced cost. Unfortunately feed manufacturers market improved feed conversion as a reason to pay more for these specialized feeds. In this case the feed conversion gains may be offset by increases in feed-related production costs.

It is also interesting to note that while recirculating systems have been assumed to beenergy intensive, a 10% change in the electric cost rate produced only a 1.5% change in the total production cost.

Engineering Performance Variables

The modelled engineering performance variables that were investigated in this study were recirculating pump efficiency, biological filter efficiency, oxygen injection system transfer efficiency, and aeration system oxygen transfer efficiency. Surprisingly, 10% changes in these variables produced no more than a 0.5% change in total production cost. It may be hard to justify expending a great deal of R & D time in improving the performance efficiency of each of these components given these results. The reader should not misunderstand this statement however. The authors are suggesting only that biological filters that "remove" 100% of the ammonia nitrogen per pass, or pumps that run at 80% efficiency may not be the key to economic viability. On the other hand a great deal of development effort is needed in improving the reliability of biological filters. Indeed, the unexpected failure of a biological filter or undetected failure of an oxygen injection system can do major damage in a short period of time to the economic viability of a commercial fish production system.

System or Fixed Cost Variables

The results of this sensitivity analysis support the findings described in Losordo (1991). Changes in either the system capacity or overall system cost had greater impacts on the total production cost that changes in any other variable. A 10% increase in system capacity was simulated without an associated change in investment. That is to say that the physical production system was not altered although the stocking rate was increased by 10%. This change caused a 4.1% decrease in the total production cost. It is interesting to note that no savings are associated in this case with variable production costs. To increase capacity we must use a proportionally higher amount of feed, electricity, pumping, aeration and operating capital. The savings come in the form of more production for the same loan, equity and labor costs. Savings in these categories as compared to the base case analysis amount to 1.4, 1.0, and 1.5% respectively.

Similarly, a 10% reduction in the overall investment for the same production capacity yielded a 3.5% change in production cost. Insight into this finding may be gained by reviewing the results from the base case production analysis in Figure 2. Over 41% of the costs associated with the production of the fish are directly related to the initial investment cost of the system. In changing the cost of the system, the calculated costsof maintenance, depreciation, loan payments and cost of equity are all directly effected.

Discussion

The simulated production results outlined above point out areas in production technology where improvements can affect the financial viability of recirculating production systems. The reader should note that the changes in the model input variables were accomplished without concurrent changes in the costs associated with making these changes within the system. In most cases improvements in any variable will have an associated cost incurred in making the change. For example, feeds that provide a higher FCR will probably cost more. Similarly pumps that are more efficient will also cost more.

The results of this study point to one area that may provide substantial production cost savings. The authors believe that these cost savings can result from reducing the overall cost of recirculating production systems while maintaining component reliability and longevity. These results can be viewed from another perspective. While we tend to think of component design criteria in performance only, the results of the performance efficiency sensitivity analysis suggest that this alone is not of primary importance. The results of this study suggest that perhaps we should combine component performance with the cost of ownership of these components (which ultimately make up the total cost of the system) into an "performance / cost" factor. For example, a biological filter should be evaluated according to the total grams of ammonia nitrogen removed per day per cost of annual ownership ((g TAN / year) / ($ / year) = g TAN / $). The costs associated with ownership on an annual basis would include annual loan or equity cost for the purchase of the component combined with depreciation and maintenance costs. That is to say, given a basic reliability factor, we should seek to maximize the calculated value of this variable. For example, the rotating biological filter at the North Carolina Fish Barn has had an average TAN removal capacity of 145 g/ day over a recent 160 day test period. Given a $3000 purchase price, an estimated component life of 7.5 years, a maintenance cost rate of 5%, interest rate of 13%, equity rate of 9.5%, and an equity-interest ratio of 1:1, the performance / cost factor can be estimated to be 59 g TAN / $. The floating bead filter utilized at the North Carolina Fish Barn removed an average of 53 g TAN / day over the same test period. With a purchase price of $3200, an estimated component life of 10 years andsimilar economic cost rate assumptions, the performance cost factor would be calculated to be 23 g TAN / $. Additionally a fluidized bed sand filter was evaluated over the same period. The performance

results indicated that an average of 50 g TAN / day were processed. Given a $1000 price, a 10 year life and similar economic cost rate assumptions, the performance / cost factor was calculated to be 69 g TAN / $. If we assume that these components have similar performance reliability, then the fluidized bed filter would be a better value based upon this evaluation criterion.

Summary and Conclusions

The challenges for recirculating systems designers and engineers are many. We must seek to design recirculating systems and system components that can be manufactured at a lower cost or seek to utilize components that are currently available to other industries at lower prices. Where possible we must increase the carrying capacity of a system without increasing cost or sacrificing system reliability.

The following specific conclusions may be reached as a result of the data presented in this study.

1. Given the assumptions of the base case simulation, the cost of producing a kilogram of tilapia in the small recirculating system described is estimated to be $2.79 ($1.27 / lb).

2. Changes in the engineering performance efficiency on the system components did not greatly change the total production cost.

3. Changes in the feed cost had the greatest impact of production cost of all of the operational variable costs investigated.

4. Improvements in the feed conversion ratio can have a significant impact on production costs. However, these savings can be offset by associated increases in feed prices.

5. The greatest gains to be realized in improving profitability are those associated with increased production capacity or decreased system cost.

Acknowledgements

The research and demonstration program in recirculating aquaculture technology at North Carolina State University is supported by the College of Agriculture and Life Sciences of North Carolina State University andthe Energy Division of the North Carolina Department of Economic and Community Development.

References

Boyd, C.E. 1979. Water Quality ir Warmwater Fish Ponds, Auburn University Agricultural Experiment Station, Auburn, Alabama, 359 pp.

Brown, J.W. 1989. An Analysis of the Economic Potential for Hybrid Striped Bass Culture, Department of Economics and Business, North Carolina State University, Ph.D. Dissertation, 228 pp.

Losordo, T.M., J.E. Easley, and P.W. Westerman. 1989. Preliminary Results of a Survey on the Feasibility of Recirculating Aquaculture Production Systems, Presented paper at the American Society of Agricultural Engineers Winter Meeting, December 12-15, 1989, New Orleans, Louisiana, USA.

Losordo, T.M. 1991. Engineering Considerations in Closed Recirculating Systems, pp. 58 - 69 In Aquaculture Systems Engineering, Proceedings of the World Aquaculture Society and the American Society of Agricultural Engineers Jointly Sponsored Session, WAS 22nd Annual Meeting, June, American Society of Agricultural Engineers, St. Joseph, MI, 138 pp.

Richmond B., S. Peterson and P. Vescuso. 1987. An Academic User's Guide to STELLA¬, High Performance Systems Inc., Lyme, N.H., 392 pp.

Soderberg, R.W. 1990. Temperature Effects on the Growth of Blue Tilapia Intensive Aquaculture, The Progressive Fish-Culturist, Vol. 52, pp. 155-157.

Table 1. Model Input Variables

Species Cultured Hybrid Striped Bass (Brown, 1989)
Catfish (Boyd, 1978)
Tilapia (Soderberg, 1990)

Variable	Selection and/or Units
Mode of Operation	continuous or batch
System Capacity	kg
Cost of Fingerlings	$ each
Fingerling Length	mm
Fingerling Weight	g each
Fish Harvest Weight	kg each
Fish Survival	
nursery	decimal fraction
growout	decimal fraction
Production Cycle Length	
nursery	days
growout	days
Feed Cost	
Frye feed	$ / kg
Juvenile Feed	$ / kg
Grower Feed	$ / kg
Finishing Feed	$ / kg
Feed Protein Content	
Frye feed	%
Juvenile Feed	%
Grower Feed	%
Finishing Feed	%
Expected Feed Conversion Ratio	dimensionless
Electricity Costs	
< 2750 kwh/ month	$ / kwh
< 4450 kwh/month	$ / kwh
> 4450 kwh/month	$ / kwh
Base Electric Rate	$ / month
Hours of Labor	hours / day
Rate of Pay for Labor	$ / hour
Oxygen Source	
Base Case 1	aeration
Base Case 2	liquid oxygen
Bulk Storage Tank Size	liters
Bulk Tank Rental	$ / month
Cost of Bulk Oxygen	$ / m^3
Oxygen Transfer Efficiency	decimal fraction
Biological Filter Efficiency	decimal fraction
Recirculation Pump Capacity	lpm each pump
Recirculating Pump Efficiency	decimal fraction
Desired Oxygen Concentration	ppm
Desired Ammonia Concentration	ppm
Desired Nitrate Concentration	ppm
Desired Water Temperature	C
Initial Loan Amount	$
Interest Rate on Loan	decimal fraction
Equity Investment	$
Current Bank CD Market Rates	decimal fraction
Short Term Loan Interest Rates	decimal fraction
Maintenance Cost Rate	%
Depreciation Cost Rate	
Building	% per year of initial cost
Nursery	% per year of initial cost

Table 2. Selected Model Output Variables

Variable	Selection and/or Units
Total Production Cost	$ / kg of fish produced
Feed Cost	$ / kg of fish produced
Labor Cost	$ / kg of fish produced
Fingerling Cost	$ / kg of fish produced
Heating Costs	$ / kg of fish produced
Water Recirculation	
Power Costs	$ / kg of fish produced
Water Costs	$ / kg of fish produced
Oxygen Cost	$ / kg of fish produced
Cost of Borrowed Capital	$ / kg of fish produced
Cost of Operating Capital	$ / kg of fish produced
Opportunity Cost of	
Equity Capital	$ / kg of fish produced
Maintenance Cost	$ / kg of fish produced
Depreciation Cost	$ / kg of fish produced
Fish Weight	g / fish
System Biomass	kilograms
Recirculating Flow Rate	lpm
Make-up Water Flow Rate	lpm
Feed Rate	kg / day

Table 3. Individual Growout System Cost and Depreciation Estimates

System Component	Cost Installed	Component Life
Fiberglass Production Tank	3,800	15 years
Floating Bead Filter	4,000	10 years
Rotating Biological Contactor	4,000	7.5 years
Aeration or Oxygen		
Delivery System	1,500	15 years
Pipes and Valves	400	10 years
Pumps (3 each @ $120)	360	3 years
Solenoid Valves & Timers	1,000	5 years
Heater and Controller	240	5 years
Feed Delivery System	1,000	5 years
Total Cost	$16,300	

Table 4. Nursery System Cost and Depreciation Estimates

System Component	Cost Installed	Component Life
Fiberglass Tanks (3 @ 2,000)	6,000	15 years
Floating Bead Filter	4,000	10 years
Rotating Biological Contactor	4,000	7.5 years
Aeration or Oxygen		
Delivery System	1,500	15 years
Pipes and Valves	400	5 years
Pumps (3 each @ $120)	360	3 years
Solenoid Valves & Timers	1,000	5 years
Heater and Controls	230	5 years
Feed Delivery System	1,000	5 years
Total Cost	$18,490	

Table 5. Base Case Model Input Variables

Variable	Selection and/or Units
Species Cultured Tilapia	
Mode of Operation	Continuous
System Capacity	1,814 kg (per tank)
Cost of Fingerlings	0.15 ($ each)
Fingerling Length	50.8 (mm)
Fingerling Weight	3.0 (g each)
Fish Harvest Weight	0.567 (kg each)
Fish Survival	
nursery	0.95 decimal fraction
growout	0.975 decimal fraction
Production Cycle Length	
nursery	90 (days)
growout	120 (days)
Feed Cost	
Frye feed	0.93 ($ / kg)
Juvenile Feed	0.79 ($ / kg)
Grower Feed	0.51 ($ / kg)
Finishing Feed	0.35 ($ / kg)
Feed Protein Content	
Frye feed	50 %
Juvenile Feed	43 %
Grower Feed	36 %
Finishing Feed	32 %
Expected Feed Conversion Ratio	1.3 (dimensionless)
Electricity Costs	
< 2750 kwh/ month	0.0705 ($ / kwh)
< 4450 kwh/month	0.0459 ($ / kwh)
> 4450 kwh/month	0.04102 ($ / kwh)
Base Electric Rate	10 ($ / month)

Variable	Selection and/or Units
Hours of Labor	(hours / day)
Rate of Pay for Labor	6.00 ($ / hour)
Oxygen Source	
Base Case 1	aeration
Base Case 2	liquid oxygen
Bulk Storage Tank Size	3,400 (liters)
Bulk Tank Rental	425 ($ / month)
Cost of Bulk Oxygen	0.247 ($ / m^3)
Oxygen Transfer Efficiency	0.75 decimal fraction
Biological Filter Efficiency	0.50 decimal fraction
Recirculation Pump Capacity	190 (lpm each pump)
Recirculating Pump Efficiency	0.335 decimal fraction
Desired Oxygen Concentration	6.0 (ppm)
Desired Ammonia Concentration	1.0 (ppm)
Desired Nitrate Concentration	400 (ppm)
Desired Water Temperature	28° C
Initial Loan Amount	93,317 ($)
Interest Rate on Loan	0.12 decimal fraction
Equity Investment	93,317
Current Bank CD Market Rates	0.095 decimal fraction
Short Term Loan Interest Rates	0.13 decimal fraction
Maintenance Cost Rate	5.0 (%)
Depreciation Cost Rate	
Building	5.0 % per year of initial cost
Nursery	10.49 % per year of initial cost
Growout System	11.01 % per year of initial cost

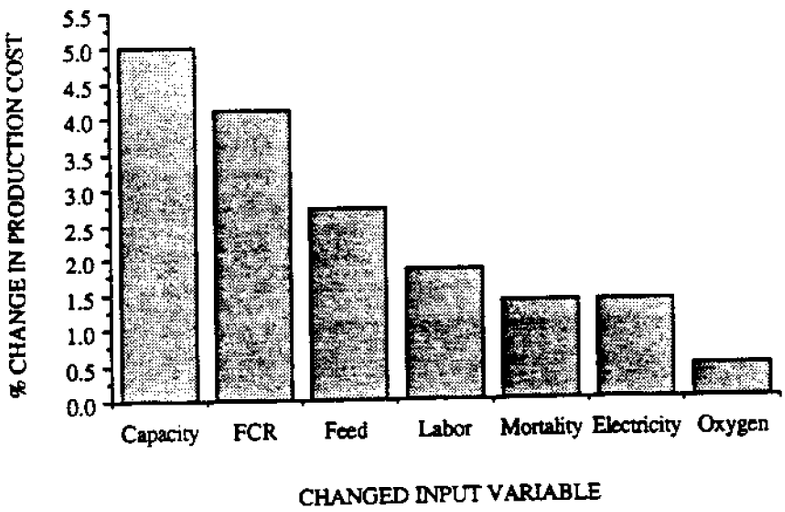

Figure 1. Simulated Change in Production Cost of Hybrid Striped Bass in a Recirculating System Due to 10% Change in Selected Input Variables.

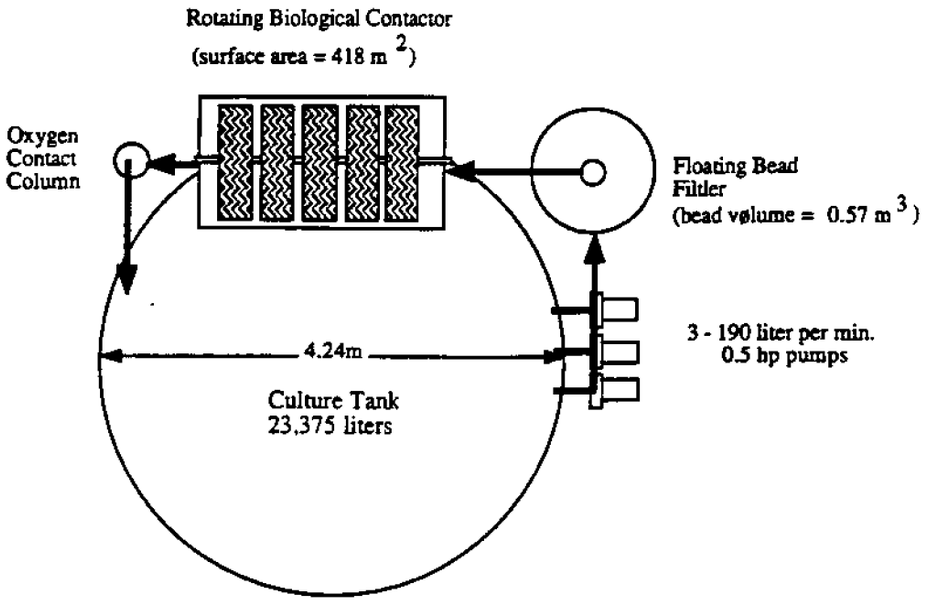

Figure 2. Simulated Typical Growout System Component Layout.

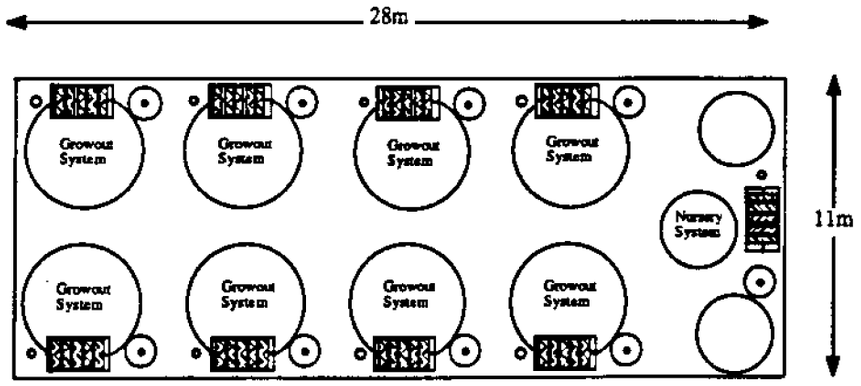

Figure 3. Simulated Recirculating Fish Production System Layout 8 Growout Tanks and 3 Nursery Tanks.

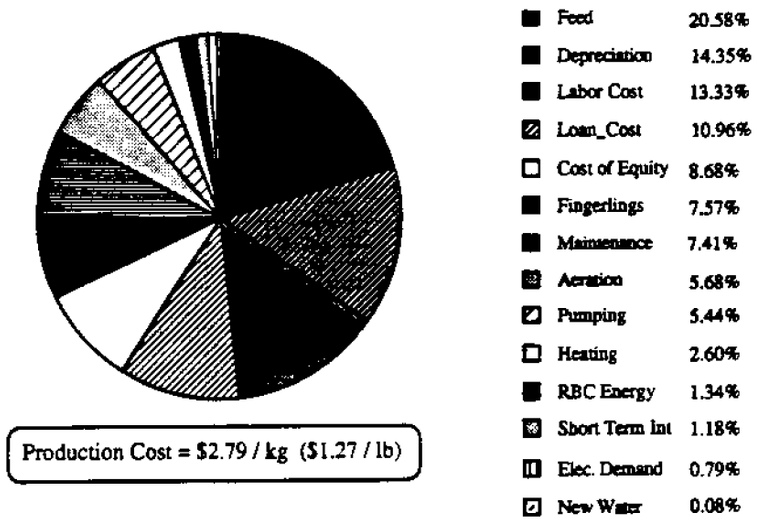

Figure 4. Simulated Base Case Production Cost of Tilapia in a Recirculating System.

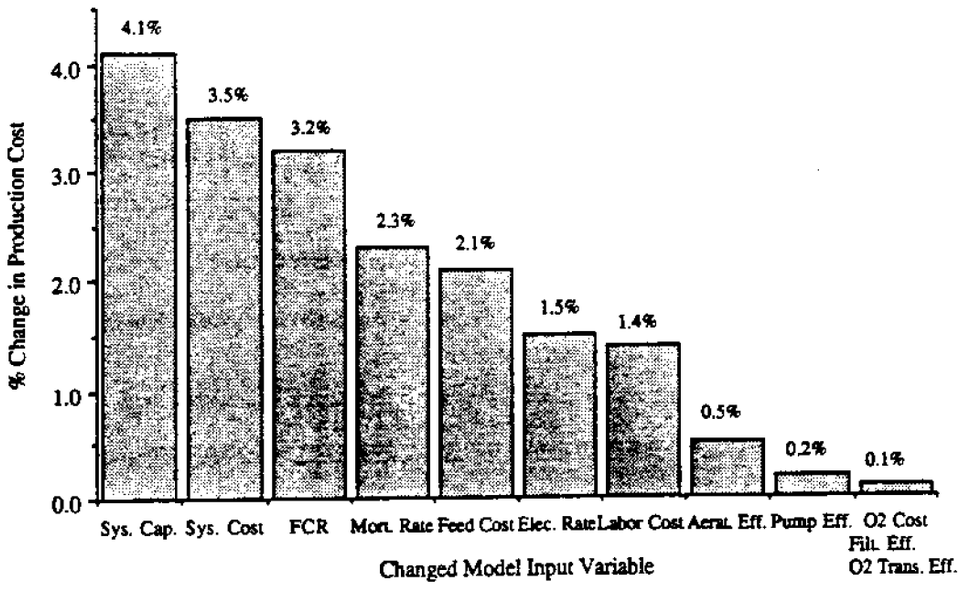

Figure 5. Results of Simulation Sensitivity Analysis: % Change in Production Cost Due to a 10% Change in a Selected Input.

Optimizing Production by Continuous Loading of Recirculating Systems

Steven Van Gorder
Fresh-Culture Systems, Inc.
Kutztown, Pennsylvania

Introduction

The advantages recognized as characteristic of recirculating aquaculture systems have been well documented for many years (Liao and Mayo 1974; Losordo 1991). The significant water resource requirements and appropriate climatic conditions required of traditional fish culture systems are virtually eliminated. A properly designed recirculating aquaculture system can be placed almost anywhere, and produce a quality-controlled product continuously throughout the year. However, with the actual implementation of various systems, the strict pre-requisites inherent in hardware and technology development, as well as the processing and marketing demands required to achieve economic viability, are becoming more clearly defined (Losordo et al. 1989).

Also, with the escalation of debate concerning recirculating technologies, it has become necessary to more carefully define the terminology that is used to describe the processes involved. "Recirculating" aquaculture can describe the reuse of water in semi-closed systems, such as raceways or flow-through tank systems, in which no water quality control technologies are employed except water exchange. A number describing percent recirculation in these cases can simply denote the amount of water that is reused on a single pass via pumping, thereby increasing the volume of water available, but still resulting in several new tank or raceway volumes of water each day. A value of 90% recirculation provided for a 1000 gpm flow-through facility would still describe the use of 100 gpm of new water, or 144,000 gallons/day.

When describing percent recirculation in a "closed-system", the number used, (by definition remaining above 90%), will describe the average percentage of the total water volume within the system which is used on a daily basis. In a 50,000 gallon facility, a 90% closed-system would employ 10% make-up water or only 5000 gallons of new water daily. In these systems, increasing the quality of the hardware and technologies used to maximize the recirculation efficiency to levels nearing 100% isoften required, not to minimize water usage, but to provide for solid waste management and to reduce the amount of heat lost in the effluents (Kugelman and Van Gorder 1991).

In order for any form of aquaculture to be economically viable, it is necessary to integrate the aquatic resources or technologies with the appropriate marketing strategies. For recirculating systems, these strategies must take advantage of the specific benefits afforded to the product. These include the availability of the fish as an absolutely fresh product (alive or fresh dead), certified as unpolluted, and available continuously throughout the year. These are "value added" aspects which, considering the higher expenses involved with recirculating techniques, must result in a more highly priced product in order to provide an economically viable alternative to traditional aquaculture. These advantages must be offered as incentives to various niche markets willing to pay a higher price (Buck 1991).

State-of-the-Art

Various methods have been employed by the industry to provide for water quality control with closed-systems at several scales of production. Because of the complexities involved, there isn't just one particular cost-effective system to be developed. Every system will differ in many ways from others, in the design of the hardware, and the use of its technologies. In evaluating any recirculating aquaculture system, the production capacity will be determined by the limitations of its weakest component. With recirculating systems there are dozens of design requirements, all of which must be carefully integrated to operate efficiently. And efficiency is the final word in the successful operation of a recirculating system.

While the scale of production is extremely important in determining what components are appropriate to a particular design, commercial systems of any size cannot use many of the off-the-shelf items that have, in the past, often been characterized as effective. The use of swimming pool sand filters for solids removal and/or biofiltration, or sewage treatment and aquarium industry hardware, is seldom appropriate for fish culture. In fact, it is necessary to integrate components designed specifically for each water quality control requirement, with the co-developed management techniques. The development of these systems is expensive, and is therefore often carried out on a pilot scale which is too limited to make objective projections. The result has been the design of many "paper systems", which have usually failed when implemented.

System Components

Optimizing production by continuous loading involves the maintenance of high levels of feeding at all times. The control of water quality continuously under these heavy loading conditions requires the use of appropriately designed hardware. And the careful integration of this hardware with co-developed management technologies is also necessary.

To date, the design of equipment to accomplish this, and the evaluation of such equipment, has usually taken place independently for each unit process (biofiltration, clarification, aeration, pH control, etc.). However, to be effective, the design of the hardware and management technologies which control each unit process must be integrated with those controlling all other unit processes. The following describes the general design characteristics of the individual components associated with the maintenance of water quality and aquacultural productivity within a recirculating system.

Tanks

The culture tanks are perhaps the first and most obvious component considered for recirculating systems. The tanks must be self-cleaning (therefore usually circular or oval), with angled floors to effluent ports, using dynamic flow characteristics for efficient removal of the culture water and waste products to the filter components. The volume of water in the tanks must correspond to the rate of flow through the filter components (for suitable retention time and number of daily passes). Other considerations are the anticipated density of the fish, smoothness of tank walls, ease of cleaning, access to the fish, and finally cost.

Biofilters

Usually considered the heart of the system, this component must remove the ammonia produced by the fish. There must be adequate surface area for the growth of nitrifying bacteria, and the filter must not clog with fish wastes or the sloughing bacterial growth. The filter media must also have the necessary characteristics to be continuously self-cleaning.

The effectiveness of the biofilter will also depend on the amount of oxygen available to the bacteria, the flow rate of the water through the filter, the concentration of ammonia coming in contact with the media over time, the mechanical reliability, the energy requirements of the filter, and the initial cost (Wheaton et al. 1990).

Clarifiers

Removal of particulates (mainly fish feces) from the water on a continuous basis is one of the most problematic of all system design requisites (Chen and Malone 1991). The settleable solids must be removed from the tank, removed from the flow of water and concentrated, and removed from the clarifier itself on a frequent basis, to preclude the break-down of these wastes. The clarifier must be integrated with the tank, biofilter and pumping components to provide the appropriate retention times required. At least a daily removal of the concentrated solids must be accomplished.

Suspended solids must be removed to reduce biological oxygen demand (BOD), maintain clarity, and provide an optimal environment for the fish. Several species of fish will demonstrate reduced respiratory efficiency, and suffer potential gill damage and secondary disease problems from excessive suspended solids levels. These solids can be removed by foam fractionation (Chen et al. 1989) or screen filtration (Makinen et al. 1988). The potential for fine screen filtration depends on the ability to preclude clogging, and the energy efficiency of the process.

Dissolved solids result in the coloration of the water, reducing visibility and possibly productivity. These solids include a BOD component, as well as an accumulation of various metabolites, such as hormones that may reduce growth rates. The only effective way to remove these is by using ozone for oxidation of this component (Wheaton 1977).

Aeration/Oxygenation

Mechanical aeration, sparging of air bubbles, and the use of air lift designs for pumping can be incorporated in a variety of appropriate designs. Consideration must be given to accomplishing the desired aeration, and secondly to the energy efficiency. All aeration considerations must relate directly to the anticipated feeding levels, and to the temperature of the water. In super-intensive recirculating aquaculture systems, which are necessary for most commercial levels of production, pure-oxygen injection systems are required to maintain adequate and stable dissolved oxygen levels (Colt and Watten 1988).

The use of pure oxygen allows for the maintenance of saturated or even super-saturated levels of DO in the water. Several methods can be used to dissolve the oxygen into the water, including U-tubes (Watten and Beck 1985), high pressure spargers with micro-bubbles (deep water systems) (Severson et al. 1987), or inverted cone oxygen-injection chambers (Speece et al. 1971). The design specifications require that the necessary volume of oxygen super-saturated water is efficiently provided to the culture tank to maintain the prescribed dissolved oxygen levels upon dilution. The efficiency is related to the percentage of pure oxygen which is dissolved without being lost to the atmosphere, and to the cost of the oxygen and electricity involved in the process. The system design must relate to the tank size, shape and volume, the temperature of the water, the levels of feeding, the cost of the equipment and the availability and cost of oxygen.

Heating/Insulation

The methods used to heat water (relating most importantly to the % recirculation previously discussed) and to conserve that heat, are vital to an energetic and economic analysis of a recirculating culture system. One choice is to heat the room air to several degrees above the desired water temperatures (taking into account evaporative cooling) and provide adequate room insulation to retain the heat economically. The second choice is to heat the water and insulate the

culture tanks, as well as the room. In this case, the tanks should be covered, as well. This results in a more comfortable working environment, with room temperatures maintained below water temperatures.

Pumping Systems

Water must be moved through system components at prescribed rates. The main concern will be electrical costs, the efficiency of pumping, the height required to pump the water, and the elimination of potential clogging problems. Air lift pumps should be used when possible to minimize cost and mechanical problems (Turk and Lee 1991).

Feeding Systems

Recirculating systems require the efficient management of feed inputs to maintain a stable andcontinuous feeding regimen throughout a carefully controlled diel light cycle. By dispersing the computed feed levels over the entire lighted period, the negative effects of feed inputs on water quality, and subsequently the demands on all of the system design components, are mitigated. This requires the use of automatic and/or demand feeding systems. Light cycles should also be maintained at optimum levels to maximize feeding periods.

Reservoir Systems

In situations where water must be treated for removal of chlorine, or to elevate incoming water temperatures for direct administration to stocked tanks, it is necessary to employ reservoir tanks. Also, quantities of water may be required for emergency use.

Holding Systems

The value of the fish produced in closed systems is related to their freshness when delivered to market. It is often necessary to keep the fish alive while awaiting transport to preserve this freshness, or to remove any off-flavors.

Emergency Electrical Systems

There must be a provision for absolutely trustworthy back-up electrical power that will sustain all water quality parameters during power outages. There should also be automatic systems for contacting key personnel in case of emergencies.

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There are many additional state-of-the-art aspects of a recirculating aquaculture system that must be specifically designed to complement all those previously discussed. These include methods or systems for pH control, carbon dioxide removal, laboratory and field analysis of water quality, nursery and/or spawning systems for continuous fingerling availability, waste management and disposal, methodologies for year-round harvests, transport capabilities, and processing and marketing strategies.

Fundamental System Requirements

There are several aspects of design and implementation that are considered of such importance that they are obligatory to the successful operation of closed/recirculating systems:

1. Levels of water recirculation must be maintained well above 90%. (Unless there is an availability of significant volumes of heated water.)

2. Efficient and continuous removal of solid wastes necessary, using clarification methods for settleable and suspended solids.

3. Ammonia levels must be controlled through efficient biofiltration.

4. Dissolved oxygen levels must be controlled through efficient aeration/oxygenation systems.

5. Systems must be maintained at threshold levels of biomass loading with sustained optimal feeding inputs, and all system water quality control components must be continuously operated near threshold design limits.

6. A combination of technologies must provide for a continuous harvest capability.

It is the last two of these imperatives that will be discussed below.

Maintaining Continuous Loading

In commercial-scale recirculating aquaculture systems, the densities involved are often much higher than with most traditional methods of aquaculture, with productivity measured in hundreds of thousands of pounds per acre. However, initial and operational expenses are necessarily much higher as well. Establishing and maintaining the levels of intensive aquaculture required for these systems is more costly. Each of the design components and technologies described above are costly to develop, build and implement. The operational expenses are higher, environmentally controlled space is expensive, and the technologies require the involvement of specially trained personnel. It is therefore necessary to continually utilize all system design elements optimally.

As an example, in most pond systems, the same number of fish are stocked that will ultimately be harvested. The fish are grown out through a single warm-weather season, and increasing levels of feed are provided to the pond as the fish biomass increases. The potential loading capacity of the pond is not reached until the very end of the season.

This situation is not acceptable in recirculating systems. Considering the costs for the equipment and space, and the increased operational expenses involved with recirculating aquaculture systems, it is necessary to always maintain fish biomass at near capacity, and therefore provide continuous feeding at some threshold level. If the same number of fish are stocked in a tank system as fingerlings, as are ultimately to be harvested, they will initially receive a small fraction of the feed that will be required when they reach a harvestable size. As with pond systems, it will be several months before the system will be loaded at the rate for which it was designed. If the water quality control systems are not being utilized at or near capacity for the majority of the growth cycle, the operational expenses will be excessive, and the system will not be economically viable.

This situation can be dealt with in several ways:

- Culture tanks utilizing individual or centralized water quality control systems can be designed to maintain several size classes of fish, with the stocking of additional fingerlings attending the harvest of fully grown fish. Biomass and feeding levels would therefore not be reduced below threshold levels following harvests.

- Many culture tanks, each with varying size classes of fish, can utilize centralized water quality control systems designed to maintain water quality within all tanks. The additive feeding levels, and the subsequent demands on the control systems of the total population of fish, are thereby not excessively reduced by the total or partial harvest of any single tank.

- Multiple tank complements with individual or centralized water quality control systems can utilize a system of density manipulation to maintain threshold levels of productivity in each tank. This requires that a significant population of small fish is initially stocked in a tank, and as the biomass of the fish and the subsequent feeding levels approach the capacity of the associated water quality control systems, the fish population is periodically sub-divided to additional tank systems, thus maintaining the biomass loading at acceptable levels in each. This way, relatively constant and efficient use is made of all tanks and water quality control systems.

The last of these scenarios is employed by Fresh-Culture Systems, Inc. (Patent Number 4,913,093), and involves the use of multiples of 15 production tanks,each with its own water quality control system, and three manipulations of fish density within a 6-month growth cycle. This system is designed to produce from 40,000 to 50,000 pounds of fish annually. The management system will also provide for the continuous availability of harvests throughout the year.

There are a number of philosophies underlying the selection of this process for maintaining continuous loading and harvesting:

- The use of multiple tanks with individual water quality control capabilities provides for the isolation of each system from all others. Any water quality, disease, management or hardware problems are usually completely isolated to a single production unit.

- The necessary integration of appropriate system components can be accomplished without relying on off-the-shelf hardware, often designed for non-aquacultural purposes. Individual water quality control requirements can be met with custom engineering appropriate to the scale of the individual tank systems. Example: The sizing of such filters as rotating biological contactors for the totality of a commercial system's demands requires the use of units that are constrained by certain associated limitations. These include the use of sewage treatment units or proportionately sized designs with the same problematic gearmotors, chain drives, shafts and pillow blocks, and the increased energy requirements. Such units will suffer from the many related malfunctions.

- Pumping requirements for circulating all water from multiple locations through a centralized water quality control center involves much greater expense and risk than carefully engineered individual tank systems. Also, to provide for continuous harvest capabilities, it is still necessary to manage multiple culture units with harvestable-sized fish populations, or use partial harvests. To manage these and the various additional production levels will require many tanks and probably multiple water quality control systems.

- The design of the water quality control systems must be integrated with themethodology for the maintenance of continuous loading. Examples: (1) Round tanks are required for optimal management of solid wastes. However, these tanks do not lend themselves to the maintenance of several size classes of fish within the same tank. (2) Oxygen injection systems using U-tubes require the mixing of the highly super-saturated water into large volumes of tank water. Therefore, this will require the mixing of all system water, either into various tank locations, or into individual very large tanks. This reduces the potential for the isolation of water quality, disease or stress problems.

- Maintaining mixed size populations within the same tank requires the use of partitioning, which is difficult in round tanks, or the mixing of sizes within the tanks (not possible with some cannibalistic species). Such mixing will exacerbate the variability within size classes of stocks and the stunting of

some percentage of the fish populations. Larger, more aggressive, fish will usually get the food. It will also require the use of graders to remove harvest-sized fish on a continual basis, resulting in more stress to the fish than with periodic manipulations of a fraction of the population.

- In order to maintain good water quality, it is as important to manage the bacterial populations growing on the biofilters as it is to manage the fish populations. This is simplified with numerous tanks requiring moderate loading levels, each with a properly scaled biofilter. Multiple tanks of tremendous additive volume and production capacity being serviced by a single bank of biofilters are more dangerous to maintain and to integrate with all other water quality control systems.

- Often, the economic viability of a project depends on the acquisition of inexpensive space, such as old factory buildings. It is seldom possible to require or afford the excavation necessary for such water quality systems as multiple U-tubes, below-grade settling systems, or angled tank floors as designated for the very large units carrying all size grades of fish, or large multiple tank units coupled with centralized water quality control.

- The required movement of significant levels of the fish population provides an opportunity to assess the size and health of the fish, determine the success of the previous growth cycle, and adjust feed levels accordingly.

For these reasons, multiple-tank density manipulation systems have been designed as described, by Fresh-Culture Systems, Inc. The potentialy! adverse consequences of density manipulations are minimized by proper management techniques and the use of appropriate hardware. The stress of handling the fish is reduced by the harvesting techniques. This includes the use of customized hardware to accomplish the move quickly, without subjecting the fish to rough handling, while maintaining optimal water quality and proper conditioning.

The use of the density manipulation system results in an increase in the production capacity of a 15 tank system from about 18,000 pounds capacity (15 tanks harvested twice per year of about 600 pounds of fish), to over 40,000 pounds per year. The system provides for the harvest of eight tanks of fish weighing over 600 pounds every six weeks.

This system also results in the maintenance of substantial feeding levels within each tank at all times. Three density manipulations provide for the initial stocking of four times as many fish as could normally be sustained. And having four size classes of fish being cultured at all times, and eight tanks available for harvest every six weeks in combination with two holding systems, results in the continuous availability of market-sized fish.

Guidelines for Implementation

Initially, a tank is stocked with the appropriate number and size of fingerlings that will, following six weeks of intensive culture at maximum feeding rates, approach the relatively fixed capacity of the growing environment to maintain water quality. At that time, half of the fish are moved to a previously harvested tank. Also at that time, another single tank is stocked with the initial number of fingerlings.

This is followed by another six weeks of intensive feeding, a subsequent density manipulation from each of the two equally stocked units and the second fingerling tank to three additional units, and the stocking of another tank of fingerlings. A third sub-division will result in the equal apportionment of the originally stocked fish within eight tanks, and the subsequent stockings similarly divided in the other seven tanks.

When in full operation, the fifteen tanks will include one tank of fingerlings stocked at a predetermined density. Two tanks will be maintained at half, four at a quarter, and eight at an eighth of this original density. Every six weeks, eight tanks are harvested, seven are sub-divided and one is restocked with fingerlings.

Operational Results

The described techniques were developed and tested over several years in full scale systems operated by Fresh-Culture Systems, Inc., using tilapia, striped bass, hybrid striped bass, catfish, carp, yellow perch, coho salmon and trout. The culture systems and density manipulation technologies have now been implemented successfully in commercial culture systems for over three years.

By incorporating appropriate handling methods, expected losses accrued during or following the movement of fish are negligible. The techniques are not excessively management intensive, and result in a much more realistic and economically viable system than all other alter-natives tested. However, the potential of this system for maintaining continuous loading and harvestable levels of fish is completely dependent on the integrated development of closed-system hardware and technologies for water quality control, and proper aquacultural management techniques. The use of several multiples of the 15 tank design provides for the up-scaling of the system on the same or different site locations.

Summary

Recirculating aquaculture systems provide many advantages towards environmentally controlled production of finfish throughout the year, removing the normal climatic and water resource restrictions. However, the systems are expensive, usually require costly indoor space, and have continuous and substantial operational requirements beyond traditional

methods. Because of the initial and operational costs of the complex water quality control systems, it is imperative that they function at near capacity at all times. The basis for determining the operational capacity of any system for growing fish is the average level of feeding that can be maintained. This will, in turn, determine the design parameters of all of the filtration and aeration equipment. Therefore, any significant fluctuation from an optimal average feeding level will reduce the operational efficiency of the system below economically acceptable levels.

The described system for maintaining an acceptable threshold level of average feeding has been tested and employed for several years, and is effective. However, to achieve and maintain this effectiveness requires not only the multiple tank/density manipulation protocols, but the use of integrated hardware and technologies for all levels of water quality control, fish and feed management, the availability of water reservoir, nursery and holding systems, and the employment of many energy conservation techniques.

References

Buck, T.R. 1991. Niche Marketing of Aquacultured Products, Network Newsletter, Vol. 10, No.1, Alternative Aquaculture Assoc., Breinigsville, Pennsylvania.

Chen, S., Timmons, M.B. and J.J. Bisogni, Jr. 1989. Fine Solids Removal by Foam Fractionation from Aquacultural Wastewater, Paper presented February 12-16 at Aquaculture '89, Westin Bonaventure Hotel, Los Angeles, California.

Chen, S, and R.F. Malone. 1991. Suspended Solids Control in Recirculating Aquaculture Systems, Pages 170-186 In Proceedings Engineering Aspects of Intensive Aquaculture Symposium, Cornell University, April 4-6, 1991, Ithaca, NY.

Colt, J. and B.J. Watten. 1988. Applications of Pure Oxygen in Fish Culture, Aquacultural Engineering, Vol. 7, pp. 397-441.

Kugelman, I.J. and S.D. Van Gorder. 1991. Water and Energy Recycling in Closed Aquaculture Systems, Pages 80-87 In Proceedings Engineering Aspects of Intensive Aquaculture Symposium, Cornell University, April 4-6, 1991, Ithaca, NY.

Liao, P.B. and R.D. Mayo. 1974. Intensified Fish Culture Combining Water Reconditioning with Pollution Abatement, Aquaculture, Vol. 3, pp. 61-85.

Losordo, T.M., J.E. Easley, and P.W. Westerman. 1989. Preliminary Results of a Survey on the Feasibility of Recirculating Aquaculture Production Systems, Presented at the American Society of Agricultural Engineers Winter Meeting, December 12-15, 1989, New Orleans, Louisiana.

Losordo, T.M. 1991. An Introduction to Recirculating Production Systems Design, Pages 32-47 In Proceedings Engineering Aspects of Intensive Aquaculture Symposium, Cornell University, April 4-6, 1991, Ithaca, NY.

Makinen, T., Lindgren, S. and P. Eskelinen. 1988. Sieving as an Effluent Treatment Method for Aquaculture, Aquacultural Engineering, Vol. 7, pp. 367-377.

Severson, R.F., J.L. Stark and L.M. Poole. 1987. Use of Oxygen to Commercially Rear Coho Salmon, Papers on the Use of Supplemental Oxygen to Increase Hatchery Rearing Capacity in the Pacific Northwest, Bonneville Power Administration, Portland, Oregon, pp. 25-34.

Speece, R.E., M. Madrid and K. Needham. 1971. Downflow Bubble Contact Aeration, Journal of Sanitary Engineers Division, Proceedings American Society of Civil Engineers, 97(SA4), pp. 433-441.

Turk, P.E. and P.G. Lee. 1991. Design and Economic Analysis of Airlift Versus Electrical Pump Driven Recirculating Aquaculture Systems, Pages 271-283 In Proceedings Engineering Aspects of Intensive Aquaculture Symposium, Cornell University, April 4-6, 1991, Ithaca, NY.

Watten, B.J., and L.T. Beck. 1985. Modeling Gas Transfer in a U-tube Oxygen Absorption System: Effects of Off-gas Recycling, Aquacultural Engineering, Vol. 4, pp. 271-297.

Wheaton, F.W. 1977. Aquacultural Engineering, New York, New York, John Wiley and Sons.

Wheaton, F.W., Hochheimer, J. and Kaiser, G.E. 1990. Fixed Film Nitrification Filters for Aquaculture, Pages 272-303 In Proceedings Aquaculture Water Quality Symposium, World Aquaculture Society, Baton Rouge Louisiana.

Preproduction Analysis of Economic Profitability for Recirculating Aquaculture Systems

Dr. Patrick D. O'Rourke
Illinois State University
Normal, Illinois

Introduction

We know we can grow fish in various controlled (non-wild) environments. We know we can clean the water, remove ammonia from the water and add dissolved oxygen to the water to increase the intensity of production. We know that genetic improvement is possible (even probable) to improve feed conversion, disease resistance and dressing percentage. The list could go on as researchers from many disciplinary backgrounds working in industry and public institutions have a wealth of knowledge and experience contributing to the development of recirculating aquaculture systems.

What we don't know (yet) is whether or not fish can be profitably produced on a commercial scale in recirculating aquaculture systems. I will propose today that there are three basic approaches to fill this gap in our knowledge of commercially viable recirculating aquaculture systems:

1. Build and operate commercial scale systems to collect the needed operational and economic data.

2. Contract with private vendors and/or producers to collect and provide the needed operational and economic data.

3. Wait for private industry to adopt recirculating systems on a broad scale or abandon such systems due to lack of commercial potential in terms of profitability. Like many other livestock production systems, commercial recirculating aquaculture systems are in a constant state of development as new knowledge and technological changes are tested for incorporation into the systems.

Commercial recirculating aquaculture systems are characterized by higher capital investment (per acre) than the more traditional pond based aquaculture systems. The hope is that the cost of this higher capital investment (per acre) is relatively lower when measured per pound of fish producedannually. Or, if higher, then is compensated by healthier fish, safer production vis-a-vis predators, improved quality vis-a-vis taste or flavor, improved feed conversion and shorter growout time, reduced labor requirements, and perhaps a price advantage deriving from the opportunity to market a stable quantity of high quality product year round from production sites geographically nearer large consumer demand centers.

Demand and Supply

The consumption of fish and fish products in the United States has grown in total and on a per capita basis over the last three decades. Given the nutritional value of fish and the expectation that the consumers' demand for nutritious and the safe fish protein sources will continue to increase on a per capita basis, the outlook for aquaculture production seems positive.

There seems to be agreement among many researchers and industry leaders that much of the expected increase in fish product demand will be met by aquaculture. There is less agreement on where it will come from (geographically) and how it will be produced. Economic history teaches us that without government intervention food is produced where the cost of production and distribution to consumers are lowest for the product (and its attributes) which consumers want to buy. The technologies employed and methods used in food crop production, once mastered, have contributed to major shifts in production. (Examples: tomatoes from Indiana to California, broilers from many small enterprises to large, geographically concentrated intensive enterprises.)

Pond production of fish will probably continue to be concentrated in the few geographic areas where climate, water supply, land, and predator and disease control are most favorable. This includes both domestic and foreign production areas. Pen and race-way aquaculture systems will also be restricted geographically by water supply and quality as well as effluent restrictions. Commercial recirculating systems, technically speaking, may be located in a larger number of geographic locations. That is, recirculation systems may be located where one or more local factors make it impossible or unprofitable, technically, for pond, pen or race-way systems to operate.

If recirculating systems are to play a major role in the commercial production of fish for food they will have to demonstrate the ability to deliver products (with the attributes consumers demand) at a lower cost per pound than other production systems. If the cost of production is relatively higher in recirculating systems their use will probably be restricted to special markets, higher valued products and/or perhaps fingerling production. If future developments lead to commercial recirculating systems with cost of production similar to other aquaculture systems their use will most likely increase dramatically, especially in areas close to input suppliers and population (demand) centers.

Public vs Private Investment

United States agriculture and consumers have benefitted from substantial research and development investments by private industry and public universities. Public investment has generally (but not always) been targeted to research and development in areas of greater risk and uncertainty because this higher level of risk and uncertainty is often a disincentive for investment by private firms necessarily concerned with profitability. This type of investment by the public sector has often been guided by an acceptable level of confidence or expectation or probability that the payoff (benefits) to the general public will be greater than the investment (cost of research).

Is research and development of commercially profitable recirculating aquaculture systems an appropriate area for public investment? I believe the answer lies partly in whether the research community can ascertain that there is an acceptable level of confidence that such systems can be commercially profitable.

Measuring Profitability and Project Value

Profitability deals with more than just the level of profit generated by an enterprise. It is a relative term measuring profit in relation to investment, and in the final analysis, the rate of return on owners' investment and the value of that investment relative to other opportunities over the life of the project.

The research community does not presently have adequate economic data for commercial recirculating aquaculture systems to describe the cost relationships or potential profitability of such systems. The data required include (in broad categories):

- capital investment requirements and the economic life of equipment and facilities,
- managerial and labor requirements,
- fixed and variable expenses of operation, per pound of production, and
- an assessment of risks, risk management strategies, and the costs of risk in terms of higher expected expenses or higher required rates of return on investment.

Ascertaining the potential profitability and value of commercial recirculating aquaculture systems may be approached with the data listed above from private operations, public operations, or as a third less desirable option, from simulation studies.

Risk and Uncertainty

The risks and uncertainties involved in commercial recirculating aquaculture systems can be put into three categories: (1) production or operations, (2) marketing or price, and (3) financial. Measuring the impact of these risks/uncertainties, developing management strategies for reducing them, and identifying appropriate management decision rules, given the risk adverseness of owners and producers is a necessary component of studies designed to ascertain the profitability of such systems. Once again the data needed may be obtained in one or more of the three approaches mentioned above.

The risks and uncertainties of a production system and what one knows about those risks and uncertainties have a predictable impact on the expected return on investment in such a system. The greater the risk, the greater the cost and/or the higher the expected rate of return on investment.

Conclusion

There is a great deal more to say on the topic of economic profitability and a great deal more detail and rigor one can apply. However, as my old major professor used to say, "Don't mess around painting the trim and polishing the door knobs if the foundation of the house is falling apart." That probably derived from the admonition to not build castles on shifting sand. Research on the biological, chemical and engineering efficiencies of fish production in recirculating aquaculture systems must continue. However, this research should be coordinated with systems research that examines the economicefficiency and potential profitability of commercial scale recirculating aquaculture systems.

We, as professionals, are very capable and well meaning in the research and development work we conduct and manage. Collectively we have improved the probability of profitable commercial recirculating aquaculture systems. The next stage, and it has already begun, is to put commercial scale recirculating aquaculture systems into production, collect the necessary data for ascertaining profitability, and continue improving the technological, chemical, biological, and economic efficiency of those systems.

Performance of Fine Sand Fluidized Bed Biological Filters

Dallas E. Weaver, Ph.D.
Scientific Hatcheries
Huntington Beach, California

Abstract

A fine sand fluidized bed biological filter operating at 143 l / (min-m^2) on a commercial aquaculture production system was tested for pH, oxygen and shock loading tolerance. The input pH was varied between 8.8 and 5.4, with no measurable ammonia (< 0.1 ppm -N) in the discharge between pH of 8.8 and 5.7. The input ammonia was increased from the normal 0.8 ppm range to as high as 4 ppm before measurable ammonia was obtained in the discharge. The oxygen in the input was decreased until the zero oxygen point had moved down into the fluidized bed within 0.8 meters of the bottom before ammonia or nitrite was measurable in the effluent.

Introduction

One major goal of aquacultural engineering research is a closed-cycle fish production system which can produce tonnage amounts of aquatic products without the use of large amounts of water or land resources. However, like all real-world engineering problems, this goal must be achieved at competitive costs.

Unlike the problems of waste removal associated with the production of terrestrial animals (poultry, cattle, pigs, etc.), the problem of separating the waste products from the growth environment of aquatic animals is non-trivial. In other words, the aquacultural engineer must create a complete life support and waste treatment system if he wants to produce aquatic animals at high densities with limited water usage.

One of the critical unit operations in designing the life support system for aquatic animals is the soluble waste removal operation. A major component of the soluble waste produced by growing aquatic animals is ammonia. Hence, a unit operation which will remove, oxidize, or incorporate ammonia is required in all closed cycle aquaculture operations. The fine-media fluidized-bed biological filter can be viewed as primarily a soluble waste removal device. Operated aerobically, it will oxidize, with suitable bacterial species, most soluble organic compounds. Anaerobic operation will result in the reduction of nitrates and sulfates.

A medium to coarse media fluidized bed biological (FBB) filter has been used for ammonia removal at a steelhead facility (Owsley 1989).

A fine-media fluidized-bed biological filter consists of a tank and a method for uniformly introducing the input water into the bottom of a tank which is partially filled with a fine medium such as a very fine sand (i.e., 60 to 90 mesh silica). As the water flows up through the tank, the sand is expanded to form a fluidized level. Mechanically, this fluidized sand behaves like a true high-density fluid. It has no shear strength, and seeks it own level.

Looking closely at the individual sand grains, they appear to be in continual free fall through the water, hence the mass transfer capability between the liquid and the particle surface is excellent. With the very small size of the sand, the surface area per unit volume available for bacterial growth is exceptionally large (about 1 ha/m^3 or 10 m^2/liter). This large unit surface area combined with high mass transfer rates for the liquid to the surface, creates an excellent habitat for bacterial growth. These bacteria will then oxidize the waste products.

If the medium is very uniform, the bed can become mixed in a time period much less than the reproductive time of the bacteria. The water passing through the bed can be described as approaching plug flow. This combination of mixed flow on the solid phase particles with their attached bacteria, and plug flow of the liquid phase, can enable a biological filter to lower the discharge concentration of a particular chemical below the minimum concentration required for bacterial survival, as long as the input concentration is maintained above the minimum. The latter can be artificially increased. This unique capability of operating below the bacterial threshold concentration can be useful in certain aquaculture designs and situations where very high water quality is mandatory (broodstock maturation and larval development).

With a fine medium, assuming that the method of water injection is gentle enough, the bacteria are not sheared from the grains by the flowing water. This lack of removal of the bacteria allows the filter to be operated with very long sludge retention times (SRT). It is the author's view that a long SRT system is more biologically stable.

A fine-medium fluidized-bed filter, when operated with low loading rates per unit volume (less than about 0.5 kg of BOD — including ammonia—per cubic meter-day), can be substrate (food) limited. Under these conditions, there is more surface area or bacterial habitat than there is food available for the bacteria. This mode of operation eliminates the microbiological competition for habitat, thereby allowing competition

and selection of bacterial strains to be determined by the ability of the bacteria to utilize the waste products at low concentrations. Under these low loading conditions, there is zero net production of biomass in the filter, no media loss, and no maintenance required.

Low loading rate operation allows the systems to handle pulse loading situations better than other types of biological filters. This can be visualized by observing that the system contains a large inventory of starving microorganisms. When presented with food, these starving organisms will be able to increase their consumption much more than well fed, fast-growing organisms. Under these conditions, the system will self-select for organisms that will remove nutrients when available, thereby handling pulse loading situations.

When operating at higher volumetric loadings (greater than about 1 kg of BOD per cubic meter day) there is a net growth of bacteria in the filter. This net growth appears as a sludge-sand particle fluidized layer on top of the fluidized sand medium layer. As high loading rates continue, this sludge layer continues to grow at the expense of the sand layer and will overflow the filter unless removed or otherwise processed. If there are other suspended solids separation unit operations in the system, it is possible to introduce a shearing device in the sludge layer which will separate the biomass from the sand particles, thereby allowing the sand grains to return to the sand layer and the biomass to leave the system in the discharge.

With a fine-media filter, the hydraulic loading is in the 150 to 400 liters per minute per sq meter range (4 to 10 gpm/ft^2). At these low rates, these fine-media filters can become oxygen limited. The author has not found a successful method of addingoxygen beyond the amount in the input water (i.e. three phase systems with oxygen, water, and fluidized sand are subject to froth flotation problems which result in media loss in the discharge). Obviously, under oxygen limited conditions, the amount of ammonia removed per minute becomes constant, independent of the input concentration.

The history and experience of aquaculturists with fine media fluidized bed biological filters is a long one. FBB filters have been in commercial aquacultural use for 17 years and have established a long term track record of decades of operation without shutdown or failure. Some smaller systems have been used on salt water tropical fish (approximately 1 m^3 of volume) for 8 years. Other systems have been used in live fish wholesale operations, with strong pulse loadings (full systems on Mondays and Wednesdays and nearly empty on Friday) for several years.

System Description

The performance of a particular unit operation of an aquaculture system is often determined by the behavior of the overall system. For example, if an experiment-sized fine-media FBB is operated in parallel with a large rotating biological contactor (RBC) on an aquaculture system, it is possible to have the FBB overloaded with near zero oxygen in the discharge. One may then wonder why the FBB seems to sludge up and quit working (Miller 1985). The normal substrate concentration in a RBC system, which only removes a fraction of the substrate per pass thru the filter, could be high enough to overload a fine-media FBB, resulting in an anerobic condition.

The experimental system consists of two FBB's (1.5 m diameter X 2.2 m high) operating at 328 l/min each. The medium in these filters is # 90 silica sand, which has about 75% bed expansion at 0.3 cm/sec plug flow vertical velocity.

These filters are part of a "feeder guppy" growout system which produces about 15 million fish per year. Along with producing the saleable fish, the system also produces new broodstock (the poor feed conversion ratios on maturing fish result in a higher percentage of the nitrogen input in the feed showing up in the water rather than as protein in the animal). The water from the FBB's is split into 2 flows, one of which is passed thru a packed column with air for oxygen addition and carbon dioxide removal. The pH of the water increases about 0.25 pH units across the column. Another part of the stream goes through pure oxygen columns to produce high-oxygen water. These two streams enter the culture tanks and can be varied according to the load on each individualtank. From the culture tanks, the water flows by gravity to settling tanks for suspended solids removal. Pumps take the water from the clarifiers and return the water to the filter.

The reader should be aware of the fact that the clarifiers are undersized and the suspended solids removal can be considered the weakest link in the system. This weakness affects the performance of the FBB by increasing the solids input, which can add to the sludge level in the filter. This weakness also adds to the oxygen consumption of the filter by creating more soluble organics from the decomposition of the suspended solids in the clarifiers. Hence, any results presented on the amount of feed input into the system, relative to the filter size and flow rate, would not be valid for another system where the suspended solids removal is excellent and no solubilization of the solid waste occurs.

The FBB's on this system introduce the flow to the bottom of the tank via a series of vertical pipes connected to a manifold on the top of the filter. Each one of the vertical pipes can be removed independently of the others and can be changed or maintained without shutting down the filter. This design concept of having the distribution manifold on the top of the tank, rather than on the bottom, is more expensive, but it is easier to maintain without shutting down the system. This type of design tradeoff is related to the system in question. In this case, where decades of continuous

production are required without shutdown, the extra cost is justified.

Since the experiments necessary to test the filters would create conditions that would stress the fish if the test conditions were applied to the system as a whole, it was necessary to test only one filter and conduct only short term dynamic tests. For example, to test the pH response of the filter, the pH of the input water to the filter was varied between 5.4 and 8.8. The duration of these tests was kept short enough to allow the fish to stay between pH 6.2 and 7.8. Considering the value of the livestock in the system, the author was not willing to push the animals beyond this range.

Experimental Design

The first objective in attempting to describe the performance of the FBB is to describe the normal operation. Since the input of nutrients to the filter and the filter's metabolic demand vary over a 24 hour period, the response of the filter has been measuredover a 24 hour period. The oxygen in, oxygen out and the associated metabolic rate of the filter were continuously measured. Other variables such as ammonia, nitrite, pH, and alkalinity were periodically measured throughout the 24 hour period.

The second objective was to measure the ammonia oxidation as a function of pH. Most biological filters demonstrate a well defined pH range within which ammonia is nitrified to nitrate (Kruner 1983; Sharma 1977). With a zero net growth biological system with decades of continuous operation, one would theoretically expect the system to have a greater dynamic range than a newly set up and very young, highly loaded filter in which surface area competition determines the fitness of the bacteria.

With two filters on the system, we could change the short term pH of one filter without swinging the whole system. This approach had a problem in that the data is only for short time ps at that pH and that the carbonate system is not at equilibrium. When decreasing the pH with HCl, the CO_2 was not allowed to escape, thereby creating a very high free CO_2 concentration. These condition are not representive of the steady state response, but can be considered indicative. At high pH, the free CO_2 concentration was abnormally low and also not representative.

While varying the pH, the metabolism of the filter, pH, ammonia, nitrite, and oxygen were measured at the input and output of the filter.

To properly describe the behavior of a FBB, it was necessary to measure the response to increases in ammonia loading. Exploratory experiments indicated that this filter produced effluent with non-detectable ammonia levels (less than 0.1 ppm TAN) as long as there was oxygen in the discharge (greater than 1 ppm). Therefore, it was decided to increase the input oxygen as much as feasible without getting too many gas bubbles from supersaturation, and increase the ammonia until we started to see some ammonia in the discharge. Due to mechanical and other limits, I was not able to maintain the high ammonia input for more than a few hours, hence the results of these experiments demonstrate the pulse loading capability of the filter rather than the steady state maximum ammonia oxidation rate.

Another viewpoint for looking at the filter would be to view the behavior as a function of the discharge oxygen levels. Previous experience has shown that this filter produces no ammonia with a zero oxygen point at the top of the sand layer. The experimental objective was to move the zero oxygen point downward in the filter tothe point where the system started to produce nitrite or ammonia.

Equipment, Materials and Methods

Oxygen and temperature measurements were made with Royce¬Instruments 9010 and 9040 oxygen meters. Probes from these meters were placed in inlet and discharge streams. The oxygen meters were connected to the serial RS485 communication bus which runs through the facility. Royce¬portable meters were used to measure verticle oxygen profiles within the filter under low oxygen conditions. Air calibration as per instructions was utilized.

The pH was measured by several different instruments which include Omega¬ PHTX-91's connected to an Opto 22¬Optomux Brain Board thru a Module AD3. A Cole Parmer series 7142 pH controller and a Jenco model 6009 portable pH meter were also utilized. All pH meters used Innovative Sensors¬1PB probes. Calibration used standard buffer solutions.

Ammonia and nitrite were measured using Hach¬ kits and Sea Test¬test kits for low range and Hach¬ nesslers reagent test kit for higher ranges (dilution and the low range kits were also used for the high ranges to check consistency).

Since there is no oxygen input to the filter other than from the input water, the metabolism of the filter can be determined from the oxygen mass balance. This was accomplished via Life Supports¬ software, which provided an online real time metabolic output in strip chart format. All data available on the RS485 bus was also collected, graphed, alarmed and archived from the same software.

The RS 485 bus connecting the instruments in the hatchery was connected to an Apple Macintosh¬ SE/30 computer via the RS422 serial port on the Mac. The Life Supports aquaculture control system software performed the data collection. This system consists of a collection of functions such as alarms, oxygen devices, temperature devices, pH devices, oxygen valves, feeders, metabolism devices, etc. where each object is

relatively independent of the other. Icons, which can be easily moved and associated with the objects are located on a schematic or layout drawing of the hatchery. Activating the icon will display the device, which can be turned on or off, modified, created, ordestroyed. Most of the functions can accomplish a wide variety of tasks such as data logging, alarms, verbal alarms with information over the PA system, archiving data, integrating the data (i.e. feed amounts), handling temperature adjustments to data or feed amounts, etc.

The same software runs all the feeders and controls the oxygen and pH levels, along with the above monitoring functions which can be used for experimental purposes. However, the normal work load on the computer is presently near maximum capacity (without either new hardware or major software changes) and there were not enough instruments or computer cycles available to test all the filters and parts of the system simultaneously. With the present high biological load on the system, the risk and cost of devoting more resources to this project were not acceptable.

Results and Discussion

Normal operation

The normal operation of the system consists of adding approximately 15 kg/day of SilverCup¬ starter and #1 feeds into the system over about a 13 hour period. How the waste products from this feed get distributed betweer the two filters varies depending upon how the tanks are harvested and restocked. Effectively, the two filters get the majority of their input water from separate tanks. This results in a range of input ammonia (-N) concentrations from 0.3 to 1.2 ppm. The discharge ammonia concentration is normally non-detectable with the test kits used (less than 0.1 ppm). The only time that measurable ammonia is detected in the discharge is when there is no oxygen in the discharge. Using a portable meter, it was determined that the zero oxygen point in the filter will move down to within 0.8 meters of the bottom before 0.1 ppm ammonia and 0.1 ppm nitrite is detected in the discharge.

When the zero oxygen point in the filter is near the bottom, some nitrite can be measured at the 0.1 ppm (-N) level. For various mechanical reasons, it was not possible to sample, measure or rationally analyze the fluid phase closer than 0.8 meters to the bottom. The near bottom conditions in this filter appear to have some large scale mixing of the input water, thereby giving very erratic measurements. It would be expected that stable operation with plug flow of the liquid phase would take a distance to establish that would be proportional to the distance between water injection points (about 0.5 meters).

Because the FBB's are followed by packed columns, it is possible to get an indication of the generalBOD removal of the filter by observing the biofouling on the packing material. As long as the discharge oxygen from the filter is under feedback control with a set point of 0.5 ppm oxygen, there is no indication of any biofouling or large scale biomass growth in the return piping system. Measurements of the mass transport coefficients associated with the packed columns indicate an alpha of 1.0 (i.e. the discharge water behaves like pure water), whereas the FBB input water will have an alpha in the 0.5 to 0.75 range when the system is being fed.

In terms of normal operation procedures, ammonia in the discharge is not monitored except when the oxygen in the discharge goes to zero. These operating procedures have produced satisfactory results without a system crash.

Input pH variation

The pH of the input water was increased by adding soda ash and decreased by adding HCl to the input water, via a variable speed chemical feed pump. The resulting pH was measured and the feed rate of the pump adjusted until the desired pH was achieved. The range of pH tested was between 8.8 and 5.4 on the inlet water. During these experiments the input ammonia (N) was between 0.8 and 0.9 ppm and the discharge oxygen was maintained above 3.7 ppm (3.7 to 6.3 ppm in the discharge). The temperature was 25°C and the starting alkalinity was 2 milliequivalents per liter, with a starting pH of 6.8. The input and output ammonia, nitrate, and oxygen were measured and the metabolism of the filter was calculated.

The results are very unexciting. No ammonia was detected in the discharge while increasing the pH. Decreasing the pH finally did show a fall-off in nitrification at a pH of 5.35 when the discharge increased to 0.5 ppm with a 0.9 ppm inlet. Once breakthrough was achieved, soda ash was pumped into the system in order to return the system to normal. Guppies will tolerate this low pH water but they are stressed. Upon increasing the input pH, the ammonia returned to nondetectable levels.

The decrease in ammonia oxidation is reflected in the decrease in metabolism rate of the filter. This effect is shown in Figure 1.

Ammonia Loading

One way to measure the performance of a biological filter is to increase the loading and monitor the response. This approach was accomplished by adding ammonium sulfate to the water at the filter input with a chemical feed pump. As previously mentioned, this filter normally produces non-detectable ammonia concentrations in the effluent as long as there is oxygen in the discharge. To obtain reasonable results with oxygen in the discharge, it was necessary to add some 200 ppm oxygen water to the input.

This series of experiments was conducted at a

temperature of 25°C with an input pH of 6.84 with 2.0 meq/l alkalinity. The input oxygen levels were increased to the 18 to 20 ppm range. These very high supersaturation levels of oxygen did create gas bubbles in the FBB, so the true metabolism rate is not fully known. The results are shown in Table 1.

Table 1. Ammonia Loading Experiments

Time	NH_3 (N) Input	NH_3 (N) Output	NO_2 (N) Output	Oxygen In	Oxygen Out	n
2:40	0.8	<0.1	ND	19.4	11	165
3:40	4.2	<0.1	ND	18.4	3.3	277
4:13	3.6	0.1	ND	17.9	2.8	300

The results are confusing in that no ammonia was detected in the effluent at 4.2 ppm input while there appeared to be 0.1 ppm at 3.6 ppm ammonia input. This behavior may indicate the possibility of luxury consumption on the part of the nitrification bacteria. However, the main factor to observe is that the peak rate of nitrification is about 4 times the average rate. This observation is consistent with the zero net growth concept previously discussed.

It is believed that the bacteria in this filter are literally starving with little or no net production of biomass. When presented with extra nutrients, it is easy for the existing bacteria to increase their consumption by a factor of 4 for a short time.

The behavior of this class of filters toward pulse loads can be very useful for certain classes of business such as temporary holding facilities, depuration facilities and similar operations where the loading varies over very short time periods. An example of such a situation is a goldfish wholesale operation. Such a facility will receive large shipments on Monday and Wednesday and be sold out by Friday. Fine sand FBB's at two such facilities have demonstrated shock load capability without crashing. One of the facilities experienced a human error which disconnected a full tank of fish from the system and killed them. The fish rotted for three days and then someone put the tank back on the recycle system. Within two hours the entire system was free of ammonia in the filter discharge water.

Other Experience

Similar FBB's are operated by other people with similar results. The Spring and Groundwater Institute in Shepherdstown, WV. operates two filters, the same size at the filter tested in this study, at 740 l/min each (larger media). A typical data set shows input TAN of 0.7 ppm with an output of 0.04 to 0.06 ppm. This system is being feed 35 lb/day of feed and has microscreens for suspended solids control. This means that the FBB's see primarily ammonia and very little general BOD. This is reflected by a change in oxygen across the filter in the 3.3 ppm range.

Filter installations on fish holding systems have shown exceptional stability in the face of highly variable loads. Two local installations were tested by the author and both had < 0.1 ppm TAN in the effluent with inlet values between 0 and 1.2 ppm. These are unfed systems and suspended solids removal is not a major problem.

Another unit on a tilapia broodstock facility has demonstrated excellent nitrite removal capability. With the large volume and internal structure in this facility, most of the ammonia is converted to nitrite in the system, which in turn became the limiting factor. Installation of a 350 l/min FBB on the system eliminated this nitrite problem.

Some very highly loaded systems have experienced some stability and performance problems that are difficult to understand except in the context of very high loading, poor suspended solids removal (slip stream SS removal rather than full flow) in the balance of the system, and a physical design that is hydraulically different than the systems described above. This system is running deeper filters at higher velocities with a less well-graded medium.

Conclusions

Fine-media fluidized-bed biological filters have demonstrated the highest level of effluent water quality and the greatest dynamic response to pulse loading of any of the biological filters used in aquaculture. Whenever high quality water is desired, fine media fluidized bed biological filters should be part of the system.

References

Owsley, David E. , Jerris, John S. , Owens, R. 1989. Ammonia Removal Allows Effluent Reuse at Fish Hatchery Using Fluidized Bed Reactors, 43rd Purdue Industrial Waste Conference Proceedings, pp. 449-457.

Kruner, G., Rosenthal, H. 1983. Efficiency of Nitrification in Trickling Filters using Different Substrates, Aquacultural Engineering, pp. 49-67.

Miller, G. E., Libey G. S. 1985. Evaluation of Three Biological Filters Suitable for Aquacultural Applications, J. World Mariculture Society, Vol. 16, pp. 158-168.

Sharma, B., Ahler, R. C. 1977. Nitrification and Nitrogen Removal, Water Research, Vol. 11, pp. 897-925.

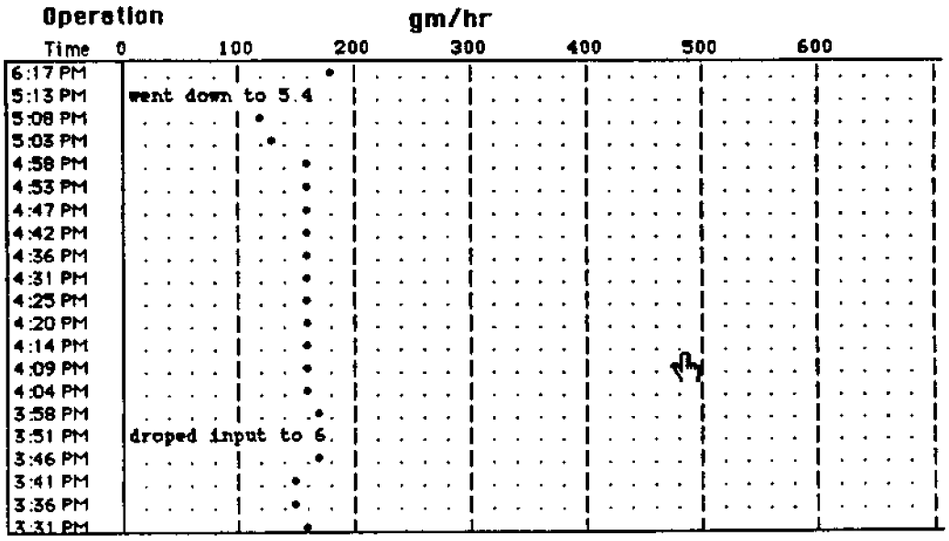

Figure 1. Screen Dump of metabolism of the filter as the pH was decreased. metabolism in gm/hr of oxygen consumption.

Maximizing Nitrification with Rotating Biological Contactors (RBC)

George S. Libey
Virginia Polytechnic Institute and State University
Blacksburg, Virginia

Abstract

Rotating Biological Contactors (RBC) have been used in sewage treatment since the 1960's. Because of the difference in the waste being treated, caution must be exercised in transferring to aquaculture the results of research conducted for sewage treatment. RBC's have been tested as a component in various recirculating aquaculture system configurations. They have been shown to possess inherent features that make them well suited for aquacultural applications. These include: self aerating - they supply most or all of the oxygen necessary for nitrification; low head requirement - head loss is minimal across the filter; reliable and stable performance - RBC's are not subject to sudden "crashes" and in all "head-to-head" tests of various biofilter configurations, RBC's have proven to be superior in overall performance.

Even though several reports have addressed RBC performance and design in aquacultural situations, optimum combinations of various factors influencing RBC performance have not yet been established. Further research will be necessary to optimize the design and operation of RBC's in recirculating aquaculture systems.

Introduction and Background

A rotating biological contactor (RBC) consists of circular plates, usually corrugated, attached to a central shaft. The plates are immersed in the water to be treated and the shaft is rotated. As the shaft revolves, the plates are alternately immersed in the liquid and exposed to the air. The shear force exerted by the liquid as the plates rotate causes continuous sloughing of excess biomass. The alternate emersion and immersion of the microbial attachment site provides oxygen necessary for nitrification.

The RBC process was first investigated in the United States and Germany in the 1920's (Dallaire 1979). These early devices used wooden plates for bacterial attachment surfaces. Incorporation of various types of plastics as the media allowed development of commercial scale units in the 1960's and 1970's. The higher specific surface area of these new plastics improved unit efficiencies and reduced project costs. An additional improvement was the use of air drives. Air supplied by a small header in the RBC tank provided buoyancy when captured in cups mounted around the perimeter of the media. This buoyant force was used to rotate the shaft.

RBC were originally developed for the treatment of domestic and industrial wastes. As a result, much of the research on RBC design has come from the civil/environmental engineering field. Because of the difference in the waste being treated, care must be exercised when transferring this information to the aquaculture field.

Some of the factors that have been identified as influencing RBC performance are: hydraulic loading, mass loading, detention time, number of stages, rotational velocity, waste concentration, and water temperature.

A common design criterion for RBC's has been hydraulic loading (m^3/m^2/day or m/day) due to the first order kinetics demonstrated in the removal of BOD and nitrification (Antonie 1976). At a specific hydraulic load, a specific percentage of substrate will be removed regardless of the influent concentration (Antonie 1976). However, this approach is not universally endorsed. European RBC's generally utilize mass loading (kg/day/1000 m^3) as a design parameter (Steiner 1980). Steiner (1980) notes that problems with undersizing are not common in Europe whereas in the United States design limits are often exceeded.

Detention time, that time required to replace the water in the RBC tank, has been shown to influence RBC performance. Overall filter performance increases with an increase in detention (Grady and Lim 1980). However, these authors point out that there is an upper limit beyond which nothing is gained.

Due to the kinetics of substrate removal, RBC efficiency has been shown to improve through the use of multiple shafts and staging the effluent from one unit to the next (Wu et al. 1980; Grady and Lim 1980). Staging can dramatically improve performance of an RBC for a given set of conditions (Wu et al. 1981).

Rotational velocity has been shown to be related to performance, with performance increasing as peripheral velocity increases (Hynek and Chou 1979; Banerji 1980). It was theorized that the water exerts the shear force necessary to continuously slough off excess biomass and maintain a more or less constant film thickness on the medium. Variable speeds have be

used to tune the efficiency of RBC stages, rotating at higher velocities where oxygen demand was greatest (Hynek and Chou 1979).

Banerji (1980) found in the studies he reviewed that the rate of ammonia oxidation through a RBC decreased with increased influent concentration. This suggests that at very high influent levels little or no oxidation would occur. However, this would be far outside the range of tolerable levels for aquaculture.

The effect of temperature on RBC performance appears to reflect the influence of temperature on microbial processes. Banerji (1980) provided a graphical presentation of the interaction of hydraulic load and temperature on removal efficiency for both BOD and nitrification. Performance increased with a rise in temperature and decreased with increased hydraulic load. The improved performance as temperature increased appeared to follow enzyme kinetics with an asymptote between 20° C and 30° C (Banerji 1980).

Aquaculture Applications

The use of RBC's in aquaculture systems have been reported by several authors. Lewis and Buynak (1976) were among the first to describe the incorporation of a RBC in a system for growing fish, channel catfish (Ictalurus punctatus). They concluded that the system they tested appeared practical for warmwater tank culture. Further studies on this basic configuration included the coupling of hydroponics as part of the water treatment system (Lewis et al. 1978; Lewis et al. 1981). The hydroponically grown plants were used to remove nitrates and phosphates from the water. Their final design "may prove attractive for private or commercial production of food fish and vegetables" (Lewis et al. 1981).

Parker (1981) described a large scale system incorporating a RBC, subsurface silos, plate clarifiers, and an air-lift pump. The report included the results from preliminary tests that indicated "commercial applications of the silo system could probably be justified for cash crops of high value."

Nunley and Libey (1991) reported on the production of reciprocal cross hybrid striped bass (Morone saxatilis X M. chrysops) at three fish densities in replicated, pilot-scale recirculating aquaculture systems incorporating a RBC. The system they tested was able to maintain excellent water quality and the RBC performance was reliable and consistent even at highest fish density tested (Table 1 and 2).

Drapcho and Brune (1984) reported on work utilizing a polyurethane media applied to a RBC. This study, utilizing artificial culture water, investigated the effect of ammonia concentration, detention time, and rotational velocity on RBC performance. The response to various detention times indicated that as detention time decreased the ammonia removal rate increased. Rotational velocity had no effect on performance at long detention times, but at short detention times a reduction in rotational velocity resulted in decreased ammonia oxidation rates.

In a field test of a RBC utilizing the polyurethane foam media, Saxton and Brune (1985) noted increasing the BOD load decreased ammonia oxidation rates. In their tests ammonia oxidation was reduced by twenty percent as a result of a 12-15 mg/l BOD. An increase in inlet ammonia concentration was accompanied by an immediate increase in nitrite levels. They found the filter could respond quickly to ammonia but there was a delayed response to elevated nitrite concentrations. They also reported that a filter not exposed to direct sunlight outperformed one that was exposed by approximately thirty percent.

Several authors have compared RBC's with various other types of biological filters. Four biological filters (RBC, biodrum, trickling filter, submerged anaerobic filter) were tested utilizing artificial culture water by Rogers and Klemetson (1985). In this study the RBC provided the best ammonia removal, better than either the biodrum or trickling filter. The submerged anaerobic filter provided denitrification, a process they suggested could be coupled with the nitrifying filters for complete removal of the inorganic nitrogen waste.

Van Gorder and Fritch (1980) tested systems incorporating either a RBC, submerged gravel, or submerged plastic media filter. Each duplicated system was operated until maximum loading capacities were obtained. The greatest standing crop, best survival, and best food conversion rate were attained in the systems with RBC's.

Miller and Libey (1985) evaluated the comparative performance of three biofilter configurations (RBC, fluidized bed reactor, packed tower) under anticipated load levels and utilizing common water from a tank containing channel catfish. They found the RBC provided the best nitrification efficiency. The packed tower was somewhat better than the fluidized bed reactor but neither could match the performance of the RBC.

Conclusions

The RBC has been shown to possess certain inherent features that make it well suited for use in recirculating aquaculture systems. Among these are:

1. Self aerating - the alternate immersion and emersion of the media provides aeration to the attached microbes and aerates the liquid;

2. Low head requirement - RBC's are low head devices and require only centimeters of water head for operation;

3. Non-clogging - due to the shear force from rotating

the media through the water, excess biomass is continuously sloughed, leading to the maintenance of a highly active biofilm of relatively uniform thickness;

4. In "head-to-head" tests of various biofilter configurations, RBC's have proven to be superior in overall performance;

5. Once established, the RBC performance is reliable and not subject to sudden "crashes".

Guideline for Implementation

A review of the literature on RBC applications in aquaculture indicates they are the best biofilter tested to date for incorporation in recirculating systems. However, most studies have concentrated on system evaluation and performance. Only two reports (Drapcho and Brune 1984; Saxton and Brune 1985) have concentrated on filter design. These studies utilized a polyurethane media, an innovative but not widely tested configuration. The amount of contact between the biofilm and the waste stream greatly impacts oxidation rate and filter performance. A comparison of two studies utilizing different detention times (Miller and Libey 1985; Nunley andLibey 1991) indicate that increasing detention was accompanied by an increased ammonia oxidation rate for a given mass loading (Figure 1). Miller and Libey (1985) utilized a 15.4 minute detention while Nunley and Libey (1991) reported on a RBC with a 5.8 minute detention. These two studies also indicate that for a given detention, a specific percentage of substrate was removed regardless of the mass load (Figure 2). In both studies, a peripheral velocity of 0.3 m/s was used so the impact of rotational velocity could not be obtained.

Two operational parameters, detention time and rotational velocity, affect biofilm/waste stream contact. An optimum combination for these two factors has not been established for various situations. Further research is necessary to optimize the design and operation of RBC's in recirculating aquaculture systems.

Literature Cited

Antonie, R.L. 1976. Fixed Biological Surfaces - Wastewater Treatment, CRC Press, Incorporated, Cleveland, Ohio.

Banerji, A.K. 1980. ASCE Water Pollution Management Task Committee Report on Rotating Biological Contactor for Secondary Treatment, In Smith, E.D., R.D. Miller, and Y.C. Wu (ed.)., Proceedings: First National Symposium/Workshop on Rotating Biological Contactor Technology, Champion, Pennsylvania.

Dallaire, G. 1979. Behind the Rapid Rise of the Rotating Biological Contactor, Journal of the Civil Engineering Division, ASCE, pp. 72-73.

Drapcho, C.M. and D.E. Brune. 1984. Application of a Retained Biomass Surface in a Rotating Biological Contactor, Proceedings: Second International Conference on Fixed Film Processes, Pittsburgh, Pennsylvania, pp. 1389-1403.

Grady, C.P.L. and H.C. Lim. 1980. Biological Wastewater Treatment, Marcel Dekker, Incorporated, New York, New York, 963 pp.

Hynek, R.J. and C.C. Chou. 1979. Development and Performance of Air-Driven Rotating Biological Contactors, Proceedings of the 34th Industrial Waste Conference, Purdue University, West Lafayette, Indiana, pp. 805-815.

Lewis, W.M. and G.L. Buynak. 1976. Evaluation of a Revolving Plate Type Biofilter for Use in Recirculating Fish Production and Holding Tanks, Transactions of the American Fisheries Society, Vol. 105, pp. 704-708.

Lewis, W.M., J.H. Ap, H.L. Schramm, Jr., and A.M. Brandenburg. 1978. Use of Hydroponics to Maintain Quality of Recirculated Water in a Fish Culture System, Transactions of the American Fisheries Society, Vol. 107, pp. 92-99.

Lewis, W.M., J.H. Ap, A.M. Brandenburg, and K.D. Schnoor. 1981. On the Maintenance of Water Quality for Closed Fish Production Systems by Means of Hydroponically Grown Vegetable Crops, Proceedings of the World Symposium on Aquaculture in Heated Effluents and Recirculation Systems, Berlin, Germany.

Miller, G.E. and G.S. Libey. 1985. Evaluation of Three Biological Filters Suitable for Aquacultural Applications, Journal of the World Mariculture Society, Vol. 16, pp. 158-168.

Nunley, C.E. and G.S. Libey. 1991. The Performance of Hybrid Striped Bass in a Recirculating Aquaculture System, (In review).

Parker, N.C. 1981. An Air-Operated Fish Culture System with Water-Reuse and Subsurface Silos, In Kinney, E.C. (ed.), Proceedings of the Bio-Engineering Symposium for Fish Culture, American Fisheries Society, Fish Culture Section, Bethesda, Maryland.

Rogers, G.L. and S.L. Klemetson. 1985. Ammonia Removal in Selected Aquaculture Water Reuse Biofilters, Aquacultural Engineering, Vol. 4, pp. 135-154.

Saxton, R.H. and D.E. Brune. 1985. Maintaining Water Quality in Recirculating Trout Culture, Field Evaluation of a Retained Biomass Nitrification Filter, ASAE Paper No. 85-5100, American Society of Agricultural Engineers, St. Joseph, Michigan.

Steiner, C.G. 1980. The Biological Approach to the Rotating Disc Process, In Smith, E.D., R.D. Miller, and Y.C. Wu. (ed.), Proceedings: First National Symposium/Workshop on Rotating Biological Contactor Technology, Champion, Pennsylvania.

Van Gorder, S.D. and J.D. Fritch. 1980. Filtration Techniques for Small-Scale Aquaculture in a Closed System, Proceedings of the Annual Conference of Southeaster Association of Game and Fish Commissioners, Vol. 34, pp. 59-66.

Wu, Y.C., R.D. Smith, and Y.T. Hung. 1980. Modeling of Rotating Biological Contactor Systems,

Biotechnology and Bioengineering, Vol. 22, pp. 2055-2064.
Wu, Y.C., E.D. Smith, and J. Gratz. 1981. Prediction of RBC Performance for Nitrification, Journal of the Environmental Engineering Division, ASCE, Vol. 107, pp. 635-652.

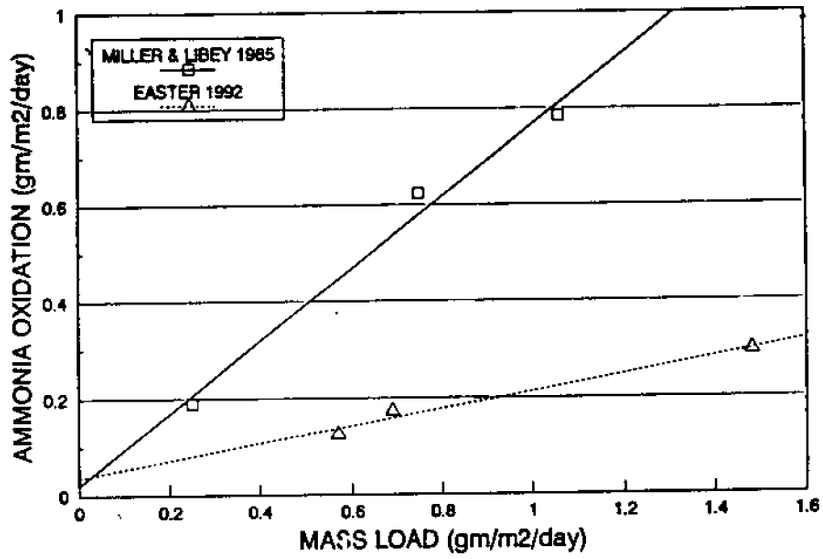

Figure 1. RBC performance at various mass loadings.

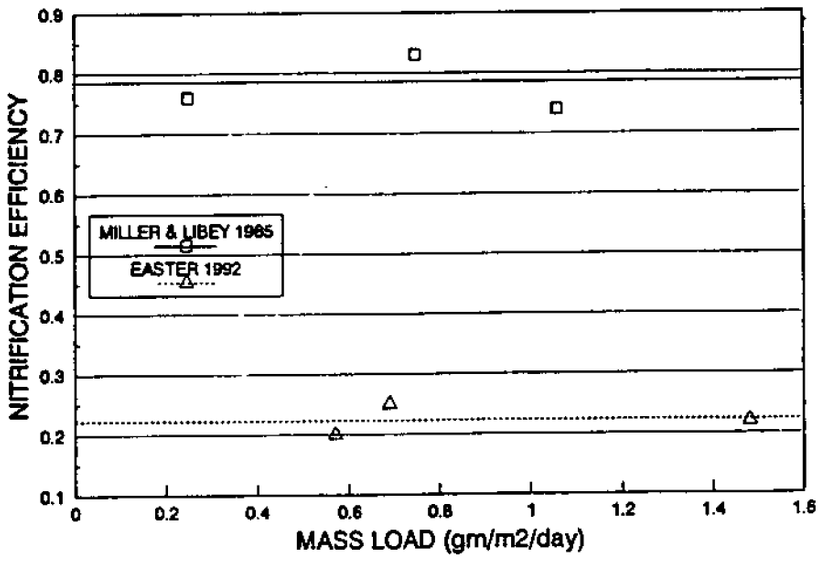

Figure 2. RBC efficiency at various mass loadings.

Table 1. Performance of reciprocal cross hybrid striped bass in a recirculating aquaculture system. Growth trial duration = 32 weeks (224 days). High density = 216 fish/m³; Medium = 108; Low = 54. (Adapted from Nunley and Libey 1991).

	Initial Wt. ea. (gm)	Initial Wt. tank (kg)	Final Wt. ea. (gm)	Final Wt. tank (kg)	Gain Wt. ea. (gm)	Gain Wt. tank (kg)	Gain / day Wt. ea. (gm)	Gain / day Wt. tank (kg)	FCR	Mort. (%)	F.W. Additions % Sys Vol/wk
High	34.9	62.8	741.9	741.9	377.3	679.1	1.68	3.03	1.34	4.2	91.3
Medium	50.8	45.7	542.1	487.9	491.3	442.2	2.19	1.97	1.39	1.8	46.5
Low	43.3	19.5	676.1	304.2	632.8	284.7	2.83	1.27	1.32	2.7	43.3

Table 2. RBC performance in a recirculating aquaculture system containing reciprocal cross hybrid striped bass. Growth trial duration = 32 weeks (224 days). High density = 216 fish/m³; Medium = 108; Low = 54. BOD and TSS, no change across RBC. Mean values. (Adapted from Nunley and Libey 1991).

	Biomass (kg fish/m^3)	RBC Mass Load (gm TAN/m^2/day)	Ammonia Oxid. (gm TAN/m^2/day)	Inlet Oxygen (mg/l)	Outlet Oxygen (mg/l)	BOD (mg/l)	TSS (mg/l)
High	89.0	1.48	0.296	5.9	6.5	15.2	21.3
Medium	58.6	0.69	0.173	5.9	6.6	9.4	13.7
Low	36.6	0.57	0.125	7.2	7.3	6.2	11.6

Biofiltration and Solids Capture with Low Density Bead Filters

Ronald F. Malone and David E. Coffin
Louisiana State University
Baton Rouge, Louisiana

Abstract

Floating bead biofilters employ filtration, expansion, and settling modes to obtain nitrification and solids removal in a single unit. Solids capture is excellent with removal rates 741.3 Kg/m^3 being documented in recent demonstration studies. Nitrification rates appear comparable to other biofilters on an areal basis averaging about 0.29 gm/m^2-day of TAN conversion. With specific surface areas in the range of 1000 m^2/m^3, nitrification capacities on a volumetric basis (gm/m^3-day) appear to fall between the rotating biological contactors and fluidized beds. Bead filters minimize water loss during washing sequences, permitting manipulation of solids and biofloc retention time. Enhanced nitrification performance has been observed for hydraulically washed units with increased washing frequency. The relationship between washing frequency and nitrification capability is not clearly understood, indicating additional research needs.

Introduction

Economic considerations are limiting the adoption of recirculating systems as a means of production of low cost food fish. High capital costs coupled with operational expenses place recirculating systems at an economic disadvantage to more extensive processes. Widespread adoption of recirculating technologies is dependent upon the development of cost effective waste treatment processes.

This paper describes the use of "floating bead" filters which perform the dual function of solids capture and biofiltration. Illustrative data collected from biofiltration units are presented and potential advantages of the approach are discussed.

Background

Bead filters are a logical extension of early filtration units which employed a submerged bed of media to provide for biofiltration and inherently solids capture (Haug and McCarty 1972). Many of the designs discussed in this paper evolved (Table 1) from submerged rock filtration units developed to support recirculating soft blue crab (Callinectus sapidus) shedding systems (Perry et al. 1979; Malone and Burden 1988a; Manthe et al. 1985). The brackish water submerged rock filters were capable of supporting large numbers of unfed premolt crabs which excreted ammonia in amounts comparable to a wide variety of aquatic organisms but contributed relatively little biochemical oxygen demand (BOD) or solids loading to the systems. Although early problems with oxygen supply (Manthe et al. 1985)were addressed by design modifications employing in-filter airlift recirculation pipes, the carrying capacity of the submerged rock filter was ultimately limited by biofloc accumulation (Manthe et al. 1988). This limitation results in rapid hydraulic failure when the filter is subject to moderate loads from species which are fed. Accumulation of large amounts of organically rich solids promotes luxurious growths of biofloc that clogs the bed's natural porosity and may inhibit nitrification (Siddall 1974; Lewis and Heidinger 1981; Paller and Lewis 1988).

The biofouling problem leads to a broad category of filters, "Expandable Granular Biofilters" (EGB's) which are designed to maintain good solids capture and biofiltration characteristics while facilitating the removal of solids by intermittent expansion of the filtration media. In addition to providing a means of rapidly removing excreted solids, expansion provides a mechanism for harvesting biofloc eliminating the threat of biofouling.

The upflow sand filter (Burden 1988) incorporated the fundamental features of an EGB. The filter has two modes of operation, i.e., filtration and washing, which are controlled by the upflow fluxrate. (unit of flow per unit of horizontal cross-sectional area). During the normal filtration mode (0.20 - 0.41 m^3/m^2), the unit operates as an effective biofilter, reflecting the high specific surface area of the sand media. As the hydraulic transitivity of the media declines due to solids capture and biofloc growth the upflowing waters gently expand the bed preventing development of compression forces which cause "caking" of organically loaded downflow, pressurized sand filters. Captured suspended solids and loosely attached biofloc are flushed from the system during the washing sequence which employs a high upward fluxrate (1.83 - 3.05 m^3/m^2) to fluidize the bed. The filter's solids capture capability eliminates the needfor separate solids capture devices. In moderately loaded soft-shell crab (Malone and Burden 1988a) or soft-shell crayfish systems (Malone and Burden 1988b), a favored filtration configuration employs an upflow sand filter as the sole filtration unit. Performance of the upflow sand filter is limited by its oxygen transport capabilities. Since the filter is submerged, oxygen delivery is controlled by the fluxrate. Fluidization of the coarse sand media (1.2 - 2.4 mm) occurs at fluxrates below 0.41 m^3/m^2 with good biofilm development. Considering a typical bed

depth of 0.38 m in the unfluidized operational mode, oxygen delivery capacity is limited to about 5.30 kg/m^3-day; this translates into about 2.25 gm/m^2-day on a specific surface area basis. Management options are further limited by the high water loss associated with the washing sequence.

The fluxrate concerns were addressed in a second EGB format which employs low density plastic (polyethylene) injection molding beads as a media, allowing pressurized operation and solids removal without high water loss. These media float; filtration occurs as upflowing waters pass through the beads which are usually retained by a screening system. Bed expansion is accomplished by air injection (Cooley 1979), hydraulic wash (Wimberly 1990) or by mechanical means (Malone 1992). Upon expansion, separation of the aggregated solids and floating beads occurs spontaneously, allowing for rapid consolidation or agglomeration of solids into a concentrated sludge (0.5 - 3%). Settling is normally accomplished within the filtration chamber.

As an example, Figure 1 illustrates the operational steps employed with a mechanically-washed bead filter. The dominant phase is filtration, which continues until solids capture and biofloc accumulation increase backpressures prompting a washing sequence. During the 30-60 second backwash phase, the circulation pump is closed off and the imbedded propellers are activated. The turbulent expansion of the bed releases captured solids while shearing excessive biofloc. The settling phase, with both pump and propellers inactivated, allows the filtration bed to reform while the solids settle. Settling is very rapid. Settling times are controlled by the degree of consolidation desired. Typical settling periods are 2 - 5 minutes in duration. Sludge is removed in the final phase, just prior to re-initiation of the filtration mode.

The bead filters are relatively insensitive to fluxrate. Fluxrates in the range of 1.43 m^3/m^2 do not adversely impact filter performance (Wimberly 1990). Concerns about oxygen transport are alleviated. Oxygen delivery capabilities exceed about 14.12 kg/m^3-day or over 11.84 gm/m^2-day on specific surface area basis for a typical 0.6 m deep bed. Oxygen supplies are sufficient to support the assimilative capacity of the biofilm, given the short detention times (20 - 60 seconds) within the bead media and the low substrate regimes associated with the recirculating systems. Nitrification performance is limited by specific surface area and biofilm retention capabilities. The biofilm thickness is controlled by the washing/settling sequences that occur within the filter eliminating concerns about water loss.

Methodology

Illustrative data were collected from a series of studies conducted under the direction of the authors. Collection of filter influent and effluent samples were used to define filter performance, avoiding complexing factors such as in-situ nitrification. Although it must be recognized that the data sets are not directly comparable, all data utilized was uniformly screened to reflect typical substrate ranges and to avoid low alkalinity (Paz 1984) or low pH (Allain 1988) which are clearly inhibitory to the nitrification process.

All analytical procedures conformed with guidelines set forth by APHA (1985; 1989). Total ammonia nitrogen (TAN) analyses employed distillation coupled with direct nesslerization. Nitrite nitrogen was determined with the sulfanilamide based calorimetric test. Dissolved oxygen levels were determined with a Yellow Springs Model 57 oxygen probe calibrated with the Winkler Titration Method. Flow rates were measured with a stop watch and bucket. Total suspended solids (TSS) were determined gravimetrically.

Discussion

Performance

Intrinsic to advocation of the EGB approach to recirculation treatment is the need to control solids and biofloc to (1) reduce the waste burden carried by the system treatment components and (2) to enhance nitrification capacity by elimination of excessive heterotrophic bacterial growths. The two are interrelated as bacteria implementing decomposition of solids and dissolved organics dominate biofiltration units (Lewis and Heidinger 1981). Increased biofilm thickness inhibits circulation through the biomedia unfavorably, shifting kinetics to a regime controlled by biofilm nutrient transport. Transport limitation implies inefficient utilization of biofilm, increasing the biofilm mass that must be supported by the aeration and degasification components. Further, it has been contended that heavy growths of heterotrophic bacteria place the critical nitrifying bacterial population in an unfavorable niche (Harremoes 1982).

Oxygen consumed during filtration, or OCF, can be effectively used to monitor levels of bacterial activity in submerged biofilters (Hirayama 1965, 1974; Manthe et al. 1988). The term is normalized to the weight of organics supported (or food consumed) empirically by the relationship:

$$\overline{OLR} = \sum_{j=1}^{N} (OCF_j / W_j)$$

where: $\overline{OLR}$ = mean oxygen demand exerted per unit mass of organisms at a fixed feeding rate, or alternately, per unit of food consumed (mg-O_2 per kg per day); W_j = weight of organisms (or food) for the j observation (kilograms).

It can be reasonably assumed that waste production from organisms maintained at a fixed feeding rate is constant (Colt 1978). A high biofiltration burden generally reflects high levels of heterotrophic activity.

Depending on the species, $\overline{OLR}$'s can be reduced significantly (30-65 percent) by efficient solids removal (Malone et al. 1990). Additional improvements are realized by controlling residence time of the biofilm grown in the biofilter.

The net effect of minimizing $\overline{OLR}$ by rapid solids removal and efficient biofilm harvesting was demonstrated (Table 2) by Wimberly (1990) with a hydraulically washed bead filter. In an experimental system holding channel catfish maintained at 1 percent feed, dramatic increases in carrying capacity were realized as backwashing frequency increased and the OLR dropped. The nitrification capacity of the biofilter increased because excessive heterophic populations were controlled by frequent removal during the washing sequence. Additionally, breakdown of the captured solids was minimized through the corresponding drop in retention time.

Use of backwash frequency as a means of optimizing filter performance is still only partially understood. Clearly, frequent backflushing reduces the impact of solids by preventing their biodegradation in the system. However, current bead filter designs link solids removal with biofloc harvesting. Wimberly (1990) observed that continuedincreases in backwash frequency would ultimately reduce the mean cell residence time of the slow growing nitrification bacteria (Sharma and Alhert 1977) causing a decline in carrying capacity. This hypothesis is not completely accepted as continually expanded fluidized beds demonstrate excellent nitrification abilities (Burden 1988; Thomasson 1990) despite the use of large (1.2 -2.4 mm) abrasive sands. However, the shear energies involved in the expansion process vary dramatically. Mechanically washed designs impart much higher shear energies than either bubble-washed or hydraulically washed configurations. With the current level of knowledge filter performance must be empirically "tuned" by varying the backwashing frequency and monitoring the TAN conversion capability.

Despite uncertainties concerning filter optimization, nitrification capacities of the bead filters examined by the authors compare favorably with other biofiltration units (Table 3). The similarities between dramatically different biofiltration units when normalized to specific surface area is striking. Noting that the data were screened to avoid recognized inhibitory conditions, all the biofiltration units seem capable of providing total ammonia nitrogen (TAN) conversions in the range of 200-300 mg/m^2. Bead filter nitrite areal conversion rates lag behind the rotating biological contactor and fluidized beds examined. The nitrobacter species responsible for nitrite conversion display long reproduction times (Sharma and Alhert 1977). These bead filters may have been operated beyond the optimum backwashing frequency for the nitrobacter species. On a volumetric basis, the bead filters displayed conversion capabilities intermediate to the RBC and fluidized beds, reflecting their good specific surface area.

The solids capture capacities of the bead filters greatly exceed their nitrification capabilities when the TSS to TAN excretion ratios generated by fish are considered (Wimberly 1990; Liao 1970; Wheaton 1977). Whereas about 21 milligram of TSS are excreted per milligram of TAN excreted, the overloaded MP&L bead filter displayed capture ratio's averaging 69 (kg TSS captured/kg TAN converted) without showing any indication of biofouling. The MP&L facility displayed volumetric solids capture rates of 741.3 kg/m^3 without being optimized for solids capture. The authors' believe much higher solids capture rates are obtainable with high fluxrates and frequent backwashing. In any case, it is clear that bead filters used for combined nitrification and solids capture should be sized according to their nitrification capacity.

Bead filters compare quite favorably against fixed media nitrification filters which are exposed to the air. High recirculating flowrates dictated by ammonia mixingconstraints in combination with the low substrate regimes eliminate concerns about oxygen transport. Trickling filters and RBC's are clearly effective biofiltration units. However, the need to maintain high porosity to avoid biofouling limits specific surface area, thus controlling their volumetric conversion capacity.

On the other hand, fluidized beds are clearly superior nitrification units displaying volumetric nitrification rates double those of bead filters. Use of a bead filter for nitrification in lieu of a fluidized bed is predicated on the assumption that integrated treatment with a bead filter will prove cost-effective. That is a bead filter sized for nitrification will prove less costly than a properly sized capture device and a fluidized bed.

In the authors' opinion, resolution of the issue will be dependent upon future clarification and refinement of bead filter capabilities through research and commercial evaluation. In particular, specific issues that need to be addressed are:

1. Development of a rationale for determining backwash frequencies which optimize nitrification under different loading regimes,
2. Documentation of solids capture capabilities with emphasis on behavior of small particles (<10 μm),
3. Identification of the impact of fluxrate on solids capture and nitrification, and
4. Clarification of the factors controlling biofilm thickness and nitrification capacities on an areal basis.

References

Allain, P. A. 1988. Ion Shifts and pH Management in High Density Shedding Systems for Blue Crabs (Callinectus Sapidus) and Red Swamp Crawfish (Procambarus Clarkii), Master's Thesis, Louisiana State University, Baton Rouge, Louisiana.

APHA. 1989. Standard Methods for the Examination of Water and Wastewater, 18th Edition, American Public Health Association, New York, New York.

Burden, D. G. 1988. Additional Waste Characterization Data on the Blue Crab (Callinectes sapidus), Technical Memorandum TM-14 (not published), Department of Civil Engineering, Louisiana State University, Baton Rouge, Louisiana.

Colt, E. J. 1978. The Effects of Ammonia on the Growth of Channel Catfish Ictalurus punctatus, PhD Dissertation, Department of Engineering, University of California Davis, Davis, California.

Cooley, P. E. 1979. Nitrification of Fish-Hatchery Reuse Water Utilizing Low-Density Polyethylene Beads as a Fixed-Film Media Type, M. S. Thesis, University of Idaho, Moscow, Idaho.

Haug, R. T. and McCarty, P. L. 1972. Nitrification in the Submerged Filter, J. Water Poll. Control Fed., No. 44, pp. 2086-2102.

Hirayama, K. 1965. Filtration in a Recirculating Hymelsystem, Bull. Jap. Soc. Sci. Fish., No. 32, pp. 983-990.

Hirayama, K. 1974. Water Control by Filtration in Closed Systems, Aquaculture, Vol. 4, pp. 369-385.

Harremoes, P. 1982. Criteria for Nitrification in Fixed Film Reactors, Water Science Technology, 14:167-187.

Liao. 1970. Salmon Hatchery Wastewater Treatment, Water and Sewage Works, Vol. 117, pp. 439-443.

Lewis, W.M. and R. C. Heidinger. 1981. Tank Culture of Striped Bass Production Manual, Illinois Striped Bass Project IDC FF-26-R, Fisheries Research Laboratory, South Illinois University, Carbondale, Illinois.

Malone, R. F., K. A. Rusch, and D. G. Burden. 1990. Kemp's Ridley Sea Turtle Waste Characterization Study: Precursor to a Recirculating Holding System Design, World Aquaculture Society, Vol. 21, No. 2, pp. 137-144.

Malone, R. F. 1992. Floating Media Biofilter, United States Patent No. 5,126,042, June 30th.

Malone, R. F. and D. G. Burden. 1988a. Design of Recirculating Blue Crab Shedding Systems, Louisiana Sea Grant College Program, Louisiana State University, Baton Rouge, Louisiana.

Malone, R. F. and D. G. Burden. 1988b. Design of Recirculating Soft Crawfish Shedding Systems, Louisiana Sea Grant College Program, Louisiana State University, Baton Rouge, Louisiana.

Manthe, D. P., R. F. Malone, and S. Kumar. 1985. Elimination of Oxygen Deficiencies Associated with Submerged Rock Filters used in Closed, Recirculating Aquaculture Systems, Pages 49-55 In Closed Blue Crab Shedding Systems, National Symposium on the Soft-Shelled blue Crab Fisheries, P.M. Perry and R.F. Malone, eds.

Manthe, D. P., R. F. Malone, S. Kumar. 1988. Submerged Rock Filter Evaluation using an Oxygen Consumption Criterion for Closed Recirculating Systems, Aquacultural Eng., 7: 97-111.

Paller, M. H. and W. M. Lewis. 1988. Use of Ozone and Fluidized Bed Biofilters for Increased Ammonia Removal and Fish Loading Rates, Progressive Fish-Culturalist, 50:141-147.

Paz, J. D. 1984. The Effects of Borderline Alkalinity on Nitrification in Natural Water Systems, Ph.D. Dissertation, Polytechnic Institute of New York.

Perry, H. M., J. T. Ogle, and L. C. Nicholson. 1979. The Fishery for Soft Crabs with Emphasis on the Development of a Closed Recirculating Seawater System for Shedding Crabs, Proceedings of the Blue Crab colloquium, October 18-19, 1979.

Sharma, B. and R. C. Alhert. 1977. Nitrification and Nitrogen Removal, Water Research, 11:897-925.

Siddall, S. E. 1974. Studies of Closed Marine Culture Systems, The Progressive Fish Culturist, 36(1).

Thomasson, Mark P. 1990. Nitrification in Fluidized Bed Sand Filters for Use in Recirculating Aquaculture Systems, Master's Thesis, Louisiana State University, Baton Rouge, Louisiana.

Wheaton, F. W. 1977. Aquacultural Engineering, 1st ed., Wiley-Interscience, New York, 708 pp.

Wimberly, Douglas M., 1990. Development and Evaluation of a Low-Density Media Biofiltration Unit for Use in Recirculating Finfish Culture Systems, Master's Thesis, Louisiana State University, Baton Rouge, Louisiana.

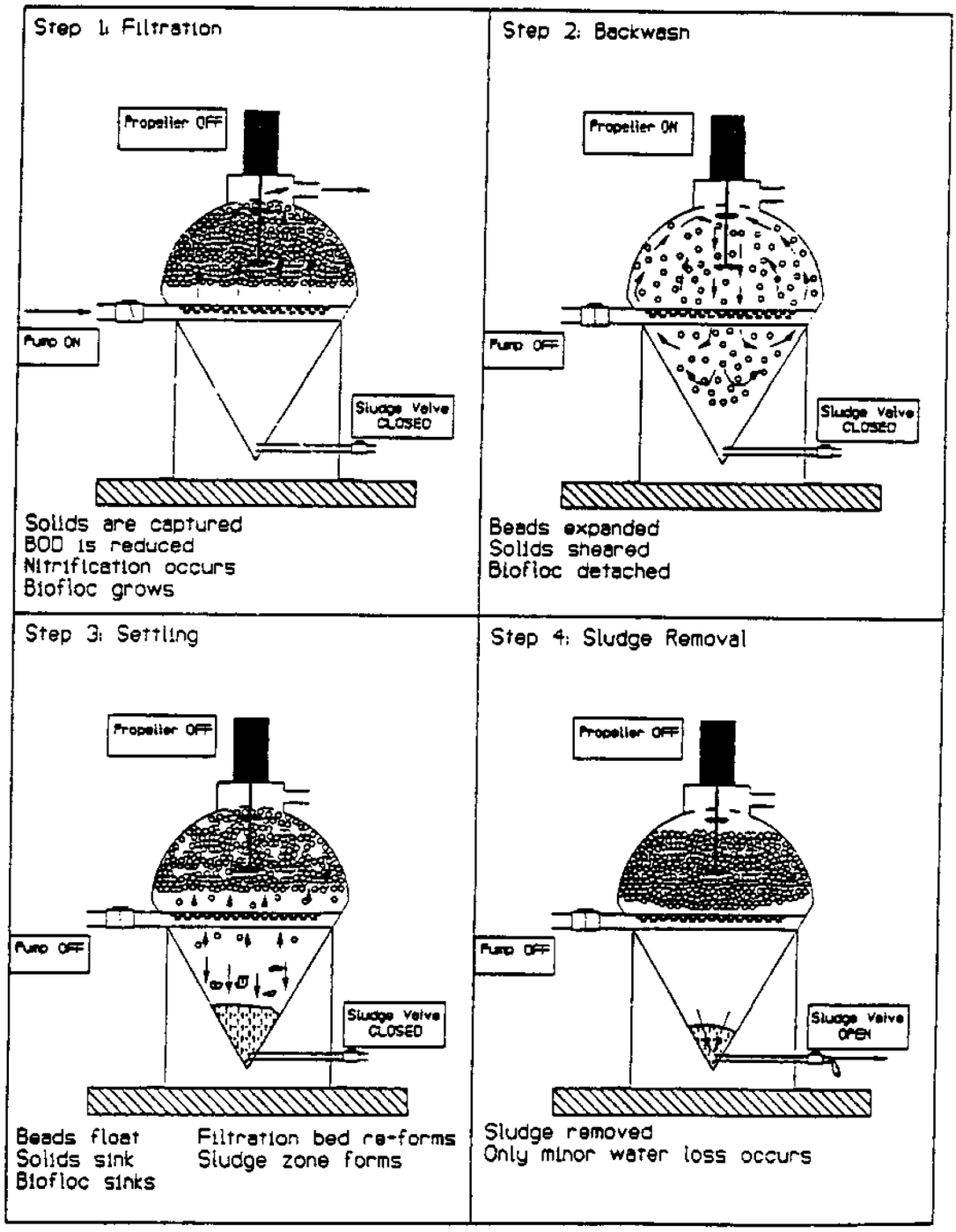

Figure 1. Operational Sequence for a Mechanically-washed Floating Bead Biofilter.

Table 1. Conceptual Evolution of Expandable Granular Biofilters

Unit	Mode(s) of Operation	Limitation	References
Submerged Rock Filter	Filtration	Biofouling	Perry et al. 1979, Manthe et al. 1988
Upflow Sand Filter	Filtration Expansion	Oxygen transport waterloss	Burden 1988
"Bead" Filter	Filtration Expansion Settling	Surface area	Cooley 1979, Wimberly 1990

Table 2. Increasing the Backflushing Frequency Increased the Carrying Capacity of Wimberly's (1990) Hydraulically Washed Bead Filter.

Backwash Frequency (day^{-1})	Carrying Capacity Kg-Fish/m^3 beads	Carrying Capacity Kg-Food/m^3 beads	OCR gmO_2/m^3 beads	MoCF Kg/m^3 beads
1	500	5	5.6	2.8
2	690	70	3.6	2.5
4	980	100	2.3	2.3
8	2180	220	1.5	3.3

Table 3. Bead Filters Display Intermediate Nitrification Capacities

	No Observations	Specific Surface Area (m^2/m^3)	Mean pH	Mean Effluent TAN (mg-N/l)	Areal TAN Conversion (g/m^2-day)	Volumetric TAN Conversion (g/m^3-day)	Description
Upflow sand	6	2350	7.15	0.31	0.064	152	Commercial soft-crayfish facility (Burden 1988)
Hydraulic washed bead filter	32	1230	7.54	0.62	0.231	286	Experimental scale catfish system (Wimberly 1990)
Mechanical washed bead filter	7	1050	7.38	1.1	0.291	308	Mississippi Power and Light demonstration facility, Greenville, Mississippi (1991)
Rotating biological contactor	6	150	7.51	0.53	0.280	41	Mississippi Power and Light demonstration facility, Greenville, Mississippi (1991)
Fluidized bed	40	2350	7.47	0.30	0.284	633	Experimental scale chemically fed (Thomasson 1991)

Sludge Management for Recirculating Aquacultural Systems

Shulin Chen, David E. Coffin, and Ronald F. Malone
Louisiana State University
Baton Rouge, LA 70803

Abstract

Recirculating systems produce a sludge discharge composed of partially stabilized excreta, uneaten food particles, and biofloc. Sludge generation ranges from 10 to 25 percent of the feeding rate on a dry weight basis. Solids concentrations range from as little as 0.02 percent to nearly 6 percent depending on the solids separation process employed. BOD_5/TSS ratios of sludge range from 0.10 to 0.2 with a TKN content of 4 to 6 percent. Following clarification, direct disposal by land application appears feasible for rural areas with dry climates. Additional stabilization in an aerated digester with disposal to landfills appears most feasible for urban areas with wet climates.

Introduction

Anticipated development of large scale recirculating finfish production systems has raised the issue of environmental impacts. Although recirculating technologies can avoid many of the impacts of dilute aquacultural wastes (Iwama 1991) by significant reductions in the volume of water discharged, they produce a concentrated sludge which can contribute to oxygen depletion and nutrient enrichment problems in receiving streams. Development of a rational approach to the sludge handling issue appears prudent and may impact the selection of internal water treatment components.

A guideline for sludge management that considers the strength and the amount of waste generated from recirculating aquacultural systems has not yet been established. The objectives of the present study were to (1) estimate the amount of sludge generated in a recirculating system, (2) investigate the characteristics of the sludge, and (3) identify treatment options.

Sludge Volume

Virtually all the wastes generated from a recirculating system originate from the feed. Assuming a typical feed conversion ratio of 1 to 2, and neglecting the impact of uneaten food, 80% of feed input to an aquacultural system (on a dry mass basis) will eventually be wasted as fish excretion products (Hopkins and Manci 1989). Sludge volume is a major factor in designing a sludge treatment system. Sludge volume generated from a recirculating system is controlled by the amount of solids produced (measured as total suspended solidsor TSS) and the degree to which the TSS is concentrated in the effluent stream. Total suspended solids production from a recirculating system can be estimated by considering direct fish excretion, solids breakdown, and biofloc production during biofiltration. The concentration is controlled by the solids removal technique employed to capture solids from the recycled stream. Solids production can be quantified through a mass balance analysis which considers the major solids fluxes:

$$dS/dt = FxP - (1-E) \times F \times P/W_f - K_d \times S + B_f (FxP) - h_s S \quad (1)$$

where,

- F = Feed rate (Kg feed/Kg fish-day)
- P = Fish in system (Kg)
- E = Solids excretion rate (Kg excreted/Kg feed consumed)
- W_f = Feed waste factor (Kg fed/Kg consumed)
- K_d = Solids biological decay constant (day $^{-1}$)
- S = Solids in system (Kg)
- B_f = Biofloc production factor from soluble BOD (Kg biofloc produced/Kg feed)
- h_s = Sludge harvest fraction (Kg solids harvested/Kg solids in system-day)

Assuming steadystate conditions and solving for S:

$$S = \frac{(1 - \frac{1-E}{W_f} + B_f) \times F \times P}{(K_d + h_s} \quad (2)$$

Neglecting the impact of uneaten feed (i.e. W_f=1), Equation 2 is simplified, clearly identifying the relationships controlling sludge production (measured as TSS) in a recirculating system:

$$S = \frac{E + B_f) \times F \times P}{(K_d + h_s} \quad (3)$$

TSS excretion rates (E) for trout and catfish are summarized in Table 1. The direct TSS excretion rate ranges from 0.40 (Speece 1973) to 0.52 kg/kg-feed (Liao and Mayo 1974) for trout and 0.43 kg/kg-feed for catfish (Wimberly 1990). Other reported ranges of TSS excretion rates for catfish ranged from 0.18 to 0.69 kg/kg-feed (Page and Andrews 1974; Gordon 1974, Ruane

et al. 1977). TSS excretion rates will clearly vary with species, temperature, and feeding rates. However, values of E in the range of 0.3 to 0.5 appear to be typical. The BOD_5 excretion rate can also generally be expressed as a ratio to the feeding rate (Table 2). The BOD_5 is excreted in soluble and particulate forms. Based upon the study on channel catfish, Murphy and Lipper (1970) reported the soluble BOD_5 as 58% of the total BOD_5 excreted; whereas, BOD_5 in particulate matter was 42%. Wimberly (1990) found that 23% of the BOD_5 excreted was in the soluble form and 77% in the particulate form.

The suspended solids production from biofiltration depends on the growth of bacterial biomass during BOD_5 removal and nitrification. Considering ammonia nitrogen excretion rates of 1.8 to 4.6% of the feeding rate (Page and Andrews 1974; Gordon 1974; Ruane et al. 1977; Wimberly 1990) and the stoichiometry of nitrification cited by Wheaton (1977), biomass production due to nitrification can be estimated as a negligible 0.03% to 0.09% of the feeding rate. The biomass production due to dissolved BOD_5 consumption, on the other hand, is more significant. For example, if 23% of the BOD_5 (0.22 kg BOD_5/kg-feed) is produced in the soluble form as reported by Wimberly (1990), the total soluble BOD_5 production will result in a biofloc production which averages 5% of the feeding rate. This portion of BOD_5 is absorbed during biofiltration, producing TSS levels equivalent to 9% of the feeding rate according to an estimation by the authors based on the stoichiometry of BOD_5 removal. Assuming the BOD has a composition similar to typical municipal sludge, values for the biofloc production constant, B_F, in the range of 0.08 to 0.12 can be reasonably assumed.

The sludge production constant S_p (Kg/day) from the system is defined simply as:

$$S_p = h_s \times S \qquad (4)$$

The concentration of the sludge stream or S_c (Kg/m^3) is determined by the efficiency of the sludge separation process and the amount of flushing or washdown waters (Q_s in m^3/day) required for the sludge removal.

$$S_c = S_p/Q_s \qquad (5)$$

Equations 2 through 5 can be used to estimate sludge production from a proposed recirculating configuration, permitting concurrent design of disposal treatment processes. Integrated design allows for overall minimization of treatment costs. Consideration given to the partitioning of the sludge stabilization burden between internal and discharge treatment processes can reduce the potential for conflicts with environmental regulatory agencies while enhancing production methodologies. Assuming a feeding rate of 2% of body weight per day, Table 2 illustrates that a recirculating aquacultural system for catfish and trout generates sludge volumes higher than other commercially cultured animals on a live weight basis.

Sludge Composition

Aquacultural sludge is characterized by the ratios of BOD_5/TSS and TKN/TSS. The BOD_5/TSS ratio is a measure of the degree of stabilization of the sludge. The sludge productions in Table 2 were calculated based on the excretion data in Table 1 and Equations 2 through 5. Calibration of Equations 2 through 5 against an experimental system in the authors' laboratory resulted in $K_d = 0.36\ day^{-1}$ and $h_s = 0.35$. The obtained K_d value is within the range of municipal waste (0.28 - 0.71 day^{-1}, Reynolds 1982). High BOD_5/TSS ratio's imply a sludge that will rapidly decay, potentially causing oxygen depletion and odor problems if it is not properly handled. The particulate BOD_5 excreted from fish is partially treated in a recirculating system by a biological filter before being discharged as sludge; therefore, the BOD_5/TSS ratio in sludge is less than that of the excreted matter. BOD transformations in a recirculating catfish system are illustrated in Figure 1, indicating that particulate BOD is the dominant form. Nitrogen production also results from fish feed. Most fish feeds contain 7.2 to 7.7% nitrogen by weight. Of the nitrogen in feed, 67 to 75% will be lost to the environment (Iwama 1991). The TKN excretion rate by fish averages approximately 5% of the feeding rate. Of the nitrogen excreted, approximately 20 to 50% is particulate. Nitrogen content is used to determine land application rates.

Results of analyses conducted in the authors' laboratory on sludge obtained from three systems using four different solids separation units are given in Table 3. These units included a pressurized sand filter, an upflow sand filter, a clarifier under a rotating biological contactor (RBC), and a low-density media filter. Samples were collected and analyzed for TSS, BOD_5, and TKN (APHA 1989). The results indicate that the BOD_5/TSS ratio of the sludge ranged from 0.09 to 0.20, while the TKN/TSS ratio ranged from 0.038 to 0.061. The variance in BOD_5 is due to the sludge age. Filters with high solids retention produce a more stabilized sludge, lowering the BOD_5/TSS ratio. The measured TKN/TSS ratios are consistent with Kugelman and Van Gorder (1991) who reported TKN/TSS ratio ranging from 0.058 to 0.063 and Olson (1991) who reported a range of 0.035 to 0.04.

Sludge Treatment and Disposal

Characterization of the sludge produced from a facility with respect to its (1) mass, (2) concentration, and (3) degree of stabilization facilitates the selection of a rational treatment scheme (Figure 2). In virtually all applications, sludge concentrations must be raised to a level of 2 to 5% by clarification within the recirculating system or as a primary treatment process to improve the economics of treatment and disposal. Direct land application of concentrated sludges is the primary management approach employed by facilities currently

addressing the issue (Mudrak 1981; Olson 1991; MacMillan 1991). However, this approach is sensitive to (1) land availability, (2) climatic conditions, and (3) environmental sensitivity in the disposal area, compatible with rural conditions, particularly in dry climates, land application is less suitable for urban applications or wet climates which can encourage odor problems. Thus, further reduction in sludge volume and stabilization (reduction in BOD_5/TSS ratio) will be desirable in many applications. Although anaerobic digestion has been examined (Kugelman and Van Gorder 1991), facultative or aerated lagoons can cost-effectively provide additional sludge treatment while facilitating intermittent (seasonal) sludge disposal in wet climates. Sludge concentrations vary widely for different sludge separation processes (Table 4). Additionally, sludge concentrations are impacted significantly by management practices, principally through water use. Recognition of the factors controlling sludge production (S_p) and sludge concentrations (S_c) as delineated in Equations 2 through 5 permits site specific estimations, allowing sizing of treatment units.

Clarification

The size of the clarifier can be determined by two methods: (1) hydraulic loading and (2) solids loading. Assuming the design hydraulic loading for the clarifier is 16.3 m^3/m^2-day (EPA 1975), the clarifier surface area needed per 1000 kg of catfish can be calculated as 0.004 to 0.04 m^2 according to the data given in Table 2. The solids loading design criteria is 72 kg/m^2-day (Metcalf and Eddy 1979), which projects a clarifier surface area of 0.054 to 0.088 m^2 per 1000 kg of catfish. Clarifier sizing is always based on the larger surface area value of the two design criteria. It is expected after clarification that the sludge will have a TSS concentration of 3 to 7% or greater, and a BOD_5/TSS ratio of less than 0.2. The overflow from the secondary clarifier will have a TSS concentration range of 10 to 30 mg/l (EPA 1975). The water overflow can be sent directly to the polishing process for further treatment if required by discharge permits.

Sludge Stabilization

Stabilization processes can reduce sludge volumes by 50 - 75% (Reynolds 1982), and provide for complete oxidation of readily degraded organics, resulting in a sludge that is unoffensive in nature. Stabilized sludges pose little problems when disposed through land application or landfilling. Several recognized methods are available to provide stabilization; their advantages and disadvantages are listed in Table 5.

Anaerobic digestion of sludges is widely used to treat municipal wastes in urban areas. The advantages of anaerobic treatment include methane generation and pathogen reduction. Digester operating complexity and cost will limit the application of anaerobic digesters in aquacultural systems. Experiments have been conducted to investigate the feasibility of using fish waste for methane generation through anaerobic digestion (Kugelman and Van Gorder 1991). Methane production ranged from 32% to 71% of the theoretical yield. The two main problems encountered were the long retention time required and free ammonia inhibition. Long retention times lead to larger digester volumes, while the ammonia inhibition requires the sludge to be diluted making the reactor even larger. Unless the digester can be designed very efficiently, and the operators are reasonably experienced in anaerobic operations, this process will not be cost effective. Therefore, anaerobic digestion is not recommended here as a viable option except in special cases.

Aerobic digestion is an alternative process for stabilizing organic sludges used primarily in smallercommercial and public waste water treatment operations (Metcalf and Eddy 1979; Reynolds 1982). Advantages of aerobic digestion include lower effluent BOD concentrations, production of a biologically stable sludge, simple operation, and lower capital costs. The disadvantages are principally the cost of aeration and the lack of a usable by-product (i.e. methane). The design criterion for an aerobic digester is based on the hydraulic retention time and the volatile solids loading rate. The hydraulic retention time for sludge at 20° C is typically 15 days, while the solids loading is 1.6 - 4.8 kg VSS/m^3-d (Metcalf and Eddy 1979). For the sludges described in Table 2, this translates to a volume range of 0.69 m^3 to 3.38 m^3 per 1000 kg of fish. The required volume decreases as the sludge concentration increases, as long as the VSS solids loading criteria is not exceeded. The air required to aerate a digester tank is 20-40 m^3/1000 m^3 digester volume per minute (Metcalf and Eddy 1979). Aerobic digesters are appropriate for use in urban areas. Aeration costs are offset by minimal space demands and odor elimination.

Anaerobic lagoons have been used to treat waste discharges from all phases of the vast agricultural industry (Middlebrooks et al. 1982), and have been considered as a suitable treatment process for manure wastes. Sludge introduced into the lagoon ranges from that containing relatively light solids concentrations (approximately 0.1% solids) to slurries containing just enough water to transport the solids into the lagoon. Anaerobic lagoons function successfully over a wide solids loading range with little maintenance. The major parameters used for anaerobic lagoon design are volatile suspended solids (VSS) or BOD_5 loading. Design criteria are highly variable. Suggested design VSS loadings for poultry manure lagoons range from 0.064 to 0.161 kg VSS/m^3/day (4 to 10 lb VSS/1000 ft^3/day). The BOD_5 loadings range from 225 to 625 kg/ha-day (200 to 1000 lbs BOD_5/acre-day). According to the sludge production rate in Table 2, the volume needed for aquacultural waste will be approximately 20.5 to 84.4 m^3/1000 kg of fish based on the VSS loading criteria. These volumes translate into a pond surface area of 8.2 to 33.8 m^2/1000 kg of fish, assuming a 2.5 meter pond depth. On a BOD_5 loading basis, the lagoon surface area ranges from 17.6 to 133.3 m^2/

1000 kg fish (Poon et al. 1986). Common problems include odor, temperature requirements, and long detention times, making anaerobic lagoons unsuitable for populated areas. It is recommended that an anaerobic lagoon should be located at least 0.8 kilometer (0.5 miles) away from neighboring residences or other sensitive locations (Overcash et al. 1983b). Anaerobic lagoons are recommended for use in the rural areas where land availability and odor generation are not issues and direct land application is not feasible.

Aerated lagoon systems are similar in performance and design to anaerobic lagoons except for the aeration process and lagoon depth. The aerobic lagoons are shallower than the anaerobic lagoons and are more expensive to operate. If land is available and odor generation is not an issue, anaerobic lagoons are recommended over aerobic lagoons because of the associated aeration costs.

Although digester/lagoon sludge stabilization processes are effective in BOD reduction, the suspended solids concentrations that meet secondary treatment level effluent quality may not be achieved due to solids and/or algae production. Algae removal will be required in order to upgrade lagoon effluents (Middlebrooks et al. 1982), while TSS removal is necessary for polishing aerobic digester effluents. Many processes can be used for effluent polishing, including constructed wetlands, sand filtration, land treatment, and microscreens (Poon et al. 1986).

Another stabilization process is composting in which organic material undergoes biological degradation to a stable end product. Composting can reduce waste volume by 50 to 85% and the properly composted sludge is an essentially pasteurized, nuisance-free, humus like material (Metcalf and Eddy 1979). This product can be marketed for use as a soil conditioner. One problem of composting aquacultural sludge is its high water content. A preparation process is needed to reduce the water content from above 90% to less than 70%. This requirement may limit the application of composting to aquacultural sludge. One solution is to use co-composting with other solid waste.

Sludge Disposal

After thickening in the clarifier, the sludge can be directly applied to land, provided land is available. High rate land application of animal manure as a waste has been proven to cause adverse environmental impacts (Overcash et al. 1983b). A better approach for animal sludge management is the utilization of the waste's fertilizer value. The high nitrogen content (4 to 6%) makes aquacultural waste valuable to crops as a fertilizer (Mudrak 1981; Willett and Jakobsen 1986; Olson 1991). Limitations of such application have also been identified (Olson 1991). The first is odor, prohibiting this option in populated areas. The second is the propensity for the applied sludge to form a crust. If the sludge is not thoroughly plowed into the soil, some plant seedlings may be unable to push through the crust. The third limitation is the expense of hauling and spreading (MacMillan 1991). The fourth is the slow nitrogen release rate. About 90% of the total nitrogen is in the organic form; consequently, only one third of the nutrients can be utilized in the first year. This makes application in high rainfall areas questionable since runoff of the unutilized nitrogen may cause problems in local surface waters.

The guidelines for application rates of aquacultural sludge on cropland have not yet been established. Studies on poultry manure indicate that crops typically remove less than 222 kg of nitrogen per hectare (Overcash et al. 1983b). Therefore, a similar application rate may eliminate potential nitrogen accumulation that would adversely impact the environment. Olson (1991) tested three application rates of trout manure in a greenhouse (111, 222, 336 kg-N/hectare). Satisfactory results were obtained from the application rate of both 222 and 336 kg-N/ hectare. Subject to further experimental verifications, 222 kg-N/hectare is recommended by Olson (1991) as a design criterion for aquacultural sludge application on land. According to this rate, the land area needed is approximately 9-144 m^2 per 1000 kg of fish based on the nitrogen concentration in the sludge.

Besides direct land application, a sanitary landfill can be used for disposal of stabilized sludge from aerobic or anaerobic digestion processes. The sanitary landfill method is most suitable if it is also used for disposal of the other solids wastes in addition to sludge. The advantages of landfill include low cost, flexibility in operation, and the possibility of land reclamation. The main disadvantages of sanitary landfills are land requirements and possible contamination of ground water by the leachate from the landfill site. Another problem in using landfills for aquacultural sludge disposal is that the stabilized sludge needs to be dewatered to reduce water content.

Summary

Integrated design of recirculating and discharge treatment processes can eliminate potential environmental impacts of sludges generated from large scale aquaculture production systems. Dilute sludges produced by backwashing or washdown operations should be concentrated by clarification processes prior to stabilization or disposal. Both aerobic and anaerobic processes with extensive track records are available to reduce the easily biodegradable portion of the sludge, minimizing the volume of sludge for final disposal. Sludge disposal through land application appears feasible for rural areas; whereas, landfilling of stabilized sludges may be most appropriate for urban areas.

References

APHA. 1989. Standard Methods for the Examination of Water and Wastewater, 17th edition, 1268 pp.

EPA. 1975. Process Design Manual for Suspended Solids Removal, U. S. EPA Technology Transfer Publication.

Gordon, J. A. 1974. Water Characterization Studies for High-Density Raceway Production of Catfish -The Galatin Project, Proceedings of the 29th Purdue Industrial Waste Conference, Lewis Publishers, Chelsea, Michigan.

Hopkins, T. A. and W. E. Manci. 1989. Feed Conversion, Waste and Sustainable Aquaculture, the Fate of the Feed, Aquaculture Magazine, 15(2):30, 32-36.

Iwama, G. K. 1991. Interactions Between Aquaculture and the Environment, Critical Reviews in Environmental Control, 21(2):177-216.

Kugelman, I. J. and S. Van Gorder. 1991. Water and Energy Recycling in Closed Aquacultural Systems, Engineering Aspects of Intensive Aquaculture, Proceedings from the Aquaculture Symposium, Cornell University, Ithaca, New York, April 4-6, 1991, Northeast Regional Agricultural Engineering Service, NRAES-49.

Liao, P. B. and R. D. Mayo. 1974. Intensified Fish Culture Combining Water Reconditioning with Pollution Abatement, Aquaculture, 3:61-85.

MacMillan, R. 1991. Economic Implications of Water Quality Management for a Commercial Trout Farm, Paper presented at National Livestock, Poultry and Aquaculture Waste Management Workshop, July 29-31, 1991, Kansas City, Missouri.

Malone, R. F. and D. G. Burden. 1988. Design of Recirculating Soft Crawfish Shedding Systems, Louisiana Sea Grant Publication, Louisiana State University, Baton Rouge, Louisiana.

Middlebrooks, E. J., C. H. Middlebrooks, J. H. Reynolds, G. Z. Watters, S. C. Reed and D. B. George. 1982. Wastewater Stabilization Lagoon Design, Performance and Upgrading, Macmillan Publishing Co., Inc., New York, 356 pp.

Murphy, J. P. and R. I. Lipper. 1970. BOD Production of Channel Catfish, The Progressive Fish-Culturist, 32:195-198.

Metcalf & Eddy, Inc. 1979. Wastewater Engineering, Treatment/Disposal/Reuse, Second Edition, McGraw Hill, 920 pp.

Mudrak, V. A. 1981. Guidelines for Economical Commercial Fish Hatchery Wastewater Treatment Systems, Allen, J. L. and E. C. Kinney (editors), Bio-engineering Symposium for Fish Culture, American Fisheries Society.

Olson, G. L. 1991. The Use of Trout Manure as a Fertilizer for Idaho Crops, Paper presented at National Livestock, Poultry and Aquaculture Waste Management Workshop, July 29-31, 1991, Kansas City, Missouri.

Overcash, M. R., F. J. Humenik and J. R. Miner. 1983a. Livestock Waste Management, Volume I, CRC Press, Inc., 255 pp.

Overcash, M. R., F. J. Humenik and J. R. Miner. 1983b. Livestock Waste Management, Volume II, CRC Press, Inc., 244 pp.

Page, J. W. and J. W. Andrews. 1974. Chemical Composition of Effluents from High Density Culture of Channel Catfish, Water, Air, and Soil Pollution, 3:365.

Poon, C. P. C., L. K. Wang and M. H. S. Wang. 1986. Waste Stabilization Pond and Lagoon, Wang, L.K and Pereire, N. C. (editors) In Handbook of Environmental Engineering, Vol.3, The Humana Press.

Reynolds, T. D. 1982. Unit Operations and Processes in Environmental Engineering, Brooks/Cole Engineering Division, Monterey, California, 576 pp.

Ruane, R. J., T. Y. J. Chu and V. E. Vandergriff. 1977. Characterization and Treatment of Waste Discharged from High-Density Catfish Cultures, Water Research, 11:789-800.

Speece, R. E. 1973. Trout Metabolism Characteristics and the Rational Design of Nitrification Facilities for Water Reuse in Hatcheries, Transactions of American Fisheries Society, 102(2):323-334.

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

Willett, L. R. and Jakobsen, P. 1986. Fertilizing Properties of Trout Farm Waste, Agricultural Wastes, 17:7-13.

Wimberly, D. M. 1990. Development and Evaluation of a Low-Density Media Biofiltration Unit for Use in Recirculating Fish Culture Systems, Master's Thesis, Louisiana State University, Baton Rouge, Louisiana.

Table 1. Fish excretion (kg/kg-feed).

Species	Excretion Products BOD	TSS	Reference
Trout	0.4 - 0.60[1]	0.4 - 0.52	Speece 1973; Liao and Mayo 1974
Channel Catfish	0.083 - 0.113[2]	0.08 - 0.28	Page and Andrews 1973; Gordon 1974
	0.22[2]	0.43	Wimberly 1990

(1) BOD_{20}
(2) BOD_5

Table 2. Waste generation (/day/100kg-LW) comparison between catfish and other commercial animals.

Animals	BOD (kg)	TSS (kg)	TKN (kg)	Sludge Volume (liter)	Reference
Fish [1]	1.1-3	3.9-6.3	0.2-0.32	65-630[2]	Present study[3]
Beef Cattle	1.6	9.5	0.32	30	Middlebrooks et al. 1982; Overcash et al. 1983a
Dairy Cows	1.4	7.9	0.51	51	
Poultry	3.4	14	0.74	37	
Swine	3.1	8.9	0.51	76	

(1) Calculated at the feeding rate of 2% body weight per day
(2) Calculated based on 1-6% TSS concentration in sludge
(3) Calculated according to the excretion value in Table 1, and assuming $K_1 = 0.36$, $h_s = 0.35$.

Table 3. The ratios of BOD_5 and total nitrogen to TSS in aquacultural sludge from different recirculating systems.

System	BOD_5/TSS	TKN/TSS	Animals
Pressure sand filter	0.20	0.053	Sturgeon (Acipenser transmontanus)
Upflow sand filter	0.17	0.038	Snapping turtle (Chelydra sertentina)
Propeller washed filter	0.09	0.049	Red swamp crawfish (Procambarus clarkii)
RBC/clarifier	0.09	0.061	Sturgeon (Ancipenser transmontanus)

Table 4. Total suspended solids concentrations in sludge generated by three typical solids removal processes.

Techniques	TSS Concentration in Sludge	Reference
Upflow sand filter	0.005 - 0.015%*	Malone and Burden 1988
Primary sedimentation	1 - 6%	Kugelman and Van Gorder 1991; present study
Low-density media filter	0.05 - 0.5%	Present study
Sand filtration	0.01 - 0.02%	Metcalf and Eddy 1979

* Calculated for 1 - 3 minute backwashing time

Table 5. Features of Sludge Stabilization Options.

Option	Advantages	Disadvantages
1. Anaerobic Lagoon	High organic loading Low maintenance	Odor
2. Aerated Lagoon	Space efficient High organic loading	Energy consumption Moderate maintenance
3. Aerobic Digesters	High loading	Energy consumption
4. Anaerobic Digesters	High loading Methane generation	Complex High maintenance
5. Composting	Useful end product	Dewatering required

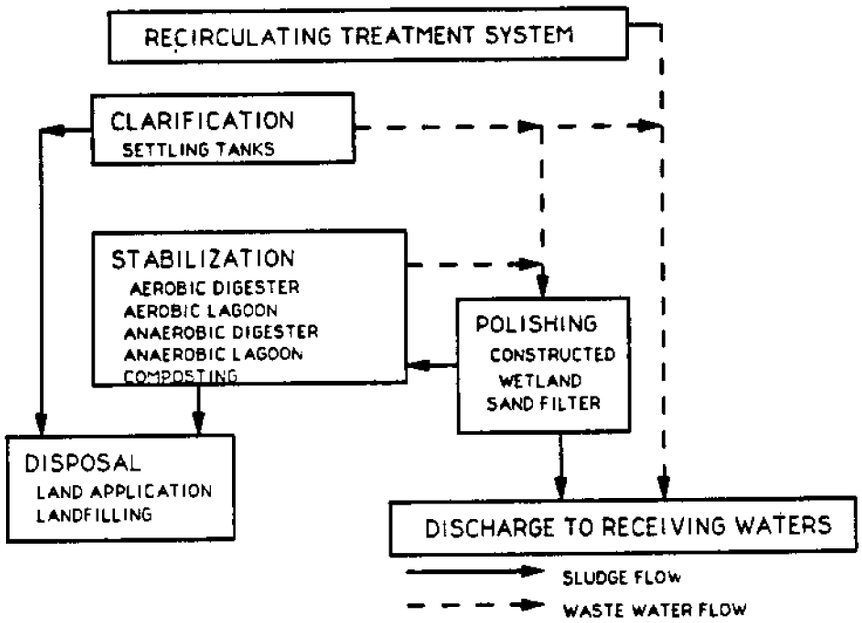

Figure 1. Approximately twenty five percent of the feed's BOD is discharged with the sludge.

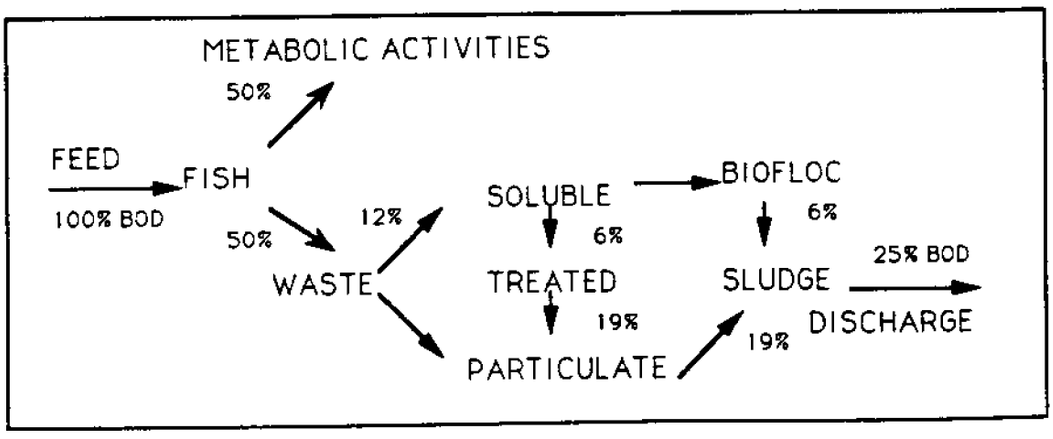

Figure 2. Sludge treatment consists of clarification, stabilization, and disposal.

Integrated Aquaculture System Design

Fred Wheaton
The University of Maryland
College Park, Maryland

Abstract

Integrated system design is the process of fitting unit processes together to form an effective and efficient aquaculture production system. Mass and energy flows and maintenance of chemical concentrations are principles used in the integration process. Well-known engineering and mathematical principles and techniques are available to assist the designer with the integration process. However, the talent of the designer will also influence how effectively the integration process is carried out.

Introduction

Any aquacultural production system is composed of a number of specific processes that carry out a single function. Examples of these simple purpose processes include pumping, filtering, settling, ammonia removal, etc. Such processes are called unit processes. Unit processes often are designed individually and then combined to form an aquaculture production system. In fact, this is probably the method used to "design" most commercial aquaculture production systems today. The quotation marks around the word design must be emphasized, because in my opinion "designing" in such a manner is not designing at all, but is the pasting or stringing together of unit processes. If systems "designed" by these methods work at all, it is due more to accident than the efforts of the "designer."

Integrated design is the intentional process by which a designer fits unit processes together to form a coherent, integrated system. Such an integrated system should not only accomplish the desired objective, but should also require the unit processes to function together in a smooth and efficient manner. The final product of integrated design will be a smoothly functioning arrangement of unit processes that will produce the desired output and do so in an efficient manner.

Unit Processes

Integrated design can not be discussed unless unit processes are understood. The concept of unit processes is widely used in the chemical and food industries. Processing of a food is typically a series of chemical, physical and biological processes that transform a raw product (e.g., tomatoes) into a consumer product (e.g., ketchup). Each of these steps or processes is called a unit process or unit operation (Geankoplis 1983). Similarly, intensive aquaculture production systems are comprised of unit processes or unit operations that are linked together to form an overall system. Examples of aquaculture unit operations include biofiltration (ammonia and nitrite removal), settling basins (solids removal), culture tanks (animal or plant growth), pumps (water movement), and many others.

Each unit process has an input (at least one) and an output (at least one). Between the input and output some change takes place in the material moving through the unit process. For example, water goes from low to high pressure as it passes through a pump, while ammonia is removed from water as it passes through a biofilter. Each of the unit processes can be designed as a separate entity using the input, desired output and the other design parameters of the unit process. However, if unit processes are to be a part of a system, designing each of them as an independent entity can lead to difficulties when the unit processes are combined to form the over all system.

Integrating Unit Processes

Unit processes are often integrated based on mass and/or energy flow requirements of the various unit processes. Because water flow is so important in aquaculture systems, mass flow of water is often used as a primary integrating process. However, intensive systems also have flows of solids and gases. Solid flowsconsist of feed, non-ingested feed, feces and other particulates, the nature of which vary depending on the type of system used. Gas flows include at least oxygen, carbon dioxide, nitrogen, and perhaps ozone. Energy demands of intensive aquaculture production systems include energy for pumping, heating and/or cooling, disinfection, waste disposal, and fish handling. Energy is usually supplied by electricity, natural gas, and/or fuel oil. Energy for the fish growth is supplied by feed.

Intensive aquaculture systems also require that the concentrations of several chemicals be maintained within acceptable limits. The concentrations of nitrogen, phosphorous, hydrogen (pH), sulfur, carbon (as CO_2, HCO_2, etc.), and several other chemicals are critical to survival of the aquatic crop. The unit processes selected

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for a system are often determined by the chemical concentrations that must be controlled (e.g., a biofilter will be selected to control ammonia concentration).

The unit operations selected for a specific application depend on the type of system, the crop to be raised, the water quality needs of the organisms, economic considerations, and other factors. Once the unit processes are selected, fitting them together into a system involves mass and energy flows and chemical concentration considerations. Unit process sizing is also a function of these same mass, energy, and chemical concentration considerations.

Example Systems

One of the easiest ways to approach understanding of system integration is to look at an example. A relatively simple example provides all of the principles while simplifying the discussion. The aquaculture production system considered here is one consisting of a culture tank, a settling basin, a biofilter, an aeration system, and a pump (or pumps) to circulate the water. Figure 1 shows one configuration, a series arrangement, of the system. All of the water is recirculated and all of the water must pass through each of the components in series. Intentional input of gases occurs only in the aeration device. However, some natural occurring gas flows (such as carbon dioxide and oxygen exchange) take place at the air-water interface. Solids flow is into (feed) and out of (waste) the culture tank, and out of the settling basin and biofilter. The crop in the culture tank and some solids in the settling basin are in temporary storage within the system.

Figure 2 shows a parallel arrangement, in terms of water flow, of the unit processes. Water flows separately from the culture tank to each of the filtration and aeration units and back to the culture tank. This parallel flow requires more pumps than does the series arrangement, but it eliminates the flow dependence of one component on other components. The parallel system increases design flexibility over the series system because, except for the culture tank, each unit process operates independently. The gas flow is a factor only in the aeration unit process. Solids exit from the culture tank, biofilter, and settling basin. Solids leaving the culture tank are assumed to be in suspension in the water and, thus, are not shown as a separate flow. The biofilter solids flow is independent of the settling basin solids flow, but the biofilter solids flow does impact the settling basin solids flow.

The systems shown in Figures 1 and 2 are at opposite ends of the design spectrum in terms of water flow through the system (i.e. one is parallel and the other series flow). There are, however, many options in between. For example, Figure 3 shows a system having the same unit operations as shown in Figures 1 and 2, but it has a significantly different flow pathway. The settling basin in Figure 3 can be operated in either the continuous or intermittent mode, a factor that will greatly influence the basin design. Obviously, there are many different ways in which the same unit operations can be combined to form an aquaculture production system. The best arrangement of unit processes will depend on the objectives of a particular design.

Integrated design takes into account the arrangement of the unit processes as well as the operation of the individual unit processes. For example, compare the operating conditions of the biofilter in Figures 1 through 3. The biofilter in Figure 1 will probably have a higher flow rate than will the biofilter in Figure 2. However, in Figure 2 the flow through the biofilter will be determined solely by the needs of the biofilter while the flow rate through the biofilter in Figure 1 will be determined by the system component requiring the highest flow rate.

Assuming the culture tanks in Figures 1 and 3 have the same solids concentration, the biofilter in Figure 3 will experience a higher solids concentration than will the biofilter in Figure 1. In a heavily loaded system the oxygen concentration in the biofilter will be lower in Figure 1 than in Figure 2. Thus, despite the fact that all three systems have the same unit processes, the operating conditions for the same unit process are different in the three systems.

The point of this discussion is that designing unit processes for an aquaculture production system independent of the rest of the system is counter productive at best. Failure to integrate the unit processes into a coherent system will usually lead to system failure.

The Integration Process

The complexity of integrating unit processes into an overall system is primarily a function of the number of unit processes that must be considered. If working with something as complex as an automobile assembly plant, there are very sophisticated plant layout techniques, such as linear programming and Monte Carlo methods, that can be used (Moore 1962). However, most aquaculture production systems have a limited number of unit processes that must be integrated and sophisticated methods are usually not necessary. System complexity may influence the methods used, but not the basic approach to integrating the system.

The integration process is an integral part of the design process. Thus, the first step is selection of design parameters for the production system. The designer must then select unit processes to accomplish the desired functions. For example, solids may be removed by several methods, including screens, settling basins, centrifuges, hydroclones, etc. From these possible unit processes the designer must select the processes he/she believes will best fit into the overall system. Next, the unit processes are arranged in what

the designer believes is the best configuration for the system. At this point a preliminary detailed design of the unit processes is carried out. The mass and energy flows for the various unit processes are then determined based on the preliminary unit process designs. Knowing the configuration of unit processes that will be used in the system design and the needs of the individual unit processes, the designer must determine the acceptable flows that must pass through each unit process.

Once the water flow is determined, the other flows must also be determined in a similar manner. Limitations imposed by flow considerations usually require modifications of the preliminary unit process design. Such a design process usually evolves into an iterative process, and several iterations are usually required to achieve a design that meets all of the design specifications and is truly integrated. However, an integrated design produces a much better system than where integration is ignored.

How the actual integration process details are carried out depends on the type and function of the system, on the design objectives, and on the designer's approach to the design. Series systems, such as shown in Figure 1, require each unit process to accept the entire mass flow of water. Because there is no significant storage of water and circulation is continuous, the mass flow discharge of one unit process is the input of the next unit process. For such a system the primary integrating parameter is the water flow. Solids and gas flows are then usually adjusted by sizing of the hardware making up the various unit processes. For example, suppose the designer determines that for the system shown in Figure 1 the biofilter requires the greatest water flow rate of all of the unit processes. This will set the water flow rate through all of the unit processes. However, the settling basin needs low velocities and quiescent flow for good settling to occur. To achieve this the settling basin size and dimensions must be chosen to achieve the desired flow conditions while accepting the required flow that was set by considerations of the biofilter.

In parallel systems, such as that shown in Figure 2, water flow through each unit process can be adjusted independently. Thus, the integration process concentrates more on achieving the water quality desired and less on flow characteristics. It is, however, difficult to determine the input water quality for each unit process in the design stage due to the indirect interaction of the various unit processes.

Systems such as that shown in Figure 3 have other considerations in the integration process. If the settling basin is operated in an intermittent fashion (i.e. water is placed in the basin for a period of time for the solids to settle out and then is pumped out and a new batch of water is allowed to enter the tank), a temporary storage function enters the design. Flow through the settling basin is thus determined by what is needed to achieve the desired separation of solids. Flow through the rest of the system is then determined by the component requiring the most flow. Integration requires consideration of all of these factors and several others.

System integration is a process that is mostly analytical, but still requires some art. Its success depends mostly on the engineering and mathematical methods employed, but the talent of the designer also enters into how well the integration is carried out. System integration is critical to successful design and operation of aquaculture systems. It is also one of the most often overlooked principles of aquaculture system design. Thus, it is a frequent cause of aquaculture systems failure and it should not be, because most of the methodologies needed to successfully integrate a system are well known.

Conclusions

1. Integrated design requires not only the design of the several unit processes making up an aquaculture production system, but also the fitting together of the unit processes into an efficient and smoothly functioning unit.

2. Combining independently designed unit processes into a system without regard to the constraints imposed by the system will lead to an inefficient design and probably to total system failure.

3. When aquaculture production systems are constructed with little or no design occurring, system constraints usually are neglected.

4. Integration of system components usually revolves around maintenance of mass and energy balances throughout the system. Chemical concentrations must also be considered and provisions made for their control in the design process.

References

Geankoplis, Christie J. 1983. Transport and Unit Operations, Allyn and Bacon, Boston, Massachusetts.

Moore, James M. 1962. Plant Layout and Design, The Macmillan Company, New York, New York.

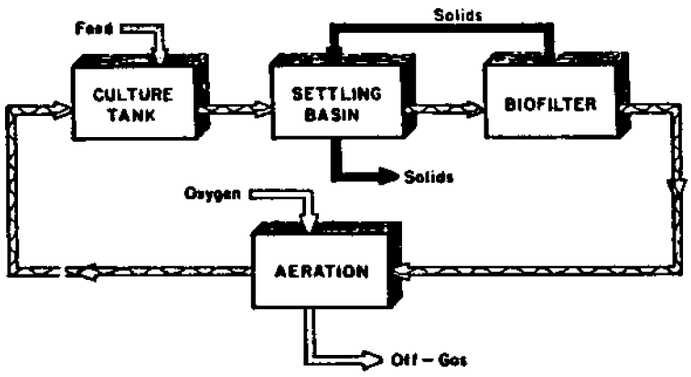

Figure 1. Series configuration of an aquaculture system.

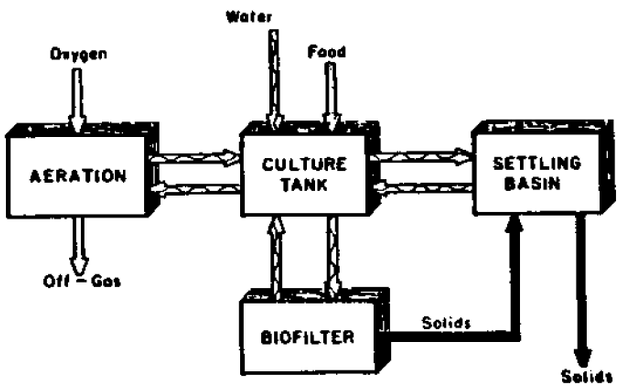

Figure 2. Parallel arrangement of aquaculture unit processes.

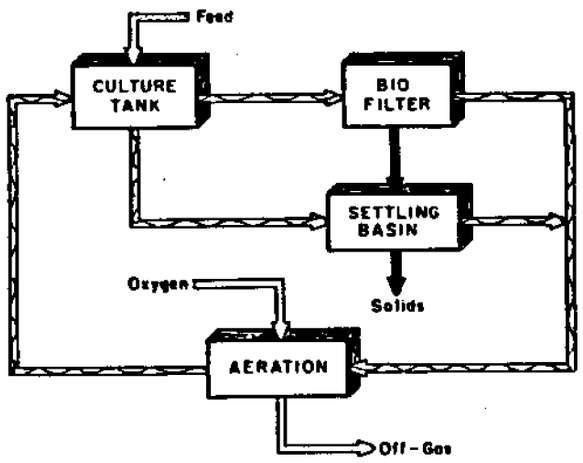

Figure 3. Parallel/series flow system.

Fish Health Management in Recirculating Systems

Ronald L. Thune
Louisiana State University
Baton Rouge, Louisiana

Thomas E. Schwedler
Clemson University
Clemson, South Carolina

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Abstract

Recirculating systems are particularly prone to disease problems due to the high densities and eutrophic conditions, yet the impact of diseases on fish in recirculating systems has largely been ignored both in terms of facility design and in the development of production protocols. Disease management in high-density, recirculating, aquaculture systems must emphasize avoidance of obligate pathogens and stress reduction practices to control facultative pathogens. Managers of recirculating systems must avoid the introduction of obligate pathogens by inspecting all new fish stocks and by implementing quarantine, batch production, and hygienic culture practices. System design should emphasize the maintainance of optimal water quality and minimal handling to reduce stress. A nutritionally balanced diet will maximize fish health and should be in a processed form to avoid the introduction of pathogens in live or unprocessed diets. Ultraviolet light or ozone can be used to reduce the numbers of facultative organisms in the circulating water, and a fish health monitoring system should be implemented that includes routine health checks and necropsy of any mortalities that occur. Frequent outbreaks of facultative pathogens indicate a need to reduce stress. Production practices that ensure that fish are growing at optimal rates will reduce time in the culture system, reduce the risk of disease outbreaks, and maximize the economic potential of the system.

Introduction

Recent advances in recirculating aquaculture production systems have resulted in increased interest in these systems for the commercial production of aquatic animals. Despite these advances, the economic viability of these systems remains to be proven. Diseases have a significant economic impact on most aquaculture production systems, including ponds, cages, and raceways. Recirculating systems are particularly prone to disease problems due to the high densities and eutrophic conditions, yet the impact of diseases on fish in recirculating systems has largely been ignored both in terms of facility design and in the development of production protocols.

A potential advantage of a properly designed closed, recirculating aquaculture system is the ability to limit serious disease problems by excluding obligate pathogens. On the other hand, a significant disadvantage of the same system is that the high densities and inherent stress associated with production at high density are conducive to the rapid spread of disease once established. Thus, prevention and stress management must be the cornerstone of disease management in any closed aquaculture system. In this discussion of fish health management in recirculating systems, the emphasis is on two levels of preventation involving system design and production protocols. The first and most important component of this fish health management plan will be the exclusion and avoidance of obligate pathogens. Should the culturist be successful in the first step, the second level of management involves careful monitoring of the system and the fish in order to minimize stress and control the incidence and spread of facultative pathogens that are almost impossible to exclude from aquatic systems. Should the culturist be successful at disease prevention, disease treatment, currently the primary method of disease management in these systems, will be minimized and utilized only as a last option.

Avoidance of Obligate Pathogens

Obligate pathogens are defined as those that require a host in order to replicate and are not capable of surviving for long periods outside of the host. Facultative or opportunistic pathogens, on the other hand, are organisms that do not require a host to replicate. They are often found as normal flora of the host or are free-living in the environment and cause disease only when the host resistance is lowered. Given the same environmental conditions, obligate pathogens are more serious than facultative pathogens and can cause significant mortality, even when water quality conditions are good. Thus, the most effective way to manage obligate pathogens is to avoid them. Due to the high densities and inherent stress asso-

ciated with closed system aquaculture, avoidance is the only practical method of management that is economically feasible. Failure to prevent the introduction of severe pathogenic organisms diseases is catastrophic in closed, high-density systems.

The avoidance of obligate pathogens requires a multifaceted approach that includes at least seven different components of design and production. These components include: (1) the design of facilities for batch production of fish; (2) the use of disease free stocks; (3) the use of pathogen free water; (4) the use of pathogen free diets; (5) the establishment of strict quarantine procedures; (6) the routine use of hygiene and disinfection procedures; (7) the establishment of a regular health monitoring program.

System Design and Operation

Disease management can be facilitated with appropriate facility design. If the species to be cultured requires frequent introduction of new stocks from external systems, quarantine facilities are essential. Effective quarantine requires a set of tanks for use as quarantine and/or nursery units. Each batch of new stockers brought into the facility should be stocked into these quarantine/nursery tanks at a moderate density and care taken to prevent cross contamination with other culture systems. Fish are grown until they are subdivided for further grow-on by removing half to a second tank. As fish grow and are subsequently split and expanded into additional tanks, care should be taken to keep individual groups of fish separate until harvest.

An obvious consideration in following this procedure in recirculating systems is the establishment and maintenance of the biological filter. The most effective method is to introduce newstockers into tanks that have recently been harvested, provided that no significant disease problems were encountered during the production of the recently harvested fish. If a previous disease problem necessitates disinfection of the system, or if the fish are being stocked into a new system, new stockers should either be stocked at low densities to allow the filter to establish as the fish grow or partial water exchanges should be made daily until the filters are established.

Disease Free Stocks

Possibly the most important decision made in the planning of a closed system production facility is the selection of the fish species to be cultured. Generally, this decision is based on biological production parameters such as reproductive behavior and capacity, growth rates, and hardiness, as well as economic parameters such as cost of production and demand for the product. An additional important consideration should be the disease susceptibility of the species, with preference given to species that are not prone to serious disease problems. For example, the channel catfish is host to two important and potentially devastating obligate pathogens, the bacterium Edwardsiella ictaluri and the channel catfish herpesvirus. Obtaining stocks of fish free of these agents should be an important consideration in a decision to produce channel catfish in closed systems. On the other hand, tilapia have no significant obligate pathogens as yet identified that are associated with their culture, and thus may be a more appropriate choice for closed system aquaculture.

Another culture parameter that has disease control implications is the reproductive behavior and capacity of the species in question. A species that is capable of producing stockers on site in a closed system allows the culturist to close the loop to the outside and maintain the pathogen-free status of a facility. If a species requires pond spawning or pond rearing to stocker size, a potential for the introduction of pathogens exists. Each time a new population of stockers is brought into the facility the danger of pathogen introduction exists. Although production practices such as quarantine minimize the risk of the entire facility to pathogen introduction, those practices have a cost associated with them that should be factored into the decision to grow a particular species.

Pathogen-Free Water and Diets

Although fish themselves are the most common source of infectious agents, a water source that is free of pathogens is also essential to the maintenance of a specific pathogen free facility. Such sources include well water and dechlorinated municipal water. Wells are expensive to drill and municipal sources require the installation and maintenance of a water dechlorinating system and the purchase of water from the municipal supplier. Surface waters from lakes and rivers frequently harbor infectious and parasitic agents and should be avoided unless a treatment system is included to render them pathogen free. Efficient systems are available for this purpose that make use of sand filters, ultraviolet light, ozone, and chlorination/ dechlorination systems in various combinations, depending on the situation. Careful comparison of the cost and efficiency of any system for providing pathogen free water is an important component of facility design for recirculating systems. The most desirable water source would be good quality well water, but in some situations either drilling costs or lack of ground water availability may make the other alternatives more cost effective.

Diets used in closed system aquaculture are either artificial ones containing substantial animal protein, usually fish meal, or natural foods, such as algae, brine shrimp, rotifers, or other live feeds, especially for larval stages. If artificial diets are not properly treated, i. e., pasteurization, a potential exists for the introduction of viral, bacterial, and parasitic pathogens. Natural diets produced in culture for larval stages have also been implicated in the transmission of some viral and bacterial pathogens.

Quarantine

In facilities where new stockers are to be brought in from external pond sites, all incoming fish should be quarantined from resident stocks. Each separate lot of quarantined fish should subsequently be raised as a distinct group or groups. Consequently, quarantine facilities must be large enough to accomodate each individual shipment of stockers for a minimum of 4 weeks. Quarantine facilities should be considered the first stage of the production cycle for each new lot of fish and should thus be stocked at a density that will require about a 1 to 2 split at the end of the 4 week quarantine period. With this strategy there would be no temptation to end the quarantine period short and place other stocks of fish at risk. Careful facilitydesign and production planning are necessary to ensure that the initial quarantine production unit and the grow-out facility are proportionately designed for maximum production efficiency. The relative size of the quarantine and production facility would be dependent on the production characteristics of the species to be raised.

Ideally, the quarantine facility should be primarily flow-through initially, with a gradually increasing percentage of recirculation. However, in practice, it is difficult to re-establish biological filters for each new batch of fish. If the previous batch of fish went through the quarantine process with no major disease problems, it may be appropriate to stock a new batch after cleaning and disinfection of the tanks. However, either U.V. or ozone sterilization in line between the biological filter and the production tank(s) is recommended in this situation.

Each lot of fish brought into quarantine should be carefully examined for disease, preferably prior to shipping and again on arrival. For some species of fish, specific problematic diseases may need to be screened according to American Fisheries Society-Fish Health Section guidelines in order to ensure minimal risk to that disease. Prophylactic treatment for protozoan parasites and columnaris should be applied after 24-48 hours in the quarantine tanks, and fish should be carefully monitored for disease throughout the quarantine process. All dead fish should be carefully necropsied and a final screening of a random sample of fish should be conducted prior to expanding them into production tanks.

Hygiene and Disinfection

A critical component of any preventative medicine program is the establishment of strict hygiene and sanitation procedures. All other preventative measures, i. e., quarantine and batch production, are negated if the same nets and equipment are used in all production systems, or if workers do not disinfect their hands between water sampling in separate systems. Special care must be taken to ensure that pathogens are not transmitted from one system to another due to the movement of equipment (nets, brushes, buckets, etc.) or personnel between systems. Disinfection of both equipment and personnel between systems is essential. For routine equipment, such as nets, it is advisablefor each system to have one dedicated for use in that system only. This is especially important for quarantine facilities.

Sanitization of rearing systems between production batches can be effectively accomplished with a variety of disinfectants. However, this approach may have a more limited role in closed aquaculture systems than in open systems, because the disinfectants would also destroy the microbes that are essential to the biological removal of nitrogenous waste. Thus, complete disinfection of a closed production system would necessitate the re-establishment of the biological filter and severely reduce the capacity of that system. However, disinfection is recommended in situations where significant disease problems were encountered in the production of the previous batch.

Health Monitoring

Fish brought into quarantine should be carefully checked for infectious and parasitic agents prior to their introduction into the facility. Specific procedures for disease certification of fish stocks are available from the Fish Health Section of the American Fisheries Society. These procedures require relatively large samples to detect specific disease agents at low levels of incidence and are primarily applicable when serious obligate pathogens are of concern. Routine application of these procedures on small lots of fish may not be practical. However, certification of broodstock is recommended for facilities where stockers are to be produced on site, particularly if serious obligate pathogens are known for that species. Where the frequent introduction of stockers from pond systems is necessary, an alternative is to purchase them from hatcheries that have been certified specific pathogen free. If hatchery certification is not available, purchases should be made from a reputable supplier known to have healthy stocks of fish. In addition, samples of each lot of fish to be introduced to the facility should be necropsied by a qualified fish pathologist to determine the presence of overt disease or a high incidence of infectious or parasitic agents. The integration of a stocker production facility with the closed system production facility is the most reliable method of obtaining healthy, disease free stocks if stocker production on site is not practical.

In addition to evaluating the health status of incoming stockers, routine sampling of production lots should be conducted, especially during quarantine and early in the production cycle. In addition, allmortalities should be necropsied to determine the cause of death.

Water Quality Maintenance for Control of Facultative Pathogens

Should the culturist be successful in excluding

obligate pathogens from the culture system, management and careful monitoring of the system and the fish are required. The goal is to minimize stress and control the incidence and spread of facultative pathogens that are almost impossible to exclude from recirculating systems. The successful production of fish at the high densities required for economic success in recirculating systems is dependent on the management of a number of environmental parameters affecting respiration, osmoregulation, and metabolism. By design, fish in recirculating systems are confined in a limited space and at very high densities typically between 5 and 15 lbs of fish per cubic foot. In this limited water volume, adequate dissolved oxygen must be maintained and the metabolic waste generated by high fish biomass under intensive feeding regimes must be removed efficiently to prevent the accumulation of toxic metabolites. Heavy nutrient loading also can result in the accumulation of organic and inorganic material that favors bacterial growth, including those that are facultative fish pathogens. In addition to the potential for water quality deterioration inherent in recirculating systems, there is obvious potential for either partial or complete system failure. Consequently, the level of management necessary to produce fish in a recirculating system and to minimize the chance of having poor environmental conditions is greater than in less intensive culture systems. In addition, fish in recirculating systems are completely dependent on artificial diets, are dependent on artificial light for photoperiod cues, and are often exposed to increased human intervention and handling. Any of these adverse environmental factors, generally referred to as "stressors", can cause poor performance in cultured fish and can have dramatic adverse effects on fish health.

Stress

The term "stress" generally refers to environmental conditions, or stressors, which cause physiological changes that make maintenance of homeostasis more difficult. The numerous stressors that have been identified ascausing a stress response in fish can be roughly divided into three groups: water quality, behavioral, and physical stressors. Individual stressors in intensive aquaculture systems will be discussed in more detail below, but their importance to fish health management is related to the physiological response of fish to stressful conditions. Selye (Selye 1936, Selye 1946, Selye 1950) presented the idea that animals exhibit a generalized stress response to adverse environmental stimuli. He also noted that the morphological, biochemical, and physiological changes that occur are consistent regardless of the stressor, and he called this response the General Adaptation Syndrome (G.A.S.). The list of physiological changes associated with stress and the G.A.S. in fish is extensive and has been recently reviewed (Barton and Iwama 1991, Heath 1987, Wedemeyer, et al. 1976). Although the interrelationships between all of the parameters are not fully understood, several primary responses are manifested in fish during stressful conditions, including the release of stress hormones such as cortisol, the disruption of osmotic balance, and an increase in branchial blood flow, blood glucose levels, and nitrogen metabolism. These primary responses are easily induced and measured in fish exposed to a stressor, but their direct effect on disease resistance is not immediately apparent. There are, however, several documented responses that have a more direct cause and effect relationship in predisposing fish to pathogenic invaders, including decreased circulating lymphocytes, decreased inflammatory response, decreased interferon production, and reduced killing of bacteria by macrophages. These changes in the fishes' specific and non-specific defense mechanisms following a stressful event indicate a compromised ability to ward off infections or to eliminate pathogens.

As we have previously stated, infectious disease agents can be separated into obligate or facultative pathogens, and the impact that a compromised immune system has on the pathogenesis of a particular disease is dependent on the specific nature of the pathogen involved. The inherent stress of high density recirculating systems requires a fish health program that is predicated on the avoidance of obligate pathogens, as discussed above. However, facultative organisms such as Flexibacter columnaris or Aeromonas hydrophila usually require a stress-induced loss of disease resistance in order to establish and maintain an infection (Figure 1) (Snieszko 1974, Walters and Plumb 1980). The prevailing theory is that many facultative bacteria are ubiquitous in water and only become a problem when the fish are compromised by environmental perturbations (Wedemeyer and Wood 1974, Wedemeyer, et al. 1976). In addition, theextensive loading of organic matter in intensively fed aquaculture systems favors high counts of potential facultative bacterial pathogens (Snieszko 1974). For example, in enriched environments Aeromonas hydrophila can reach densities of over 2×10^7 cells per ml compared with 2×10^4 cells per ml in control waters. The increased facultative bacterial pathogen load observed in enriched water is also pertinent to recirculating systems because the static nature of the system results in the presence of bacterial "sinks" in the biofilter, sumps, and other areas where nutrients accumulate. Thus, the rapid removal of excessive nutrients through the efficient removal of solids and metabolic waste will not only reduce stressful water quality conditions and improve fish health, but will reduce the numbers of facultative pathogens.

Although obligate pathogens must be avoided in recirculating systems because they are capable of causing disease without the addition of stress, a dramatic effect on the severity of disease and mortality occurs when a stressor is added. For example, stress will increase the mortality of channel catfish when challenged with the obligate bacterial pathogen E. ictaluri (Figure 2). Fish in recirculating systems are inherently stressed due to the high stocking densities,

which also facilitates the transmission of disease from one host to another. Marginal water quality resulting from efforts to maximize production and increased handling neccessary in recirculating tank systems also causes stress. The nature of obligate pathogens and their enhanced activity in stressed fish mandate the avoidance approach presented earlier.

Stressors

Stressors can be classified by the effect they have on the animal's metabolic rate at the cellular level. Hoar (Hoar 1966) classified stressors into three categories: (1) limiting stressors, which deprive the cell of an important input, thereby reducing metabolic rate. Low dissolved oxygen and nutritional deficiencies would fit into this category; (2) inhibiting stressors, which suppress the rate of metabolism, such as poisons and low temperature; (3) loading stressors, which increase the rate of metabolism past the normal range. Examples are increased temperature, salinity, and excessive exercise.

Water Quality

The quality of water in a recirculating aquaculture system has to be maintained within the specific limits of the fish species being cultured. There are water quality guidelines for individual species such as the catfish (Tucker 1985) and for general aquaculture systems (Agency 1973, Brune and Tomasso 1991, Piper, et al. 1983). Most fish should be cultured in waters containing between 75 and 300 mm Hg dissolved oxygen (Colt and Orwiscz 1991, Doulos and Kindschi 1990). The pH should be maintained between 5 and 9 (Randall 1991) and the carbon dioxide below 20 mg/L (Beamish 1964). The un–ionized ammonia should be maintained below 0.05 mg/L , and the nitrites should be maintained at or below a nitrite to chloride molar ratio appropriate for the fish species being reared (Colt and Tchobanoglous 1978, Russo and Thurston 1991). Other compounds that can cause stress are hydrogen sulfide, heavy metals, and organic or introduced toxicants (Schwedler, et al. 1985). Adverse water quality conditions are probably the most common stressor that fish encounter in confined culture, particularly in recirculating systems. However, it is important to note that published water quality parameters for a given species may not adequately consider reduced disease resistance at the extremes of the optimal ranges, especially when the impact on circulating levels of facultative pathogens is considered. Continued outbreaks of disease associated with facultative pathogens in an "acceptable" water quality range indicate that water quality parameters need to be re-evaluated. This evaluation should consider the effects of water quality both on the fish and on facultative pathogen levels in the water.

In confined aquaculture systems, it is also common for there to be more than one stressor encountered simultaneously. As can be seen in Figure 3, there is an additive or synergistic effect when two stressors are applied to fish at the same time. Lymphocyte stimulation indices, which measure lymphocyte activity, were lower in net confined (behaviorally stressed) control fish than in control fish left loose in the same tank. The application of a toxicant decreased the stimulation index when compared with controls, and the response was depressed even further when the intoxicated fish were also net stressed. Thus, while individual water quality parameters may be within optimal ranges, the sum total of the parameters may be in a stressfull range, especially when the other stressors associated with intensive aquaculture are factored in.

Physical Stressors

Physical stressors that cause increased incidence of disease and impaired immune function are primarily related to management. For example, handling or net confinement initiates a classical stress response with elevated cortisol levels, osmoregulatory dysfunction, and impaired lymphocyte function (Ellsasser and Clem 1987, Redding and Schreck 1983). Barton and Iwama (Barton and Iwama 1991), in a recent review of the physiological changes in fish associated with handling and transport, indicate that as the duration of handling increases, cortisol levels and the degree of immune impairment increase. Other physical stressors include gas supersaturation, which can also be a primary etiologic agent (Colt, et al. 1986); incorrect artificial lighting and photoperiod manipulation; high velocity flow rates which cause excessive exercise (Casillas and Smith 1977); startling noises; and rapid temperature changes.

Behavioral Stressors

Behavioral stressors can be related to stocking density in the rearing tanks (Li and Brocksen 1977). It is felt that behavioral characteristics of certain ages and species of fish are related to dominance establishment. When fish densities are too low, dominant fish chase subordinate fish and cause injuries that occur during fighting. When densities are in an optimal range for the species, the incidence of dominance chasing and fighting is reduced. However, if densities exceed a threshold, water quality management is impaired and environmental deterioration results. Densities can also affect the incidence and severity of infection (Schwedler and Plumb 1982), probably because of enhanced transmission due to the proximity of the fish and an increase in water borne pathogens, as well as the imposed stress due to crowding.

Nutritional Deficiencies

While nutritional deficiencies are not generally regarded as inducing the classical stress response, they can cause or exacerbate infectious disease in a similar manner. Nutritional deficiencies have been reported as primary etiologic (causative) agents (Roberts and Bullock 1989) and as predisposing factors

for infectious and parasitic agents due to reduced disease resistance (Blazer 1991, Durve and Lovell 1982, Ghittino 1989, Landolt 1989, Li and Lovell 1985). The nutritional status of fish in a recirculating system or other dense culture system isdependent on the quality of the feed and the feeding practices employed by the fish culturist. High quality feed and careful feeding practices are of paramount importance in maintaining fish in good condition. Healthy fish are better able to adapt to unfavorable conditions and resist infectious disease agents. In recirculating systems, as with other confined production systems, there is no opportunity for the fish to acquire nutritional inputs from outside sources. Therefore, all macro and micro-nutrients, vitamins, and minerals must be provided in the feed.

Summary

Disease management in high-density, recirculating aquaculture systems must emphasize avoidance of obligate pathogens and stress reduction practices to control facultative pathogens. Some simple approaches to minimizing the impact of disease include:

1. Strickly avoid the introduction of obligate pathogens through disease inspection of all new fish stocks, quarantine, batch production, and hygienic culture practices.

2. Reduce stress due to poor water quality by designing the system to maintain optimal levels.

3. Avoid excessive handling and allow 1-2 weeks between handling disturbances to allow recovery from the stress.

4. Provide a nutritionally balanced diet, preferably in a processed form to avoid the introduction of pathogens in live or unprocessed diets.

5. Utilize ultraviolet light or ozone to reduce the numbers of facultative organisms in the circulating water.

6. Institute a fish-health monitoring system to do routine health checks and necropsy any mortalities that occur. Frequent outbreaks of facultative pathogens indicate a need to reduce stress.

7. Make sure fish are growing at optimal rates to avoid excess time in the culture system.

Literature Cited

Agency, E. P. 1973. Water Quality Criteria. EPA-R-73-033. U.S. Environmental Protection Agency.

Barton, B. A. and G. K. Iwama. 1991. Physiological changes in fish from stress in aquaculture with emphasis on the response and effects of corticosteroids. Annual Review of Fish Diseases 1:3-26.

Beamish, F. W. H. 1964. Respiration of fish with special emphasis on standard oxygen consumption, IV. Influence of carbon dioxide and oxygen. Canadian Journal Zoology 42:847-856.

Blazer, V. S. 1991. Piscine macrophage function and nutritional influence: A review. Journal Aquatic Animal Health 3:77-86.

Casillas, E. and L. S. Smith. 1977. Effect of stress on blood coagulation and hematology in rainbow trout (Salmo gairdneri). Journal Fishery Biology 10:481-491.

Colt, J., G. Bouck and L. Fidler. 1986. Review of current literature and research on gas supersaturation and gas bubble trauma. Special Publication Number. Bioengineering Section, American Fisheries Society, Bethesda, Maryland, USA.

Colt, J. and C. Orwiscz. 1991. Aeration in intensive culture. Pages 90-104 in D. E. B. a. J. R. Tomasso, editor. Aquaculture and Water Quality. World Aquaculture Society, Baton Rouge, Louisiana, USA.

Colt, J. and G. Tchobanoglous. 1978. Chronic exposure of channel catfish, Ictalurus punctatus, to ammonia effects on growth and survival. Aquaculture 15:353-372.

Doulos, S. K. and G. A. Kindschi. 1990. Effects of oxygen supersaturation of cutthroat trout, Oncorhynchus clarkii Richardson, and rainbow trout Oncorhynchus myskiss Richardson. Aquaculture and Fisheries Management 21:39-46.

Durve, V. S. and R. T. Lovell. 1982. Vitamin C and disease resistance in channel catfish, Ictalurus punctatus. Canadian Journal of Fisheries and Aquatic Science 39:948-951.

Ellsasser, C. F. and L. W. Clem. 1987. Cortisol-induced hematologic and immunologic changes in channel catfish (Ictalurus punctatus). Comparative Biochemical Physiology 87A:405-408.

Ghittino, P. 1989. Nutrition and Fish Diseases. Pages 682-713 in J. E. Halver, editor. Fish Nutrition. Academic Press, Inc., San Diego, California.

Hayasaka, S. S. and T. E. Schwedler. 1991. Effects of trichlorofon on the immuneresponse of channel catfish. Report Number 13. South Carolina Water Resources Institute, Clemson, South Carolina.

Landolt, M. L. 1989. The relationship between diet and the immune response of fish. Aquaculture 79:193-206.

Li, W. H. and W. Brocksen. 1977. Approaches to the analysis of energetic cost of intraspecific competition for space in rainbow trout (Salmo gairdneri). Journal Fishery Biology 11:329-341.

Li, Y. and R. T. Lovell. 1985. Elevated levels of dietary asorbic acid increases immune response in channel catfish. Journal of Nutrition 115:123-131.

Randall, D. 1991. The impact of variations in water pH on fish. Pages 90-104 in D. E. Brune and J. R. Tomasso, editor. Aquaculture and Water Quality. World Aquaculture Society, Baton Rouge, LA.

Redding, J. M. and C. B. Schreck. 1983. Influence of ambient salinity on osmoregulation and cortisol concentration in yearling coho salmon during stress. Transactions American Fisheries Society 112:800-807.

Roberts, R. J. and A. M. Bullock. 1989. Nutritional Pathology. Pages 424-473 in J. E. Halver, editor. Fish Nutrition. Academic Press, Inc., San Diego, California, USA.

Schwedler, T. E. and J. A. Plumb. 1982. Golden shiner virus: Effects of stocking density on incidence of infection. Progressive Fish-Culturist 44:151-152.

Selye, H. 1936. A syndrome produced by diverse noxious agents. Nature 138:32.

Selye, H. 1946. The general adaption syndrome and the diseases of adaptation. Journal of Clinical Endocrinology 6:117-230.

Snieszko, S. F. 1974. The effects of environmental stress on outbreaks of infectious diseases of fishes. Journal of Fish Biology 6:197-208.

Tucker, C. S. 1985. Water quality. Pages 135-221 in C. S. Tucker, editor. Channel Catfish Culture. Elsevier, New York.

Walters, G. R. and J. A. Plumb. 1980. Environmental stress and bacterial infection in channel catfish Ictalurus punctatus. Journal of Fish Biology 17:177-186.

Wedemeyer, G. and J. W. Wood. 1974. Stress as a predisposing factor in fish diseases. Fish Disease Leaflet 30. U. S. Fish and Wildlife Service, Division of Fishery Research, Washington, D. C.

Wise, D. J., T. E. Schwedler and D. Otis. 1993. Effects of stress and susceptibility of naive channel catfish to environmental, dip-exposures with Edwardsiella ictaluri. Journal of Aquatic Animal Health in press.:

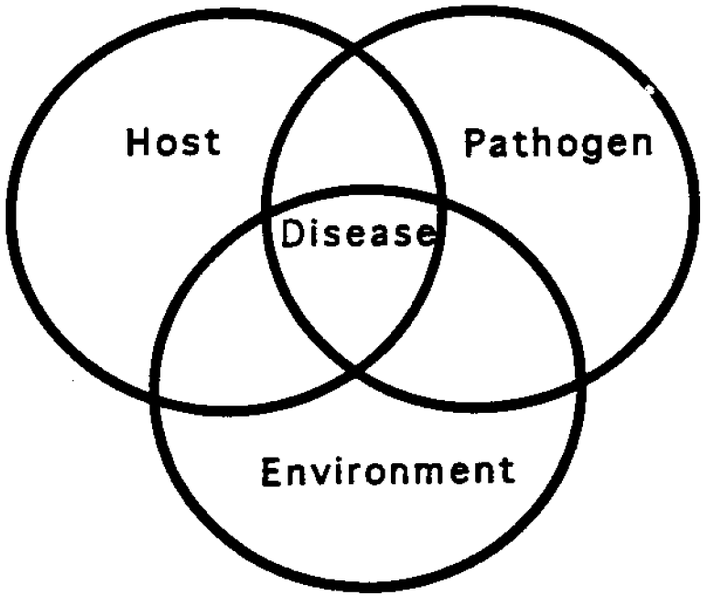


Figure 1. Representation of the interactions of facultative pathogens with environmental perturbations and the susceptible host to create a disease state (Snieszko 1974).

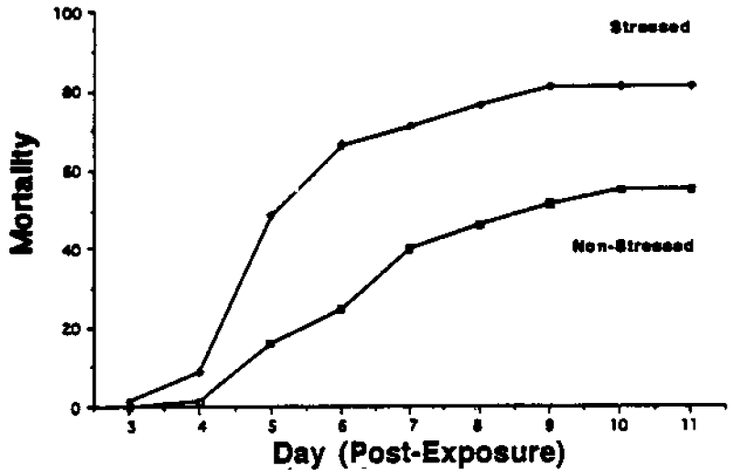


Figure 2. The effect of stress on mortality of channel catfish exposed to Edwardsiella ictaluri. From Wise et al. (Wise, et al. 1993).

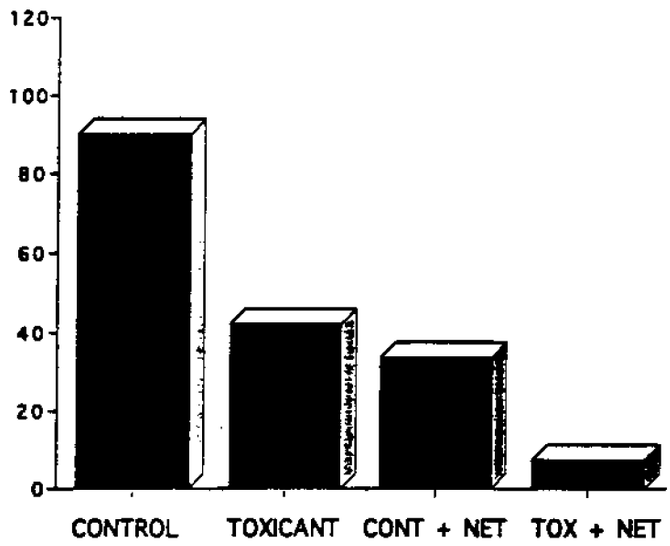


Figure 3. Response of channel catfish lymphocytes to individual and combined stressors. From Hayasaka and Schwedler (Hayasaka and Schwedler 1991).

Potential Effects of Therapeutic Agents on Biological Filtration In Closed Aquaculture Systems

John P. Hawke
Department of Veterinary Microbiology and Parasitology
School of Veterinary Medicine
Louisiana State University
Baton Rouge, Louisiana

Introduction

Ideally, fish health management in closed aquaculture systems involves the acquisition or establishment of disease free stocks through quarantine and prophylaxis. These fish may then be stocked into growout units. A state of heightened immunity of the cultured fish may be achieved and maintained through the use of vaccines. Others have discussed the maintenance of disease free status of fish in closed systems by water treatment through ozonation, ultraviolet sterilization and reduced stress on the cultured population. In reality, disease free stocks are rarely available and quarantine of incoming stocks from pond sources is not foolproof due to the introduction of asymptomatic carriers of obligate pathogens. Opportunistic pathogens may naturally occur in the culture facility environment and may be introduced into the system via the air, feed or hands of workers. Once a disease is introduced and established in a culture unit, water sterilization measures are somewhat ineffective in controlling the infection within that unit, however, they may prevent the spread of the disease to other units. This paper will focus on the potential course of action when the aforementioned methods have failed, survival of the fish is in jeopardy due to infectious disease and treatment of the fish with a drug or chemical is the only recourse.

The Decision to Use Drugs or Chemicals

The decision to use therapeutic agents in a closed aquaculture system is not always an easy one. There are many factors one must consider before making a final decision which may include; the species of fish being cultured, the identity of the specific pathogen or pathogens involved, the susceptibility of the fish to the particular pathogens, the susceptibility of the pathogen to various therapeutic agents, the effect of the therapeutic agent on the species of fish being cultured and the effect of the therapeutic agent on the nitrifying bacteria in the biological filtration system. If fish in a system become infested with ectocommensal protozoa at light to moderate levels and no mortalities are occurring, the fish should be carefully monitored for reduction in feeding rate. If the infestation remains at a low to moderate level and feeding activity remains good, treatment would probably not be warranted. However, if feeding rates decline as a result of the ectocommensal infestation or a diagnosis reveals a bacterial infection and mortality rates are in the range of 0.5 - 1 % per day and increasing, application of a therapeutic agent will be required.

The first step in the decision making process is to obtain a complete diagnosis from a fish disease diagnostic laboratory. Only after a complete diagnosis is made can the proper treatment be selected. The second step is to examine the EPA and FDA status of the drug or chemical chosen to treat the disease. If the chemical or drug is not cleared for use on the species of fish being cultured as a food item, then special permission will have to be obtained from the Center for Veterinary Medicine, Rockville, Maryland or a veterinary prescription will be required. The third step is to determine if the manner in which the drug or chemical is being applied will adversely affect the population of nitrifying bacteria in the biological filtration system.

Nitrifying Bacteria

A discussion of the effects of therapeutic agents on biofiltration should begin with a characterization of the nitrifying bacteria themselves. Spotte (1979) listed the factors which influence the activity of nitrifying bacteria as temperature, pH, dissolved oxygen concentration, salinity, surface area for attachment and toxic substances in the water (including medications). The nitrifying bacteria are comprised of several genera of chemolithotropic bacteria in the family *Nitrobacteriaceae*. These bacteria can use inorganic compounds, primarily ammonia and nitrite, as energy sources and can fix CO_2 as a carbon source. The organisms in this family are composed of two physiological groups of bacteria which are not phylogenetically related (Woese et al., 1984). The species in one group oxidize ammonia to nitrite and those in the other group oxidize nitrite to nitrate. The genus of ammonia oxidizing bacteria which has received the most study is *Nitrosomonas* and the genus of nitrite oxidizing bacteria investigated most thoroughly is *Nitrobacter*. Both of these genera of bacteria have a gram negative type of cell wall. In nature and in aquaculture systems, many of the nitrifying bacteria occur in cell aggregates referred to as zoogloeae or cysts. Zoogloeae are composed of loosely associated

cells embedded in a soft slime layer while cysts are composed of 5 or 6 closely packed cells and surrounded by a firm slime layer. This mechanism may serve to protect the bacteria from toxic chemicals present in the water or dessication as a result of fluctuating water levels. These forms are usually found attached to tank walls or filter media. Nitrification may occur in the water column as a result of detached cysts or individual cells. This physical phenomenon may account for the varied effects of certain chemicals and biologics in different types of testing systems.

Therapeutic Agents

Therapeutic agents will be divided into two basic groups for the purpose of this paper, antibacterial agents and parasiticides. Unfortunately there is limited published data on the effects of therapeutic agents on biological filtration but I will try to keep anecdotal accounts of the results of their use to a minimum.

Antibacterial Agents

Antibacterial agents may be used to treat infectious bacterial diseases of fish by adding them directly to the water or feeding them to the fish in a medicated feed. Valuable brood fish may be injected with antibiotics but this practice is not economically feasible for large numbers of fish. Most antibiotics, with the exception of Furanace® (nifu pirinol), are not taken up very efficiently from the water and are only effective against external infections of the skin, fins and/or gills. For this reason antibiotics are rarely added directly to the water and instead are administered via the feed. Levels of drug introduced into the system in this manner are normally much lower than if administered as a bath treatment. The activity of the individual drug will determine it's safety for use in an aquaculture system. Although most of the antimicrobial compounds used in aquaculture are designed to control gram negative organisms, they are bacteriostatic in nature and in most cases the nitrifying bacteria should recover after the treatment is removed.

A hypothetical example of the amount of drug that would enter a closed system following a 10 day oral treatment with Terramycin® medicated feed would be as follows: A 2000 l system containing 200 lbs of catfish, treated at 1.5% body weight per day, would receive a total of 30 lbs of feed. The feed contains 2.5 g oxytetracycline per pound resulting in a total of 75g of drug entering the system. This would equal 37.5 mg/l or 3.75 mg/l/day. This would be considerably less than the single bath treatment rate of 50 mg/l recommended by various sources (Collins et al. 1976).

The amount of drug that would accumulate in the water following oral administration would vary according to the bioavailability and metabolism of the individual drug and the fish species being treated. Plakas et al. (1990) studied the tissue distribution and renal excretion of ormetoprim in the channel catfish. Ormetoprim is used to potentiate sulfadimethoxine in the drug Romet®. The percentage of radiolabeled ormetoprim 24 hr after an oral dose of 4 mg/kg was 6.7% in the tissues, 19.3% in the urine, 5.8% in the bile, 1.4% in the gut and 54.2% in the tank water. They reported that ormetoprim was extensively metabolized by channel catfish with less than 4% of the drug being eliminated in the urine and bile as the parent compound. In a separate study, Squibb et al. 1988, found that Sulfadimethoxine was present at > 95% as the parent compound in the muscle tissue and plasma but was excreted into the bile at approximately 90% in the N-acetyl form and 10% in the form of other metabolites. Samples of urine taken at 24 hrs after dosing revealed that approximately 45% of the sulfadimethoxine was in the parent form and the remainder in the form of metabolites. From these studies it can be seen that the effect of a dose of an antimicrobial drug to a fish population in a closed system will be greatly influenced by the bioavailability of that drug and the extent ot which the drug is metabolized to other forms which may have limited activity against bacteria in the biological filter.

For the tables to be more informative, a description of the systems and the conditions under which the experiments were conducted follows:

(1) The initial tests of Bower and Turner (1982) were conducted in glass aquaria containing 15 L of water with undergravel filtration and 4000 cm^3 dolomitic limestone as the filter medium. The salinity of the water was 30 ppt prepared with Instant Ocean® and organic loading was achieved by stocking mummichogs *Fundulus heteroclitus* (average total weight 7.5 g per tank) and feeding at 2% body weight per day. A second set of experiments to determine the effects during and after treatment were conducted in 38 liter aquaria containing 25 liters of artificial seawater and 6000m^3 of conditioned filtrant. Each tank was stocked with 7.7 g of fish per aquarium. The duration of the treatments were: chloramphenicol and neomycin 7 days, methylene blue and copper sulfate 14 days. At the end of the respective treatment period, the water in the aquarium was replaced.

(2) The tests of Collins et al. (1975 and 1976) were conducted in 70 liter glass aquaria equipped with a commercial filter (Cosmic Industries) containing 500 cm^3 of seasoned crushed oyster shell and 500 cm^3 #4 quartz gravel as the filter medium. The recirculation rate was approximately 15 liters per minute. The systems were filled with declorinated tap water pH 7.2 and hardness 30 ppm. Each

aquarium was stocked with 20 channel catfish fingerlings *Ictalurus punctatus* average weight 16.3g fed a commercial diet at 1.5% of total body weight per day. Systems were operated for 14 days before treatment and monitored for 26 days after treatment. Treatments were administered once by direct addition to the water.

(3) The tests of Levine and Meade were conducted in 500 ml Erlenmeyer flasks containing 275 ml of a medium specific for culturing nitrifiers (Meiklejohn, 1954). Each flask was inoculated with semi-pure cultures of nitrifiers from a batch culture tank prior to the addition of the chemical to be tested. The flasks were incubated at 27°C on a reciprocating shaker for for 72 hrs. Dissolved oxygen was maintained above 7.5 ppm.

(4) The system in which the tests of Muir and Roberts were conducted was not described in detail but is believed to be a large aquaculture system with high organic loading.

(5) The system of Rosenthal and Otte, 1979 was a 5.4 m^3 system in which biological filtration was achieved using a trickling type filter filled with Hydropak plastic media. The system was conditioned as a fresh water system for 15 weeks prior to increasing the salinity of the water 2 ppt in daily increments. The fish load during the experiment was 92 kg of *Tilapia sp.*

Table 1 lists some of the antibacterial agents that have been evaluated as to their effect on nitrification in aquatic systems.

Table 1. Antibacterial agents.

Compound	(Conc. mg/l)	Inhibition	Source
Chloramine T	10.0	N	Muir and Roberts
Chloramphenicol	13.3	S	Bower and Turner
Chloramphenicol	50.0	E	Levine and Meade
Chloramphenicol	50.0	N	Collins et al.
Chloratracycline	10.0	E	Levine and Meade
Erythromycin	50.0	E	Collins et al.
Gentamycin	5.0	N	Bower and Turner
Neomycin	66.0	M	Bower and Turner
Nifurpirinol	1.0	N	Collins et al.
Nifurpirinol	0.1	S	Levine and Meade
Nifurpirinol	0.1	N	Bower and Turner
Nifurpirinol	0.1	S	Coffin and Malone
Oxytetracycline	50.0	N	Collins et al.
Sulfamerazine	50.0	N	Collins et al.
Sulfanilamide	25.0	M	Levine and Meade
Sulfadiazine + Trimethoprim	10.0	S	Muir and Roberts

N=none, S=slight, M=moderate, E=extreme effects on nitrification.

Since different publications reported the data in different ways, I decided to report the inhibition in terms of None = 0% or negligable inhibition, Slight = 10 - 20% inhibition, Moderate = 30 - 70% inhibition and Extreme = 70 - 100% inhibition.

Parasiticides

Parasiticides may be administered to fish orally or by adding them directly to the water. The vast majority are added to the water, however the target parasite dictates the proper method of treatment. It is imperitive that any chemical treatment be evaluated in terms of efficacy and effect on nitrification. The efficacy of various disease treatments were reviewed by Hoffman and Meyer, 1974. An interesting complication in the measurement of ammonia exists when systems are treated with formalin. Formaldehyde reacts with ammonia in an aqueous solution to produce hexamethylenetetramine (Levine and Meade, 1976). This reaction would pose a problem in the accurate measurement of ammonia. Other chemicals should not interfere with measurements.

Table 2 lists some therapeutic agents that have been evaluated as to their effect on nitrification in aquatic systems.

Table 2. Paraciticides

Compound	(Conc. mg/l)	Inhibition	Source
Copper Sulfate	1.0	N	Collins et al.
Copper Sulfate	0.3	M	Bower and Turner
Copper Sulfate	5.0	N	Levine and Meade
Formalin	10.0	N	Muir and Roberts
Formalin	25.0	N	Collins et al.
Formalin	15.0	S	Levine and Meade
Malachite Green	0.1	S	Muir and Roberts
Malachite Green	0.1	N	Collins et al.
Malachite Green	0.5	S	Levine and Meade
Methylene Blue	5.0	E	Levine and Meade
Methylene Blue	8.0	E	Bower and Turner
Methylene Blue	5.0	E	Collins et al.
Potassium Permanganate	4.0	N	Collins et al.
Potassium Permanganate	1.0	E	Levine and Meade
Quinacrine	12.0	N	Bower and Turner
Salt (NaCl)	5000.0	N	Collins et al.
Salt (NaCl)	8000.0	S	Rosenthal, Otte
Salt (NaCl)	30000.0	E	Spotte

N=none, S=slight, M=moderate, E=extreme effects on nitrification

Increasing Salinity

Increasing the salinity to treat external parasites and bacterial infections is a common practice with euryhaline species of fish. The effects of increasing salinity at 2ppt intervals on nitrification in a fresh water system are depicted in Figure 1.

In this study increasing salinity had an immediate effect on the populations of nitrifying bacteria, however the *Nitrosomonas* populations were quick to adapt.

Long Term Effects of Therapeutic Agents on Nitrification

The long term effects of selected therapeutic agents on ammonia oxidation in marine systems is depicted in Figure 2 which was adapted from data by Bower and Turner. The conditions of the tests have been previously described. The water replacements to terminate various treatments were done on day seven (arrow) and on day 14 (asterisk). Post treatment ammonia levels decreased in tanks recieving chloramphenicol but remained about the same in those treated with neomycin. No reduction in the level of ammonia was noted after water exchange in tanks treated with methylene blue and copper sulfate when monitored over 21 days post initial treatment. Of the four compounds tested in the study by Bower and Turner, only neomycin and copper sulfate caused significant increases in nitrite (data not shown).

In contrast, Figure 3. shows the severe impact of Erythromycin on ammonia oxidizing bacteria and the delayed affect on the nitrite oxidizing bacteria (adapted from Collins et al. 1976).

Regulations Concerning Drugs, Biologicals, and Chemicals

The only therapeutic agents registered for use on food fish are; Acetic acid, Formalin-F, Paracide-F, Romet-30, Sulfamerazine, Terramycin (Oxytetracycline) and Salt (NaCl). Most of the chemicals and drugs listed in the tables are labeled for non-food fish culture or may not be legal to use at all therefore the mention of a drug or chemical in this publication does not in any way endorse its use. In order to use a non-registered therapeutant for a particular species of fish, a request must be filed with the Center for Veterinary Medicine, Rockville, Maryland, or a veterinary prescription for extra label use must be obtained.

For those drugs that are approved, the treatment rate prescribed on the label should be followed for maximum safety and efficacy.

References

Bower, C.E. and D.T. Turner. 1982. Effects of seven chemotherapeutic agents on nitrification in closed seawater culture systems. Aquaculture 29:331-345.

Collins, M.T., J.B. Gratzek, D.L. Dawe, and T.G. Nemetz. 1975. Effects of parasiticides on nitrification. J. Fish. Res. Board Can. 32:2033-2037.

Collins, M.T., J.B. Gratzek, D.L. Dawe, and T.G. Nemetz. 1976. Effects of antibacterial agents on nitrification in an aquatic recirculating system. J. Fish. Res. Board Can. 33:215-218.

Hoffman, G.L. and F.P. Meyer. 1974. Parasites of freshwater fishes. T.F.H. Publications, Inc. Neptune City, NJ. 224 pp.

Levine, G. and T.L. Meade. 1976. The effects of disease treatment on nitrification in closed system aquaculture. In. J.W. Avault Jr. (Editor), Proc. 7th Ann. Meet. World Maricult. Soc. Louisiana State Univ., Baton Rouge LA. p 483-493.

Meiklejohn, J. 1954. Some aspects of the physiology of the nitrifying bacteria. In autotrophic microorganisms: 4th symposium of the Society of General Microbiology, ed. B.A. Fry and J.L. Peel. p 74, Cambridge University Press.

Muir, J.F., and R.J. Roberts. 1988. Recirculated water systems in aquaculture. In: Recent Advances in Aquaculture Vol. 3. J.F. Muir and R.J. Roberts (Editors). Westview Press, Boulder, CO.

Plakas, S.M., R.W. Dickey, M.G. Barron, and A.M. Guarino. 1990. Tissue distribution and renal excretion of ormetoprim after intravascular and oral administration in the channel catfish *Ictalurus punctatus*. Can. Jour. of Fish. and Aquatic Sci. 47(4):766-771.

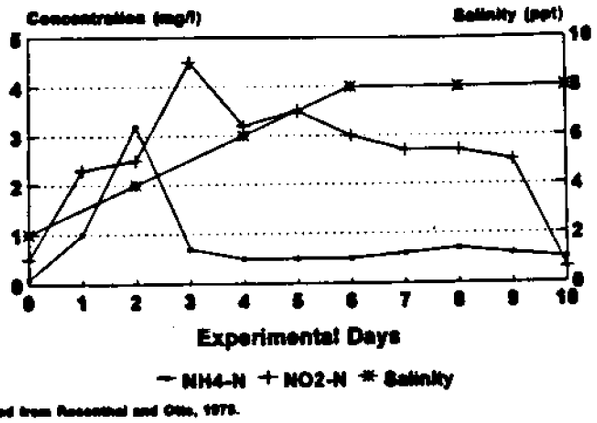
Rosenthal, H. and G. Otte. 1979. Adaptation of a recirculating fish culture system to salinity change. Biologische Anstalt Helgoland, Zentrale, Hamburg. p 203-206.

Spotte, S., 1970. Fish and Invertebrate Culture: Water Management in Closed Systems. Wiley, New York, NY. 145 pp.

Spotte, S., 1979. Seawater Aquariums: The Captive Environment. Wiley, New York, NY. 413 pp.

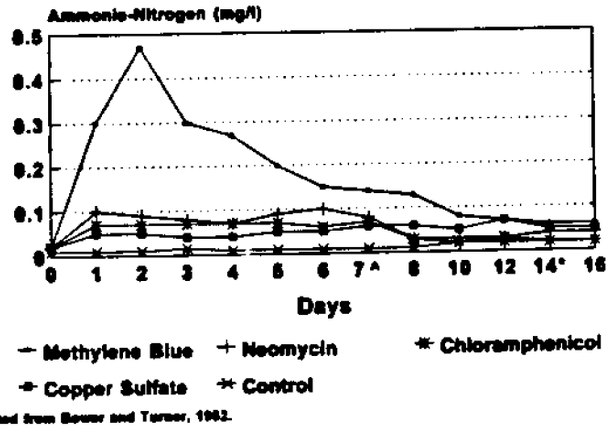
Squibb, K.S. C.M.F. Michel, J.T. Zelikoff, and J.M. O'Connor. 1988. Sulfadimethoxine phamacokinetics and metabolism in the channel catfish *Ictalurus punctatus*. Vet. Human Toxicol. 30:31-35.

Woese, C.R., W.G. Weisburg, B.J. Paster, C.M. Hahn, R.S. Tanner, N.R. Kreig, H.-P. Koops, H. Harms, and E. Stackebrandt. 1984. The phylogeny of the purple bacteria: the beta subdivision. Syst. and Appl. Microbiol. 5: 327-336.



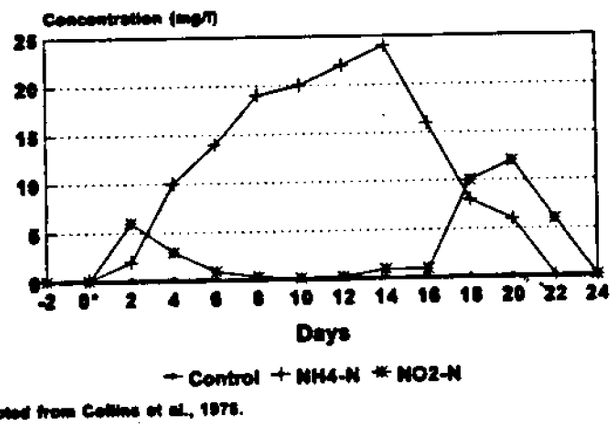
Adapted from Rosenthal and Otte, 1975.

Figure 1. Effects of salinity change on nitrification.



Adapted from Bower and Turner, 1982.

Figure 2. Effects of therapeutic agents on nitrification.



Adapted from Collins et al., 1975.

Figure 3. Effect of erythromycin on nitrification.

Disease Prevention by Water Treatment with Ultraviolet, Ozone and Filtration

David E. Owsley, P.E.
Dworshak National Fish Hatchery
Ahsahka, Idaho

Position Statement

The objective of disinfecting a recirculating water supply or the make up water supply to a recycle system is to achieve the functional inactivation of infectious fish pathogens. There are three major criteria for providing a disinfected water supply in a hatchery (Montgomery Engineers, 1991).

1. Kill or inactivate all bacteria, protozoan and viral agents capable of causing sickness or mortality in the hatchery population.

2. Insure that the disinfected water supply to the incubation, rearing and holding facilities of the hatchery does not contain any residue or significant concentration of chemicals that will impact fish survival, growth or development.

3. Provide a treated water supply that is essentially (significantly) unchanged from the influent water quality other than with regard to the destruction of pathogens, reduction of residual organics and, as possible, improved dissolved oxygen availability in the product water.

A number of disinfection systems are available to the fish culturist today. Some have been tried and work well, while some hold promise and some have failed.

1. Ozone
2. Ultraviolet light
3. Chlorine
4. Iodine
5. Hydrogen peroxide
6. Chlorine Dioxide
7. Photozone
8. Perozone
9. Nuclear Radiation
10. Microfiltration
11. Heat
12. Pure oxygen

Ozone

Owsley (1988) reported on ozone being used at Dworshak National Fish Hatchery. Ozone is a three atom allotrope of oxygen. It is a colorless gas and can be readily detected by its odor at very low concentrations. It is an unstable gas and the strongest oxidizing agent commercially available. Ozone is produced by passing air or oxygen through a high frequency electric field. Ozone has to be generated at the point of application.

Commercial ozone generators are available in many designs. Most generators use the high voltage corona discharge system. This system consists of two surfaces separated by a space. A high voltage is impressed across this space. Air or oxygen is passed between the surfaces where the oxygen molecules are excited sufficiently enough to form ozone. Ozone can be produced by ultraviolet radiation. Ultraviolet radiation does not produce the quantity of ozone that an ozone generator can achieve. Ozone is also produced by lightning or any high voltage with air contact such as welding.

Ozone is directly toxic to aquatic organisms and to man. Ozone is highly effective as a disinfectant, having about twice the oxidizing capabilities as chlorine. Ozone is toxic on contact killing both bacteria and viruses with equal effectiveness and speed. Ozone reacts very quickly as compared to compounds such as chlorine. Ozone effectiveness is much less affected by pH and temperature than chlorine.

Ozone is not limited to low turbidity water like ultraviolet radiation. Ozone does not appear to leave harmful residues in water such as chloramines produced by chlorine. Both organic and inorganic materials exhibit a demand for ozone. Ozone is used for removal of color, odor and turbidity. Water containing organic matter must be treated with a higher ozone level than similar water without organic matter to achieve the same disinfection level. Inorganic materials such as iron and manganese can be oxidized to the insoluble oxide forms by ozone.

Ozone decomposes back to oxygen. The decomposition rate is temperature dependent, rapidly increasing with increased temperature. An ozone destruction unit is basically a heater. Ozone has been widely used in Europe for water disinfection for many years. Europe does not have the luxury of abundant clean water that exists in the United States and Canada. The use of chlorine and ultraviolet radiation was moreeconomical for the clean water. We are finding, however, that there are certain diseases that chlorine and ultraviolet radiation cannot effectively combat. One example for humans is giardiasis. This disease is caused by a protozoan, Giardia lamblia, commonly found in high mountain streams and lakes. Ozone will effectively kill Giardia and its cysts, whereas chlorine has been found ineffective. An example in aquaculture is the virus called infectious hematopoietic necrosis (IHN). Infectious hematopoietic necrosis is an epizootic virus causing high mortality with fish that survive becoming carriers. Adult fish shed the virus during

spawning. Natural transmission occurs through the water. External symptoms of fish with IHN include hemorrhaging under the skin, exophthalmia (protruding eyes), swollen abdomens, lethargy, darkening of skin color, and hemorrhaging at the base of the fins. Internally, the liver, spleen and kidneys are usually pale. The stomach and intestine may be filled with fluid.

The work at the Seattle Lab indicated that ozone destroyed the IHN virus at low dosage rates and contact times (Wedemeyer et al. 1978).

Table 1. Ozone dosage rates and contact times for different water sources.

Ozone Level	Contact Time	Water Source @ 10°C
0.01 mg/L	0.5 1.0 minute	phosphate-buffered distilled water
0.01 mg/L	10.0 minutes	soft lake water
0.01 mg/L	10.0 minutes	hard lake water

Other aquaculture studies have shown positive results using ozone to treat water supplies. Ceratomyxa shasta was controlled at the Cowlitz Hatchery in Washington (Tipping and Kral 1985). Coleman National Fish Hatchery in California has had some success in controlling Myxosoma cerebralis (whirling disease) using ozone (Baker 1986). Ozone was found superior to chlorine for inactivating the fish pathogens Aeromonas salmonicida (furunculosis), Yersinia ruckeri (enteric redmouth) and the pathogenic viruses infectious pancreatic necrosis (IPN) (Wedemeyer et al. 1977).

The combination of ozone and reuse is not a new concept. Morrison (1977) achieved a 70% decrease in ammonia using ozone on a pilot plant reuse system at Dworshak. Plate counts showed that ozone consistently provided better disinfection of make-up water than the existing ultraviolet system.

A similar study by Oakes et al. (1978) at Dworshak showed that nitrite could be virtually eliminated in reuse water.

Williams et al. (1982) demonstrated at the Tunison Lab in New York that ozone and reuse were an excellent combination for reuse with only 1% make-up water.

Rosenthal and Kruner (1985) determined oxidation rates for ammonia, nitrite and biological oxygen demand (BOD) separately and in combination using an improved contact chamber design. The upflow design with an improved foam removal system demonstrated good results in the study.

Foam can be a problem in an ozone system. The foam is made up of protein, suspended and dissolved solids, organic and inorganic compounds. It is mainly found in the contact chamber and most systems are designed with some type of removal such as a protein skimmer. Foam in incubation and especially during nursery rearing can be a hindrance to fish culture operations. In nursery tanks, the foam prevents the starter feed from getting to the fish. A foam remover such as Dow Corning anti-foam FG-10 can be used to clear the water.

Ozone is unstable and samples should be fixed at the sampling site. There are several procedures available to sample ozone.

The DPD (N,N-diethyl-P-phenylenediamine) method (Paulin 1967) is easy to use but has limitations. The test must be completed between 1 and 6 minutes after fixing the sample. It has numerous interferences and should only be used as an indicator in water that is similar to distilled type water.

A modified DPD method (Wedemeyer et al. 1978) showed good results at lower ozone levels. This method used a longer sample cell for increased accuracy.

The Indigo Blue method (Bader and Hoigne 1982) is the standard procedure for measuring ozone in water. It is stable up to 4 hours and has the least interferences of any methods available. Accuracy is good and color development is excellent.

The feasibility of treating lesser quality water as opposed to developing good quality supplies was studied at Coleman National Fish Hatchery in northern California (Sverdrup and Parcel Engineering 1986). Four water supply systems were evaluated: wells, chlorination/dechlorination, ultraviolet sterilization and ozonation. Table 2 summarizes the findings of that study using ultraviolet sterilization as the base. At this facility,ozonation was the logical choice to pursue based upon cost/effectiveness.

Table 2.

Water Supply	Construction Costs	Annual Operation and Maintenance
Wells	366%	45%
Chlorination/ dechlorination	70%	61%
Ultraviolet sterilization	100%	100%
Ozonation	94%	36%

As new developments arrive, ozonation will become more cost/effective. One new development, the aquatector (Schutte 1986) could greatly improve or eliminate the need for ozone contact basins. The aquatector uses micro bubble technology which could improve the efficiency of getting ozone into the water. This would in turn reduce the size of equipment needed and reduce costs.

On-site oxygen generating equipment can double the production of ozone and reduce costs accordingly. Liquid oxygen is also an alternative that can be considered.

New technology from the ozone industry is occurring. Better, more reliable equipment that will be smaller, more energy efficient and produce more ozone is being developed.

Ozone is a very toxic compound. The toxicity of ozone is a function of time and concentration. The maximum allowable concentration for an 8-hour day is 0.1 mg/L. Ozone can normally be detected in the air by the human nose in the range of 0.05 mg/L. It is very important that the work area be free of ozone. Ozone can be converted back to oxygen using a heat system prior to being discharged into the atmosphere.

Ozone is also very toxic to fish and must be removed from the water supply. Wedemeyer et al. (1979) determined that the permissible safe exposure level was 0.002 mg/L. Several other studies have verified this level while others have reported much higher exposure levels. This discrepancy from Wedemeyer's work is probably due to sampling accuracy and production versus laboratory conditions. Because of the low exposure level, ozone must be removed from the water. The conventional method to remove the ozone is detention chambers. Ozone has a short half-life and can be removed by allowing the water to be held in a chamber for a period of time.

A faster, more economical way to remove the ozone is to strip it out of the water using packed columns (Owsley 1979). Efficiency of packed columns varies from 70% to 95% removal. Complete removal can be accomplished by using stripping towers (Montgomery Engineers 1987). A stripping tower is a packed column using a counter-current air flow. Stripping towers raise the energy costs due to the additional height and blower requirements.

Carbon filters will also remove ozone very effectively. These filters accommodate small systems and are not as economical as packed columns or stripping towers.

Conclusion and Recommendations

Ozone is a viable means of disinfecting water for aquaculture. It has been proven effective against fish and human pathogens at very low dosage rates.

Ozone is not as restricted as chlorination or ultraviolet radiation in the quality of water that can be treated. It has been used successfully in Europe for many years. Pilot studies in aquaculture have shown a lot of promise in the United States.

On-site production of oxygen will double the capacity of ozone generators and reduce costs accordingly as compared to compressed air systems. It will also eliminate the build-up of nitrous oxide and reduce annual maintenance.

The aquatector could improve efficiency and eliminate the need for a contact chamber. This, in turn, would reduce the size of the oxygen generating equipment.

Packed columns and stripping towers remove residual ozone and eliminate the need and costs of detention tanks.

By incorporating the above features in the design of an ozone system, ozone can be feasible as well as effective.

Ultraviolet light

Ultraviolet (UV) light or energy in the wavelength of approximately 254 nm has been shown to be effective in killing both bacterial and viral organisms. Light in the UVB and UVC spectra are responsible for the majority of the disinfection/sterilization attributed to this type of system. High quality bacteria free water (99.9% removal), has reportedly been produced using UV disinfection for low turbidity supply sources.

Since UV light must effectively penetrate a water source to ensure that the short wave energy is imparted to the biological organism, it is critical to ensure that the water source entering the UV contactor is very low in turbidity and suspended solids, and that turbulent conditions are maintained for adequate mixing. In order to achieve the influent water quality necessary, prefiltration is often recommended to remove particulate matter capable of blocking light penetration or fouling the equipment.

The EPA has not recognized UV treatment as an allowable method of potable water disinfection without other treatment, since it is impossible to develop a corollary concentration and detention time relationship to chemical oxidation. This does not mean that UV disinfection would not be effective: only that it has not been widely utilized for treating water supplies destined for human consumption.

The use of ultraviolet (UV) energy to deactive waterborne microorganisms in the aquaculture industry is a well known and understood technology. UV systems have become an integral part of many aquaculture operations providing disinfected water to areas of the hatchery and rearing operation where there is an established need to maintain control of fish pathogens (bacteria, viruses, molds and protozoans). Advantages of UV are:

1. Non-toxic.

2. Adds nothing to the water: thereby preventing formation of toxic chemical residuals.

3. Does not affect water chemistry.

4. Is effective against a wide range of organisms.

5. Can be designed to suit any flow rate.

6. Can be supplied with automatic duty stand-by capability.

It is generally accepted that the germicidal nature of UV energy is the result of disruption of the microorganisms' DNA molecules by irradiations of wavelengths 220-290 nm. With most organisms this effect is maximized at 260 nm. It has also been shown that the survival ratio after UV treatment is related to the UV dose applied (normally reported in milliwatt seconds or millijoules per square centimeter - $mWsec/cm^2$ or mJ/cm^2).

The UV dose received by the organism is dependent on:

1. The energy output of the UV source.

2. The flow rate of the water and its residence time under the influence of the UV Source.

3. The ability of the fluid to transmit the germicidal wavelengths; often referred to as "UV Transmittancy".

4. The geometry of the radiation chamber.

In many cases, UV systems have been designed to provide a high kill rate for organisms that are easy to kill with UV such as E. coli. E. coli is a common bacteria found in waster water treatment plants. A theoretical UV does of 3.2 mJ/cm^2 will provide a 90% kill, or one log reduction. (The actual UV dose applied will depend on factors such as UV transmittancy, TSS, etc.)

Look at the UV dose levels needed to achieve a 90% and a 99.9% reduction of the following fish pathogens: (Aquionics 1988)

	90% Reduction	99.9% Reduction
1. Viral hemorrhagic septicaemia (VHS)	10 mJ/cm^2	30 mJ/cm^2
2. Saprolegnia (fungal disease)	13 mJ/cm^2	39 mJ/cm^2
3. Ichthyophthirius (White Spot or "Ich")	40 mJ/cm^2	120 mJ/cm^2
4. Infectious pancreatic necrosis (IPN)	60 mJ/cm^2	190 mJ/cm^2

In many aquaculture applications there are seasonal changes in water quality: concentration of undesirable organisms and high suspended solids/turbidity levels that impact UV treatment. These facts, plus the lack of accurate, continuous monitoring/validation of the UV system operation have resulted in poor performance of UV equipment which was originally designed to treat consistently clear water. Several hatcheries, over the years,have installed large scale UV systems including the Dworshak National Fish Hatchery. However, the Trail Lake Hatchery (AK), which used UV for post hatchery disinfection, is the only major facility in the Northwest still using UV on a routine basis as far as can be determined. The Trail Lake system (2,000 gpm), based upon a recent conversation with the manager, is working satisfactorily.

Chlorine

According to the report by Montgomery Engineers (1990), chlorine is not a viable disinfectant alternative for most fish production facilities. Chlorination is the most common type of disinfection utilized for water treatment. The use of chlorine gas, HTH (hypochlorite), a chloramine or other materials to produce a solution containing the oxidant is practiced by the majority of domestic water purveyors throughout the United States. Chlorine disinfection has been utilized on several aquaculture projects within the Northwest, but has several inherent drawbacks. The most serious is the necessity to remove the residual chlorine prior to introducing the disinfected water into the hatchery (fish) in order to avoid the possibility of creating a toxic environment impacting fish production. However, Wedemeyer et al. (1978) and Bedell (1971) found chlorine effective in inactivating both C. shasta and IHN virus. The only large scale facility in the Northwest using chlorine disinfection in salmonids (Ore-Aqua) relies on careful control and constant monitoring in combination with sufficiently large post chlorination storage to provide the extended detention necessary for dissipation of any harmful chlorine residual. Chloramine (ammoniated chlorine) is not suitable for aquaculture systems due to its known toxicity to aquatic organisms and will not be considered as a feasible method of hatchery disinfection.

Iodine

Iodine is a nonmetallic element with an atomic weight of 126.92. It is the heaviest of the halogen group (chlorine, bromine, iodine, fluorine). The halogens are a group of elements that form with metal compounds like common salt—sodium chloride. It is the only halogen that is solid at room temperature. Iodine is a shining blackish brown crystal solid with a specific gravity of 4.93 and peculiar chlorine-like odor. Iodine is always found combined and can be prepared from kelp or crude chile saltpeter. It is only slightly soluble in water.

Iodine and its compounds have been used in medicine since the early 1800s. The first use of iodine in water was in World War I for sterilization of water for troops. Iodine as a disinfectant for water supplies has been recognized for a long time but has never been as feasible to use as chlorine. Iodine does have some advantage over chlorine as a water disinfectant. In the

late 1970s, the Idaho Department of Health and Welfare recommended that the domestic water supply treatment at Dworshak National Fish Hatchery be changed from chlorine to iodine. This change was to combat a parasite Giardia lamblia which is chlorine resistant.

Iodine is widely used in fish culture practices for egg disinfection. The term "iodophores" includes commercial forms of iodine with the two most common being Wescodyne and Betadir e. Wescodyne is an iodine solution with a detergent base. Betadine is an iodine solution that contains povidone as the organic base. Both iodophores contain iodine as the active ingredient with different organic bases. Both are considered to be effective bactericides and viricides.

Due to a severe Infectious Hematopoietic Necrosis Virus (IHNV) problem at Dworshak, since 1982, steelhead eggs have been water hardened in an iodophor solution each year to prevent the virus. Based upon personal communications with Dr. Bob Busch, Director of Rangens Laboratory in Buhl, Idaho, iodine was selected to be tested on a continuous drip method on a small group of fish from Brood Year 1989 steelhead egg production. Dr. Busch had been successful in reducing losses to IHNV by applying a continuous dosage of iodine between 0.3 to 0.5 mg/l to fry and fingerling rainbow trout. Dr. Busch also stated that iodine was an effective treatment against bacteria gill disease. This report on iodine was further substantiated by Dr. Jim Winton of the Service's National Fisheries Research Center in Seattle. Dr. Winton was able to effectively kill the IHN virus in a laboratory setting at the levels (0.3 to 0.5) that Dr. Busch had reported. With this data, a pilot study was set up for the Dworshak hatchery (Owsley 1989).

Results of the test showed that iodine was not successful in alleviating the IHN virus in the water with a continuous level of iodine present. Ozone and control groups both showed very little sign of the virus. The reason for these results are theorized based upon literature and fish health experience. Dworshak's water supply has a pH value that can range form 6.3 to 7.3 over a complete rearing cycle. Iodine is not a good viricide with this low to neutral pH. (see memo dated August 24, 1989). Iodine, however, is a good bactericide at these pH levels. The Ph theory might explain why Dr. Busch has success in southern Idaho where the water supply has a pH value of 8.2. Iodine is a good viricide at this level. Secondly, the low level of iodine on a continuous exposure could have created a stress on the fish that may have initiated an IHN outbreak in this study.

Further evaluation of iodine needs to be conducted at various levels of pH and water quality. Iodine does have the potential to help alleviate a serious problem in fish culture, the IHN virus.

Hydrogen Peroxide

Hydrogen peroxide is clear, colorless, waterlike in appearance, and has a characteristic pungent odor. Non-flammable, it is miscible with water in all proportions and is sold as a water solution. The amount of hydrogen peroxide in commercial solutions is expressed as a percentage of the solution's weight. Thus, a 35 percent solution contains 35 percent hydrogen peroxide and 65percent water by weight. Most industrial applications call for 35, 50 or 70 percent concentrations.

Hydrogen peroxide is not a particularly hazardous substance. It is considerably safer to handle and store than chlorine gas, which is so widely used in wastewater treatment. It does not have the highly corrosive and dangerously toxic characteristics of chlorine.

In fact, when properly handled and contained, hydrogen peroxide has been safely stored on street corners and the center strips of residential streets, where it is injected into sewer mains.

At the same time, a basic understanding of the properties of hydrogen peroxide is necessary for proper handling.

The U.S. Department of Transportation classifies solutions of hydrogen peroxide as "Oxidizer" (DOT yellow label). In addition, the chemical is a strong oxidizing agent, liberating oxygen and heat when it decomposes. In dilute solutions the heat is readily absorbed by the water, but in more concentrated solutions the heat raises the temperature of the solution and accelerates the decomposition rate. Hydrogen peroxide itself will not burn, but its decomposition liberates oxygen which supports combustion. This agent has a lot of potential for water treatment, but needs more research work.

Chlorine Dioxide

Chlorine dioxide ($C10_2$) is a powerful oxidant and disinfectant. It is also very unstable, and is explosive in air in concentrations above 4%. Because of its instability, it is always generated in solution on-site, and is used immediately without storage. This can be considered a disadvantage for some applications. As long as care is taken to keep chlorine dioxide in solution and its storage is avoided, its explosive potential can be vegetated. Chlorine dioxide is readily soluble in water and is decomposed by sunlight.

The material is more a powerful oxidizing agent and a better biocide than is chlorine. In addition, when chlorine dioxide is prepared in the absence of excess free chlorine, its use will not produce trihalomethanes, or other chlorinated organicby-products of current public health concern. Additionally, chlorine dioxide has been used in pretreatment to oxidize phenolic

compounds and to separate iron and manganese from organic complexes which are stable to chlorination.

Distribution system residuals of dissolved chlorine dioxide are longer-lasting than those of chlorine because there is no reaction with ammonia or formation of chlorinated organic materials. Additionally, chlorine dioxide is not known to impart tastes and odors to water, as does chlorine.

Recent health effects studies have shown (NAS 1987) that chlorine dioxide produces hematological effects in both humans and laboratory animals. For these reasons, the U.S. EPA (1981) advises that the total concentration of chlorine dioxide and its decomposition products (chlorite and chlorate ions) be maintained below 1 mg/L.

Gaseous chlorine dioxide has a strong, disagreeable odor, similar to that of chlorine gas, and is toxic to humans when inhaled. It is detected by the human nose at 4.5% concentrations, it irritates the respiratory mucous membranes and may cause severe headaches. At concentrations below 6% in air, it may be compared with chlorine with respect to its toxicity (Masschelein 1979). Eventual intoxications appear by local irritations of the nervous system, ocular and respiratory mucous membranes, without substantial resorption or systemic poisoning (Ehrlicher 1964). There are no cumulative effects in cases of repeated exposure (Haller & Northgraves 1955).

Photozone

For many organic compounds that are refractory even to so strong an oxidizing agent as ozone, the simultaneous application of ultraviolet radiation along with the ozone can accelerate otherwise sluggish reaction rates significantly. Acceleration is brought about by catalytic formation of hydroxyl free radicals, which are stronger oxidizing agents than ozone itself.

The ozone and the ultraviolet radiation compliment each other, requiring lower doses of ozone and better operation and maintenance of the ultraviolet radiation system.

Perozone

Like photozone, the simultaneous application of hydrogen peroxide along with the ozone can accelerate otherwise sluggish reaction rates significantly. Acceleration is brought about by catalyticformation of hydroxyl free radicals, which are stronger oxidizing agents than ozone itself.

This disinfection system also has a lot of potential but needs more research.

Nuclear Radiation

Definitely something to consider in the future. Needs more research!

Microfiltration

Microfiltration does a good job of cleaning a water supply but does not kill viruses or bacteria. It is limited to small flows of water and is generally used with a disinfectant.

Heat

Will destroy all bacteria and viruses but is not practical or economical.

Pure Oxygen

Although not considered as a disinfectant, high levels of pure oxygen can relate to a clean environment and reduce the toxic effects of some bacteria and viruses.

References

Bader, H. and J. Hoigne. 1982. Determination of Ozone in Water by the Indigo Method: A Submitted Standard Method, Ozone: Science and Engineering, 4:169-176.

Baker, B. 1986. Ozonation of Hatchery Water Supply at Coleman National Fish Hatchery, Proceedings of the Northwest Fish Culture Conference, Eugene, Oregon.

Bedell, J. 1971. Eradicating Ceratomyxa shasta from Infected Water by Chlorination and Ultraviolet Radiation, Prog. Fish Cult., 33:51-54.

Blake, M. 1988. Aquionics Aquaculture, Technical Report 1988.

Erlicher, H. 1964. Zentr. Arbeitsmed., Arbeitsschutz, 14:260.

Haller, J.F., Northgraves, W.W. 1955, Tappi, 38:199.

Masschelein, W.J. 1979. Chlorine Dioxide: Chemistry and Environmental Impact of Oxychlorine Compounds (Ann Arbor, MI: Ann Arbor Science Publishers, Inc., 1979).

Montgomery Engineers. 1987. Ozone Removal Methods and Review of the Design Concept, Corps of Engineers, Information Report, Walla Walla, Washington.

Montgomery Engineers. 1991. Cold Lake Fish Hatchery Water Treatment Modifications Conceptual Design Report.

Morrison, T. J. 1977. A Pilot Plant Trial for Ozone Sterilization of Fish Hatchery Water, Master's Thesis, University of Idaho, Moscow.

National Academy of Sciences. 1987. Drinking Water and Health, Disinfectants and Disinfectant By-Products, Volume 7, (Washington, DC: National Academy Press.

Oakes, D. P. Cooley and L. Edwards. 1978. Ozone Disinfection of Make-Up and Recycle Water at Dworshak National Fish Hatchery, University of Idaho, Chemical Engineering Department, Moscow.

Owsley, D. E. 1979. Packed Columns for Nitrogen Gas Removal, Bioengineering Symposium for Fish Culture, Traverse City, Michigan.

Owsley, D. E. 1984. Operation of the Ozone Pilot System at Dworshak National Fish Hatchery, Proceedings of the Northwest Fish Culture Conference, Tacoma, Washington.

Paulin, A. T. 1967. Procedures for Measuring Ozone in Water, J. Institute Water Engineering, 21:537.

Rosenthal, H. and G. Kruner. 1985. Treatment Efficiency of an Improved Ozonation Unit Applied to Fish Culture Situations, Ozone: Science and Engineering, 7:179-189.

Schutte, A. R. 1986. Evaluation of the Aquatector, An Aeration System for Intensive Fish Culture, U.S. Fish and Wildlife Service Information Leaflet 42, Bozeman, Montana.

Sverdrup and Parcel Engineering. 1986. An Evaluation of Alternative Water Supply Systems, U.S. Fish and Wildlife Service, Information Report, Portland, Oregon.

Tipping, J. M. and K. B. Kral. 1985. Evaluation of a Pilot Ozone System to Control Ceratomyxa shasta at the Cowlitz Trout Hatchery, Washington State Game Department Bulletin, 85-18, Olympia.

U.S. Environmental Protection Agency. 1981. Technologies and Costs for the Removal of Trihalomethanes from Drinking Water, Washington, DC, Science and Technology Branch, Criteria and Standards Division, Office of Drinking Water.

Wedemeyer, G. A., N. C. Nelson, and C. A. Smith. 1977. Survival of Two Bacterial Pathogens Aeromonas salmonicida and the Enteric Redmouth Bacterium in Ozonated, Chlorinated, and Untreated Waters, J. Fish. Res. Board Can., 34:429-432.

Wedemeyer, G. A., N. C. Nelson, and C. A. Smith. 1978. Survival of the Salmonid Viruses Infectious Hematopoietic necrosis (IHNV) and Infectious Pancreatic necrosis (IPN) in Ozonated, Chlorinated and Untreated Waters, J. Fish. Res. Board Can., 35-875-879.

Wedemeyer, G. A., N. C. Nelson, and W. T. Yasutake. 1979. Potentials and Limits for the Use of Ozone as a Fish Disease Control Agent, Ozone: Science and Engineering, 1:295-318.

Williams, R. C., S. G. Hughes, and G. L. Rumsey. 1982. Use of Ozone in a Water Reuse System for Salmonids, Prog. Fish-Cult., 44:102-105.

An Application Model Of Foam Fractionators Used In Aquaculture

M.B. Timmons
Department of Agricultural and Biological Engineering
Cornell University
Ithaca, NY 14850;

S. Chen
Department of Civil Engineering
Louisiana State University
Baton Rouge, LA 70803-6405

N.C. Weeks
Steams and Wheler Environmental Engineers
Cazenovia, NY 13035.

Abstract

Closed cycle aquaculture production is subject to an accumulation of suspended solids (<100 mm) and dissolved organics. Of particular concern are solids in the 5 to 10 mm size, since these solids have been identified as causing mortality and poor performance due to gill infections in salmonids, and are not easily removed by conventional methods of screening. Foam fractionators have been used with some success to remove fine suspended solids and excessive nutrient concentrations. Foam from fractionator columns needs to be concentrated to prevent excessive water loss. A mathematical model is developed that can be used to predict volatile solids removal rates and fine suspended particles from a typically configured foam fractionator that uses an inverted funnel to concentrate foam. The mathematical model includes the effects of superficial gas velocity, particle diameter, bubble diameter, protein concentration, pH, gas void ratio, column liquid flow rate and foam over flow height as well as column geometric variables such as diameter and submergence depth.

Intensive aquaculture is contributing an increasing amount of the annual fish and seafood being marketed in the USA and world markets. Lack of suitable water supplies and more stringent control of waste and nutrient discharge from hatchery facilities accentuates the demand for closed system production units that can be used to produce fingerlings for growout facilities or food fish directly on an economically competitive basis.

Closed cycle aquaculture production systems are usually defined as exchanging 10% or less of the system's water volume on a daily basis. Operating a closed-cycle system under economically competitive conditions, e.g., one pound or greater of fish per gallon of water and daily feeding rates that are 1 to 3% of the total fish biomass, presents significant engineering challenges to maintain water quality conditions that can sustain high fish productivity. Typically closed production units are subject to an accumulation of suspended solids (<100 µm) and dissolved organics (protein). Of particular concern are solids in the 5 to 10 µm size, since these solids have been identified as causing mortality and poor performance because of gill infections in salmonids (Chapman et al., 1987). Others have reported on the adverse effects of solids on fish health and gill damage (Stickney, 1979; Wickins, 1980). Particles >100 µm are settleable (Rudolfs and Balmat, 1952) and because of their size are not generally viewed as being a primary object of removal by foam fractionation.

Foam fractionation has been used with some success to remove fine suspended solids and excessive nutrients. Lomax (1976) compared fish culture systems that used a biofilter in combination with either a sedimentation tank, foam fractionation units or mechanical filters. Lomax's opinion was that in terms of cost and effectiveness, the biolfilter with fractionation was the best design combination. Lawson (1978) analyzed the effects of flow rate, geometry and aeration rates on fractionator performance, but used Triton X-100 as the water contaminant as opposed to using fish culture water which could greatly affect fractionator performance. Kown (1971) reported that increasing fractionator column diameter decreased solids removal rates. There has been considerable mathematical modelling of the fractionation process (Wilson et al., 1976; Lemlich, 1966; Sastry and Fuerstenau, 1970). Generally, the models take the form of a model presented by Dwivedy (1973):

$$C(t) = C_o \exp(-b\,t) \qquad (1)$$

where

C = concentration of pollutant, mg/L
C_o = initial concentration of pollutant (t = 0), mg/L
t = time, s
b = closure constant, 1/s

Unfortunately, these studies are restrictive to particular applications, with none of the studies to date

specifically using waste waters that could be considered typical of closed cycle aquaculture systems. Also, the models have failed to separate the effects of foam fractionator operational and/or design parameters upon fractionator performance.

The objective of this paper is to develop a predictive model of a practical foam fractionator design.

Materials and Methods

Mathematical Analysis

Chen (1991) developed a mathematical model which described dissolved solids (C_d) and fine suspended solids (C_p) removal rates as affected by both design and operating parameters. Dissolved proteins were identified as the primary surfactant material in the fractionation process and the surfactant proteins constituted 11% (SE = 2.5%) of the total protein levels in the fish waste water. Without surfactant material, foam fractionation ceases. Removal of fine solids (C_p) cannot occur without concomitant removal of dissolved proteins (C_d). Thus, although uncoupled equations were presented to describe removal rates of C_d and C_p, the processes are in fact coupled. Chen (loc cit) modelled the foam fractionation process in two parts, dissolved nutrient and suspended particle transfer to air bubbles as they traveled from the air stone up through the column length.

A typical fractionator design is depicted in Figure 1. Simply stated, the model developed by Chen (loc cit) states that the loss of solids from the liquid is equal to the solids accumulated on the air bubbles, one being the negative of the other:

$$dC_l/dt = -R_l \quad (2a)$$

$$dC_g/dt = R_g \quad (2b)$$

where

C_l = concentration of solids in liquid, mg/L
R_l = removal rate of solids from liquid, mg/L per sec
C_g = concentration of solids in gas, mg/L
R_g = removal rate of solids from gas, mg/L per sec

Chen (loc cit) divided the solids removal rates into dissolved and suspended fine solids:

$$R_l = R_{l,d} + R_{l,d} \quad (3a)$$

$$R_g = R_{g,d} + R_{g,p} \quad (3b)$$

where

$R_{l,d}$ = dissolved solids removal rate from liquid, mg/L per sec
$R_{l,p}$ = suspended solids removal rate from liquid, mg/L per sec
$R_{g,d}$ = dissolved solids removal rate from gas, mg/L per sec
$R_{g,p}$ = suspended solids removal rate from gas, mg/L per sec

Chen (loc cit) developed mathematical expressions for Equations 3a,b by assuming that gradients over a short distance in the fractionator in the direction of flow (z) could be neglected (high mixing rates), and that net water flow over small increments of z was zero:

$$R_{l,d} = d(C_d)/dt = -1.59\,(C_d - k\,T_d)D^{0.5}\,U_g / \{(1-E_g)U_b^{0.5}B_r^{0.5}\} \quad (4)$$

$$R_{l,p} = d(C_p)/dt = -S\,(C_p)U_g\{P_r/(B_r^2\,(1-E_g)\,U_b)\} \quad (5)$$

$$R_{g,d} = d(T_d)/dt = 0.53(C_d - k\,T_d)D^{0.5}\,(U_g)/(E_g\,U_b^{0.5}B_r^{0.5}) \quad (6)$$

$$R_{g,p} = d(T_p)/dt = 0.33(S\,C_p\,U_g)\{P_r/(B_r\,E_g\,U_b)\} \quad (7)$$

where

C_d = dissolved solids concentration in bulk solution, mg/L
C_p = fine solids concentration in bulk solution, mg/L
T_p = surface concentration of fine solids, g/m^2
k = adsorption coefficient, 1/m
T_d = surface concentration of dissolved solids (surfactants), g/m^2
D = dissolved solids (surfactants) diffusion coefficient, m^2/s
U_g = superficial gas velocity, m/s
U_b = rising velocity of a single bubble, m/s
B_r = radius of individual gas bubble, m
S = fine suspended particle removal coefficient, m/s
P_r = radius of suspended particles, m
E_g = gas holdup, dimensionless

Superficial gas velocity is defined as the ratio of volumetric air flow through the fractionator column and the cross sectional area of the column:

$$U_g = Q/A \quad (8)$$

where

U_g = superficial gas velocity, m/s
Q = air flow rate through column, m^3/s
A = cross sectional area of fractionator column, m^2

Gas holdup is the ratio of fractionator column height without aeration to height with aeration (typical values are around 0.1 to 0.2).

Equation 4 describes the product of the flux of dissolved solids from the column liquid onto a bubble surface and the flux of gas bubbles through a cross

section of the fractionator column. The constant in Equation 4 is the reduced value of several geometric terms describing area and volume, i.e., it is not an empirical constant. Equation 5 represents the area of bubbles flowing through the fractionator column and their impact with suspended fine solids during the travel through the column, with removal rates being predicted based upon the collection efficiency term, S. Chen neglects diffusion of solid particles to the bubble surface, i.e. Chen models impact only. Reay and Ratcliff (1973) reported that for particles > 3-4 micron, bubble collection of solids is governed by impact.

Equation 7 is never used, since Chen has assumed particle loads on bubbles do not affect removal rates of particles from the bulk liquid (Equation 5). Equation 6 (the constant 0.53 is the result of the same geometric constants in Equation 4, except it is divided by three to convert a surface concentration, T_d, to a volumetric concentration of the gas bubble) is solved for T_d in terms of C_d and then substituted back into Equation 4. The final set of equations are as follows (see Chen, loc cit, for a complete mathematical derivation):

$$d(C_d)/dt = -1.59(C_d)(D^{0.5})(U_g)\exp\{1.59\ k\ (D^{0.5})(U_g)(H)/[3(E_g)(U_b^{1.5})(B_r^{0.5})]\}/[(1-E_g)(U_b^{0.5})(B_r^{1.5})] \quad (9)$$

$$d(C_p)/dt = -(S)(C_p)(U_g)(P_r)/[(B_r^2)(1-E_g)(U_b)] \quad (10)$$

where

H = column length, m

Equations 9 and 10 can both be solved easily and have the form:

$$C(t) = C_o \exp(C - bt) \quad (11)$$

Chen (loc cit) experimentally determined the unknown coefficients in Equations 9 and 10 as:

D = 1.11 E-12 m^2/s (standard deviation of 0.71E-12, 9 test runs)

k = 1.03 E+4 , 1/m

S = 3.88E-3 (standard deviation of 1.20 E-3)

(*$k=1/k_e$: k_e is the coefficient (units of length) relating $T_{d,e}$, which is the adsorption isotherm of protein on a bubble surface, and C_d, the bulk solution concentration of dissolved proteins; Chen (loc cit) presented the following regression of data to relate C_d and $T_{d,e}$ for bubble surface concentrations after bubbles have travelled through a 2 meter high column filled with bulk solution: $T_{d,e} = (k_e)(C_d)$ where K_e = 9.7 E-5 (r^2 of 0.84, SE of coefficient of 9.9 E-6, DOF (Degree of Freedom) of 16).

Both Equations 9 and 10 are first order. Chen (loc cit) verified that both dissolved and suspended particles behaved in a first order manner in a series of experiments operated in batch mode at various concentrations for both variables of interest.

Note that Equations 9 and 10 contain directly or indirectly most of the operating parameters involved in using a foam fractionator. These equations omit the effects of the collection process, and only model the bubble enrichment process as bubbles travel from an air diffuser up through the fractionator water column to the air/water interface. Thus, predicting actual operation of a practical foam fractionation design is still not possible.

Using Equation 10 requires a characterization of the size of the fine suspended particles contained in the bulk solution. Chen et al. (1992) analyzed solids from several water reuse systems and found that the dominant size of particles by number were 8 to11 micron. This value of course could be assigned differently to the model, if the bulk solution is known to be different.

Development of an Applied Model

Modelling the removal rate of suspended particles also produces some redundancy in modelling protein removal rates, since suspended particles are in part made up of protein materials. Weeks et al. (1992) reported that suspended solids were approximately 50% protein and that the condensate enrichment resulted almost entirely from collection of volatile solids; fixed solids remained the same between fish culture water and foam condensate, although the foam condensate increased in total solids by approximately 600 mg/L, from 900 mg/L to 1,500-1,600 mg/L.

Unfortunately, Equations 9 and 10 cannot be applied directly since they model production rates at the water air interface at the top of the column. Some means are generally used to concentrate the foam in order to minimize water losses. Weeks et al. (1992) presented a foam fractionation design (see Figure 2) that has been applied in commercial settings. As can be seen in the figure, an inverted 60° funnel is placed at the top of the column to concentrate the foam and to utilize the velocity of the escaping air bubbles to push the foam on through the funnel for condensate storage or disposal.

Weeks et al. (loc cit) found that only a fairly narrow range of overflow heights could be used in practical applications (placing the mouth of the funnel at the air-water interface was defined as a zero height) for the 15 cm diameter column design studied. Surprisingly, using zero overflow heights resulted in condensate that was only 60% higher in volatile solids, although production rates of foam were high, and an 8 cm overflow height resulted in a 270% increase in volatile solids concentration but very low levels of foam production (mL/min). The ratio of solids removal rates between the zero overflow and the 8 cm height was 2.88. Clearly, this implies management options are available as to how best to operate a foam fractionator, which would depend upon what is desired to be

accomplished, e.g. minimizing effluent or maximizing solids removal.

Variables Affecting Removal Rates

Gas Holdup (E_g)

Foam fractionator performance is dependent upon gas hold up, which is the fractional increase in column liquid height due to aeration. The increase is obviously related to the amount of aeration. Chen (loc cit) obtained the following relationship between E_g and U_g:

$$E_g = 4.1\ U_g^{0.83} \qquad (11)$$

(original regression equation was: $\ln(E_g) = 1.41 + 0.83 \ln(U_g)$; r^2=0.98, SE coeff = 0.15, N = 9)

Equation 11 was used in the application model of this paper to describe E_g as a function of U_g.

Bubble Rising Velocity

Equations 9 and 10 employ the velocity associated with a single bubble rising in a column. A fractionator operates using a bubble swarm. The decreased density difference between the bubble and the surrounding fluid decreases the buoyancy force, which slows the velocity of the bubbles in the bubble swarm. Chen (loc cit) video taped rising bubbles in a column with a bulk fluid dissolved protein concentration of 120 mg/L. His data are given in Table 1. Data in Table 1 was regressed to give the following expression:

$$\ln(U_b) = -1.55 - 25.8\ (U_g) \qquad (12a)$$

(r^2 = 0.97, SE coeff = 3.12, N=4)

Equation 12 can be more conveniently expressed by taking the exponential of both sides of the equation:

$$U_b = 0.21 \exp(-25.8\ U_g) \qquad (12b)$$

Shah et al. (1982) presented a simpler form to predict bubble rising velocity for bubble swarms:

$$U_b = U_g/E_g \qquad (13)$$

Table 2 gives a comparison of the predicted bubble rising velocities using Equation 12 or 13. As can be seen from the comparison between the two model predictions in Table 2, Shah's method (Equ. 13) predicts the opposite effect as observed from the Chen's data and as described by the regression equation, Equ. 12. Therefore, Equation 12 will be used in the application model for this paper to predict foam fractionator performance.

A practical range exists for Ug since continued increases in U_g will eventually result in the formation of Taylor bubbles or gas slugs (thus not suited for fractionation). Reinemann and Timmons (1989) reported that gas void ratios (E_g) above 25% will begin to form slug flow in tap water. It would be expected that due to higher surface tension effects from dissolved solids in the fish culture water, the E_g value could be extended and still maintain bubbly flows.

Bubble Size

Chen (loc cit) presented data (Table 3) which can be used to predict bubble size as affected by protein concentration (PC) and superficial air velocity (U_g). Neither Equation 12 or 13 had included the effects of protein concentration.

A regression of the data given in Table 3 provides the following relationship to describe the effects of superficial gas velocity and protein concentration on bubble diameter, B_d:

$$Bd(mm) = 2.58 + 28.1\ U_g(m/s) - 0.0098\ (PC, mg/L) \qquad (14)$$

(use only for protein concentrations up to 137 mg/L)

(r^2 = 0.82, SE coefficients = 5.4 and 0.0013, DOF = 17)

Since equation Equation 12 predicts bubble rising velocity at a PC of 120 mg/L, Equation 14 could be used to adjust the rising velocity based upon change in bubble diameter. Roughly, the bubble rising velocity will change proportionally to the change in bubble diameter, since the rising velocity is proportional to the volume of the bubble (buoyancy effect) and inversely proportional to the drag area of the bubble—or B_d^3/B_d^2.

Effects of pH

The pH of the bulk solution has been seen to affect the foam fractionation process (Grieves, 1972). Chen (loc cit) analyzed the effects of pH upon the equilibrium concentration of dissolved proteins on the bubble surface (Table 4) as reflected in the adsorption equilibrium coefficient, k_a. In applying the application model, pH effects can be included by using a more specific k_a value in Equation 9 rather than the average ka value as shown in Table 4. A regression of the data in Table 4 (trivial since only 3 data points) is:

$$k_a = -3.50E\text{-}5 + 1.68E\text{-}5(pH) \qquad (15)$$

(r^2 = 0.86, SE coeff = 6.7xE-6, DOF = 1)

Note that Equation 9 uses the inverse of k_a, 10,300 m.

Closure of the Application Model

Chen's (loc cit) model was specifically developed to model removal rates of proteins. In fact, Chen argued that the dissolved solids removed via fractionation were exclusively proteins. Recent data pre-

sented by Weeks et al. (1992) indicated that the condensate solids were a combination of dissolved proteins and other solids, and could not be strictly accounted for by proteins alone. Weeks et al. (loc cit) reported the volatile solids removal rates, foam-condensate production rates and condensate solids characteristics for a variety of operating conditions and foam overflow heights (Table 5 provides the physical characteristics and operating conditions of the fractionator design used, and the performance data is presented in Table 6).

An application model was developed from the data in Tables 5 and 6 by using Equations 9 and 10 to predict the dissolved and fine particle removal rates and then developing a correction factor to account for the collection efficiency of the foam fractionator design (simply the ratio of the measured to the predicted amount of solids removed). Since Chen's models were strictly developed for batch operation, data given by Weeks et al. (loc cit) was expressed as removal rates per residence time of fluid in the column (volume of column divided by water flow rate through column). The removal rate data reported by Weeks et al. (loc cit) was for a concentration of total suspended solids in the bulk fluid of 10.1 mg/L. The time of residence as calculated from the Weeks et al.'s data was substituted into Chen's exponential equations to calculate the removal rates per unit residence time of fluid in the foam fractionator column.

Results and Discussion

A multiple regression was performed using the data in Table 6 to develop correction terms to reflect the collection efficiency associated with a concentrating funnel placed on top of a foam fractionator column. The measured condensate rates of removal were regressed with the predicted data (Equations 9 and 10) using the operating conditions of overflow height (H_t) and superficial gas velocity (U_g) as independent variables. The correction factors for the dissolved solids and the suspended fine particles are referred to as B_d and B_p, respectively, and are as follows:

$$B_d = 0.012 - 0.00076\,(H_t) - 0.086\,(U_g) \qquad (16)$$

($r^2 = 0.56$, SE coeffs = 0.00015, 0.028 , DOF = 51)

$$B_p = 0.19 - 0.017\,(H_t) \qquad (17)$$

($r^2 = 0.49$, SE coeffs = 0.0024, DOF = 52)

where

H_t = distance from top of liquid to bottom edge of funnel, cm.

Including an interaction variable of overflow height and superficial air velocity did not improve the regression for B_d or B_p. The superficial gas velocity was not statistically significant (at the 10% level or smaller) in the regression for B_p.

Application of Models

Equations 9 and 10 can be applied to predict the performance of a typically configured foam fractionator. These results must be viewed in the context that the collection efficiency has not been included. Thus, the figures will show removal rates as if there was 100% collection of solids at the top of the fractionator column, and will therefore be used only to illustrate relative effects. The effects of bubble size (Figure 3) can be shown by applying Equations 9 and 10 directly (assumes bubble size is not a function of the other operating parameters). The control of bubble size is impractical when glass bonded diffusers are used, since all bubbles will be within a narrow range (usually from 2.5 to 3.5 mm diameter bubbles), even when different diffuser types are used (Chen, loc cit). Thus, Equations 9 and 10 can express bubble size as a function of superficial gas velocity and the protein concentration of the bulk fluid (Equation 14). After making this substitution, Equations 9 and 10 are a function of only one operational parameter, superficial gas velocity. Effects of superficial gas velocity (where B_r, E_g, U_b are all expressed as a function of U_g) are shown in Figures 4-7. All results are shown as the ratio of current concentration (at time t=10, 30, or 60 sec) to initial concentration (t=0) of either dissolved solids (Equation 9) or fine particles (Equation 10).

Actual performance of foam fractionators can be predicted by including correction terms for collection efficiency as developed in the preceding section. The following is an example to demonstrate the application of the equations presented in this paper to predict solids removal rates from a foam fractionator where a funnel is used to concentrate foam.

Problem Statement

Calculate the initial removal rates of volatile solids (VS) for a fractionator that is 1.0 m in height, 10 cm in diameter, and is operated with a 20 L/min bulk fluid flow rate. The fractionator is as described by Figure 1 (glass bonded diffuser with inverted funnel to trap and concentrate foam). The funnel is set for a 4 cm overflow height (bottom edge of funnel is 4 cm above the water level in the column) and the concentration of VS in the fish tank is 300 mg/L with a total protein concentration (PC) of 50 mg/L and a pH of 8.0. Assume the fine suspended solids average 10 micron in diameter (considered typical). Calculate the solids removal rate from a single fractionator column.

Solution: $\Delta VS = \Delta C_d + \Delta C_p$

Assume that ΔC_d that can be predicted based upon the volatile solids concentration (VS) and ΔC_p can be predicted from the concentration of total sus-

pended solids (TSS). Foam production rates are a function of U_g, E_g, U, D_b:

U_g(m/s, Equ. 8) = Q/A=(20 L per min/(π 0.052)){min/60 s}{m^3/1,000 L}
= 0.04 m/s

E_g(dimensionless, Equ. 11) = 4.1 $(U_g)^{0.83}$
= 4,1 $(0.04)^{0.83}$
= 0.28

U (m/s, Equ. 12b)* = 0.21 exp(-25.8 U_g)
= 0.21 exp(-25.8 x 0.04)
=0.07 m/s

*Assumes U_b is equal to U)

B_d (mm, Equ. 14) = 2.58 + 28.1(0.04 m/s)-0.0098 (50 mg/L)
= 3.2 mm (B_f=B_d/2=3.2/2=1.6 mm or 0.0016 m)

k_s (m, Equ. 15) = -3.50E-5 +1.68E-5(8)
= 9.94E-5 or k = 1/k_s = 10,060

Other values needed to use Equations 9 and 10:

$1.59xD^{0.5}$ = 1.7 E-6; $1.59D^{0.5}$ x k/3 = 5.7 E-3

Substituting the above values into Equation 9 (where $C_{d,o}$ is the initial concentration of C_d):

$d(C_d)/dt$ = -$C_{d,o}$[(1.7E-3 U_g)/[(1-E_g)($U^{0.5}B_f^{1.5}$)]]x exp{(5.7E-3 U_g H)/(E_g $U^{1.5}$ $B_f^{0.5}$)}

= -$C_{d,o}$(1.76E-6 x 0.04/(1-0.28) x $0.07^{0.5}$ x $0.0016^{0.5}$) x exp{5.7E-3 x 0.04 x 1.00/(0.28 x $0.07^{0.5}$ x $0.0016^{0.5}$)}

= -$C_{d,o}$(5.77E-3) x exp(1.10)
= -$C_{d,o}$(1.73E-2)

Solving: $C_d(t)/C_{d,o}$ = exp(-1.73E-2 t)

The model has been developed to represent solids removal rates per unit residence time (time required for an amount of bulk liquid to pass through a column equal to the column volume). Calculate residence time:

t (s) = Volume/flow rate
= (1 m)(π x 0.05^2m^2)/{(20L/min)(min/60 s)(m^3/1,000 L)}
= 24 s

Substitute t = 24 s into exponential:

C_d(t=24 s)/$C_{d,o}$ = exp(-1.73 x 24) = 0.66
or removed C_d is 1.0-0.66 = 0.34 $C_{d,o}$

The removal must be corrected for collection efficiency:

B_d (Equ. 16) = 0.012 -0.00076 (4 cm)
-0.086 (0.04 m/s)
= 0.0055

The dissolved volatile solids removed per retention time can now be calculated using an initial concentration of volatile solids of 300 mg/L:

ΔC_d x B_d = 0.34 ::300 mg/L x 0.0055 x 20 L/min x (min/60 s) x 24 s

= 4.6 mg/retention time (0.19 mg/L per second) VS.

Now, calculate the fine solids removal rate (radius of particles is 0.000005 m and where $C_{p,o}$ is the initial concentration of C_p):

$d(C_p)/dt$ (Equ. 10) = -$C_{p,o}$(3.88E-3)(0.04 m/s)
(0.000005 m)/(0.00162 m^2)(1-0.28)(0.07 m/s)

= -$C_{p,o}$ (6.01E-3)

Solving the last differential equation:

$C_p/C_{p,o}$ = exp{(-6.01E-3)(t = 24 s)}
= 0.87 or removed C_p is 1-0.87 = 0.13 $C_{p,o}$

The removal rate must be corrected for collection efficiency:

B_p (Equ. 17) = 0.19 - 0.017 (4 cm)
= 0.12

The fine suspended solids removed per retention time can now be calculated using an initial concentration of fine solids of 10 mg/L:

ΔC_p x B_p = 0.13 x 10 mg/L x 0.12 x 20 L/min x (min/60 s) x 24 s

= 1.25 mg/retention time (0.052 mg/L per second) of TSS.

If the concentration in the bulk solution remained at 300 mg/L VS and 10 mg/L TSS (steady state condition where production rate of VS and TSS was balanced by the removal rates from the foam fractionators), then the 24 hour removal rates would be:

$\Delta C_{d24\ hr}$ = 0.19 mg/L/s (60 s/min)(60 min/hr)(24 hr/day)
= 16,400 mg/day or 16.4 g/day

$\Delta C_{p24\ hr}$ = 0.052 mg/L/s (60 s/min)(60 min/hr)(24 hr/day)
= 4,500 mg/day or 4.5 g/day

As a "rule of thumb" check of these predictions, Weeks et al. (1992) reported the average TVS concentrations in foam condensate from all operating conditions analyzed as 872 mg/L. Using this condensate concentration and the previously calculated $\Delta C_{d24\,hr}$, the volume of condensate collected in a 24 hour period from one 10 cm diameter column would be estimated as :

$$\text{Volume Condensate} = (16{,}400\ \text{mg/L}) / (872\ \text{mg/L}) = 18.8\ \text{L}$$

This value is consistent with the author's observations of fractionator performance over the last several years. Other observations include erratic performance from fractionator units. Lack of consistency is attributed in large to anti-surfactant properties of fish feeds, e.g. oil top dressings used on pelleted feeds. Predictions using the developed equations in this paper should be used as estimates of performance.

Literature Cited

Chapman, P.E., Popharnn, J.D., Griffin, J., and J. Michaelson. 1987. Differentiation of physical from chemical toxicity in solid waste fish bioassay. Water, Air, and Soil Pollution, 33:295-308.

Chen, S. 1991. Theoretical and experimental investigation of foam separation applied to aquaculture. PhD Thesis. Cornell University, Ithaca, NY, USA.

Chen, S., Timmons, M.B., Bisogni, J.J. Bisogni, and D.J. Aneshansley. 1993. Suspended solids characteristics from recirculating systems and design implications. Aquaculture 112: 143-155.

Dwivedy, R.C. 1973. Removal of dissolved organics through foam fractionation in closed cycle systems for oyster production. ASAE Paper No. 73-561, ASAE, St. Joseph, MI.

Grieves, R.B. 1972. Foam fractionation of surfactants, orthophosphate, and phenol. In Adsorptive Bubble Separation Techniques, Lemlich, R. (editor), Academic Press, New York.

Kown, B. 1971. The effects of column height and diameter on the effectiveness of a continuous bubble fractionation system. Water Research 5:93-102.

Lawson, T.B. 1978. Venturi design parameters for air injection into a foam fractionation system. PhD dissertation. The University of Maryland, College Park, MD, USA.

Lemlich, R. 1966. A theoretical approach to nonfoaming adsorption bubble fractionation. AIchE Journal, 12(4):802-804.

Lomax, K.M. 1976. Nitrification with water pretreatment on a closed cycle catfish culture system. Unpublished PhD Thesis, University of Maryland, College Park, MD.

Reay, D and G.A. Ratcliff, 1973. Removal of fine particles from water by dispersed air flotation: effects of bubble size and particle size on collection efficiency. Can. J. Chem. Eng., 51:178-185.

Reinemann, D. J. and M.B. Timmons. 1989. Prediction of oxygen transfer and total dissolved gas pressure in airlift pumping. Aquacultural Engineering 8:29-46.

Rudolfs, W. and J.L. Balmat. 1952. Sewage Ind. Wastes, 24(3):247-256.

Sastry, K.V.S. and D.W. Fuerstenau, 1970. Theoretical analysis of a counter current flotation column. Trans. of Soc. Mining Engineers., AIME, 247:47-52.

Shah, Y.T., Kelkar, B.G., Godbole, S.P. and W.D. Deckwer. 1982. Design parameters estimations for bubble column reactors. AIhE Journal, 28(3):353-379.

Stickney, R.R. 1979. Principles of Warm Water Aquaculture. Wiley Interscience, New York, 375 pp.

Weeks, N.C., Timmons, M.B. and S. Chen. 1992. Feasibility of using foam fractionation for the removal of dissolved and suspended solids from fish culture water. Aquacultural Engineering 11: 251-265.

Wickins, J.F. 1980. Water quality requirements for intensive aquaculture: a review, Symposium on new developments in the utilization of heated effluents and recirculation systems or intensive aquaculture. EIFAC, 11th Session, Stavanager, Norway, 28-30 May.

Wilson, J.W., Wilson, D.J., and J.H. Clarke. 1976. Electrical aspects of adsorbing colloid flotation IV: stripping column operation. Separation Science, 11(3):223-239.

Table 1. Measured single bubble rising velocity as part of a bubble swarm for various superficial air velocities in a fractionator column (protein concentration in bulk fluid of 120 mg/L; Chen, 1991)

U_g(m/s)	U(m/s)
0.0011	0.22
0.0093	0.16
0.022	0.11
0.038	0.085

Table 2. Predicted bubble rising velocity (Ub) using data from Chen (1991) versus prediction given by Shah et al. (1982)

	U_b (m/s)		
U_g(m/s)	E_g (Equ 11)	Chen (Equ. 12)	Shah (Equ. 13)
0.005	0.06	0.18	0.09
0.01	0.10	0.16	0.10
0.02	0.17	0.12	0.11
0.04	0.30	0.07	0.13
0.06	0.42	0.04	0.14

Table 3. Changes in measured bubble diameter (Bd, mm) at various superficial air velocities (Ug) and protein concentrations (PC) (from Chen, 1991)

Bubble Diameter, mm Protein Concentration, mg/L				U_g (m/s)
0	57	117	137	
2.3	2.3	2.0	1.2	0.0075
2.7	2.6	2.2	1.4	0.015
3.2	2.8	2.6	1.6	0.022
3.7	3.2	2.75	1.8	0.034
3.8	3.4	2.7	2.0	0.045

Table 4. Regression coefficients developed by Chen (1991) to describe the effects of pH on the protein adsorption equilibrium coefficient (k_a).

pH	k_a (m)
5.3	5.78xE-5
7.4	7.80xE-5
8.3	11.3xE-5
avg	9.7xE-5

Table 5. Operating and design characteristics of the foam fractionators used by Weeks et al. (1992)

Piper diameter	0.152 m
Submergence depth	0.69 m
Airflow rates	33.0, 37.8,42.5,51.9, 66.1 L/min (expressed later as U_g m/s)
Water flow rates	11.4, 22.7, 34.1 L/min
Overflow heights	0,3,4,5,7, 8 cm

Table 6. Effects of superficial gas velocity (U_g, m/s) and overflow height (H_f, cm) on foam fractionator performance in terms of volatile solids removal rates (VSSR, mg/min) and dissolved and fine particle removal rates per unit residence time (ΔC_d and ΔC_p) (from Weeks et al., 1992; culture water characteristics: 307 mg/L volatile solids, 625 mg/L fixed solids, 10 mg/L suspended solids and 0.8 mg/L total kjeldahl nitrogen).

Trial #	TotVol/ Batch	B_r (L)	U_b (m)	E_g (m/s)	H_f (cm)	U_g (m/s)
1	22.7	0.0017	0.10	0.22	0	0.030
2	22.7	0.0017	0.10	0.22	4	0.030
3	22.7	0.0017	0.10	0.22	8	0.030
4	22.7	0.0018	0.08	0.28	0	0.039
5	22.7	0.0018	0.08	0.28	4	0.039
6	22.7	0.0018	0.08	0.28	8	0.039
7	22.7	0.0019	0.06	0.33	0	0.048
8	22.7	0.0019	0.06	0.33	4	0.048
9	22.7	0.0019	0.06	0.33	8	0.048
10	34.1	0.0017	0.10	0.22	0	0.030
11	34.1	0.0017	0.10	0.22	4	0.030
12	34.1	0.0017	0.10	0.22	8	0.030
13	34.1	0.0018	0.08	0.28	0	0.039
14	34.1	0.0018	0.08	0.28	4	0.039
15	34.1	0.0018	0.08	0.28	8	0.039
16	34.1	0.0019	0.06	0.33	0	0.048
17	34.1	0.0019	0.06	0.33	4	0.048
18	34.1	0.0019	0.06	0.33	8	0.048
19	11.4	0.0017	0.10	0.22	0	0.030
20	11.4	0.0017	0.10	0.22	4	0.030
21	11.4	0.0017	0.10	0.22	8	0.030
22	11.4	0.0018	0.08	0.28	0	0.039
23	11.4	0.0018	0.08	0.28	4	0.039
24	11.4	0.0018	0.08	0.28	8	0.039
25	11.4	0.0019	0.06	0.33	0	0.048
26	11.4	0.0018	0.06	0.33	4	0.048
27	11.4	0.0019	0.06	0.33	8	0.048
28	22.7	0.0018	0.09	0.25	3	0.035
29	22.7	0.0018	0.09	0.25	5	0.035
30	22.7	0.0018	0.09	0.25	7	0.035
31	22.7	0.0019	0.06	0.33	3	0.048
32	22.7	0.0019	0.06	0.33	5	0.048
33	22.7	0.0019	0.06	0.33	7	0.048
34	22.7	0.0021	0.04	0.40	3	0.060
35	22.7	0.0021	0.04	0.40	5	0.060
36	22.7	0.0021	0.04	0.40	7	0.060
37	22.7	0.0018	0.09	0.25	3	0.035
38	22.7	0.0018	0.09	0.25	5	0.035
39	22.7	0.0018	0.09	0.25	7	0.035
40	22.7	0.0019	0.06	0.33	3	0.048
41	22.7	0.0019	0.06	0.33	5	0.048
42	22.7	0.0019	0.06	0.33	7	0.048
43	22.7	0.0021	0.04	0.40	3	0.060
44	22.7	0.0021	0.04	0.40	5	0.060
45	22.7	0.0021	0.04	0.40	7	0.060
46	22.7	0.0018	0.09	0.25	3	0.035
47	22.7	0.0018	0.09	0.25	5	0.035
48	22.7	0.0018	0.09	0.25	7	0.035
49	22.7	0.0019	0.06	0.33	3	0.048
50	22.7	0.0019	0.06	0.33	5	0.048
51	22.7	0.0019	0.06	0.33	7	0.048
52	22.7	0.0021	0.04	0.40	3	0.060
53	22.7	0.0021	0.04	0.40	5	0.060
54	22.7	0.0021	0.04	0.40	7	0.060

*Vol of liquid treated (volume of bulk fluid passing through column per unit residence time)

Trial #	Retention time (sec)	Prod rate foam (mL/min)	VSRR mg/min	ΔC_d mg/ft*	ΔC_p mg/ft*
1	33.3	28.6	17.3	604	10.9
2	33.3	6.4	8.2	604	10.9
3	33.3	4.0	5.8	604	10.9
4	33.3	57.0	35.1	960	16.3
5	33.3	12.7	10.1	960	16.3
6	33.3	8.7	11.0	960	16.3
7	33.3	107.3	22.1	1503	23.0
8	33.3	22.8	15.0	1503	23.0
9	33.3	15.7	14.4	1503	23.0
10	22.2	30.4	17.7	414	7.4
11	22.2	9.3	8.6	414	7.4
12	22.2	6.3	8.0	414	7.4
13	22.2	77.5	33.3	670	11.1
14	22.2	21.0	18.1	670	11.1
15	22.2	12.4	13.4	670	11.1
16	22.2	140.0	44.5	1081	15.9
17	22.2	30.3	19.0	1081	15.9
18	22.2	19.1	14.8	1081	15.9
19	66.3	9.2	7.6	1111	20.8
20	66.3	9.5	11.3	1111	20.8
21	66.3	3.1	4.5	1111	20.8
22	66.3	44.0	25.4	1681	30.3
23	66.3	8.3	7.7	1681	30.3
24	66.3	5.2	6.7	1681	30.3
25	66.3	57.0	27.0	2428	41.7
26	66.3	15.2	15.1	2428	41.7
27	66.3	8.3	9.5	2428	41.7
28	33.3	11.0	8.9	784	13.7
29	33.3	6.8	7.0	784	13.7
30	33.3	5.7	6.3	784	13.7
31	33.3	29.5	17.2	1503	23.0
32	33.3	19.2	15.6	1503	23.0
33	33.3	14.7	12.2	1503	23.0
34	33.3	80.0	34.2	2652	34.8
35	33.3	33.3	19.5	2652	34.8
36	33.3	22.3	20.5	2652	34.8
37	33.3	10.0	7.8	784	13.7
38	33.3	7.5	7.6	784	13.7
39	33.3	5.6	5.2	784	13.7
40	33.3	22.7	15.6	1503	23.0
41	33.3	15.7	12.3	1503	23.0
42	33.3	12.2	10.3	1503	23.0
43	33.3	41.0	13.3	2652	34.8
44	33.3	13.8	6.2	2652	34.8
45	33.3	18.4	14.2	2652	34.8
46	33.3	9.9	8.4	784	13.7
47	33.3	7.4	8.6	784	13.7
48	33.3	5.7	8.2	784	13.7
49	33.3	19.2	13.9	1503	23.0
50	33.3	10.3	9.8	1503	23.0
51	33.3	10.2	10.1	1503	23.0
52	33.3	42.0	19.9	2652	34.8
53	33.3	45.0	40.7	2652	34.8
54	33.3	19.4	16.0	2652	34.8

*Expressed as mg per unit retention time of bulk fluid through fractionator column.

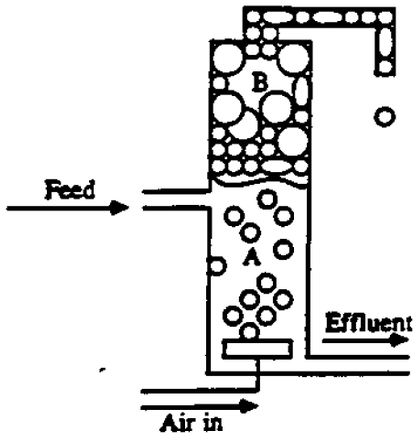


Figure 1. Typical foam fractionator design (from Chen, loc. cit.).

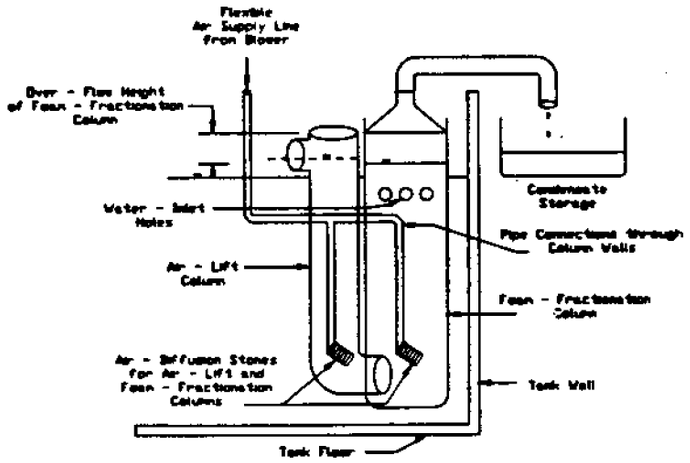


Figure 2. Foam fractionator design used by Weeks et al. (1992).

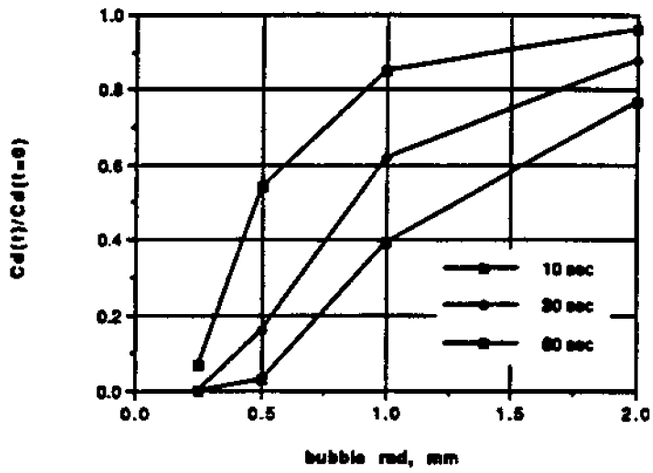


Figure 3. Relative change in dissolved or fine solids concentration with time (Cd(t)/Cd(t=0) or Cp(t) Cp(t=0) as affected by bubble radius (effect is exactly the same) (Ug=0.03 m/s, L=1.0 m, bubble size is not a function of protein concentration or superficial gas velocity).

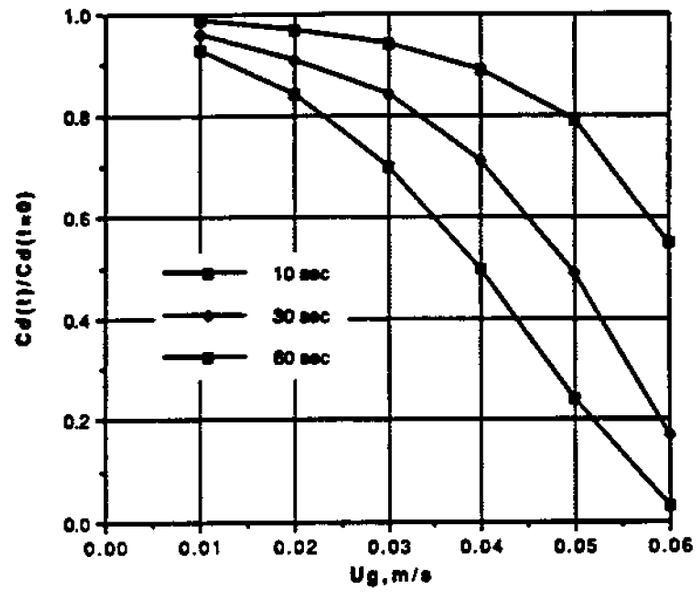


Figure 4. Relative change in dissolved or fine solids concentration with time (Cd(t)/Cd(t=0) or Cp(t) Cp(t=0) as affected by superficial gas velocity (Ug, m/s) and where bubble size is a function of superficial gas velocity and protein concentration (PC) (L=1.0 m, Pr=5 micron, PC=0 mg/L).

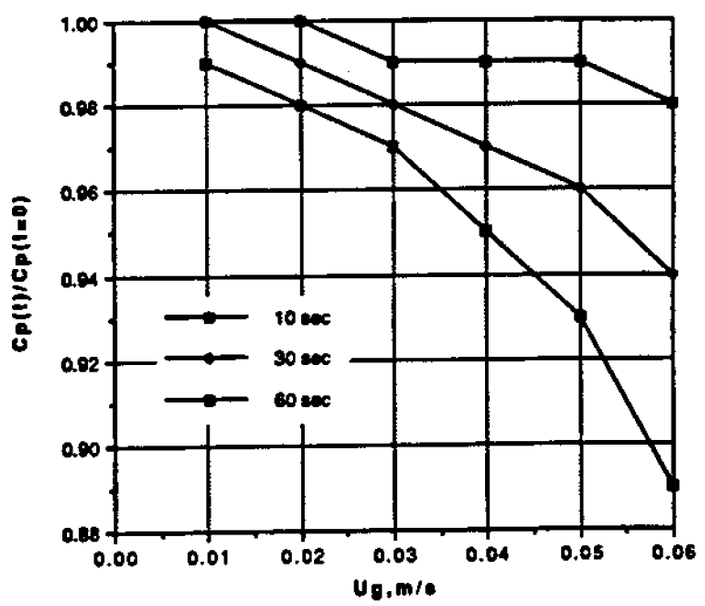


Figure 5. Relative change in dissolved or fine solids concentration with time (Cd(t)/Cd(t=0) or Cp(t) Cp(t=0) as affected by superficial gas velocity (Ug, m/s) and where bubble size is a function of superficial gas velocity and protein concentration (PC) (L=1.0 m, Pr=5 micron, PC=0 mg/L).

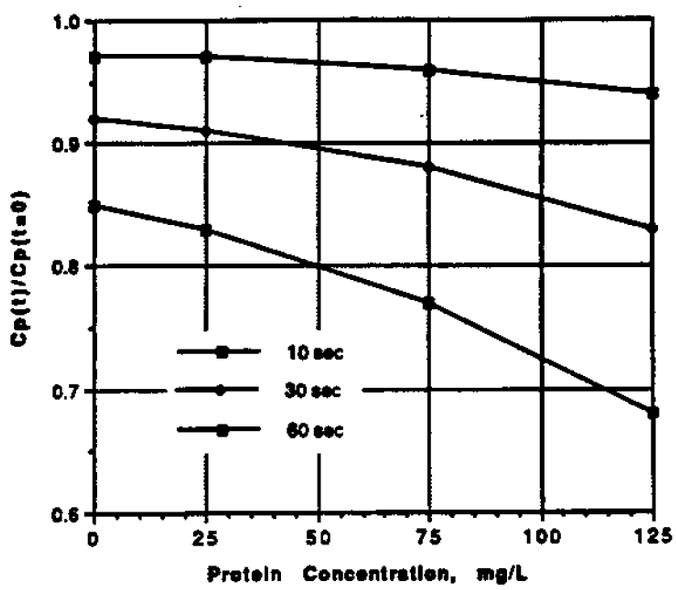


Figure 6. Relative change in dissolved or fine solids concentration with time (Cd(t)/Cd(t=0) or Cp(t) Cp(t=0) as affected by protein concentration (PC) (L=1.0 m, Pr=5 micron, Ug=0.03 m/s) where bubble size is a function of Ug and PC.

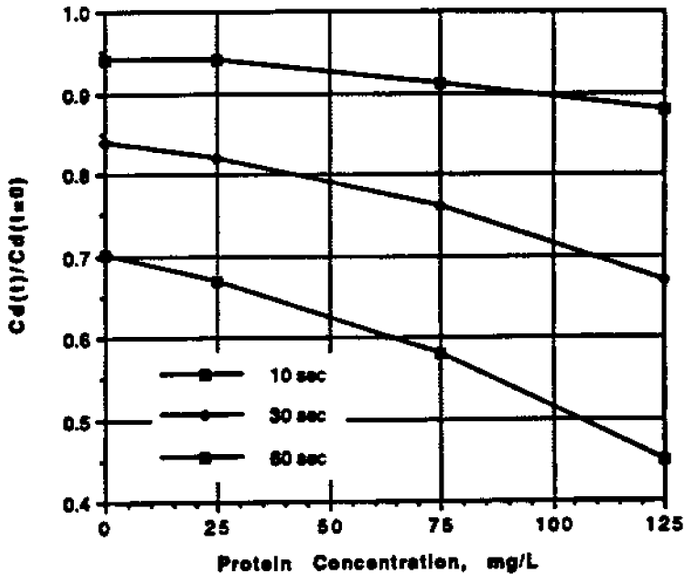


Figure 7. Relative change in dissolved or fine solids concentration with time (Cd(t)/Cd(t=0) or Cp(t) Cp(t=0) as affected by protein concentration (PC) (L=1.0 m, Pr=5 micron, Ug=0.03 m/s) where bubble size is a function of Ug and PC.

Ozone Use in Recirculating Systems: Comparisons With Other Disinfections Options

T.B. Lawson and G. Merry
Louisiana State University
Baton Rouge, Louisiana

ABSTRACT

Ozone has been widely used in the European wastewater treatment and drinking water industries since the early 1900's. For over two decades it has been used by over 60% of European aquarists. Interest in using ozone in recirculating aquaculture systems has been revived recently. Ozone has proven to be effective against a variety of fish and human pathogens at low dosages. Ozone is also is effective at removing turbidity, color, odor and organics from culture water. Water quality does not affect ozone as it does chlorine and UV irradiation. Ozone also does not combine with ammonia as chlorine does. On-site pure oxygen production will double the quantity of ozone generated as compared to air, reducing system and operating costs. Stripping towers remove residual ozone and eliminate the need for detention chambers.

Introduction

Disinfection is the reduction of infectious organisms in aquaculture systems. It differs from sterilization, which is the complete destruction of all living organisms in a system. Disinfection is usually necessary in heavily loaded systems to minimize the transmission of disease organisms and parasites, consisting of bacteria, viruses, protozoans, fungi and worms.

Disinfection is used primarily in hatcheries but is becoming more prevalent in flow-through and recirculating systems. Higher stocking densities, temporary declines in dissolved oxygen, rapid temperature and/or pH changes, and increases in ammonia, nitrite, carbon dioxide or organics cause stresses on the cultured animals, making them more susceptible to infections. Normally harmless organisms may become infectious when environmental factors cause a sudden increase in their numbers. The environment and health of the host determine whether the organisms remain latent or become infectious. Heavy infestations often occur on seemingly healthy fish despite carefully maintained and controlled environmental factors (Spotte 1970). Disinfection in large recirculating systems is expensive, but as discharge regulations tighten and recreational use of natural water increases, recirculating system aquaculture with disinfection becomes more attractive.

The need for control of infectious microorganisms in recirculating systems is clear. Because of dense stocking rates, stress factors and water reuse in recirculating systems, infections can spread rapidly. All things considered, we feel that ozone has the greatest potential for disinfection in recirculating systems. This paper discusses the various options and makes comparisons of ozone to other disinfection methods. Finally, we close our argument with conclusions and recommendations.

Disinfection Options

Basically, four disinfection options are considered feasible for recirculating aquaculture systems: heat, chlorination, ultraviolet (UV) irradiation, and ozone. For various reasons, mostly economical, other disinfection processes have not proven to be feasible. The effectiveness of a given disinfecting agent or method depends on concentration used, contact time, temperature, turbidity, particulate concentration and type of disease organism or parasite (Huguenin and Colt 1989).

Heat

Water in recirculating systems can be sterilized or disinfected with steam or by elevating the water temperature. Small steam generators are commonly used to sterilize equipment and tanks before putting in fish or after a disease incident. Steam is also used to sterilize boots, seines, nets, buckets and other equipment which are avenues for infection.

Culture water can be disinfected by elevating the temperature with a steam boiler, electric or gas-fired heater or heat exchanger. The percent kill of the microorganisms depends on the final temperature, holding time and the species of target microorganism. Since water has a high specific heat, considerable energy is required to heat to the proper disinfecting temperature. It must then be cooled prior to biological filtration or before it re-enters culture units since fish and nitrifying bacteria cannot tolerate the high temperatures required for disinfection (Wheaton 1977). A major disadvantage to using heat disinfection methods

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in recirculating systems is the energy required for heating and cooling (Dupree 1981).

Chlorine

Chlorine is an inexpensive and readily available chemical disinfectant, and equipment for its application is readily available (Dupree 1981). Chlorine is added as chlorine gas (Cl_2), calcium hypochlorite ($Ca(OCl)_2$), or sodium hypochlorite ($Na(OCl)$). In all cases the active disinfecting agent is the hypochlorite ion (OCl^-) or hypochlorous acid ($HOCl$). These exist in a pH-dependent chemical equilibrium (Piedrahita 1991):

$$HOCl \leftrightarrow OCl^- + H^+; \quad (pK_a = 7.54 \text{ @ } 25°C) \quad (1)$$

Chlorine, HOCl and OCl^- are strong oxidizing agents. A disadvantage is that they react with a variety of organic and inorganic materials, one of which is ammonia. NH_3 - HOCl reactions may produce mono-, di-, and trichloramines, substances which are toxic to fish (Wheaton 1977). The HOCl and OCl^- compounds are called **free available chlorine**, and chloramines are referred to as **combined available chlorine**. Sufficient contact time must be allowed for chlorine to be effective. It must first dissociate into the water before it becomes toxic. It then diffuses through cell walls and kills the microorganisms by inactivating certain enzymes.

Chlorine use in recirculating systems is risky since fish and invertebrates are highly sensitive to both chlorine and chloramines. It is reported to cause gill damage in fish. Residual chlorine and chloramines must be removed from culture systems by dechlorination methods. The reported safe level of residual chlorine in aquaculture systems is 0.5 to 3 µg/l (Piedrahita 1991).

Dechlorination can be accomplished using one of several procedures available. Reducing agents like sodium thiosulfate ($Na_2S_2O_3$), sodium sulfite (Na_2SO_3) and ferrous salts will reduce chlorine to inactive compounds, but they have no effect on chloramines (Dupree 1981). Reducing agents should be used cautiously since they have toxic properties of their own. Piedrahita (1991) reported that reducing agents should be used in strengths no greater than 1 to 5 mg/l. A limited amount of chlorine can be destroyed with UV light, but UV light effectiveness is inhibited in systems where sodium thiosulfate is used for dechlorination (Seegert and Brooks 1978).

Aeration is often suggested as a dechlorination method; however, it is doubtfull that chloramines are completely removed by aeration. Bedell (1971) reported that aeration is not effective in water high in organics. He stated that both chlorine and chloramines can best be removed with activated carbon. This may be practical for small systems, but activated carbon use has not been shown to be economical in large recirculating systems.

Ultraviolet Light

Ultraviolet (UV) irradiation, also referred to as UV light, is a popular disinfection method and is routinely used in hatcheries. It is effective in the control of bacteria, viruses and other microorganisms (Dupree 1981). Its killing effect is largely a function of light wavelength. Koller (1965) and Wheaton (1977) reported the most effective light wavelength to be 2500 to 2600 Å (angstroms). The effectiveness drops rapidly on either side of the peak; 0.4 and 0.002% at 3200 and 7000 Å, respectively. Hoffman (1974) stated that 2537 Å is not dangerous to fish and is effective in killing a variety of fish pathogens. The mechanism by which microorganisms are killed is not completely understood, but it is believed that UV energy interrupts the genetic chemistry (DNA) of living cells (Moe 1989).

Ultraviolet light is measured in microwatts per second per square centimeter of contact area (µW/s/cm^2). Spotte (1979) recommended a minimum dosage of 35,000 µW/s/cm^2 for disinfecting aquarium water. Dosages required to achieve 99 - 100% kill vary from 35,000 to 156,000 µW/s/cm^2, which is adequate for killing practically all disease-causing organisms which trouble aquaculture systems. Yeasts appear to be more resistant than bacteria (Huguenin and Colt 1989). The lethal dose varies with age, size and species of target microorganism, water turbidity, ionic strength and water depth.

The efficiency of UV bulbs decreases with age due to degradation of the electrodes, caused by switching on and off, and because of the gradual darkening of the inner glass surfaces of bulbs. This darkening effect is called **solarization**. Because of solarization, the effective life of most bulbs is 6 to 12 months, after which they must be replaced (Moe 1989). Bulb efficiency is also reduced by biological slime and mineral deposits which collect on the outside.

Impurities in the water are the major limiting factor determining bulb effectiveness. Turbidity, organic material and color absorb UV light rays and prevent exposure to target organisms. Suspended particulates shield smaller microorganisms from the light, thus further reducing its effect. UV light can only penetrate water to a depth about 50 mm (2 in). Therefore, it is recommended that water being treated should be no deeper than 25.4 mm (1 in) to insure complete penetration (Moe 1989). Slow flow rates through UV light units are recommended for proper exposure times. Spotte (1970) recommended flow rates no greater than 3 lpm per mm of bulb (2 gpm/in).

Three types of UV irradiation units are manufactured for aquaculture use. The **suspended** type is the oldest and most simple. These units are suspended about 50 to 150 mm (2 to 6 in) above the water surface, and water flows slowly beneath the lamps in a thin layer. Figure 1 illustrates this type of UV unit. Water is switched back and forth beneath the UV bulbs by

baffles, thus receiving a longer exposure to the light rays. Units of this type are subject to coating by biological slime and mineral deposits. Other types are the submerged and jacketed types. In submerged units the UV bulbs are contained inside a PVC tube, and the water flows between the tube and the bulb. The bulb operates at the same temperature as the culture water. Jacketed bulb types have a quartz sleeve over the UV bulb, and water flows on the outside of the sleeve. Most commercial UV light units available today are of this type. UV bulbs which do not contact the water (such as in jacketed and suspended bulb units) operate at their most effective temperature around 40°C (104°F). Bulb efficiency declines as the temperature declines and is only about 50 % at 21°C (70°F) (Moe 1989). Thus, submerged bulb units are the least efficient type.

Disadvantages to UV light use are numerous. Since UV light leaves no residuals in the water any interruption in power will cause disruption of the disinfection process and leave the system unprotected. Even when functioning properly UV light will not affect microorganisms in pipes, pumps, culture units, etc. since they will not be exposed to the light. UV units are maintenance-intensive since the bulbs require frequent cleaning. Additionally, water impurities severely reduce UV light effectiveness. Finally, the applicability of UV light on a large scale is questionable because of the low flow rates recommended.

Ozone

Ozone (O_3) is a three atom allotrope of oxygen. The three oxygen atoms are loosely held together in an unstable bond. A single oxygen atom is quick to break away and reacts with most of the organic and inorganic molecules it comes into contact with. Hence, ozone is a powerful oxidizing agent, second only to fluorine.

Ozone has been used in the sewage treatment industry in Europe since the early 1900's. It was later used for sterilizing potable water supplies. Two decades ago it came into widespread use in the aquarium trade. In the 1970's 6 out of 10 European aquarists were using ozone (Stopka 1975). It has been slow to be adopted in the United States, however. A flurry of research expounded the use of ozone in the 1970's and early 1980's, but it was considered to be too expensive. Interest in ozone use has since been revived.

In aquaculture systems ozone is reported to provide many benefits and has distinct advantages (Wedemeyer et al. 1979; Williams et al. 1982 and Rosen 1972). Ozone use in aquaculture systems has been cautious due to its potential toxic effects on fish, invertebrates and bacteria in biofilters (MacLean et al. 1973; King and Spotte 1974). Ozone toxicity has been reviewed by Rosenthal (1980), Wedemeyer et al. (1979) and other researchers. These studies conclude that, if residual concentrations remain low, ozone can be safely used with significant advantages.

The effectiveness of ozone as a disinfecting agent is a function of dosage and contact time. Contact time is acquired by dispersing ozone throughout the culture water. Contact time and ozone concentration vary with the target microorganism and water quality. It effectively destroys bacteria, viruses, fungi, algae and protozoa (Lohr et al. 1986). Ozone kills by "burning" delicate cell membranes and actually enters the cells, destroying the cell nuclear chemistry (Moe 1989). Effects of ozone on various organisms are reviewed by Hoffman (1974), Colberg and Lingg (1978), Lohr and Gratzek (1986), Wedemeyer et al. (1978) and others.

In addition to its disinfecting properties, ozone is reported to reduce nitrite (Evans 1972; Colberg and Lingg 1978), BOD and COD (Colberg and Lingg 1978) and causes the precipitation of iron and manganese (Wheaton 1977). Ozone "breaks" long chain organic molecules into shorter chain molecules which are more easily degraded by bacteria. The gelatinous coating on tank walls and in pipes and sumps is reduced, thus lessening the chances for clogging and short-circuiting in these units. Total system solids loading is reduced with ozone. Its oxidative potential is also a disadvantage, however, in that ozone is very corrosive. All system components, such as PVC, plastics, O-rings, pumps, etc. having direct contact with ozone must be fabricated from ozone-resistant material.

Effect of pH. The decomposition of ozone is accelerated in the presence of hydroxyl ions (Weiss 1935). Therefore, rapid reduction of ozone is reported at pH 8.0 and above. Colberg and Lingg (1978) reported that, in their experience, even though the concentration of ozone was lowest at higher pH values, the oxidation capacity was greater. Ozone seems to be more effective at pH below 7.0 (Wheaton 1977).

Effect of Organic Matter. Since color, odor and turbidity can be removed by ozone, this indicates that they also have an ozone demand (Dupree 1981). Water containing high organic loads must be treated with higher dosages of ozone. As organic matter is oxidized, more free ozone becomes available. Therefore, it is wise to cut back on the ozone output a few days after installation in older systems (Moe 1989).

Effect of Salinity. Evidence supports the use of ozone in marine systems (Honn and Chavin 1976; Sutterlin et al. 1984; Moe 1989). However, since marine systems normally have higher pH values, ozone decomposition is more rapid. Ozone may have detrimental effects on seawater. It is reported to deplete certain trace elements, particularly manganese (Spotte 1970) and calcium (Moe 1989). Ozone reacts with the chloride and bromide ions, forming toxic hypochlorites and hypobromites (Moe 1989).

Comparison of Methods

Ozone vs. Chlorine Disinfection

Ozone does not leave harmful residuals (Hewes and Davison 1971), as chlorine does, since it rapidly degrades to molecular oxygen (Layton 1972). The half-life of ozone is about 15 minutes (Honn et al. 1976). Ozone has twice the chemical oxidative capacity of the hypochlorite ion (Rosen 1972). Ozone does not react with ammonia like chlorine, allowing it to react much quicker. The effectiveness of chlorine on a mg/l basis is much less than that of ozone (Venosa 1972). Ozone is not as affected by pH and temperature changes as chlorine.

Ozone vs. UV Irradiation

Ozone rapidly oxidizes turbidity, organics and color-causing agents. Its efficiency is therefore not affected by these materials. Impurities in the water severely reduces the efficiency of UV light since they absorb the light rays and shield microorganisms from the killing effects of the light. Thus, ozone does not require costly pre-filtration like UV light systems to be effective (Lohr and Gratzek 1986).

Generation of Ozone

Ozone is a very unstable gas. Thus, it must be generated on-site and used immediately. It's production is measured as mg ozone per hour. Most ozone generators used in the aquaculture industry use one of two methods to produce ozone: UV irradiation or silent electrical (corona) discharge. The UV irradiation method is the less efficient of the two and is used primarily in small ozone generators used in the aquarium trade. In this method air is passed through a chamber containing a UV bulb which emits light in the 1,000 to 2,000 Å range (Klein et al. 1985). The UV light splits some of the oxygen molecules. The single oxygen atoms then attach to other O_2 molecules and form ozone (O_2). The UV approach is only applicable to small systems because of the low concentration ozone produced (Honn et al. 1976). The feed gas must be dry since moisture decreases the amount of ozone produced and accelerates the formation of corrosive nitrous oxides. Also, air compressors must be free of residual oil since any hydrocarbons present will reduce ozone production (Honn et al. 1976).

Corona discharge generators are capable of large-scale ozone production. A high voltage is impressed across two plates through which air or pure oxygen passes. The oxygen molecules are "excited" by the electrical charge and form ozone. The process is energy-consuming and produces heat.

Either air or pure oxygen can be used to produce ozone. The same ozone generator uses twice the power to produce ozone from air as it does to produce ozone from pure oxygen. Even though oxygen must be purchased, the cost is still usually less than ozone production using air (Wheaton 1977).

Ozone Dosage

Sufficient time must be allowed for ozone to be dispersed through the water. This brings about the required contact between the ozone and the target organism. Contact time and ozone concentration vary with the target organism and water quality.

The scientific literature indicates that ozone concentrations of 0.56 to 1.0 mg O_3/l and contact times of 1 to 3 minutes are sufficient to kill most pathogens in aquaculture systems (Dupree 1981), however this varies considerably. Wedemeyer et al (1978) reported that the minimum dosage required to control Saprolignia on fish eggs without damage to eggs or fry was 0.03 mg/l. Ozone was toxic to eggs at 0.3 mg/l. Wedemeyer et al. (1979) reported that an ozone concentration of 0.01 mg/l caused complete inactivation of enteric red mouth (ERM) bacterium in 30 seconds while 10 minutes was required to inactivate A. salmonicida. It was felt that pH and other water quality factors could have caused the differences. At the Dworshak National Fish Hatchery in Idaho an ozone residual of 0.20 mg/l for 10 minutes achieved control over the virus infectious hematopoietic necrosis (IHN) (Owsley 1988).

Ozone treatment normally does not leave a residual in the treated water. However, for disease protection some researchers have advised that a very low residual is not harmful to fish and may prevent disease from reoccurring. Wedemeyer et al. (1979) determined that a safe permissible exposure level of ozone to fish was 0.002 mg/l. Toxicity effects vary considerably in the literature. Arthur and Mount (1975) reported that ozone was toxic to fathead minnows at 0.2-0.3 mg/l. Roselund (1974) and Honn et al (1976) reported gill damage to rainbow trout at 0.01-0.06 mg/l. In other toxicity testing Wedemeyer et al. (1978) observed no differences between salmon exposed for three months to 0.023 mg O_3/l and the control. However, damage was observed at 0.05 mg/l. Sutterlin et al. (1984) reported that 1.62 g O_3 per kg fish with a one hour exposure time was sufficient to maintain water clarity with no ill effects on Atlantic salmon smolts. Treatment still had no effects when extended to 8 hours, but high mortalities occurred when the ozone generator was accidentally left running for 15 hours. Honn and Chavin (1976) noted no deleterious effects of 0.132 mg O_3/hr/l in a seawater system containing nurse sharks. With so much variability being reported, the only positive method to determine ozone dosages and contact times is with on-site bench scale testing.

Precautions must be taken not to expose humans to high levels of ozone. Ozone can be detected in air by its odor, often described as "sharp" and "fresh". Humans can smell ozone in the range of 0.05 mg/l. The maximum safe concentration for an 8-hour work day is 0.1 mg/l (Owsley 1988). Ozone in air can cause headaches,

nausea, and eye irritation (Moe 1989). Human exposure can be minimized by applying ozone for a few hours at night or during "off" periods when personnel will not be exposed.

Analytical Methods

Because of its recommended limitations, ozone is measured in very low concentrations, and is, at times, difficult to detect. Instrumentation to directly measure ozone in water is commercially available; however, it is expensive and the sensitivity is usually not great enough to measure ozone in 0.001 mg/l increments.

Several wet chemistry methods are available (APHA 1988; Owsley 1988). The DPD (N,diethyl-P-phenylenediamine) method is relatively simple but has limitations (Paulin 1967). The test must be completed within 6 minutes after fixing the sample and has many interferences (Owsley 1988). A modified DPD method showed good results and increased accuracy (Wedemeyer et al. 1978).

The standard accepted procedure for measuring ozone in water is the Indigo Blue method (Bader and Hoigne 1982). It is stable for up to 4 hours and has the least interferences of any of the analytical methods available.

Another method which can be used as an indicator is **redox potential**, which is a measure of the relative amount of the positive and negative charges on the oxidized and reduced molecules in solution. A high redox potential is necessary in culture water for optimum cell respiration. High organic loads depress the redox potential in aquaculture systems. Thus, most aquaculture waters have a low redox potential. The addition of ozone will oxidize more of the reduced molecules, thus elevating the redox potential. The higher the redox potential, the more "pure" the water.

Redox potential is reported in mv (millivolts). Healthy aquaculture systems should have a redox potential ranging from 200 to 350 mv. Levels below 200-250 mv indicate the presence of toxic, reduced compounds while levels above 400-450 mv indicate too active an oxidative environment, which can potentially damage plant and animal tissues and cells. Over-treatment with ozone can produce too high a redox potential (Moe 1989). Redox potential can be maintained at a comfortable 300-350 mv with careful ozone regulation. Commercial automatic control units are available which continuously monitor redox potential and adjust ozone output to maintain redox potential at specified levels.

Applications

Ozone units can be incorporated into recirculating systems in a number of schemes. The choice is left to the aquaculturist and is dependent on system design, site conditions and economics. It is generally agreed that ozonation should precede solids removal and/or biological filtration (Fig. 2). Since dissolved and suspended organic materials are rapidly oxidized by ozone, the load going to settling basins, tube settlers, or sand filters (whichever method is used for solids removal) is reduced, lessening the chance for clogging or short-circuiting in these units. Due to the decrease in solids it may be possible to decrease the size of these units, reducing the initial capital investment and operating costs.

Ozone has the effect of splitting large organic molecules into smaller biodegradable materials which are more easily removed by heterotrophic bacteria. Organic matter reduction prior to biological filtration has the effect of reducing the population of heterotrophic bacteria in the filters. Heterotrophs feed on organics and are the first to become established in biofilters. They compete for space on biofilter substrate with nitrifying bacteria. Too high a population of heterotrophs may inhibit nitrification (Palier and Lewis 1988).

The direct ozonation of water in the culture unit is the most simple method of application. In this method ozone is introduced through airstones or diffusers at the bottom of a water column. This method is most frequently used in home aquaria and small recirculating systems, however, caution must be used to guard against the potentially toxic effects of ozone residuals. Ozone can be applied as a "dose", lasting just a few hours. Continuous direct ozonation is not recommended. In lieu of direct application in culture units, ozone can be applied through a separate mixing chamber following the culture unit and preceding solids removal/biological filtration. In this technique, large doses can be applied and residuals can be removed by aeration. Aeration cuases the ozone to degrade to molecular oxygen. Contact time is governed by the size of the chamber.

Some recirculating systems are designed for easy retrofit of an ozone generator. For existing systems the sidestream method of injection (Fig. 3) generally works best since it requires a minimum amount of plumbing changes. An alternate method is to combine ozone with a foam fractionator so that the benefits of both can be realized (Rosenthal 1980). The benefits of foam fractionation in aquaculture systems are well known (Wheaton et al. 1979; Lawson and Wheaton 1980), and the combination of foam fractionation and ozone can be very effective. Rosenthal (1980) reported that heavy metals were removed from culture water by foam fractionation and ozonation used in combination. Ozone changes some complex organics into more surface-active compounds which produce a more stable foam and increase the efficiency of the operation (Moe 1989).

Figure 4 illustrates a counter-flow foam fractionator with ozone injection. The water enters the foam column near the water surface and flows downward against the upward flow of air (or pure oxygen) and ozone, which is introduced through a fine bubble diffuser near the

bottom of the column. The water current is generated by a second diffuser placed in the smaller column. Foam is collected in the conical-shaped collector at the top of the column. Foamate is then collected in a container and saved for disposal. Foam units can be used individually or in multiple units, depending on system loading. They may be hung on the inside or outside of culture units, or they may be installed into a separate mixing chamber. Several models of foam fractionators incorporating ozone are commercially available.

Ozone Removal

Because of its toxicity to fish ozone must be removed from culture water. Since ozone decomposes back to oxygen in a short period of time it can be removed by allowing culture water to be held in an aeration chamber for a period of time. However, this may not be practical in some systems due to the size of chamber required. Carbon filters are also very effective for removing ozone (Owsley 1988). However, for reasons previously discussed, carbon adsorption is not normally economical in large recirculating systems.

A faster, more economical method for ozone removal is to strip it from solution using a stripping tower (Owsley 1981). Complete removal occurs in properly designed units. Figure 5 illustrates a stripping tower. Tower packing material consists of plastic modules or other rigid, non-toxic material which has a high surface area. Ozone-laden culture water enters the top of the tower. Air enters at the bottom of the column and flows upward through the packing material. Ozone is "stripped" from the water by the counter-current action of the process. "Clean" culture water collects in the bottom of the tower and is returned to the culture unit. If desired, the unit can be covered and the ozone off-gas vented to the outside. A word of caution is in order here; the designer should insure that the level of the water at the bottom of the tower is below the level of the air inlet so that water does not back up into the air compressor when power is off.

Conclusions and Recommendations

Ozone is a viable method for disinfecting and removing organics from culture water. It has also been proven an effective disinfectant. Next to fluorine, it is the most powerful oxidizing agent known.

Ozone has been successfully used in Europe for many years. It reacts twice as fast as chlorine and is not as restrictive in water quality requirements as is chlorine and UV irradiation. Ozone does not react with ammonia like chlorine and leaves no harmful residuals in the water.

Ozone must be generated on-site since it is an unstable gas. Production is doubled using pure oxygen instead of air.

Ozone is highly corrosive. All materials which directly contact ozone must be ozone-resistant.

The concentration of ozone can be determined using a variety of analytical methods. The most common is the Blue Indigo method. Another method showing potential is redox potential.

Caution should be exercised when using ozone. Residuals in culture units should not exceed 0.002 mg O_3/l. Ozone in air should not exceed 0.1 mg/l for an 8-hour shift to protect humans who may be exposed. Ozonation should be done at off-hours when employees are not normally present.

Potentially, the most promising method of ozone injection is in combination with foam fractionation.

Stripping towers remove residual ozone and eliminate the need for costly detention chambers.

Literature Cited

APHA. 1988. Standard Methods for the Examination of Water and Wastewater. American Public Health Association, Water Pollution Control Federation,. Washington, DC. 1134 pp.

Arthur, J.W. and D.I. Mount. 1975. Toxicity of a disinfected effluent to aquatic life. First International Symposium on Ozone in Water and Wastewater Treatment, Intern. Ozone Inst., Syracuse, NY.

Bader, H. and J. Hoigne. 1982. Determination of ozone in water by the Indigo Method: a submitted standard method. Ozone: Science & Engineering 4:169-176.

Bedell, G.W. 1971. Eradicating Ceratomyxa shasta from infected waters by chlorination and ultraviolet irradiation. Prog. Fish Cult. 33:51-54.

Colberg, P.J. and A.J. Lingg. 1978. Effects of ozonation on microbial fish pathogens, ammonia, nitrate, nitrite and BOD in simulated reuse hatchery water. J. Fisheries Res. Board of Canada 35:1290-1296.

Dupree, H.K. 1981. An overview of the various techniques to control infectious diseases in water supplies and in water reuse aquacultural systems. Bioengineering Symposium for Fish Culture (FCS Pub. 1): 83-89.

Evans, F.L. 1972. Ozone technology: current status. IN: F.L. Evans, ed. Ozone In Water And Wastewater Treatment, Ann Arbor Science Publishers, Inc., Ann Arbor, MI.

Hewes, C.G. and R.P. Darrison. 1971. Kinetics of ozone decomposition and reaction with organics in water. Orig. not available. IN: Lohr and Gratzek 1986.

Hoffman, G.L. 1974. Disinfection of contaminated water by ultraviolet radiation, with emphasis on whirling disease (Myxosoma cenebralis) and its effects on fish. Trans. Amer. Fish. Soc. 103:541-550.

Honn, K.V. and W. Chavin. 1976. Utility of ozone treatment in the maintenance of water quality in a closed marine system. Marine Biol. 34:201-209.

Honn, K.V., G.M. Glezman and W. Chavin. 1976. A high capacity ozone generator for use in aquaculture and water processing. Marine Biol. 34:211-216.

Huguenin, J.E. and J. Colt. 1989. Design and Operating Guide for Aquaculture Seawater Systems. Elsevier, Amsterdam, 264 pp.

King, J.M. and S. Spotte. 1974. Marine aquariums in the research laboratory. IN: Honn et al. 1976.

Koller, L.R. 1965. Ultraviolet Radiation. John Wiley and Sons, New York, 312 pp.

Lawson, T.B. and F.W. Wheaton. 1980. removal of organics from fish culture water by foam fractionation. J. World Mariculture Soc. 11:128-134.

Klein, M.S., R.I. Brabets and L.C. Kinney. 1985. Generation of ozone. Proc. First International

Symposium on Ozone for Water and Wastewater Treatment, International Ozone Institute, Waterbury, CT.

Layton, R.F. 1972. Analytical methods for ozone in water and wastewater applications. IN: F.L. Evans, ed. Ozone In Water And Wastewater Treatment, Ann Arbor Science Publishers, Inc., Ann Arbor, MI.

Lohr, A.L. and J.B. Gratzek. 1986. Bactericidal and parasitic effects of an activated air oxidant in a closed aquatic system. J. Aquariculture and Aquatic Sciences 4:1-8.

MacLean, S.A., A.C. Longwell and W.J. Blogoslawski. 1973. Effects of ozone-treated seawater on the spawned, fertilized, meiotic and cleaving eggs of the commercial American oyster. Annals App. Biol. 45:521-527.

Moe, M.A. 1989. The Marine Aquarium Reference: Systems and Invertebrates. Green Turtle Publications, Plantation, FL, 510 pp.

Owsley, D.E. 1981. Nitrogen gas removal using packed columns. Bio-Engineering Symposium for Fish Culture (FCS Pub. 1):71-82.

Owsley, D.E. 1988. Ozone for disinfecting hatchery rearing water. Presented at the Bio-Engineering Symposium, Amer. Fish. Soc., Portland, OR, Oct. 24-28.

Piedrahita, R.H. 1991. Engineering aspects of warmwater hatchery design. IN: Aquaculture Systems Engineering, American Society of Agricultural Engineers, St. Joseph, MI, 138 pp.

Paller, M.H. and W.M. Lewis. 1988. Use of ozone and fluidized-bed biofilters for increased ammonia removal and fish loading rates. Prog. Fish Cult. 50:141-147.

Papko. S.I. 1957. Action of certain heterogeneous catalysts on the oxidation of ammonia in aqueous solution by ozonated oxygen. J. Applied Chem. 30:1361-1367.

Paulin, A.T. 1967. Procedures for measuring ozone in water. J. Institute Water Engineering 21:537.

Rosen, H.M. 1972. Ozone generation and its relationship to the economical application of ozone in wastewater treatment. IN: F.L. Evans, ed. Ozone In Water And Wastewater Treatment, Ann Arbor Science Publishers, Inc., Ann Arbor, MI.

Roselund, B. 1974. Disinfection of hatchetry effluent by ozonation and the effect of ozonated water on rainbow trout. IN: W.J. Blogoslawski and R.G. Rice, eds. Aquatic Applications of Ozone, International Ozone Institute, Syracuse, 226 pp.

Rosenthal, H. 1980. Ozonation and Sterilization. European Inland Fisheries Advisory Council (EIFAC) Symposium on New Developments in the Utilization of Heated Effluents and of recirculation Systems for Intensive Culture, Stavanger, Norway.

Sander, E. and H. Rosenthal. 1975. Application of ozone in water treatment for home aquaria, public aquaria and for aquaculture purposes. IN: W.J. Blogoslawski and R.G. Rice, eds. Aquatic Applications of Ozone, International Ozone Institute, Syracuse, 226 pp.

Seegert, G.L. and A.S. Brooks. 1978. Dechlorination of water for fish culture: comparison of activated carbon, sulfite reduction and photochemical methods. J. Fish Res. Brd. of Canada 35:88-92.

Spotte, S.H. 1970. Fish and Invertebrate Culture: Water Management In Closed Systems. John Wiley and Sons, New York, 145 pp.

Spotte, S.H. 1979. Seawater Aquariums: Captive Environment. John Wiley and Sons, New York, 413 pp.

Stopka, K. 1975. European and Canadian experiences with ozone in controlled closed circuit fresh and salt water systems. IN: W.J. Blogoslawski and R.G. Rice, eds. Aquatic Applications of Ozone, International Ozone Institute, Syracuse, 226pp.

Sutterlin, A.M., C.Y. Couterier and T. Devereaux. 1984. A recirculation system using ozone for the culture of Atlantic salmon. Prog. Fish Cult. 46:239-244.

Venosa, A.D. 1972. Ozone as a water and wastewater disinfectant: a literature review. IN: F.L. Evans, ed. Ozone In Water And Wastewater Treatment, Ann Arbor Science Publishers, Inc., Ann Arbor, MI.

Wedemeyer, G.A., N.C. Nelson and C.A. Smith. 1978. Survival of the salmonid viruses infectious hematopoeitic necrosis (IHNV) and infectious pancreatic necrosis (IPNV) in ozonated, chlorinated and untreated waters. J. Fish Res. Brd. Canada 34:433-437.

Wedemeyer, G.A., N.C. Nelson and W.T. Yasutaka. 1979. Potentials and limits for the use of ozone as a fish disease control agent. Ozone:Science and Engineering 1:295-318.

Weiss. J. 1935. The radical HO_2 in solution. Trans. Farraday Society 31:668-681.

Wheaton, F.W. 1977. Aquacultural Engineering. John Wiley and Sons, New York, 708pp.

Wheaton, F.W., T.B. Lawson and K.M. Lomax. 1979. Foam fractionation applied to aquacultural systems. J. World Maric. Soc. 10:795-808.

Williams, R.C., S.G. Hughes and G.L. Rumsey. 1982. Use of ozone in water a reuse system for salmonids. Prog. Fish Cult. 44:102-105.

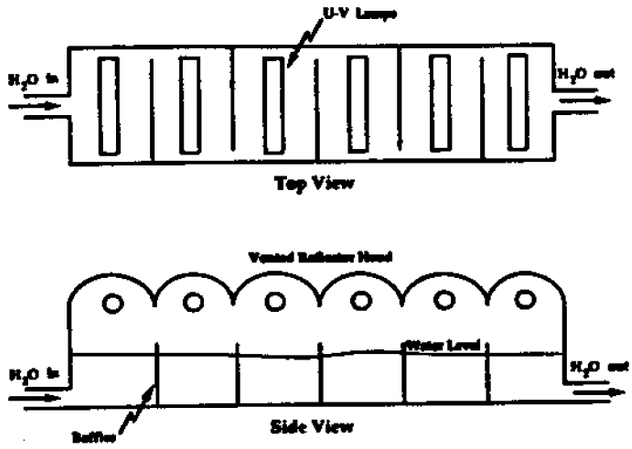

Figure 1. A suspended bulb UV irradiation unit.

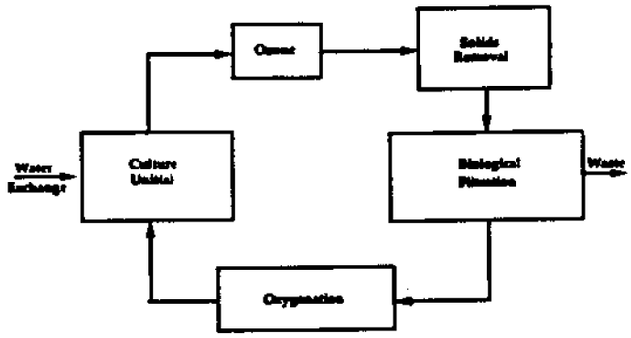

Figure 2. Sequence of treatment processes in a semi-closed recirculating system loop.

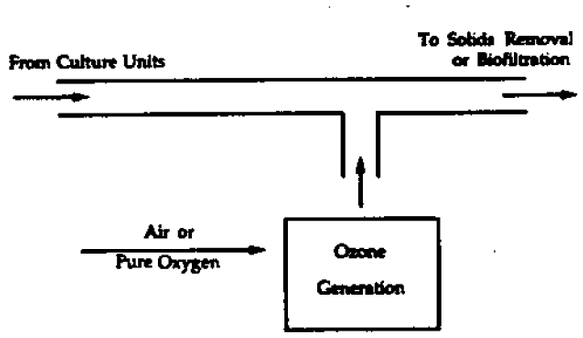

Figure 3. Ozone injected as a side stream.

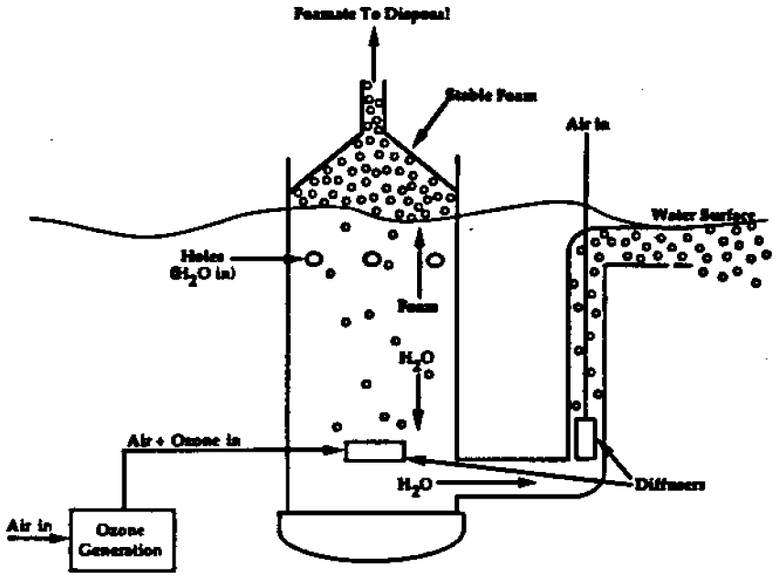

Figure 4. Ozone used in combination with a counter-current foam fractionator.

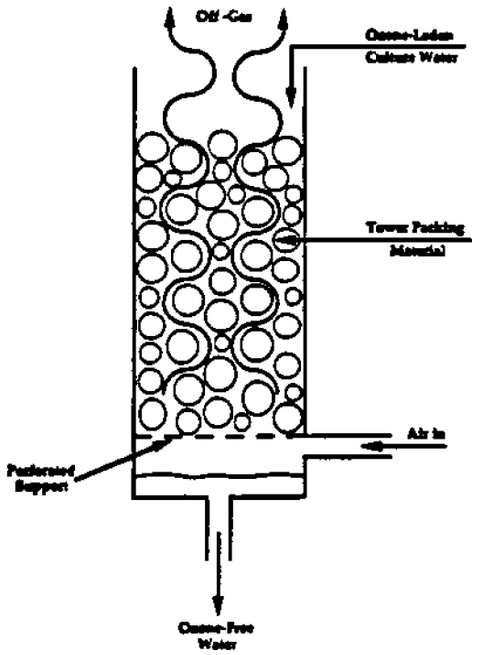

Figure 5. A packed tower used for ozone stripping.

Design Procedure for Hooded Surface Oxygen Absorption Systems

Barnaby J. Watten
U.S. Fish and Wildlife Service
National Fishery Research and Development Laboratory
Wellsboro, Pennsylvania

Abstract

A design procedure addressing effluent total dissolved gas pressure limits along with standard performance indicators such as oxygen absorption efficiency (kg absorbed/kg applied), and transfer efficiency (kg/kW•h), is presented for surface agitation equipment operating in an oxygen-enriched atmosphere. Application of this contactor type in closed culture systems is attractive given its insensitivity to biological fouling and ability to operate without the need for a significant hydraulic gradient. Performance algorithms were developed through application of chemical reactor theory, Henry's Law and the Ideal Gas Law. In the analysis, gas and liquid phases were treated as being homogeneous. The design steps presented are unique in that required mass transfer coefficients (K_La), operating pressures (CP), and oxygen feed rates (G/L) are calculated for target changes in both dissolved oxygen and nitrogen without the use of iterative numerical procedures. A second calculation sequence establishes the sensitivity of system performance to changes in G/L when CP and K_La are known.

Introduction

Successful operation of closed or semi-closed culture systems requires regulation of dissolved gas levels within acceptable limits. Regulation generally demands oxygen (DO) supplementation at rates that can approach the mass rate addition of feed per day, i.e., 1 kg O_2/kg feed (Colt et al. 1991). Oxygen supplementation is frequently achieved by exposing culture water to oxygen-enriched gas within equipment designed to provide large gas-liquid interfacial areas as well as intensive mixing (Colt and Watten 1988; Boyd and Watten 1989). Equipment of this type has the unique ability to economically saturate or supersaturate water with DO, while concurrently stripping dissolved nitrogen (Watten and Beck 1985; Watten et al. 1991). Supersaturation with DO can increase allowable fish loading (kg/l•min) or rearing densities (kg/m^3) several fold (Colliens et al. 1984; Gowan 1987; Colt et al. 1991). These increases, in turn, can lower production costs by minimizing required rearing vessel volumes, reducing the energy used to circulate water among system components, and by decreasing the size of treatment units that are sensitive to hydraulic loading (e.g. clarifiers and biological filters). Supersaturation with DO, however, may result in gas bubble trauma or, in extreme cases, oxygen toxicity. The latter results from the oxidation of cellular components and leads to respiratory failure (Sebert et al. 1984; Colt et al. 1991). Oxygen toxicity can be avoided by keepingthe tension of DO below 300 mm Hg (Table 1). The pathology of gas bubble trauma includes the formation of gas emboli and hemostasis, which act together to reduce fish growth and increase mortality (Bouck 1980; Weitkamp and Katz 1980; Krise 1991). Gas bubble trauma is caused by exposure to a total dissolved gas pressure that exceeds the sum of hydrostatic pressure and local barometric pressure—i.e., when $\Delta P > 0$ where

$$\Delta P = (\Sigma F_iC_i) - BP - HP + VP \qquad (1)$$

The net change in ΔP across an oxygen absorber will be fixed by the relative rate of dissolved nitrogen (DN) desorption as indicated by the ΔDODN stripping ratio (Watten et al. 1991):

$$\Delta DODN = (\Delta DO)_{AB} / (\Delta DN)_{AB} \qquad (2)$$

Stripping ratios greater than -2.2 provide a net reduction in dissolved gas pressure whereas ratios less than -2.2 result in an increase. Therefore excessive ΔP can be avoided by designing equipment that provides the required nitrogen as well as oxygen transfer rates. A design procedure that addresses total gas pressure limits along with standard performance indicators (AE, TE, and transfer costs) is presented in this paper for hooded surface oxygen absorption equipment. Application of this contactor type in closed culture systems is attractive given its relative insensitivity to biological fouling and ability to operate without the need for a significant hydraulic gradient. The design procedure is unique in that operating conditions are identified for target changesin dissolved gases directly without the need for the iterative numerical procedures described by Watten et al. (1990).

Principles of Operation

Major components of an enclosed surface oxygenation system are shown in Figure 1 (Watten et al. 1990). The rate of gas absorption or desorption achieved will be governed by the product of the effective mass transfer coefficient, K_La, and the prevailing dissolved gas deficit (Lewis and Whitman 1924):

$$\frac{dC}{dt} = (K_La)_{LT} (C^*_i - C_i)_{AB} \qquad (3)$$

The coefficient K_La reflects operating conditions within the enclosure. Important conditions include

characteristics of the liquid, turbulence, and the extent of the gas-liquid interfacial area present:

$$(K_L a)_{I,T} = \frac{D_I A}{\Delta V} \qquad (4)$$

Turbulence and interfacial area in turn are related to reactor geometry, agitator type, and power input. Figure 1 shows mixing by a rotating propeller. Surface agitation can also be provided by rotating turbines, paddles or by jetting water (Petit 1981; Meade et al. 1991). The effective $K_L a$ for a specific agitator type can be varied by changing the extent of agitator or spray head submergence (Watten et al. 1990; Meade et al. 1991).

The $K_L a$ value for a system is typically identified through analysis of steady-state oxygen transfer data given the difficulty of measuring the variables D, A, or Δ independently (Equation 4). In the case of the surface oxygenation system, $K_L a$ is obtained from the following expression assuming both gas and liquid phases are homogeneous, i.e. the system is operating as a mixed-flow reactor (Watten et al. 1990):

$$(K_L a)_{O2,T} = \frac{(GTR)_{O2}}{(C^* - C_{out})_{O2,AB}\, V\, 10^{-3}} \qquad (5)$$

Once established, $(K_L a)_{O2}$ values can be corrected for the effects of non-standard liquid characteristics such as temperature, surfactants, or dissolved gas species (N_2) using the following expression(Tsivoglou et al. 1965; Stenstrom and Gilbert 1981; APHA 1985):

$$(K_L a)_{I,T} = (K_L a)_{O2,20^\circ}\, 1.024^{T-20} \propto \emptyset_I \qquad (6)$$

In addition to the enhancement of $K_L a$, gas transfer rates within the enclosed surface oxygenation system are accelerated by elevating dissolved gas deficits ($C_i^* - C_i$) through enrichment of the gas phase with commercial oxygen. Following Henry's Law, the change in mole fraction (χ_i) of oxygen in the gas phase increases the saturation concentration (C) of DO and decreases the C* of DN (Colt 1984):

$$C^*_i = \chi_i K_i \beta_i\, 1000\, \frac{(CP - VP)}{760} \qquad (7)$$

This operating characteristic, common to oxygen contact equipment, allows for an effluent DO above air saturation concentrations. The decrease in C^*_{N2} also enables DN to be stripped, thereby controlling total dissolved gas pressures. Nitrogen stripped from solution enters the gas phase then is purged from the enclosure by venting off-gas. Positive enclosure gauge pressures allow for passive off-gas venting. Negative gauge pressures require energy to remove off-gas either through application of a vacuum pump or water jet exhauster (Fig. 1).

Design factors affecting the rate of oxygen addition and nitrogen desorption include the oxygen feed rate (G/L), enclosure pressure (CP), $K_L a$, water residence time ($V/Q_L \cdot 0.06$), and influent DO and DN concentrations. Carbon dioxide (DC) stripping over typical G/L ratios (< 5%) is negligible. Therefore the effect of DC on system performance can be ignored (Watten et al. 1991). Application of reactor theory provides a means of correlating important design conditions with system performance. For example, effluent DO or DN can be predicted under steady-state conditions by treating gas transfer as a first order reaction with respect to the dissolved gas deficit (Equation 3), and by using the general conversion model for a mixed flow reactor (Levenspiel 1979):

$$\frac{(C^* - C_{out})_{AB,I}}{(C^* - C_{in})_{AB,I}} = \frac{1}{1 + (K_L a)_I\, (V/Q_L\, 0.06)} \qquad (8)$$

Alternatively, the residence time (V/Q_L 0.06) required to achieve a desired change in the dissolved gas deficit can be obtained from:

$$(V/Q_L\, 0.06) = \frac{1}{(K_L a)_I} \left[\frac{(C^* - C_{in})_{AB,I}}{(C^* - C_{out})_{AB,I}} - 1 \right] \qquad (9)$$

The $(C^*)_{AB,i}$ used in Equations 8 and 9 represents the saturation concentration for a specific gas species within the enclosure. These concentrations will vary with enclosure pressure (CP) and gas composition (χ_i)(Equation 7). Therefore, when designing equipment to satisfy selected changes in both DO and DN, it is necessary to find the combination of CP and χ_i that meets the $(C^*_{O2})_{AB}$ and $(C^*_{N2})_{AB}$ requirements simultaneously (Fig. 2) and then solve for the appropriate G/L. Application of Equation 8 reveals the fraction of the dissolved gas deficit remaining after treatment will decrease exponentially with increases in $(K_L a)_i$ or (V/Q_L 0.06) (Fig. 3). It is also apparent that increasing the product $(K_L a)_i$ (V/Q_L 0.06) beyond 3 or 4 does little to improve the extent of gas transfer (Fig. 3).

Along with an analysis of gas transfer rates, performance of oxygen contact equipment is often assessed in terms of oxygen absorption efficiency, the ratio of mass of oxygen absorbed to mass oxygen applied (AE, %), and transfer efficiency, the ratio of mass oxygen absorbed to energy input (TE, kg O_2/kW•h). Elevating G/L generally increases at a diminishing rate ΔDO, ΔDN, and TE but reduces AE (Fig. 4). Use of a sub-atmospheric enclosure pressure (CP) will enhance nitrogen desorption while decreasing both AE and TE (Fig. 4). Thus both oxygen and power requirements increase as the required nitrogen desorption rates increase. The performance of the mixer or surface agitator employed can be characterized by calculating standard aeration efficiency (SAE) (Boyd and Watten 1989):

$$SAE = \frac{[(K_L a\, C^*)_{O2,20^\circ C}\, V\, 10^{-3}]}{PW_{mixer}} \qquad (10)$$

Here the C^*_{O2} used represents the standard air saturation concentration at 20°C and one atmosphere of pressure. Typical SAE values for propeller and paddle wheel type agitators range between 1.15 and 2.25 kg O_2/kW·h (Boyd and Watten 1989).

Design Procedures

The design of an enclosed surface agitation system must include a value for $(K_La)_{20}$ that corresponds to the selected agitator type. Enclosure pressure and the oxygen feed rate must also be identified. A required change in DO and DN can be achieved using a number of different combinations of these variables. Therefore, the engineers task is to identify the site specific combination that minimizes operating costs as well as the risk of system failure. This is accomplished with the design procedure developed here (Fig. 5, Procedure 1) by establishing oxygen and nitrogen stripping rates, selecting a $(K_La)_{20}$ value and then calculating the required CP, G/L, and resultant performance indicators. The calculation procedure is then repeated using alternate values of $(K_La)_{20}$, and the results plotted as in Figure 6. Establishing performance indicators such as TE or transfer costs will require a previously identified correlation among agitator cost, power requirements, and $(K_La)_{20}$.

A second design procedure (Fig. 5, Procedure 2) was developed to establish the sensitivity of a given systems performance to changes in G/L when CP is known. This second procedure also provides a means of checking values of CP and G/L calculated for selected $(\Delta DO)_{AB}$ values using Design Procedure 1. For Design Procedure 2, CP and $(K_La)_{20}$ are identified for the system being modeled. An initial selection of the mole fraction of oxygen χ_{O2} within the enclosure is made allowing resultant changes in DO and DN to be calculated along with the required G/L. The procedure is then repeated by using increasing or decreasing values of χ_{O2} and the results are plotted as in Figure 7. Selecting a low value of χ_{O2} will result in a relatively high AE, and relatively small values of $(\Delta DO)_{AB}$, $(\Delta DN)_{AB}$, G/L and TE (Fig. 7).

Performance algorithms used in both Design Procedures 1 and 2 were developed by applying a materials balance on the gas and liquid phases of the system along with the Ideal Gas Law and Equations 5-8. In the analysis both gas and liquid phases of the system were treated as being homogenous. Further, the effect of DC on system performance was ignored (Watten et al. 1991) as was the effect of other gases present in trace concentrations. Performance predictions agree with those established by using the computer simulation program of Watten et al. (1990) previously verified by 114 observed versus model predicted effluent DO and DN comparisons. Several calculation steps are identical to those used in the packed column design protocol described previously (Watten 1990). Steps that are common to both contactor types are included here for the readers convenience.

Design Procedure 1

Step 1-1. Select target changes in DO and DN concentration on the basis of a desired oxygen drop across the rearing vessel, expected rearing unit hydraulics, and criteria for individual and total dissolved gas pressures (Table 1). Calculate the required change in DO (mg/l) as follows:

$$(\Delta DO)_{AB} = ECOC + (DO_{out})_{RV} - (DO_{in})_{AB} \quad (11)$$

If plug-flow rearing units are used, the required $(\Delta DN)_{AB}$ is based on the ΔP of water being treated, and a selected value of ΔP for the water entering the head end of the rearing vessel — i.e.,

$$(\Delta DN)_{AB} = -\left(\frac{\Delta DO\, F_{O2})_{AB} + \Delta P_{in})_{AB} - (\Delta P_{out})_{AB}}{F_{N2}}\right) \quad (12)$$

Alternatively, if mixed-flow rearing units are employed, $(\Delta DN)_{AB}$ can be based on the DN concentration in the supply water and a target ΔP and DO concentration in the rearing unit effluent:

$$(\Delta DN)_{AB} = -\left(\frac{BP + (\Delta P_{out})_{RV} - (DO_{out})_{RV}(FO_{O2}) - VP}{F_{N2}}\right) \cdot (DN_{in})_{AB} \quad (13)$$

If DO concentrations must be increased without a change in ΔP, the required ΔDN is then simply the product of ΔDO and the critical stripping ratio:

$$(\Delta DN)_{AB} = (\Delta DO)_{AB}(-F_{O2}/F_{N2}) \quad (14)$$

The calculation procedure requires that the calculated $(\Delta DO)_{AB}$ value is positive and that the corresponding $(\Delta DN)_{AB}$ value is negative. Further, the required change in DN must be less than the influent DN concentration.

Step 1-2. Calculate the required dissolved oxygen and nitrogen transfer rates using $(\Delta DO)_{AB}$, $(\Delta DN)_{AB}$, and the system water flow rate:

$$(GTR)_{O2} = (\Delta DO)_{AB}\, Q_L\, (6x10^{-2}) \quad (15)$$

$$(GTR)_{N2} = (\Delta DN)_{AB}\, Q_L\, (6x10^{-2}) \quad (16)$$

Step 1-3. Using a selected value of $(K_La)_{20}$ and $(GTR)_{O2}$ and $(GTR)_{N2}$ values from Step 1-2, solve for the required saturation concentrations of DO and DN within the enclosure:

$$C^*_{O2} = \left(\frac{(GTR)_{O2}}{(K_La)_{20} \propto 1.024^{T-20} V 10^{-3}}\right) + (DO_{out})_{AB} \quad (17)$$

$$C^*_{N2} = \left(\frac{(GTR)_{N2}}{(K_L a)_{20} \varnothing_{N2} \propto 1.024^{T-20} V 10^{-3}} \right)$$

$$+ (DN_{out})_{AB} \quad (18)$$

Selecting a relatively high value of $(K_L a)_{20}$ will result in a relatively low CP and G/L and a relatively high required power input.

Step 1-4. Using Henry's Law, calculate gas phase partial pressure ratios for both O_2 and N_2 that correspond to the saturation concentrations established in Step 1-3. The partial pressure ratios are defined here as:

$$P_{O2} = ((C^*)_{O2} / \beta_{O2} K_{O2} 1000)_{AB} \quad (19)$$

$$P_{N2} = ((C^*)_{N2} / \beta_{N2} K_{N2} 1000)_{AB} \quad (20)$$

The product of $\beta_i K_i 1000$ is given in Table 2 as a function of temperature (fresh water) for both oxygen and nitrogen.

Step 1-5. The required absolute pressure within the enclosure can now be calculated by using P_{O2}, P_{N2}, and a water vapor pressure (VP) value from Table 2:

$$CP = 760 (P_{O2} + P_{N2}) + VP \quad (21)$$

Step 1-6. Calculate the gas phase mole fraction (χ_i) of both oxygen and nitrogen, using CP and the C* values from Step 1-3:

$$\chi_{O2} = (C^*_{O2})_{AB} / \left(\frac{CP - VP}{760} \right) (\beta_{O2} K_{O2} 1000) \quad (22)$$

$$\chi_{N2} = 1 - \chi_{O2} \quad (23)$$

Step 1-7. Establish the required molar flow rate of oxygen $(QM)_{O2}$ into the enclosure:

$$(QM)_{O2} = \left(\frac{\chi_{O2}}{\chi_{N2}} \right) \left(\frac{|(\Delta DN)_{AB}| QL}{(MW)_{N2}} \right) + \frac{(\Delta DO)_{AB} QL}{(MW)_{O2}} \quad (24)$$

Step 1-8. Convert the molar flow rate of oxygen to a standard volumetric flow rate (21.1°C and 760 mm Hg) using the Ideal Gas Law:

$$QV = 0.0821 QM (273.15 + 21.1) \quad (25)$$

Step 1-9. Calculate the effective gas-liquid ratio (G/L) as follows:

$$G/L = (QV/QL) 100 \quad (26)$$

Step 1-10. Standard performance indicators can now be established, including AE and TE:

$$AE = \frac{(\Delta DO)_{AB} QL 100}{(MW)_{O2} QM} \quad (27)$$

$$TE = \frac{(\Delta DO)_{AB} (QL) (6 \times 10^{-5})}{PW} \quad (28)$$

Total energy input (PW) represents the sum of the power required to operate the agitator ($PW_{agitator}$), to pump water through the enclosure (PW_{pump}) and to extract off-gas ($PW_{compressor}$) when the enclosure pressure is less than the local atmospheric pressure. Neglecting the effects of friction in the water supply line and assuming the combined pump and motor efficiency is 0.7, PW_{pump} can be estimated by using the following equations:

$$PW_{pump} = [H (\gamma)_T (QL/60{,}000)] / 0.7 \quad (29)$$

$$H = (|CP - BP| 0.0136) + Z \quad (30)$$

The specific mass of water $(\gamma)_T$ required is given in Table 2. An estimate of the energy required to extract off-gas can be obtained from the adiabatic compression formula described by Yunt (1979):

$$PW_{compressor} = \frac{QRT_i}{Ne} \left[\left(\frac{P_O}{P_i} \right)^N - 1 \right] \quad (31)$$

The mass flow rate (Q) of off-gas required above is related to the composition and molar flow rate of the off-gas as follows:

$$Q = (QN) [(\chi_{O2} 0.032) + (\chi_{N2} 0.028)] \quad (32)$$

The molar flow rate of off-gas, QN, is obtained from the required nitrogen desorption rate, water flow rate, and χ_{N2} by using the expression:

$$QN = \left(\frac{(|\Delta DN|)_{AB} (QL/60)}{(MW)_{N2}} \right) / \chi_{N2} \quad (33)$$

At this time total, variable, and fixed costs should be calculated using local power and commercial oxygen costs along with calculated values of AE and TE.

Design Procedure 2

Step 2-1. Identify $(K_L a)_{20}$ for the selected surface agitation system.

Step 2-2. Select a gas phase χ_{O2} value (0.25 to 0.9) and calculate the corresponding value of χ_{N2}—i.e.,

$$\chi_{N2} = 1 - \chi_{O2} \quad (34)$$

Step 2-3. Given χ_{O2} and χ_{N2}, calculate the resultant saturation concentrations of DO and DN within the enclosure by using Henry's Law, the selected CP (absolute), and solubility coefficients presented in Table 2:

$$(C^*_i)_{AB} = 1000\, K_i\, \beta_i\, \chi_i\, \frac{(CP - VP)}{760} \qquad (35)$$

Step 2-4. Calculate the absorber effluent DO and DN using $(C)_{AB}$ values from Step 2-3, $(K_L a)_{20}$ values from Step 2-1, and the following expression:

$$(C_{out})_{AB} = (C^*_i)_{AB} - \left[\left(\frac{1}{1 + ((K_L a)_{20}\, \Theta_i\, \alpha\, 1.024^{T-20}\, (V/Q_L 0.06))}\right) \cdot ((C^*_i)_{AB} - (C_{in})_{i-AB})\right] \qquad (36)$$

Step 2-5. Given influent and effluent DO and DN, calculate $(\Delta DO)_{AB}$ and $(\Delta DN)_{AB}$, and then solve for the required molar flow rate of oxygen $(QM)_{O2}$ into the enclosure as in Step 1-7.

Step 2-6. Convert the molar flow rate of oxygen to a standard volumetric flow rate as in Step 1-8.

Step 2-7. Calculate the effective gas-liquid ratio as in Step 1-9.

Step 2-8. Calculate the standard performance indicators as in Step 1-10.

Step 2-9. Establish a ΔP value for the contactors effluent using Equation (1) to ensure that total dissolved gas pressures are acceptable (Table 1).

Step 2-10. If desired, select a new value of χ_{O2} and repeat the design calculations starting at Step 2-2.

Literature Cited

APHA. 1985. Standard methods for the examination of water and wastewater. 16th ed. American Public Health Association, Washington, D.C.

Bouck, G.R. 1980. Etiology of gas bubble disease. Trans. Am. Fish. Soc. 109:703-7.

Boyd, C.E., and B.J. Watten. 1989. Aeration systems in aquaculture. CRC Crit. Rev. Aq. Sci. 1:425-72.

Collins, C., G. Burton, and R. Schweinforth. 1984. Evaluation of liquid oxygen to increase channel catfish production in heated water raceways. Tennessee Valley Authority, Office of Natural Resources and Economic Development, Division of Air and Water Resources 84/8, Gallatin, Tennessee.

Colt, J. 1984. Computation of dissolved gas concentrations in water as functions of temperature, salinity and pressure. Am. Fish. Soc., Bethesda, Maryland, Special Publ. No. 14.

Colt, J., and B.J. Watten. 1988. Applications of pure oxygen in fish culture. Aquacult. Eng. 7:397-41.

Colt, J., K. Orwicz, and G. Bouck. 1991. Water quality considerations and criteria for high density fish culture with supplemental oxygen. Am. Fish. Soc. Symposium 10:372-38.

Gowan, R. 1987. Use of supplemental oxygen to rear chinook in seawater. Pages 35-39 in G. Bouck, editor. Papers on the use of supplemental oxygen to increase hatchery rearing capacity in the Pacific Northwest. Bonneville Power Administration, Portland, Oregon.

Krise, W.F. 1991. Hatchery management of lake trout exposed to chronic dissolved gas supersaturation. Am. Fish. Soc. Symposium 10:368-71.

Levenspiel, O. 1979. The chemical reactor omnibook. Oregon State University Book Stores, Inc., Corvallis, Oregon.

Lewis, W.K., and W.C. Whitman. 1924. Principles of gas adsorption. J. Ind. Eng. Chem. 16:1215-20.

Meade, J.W., B.J. Watten, W.F. Krise, and P.W. Halliburton. 1991. Use of oxygen-filled enclosures for aeration and degassing. Am. Fish. Soc. Symposium 10:459-64.

Petit, J. 1981. Utilization de l'oxygene pur en pisciculture. Pages 429-45 in K. Tiews, editor. Proceedings of the World Symposium on aquaculture in heated effluents and recirculation systems, Heenemann Gmbh, Berlin.

Sebert, P., L. Barthelemy, and C. Peyrand. 1984. Oxygen toxicity in trout at two seasons. Comp. Biochem. Physiol. 78A:719-22.

Stenstrom, M.K., and R.G. Gilbert. 1981. Effect of alpha, beta and theta factors upon the design, specification and operation of aeration systems. Water Res. 15:643-54.

Tsivoglou, E.C., R.L. O'Connel, C.M. Water, P.J. Godsil, and G.S. Logsdon. 1965. Tracer measurements of atmospheric reaeration-1. Laboratory Studies, J. Wat. Poll. Contr. Fed. 37:1343-62.

Watten, B.J. 1990. Design of packed columns for commercial oxygen addition and dissolved nitrogen removal based on effluent criteria. Aquacult. Eng. 9:305-28.

Watten, B.J., and L.T. Beck. 1985. Modeling gas transfer in a U-tube oxygen absorption system: effects of off-gas recycling. Aquacult. Eng. 4:271-97.

Watten, B.J., J.W. Meade, and C.E. Boyd. 1990. Modeling the effects of serial off-gas reuse on the performance of a hooded surface oxygen absorption systems. Aquacult. Eng. 9:97-120.

Watten, B.J., J. Colt, and C.E. Boyd. 1991. Modeling the effect of dissolved nitrogen and carbon dioxide on the performance of pure oxygen absorption systems. Am. Fish. Soc. Symposium 10:474-81.

Weitkamp, D.E., and M. Katz. 1980. A review of dissolved gas supersaturation literature. Trans. Am. Fish. Soc. 109:659-702.

Yunt, F.W. 1979. Gas flows and power measurements. Pages 105-27 in W.C. Boyle, editor. Proceedings Workshop toward an oxygen transfer standard, EPA-600/9-78-021, U.S. Environmental Protection Agency, Cincinnati, Ohio.

Table 1. Dissolved gas criteria for intensive culture conditions (Colt et al. 1991).

	CONDITIONS	
Parameter	Cold water (12°C)	Warm water (25°C)
DO (low)	5-6 mg/l	3-4 mg/l
DO (high)	21 mg/l	16 mg/l
	300 mm Hg	300 mm Hg
DC	20 mg/l	20 mg/l
ΔP (all life stages)	10 mm Hg	20 mm Hg
ΔP (specific life stages)		
eggs	45 mm Hg	*
sac fry	35 mm Hg	20 mm Hg
early juveniles	10 mm Hg	50 mm Hg
advanced juveniles	< 30 mm Hg	*
*Unknown.		

Table 2. Water vapor pressure and dissolved oxygen and nitrogen solubility parameters as a function of water temperature (Colt 1984). T = water temperature, °C; γ = specific mass of water, kN/m³; VP = water vapor pressure, mm Hg; β = Bunsen's coefficient, liter gas/l water at 760 mm Hg pressure; β•K•1000 = gas solubility, mg/l at a partial pressure of 760 mm Hg; F = gas tension per mg/l, mm Hg.

		DISSOLVED OXYGEN			DISSOLVED NITROGEN			
T	γ	VP	β	βK1000	F	β	βK1000	F
5	9.806	6.54	0.04302	61.477	12.363	0.02100	26.259	28.937
6	9.806	7.01	0.04196	59.962	12.675	0.02053	25.671	29.611
7	9.806	7.51	0.04095	58.519	12.989	0.02007	25.096	30.285
8	9.805	8.05	0.03998	57.133	13.304	0.01963	24.548	30.961
9	9.805	8.61	0.03905	55.804	13.621	0.01921	24.021	31.637
10	9.804	9.21	0.03816	54.532	13.938	0.01881	23.521	32.312
11	9.803	9.85	0.03730	53.303	14.257	0.01842	23.033	32.988
12	9.802	10.52	0.03649	52.145	14.576	0.01806	22.583	33.662
13	9.801	11.23	0.03570	51.016	14.895	0.01770	22.133	34.335
14	9.799	11.99	0.03495	49.945	15.215	0.01736	21.707	35.006
15	9.798	12.79	0.03423	48.916	15.535	0.01704	21.307	35.676
16	9.796	13.64	0.03354	47.930	15.855	0.01672	20.907	36.342
17	9.795	14.53	0.03288	46.987	16.175	0.01642	20.532	37.006
18	9.793	15.48	0.03224	46.072	16.494	0.01614	20.182	37.666
19	9.791	16.48	0.03163	45.200	16.812	0.01586	19.832	38.322
20	9.789	17.54	0.03105	44.371	17.130	0.01559	19.494	38.974
21	9.787	18.66	0.03048	43.557	17.446	0.01534	19.182	39.621
22	9.785	19.83	0.02994	42.785	17.762	0.01510	18.881	40.264
23	9.783	21.08	0.02942	42.042	18.076	0.01486	18.851	40.907
24	9.780	22.39	0.02892	41.328	18.388	0.01460	18.294	41.532
25	9.778	23.77	0.02844	40.642	18.699	0.01442	18.031	42.156
26	9.775	25.22	0.02798	39.984	19.007	0.01421	17.769	42.775
27	9.772	26.75	0.02754	39.355	19.314	0.01401	17.519	43.386
28	9.770	28.36	0.02711	38.741	19.618	0.01382	17.281	43.990
29	9.767	30.06	0.02670	38.155	19.919	0.01363	17.043	44.587
30	9.764	31.84	0.02630	37.583	20.218	0.01345	16.818	45.175

Nomenclature

A	Gas-liquid interfacial area (m^2)
AE	Oxygen absorption efficiency (%)
BP	Atmospheric pressure (mm Hg)
C	Dissolved gas concentration (mg/l)
CP	Enclosure pressure, absolute (mm Hg)
C*	Dissolved gas saturation concentration (mg/l)
d	Rate of change (dimensionless)
D	Diffusion coefficient (m^2/h)
DC	Dissolved carbon dioxide concentration (mg/l)
DN	Dissolved nitrogen concentration (mg/l)
DO	Dissolved oxygen concentration (mg/l)
e	Combined efficiency of compressor and motor (dimensionless)
ECOC	Cumulative dissolved oxygen drop across the rearing unit(s) (mg/l)
F	Dissolved gas tension (mm Hg/mg•l)
GTR	Gas transfer rate (kg/h)
G/L	Volumetric gas liquid ratio (%)
H	Total hydraulic head (m H_2O)
HP	Hydrostatic pressure (mm Hg)
k	Isentropic index for gas mixture (dimensionless)
K	Ratio of molecular weight to volume for a gas (mg/ml)
K_La	Overall mass transfer coefficient (h^{-1})
MW	Molecular weight (mg/mole)
N	(k-1)/k (dimensionless)
P	Partial pressure ratio, $(C)_{AS}/\beta_i \bullet K_i \bullet 1000$ (dimensionless)
P_i	Absolute compressor inlet pressure (kPa)
P_o	Absolute compressor outlet pressure (kPa)
PW	Total power input (kW)
$PW_{compressor}$	Power used to exhaust off-gas (kW)
PW_{mixer}	Power used to agitate water (kW)
PW_{pump}	Power used in pumping water (kW)
Q	Mass flow rate of off-gas (kg/s)
QL	System water flow rate (l/min)
QM	Molar flow rate of oxygen (moles/min)
QN	Molar flow rate of off-gas (moles/s)
QV	Volumetric flow rate of oxygen (l/min @ 21.1°C and 760 mm Hg)
R	Gas constant (0.287 kJ/kg•°K)
SAE	Standard aeration efficiency (kg O_2/kW•h)
t	Time (h)
T	Water temperature (°C)
T_i	Absolute temperature of gas at compressor inlet (°K)
TE	Oxygen transfer efficiency (kg/kW•h)
V	Volume of liquid in contactor (m^3)
VP	Vapor pressure of water (mm Hg)
X	Mole fraction in gas phase (dimensionless)
Z	Head loss across the contactor (m)
α	$(K_La)_{20,FW}/(K_La)_{20,CW}$ (dimensionless)
β	Bunsen coefficient (litres gas/l water at 760 mm Hg absolute pressure)
γ	Specific mass of water (kN/m^3)
Δ	Liquid film thickness (m)
ΔP	Uncompensated differential dissolved gas pressure (mm Hg)
ΔDO	Change in DO concentration (mg/l)
ΔDODN	Stripping ratio, _DO/_DN (dimensionless)
ΔDN	Change in DN concentration (mg/l)
$Ø_i$	(dimensionless)

Subscripts and Superscripts

AS	Packed column absorber
CW	Clean water
FW	Field water
i	Gas species identifier
in	Influent
out	Effluent
OX	Oxygen
NI	Nitrogen
RV	Rearing vessel
T	Temperature (°C)

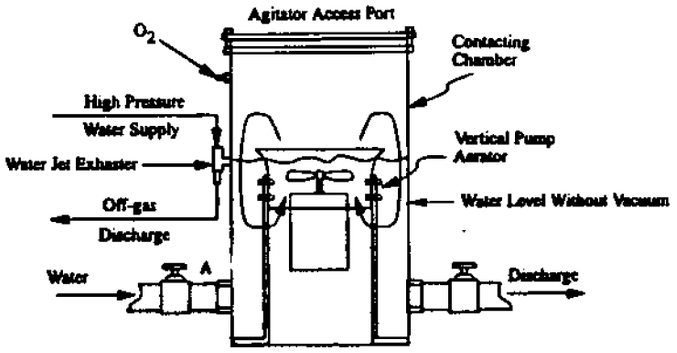

Figure 1. Major components of an enclosed propeller type surface agitator designed for contacting oxygen with water at subatmosphere pressure.

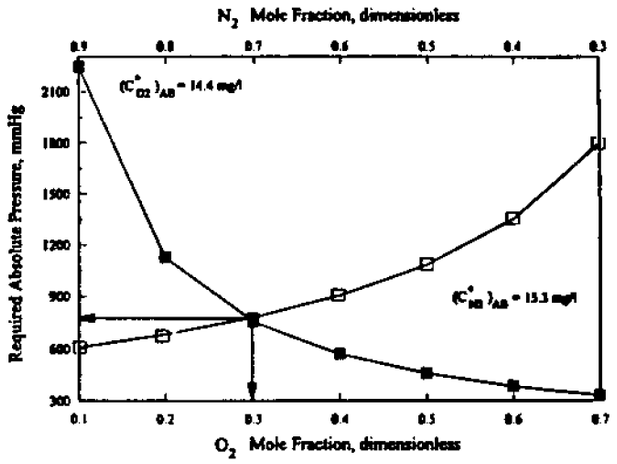

Figure 2. Combinations of gas composition (χ_i) and enclosure pressure (CP) that provide dissolved oxygen (DO) and dissolved nitrogen (DN) saturation concentrations of 14.4 and 15.3 mg/l, respectively (T=15°C). The design point marked represents the combination χ_i and CP that provides the required DO and DN saturation concentrations simultaneously.

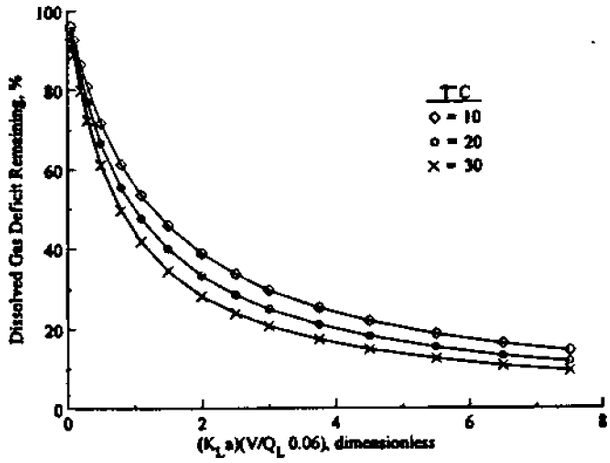

Figure 3. Dissolved gas deficit remaining after treatment $(C - C_{out})_{i,AB}$ versus $K_L a$ (V/Q 0.06) at three temperatures.

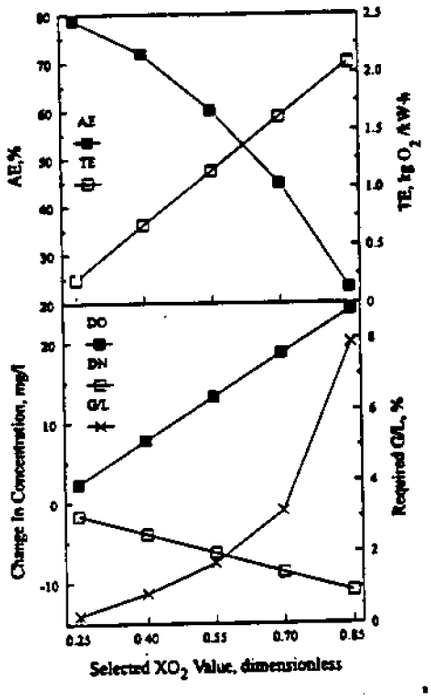

Figure 4. Effect of enclosure pressure and oxygen feed rates on the performance of an enclosed surface oxygenation system (T = 15°C; Q_L = 190 l/min; $(K_La)_{20}$ = 100/h; V = 0.085 m^3; DO_{in} = 6.0 mg/l; DN_{in} = 16.36 mg/l).

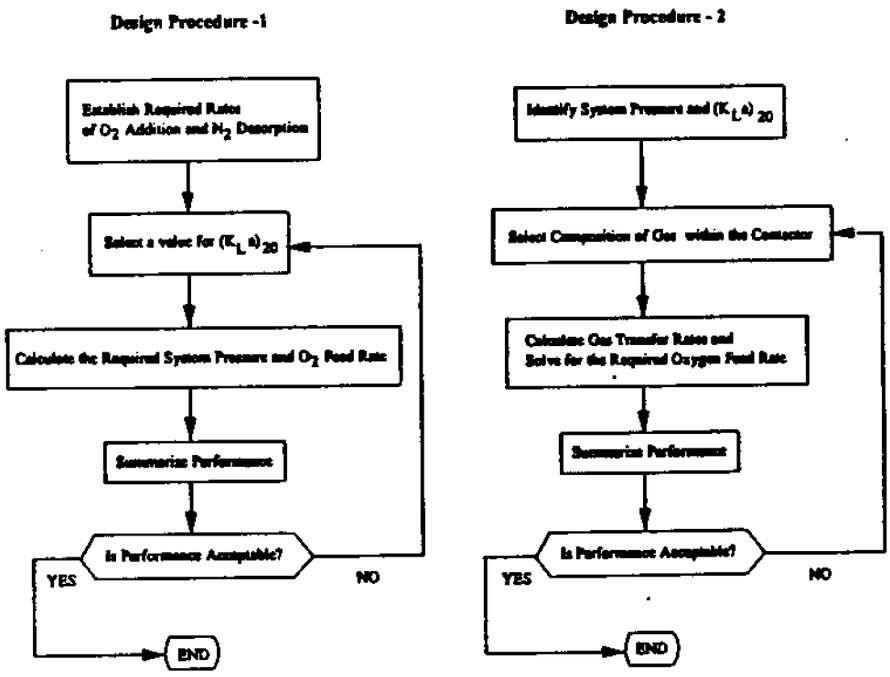

Figure 5. Computation sequence for design procedure 1 and 2.

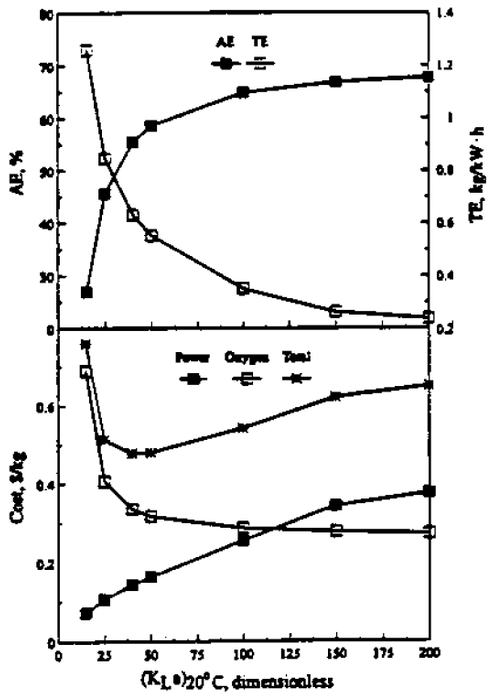

Figure 6. Effect of selected $(K_L a)_{20}$ values on transfer costs, oxygen absorption efficiency (AE), and transfer efficiency (TE) when operating with an effluent DO and DN of 16.0 mg/l and 14.6 mg/l, respectively (T = 15°C; barometric pressure = 760 mm Hg; DO_{in} = 5 mg/l; DN_{in} = 19.6 mg/l; V - 2.5 m^3; Q_L = 1000 l/min; power cost = $0.09/kWh; oxygen cost = $0.1856/kg).

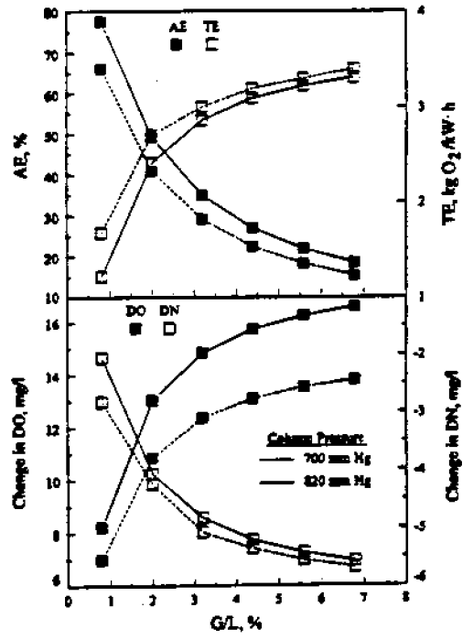

Figure 7. Effect of selected oxygen mole fraction values ($\chi 2_{O2}$) on effluent dissolved oxygen (DO), effluent dissolved nitrogen (DN), oxygen absorption efficiency (AE), transfer efficiency (TE) and the required gas-liquid ratio (G/L) (T = 15°C; enclosure pressure = 760 mm Hg; $(K_L a)_{20}$ = 50/h; power applied = -.125 kW; Q_L = 189 l/min; V = 0.8 m^3; DO_{in} = 9 mg/l; DN_{in} = 17.9 mg/l).

Carbon Dioxide Removal for Intensive Aquaculture

Raul H. Piedrahita
Gary R. Grace
University of California
Davis, California

Abstract

The management of carbon dioxide in intensive aquaculture tends to be ignored and may be achieved more as an incidental outcome of other management activities than as a deliberate objective. Carbon dioxide dissolved into the water tends to depress the pH, and as the intensity of culture operations increases, so does the magnitude of the pH drop caused by the carbon dioxide produced by the fish.

Carbon dioxide exists in solution as part of a chemical equilibrium system, the carbonate system, and this makes the analysis and study of removal mechanisms more difficult than for other gases such as oxygen or nitrogen. Estimates of the amount of carbon dioxide that must be removed to achieve a given pH or carbon dioxide concentration must take into account considerations of water chemistry as well as gas exchange. In addition, carbon dioxide constitutes a very small proportion of a normal atmosphere (~ 0.03 %) and this presents special difficulties in designing carbon dioxide removal techniques for aquaculture.

Procedures are recommended for carbon dioxide removal and are based on studies carried out with a packed column aerator. The test column was 3.0 m high, 0.2 m in diameter, was filled with 1.59 cm Pall™ rings, and was used with a counterflow of air. The column was open to the atmosphere at the top which prevented pressurization. Tests were carried out at hydraulic loading rates comparable to those recommended for oxygen addition. Gas (air) flow rates were varied over a broad range and a minimum gas to liquid (volume to volume) flow rate ratio of 5.0 was found to be suitable for carbon dioxide removal. The columns need not be deeper than approximately 1.5 m. Removal of carbon dioxide under these conditions caused a reduction in CO_2 concentration of over 90 % of the incoming value. The alkalinity of the water (and the total carbonate carbon concentration), the depth of the column, the G/L ratio and the initial carbon dioxide concentration were found to affect the efficiency of removal.

Introduction and Background

The removal of carbon dioxide from culture water in intensive aquaculture systems has not been given much attention in the commercial or scientific communities. For the most part, whatever carbon dioxide control is exerted in intensive aquaculture systems takes place as an incidental result of management actions designed for other purposes, such as aeration. As a result, there is a dearth of published information on CO_2 removal and control in intensive systems. The design guidelines presented here are based on laboratory experiments carried out with a 3.0 m tall, 0.2 m diameter laboratory column filled with 1.59 cm Pall ¬ rings, and on very limited field application of various types of packed columns. A packed column was selected as the degassing method because it facilitated the quantification of the carbon dioxide removal rates, the study of the effect of water quality and operational parameters on the rates of removal, and analysis of the effects of chemical reactions and gas exchange rates on the overall rate of carbon dioxide removal from culture water. In addition, packed columns have been extensively studied and are widely used in aquaculture for both oxygenation and degassing. An initial consideration in the work was to determine if carbon dioxide removal was compatible with aeration (oxygen addition) as simultaneous operations in a packed column.

The operation of PCA's has been well described (e.g., Hackney and Colt 1982, Nirmalakhandan et al. 1988; Watten and Boyd 1989; Watten and Boyd 1990; Watten 1990) for gases such as nitrogen (N_2) and oxygen (O_2). Traditionally, CO_2 transfer has been approximated following an approach similar to that used for nitrogen gas, where mass transfer rates are assumed to be proportional to molecular diameter or molecular diffusivity of the gases (Colt and Bouck 1984; Watten 1990). In addition, CO_2 removal has been estimated by assuming the equilibrium reaction between CO_2 and H_2CO_3 (carbonic acid) to be instantaneous (Colt andBouck 1984; Howe 1986). These studies have noted that the actual transfer may be slower due to kinetic effects.

In a packed column, the gas exchange rate is limited by transfer at the gas-liquid interface and can be modeled using the Whitman two-film model (Hackney and Colt 1982). Hackney and Colt (1982) adapted the two-film model to PCA's by noting that within a certain range (flow rates less than that which caused flooding)

the transfer of oxygen was independent of flow rate, but depended on column depth and on an overall mass transfer coefficient. A simplified form of the equation is:

$$\ln\{(C_s-C_i)/(C_e-C_i)\} = KZ \qquad (1)$$

where

C_s = saturation concentration ($g.m^{-3}$)
C_i = influent concentration ($g.m^{-3}$)
C_e = effluent concentration ($g.m^{-3}$)
K = overall PCA mass transfer coefficient (m^{-1})
Z = column height (m)

The value of K has been found to vary with temperature and with water quality, and measured values are normally standardized to 20° C and clean water (Colt and Bouck 1984; Boyd and Watten 1989; Watten 1990):

$$K(20)_{ow} = K(T)_{fw} * 1.024^{(20-T)} / \alpha \qquad (2)$$

where

$K(20)_{ow}$ = overall clean water PCA mass transfer coefficient at 20° C (m^{-1})
$K(T)_{fw}$ = overall mass transfer coefficient as measured in the field (m^{-1})
T = water temperature during determination of K(T)fw (° C)
α = ratio between the gas transfer coefficient in field water and in clean water

The value of K can be estimated from Equation (1) using measurements of influent and effluent concentrations to a column and information on saturation concentration inside the column. Typical K values for oxygen have been estimated (eg. Hackney and Colt 1982; Watten 1990) for various packing media types and sizes. An alternate method for estimating the transfer coefficient may be adapted from the procedure suggested by ASCE (1984). In this procedure, a non-linear regression method isused to estimate the gas transfer coefficient and the saturation concentration (Grace and Piedrahita 1989). The saturation concentration obtained with the non-linear method constitutes an "effective" value for the mass transfer model, but does not necessarily represent the saturation concentration at a specific point in the column.

The mass transfer coefficient generally is considered to be constant under operating conditions in which gas flows and liquid flows are less than those which cause flooding for a specific gas transfer system . In addition, reduced gas transfer may be caused by G/L ratios so low that the gas composition inside the column is significantly altered resulting in changes in the saturation concentration. For gases other than oxygen, the overall transfer coefficient is usually considered to be proportional to the molecular diffusivity or molecular diameter of the species transferred (Tsivoglou et al. 1965; Colt and Bouck 1984; Watten 1990). Based on molecular diffusivities, the mass transfer rate coefficient for carbon dioxide should be 0.78, with a possible range between 0.65 and 0.98 associated with the uncertainties in the estimates of molecular diffusivities available (Perry et al. 1984), whereas based on molecular diameter, the ratio should be approximately 0.91 (Weast and Astle 1980). These values contrast with that recommended in the literature, of 1.00 (Speece and Humenick 1973; Colt and Bouck 1984'. Other effects such as chemical reactions may produce an apparent transfer coefficient which differs from the true mass transfer coefficient. While the true mass transfer coefficient may be difficult to determine, the apparent transfer coefficient can be approximated with a linear or non-linear regression of sample data using Equation 1.

The Need for Carbon Dioxide Removal

Carbon dioxide is a waste product of metabolism, and under certain conditions in intensive aquaculture operations may be the factor limiting fish density. If dissolved oxygen is eliminated as a possible water quality limiting factor by the use of pure oxygen injection, then unionized ammonia and carbon dioxide concentration may become the limiting factors. Simplified mass balances may be carried out to illustrate these situations. Dissolved oxygen consumption, total ammonia nitrogen and carbon dioxide production rates may be expressed as functions of feed in a tank as 0.2, 0.03 and 0.28 kg/kg of feed respectively (Colt 1986). If pure oxygen is used and dissolved oxygen concentration is maintained at safe levels, and if no carbon dioxide is removed in the process of oxygen injection, then a mass balance on carbon dioxide and unionized ammonia can be carried out as illustrated in Figure 1 for fish being fed at 2 % body weight per day, and for an alkalinity of 4 meq/L. InFigure 1, the unionized ammonia and carbon dioxide concentrations are allowed to rise without treatment as the fish loading increases. As a result, carbon dioxide depresses pH and unionized ammonia concentration is kept below 0.010 mg-N/L considered as a "safe" level (Colt 1987) for fish loading rates over 15 kg/L/min. Carbon dioxide, on the other hand, rises over the recommended "safe" level of 30 mg/L as the fish loading rate exceeds approximately 3.7 kg/L/min, which would become the calculated maximum loading rate in the system. Similar calculations can be made if the carbon dioxide concentration in the culture system is controlled and not to allowed to exceed 30 mg/L. The resulting unionized ammonia concentration reaches a level of 0.010 mg-N/L at a loading rate of approximately 11.5 kg/L/min, an increase of over 200 % over the original allowable loading rate of 3.7 kg/L/min (Figure 2).

A related factor that can become limiting is the pH value. As carbon dioxide is added to the water, the pH is depressed. The magnitude of the drop depends primarily on the alkalinity of the water. The pH value for the same conditions as in Figure 2 is shown in Figure 3. for cases in which the CO_2 concentration is controlled

at 30 mg/L. At an alkalinity of 4.0, pH reaches a value of 6.5 at approximately the same fish loading rate as the 30 mg/L CO_2 concentration is reached. For lower alkalinities, the pH values drop rapidly (Figure 3), reaching values that may be considered stressful to fish at fish loading rates that are likely to be lower than for carbon dioxide or unionized ammonia concentrations. As an example, the unionized ammonia concentration for a system in which carbon dioxide concentration is not allowed to rise above 30 mg/L is shown for three alkalinity values (Figure 4). At the low alkalinities, unionized ammonia nitrogen remains low for the range of fish loading rates considered.

The sensitivity of nitrifier bacteria to pH is an additional reason for controlling pH and carbon dioxide concentration in systems with biological filters. Nitrifier bacteria have been shown to have a narrow pH range for optimum activity (eg. approximately 7.0 to 8.0 for nitrosomonas and 7.5 to 8.5 for nitrobacter (Grady and Lim 1980). The decline in nitrification activity also has been shown to be very sensitive to pH outside the optimum ranges mentioned, with decreases as high as 50 % over less than 0.5 pH units (Grady and Lim 1980).

It is clear then, that controlling carbon dioxide concentration can result in increased loadingrates under certain conditions. The increased loading rates achievable with carbon dioxide control will depend on the particular situation and specifically on the rates of CO_2 and ammonia production by the fish and on the "safe" levels for those parameters. Additional factors affecting the loading rates and how they are affected by carbon metabolite production, are parameters such as alkalinity and temperature, that determine how the carbon dioxide produced by the fish affects pH.

Carbon Dioxide Removal

Unlike other important dissolved gases (N_2, O_2), carbon dioxide exists in solution as part of a chemical equilibrium system. As CO_2 is stripped from solution, a shift in the carbonate carbon equilibrium occurs. The carbonate carbon equilibrium is described by a pH dependent set of relationships between carbon dioxide, carbonic acid (H_2CO_3), bicarbonate (HCO_3^-), and carbonate (CO_3^{-2}) (Stumm and Morgan 1981, Snoeyink and Jenkins 1980).

$$CO_{2(gas)} \_ CO_{2(aq)} \quad (3)$$

$$CO_{2(aq)} + H_2O \_ H_2CO_3 \quad (4)$$

$$H_2CO_3 \_ H^+ + HCO_3^- \quad (5)$$

$$HCO_3^- \_ H^+ + CO_3^{-2} \quad (6)$$

Total carbonate carbon (C_tCO_3) is defined as the sum of aqueous carbon dioxide, carbonic acid, bicarbonate and carbonate concentrations:

$$C_tCO_3 = [H_2CO_3^*] + [HCO_3^-] + [CO_3^{-2}] \quad (7)$$

where:

$$[H_2CO_3^*] = [H_2CO_3] + [CO_{2(aq)}] \quad (8)$$

and the square brackets denote molar concentration. An additional term that is often used in calculations involving the carbonate system is alkalinity (ALK), which may be defined in a simplified way as:

$$ALK = [HCO_3^-] + 2[CO_3^{-2}] + [OH^-] - [H^+] \quad (9)$$

and has units of equivalent per liter (eq/L, or more commonly meq/L) (Stumm and Morgan 1981, Snoeyink and Jenkins 1980). Alkalinity is easily measured and is often used to estimate total carbonate carbon concentration and the concentration of the various species of the carbonate system. The concentration of each system species is dependent on the total carbonate carbon (or alkalinity) and a pH dependent "ionization" fraction indicating the proportion of the total carbon that is present in each form. As an example, the concentration of carbon dioxide (in mg/L) as a function of pH and alkalinity of the water is shown in Figure 5 for 25° C. From this figure, one can see that for a given pH, the concentration of carbon dioxide present is strongly dependent on alkalinity, and will be higher as the alkalinity increases. Conversely, a given concentration of carbon dioxide may be present in solution at different pH values, depending on the alkalinity of the water. In addition, the pH change associated with a given carbon dioxide concentration change depends on the initial conditions (alkalinity, pH and CO_2 concentration) and on the magnitude of the concentration change. If the alkalinity is relatively high, pH may be within biologically acceptable levels, but CO_2 may be present at concentrations that can affect oxygen transfer to the blood (Spotte 1979) (see Figure 5).

Temperature is an additional factor affecting the relationship between carbon dioxide and pH. Figure 5 corresponds to water at 25° C. Use of Figure 5 to estimate the carbon dioxide concentration at a temperature of 15° C for example, would result in an underestimation of CO_2 concentration of approximately 18 %. For approximate calculations, the CO_2 values obtained from Figure 5 can be increased by 1.8% for each degree below 25° C, and decreased by 1.8 % for each degree over 25° C.

When carbon dioxide is removed from solution, the total carbonate carbon is reduced by an amount equal to the carbon removed as CO_2. The removal of CO_2, however, causes a pH shift and the ionization fractions change. The result is that the amount of CO_2 transferred is not equal to the change in CO_2 concentration. For a packed column, or a segment of one:

$$\Delta[CO_2] = C_tCO_3^{in} * a_{in} - C_tCO_3^{out} * a_{out} \quad (10)$$

$$CO_2 \text{ removed} = C_tCO_3^{in} - C_tCO_3^{out} \quad (11)$$

where,

$\Delta[CO_2]$ = change in CO_2 concentration
C_iCO_{3i} = initial total carbonate carbon
C_iCO_{3f} = final total carbonate carbon
α_{oi} = initial CO_2 ionization fraction
α_{of} = final CO_2 ionization fraction

The magnitude of the pH change for a given carbon dioxide addition or removal will depend on the alkalinity of the water. In general, the higher the alkalinity, the less pronounced the pH change is for a given addition or removal of CO_2.

Design Guidelines

As has been mentioned, the design guidelines presented here are based on experiments carried out in a 3 m tall, 0.2 m diameter column filled with 1.59 cm Pall rings (Figure 6). To facilitate extrapolation of the data to other packed column designs and packing media, an attempt was made to relate the CO_2 transfer properties to dissolved oxygen transfer characteristics that may be readily available or easily obtained experimentally. All oxygen transfer rate coefficients were obtained using the non-linear approximation method based on recommendations developed by ASCE (1984), whereas carbon dioxide transfer rate coefficient calculations were based on mass balances for column sections between ports where samples were obtained. Details of the experimental set up and calculation procedures are available elsewhere (Grace and Piedrahita, 1989; Grace, 1991).

Data analysis for oxygen runs included determination of the overall transfer coefficient and a mass balance between gas and liquid phases for each column segment between sample ports. A spreadsheet was developed to perform the mass balance and calculate the following information at each sample port (ie. specific column depth Z): gas mole fraction, oxygen saturation concentration, K for the column segment and K from the top of the column to depth Z.

Data analysis for CO_2 included the calculation of CO_2 from measurements of pH and alkalinity and mass balance calculations to estimate the CO_2 content of the gas phase inside the column based on the measured CO_2 concentrations at the sampling ports. The mass balance spreadsheet was used to calculate the same parameters as in the O_2 mass balance including the gas transfer coefficients.

Twenty three oxygen runs and 54 carbon dioxide runs were performed with various G/L ratios and liquid flow rates. Oxygen runs provided useful results which described the performance of the laboratory PCA and could be used as a reference for the carbon dioxide runs. The oxygen runs also allowed comparison with previously published PCA studies (eg. Hackney and Colt 1982). Oxygen transfer was measured in the PCA for liquid flow rates between 18.9 and 66.2 L/min (hydraulic loading rates of 39.7 to 139.3 $m^3m^{-2}h^{-1}$) and G/L ratios of 0.16 to 8.19. Average mass transfer coefficients, as determined by the non-linear regression method, and corrected to 20° C (K_{20}) ranged from 1.86 to 2.31 m^{-1}. The lower values were obtained for G/L ratios less than 1.0, and are probably distorted values due to changes in the dissolved oxygen saturation concentration in the column caused by the low G/L ratio (Hackney and Colt 1982). Differences in the transfer coefficients also were caused by media packing and by temperature. The limited oxygen data generated in this study suggests that a value of 1.007 as opposed to the 1.024 used in Equation 2 would provide a more accurate description of gas transfer coefficient changes as affected by temperature in the column used.

Removal of carbon dioxide was investigated for G/L ratios between 0.14 and 17.9 and for alkalinities between 0.36 and 5.93 meq/L. The calculated mass transfer coefficients ranged from 0 to 2.09 m^{-1} for the top 1.07 m section of the column, 0.11 to 1.85 m^{-1} for the top 1.54 m, and from 0.17 to 1.32 m^{-1} for the full column depth. The wide range of transfer coefficients calculated suggests that there were factors that were not properly accounted for in the calculations of mass transfer coefficients for CO_2 in the column. The CO_2 percent removal at a given depth in the column was found to be affected primarily by G/L ratio (Figures 7 and 8). It was also affected by alkalinity and by temperature. The G/L ratio significantly affected the carbon dioxide stripping capability of the laboratory PCA, particularly for G/L ratios less than 5.0 (Figures 7 and 8). The decreased efficiency of transfer at the low G/L ratios was due to the increase in CO_2 in the gas phase, and the resulting change in saturation concentration. As an example, at an alkalinity of 0.50 and a G/L ratio of 0.52, the influent carbon dioxide mole fraction was measured as 0.00041 (saturation concentration = 0.69 mg/L) and the effluent mole fraction was calculated as 0.0187 (saturation concentration = 31.5 mg/L) based on a mass balance for the PCA. The influence of G/L for ratios above 5.0 was greatly reduced but still noticeable. Again, as an example, for an alkalinity of 0.50 meq/L, the CO_2 mole fraction in the gas phase increased from 0.00040 at the base of the column to 0.0023 and 0.0012 (corresponding to CO_2 saturation concentrations of 3.91 and 2.08 mg/L) for G/L ratios of 7.6 and 17.1 respectively. The effect of alkalinity was minor when compared to that of G/L ratios (Figures 7 and 8), but resulted in noticeable differences in CO_2 removal efficiencies for the column. The efficiency of carbon dioxide removal decreased as the alkalinity increased (Figure 9), and although the decrease is not very high, it represents important differences that must be kept in mind during the design process. At the highest alkalinities tested (4.7 and 5.9 meq/L), the efficiency of removal remains below 90 % even at the end of three meter column, while it exceeds 90 % by the 1.5 m depth for the lower alkalinity waters (0.5 and 2.1 meq/L) (Figure 9). The decline apparently is due to the increased reservoir of carbonate carbon present at higher alkalinities. While the actual rate of

mass transfer may not be affected by the carbon present in forms other than CO_2 (due to the relative speed of the hydroxylation reactions when compared to the rate of mass transfer or the residence time in the column), the efficiency of removal is affected since some of the carbon present as carbonate and bicarbonate will react and form CO_2 after the water exits the column. The carbon present as bicarbonate or carbonate does not have enough time to react within the column and readjust to the new pH created by the CO_2 removal and result in increased carbon dioxide removal from the column. Achievement of chemical equilibrium takes place after the water has exited the column, and some CO_2 is produced from reactions of the carbonate system. As a result, as alkalinity increases and the reservoir of total carbonate carbon is greater for a given carbon dioxide concentration, the efficiency of carbon dioxide removal decreases, if this efficiency is estimated based on the change in CO_2 concentration between the influent and effluent streams for the column (Figure 9).

Calculation of mass transfer coefficients was based on mass balances and on the application of Equation 1. Estimates of saturation concentration in the column are needed in addition to measurements of carbon dioxide concentration to be able to calculate the gas transfer coefficient. In addition, information is needed on the rate of the chemical reactions affecting carbon dioxide in solution, and on how that rate compares to the mass transfer rate and to the flow rate through the system. As indicated above and elsewhere (Grace 1991), evidence from the column suggests that the rate of chemical hydroxylation of carbon dioxide is slow relative to the rate of flow through the column and to the rate of mass transfer. Under that set of conditions, mass transfer rates may be estimated from Equation 1 using carbon dioxide concentrations and assuming that no chemical reaction takes place inside the column.

The gas transfer coefficients obtained for CO_2 using the methods described above were observed to change primarily with G/L ratio, with column depth and with alkalinity of the water (Grace and Piedrahita 1989; Grace 1991). The values declined rapidly for G/L ratios below approximately five. They also tended to stay approximately constant for the top 1.8 m of the column, but decreased for the bottom column segments. The calculated coefficients also decreased as alkalinity increased. Calculations were carried out with an implicit assumption of plug flow for the gas phase being used in the mass balances. The calculated values of gas transfer coefficient were expected to remain constant for thehydraulic loading rate used. Variations obtained in the calculated results are thought to be due primarily to gas flow characteristics that are not accounted for with the assumption of plug flow used for the mass balances and the calculation of gas transfer rates.

Accurate calculation of the mass transfer coefficient is dependent on a close estimate of the saturation concentration and how it changes in the column. Saturation concentrations, in turn, depend on the mole fraction of the gas in the column. For relatively high G/L ratios, the mole fraction of the gases does not change significantly inside the column, hence saturation concentration stays approximately constant throughout the column and a single value may be used for calculating the gas transfer coefficient at any point in the column. This is the case for oxygen transfer in aeration columns at G/L ratios over approximately 1 (Hackney and Colt 1982). For carbon dioxide stripping, on the other hand, the mole fraction in a normal atmosphere is low (~0.03 %) and the amount removed from solution is sufficient to cause a significant change in the saturation concentration through the column. Not being able to estimate the changing saturation concentration at different points in the column makes the calculation of the gas transfer coefficient uncertain. The change in saturation concentration is noticeable for all G/L ratios tested, but the difficulty is especially noticeable as G/L ratios decrease below approximately five and the saturation changes become larger.

Predicting how the saturation concentration changes in the column requires detailed knowledge of the gas flow properties in the column: ie. whether it is plug or mixed flow, or some intermediate model. If the changes in saturation concentration can be predicted, then accurate estimates of gas transfer rate coefficients can be made, and the rate coefficients should be independent of alkalinity and G/L ratio, and should be constant for hydraulic loading rates that are below the critical flooding value. With accurate estimates of the gas transfer coefficient for carbon dioxide, CO_2 concentrations can be estimated given values of G/L ratio, column depth, CO_2 content in air being injected into the column, and incoming water pH and alkalinity (or CO_2 concentration) values. The procedure follows a iterative process to arrive at CO_2 concentrations that are consistent with changes in CO_2 content in the gas phase. Uncertainties in the estimation of the CO_2 gas transfer coefficient for packed columns, and the need to use a complex iterative calculation procedure make the process impractical for designpurposes at the present time. At the present time, a more practical approach is to base estimates of carbon dioxide removal in packed columns on information such as that presented in Figures 7 and 8. The iterative procedure will become more useful and reliable as more information becomes available and as procedures are developed to relate the oxygen and carbon dioxide transfer coefficients, and to predict gas flow conditions inside the packed column.

Summary

In implementing a carbon dioxide removal scheme, it is important to realize that the flow and operational requirements are quite different from those for oxygen addition through aeration with atmospheric air or from pure oxygen injection. As has already been mentioned, even if a PCA is used for oxygen injection or aeration, the G/L requirements for carbon dioxide removal are

one to two orders of magnitude greater than for aeration or oxygen injection. In addition, the flow rates required for the two operations can be very different depending on efficiencies and on operational requirements. Figure 10 is an example of the ratio between the flow rate required for carbon dioxide control and for oxygen injection when pure oxygen is used (calculation details are given in the figure legend). In this example, the flow requirement for oxygen addition is greater than for carbon dioxide only at fish loading rates of less than about 5 kg/L/min. At higher loading rates, the flow required to control carbon dioxide concentration and maintain it at 30 mg/L is higher than for oxygen injection.

Carbon dioxide stripping appears to have been affected significantly by changing gas composition inside the packed column. As CO_2 transfers out of solution into the countercurrent gas flow, the partial pressure of CO_2 in the gas phase increases and the saturation concentration increases, reducing the driving force for gas transfer. The lower the G/L ratio the greater the effect on the total gas composition. For the lowest G/L ratios the effluent gas had a CO_2 partial pressure and saturation concentration approximately equivalent to the influent dissolved concentration (e.g., mole fraction of 0.035 in the gas leaving the column and corresponding saturation concentration of 53 mg CO_2/L, for an influent concentration of 74.6 mg CO_2/L). For the high G/L ratios such as 17.2, effluent partial pressure of CO_2 corresponded to a saturation concentration of 2.07 mg/L (influent and effluent CO_2 concentrations of 30.0 and 2.19 mg/L respectively).

The importance of describing CO_2 transfer will increase as fish densities in culture systems are increased. Carbon dioxide is a very soluble gas and it isrelatively simple to remove from solution. System designers must be aware, however that conventional equations for packed column aerators overestimate the amount of CO_2 removed. The result will be a higher CO_2 concentration in the culture system than predicted.

Packed columns used for pure oxygen may not offer sufficient opportunity for CO_2 to be removed. At the low G/L ratios characteristic of pure oxygen or of aeration systems (usually below G/L = 1.0), very little CO_2 is transferred as the test results indicate. A G/L ratio greater than 5.0 is recommended to remove CO_2 in a PCA, and the higher the alkalinity, the higher the G/L ratio needs to be. Until the gas flow characteristics in the column can be described better, column design may be carried out using guidelines as shown in Figures 7, 8 and 9, where over 90 % of the influent carbon dioxide is removed in a column that is less than 1.5 m deep as long as the G/L ratio is above approximately 5 and the alkalinity is low (2.1 meq/L or lower in the experimental trials). The efficiency or removal is noticeably lower for higher alkalinities. In all cases, an increase in removal efficiency of approximately 10 % is achieved by increasing the column depth from 1.0 to 1.5 m, while the increase obtained by increasing the column depth to 3.0 m is less than 5 %, and shallow columns (depth less than 1.5 m) are recommended for carbon dioxide removal. Further work to characterize gas flow rates should make possible the use of more accurate design procedures such as those used for oxygen injection systems in packed columns.

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Literature Cited

American Public Health Association, American Water Works Association and Water Pollution Control Federation (1985). Standard methods for the Examination of Water and Wastewater. 16th Edition. 1268 pp.

American Society of Civil Engineers (1984). A Standard for the Measurement of Oxygen Transfer in Clean Water. ASCE, New York, 39 pp.

Boyd, C.E. and Watten, B.J. (1989). Aeration system in aquaculture. CRC Crit. Rev. Aq. Sci., 1:425-472.

Colt, J. (1984). Computation of Dissolved Gas Concentrations in Water as Functions of Temperature, Salinity, and Pressure. American Fisheries Society Special Publication 14.

Colt, J. (1986). An introduction to water quality management in intensive aquaculture. In Oxygen supplementation, a new technology in fish culture. U.S. Department of the Interior, Fish and Wildlife Service, Region 6, Information Bulletin #1. Denver, Colorado.

Colt, J. and Bouck, G. (1984). Design of Packed Columns for Degassing. Aquacultural Engineering 3: 251-273.

Grace, G.R. (1991). Carbon dioxide stripping in a packed column aerator. M.S. thesis (in preparation).

Grace, G.R. and Piedrahita, R.H. (1989). Carbon dioxide removal in a packed column aerator. American Society of Agricultural Engineers Paper No. 89-7011. ASAE, Saint Joseph, Michigan. 21 pp.

Grady, C.P.L. and Lim, H.C. (1980). Biological wastewater treatment. Theory and applications. Marcel Dekker, Inc. New York. 963 pp.

Hackney, G. and Colt, J. (1982). The Performance and Design of Packed Column Aeration Systems for Aquaculture. Aquacultural Engineering 1: 275-295.

Howe, K. J., (1986). An Equilibrium Approach to pH Effects in Gas Transfer. M.S. Thesis. University of Texas, Austin. 141pp.

Nirmalakhandan, N., Lee, Y. H., and Speece R. E., (1988). Optimizing Oxygen Adsorption and Nitrogen Desorption in Packed Towers. Aquacultural Engineering, 7: 221-234.

Perry, R.H., Green, D.W. and Maloney, J.W., editors (1984). Perry's Chemical Engineer's Handbook, 6th edition. McGraw-Hill, New York.

Snoeyink, V. L., and Jenkins, D. (1980). Water Chemistry. Wiley & Sons, New York. 463 pp.

Speece, R.W. and Humenick, M.J. (1973). Carbon dioxide stripping from oxygen activated sludge systems. J. Wat. Poll. Contr. Fed. 45:412-423.

Spotte, S., (1979). Fish and Invertebrate Culture. 2nd Edition, Wiley & Sons, New York. 179 pp.

Stumm, W. and Morgan, J. J., (1981). Aquatic Chemistry. 2nd Edition. Wiley & Sons, New York. 780 pp.

Tsivoglou, E.C., O'Connell, R.L., Walter, C.M., Godsil, P.J., and Logsdon, G.S. (1965). Tracer measurements of atmospheric reaeration- I. Laboratory studies. J. Wat. Poll. Contr. Fed. 37:1343-1362.

Watten, B.J., and Boyd, C.E., (1989). Gas phase axial dispersion in a packed column oxygen absorber: Model development and application. Aquacultural Engineering, 8:421-434.

Watten, B.J., and Boyd, C.E. (1990). Gas transfer within a multi-stage packed column oxygen absorber. Aquacultural Engineering, 9:33-59.

Watten, B.J. (1990). Design of packed columns for commercial oxygen addition and dissolved nitrogen removal based on effluent criteria. Aquacultural Engineering, 9:305-328.

Weast, R. C. and Astle, M.J., editors (1980). CRC Handbook of Chemistry and Physics. 60th edition. CRC Press, Boca Raton, Florida.

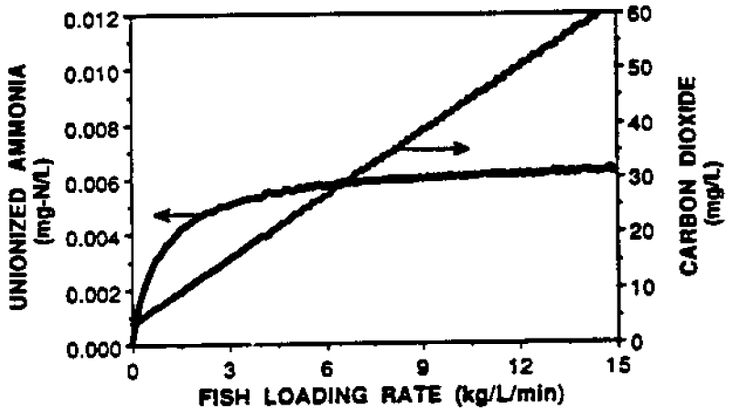

Figure 1. Carbon dioxide and unionized ammonia concentrations as functions of fish loading rate. Influent water supply is at 25°C, pH of 7.5 and free of total ammonia nitrogen. Additional assumptions and conditions for the calculation are stated in the text.

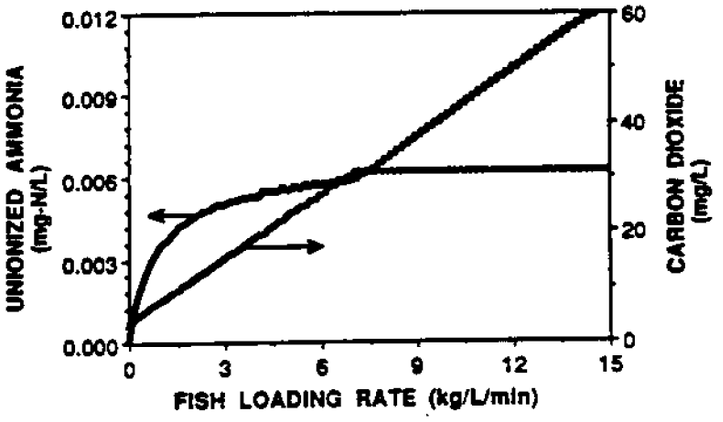

Figure 2. Carbon dioxide and unionized ammonia concentrations as functions of fish loading rate when the CO_2 concentration is controlled and maintained at 30 mg/L. Influent water and other assumptions as in Figure 1.

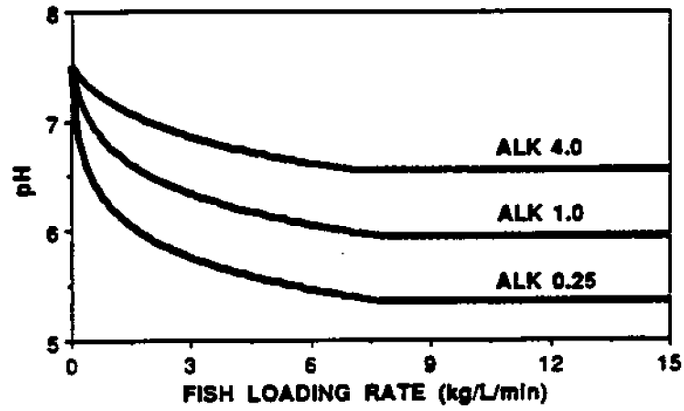

Figure 3. Effluent pH resulting from CO_2 production by fish. The calculations are repeated for alkalinity values of 0.25, 1.0 and 4.0 meq/L. Other assumptions and conditions for the calculations are as in Figure 2, where CO_2 is not allowed to rise over 30 mg/L.

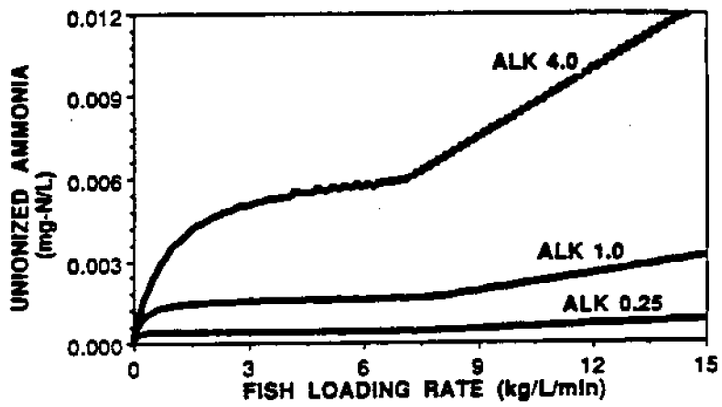

Figure 4. Unionized ammonia nitrogen concentration for fish stocked in three different waters. In all cases the carbon dioxide concentration is not allowed to rise over 30 mg/L. Assumptions and conditions are as for the previous figures.

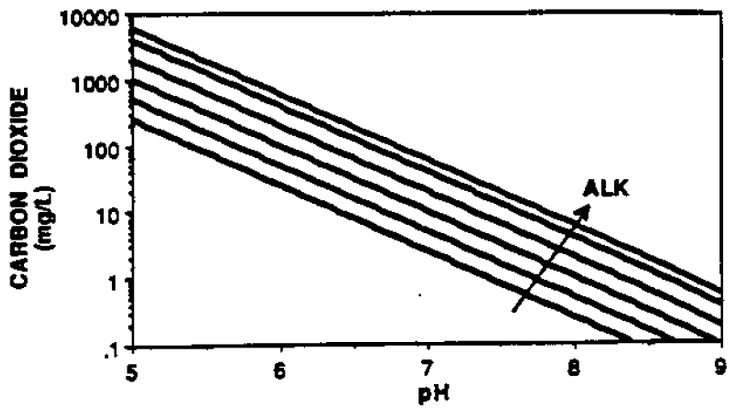

Figure 5. Carbon dioxide concentration as a function of pH and alkalinity for fresh water at 25°C. The lines correspond to alkalinities of 0.25, 0.50, 1.0, 2.0, 4.0 and 6.0 meq/L, with increasing alkalinities as indicated in the graph.

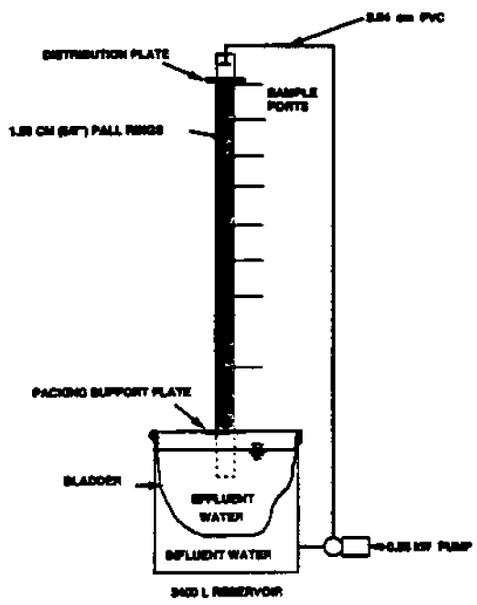

Figure 6. Packed column installation used for aeration and degassing tests.

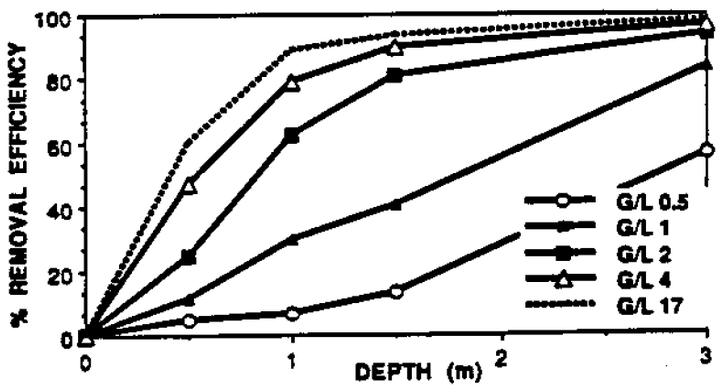

Figure 7. Carbon dioxide removal efficiency at different depths in the column, for G/L ratios between 0.5 and 17. Water alkalinity for these runs is 0.5 meq/L.

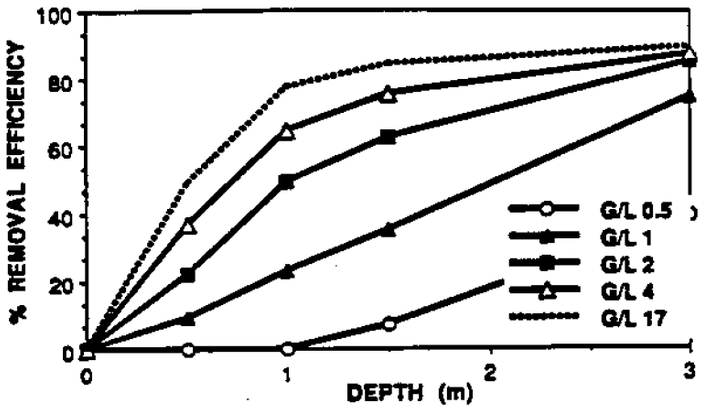

Figure 8. Carbon dioxide removal efficiency at different depths in the column, for G/L ratios between 0.5 and 17. Water alkalinity for these runs is 5.9 meq/L.

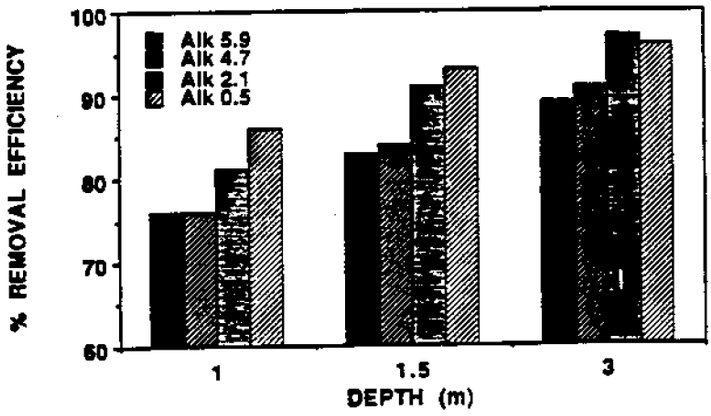

Figure 9. Maximum carbon dioxide removal efficiency obtained for different alkalinities. Removal efficiency was estimated from measured CO_2 concentrations.

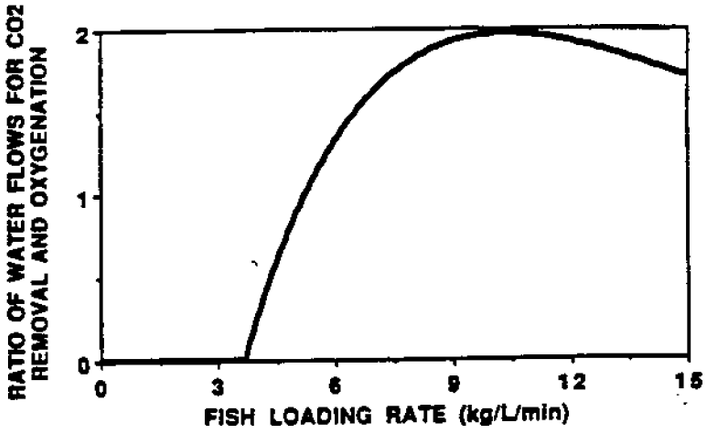

Figure 10. Ratio between the flow required for carbon dioxide control and for oxygen injection. Return oxygen concentration from the oxygen injection system is 700% of saturation; dissolved oxygen concentration in the culture tank is maintained at 150% of saturation; carbon dioxide is not allowed to rise above 30 mg/L; and the removal efficiency of the CO_2 removal process is 90%. Influent water and other assumptions are as for Figure 1.

LIST OF PARTICIPANTS

Supporting Agencies

Dr. James McVey, Program Director (Aquaculture)
U.S. Department of Commerce
NOAA
National Sea Grant College Program
1335 East West Highway
Silver Spring, MD 20910
Phone: (301) 427-2451
FAX: (301) 427-2799

Dr. John Nickum, Aquaculture Coordinator
U. S. Fish and Wildlife Service
4401 N. Fairfax Drive
Arlington, VA 22203
Phone: (703) 358-1878
FAX: (703) 358-2210

Dr. O'Neil Smitherman, Aquaculture Specialist
U. S. Department of Agriculture
Cooperative Research Service
901 D Street S. W.
Washington, D. C. 20024
Phone: (202) 401-4929
FAX: (202) 401-5179

Organizational Committee

Dr. Ronald F. Malone, Workshop Chairman
Department of Civil Engineering
Louisiana State University
3508 CEBA
Baton Rouge, LA 70803-6405
Phone: (504) 388-8666
FAX: (504) 388-8662

Mr. Ronald E. Becker, Associate Director
Louisiana Sea Grant College Program
Louisiana State University
Baton Rouge, LA 70803
Phone: (504) 388-6345
FAX: (504) 388-6331

Dr. Shulin Chen
Department of Civil Engineering
Louisiana State University
2401 CEBA
Baton Rouge, Louisiana 70803-6405
Phone: (504) 388-6498
FAX: (504) 388-8662

Mr. Douglas Drennan
Department of Civil Engineering
Louisiana State University
Baton Rouge, LA 70803-6405
Phone: (504) 388-8528
FAX: (504) 388-8662

Ms. Pamela Rupert,
Speaker/Manuscript Coordinator
Department of Civil Engineering
Louisiana State University
3508 CEBA
Baton Rouge, LA 70803-6405
Phone: (504) 388-8662
FAX: (504) 388-8662

Ms. JoPaula Lantier, Conference Coordinator
Louisiana Sea Grant College Program
Louisiana State University
Baton Rouge, LA 70803-7507
Phone: (504) 388-6445
FAX: (504) 388-6331

Speakers

Mr. John Hawke
School of Veterinary Medicine
Louisiana State University
Baton Rouge, LA 70803
Phone: (504) 346-3281
FAX: (504) 346-5715

Dr. Tom Lawson
Department of Agricultural Engineering
Louisiana State University
Baton Rouge, LA 70803-4505
Phone: (504) 388-3153
FAX:

Dr. George Libey
Department of Fisheries & Wildlife Services
Virginia Polytechnic Institute and State University
146 Cheatham Hall
Blacksburg, VA 24061-0321
Phone: (703) 231-6400
FAX: (703) 231-3330

Dr. Thomas Losordo
College of Agriculture and Life Sciences
North Carolina State University
Campus Box 7646
Raleigh, North Carolina 27695-7646
Phone: (919) 515-7587
FAX: (919) 515-5110

Dr. Patrick O'Rourke
Departmentof Agriculture
Illinois State University
Rm. 150 Turner Hall
Normal, IL 61761-6901
Phone: (309) 438-5654
(309) 438-3492
FAX: (309) 438-5037

Mr. David Owsley
U.S. Fish and Wildlife Service
Dworshak National Fish Hatchery
P.O. Box 18
Ahshaka, ID 83520-0018
Phone: (208) 476-4591
FAX: (208) 476-3252

Dr. Raul Piedrahita
Department of Agricultural Engineering
University of California
Davis, CA 95616
Phone: (916) 752-2780
FAX: (916) 752-2640

Dr. Ken Roberts
Louisiana Cooperative Extension Service
Knapp Hall
Louisiana State University
Baton Rouge, LA 70803
Phone: (504) 388-2145 or 388-6348
FAX: (504) 388-2478

Dr. Tom Schwedler
Department of Aquaculture, Fish & Wildlife
Clemson University
Clemson, SC 29631
Phone: (803) 656-2810
FAX: (803) 656-0678

Dr. Ron Thune
Department of Veterinary Microbiology
& Parasitology
Louisiana State University
Baton Rouge, LA 70803
Phone: (504) 346-3308
FAX: (504) 346-5715

Dr. Michael B. Timmons
Cornell University
Riley Robb Hall
Ithaca, NY 14853
Phone: (607) 255-2801
FAX: (607) 255-4080

Mr. Steven Van Gorder
Fresh-Culture Systems, Inc.
12 Peach Street
Kutztown, PA 19530
Phone: (215) 683-7933
FAX: (215) 683-9280

Dr. Barnaby Watten
U.S. Fish and Wildlife Service
NFRDL
RD4 Box 63
Wellsboro, PA 16901
Phone: (717) 724-3322
FAX: (717) 724-2525

Dr. Dallas Weaver
Scientific Hatcheries
8152 Evelyn Circle
Huntington Beach, CA 92646
Phone: (714) 960-4171
FAX: (714) 960-8281

Dr. Fred Wheaton
University of Maryland
Ag Engineering Department
College Park, MD 20742
Phone: (301) 405-1189
FAX: (301) 314-9023

Participants

Dr. James Avault
School of of Forestry, Wildlife and Fisheries
Louisiana State University
B.R., LA 70803
Phone: (504) 388-4168

Mr. Jimmy Avery
Agricultural Extension Service
202-A Knapp Hall
Louisiana State University
Baton Rouge, LA 70803
Phone: (504) 388-2152
FAX: (504) 388-2478

Dr. Meryl Broussard
USDA/CGRS
Aerospace Center, Room 330-D
901 D Street
Washington, D.C. 20250
Phone: (202) 401-4061
FAX: (202) 401-4555

Dr. Earle Buckley
NCRI
P. O. Box 751
528 SW Mill, Suite 222
Portland, OR 97207
Phone: (503) 725-5725
FAX: (503) 725-5709

Mr. Babu Chitta
Louisiana State University
Department of Civil Engineering
Baton Rouge, LA 70803-6405
Phone: (504) 388-1154
FAX: (504) 388-8662

Dr. George Cline
Biology Department
University of Alabama at Birmingham
University Station
Birmingham, AL 35294
Phone: (205) 934-5250

Mr. Dave Coffin
Louisiana State University
Department of Civil Engineering
Baton Rouge, LA 70803-6405
Phone: (504) 388-8412
FAX: (504) 388-8662

Dr. Dudley Cuffey
School of Forestry, Wildlife and Fisheries
117 For-Wild-Fish Bldg.
Louisiana State University
Baton Rouge, LA 70803
Phone: (504) 388-4169

Mr. James Ebeling
Piketon Research & Extension Center
Ohio State University
P. O. Box 549
Piketon, OH 45661

Mr. Larry Fall
University of Southwestern Louisiana
Box 44432
Lafayette, LA 70504
Phone: (318) 231-6641

Dr. Harry Fischer
Rt. 665, Box 14
Moon, VA 23119
Phone: (804) 725-5820
FAX: (804) 725-5880

Mr. John Hargreaves
Department of Forestry, Wildlife and Fisheries
Louisiana State University
734 Stoney Creek
Baton Rouge, LA 70808
Phone: (504) 766-7916

Dr. John Heinen
Zeigler Brothers/Freshwater Institute
P.O. Box 1746
Shepherdstown, WV 25443
Phone: (304) 876-2125

Dr. Jay Huner
University of Southwestern Louisiana/
Crawfish Research Center
P.O. Box 44650
Lafayette, LA 44650-4650
Phone: (318) 281-5239
FAX: (318) 231-5395

Dr. Gary L. Jensen
USDA-Extension Service
Rm. 3863 S. Bldg.
Washington, DC 20250-0900
Phone: (202) 447-5004
FAX: (202) 475-4969

Dr. Wendell Lorio
Cooperative Extension Service
202 Knapp Hall
Louisiana State University
Baton Rouge, LA 70803
Phone: (504) 388-2152
FAX: (504) 388-2478

Dr. Greg Lutz
Cooperative Extension Service
202 Knapp Hall
Louisiana State University
Baton Rouge, LA 70803
Phone: (504) 388-2152
FAX: (504) 388-2478

Mr. Brian McCarthy
Louisiana State University
Veterinary Science
Baton Rouge, LA 70803
Phone: (504) 388-2575

Mr. Bruce McGoogan
940 Stanford Ave., Apt. 309
Baton Rouge, LA 70808
Phone: (504) 767-5880

Mr. Paul Medley
1006 S. Kenilworth Pkwy. #201
Baton Rouge, LA 70820
Phone: (504) 766-8810

Ms. Gwenn Merry
2000 Brightside Lane #2323
Baton Rouge, LA 70820
Phone: (504) 769-2450

Mr. John Ogle
Gulf Coast Research Lab
P.O. Box 7000
703 E. Beach Blvd.
Ocean Springs, MS 39564-7000
Phone: (601) 872-4280

Mr. Michael Pena
Louisiana State University
Veterinary School Bldg.
Baton Rouge, LA 70803
Phone: (504) 388-2575

Dr. Robert C. Reigh
School of Forestry, Wildlife, and Fisheries
Louisiana State University
Baton Rouge, LA 70803
Phone: (504) 388-4177

Mr. Vernon Rodrigue
Armant Aquaculture, Inc.
P. O. Box 2
Vacherie, LA 70090
Phone: (504) 265-4372
FAX: (504) 265-7212

Dr. Robert Romaire
School of Forestry, Wildlife and Fisheries
Louisiana State University
Baton Rouge, LA 70803

Ms. Kelly A. Rusch
Department of Civil Engineering
Louisiana State University
Baton Rouge, Louisiana 70803-6405
Phone: (504) 388-8528
FAX: (504) 388-8662

Dr. Godfrey Savage
Mechanical Engineering Department
University of New Hampshire
Durham, NH 03824
Phone: (603) 862-1356
FAX: (603) 862-2030

Mr. Curtis D. Stutzman
Aquaculture Learning Center
1375 Baxter Ave., N. W.
Amana, IA 52203
Phone: (319) 846-8098
FAX: (319) 846-8099

Mr. C. Theegala
Department of Civil Engineering
Louisiana State University
Baton Rouge, LA 70803-6405
Phone: (504) 388-8412
FAX: (504) 388-8662

Dr. Philip E. Turk
University of Texas Medical Branch
Marine Biomedical Institute
200 University Blvd. H-63
Galveston, TX 77551
Phone: (409) 772-2133

Dr. Jack Van Lopik, Executive Director
Louisiana Sea Grant College Program
Louisiana State University
Baton Rouge, Louisia 70803-7507
Phone: (504) 388-6710
FAX: (504) 388-6331

Dr. William Wolters
Department of Wildlife and Fisheries
Louisiana State University
Baton Rouge, LA 70803

Dr. Philip Westerman
Department of Biological and
Agricultural Engineering
North Carolina State University
Box 7625
Raleigh, NC 27695-7625
Phone: (919) 515-3121

Mr. Arthur M. Williams
Louisiana Department of Wildlife and Fisheries
P. O. Box 98000
Baton Rouge, LA 70803-9000
Phone: (504) 765-2329