Performance of Fine Sand Fluidized Bed Biological Filters

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Abstract

A fine sand fluidized bed biological filter operating at 143 l/min/m² 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 (Owlesly 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 its own level.

Looking closely at the individual sand grains, they appear to be in continual tree 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/m³ or 10 m²/filter). 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/R²). At these low rates, these fine media
filters can become oxygen limited. The author has not
found this to be a successful method of adding oxygen
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 m² 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 anaerobic condition.

The experimental system consists of two FBB's
(1.5 m diameter X 2.2 m high) operating at 326 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 individual tank. 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, should be regarded as a
minimum 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 measured over 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 (Kruener 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 of pH 5.4 to 7.8 when the filter is 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 CO₂ was not allowed to escape, thereby creating a very high free CO₂ concentration. These conditions are not representative of the steady state response, but can be considered indicative. At high pH, the high CO₂ 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 to the point where the system started to produce nitrite or ammonia.

Equipment, Materials and Methods

Oxygen and temperature measurements were made with Royce-Insutum 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 vehicle 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—Optionx Brain Board thru a Module A03. A Cole Parmer series 7142 pH controller and a Jenco model 98 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 Saa Test—test kits for low range and Hach reagents 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 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, or destroyed. Most of the $\# 1$ feeds 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 SilverCap starter and $\# 1$ feeds into the system over about a 13 hour period. How the waste products from this feed get distributed between 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 rationize 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 general BOD removal of the filter by observing the bioflocculation 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 bioflocculation 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 6.8 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 mEq/L equivalents per liter, with a starting pH of 6.5. 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 nitritification 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.

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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 FBBs 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 FBBs 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 unified 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


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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)

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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 aerosol, 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 (m³/m²/day or m³/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 m²) 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 (Hynel and Chou 1978).

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 foam 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 Frick (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 aeration - 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 and Libey 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 of the 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


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).

<table>
<thead>
<tr>
<th></th>
<th>Initial Wt. (gm)</th>
<th>Initial Wt. tank (kg)</th>
<th>Final Wt. (gm)</th>
<th>Final Wt. tank (kg)</th>
<th>Gain Wt. (gm)</th>
<th>Gain Wt. tank (kg)</th>
<th>Gain / day Wt. (gm)</th>
<th>Gain / day Wt. tank (kg)</th>
<th>FCR</th>
<th>Mort. (%)</th>
<th>F.W. Additions % Sys Vol/wk</th>
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<tr>
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<td>62.8</td>
<td>741.9</td>
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<td>377.3</td>
<td>679.1</td>
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<td>1.34</td>
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<td>45.7</td>
<td>542.1</td>
<td>487.9</td>
<td>491.3</td>
<td>442.2</td>
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<td>1.97</td>
<td>1.39</td>
<td>1.8</td>
<td>46.5</td>
</tr>
<tr>
<td>Low</td>
<td>43.3</td>
<td>19.5</td>
<td>676.1</td>
<td>304.2</td>
<td>632.8</td>
<td>284.7</td>
<td>2.83</td>
<td>1.27</td>
<td>1.32</td>
<td>2.7</td>
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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).

<table>
<thead>
<tr>
<th></th>
<th>Biomass (kg fish/m³)</th>
<th>RBC Mass Load (gm TAN/m²/day)</th>
<th>Ammonia Oxid. (gm TAN/m²/day)</th>
<th>Inlet Oxygen (mg/l)</th>
<th>Outlet Oxygen (mg/l)</th>
<th>BOD (mg/l)</th>
<th>TSS (mg/l)</th>
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<tr>
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28
Biofiltration and Solids Capture with Low Density Bead Filters

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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 g/m$^2$-day of TN conversion. With specific surface areas in the range of 1000 m$^2$/m$^3$, nitrification capacities on a volumetric basis (g/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 producing low cost 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 (Callinectes 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 unflagged prawns 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 many of rapidly removing excreted solids, expansion provides a mechanism for harvesting biofloc eliminating the threat of biofouling.

The upflow sand filter (Burdon 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 fltrate. (unit of flow per unit of horizontal cross-sectional area). During the normal filtration mode (0.20 - 0.41 m$^3$/m$^3$), the unit operates as an effective biofilter, reflecting the high specific surface area of the sand media. As the hydraulic transduction 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 fltrate (1.83 - 3.05 m$^2$/m$^3$) to fluidize the bed. The filter's solids capture capability eliminates the need for 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 fltrate. Fluidization of the coarse sand media (1.2 - 2.4 mm) occurs at fltrates below 0.41 m$^3$/m$^3$ 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/m²-day; this translates into about 2.25 g/nm²-day on a specific surface area basis. Management options are further limited by the high water loss associated with the washing sequence.

The fluctuate 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 1980) or by mechanical means (Malone 1982). Upon expansion, separation of the agglomerated 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 excess biofloc. The settling phase, with both pump and propellers inactive, 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 reinitiation of the filtration mode.

The bead filters are relatively insensitive to fluctuate. Fluxrates in the range of 1.43 m³/m² do not adversely impact filter performance (Wimberly 1990). Concerns about oxygen transport are alleviated. Oxygen delivery capabilities exceed about 14.12 kg/m²-day or over 11.84 g/m²-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. Nitrate nitrogen was determined with the sulfanilamide based colorimetric test. Dissolved oxygen levels were determined with a Yellow Springs Model 57 oxygen probe calibrated with the Winkler Titrination Method. Flow rates were measured with a stop watch and bucket. Total suspended solids (TSS) were determined gravimetrically.

Discussion

Performance

Intrinsic to advocate for 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 biomass 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 (Harrmcoes 1982).

Oxygen consumed during filtration, or OCF, can be effectively utilized to monitor levels of bacterial activity in submerged biofilters (Hiroyama 1965, 1974; Manthe et al. 1986). The term is normalized to the weight of organisms supported (or food consumed) empirically by the relationship:

$$\frac{\text{OLR}}{W} = \frac{\text{OCF}}{W}$$

where: OLR = mean oxygen demand exerted per unit mass of organisms at a fixed feeding rate, or alternatively, per unit of food consumed (mg-O₂ per kg per day); W = weight of organisms (or food) for the i observation (kilograms).

It can be reasonably assumed that waste production from organisms maintained at a fixed feeding rate is constant (Coit 1978). A high biofiltration burden generally reflects high levels of heterotrophic activity.
Depending on the species, OLRT 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 OLRT 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 OLRT dropped. The nitrification capacity of the biofilter increased because excessive heterotrophic 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 continued increases in backwash frequency would ultimately reduce the mean cell residence time of the slow-growing nitrification bacteria (Sharma and Ahlert 1977) causing a decline in carrying capacity. This hypothesis is not completely accepted as continually expanded fluidized beds demonstrate excellent nitrification abilities (Burden 1986; 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/NL. Bead filter nitrifier aerial 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 Ahlert 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 MPAL bead filter displayed capture ratios of averaging 69 (kg TSS captured/kg TAN converted) without showing any indication of biofouling. The MPAL facility displayed volumetric solids capture rates of 741.3 kg/m² without being optimized for solids capture. The assumption that much higher solids capture rates are obtainable with high flux rates 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 mixing constraints 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 flux rate on solids capture and nitrification, and
4. Clarification of the factors controlling biofilm thickness and nitrification capacities on an areal basis.
References


Figure 1. Operational Sequence for a Mechanically-washed Floating Bead Biofilter.
Table 1. Conceptual Evolution of Expandable Granular Biofilters

<table>
<thead>
<tr>
<th>Unit</th>
<th>Mode(s) of Operation</th>
<th>Limitation</th>
<th>References</th>
</tr>
</thead>
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<tr>
<td>Submerged Rock Filter</td>
<td>Filtration</td>
<td>Bioluminescence</td>
<td>Perry et al. 1979, Manthe et al. 1988</td>
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<tr>
<td>Uplow Sand Filter</td>
<td>Filtration Expansion</td>
<td>Oxygen transport waterloss</td>
<td>Burden 1988</td>
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<tr>
<td>&quot;Bead&quot; Filter</td>
<td>Filtration Expansion settling</td>
<td>Surface area</td>
<td>Cooley 1979, Wimberly 1990</td>
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Table 2. Increasing the Backflushing Frequency Increased the Carrying Capacity of Wimberly’s (1990) Hydraulically Washed Bead Filter.

<table>
<thead>
<tr>
<th>Backwash Frequency (day⁻¹)</th>
<th>Carrying Capacity Kg-Fish/m³ beads</th>
<th>Carrying Capacity Kg-Food/m³ beads</th>
<th>OCR gmO₂/m³ beads</th>
<th>MoCF Kg/m³ beads</th>
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<tr>
<td>1</td>
<td>500</td>
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<td>8</td>
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<td>Observation</td>
<td>Specific Surface Area (m²/m³)</td>
<td>Mean pH</td>
<td>Mean Effluent TAN (mg-N/l)</td>
<td>Annual TAN Conversion (g/m²-day)</td>
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<td>2350</td>
<td>7.47</td>
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Table 3. Bead Fillers Display Intermediate Nitritification Capacities
Sludge Management for Recirculating Aquacultural Systems

Shulin Chen, David E. Coffin, and Ronald F. Malone
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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. BOD7/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 fish production systems has raised the issue of environmental impacts. Although recirculating technologies can avoid many of the impacts of dilute aquacultural wastes (Iwama 1981) 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 solids or 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:

$$\frac{dS}{dt} = F \cdot (1-E) \times F \times P \cdot W_t \cdot K_d \times S + B_t (F \cdot x \cdot P) - h_s S$$

(1)

where,

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

Assuming steady-state conditions and solving for $S$:

$$S = \frac{(1-E) \times F \times P}{W_t \cdot K_d + h_s}$$

(2)

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

$$S = \frac{E + B_t \times F \times P}{K_d + h_s}$$

(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 BODg excretion rate can also generally be expressed as a ratio to the feeding rate (Table 2). The BODg is excreted in soluble and particulate forms. Based upon the study on channel catfish, Murphy and Lippert (1970) reported the soluble BODg as 58% of the total BODg excreted; whereas, BODg in particulate matter was 42%. Wimberly (1980) found that 23% of the BODg 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 BODg 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 1980) 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 BODg consumption, on the other hand, is more significant. For example, if 23% of the BODg (0.22 kg BODg/kg-feed) is produced in the soluble form as reported by Wimberly (1990), the total soluble BODg production will result in a bioclog production which averages 5% of the feeding rate. This portion of BODg 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 BODg removal. Assuming the BOD has a composition similar to typical municipal sludge, values for the bioclog production constant, BP, in the range of 0.08 to 0.12 can be reasonably assumed.

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

$$S_P = S_0 \times S$$  (4)

The concentration of the sludge stream or SC (Kg/m³), is determined by the efficiency of the sludge separation process and the amount of flushing or washdown waters (Qw in m³/day) required for the sludge removal.

$$S_C = S_P / Q_w$$  (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 BODg/TSS and TKN/TSS. The BODg/TSS ratio is a measure of the degree of stabilization of the sludge. The sludge production 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 Kp = 0.36 day⁻¹ and h = 0.35. The obtained Kp value is within the range of municipal waste (0.28 - 0.71 day⁻¹), Reynolds et al. (1974), and Hightower and Perry (1970). 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 BODg excreted from fish is partially treated in a recirculating system by a biological filter before being discharged as sludge; therefore, the BODg/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 feed contains 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 solid 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 flocculent filter. Samples were collected and analyzed for TSS, BODg, and TKN (APHA 1989). The results indicate that the BODg/TSS ratio of the sludge ranged from 0.09 to 0.20, while the TKN/TSS ratio ranged from 0.03 to 0.05. The variance in BODg is due to the age of the sludge. Filters with high solids retention produce a more stabilized sludge, lowering the BODg/TSS ratio. The measured TKN/TSS ratios are consistent with Kugelman and Van Gorder (1991) who reported TKN/TSS ratio ranging from 0.05 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 2.0 to 5.0% 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 sludge is the primary management approach employed by facilities currently
addressing the issue (Mudrack 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 BOD₅/TSS ratio) will be desirable in many applications. Although anaerobic digestion has been examined (Kugelmann and Van Gorden 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 (Sₑ) and sludge concentrations (Sₑ) 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 m³/m²·day (EPA 1975), the clarifier surface area needed per 1000 kg of catfish can be calculated as 0.004 to 0.04 m² according to the data given in Table 2. The solids loading design criteria is 72 kg/m²·day (Metcalf and Eddy 1979), which projects a clarifier surface area of 0.054 to 0.058 m² 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 BOD₅/TSS ratio of less than 0.2. The overflow from the secondary clarifier will have a TSS concentration range of 10 to 50 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 ineffective 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 (Kugelmann and Van Gorden 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.

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 surmive 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 BOD₅ loading. Design criteria are highly variable. Suggested design VSS loadings for poultry manure lagoons range from 0.064 to 0.161 kg VSS/m²·day (4 to 10 lb VSS/1000 ft²·day). The BOD₅ loadings range from 225 to 625 kg/hr·day (200 to 1000 lbs BOD₅/ac·day). According to the sludge production rate in Table 2, the volume needed for aquacultural waste will be approximately 20.5 to 84.4 m³/1000 kg of fish based on the VSS loading criteria. These volumes translate into a pond surface area of 8.2 to 33.8 m²/1000 kg of fish, assuming a 2.5 meter pond depth. On a BOD₅ loading basis, the lagoon surface area ranges from 17.8 to 133.3 m².
1000 kg fish (Poon et al. 1985). 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 (Oversch 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 (Oversch 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 aquatic waste valuable to crops as a fertilizer (Mudraik 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 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 to 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 22 kg of nitrogen per hectare (Oversch 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 m² 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**


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

<table>
<thead>
<tr>
<th>Species</th>
<th>BOD</th>
<th>Excretion Products</th>
<th>TSS</th>
<th>Reference</th>
</tr>
</thead>
<tbody>
<tr>
<td>Trout</td>
<td>0.4 - 0.60(1)</td>
<td>0.4 - 0.52</td>
<td></td>
<td>Speece 1973;</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td>Liao and Mayo 1974</td>
</tr>
<tr>
<td>Channel Catfish</td>
<td>0.083 - 0.113(2)</td>
<td>0.08 - 0.28</td>
<td></td>
<td>Page and Andrews 1973;</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td>0.22(2)</td>
<td>Gordon 1974</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td>0.43</td>
<td>Wimberly 1990</td>
</tr>
</tbody>
</table>

(1) BOD$_{20}$  
(2) BOD$_{5}$

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

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

(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 $K_1 = 0.36$, $h_5 = 0.35$. 
Table 3. The ratios of BOD$_5$ and total nitrogen to TSS in aquacultural sludge from different recirculating systems.

<table>
<thead>
<tr>
<th>System</th>
<th>BOD$_5$/TSS</th>
<th>TKN/TSS</th>
<th>Animals</th>
</tr>
</thead>
<tbody>
<tr>
<td>Pressure sand filter</td>
<td>0.20</td>
<td>0.053</td>
<td>Sturgeon <em>Acipenser transmontanus</em></td>
</tr>
<tr>
<td>Upflow sand filter</td>
<td>0.17</td>
<td>0.038</td>
<td>Snapping turtle <em>Chelydra serpentina</em></td>
</tr>
<tr>
<td>Propeller washed filter</td>
<td>0.09</td>
<td>0.049</td>
<td>Red swamp crawfish <em>Procambarus clarkii</em></td>
</tr>
<tr>
<td>RBC/clarifier</td>
<td>0.09</td>
<td>0.061</td>
<td>Sturgeon <em>Acipenser transmontanus</em></td>
</tr>
</tbody>
</table>

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

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

* Calculated for 1 - 3 minute backwashing time

Table 5. Features of Sludge Stabilization Options.

<table>
<thead>
<tr>
<th>Option</th>
<th>Advantages</th>
<th>Disadvantages</th>
</tr>
</thead>
<tbody>
<tr>
<td>1. Anaerobic Lagoon</td>
<td>High organic loading</td>
<td>Odor</td>
</tr>
<tr>
<td></td>
<td>Low maintenance</td>
<td></td>
</tr>
<tr>
<td>2. Aerated Lagoon</td>
<td>Space efficient</td>
<td>Energy consumption</td>
</tr>
<tr>
<td></td>
<td>High organic loading</td>
<td>Moderate maintenance</td>
</tr>
<tr>
<td>3. Aerobic Digesters</td>
<td>High loading</td>
<td>Energy consumption</td>
</tr>
<tr>
<td>4. Anaerobic Digesters</td>
<td>High loading</td>
<td>Complex</td>
</tr>
<tr>
<td></td>
<td>Methane generation</td>
<td>High maintenance</td>
</tr>
<tr>
<td>5. Composting</td>
<td>Useful end product</td>
<td>Dewatering required</td>
</tr>
</tbody>
</table>
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

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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 (Gaskoplis, 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 overall 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 flows consist 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 CO2, HCO2, etc.), and several other chemicals are critical to survival of the aquatic crop. The unit processes selected.
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 operations 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 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 controlling parameter 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 controlling 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.

Figure 1. Series configuration of an aquaculture system.

Figure 2. Parallel arrangement of aquaculture unit processes.

Figure 3. Parallel/series flow system.