Biological Filters: Trickling and RBC Design

John N. Hochheimer
National Committee for Quality Assurance
2000 L Street, NW Suite 500
Washington, DC 20036

Fred Wheaton, Professor
Biological Resources Engineering Dept.
University of Maryland
College Park, MD 20742

Fish produce a variety of wastes including solids, ammonia, carbon dioxide and other materials. These wastes must be removed from the culture water or they become toxic to the fish. Many methods have been developed to remove the wastes fish produce. However, in this paper we will concentrate on removal of ammonia and nitrite, both of which are highly toxic to fish and other aquatic organisms.

In fresh water systems there are two common methods used to remove ammonia: ion exchange and biological filters. In brackish or salt water systems ion exchange is not a viable alternative because salt in the culture water quickly (usually in a matter of minutes) saturates all of the adsorption sites on the ion exchange media. Thus, biological filters are the only widely used method of removing ammonia and nitrite from all types of aquacultural systems.

Biological filters consist of some solid media that serves as a surface on which bacteria can attach and live. Water containing ammonia and/or nitirite flow over this media (and the bacteria attached to it). The bacteria remove the ammonia from the water and use it as an energy source to drive their life processes. These bacterial excrete nitrite, require oxygen and produce carbon dioxide as byproducts of their respiration. A different group of bacteria remove the released nitrite and convert it to nitrate. These bacteria use the nitrite to nitrate conversion for an energy source, they use nitrite and oxygen, and produce nitrate and carbon dioxide. The ammonia to nitrite conversion produces hydrogen and uses up alkalinity. Although many bacteria species can participate in these conversions it is usually assumed that the ammonia to nitrite conversion is carried out primarily by *nitrosomona* sp. and the nitrite to nitrate conversion by *nitrobacter* sp. (Water Pollution Control Federation, 1983). Thus, the primary purpose of biological filters is to remove ammonia and nitrite from aquatic culture systems.

Management of water chemistry is one of the most important considerations in recirculating aquacultural systems. Proper system management results in the minimization of stress, which in turn leads to healthier fish and more profitability. The different components in a recirculating system are designed to control one or more water quality functions, such as ammonia, temperature, dissolved oxygen, or solids. Biological filters are designed to maintain the various forms of inorganic nitrogen (e.g., ammonia, nitrite, and nitrate) at levels that are healthy for the fish being cultured.

Basic Nitrogen Cycle

Nitrogen plays an important role in the structure and make-up of all living organisms. In the aquacultural environment, nitrogen exists in the inorganic forms of nitrate, nitrite, ammonia, and nitrogen gas and in many forms of organic nitrogen. The nitrogen cycle in recirculating system aquaculture can be described pictorially by Figure 1. Nitrogen originates from the atmosphere primarily in the form of nitrogen gas. Animals excrete nitrogen in the form of ammonia, amino acids, urea, and uric acid. Plants excrete nitrogen in the form of amino acids and proteins. Also, nitrogen is released through decomposition of dead animals and plants, uneaten feed, and bacterial cells and wastes.

The presence of nitrogen gas in recirculating system waters is usually of little importance, because very little of it is fixed into organic matter. The concentration of nitrogen gas in recirculating system waters depends on the partial pressure of atmospheric nitrogen compared to dissolved nitrogen, as well as temperature and salinity of the water. Only when nitrogen gas becomes super-saturated does it become problematic to recirculating system aquaculture.

Plants in recirculating system waters release amino acids and peptides. However, compared to the amounts of nitrogenous compounds released by animals, those released by plants are of little consequence (Spotte, 1979). Wastes in recirculating aquacultural systemsCurea, amino acids and uric acidCare rapidly broken down (in a process called mineralization) into ammonia by heterotrophic bacteria. This mineralization process is depicted in the nitrogen cycle (Figure 1) where the organic compounds are broken down into their inorganic components of which ammonia predominates.

The two processes in the nitrogen cycle that are of major importance in recirculating system aquaculture are nitrification and denitrification. Ammonia is oxidized to nitrite and then to nitrate through a series of biochemical reactions called nitrification. Denitrification is primarily a reduction of nitrate to nitrogen gas by anaerobic bacteria. Nitrification as it relates to biological filtration in recirculating aquacultural systems will be the focus of this paper.

Ammonia, nitrite, and nitrate are all highly soluble in water. Ionized ammonia, NH_4, exists at equilibrium with un-ionized ammonia, NH_3, in water. The relative concentration of ionized and un-ionized ammonia depends primarily on temperature and pH**C**the higher the temperature and pH, the higher the concentration of un-ionized ammonia. Unless otherwise noted in the text, ammonia will refer to total ammonia, which is the sum of ionized and un-ionized ammonia (often refereed to as total ammonia nitrogen or TAN). Nitrite exists at equilibrium with nitrous acid in water, with the relative concentration again depending on pH and temperature. Nitrite, when mentioned in the text, will refer to the sum of nitrite and nitrous acid. Nitrate is the conjugate base of nitric acid, a strong acid. Since strong acids usually dissociate completely in water, nitrate exists in its conjugate base form only.

Nitrogen Control

Of the many forms of nitrogen present in aquacultural system waters, ammonia, nitrite, and nitrate are considered to be of major importance. Nitrogen gas, if super-saturated, can cause morbidity and mortality. However, nitrogen gas can easily be stripped from system waters by agitating the water in some fashion (Speece, et al., 1988; Parker, et al., 1984). Organic forms of nitrogen are rapidly broken down by bacteria in aquatic systems to inorganic forms of nitrogen; the primary inorganic form is ammonia. Aqueous ammonia can be toxic to fish and other aquatic organisms at relatively low concentrations (see Boyd, 1979 for a good summary of the toxic effects of ammonia). Therefore, ammonia must somehow be controlled, converted to a non-toxic form, or removed from aquacultural system waters.

Nitrification Kinetics

Nitrification is the oxidation of ammonium to nitrate via nitrite. Nitrification is carried out by a few species of autotrophic bacteria; bacteria that derive their energy from these oxidations and not from oxidation of carbon compounds (Painter, 1970). The importance of nitrification is that it produces an oxidized form of nitrogen (i.e., nitrate) that may participate in denitrification reactions. When denitrification is complete, the result is the loss of readily available nitrogen from water.

Nitrification in recirculating aquacultural systems typically occurs by the action of two genera of autotrophic bacteria: *Nitrosomonas* and *Nitrobacter*. Autotrophic bacteria derive their energy from inorganic compounds, as opposed to heterotrophic bacteria that derive energy from organic compounds. Ammonia removal is a two step process, where ammonia is converted to nitrite by *Nitrosomonas* and nitrite is converted to nitrate by *Nitrobacter*. Equations 1 and 2 (Water Pollution Control Federation, 1983, USEPA, 1975, and Boyd, 1979) show the chemical reaction of this conversion:

$$NH_4^+ + 1.5\,O_2 \rightarrow 2\,H^+ + H_2O + NO_2^- \quad \textbf{(1)}$$

$$NO_2^- + 0.5\,O_2 \rightarrow NO_3^- \quad \textbf{(2)}$$

The reactions shown in Equations 1 and 2 release energy that is used by *Nitrosomonas* and *Nitrobacter* for cellular growth and maintenance. Oxygen serves as an electron acceptor and is the only electron acceptor that can be used by *Nitrosomonas* and *Nitrobacter* (Water Pollution Control Federation, 1983).

Growth of nitrifiers is very slow and cell yield per unit of energy is low. Most of the energy produced in the oxidation of ammonia and nitrite is used by nitrifying bacteria to produce new bacterial cells. One school of thought considers the slow growth to be inevitable while others believe that higher growth rates are possible given the right, but currently unknown conditions (Painter, 1970). Autotrophs are relatively inefficient, compared to heterotrophs, in energy

usage to form cellular material. Ecologically, low energy efficiency yields a small biomass production capable of oxidizing a large quantity of ammonia (Fenchel, 1979). Nitrifiers use a lot of energy to produce a small amount of cell mass. In biofilter performance, this is a desirable characteristic, since much ammonia and nitrite are removed with relatively few cells produced. Biofilters will be slower to clog and small volumes of sludge will be produced.

Stoichiometry

The stoichiometric (chemical balance) requirements of ammonia oxidation were described by Gujer and Boller (1986) as:

$$NH_4 + 1.90_2 + 2\ HCO_3 \rightarrow NO_3 + 1.9\ CO_2 + 2.9\ H_2O + 0.1\ CH_2O \qquad \textbf{(3)}$$

where CH_2O represents cell biomass. Equation 3 can be used to predict three stoichiometric requirements of nitrification: oxygen requirements, alkalinity consumption and biomass production. The oxidation of 1 g of ammonia requires 4.34 g oxygen and 7.14 g alkalinity and produces 0.21 g of bacterial cells, 1.98 g acid, and 4.43 g nitrate.

When oxygen, alkalinity, and micro-nutrients are excluded, the growth limiting substrate for *Nitrosomonas* is ammonia and for *Nitrobacter* is nitrite. *Nitrobacter* also grow faster than *Nitrosomonas*. Because nitrification proceeds from ammonia oxidation to nitrite oxidation, the overall kinetics of nitrification are usually controlled by ammonia oxidation (Water Pollution Control Federation, 1983).

Factors Limiting Nitrifier Growth

The ability of biological nitrification to adequately control ammonia and nitrite in recirculating aquacultural systems depends on a variety of factors that limit nitrifier growth. Studies on the kinetics of nitrification show how effective nitrifiers are under ideal experimental conditions, i.e., oxygen and alkalinity are sufficient and ammonia is the only limiting factor. However, under normal operating conditions, there are a variety of factors that, individually or in combination with each other, will reduce the efficiency of biofilter operation. One important consideration about biofilter performance is acclimation. Biofilters can usually be adapted to water quality and operating conditions that, according to theory, should normally cause the nitrifying system to fail. Most successful biofilter systems operate outside of ideal laboratory conditions much of the time. However, these operating biofilters are running under a balance of conditions within the biofilter and culture system. Operating at or just outside of the limits normally considered safe in laboratory experiments can be done, but be careful! You must monitor water quality and be able to correct problems when they occur, there is not much forgiveness and catastrophic failures can happen.

pH

The interactions of pH, nitrification, and water quality can be quite complex. In general, nitrification is most efficient at pH levels ranging from about 7.5 to 9.0. At the higher pH ranges (8.5 - 9.0), nitrification rates are fastest given sufficient ammonia. However, at the low ammonia concentrations usually found in aquacultural systems, operating at a pH of about 7.0 can be efficient. Because pH also effects the relative concentration of ionized and un-ionized ammonia in water and nitrifying bacteria use the ionized form, operating at a pH of about 7.0 usually increases the efficiency of the recirculating aquacultural system. Another positive effect of operating at the lower pH is that the toxicity of ammonia to fish increases with increasing pH, so operating in the lower range also reduces ammonia toxicity.

Alkalinity

Equations 1 and 3 illustrate the relationships of acid produced in the oxidation of ammonia and destruction of alkalinity by the consumption of bicarbonate ions in cell production and neutralization of generated acid. The stoichiometric relationship (Equation 3) indicates that 7.14 g of alkalinity (as $CaCO_3$) are used to oxidize 1 g of ammonia. Therefore, nitrification produces acid and uses alkalinity, so alkalinity in recirculating system aquaculture must be continually monitored and adjusted. Many basic solutions can be used to buffer, or add alkalinity, to recirculating systems, including sodium bicarbonate, calcium carbonate, sodium hydroxide, etc. Care is needed in the selection of the buffer, as too much of a strong buffer (e.g., sodium hydroxide) can lead to wide swings in pH, which is stressful to the fish and nitrifying bacteria. Buffers that contain calcium (e.g., lime or calcium carbonate) can lead to excessive calcium build-up in the recirculating system. When the calcium becomes supersaturated, subsequent additions of lime cause precipitation of calcium carbonate in the system, unavailable alkalinity, and a mess. Many aquaculturists prefer to use sodium bicarbonate (baking soda) as an alkalinity supplement because it produces small changes in system pH , is readily available, and inexpensive.

Temperature

Temperature directly affects growth and nitrification rates of nitrifying bacteria. Jones and Morita (1985) isolated an ammonia oxidizing bacteria capable of nitrification and growth at temperatures of -5° C. Optimal growth occurred at 22° C for cells grown at 5° C and lethal temperatures were about 29° C. Cells grown at 25° C had optimal growth temperatures of 30° C and lethal temperatures of 38° C (Jones and Morita, 1985). Basically, research on temperature and its effects on nitrification show that nitrification occurs and can be acclimated to conditions that are also favorable to aquatic species. Nitrification rates are slower at lower temperatures and increase linearly through the range of temperatures found in most aquacultural applications (Wortman, 1990).

Dissolved Oxygen

Dissolved oxygen is critical for nitrification to occur. As dissolved oxygen levels decrease to 1.0 mg/L in biological filters, dissolved oxygen rather than ammonia becomes the growth limiting factor. To prevent dissolved oxygen from becoming a limiting factor, water entering a biofilter should have minimum oxygen levels of 2.0 mg/L (Water Pollution Control Federation, 1983). Biofilter designs using trickling or rotating biological contactors benefit from natural oxygenation occurring as air flows past media covered with biofilms.

Light

Designs for biological filters should prevent too much light from contacting the bacterial surfaces. Olson (1981) found light intensities less than 1% of sunlight intensities were inhibitory to nitrifying bacteria. Light is believed to oxidize cytochrome C in both species of bacteria. *Nitrobacter* is more sensitive to light because it contains less cytochrome C than *Nitrosomonas* (Olson, 1981). Horrigan, et al. (1981) found similar results for light inhibitors and concluded complete darkness was superior to diurnal cycling of light regimes for nitrifying bacteria.

Salinity

Kawai, et al. (1965) found that nitrification in saline waters was maximal when done at constant salinities. Fresh water nitrifiers were completely inhibited in saline waters. Salt water nitrifiers were also found to be more sensitive to oxygen concentrations than fresh water nitrifiers. Bower and Turner (1981) noted that abrupt changes in salinity probably shocked nitrifiers, thus reducing their ability to remove ammonia and nitrite. Slowly acclimating working biological filters to salinity conditions results in successful transitions from one salinity level to another. A maximum change of 5 ppt should not adversely affect biofilter operation. However, gradual changes in salinity over several weeks is preferable.

Other Water Chemistry Concerns

Many chemicals have been found to be inhibitory or toxic to nitrifying bacteria. A general rule is if a substance is toxic to fish, then it is probably toxic to the bacteria. Chemicals used to treat fish for a variety of diseases and parasites can be toxic to nitrifying bacteria at therapeutic levels for fish. Antibiotics are generally toxic. Treatments used to remove external parasites, such as formalin, potassium permanganate, or peroxide, oxidize bacteria, as well. System design should include a means to take biofilters off-line during short treatments and allow for water to be flushed from the system prior to reestablishing flow.

Particulates in system waters can have several effects on biofilters. Particles that are larger than the pore sizes in the filter media can clog the filter, and lead to reduced filtering capacity and efficiency. Some nitrifiers will grow on particles that reside in the system for extended periods of time and may actually perform the majority of nitrification occurring in the system (at the expense of the nitrifier populations on the biofilter media). If the system is flushed or

filtered for particulate, the nitrifiers are removed from the system and nitrification may essentially cease for a period of time. Most of the particulate are made up of organic compounds that will break down rapidly in the system from heterotrophic bacterial activity. This break down consumes oxygen needed by the nitrifiers and fish in the system.

Filter Configurations Used

There are litterly hundreds of biofilter configurations. However, they can be classified into one of several groups, the groupings based primarily on how they operate. Submerged filters are designed to keep the solid media in the filter continuously submerged in the water. Upflow submerged filters have the water flow from bottom to top, while downflow submerged filters have the water flow from the top toward the bottom. The oxygen supply for the bacteria in a submerged filter must be supplied from the water, a factor that often sets the flow rate through the filter at a higher value than would be necessary if the flow rate were dictated by ammonia removal only. Submerged biological filters tend to plug fairly easily unless the media has a high void percentage and is at least 2 cm in diameter.

Trickling filters look much like a submerged filter (i.e. they consist of a tube or tank filed with media through which water is passed). However, they are operated differently in that the free water surface in the filter is maintained below the media. The culture water is pumped to the top of the filter and uniformly distributed over the top of the filter. As the water trickles down over the media it absorbs oxygen from the air in the filter and supplies ammonia and/or nitrite to the bacteria growing on the media. If properly designed, trickling filters rarely plug, they are quite stable over time, but they require some pumping head (at least the height of the filter). Their primary advantage is the oxygen for the bacteria in the filter comes from the air in the filter. Thus, they are well aerated, and the water flow rate through the filter is independent of oxygen supply.

Rotating biological contactors (RBC=s) typically are designed in one of two configurations. The first consists of a horizontal shaft that has flat or corrugated circular plates attached to it. The plates are typically spaced at least one cm apart along the shaft. The shaft is attached to bearings and mounted above a tank such that about 40 to 45 percent of each plate surface is below the top of the tank. Waste water is pumped into the tank and the RBC is rotated by a motor such that the plates rotate in and out of the water during each revolution. Bacteria grow on the plate surfaces and as the bacteria enter the water they remove ammonia or nitrite and as they rotate through the air they extract oxygen from the air. The second RBC configuration replaces the disks with a drum filled with some light weight media (e.g. plastic) that has a high surface to volume ratio and a high void ratio. As the drum rotates the bacteria on the media are alternately supplied with ammonia or nitrite from the wastewater and oxygen from the air. RBC=s are generally quite stable in operation, have a high ammonia removal efficiency compared to some other biofilters, and they require very little head loss (typically 2 to 3 cm of water). Their primary disadvantage is that they require a power source to turn them and mechanical breakdown can be a problem, particularly with a poorly designed unit (Hochheimer, 1990; Wheaton, et al., 1991).

Fluidized bed biological filters consist of a bed of sand or other heavier than water media that is small in diameter. Water is pumped up through the sand at a fast enough velocity to Afluidize@ the sand (i.e. suspend the sand grains in the vertical column of flowing water). Bacteria grow on the sand grains and as the water passes by the fluidized sand the bacteria extract ammonia and/or nitrite. Fluidized sand filters require a small foot print for the size of the filter because the small sand grains provide a very high specific surface area per unit of volume of filter. These filters require continuous pumping and have an essentially constant pressure drop across the filter the pump must overcome (Summerfelt and Cleasby, 1996).

There are several types of bead filters including those using heavier than water and those using lighter than water beads (Timmons, 1997; Delos Reyes and Malone, 1998). Most systems use a small tank specifically designed to provide the flow and operation desired for the bead filter. The tank typically has an upflow configuration and a screen across the top of the bead bed to prevent the beads from exiting the filter. Wastewater is pumped upward through the bead bed. Bacteria on the bead surfaces provide nitrification of the ammonia and nitrite and the beads provide a screening effect that traps considerable solids. Thus, bead filters can be used as solids removal devices or as biofilters (Beecher et al., 1997).

Start-up

Start-up must be considered when designing and operating recirculating aquacultural systems. Establishing and maintaining a robust population of nitrifying bacteria that is capable of removing the intended ammonia load is critical to success. Operators of recirculating aquacultural systems must acclimate the nitrifying bacteria population to unique conditions and develop a population that will be sufficient to remove levels of ammonia produced when fish are introduced into the system.

Bower and Turner (1981 and 1984) concluded from their studies that seeding filters with filter media from established filters could significantly reduce new system start-up times. Addition of 10% wet filter media from established seawater systems reduced start-up time 81% (4 days compared to 21 days) for ammonia removal and 89% (4 days compared to 37 days) for nitrite removal compared to controls. The use of dry filter media from established filters, seawater from established filters or wet filter media from freshwater filter systems produced considerably less reductions in filter start-up time than did the addition of wet filter media from seawater systems. Additions of commercial additives provided variable results, none of which were as rapid as the wet filter media additions.

Seeding of freshwater systems was examined by Carmignani and Bennett (1977). The authors found that addition of approximately 3% wet filter media from an established filter decreased start-up time by 48% compared to control filters. Ammonia and nitrite at levels above 15-20 mg/L can become toxic to nitrifying bacteria. Ammonia and nitrite levels must be monitored at least daily during start-up to prevent toxic levels from building up.

System start-up presents problems for many recirculating aquacultural system operators. Nitrifying bacteria populations grow slowly and do not quickly adapt to change.

Heterotrophic bacteria (bacteria that populate systems and remove organic carbon substances from the water) can out compete nitrifying bacteria. Starting the biological filter with media from existing and similarly operating filters coupled with inorganic ammonia additions, in lieu of fish, can be an efficient way to start biofilters in less than a month=s time.

Design of a Trickling Filter for Ammonia Removal

The basic concept of trickling filters is to provide a surface on which microbial films grow. Trickling filters come in many configurations and contain various media types. Traditional wastewater treatment trickling filters use rock for a media and are typically short in height, large in diameter, and cylindrically shaped. Trickling filters for aquaculture are predominately cylindrical and, with the advent of light weight, plastic media, the filters can be made tall in relation to their diameter. Media types are either dumped or fixed. Dumped media allows for randomly packed filters and media shapes are usually some configuration of hollow cylinders, spheres, or other regular shapes. Fixed media resemble corrugated fiberglass roofing materials and are arranged in vertical, horizontal, or angular orientations to water flow.

Water containing a dissolved substrate flows over an exposed microbial film in a trickling filter and is biologically oxidized to form a more stable material. The biofilm cannot utilize a substrate unless it is transported to the microorganisms. Substrate flux (in this case ammonia removal) within the biofilm results in a lower substrate concentration surrounding the microorganisms than the concentration of ammonia in the bulk liquid. Closed aquaculture systems are ammonia limiting (as opposed to those in wastewater treatment, which are oxygen limiting) and rely on physical processes for mass transport of ammonia to the biofilm and not on diffusion. Therefore, it is essential that ammonia is constantly made available to the microorganisms for maximum ammonia removal.

Factors Affecting Trickling Filter Performance

Hydraulic Loading

Hydraulic loading rates are very important design considerations for both trickling and rotating biological contactor filters. The total influent flow rate per unit of biofilter cross sectional area is defined as the hydraulic loading rate and is expressed as flow per unit area ($m^3/m^2 \cdot d$). The lower limit of hydraulic loading is the minimum wetting rate (MWR), which is the lowest flow rate that wets all of the media in the filter. The MWR is important since media not wetted will not support bacterial growth. Grady and Lim (1980) reported that one manufacturer of random packed media recommended a minimum hydraulic loading of 29 $m^3/m^2 \cdot d$ (Norton Actifil). Roberts (1985) reported minimum hydraulic loadings of 32 to 55 $m^3/m^2 \cdot d$ for random packed media (plastic pall rings). For design purposes, a MWR of 50 $m^3/m^2 \cdot d$ is considered safe.

The upper irrigation rate (UIR) is the maximum flow rate in a filter before scouring of the

biofilm occurs. In high void fraction media, like those used in many biofilters, exceeding the (UIR) usually causes scouring of the biofilm and loss of active nitrifying surfaces. Roberts (1985) reported UIR values of 72 to 188 $m^3/m^2 \cdot d$ for randomly packed plastic media (plastic pall rings). Grady and Lim (1980) reported a UIR range of 234 to 350 $m^3/m^2 \cdot d$ (Dow Surfpac). A design UIR of 300 $m^3/m^2 \cdot d$ should be acceptable.

The relationship between filter performance and hydraulic loading should not be mistakenly considered as synonymous with substrate loading. At a constant substrate influent concentration, increases in the hydraulic loading rate decreases the percent substrate removed. For the same conditions, the mass substrate removal rate increases (Grady and Lim, 1980). This is logical since, as the flow increases, the residence time decreases in the filter and for a constant concentration, the mass of substrate input to the filter increases. Research by Hochheimer (1990) shows that mass loading of ammonia to biofilters is a limiting factor (i.e., because ammonia is a limiting nutrient and diffusion of ammonia to bacteria in the biofilter limits removal). When operating and designing biofilters, this means that flow rates to biofilters should be as high as possible (but under UIR limits of 300 $m^3/m^2 \cdot d$) while minimizing pumping costs.

Mass Transport

Movement of substrate (ammonia, nitrite, oxygen, etc.) to and wastes (nitrite, nitrate, etc.) from bacterial cells is often a limiting factor in trickling filter performance. Hochheimer (1990) developed equations to describe the diffusive and mass transport relationships in a trickling filter. Media in a working biofilter becomes coated with a biofilm, resulting from the growing bacterial population. The two components where mass transport become important are within the biofilter (transfer of substrate(s) to the surface of the biofilm) and then within the biofilm. Getting substrate to the surface of the biofilm is associated with concentrations of the individual substrates in the culture system water and movement of the water through the biofilter. Work from Hochheimer (1990) indicates that ammonia concentrations are too low to be influenced by diffusion in the water flowing through the biofilter. Physical mass transport of the ammonia then becomes the dominant factor in determining availability of ammonia to the nitrifying bacteria. The other substrates (oxygen and alkalinity), if kept at recommended levels, are dominated by diffusion.

Bacterial cells growing within the biofilm require all nutrients to diffuse into the biofilm to become available to the growing cells. Similarly, waste products from the bacteria must diffuse out of the biofilm. Within the biofilm, diffusion is the primary transport method of substrate to bacterial cells. Again, because ammonia concentrations are so low, diffusion becomes the most limiting factor in ammonia removal. Nitrite behaves similarly to ammonia in both the water and biofilm components.

The significance of these transport processes is that flow rate of water through the biofilter becomes important in determining the effectiveness of a trickling filter (or any nitrification processes) in recirculating aquacultural systems. Designs of trickling filters should strive for

flow rates that are near the UIR so that maximum mass transport of substrate to the biofilm is achieved. Designs should also allow for air flows through the trickling filter to maximize oxygen availability.

Depth

Trickling filter depth is primarily determined by available space in which the filter is being placed and the weight of media. Filter containers for biofilters with heavy media must be adequately constructed to hold the combined weight of media, water, and biofilms. The filter depth must be adequate to allow for both steps**C**ammonia and nitrite removal**C**to take place. However, no good design information is available for determining the most efficient depth. Presently, most designs consider available area, weight of media, and costs for filter containers to determine filter depth. In trickling filters depth is usually directly proportional to pumping costs.

Cross Sectional Area

Cross sectional area of a biofilter is defined as the top area of the filter container. For systems with multiple filters, the total cross sectional area is determined by summing the individual areas. The cross sectional area is important in the calculation of the hydraulic loading rate.

Void Ratio

Void ratio is the proportion of free space volume in a filter to the total filter volume. In a trickling filter there are voids that are not filled in by the media. High void ratios reduce clogging and allow for air to move more freely in the filter. Remember, trickling filters work by allowing a thin film of water to flow across media surfaces. Filters with low void ratios tend to interrupt this thin flow of water and trap many solid particles. Clogged filters must be cleaned, which often leads to reduced filter efficiency.

Specific Surface Area

Specific surface area is defined as the surface area of a particular media per unit volume. Since bacteria attach to the surfaces of the media, it is the surface area that determines how much nitrification can occur. It is desirable to have a large specific surface area to minimize the volume of filters required in a particular system. Floor space and ceiling heights usually determine available space for biofiltration, so adjusting specific surface area is one way to obtain a desired filtration capacity in a given volume. However, most media costs are proportionate to specific surface area. The challenge in design is to maximize specific surface area while concurrently maintaining relatively high void fractions, low costs, and adequate filter flow rates. Usually filter design involves an iterative process of selecting media, calculating volumes and cost, and then evaluating outcomes.

Media Type and Size

There are a wide variety of media types and sizes available for trickling filters. These include rocks, sand, plastic media (designed for biofilters), packing materials, and corrugated plastic shapes. Any material that is non-toxic to the bacteria and fish and is stable in a water environment should be acceptable.

Trickling Filter Design Example

The following section will show an example for the basic design of a trickling filter for a recirculating aquacultural system. Water quality requirements and design estimates are conservative. The design example will be for a single culture system to grow hybrid striped bass with a maximum carrying capacity of 9072 kg (20,000 pounds).

Basic Design Assumptions

Several assumptions are made about the recirculating aquacultural system. Water exchanges are necessary to replenish water that evaporates and water lost from the system due to solids removal. Critical variables in the design that impact filter sizing are total mass of fish at maximum loading, system temperature, daily feeding rate, and density of fish in the system.

1. There is 100% reuse with weekly exchanges of 20% of the system water volume and daily additions of water to maintain system volume.

2. Hybrid striped bass in the system average 0.7 kg (1.5 pounds) each at the end of the growing season.

3. The system temperature is to be maintained at 24°C (75°F).

4. The fish are fed at 2% of body weight on a daily basis.

5. The maximum fish density in the system is 120 kg/m**;** (1 pound per gallon).

6. The system exchanges water at least 2-3 times per hour with the biological filter.

Media Data

The type of media and its specific surface area will directly affect the filter volume. Selection of the media should consider specific surface area, weight, void ratio, cost per unit of surface area (not volume), availability, type of material, and durability. Design of the filter is an iterative balance between cost and volume. Remember, media that is too small will tend to trap particles and clog from bacterial film growth. Also, the material must be strong enough to support the weight of media above it, not be toxic to bacteria or fish, and must not break down under normal operating and cleaning conditions.

Parameter	Value

Type	plastic rings
Diameter	2.5 cm (1 inch)
Void Fraction	0.92
Specific Surface Area	220 m**5**/m**;** (67 ft^2/ft^3)

Water Quality Requirements

The following water quality requirements apply to hybrid striped bass (Hochheimer and Wheaton, 1997):

Parameter	Limit
Dissolved Oxygen	> 5.0 mg/L
pH	6.5 - 9.0
Alkalinity (total as $CaCO_3$)	50 - 400 mg/L
Ammonia (as NH_3-N)	0.0125 mg/L
Nitrite (as NO_2-N)	0.1 mg/L
Carbon Dioxide	0- 15 mg/L
Nitrogen (gas)	< 110% total gas saturation
Total Suspended Solids	< 80 mg/L

Design Calculations

Water Volume

At a density of 120 kg/m**;**, the total volume of water required is:

$$Volume_{Water} = 9072 \text{ kg of fish} \div 120 \text{ kg/m}^3$$
$$Volume_{Water} = 75.7 \text{ m}^3$$

Feed Consumption at Maximum Production (per day)

This design uses ammonia loading calculations that are based on the amount of feed fed on a daily basis. It is assumed that ammonia excreted by the fish is proportional to the amount of feed put in the system. The amount of feed fed per day varies with the size of the fish. At early life stages, the fish require feedings of about 6% or more of their body weight. For fish close to harvest weight (and when the system is likely to have the greatest loading of ammonia) feeding rates range from 1.5% to 3.0% of body weight per day. The greatest total mass of feed that will be put in the system is assumed when the fish are at harvest weight and the maximum mass of fish will be in the system. A feeding rate of 2% of body weight is assumed for this example. Thus, for the production of 9,072 kg of fish:

$Mass_{Feed}$ = 9,072 kg of fish x 0.02 kg /kg of fish

$Mass_{Feed}$ = 181 kg

Waste Production and Oxygen Consumption

Colt (1986) proposed a mass balance approach to oxygen requirements and waste production. This approach determines the oxygen requirements of a fish culture system (including biological filtration) based on the mass of feed fed on a daily basis. The following show the relationships developed by Colt (1986):

1 kg of feed requires	0.21 kg of oxygen
1 kg of feed produces	0.28 kg of carbon dioxide 0.30 kg of solids 0.03 kg of total ammonia

Oxygen Requirements

With 187 kg of feed fed per day, and assuming that 0.25 kg of oxygen (0.21 kg of oxygen per kg of feed (Colt, 1986) plus an additional 20% as a safety factor) are required by the fish in the system for respiration and by bacterial respiration for nitrification and carbonaceous biochemical oxygen demand. The oxygen requirements will be:

Oxygen = 0.25 kg O_2 / kg feed x 181 kg feed/day

Oxygen = 45.3 kg O_2 / day

Typical oxygen transfer efficiencies range from about 5% to greater than 90%. The transfer efficiency will have to be considered when determining the total oxygen requirements of the system. For the purposes of this example, a system will be needed to supply at least 45.3 kg of oxygen per day.

Ammonia Production

Ammonia is found in two forms within aquatic systems, ammonia (often reported as NH_3, NH_3-N, or un-ionized ammonia) and ammonium (often reported as NH_4 , NH_4 -N, or ionized ammonia). Ammonia and ammonium exist in chemical equilibrium in the aquatic system. Un-ionized ammonia is highly toxic to aquatic animals and must be removed from the system. The pH and temperature of the system determines the relative concentrations of ionized and un-ionized ammonia in an aquatic system. At low pH values and temperatures, ionized ammonia predominates and at high pH values and temperatures un-ionized ammonia predominates. Total ammonia nitrogen (TAN) is the measurement of the combined concentrations of un-ionized and ionized ammonia, which accounts for pH and temperature.

The TAN produced in the example system is:

TAN = 181 kg feed/day x 0.03 kg TAN/kg feed
TAN = 5.4 kg TAN/day

Thus, a filter system would be required to remove 5.4 kg TAN per day to keep the fish healthy.

Ammonia production is not constant throughout the day. Typically ammonia production from fish in a closed system is cyclic with peaks occurring several hours after feeding. Experience has shown that an average hourly ammonia loading, based on a daily ammonia loading rates, is adequate to determine when for ammonia concentrations may become lethal in a closed system.

The maximum concentration of total ammonia ([TAN] in mg/L) in this example is the estimated hourly load divided by the volume of water in the system and the estimated filter exchange rate (2 times per hour):

[TAN] = (5.4 kg TAN/day x 1000 g/kg) ÷ (24 hours/day x 75.7 m^3 x 2 exchanges/hr)
[TAN] = 1.5 mg/L

Filter Design

Design of the physical requirements for a trickling biofilter can often require an iterative process and is based on water quality requirements and production levels.

Ammonia Removal Rate

At the design temperature of 24°C and a TAN concentration of about 1.5 mg/L, the ammonia removal rate is estimated to be 1.0 g TAN/$m^2 \cdot d$ (Wortman, 1990; Gujer and Boller, 1986).

Filter Surface Area

The surface area of the trickling filter required to remove the ammonia produced in the closed system is:

$Area_{Filter}$ = 5.4 kg TAN/d x 1000 g/kg ÷ 1 g TAN/$m^2 \cdot d$

$Area_{Filter}$ = 5,400 m^2

Filter Volume

The volume of media needed is a function of the surface area required and the specific surface area of the media from media data:

$Volume_{Media}$ = 5,400 m^2 ÷ 220 m^3/m^2

$$Volume_{Media} = 24.6\ m^3$$

Filter Dimensions

The determination of the dimensions and number of filters to use in a system is based on space requirements and limitations within a filter unit. Since recirculating aquacultural systems are typically ammonia limited, mass transport of ammonia is a crucial factor in filter performance. In general, the greater the flow rate of water through a filter, the greater the ammonia removal rates will be because more turbulence is created. Research by Hochheimer (1990) revealed that there is an upper and lower limit for hydraulic loading in a biofilter. Hydraulic loading is a function of flow rate and the cross-sectional area of the filter.

The minimum hydraulic loading for a filter ensures that all media in the filter is continually wetted, thus preventing bacteria from drying out. The maximum hydraulic loading rate prevents scouring of bacteria from the media in a filter. For randomly packed media, a minimum hydraulic loading of 30 $m^3/m^2 \cdot d$ and a maximum hydraulic loading of 225 $m^3/m^2 \cdot d$ can be used for design purposes (Hochheimer, 1990). Design of the filters requires a balance of the number of filter units, diameter and height of each individual filter, and the total flow rate of water through the filter system. The determination of filter dimensions can be iterative and is as follows.

Total flow through the filter per day is the volume of the culture system multiplied by the number of filter exchanges per day. It is desired to have at least 2 exchanges per hour, so the total flow rate is:

$$\text{Total Flow} = 75.7\ m^3 \times 24\ \text{hours} \times 2\ \text{exchanges per hour}$$

$$\text{Total Flow} = 3{,}634\ m^3/d$$

Using a configuration to filter total system flow, assume 6 filters and then the flow rate per filter is:

$$Flow\ Rate_{Filter} = 3{,}634\ m^3/d \div 6\ \text{filters}$$

$$Flow\ Rate_{Filter} = 605.6\ m^3/d$$

The volume needed for each filter unit is:

$$Volume_{Unit} = 24.6\ m^3 \div 6\ \text{filters}$$

$$Volume_{Unit} = 4.1\ m^3$$

The dimensions of each filter unit can be calculated from the maximum hydraulic loading rate. To determine the cross-sectional area:

$$Area_{Cross\text{-}Sectional} \geq 605.6\ m^3/d \div 225\ m^3/m^2 \cdot d$$

$$Area_{Cross\text{-}Sectional} \geq 2.7\ m^2$$

Assuming a cylindrical shape, the diameter of the cylinder for each filter unit would need to be:

$$Diameter^2 \geq (4\ x\ Area_{Cross\text{-}Sectional}) \div \pi$$

$$Diameter^2 \geq (4 \times 2.7\ m^2) \div \pi$$

$$Diameter \geq 1.85\ m$$

Thus, if a diameter of 2.0 m is assumed, then the height of the filter unit is:

$$Height = Volume_{Unit} \div Area_{Cross\text{-}Sectional}$$

$$Height = 4.1\ m^3 \div (\pi \times (2.0\ m)^2 \div 4)$$

$$Height = 1.31\ m$$

Then, the filter dimensions would be:

Height	= 1.3 m
Diameter	= 2.0 m
Volume	= 4.1 m^3
Cross-Sectional Area	= 3.1 m^2
Number of Filter Units	= 6

If the filter dimensions are not suitable for the physical conditions of the closed system, the above calculations can be reiterated with new values to fit the particular situation.

The biofilter units are usually filled with dumped plastic media. This type of media is specified because it provides a large specific surface area at a relatively economical cost. Other media could be used but care must be taken to provide enough surface area for the complete removal of ammonia in the system. Media of smaller specific surface area will require more voluminous filter units, thus more floor space.

The design presented here is conservative, but allows for flexibility resulting from many of the unknowns in biofilter design. There are many designs and configurations for biofilters that can be used for aquaculture. All of the different biofilter designs have positive, as well as negative traits. However, proper operation of closed systems is essential for success and a good operational plan can accommodate for the negative aspects of a particular design.

Rotating Biological Contactors

Design of RBC=s is very similar to other biofilters. The object is to get the waste water to move past the RBC so the nitrifying bacteria can remove the ammonia and the nitrite from the water. The factors noted above and by several other authors (Hochheimer, 1990; Hochheimer and Wheaton, 1991; Wheaton et al., 1991) effect the operation of an RBC and the bacteria on the RBC the same as they do any other nitrifying bacteria. Thus, this discussion will not be repeated here. The focus of this section will be on those design factors unique to RBC=s.

Because an RBC rotates through both an aquaous and an air phase, the oxygen is supplied by the air and the ammonia and nitrite by the water. The RBC is typically operated with a 35 to 45 percent submergence. Thus slightly less than one-half of the time the bacteria will be in the water and slightly more than one-half of the time the bacteria will be in the air. There are several constraints on the rotational speed of the RBC. The bacteria grow on the plate or media surfaces of the RBC. If the rotational speed gets too high the shear forces generated by the plates moving through the water will exceed the adhesion of the bacteria for the plate surface and the bacteria will be stripped off of the plates. Thus maximum rotation speed (i.e. revolutions per minute (RPM)) is generally limited by the lineal velocity of the fastest moving part of the plate as it moves through the water. This maximum velocity is ill defined because it is dependent on the plate surface characteristics (which is a function of the construction materials and the geometric design of the plate or media surfaces), the health and age of the bacteria, and other factors. It should also be noted that the maximum lineal velocity is a fixed value for a given application, but one of the design variables is the diameter of the RBC plates or drum. The larger the diameter, the greater the lineal velocity of the outer rim of the plates for a given number of revolutions per minute (RPM) of the RBC. For example, a two meter diameter RBC operating a 4 RPM has a much higher lineal speed at the rim of the plates than a 1 meter diameter RBC operating at the same RPM.

Another limit on the speed of rotation of the RBC is related to the oxygen concentration in the wastewater and the drying rate of the air. Any one bacteria must not be left in the water phase so long that it runs out of oxygen before remerging into the air. Similarity the period of time the bacteria is in the air must not be sufficient to dry the bacteria out so it can no longer function. These two factors place a lower limit on the speed of the RBC.

Between the two limitations discussed above is a wide range of rotational speeds that can be used in design of RBC=s. Selection of the optimum RPM of the rotor appears to be more of an art than a science. However, Antonie (1976) showed that ammonia removal by RBC=s was enhanced at peripheral speeds up to 0.305 m/sec (1 ft/sec), but above this value the ammonia removal was constant with increased peripheral speeds. Wortman (1990) in his biodrum studies used 10.37 cm/sec (0.34 ft/sec) peripheral speed. Paolini (1985) showed when RBC=s were used for COD (Chemical Oxygen Demand) removal, the removal of soluable BOD, influent BOD and RBC rotational speed interacted. For a given influent BOD with all other variables remaining constant increased rotational speed increased COD removal. Paolini (1985) also concluded that under limiting oxygen transfer conditions, the maximum COD removal rate is an approximately linear function of the square root of the disk rotational

speed, regardless of the wastewater type and the RBC system used. Friedman et al. (1979) showed that RBC removal leveled off, the value where the removal leveled off was a function of the influent COD. Weng and Molof (1974) found that nitrification by a RBC increased when speed was increased from 0.1 to 0.34 m/sec (0.3 to 1.1 feet/sec). Easter (1992) indicates that the Libey system used at Virginia Polytechnical and State University operated their RBC=s at 3 RPM (peripheral velocity of approximately 0.94 ft/sec). Gilbert et al. (1986) found in commercial installations most RBC=s were driven at a peripheral speed of about 0.3 m/sec (1 ft/sec), but this varied somewhat over the 105 units they surveyed.

Power consumption has been shown to increase as the RBC=s rotational speed increases (Fujie et al., 1983; Gilbert et al., 1986). Gilbert et al. (1986) surveyed 29 sewage treatment plants that had RBC=s. They found that energy usage was a function of rotational speed, wastewater temperature, the amount and configuration of media surface area, degree of submergence of the media, amount of biofilm growth on the RBC plates, and the efficiency of the motor and drive systems. Fujie et al. (1983) developed equations to predict the power consumption by RBC=s operating in sewage systems, primarily for BOD or COD removal. They found that power consumption per unit area of RBC surface at low speeds was proportional to the RPM squared and at high speeds was proportional to RPM cubed. This was attributed to the fact that at low speed the flow in the RBC tank was laminar and at higher speeds the flow became turbulent.

In contrast to the effect of rotation speed on power consumption, higher rotational speeds result in greater oxygenation in the RBC tank and usually better removal rates. Biodrums will provide considerably more oxygenation than will plate type RBC=s at the same RPM, but they will require greater power consumption. Fujie et al. (1983) found that the power consumption dropped as the RBC diameter decreased, but the COD (and probably ammonia) removal per square meter of floor space also decreased.

Thus, the concensus of those using RBC=s appears to be to maintain a peripheral velocity for the RBC of approximately 0.30m/sec (1 ft/sec) and to adjust the RPM to as low a value as possible while maintaining the peripheral speed. Disk diameter is limited by physical strength of the shaft and bearings and the space and power needed to house and operate the RBC. Most RBC=s used in aquacultural applications are in the 3 m (9 ft) or smaller in diameter. The length is usually determined by the lengths supplied by manufacture and the surface area needed for ammonia and nitrite removal.

Submergence Depth

RBC=s operate such that some proportation of the discs or drum is submerged in the water. Grady and Lim (1980) presented data to show the optimal submergence for an RBC is 35 to 50 percent. Practically constructing the RBC so the rotor bearings are above the water level is much easier. Thus, the submergence of the rotor is almost always somewhat less than 50 percent, in the optimal range as found by Grady and Lim (1980). The exact percent submergence depends on the bearing and shaft design as much as anything else. Most designers attempt to maximumize submergence of the rotor while keeping the bearings out

of the water.

Ammonia and Nitrite Removal Rates

Westerman et al. (1993) used an upflow sand filter in combination with an RBC on a full scale tilapia culture systems. They found that TAN (Total Ammonia Nitrogen) removal rates by the RBC ranged from 5.5 to 18.5 g/hr and nitrite removal rates varied from 9.4 to 22.6 g/hr. The RBC they used had a surface area of 470 square meters and a hydraulic loading rate of 0.28 L/m^2-min.

Easter (1992) developed an equation to predict the TAN removal for a RBC operating on hybrid striped bass recirculating system. For his three stage RBC the perdiction equation was:

$$S/So=e^{(-K(\text{Stage Number})/W^n)}$$

Where,

S = Effluent ammonia Concentration (mg TAN/L)
S_o = Influent ammonia concentration (mg TAN/L)
W = Mass loading of substrate (mass/area of biofilter) (g TAN/m^2 biofilter/day)
K = Empirical constant
n = Empirical constant

Values for the empirical constants are given in Table 1 below as provided by Easter (1992).

Table 1. Values for the constants in Easter's (1992) ammonia removal equation.

Filter Configuration Analyzed	n	K
RBC Stage 1 Only	0.55	0.14
RBC Stage 2 Only	0.14	0.11
RBC Stage 3 Only	0.18	0.06
All 3 Stages Together	0.36	0.08

Easter (1992) used detention times of 2.3 minutes per stage. The tank containing all three stages had a volume of 1,930 L and the flow rate was approximately 285 L/min. Each stage of the three stage RBC had 536 m^2 of surface area thus providing a hydraulic loading rate of 0.18 L/min/sq meter of filter surface area Using these data Easter (1992) found that the three stage RBC removed approximately 30 percent of the TAN on one pass as long as the influent TAN was above 0.2 mg/L. He also found that the mass removal of TAN by the filter was linearly related to the influent TAN loading in g of TAN/ m^2 of media/day and that the effluent TAN concentration rose quite rapidly from zero to about 0.2 mg/L and then followed an a nonlinear relationship with further increases in influent TAN.

Westerman et al. (1993) used a RBC with 470 m^2 of surface area, a flow rate of 130 L/min and a hydraulic loading per tank cross-sectional area of 160 L/min-m^2. The hydraulic loading calculated by dividing the flow rate by the filter or specific surface area was 0.28 L/min-m^2. He later indicated that the RBC could probably have been loaded more heavily. Fujie et al. (1983) reported the hydraulic loading of several treatment plants using RBC's for BOD (biochemical oxygen demand) removal. Their results showed hydraulic loading rates ranging from 0.024 to 0.06 L/min-m^2. Thus, loading rates vary widely depending on the design, the material being removed, and the effluent concentration desired. However, loading guidelines for ammonia and nitrate removal from aquacultural systems by RBC's is very limited.

Miller and Libey (1985) found mass removal rates of an RBC to vary some with the loading rate. At a fish stocking density of 227 kg/m^3 of catfish the RBC removed 0.78 g N/m^2 /day; at a stocking density of 118 kg/m^3 the RBC removal rate was 0.63 g N/m^2 /day; and at a stocking density of 57 kg/m^3 the RBC removed 0.19 g N/m^2 /day. The culture tank TAN concentrations were: 1.46, 1.26, and 0.36 mg/L for the high, medium and low feeding rates, respectively. This data was collected when the system water temperature was approximately 27.5 ^{0} C. As one might expect the RBC removed greater amounts of ammonia when the stocking density and the concentration of ammonia were higher.

Number of Stages

The number of stages used in a RBC can, theorically, be infinite. However, practical limitations usually limit the number of stages to three to five. Each stage is really a separate filter but works on the same wastewater stream in series. Thus, the raw influent to the first stage is what exits the tank immediately upstream of the RBC. The second stage sees the effluent of the first stage as its influent and so on for all other stages. Because heterotrophic bacteria usually grow faster than nitrifiers, the first stage of an RBC tends to be primarily a BOD or COD removal device unless the wastewater organic content is very low. As the wastewater moves to the second and subsequent stages the RBC tends to first begin removing ammonia and then nitrite with the final product being nitrate, assuming the RBC is sized and operated correctly.

In the ammonia and nitrite removal process it is interesting to compare the results observed by several authors. Easter (1992) using hybrid striped bass systems found that the TAN concentration in the RBC effluent was linearly related to the TAN in the influent and that the mass removal of TAN by a RBC was linearly related to the TAN loading on the RBC. He also stated that equation 1 above could be used to predict the ratio of the effluent to the influent TAN concentration. Westerman et al. (1993) found that their RBC removed about 250 mg TAN/ m^2 -day and provided 67 $\pm$ 18 percent removal of TAN and 59 $\pm$11 percent removal of nitrite-nitrogen when operated at 27-28 ^{o}C . Wheaton et al. (1994) in their development of RBC designs used TAN removal rates in mg TAN/m^2 /day of: 379, 193, and 122 at 30, 25, and 15 ^{o} C, repetitively. In the way of contrast Jansen et al. (1995) cited nitrification rates of between 2 and 4.48 g N/m^2 (2000 to 4480 mg N/m^2) for RBC:s operating on municipal wastewater. The large differences in these values result from the ammonia concentrations used in the two applications. Ammonia concentrations in municipal wastewater is many times higher than is allowable in aquacultural systems. Thus, the lower removal rates are what may be expected in aquacultural applications.

Design of RBC

The problem is to design an RBC for a striped bass culture system containing 20,000 lbs (9071 kg) of fish that are being fed 2 percent of body weight per day with a pelleted feed. It is further assumed that 3 percent of the feed becomes ammonia and the system operates at 24 ^{o}C. Based on this temperature and several other assumptions relative to flow rates, ammonia concentrations, and other variables it will be assumed that the RBC will remove 0.75 g TAN/m^2 -day (this is higher than what Miller and Libey (1985) found).

Ammonia production = (fish weight) (% body weight fed per day) (TAN produced per kg feed fed)

Ammonia produced = (9071 kg fish) (0.02) (0.03)

= 5.44 kg ammonia produced per day

Specific surface area needed = (Ammonia produced/ day)/ (Ammonia removal /m^2 /day

= (5.44 kg/day) / (0.75 g TAN/ m^2 /day)

= 7253 m^2

At this point one has to decide the diameter of the disks and calculate the length of unit needed based on the disk spacing along the shaft. This decision will be influenced by whether the unit will be purchased or will be constructed from available materials. Let us assume that we will use a commercially produced unit that is 3 m in diameter and has plates spaced every one cm along the shaft. Assuming the plates are flat the area of each plate is:

Area per plate = π (Radius)^2 (two sides of plate)

= π (1.5m)^2 (2)

= 14.2 m^2

The number of plates needed are:

Number of Plates = 7253 m^2 / 14.2 m^2

Number of Plates = 511 plates that are 3 m in diameter

Assume the plates are spaced every cm along the RBC shaft. Then the length of the RBC can be determined:

Filter length = (511 plates) (1 cm per plate)

= 511 cm = 5.11 m = 201.1 inches = 16.8 ft

The volume occupied by the filter is:

Filter volume = π R^2 (length)
= 36.12 m^3

Based on Easter-s (1992) tank volume to filter volume recommendation of 2.14:1 tank volume to filter volume, The tank to contain the filter must be:

Tank Volume = (36.12 m^2) (2.14)

= 77.3 m^3

The rotational speed of the filter is based on a peripheral velocity of the disks of 0.3 m/sec (1 ft/sec) as discussed above. Therefore, the disk rotational speed is:

(RPM) (Circumference of RBC) = 0.3 m/sec

$$RPM = \frac{(0.3 \text{ m/sec})\ (60 \text{ sec/min})}{\pi\ (3\text{m})}$$

RPM = 1.91 Revolutions per minute

Loading rate on the filter must then be calculated to determine the flow through the filter. In some cases the flow through the filter may be dictated by other than the filter design in which case the filter would have to be designed about a flow rate as a design constraint. For purposes here it is assumed there are no external constraints on the flow rate. Loading rates vary considerably depending on the use (i.e. BOD or ammonia reduction), influent concentration and other variables. However, Easter (1992) reported a loading rate in the VPI system of 0.18 L/min-m^2, Westerman et al. (1993) reported a loading rate of 0.28 L/min-m^2, and Miller and Libey (1985) used a loading rate of approximately 0.8 L/min-m^2. In this example a loading rate of 0.2 L/min-m^2 will be used. Thus the flow rate will have to be:

Flow rate through the filter = (0.2 L/min-m^2) (7253 m^2)

= 1450 L/min = 383 gal/min

The number of stages to be used depends on the organic content of the water, flow rate and several other variables. However, for our example four stages should be ample. The first stage could be larger than the others if there is a high organic content. If not, the four stages should be about the same size.

The structural design is all that remains for the RBC. Structural design is beyond the space allowed here but the following comments will be offered. During early use of RBC-s in aquacultural applications there were a considerable number of failures of the shaft and/or drive train of the filter. One must design the mechanical parts of the RBC to withstand 24 hour per day operation over long periods of time. The shaft and bearings must be designed to withstand the load of the RBC when it is completely broken in (not just its dry weight) and the shaft must be designed based on fatigue and not just strength. The wet weight of a broken in filter includes the bacteria growing on the filter. Film thickness data is not plentiful, but several authors have suggested that the films may get to be 4 or more mm thick (Grady, 1982). The density of these films are essentially the same as that of water. Thus, these films add significant weight to the RBC.

References

Antonie, R.L. 1976. Fixed biological surfaces-wastewater treatment: the rotating biological contactor. CRC Press, Inc. Cleveland, Ohio.

Beecher, L.E., A.A. Delos Reyes, D. G. Drennan, and R.F. Malone. 1997. Alternative media for enhanced nitrification rates in propeller-washed bead filters. pp. 263-275. IN: Advances in Aquacultural Engineering. Publication Number NRAES-105. Northeast Regional Agricultural Engineering Service.

Cornell University. Ithaca, New York.

Bower, C.E. and D.T. Turner. 1981. Accelerated nitrification in new seawater culture systems: effectiveness of commercial additives and seed media from established systems. Aquaculture 24:1-9.

Bower, C. E. and D. T. Turner. 1984. Evaluation of two commercial nitrification accelerators in closed seawater culture systems. Aquaculture 41:155-159.

Boyd C. E. 1979. Water Quality in Warm Water Fish Ponds. Auburn University Agricultural Experimentation Station, Auburn AL.

Carmigiani, G. M., J.B. Bennett. 1977. Rapid start-up of a biological filter in a closed aquaculture system. Aquaculture 11:85-88.

Colt, J. 1986. An introduction to water quality management in intensive aquaculture. Presented at the Northwest Fish Culture Conference, Springfield, OR.

Delos Reyes, A. A., and R. F. Malone, 1998. Design and evaluation of a commercial-scale, paddle-washed floating bead filter. pp. 196-205. IN: Success and failures in commercial recirculating aquaculture. Publication Number NRAES-98. Northeast Regional Agricultural Engineering Service. Cornell University, Ithaca, New York.

Easter, C. 1992. Water chemistry characterization and component performance of a recirculating aquaculture system producing hybrid striped bass. Unpublished M.S. thesis. Virginia Polytechnic Institute and State University, Blacksburg, Virginia.

Fenchel, T. 1979. Bacteria and Mineral Cycling, Academic Press, Inc. London.

Friedman, A.A., L.E. Robins, R.C. Woods, and J.R. Wauford. 1979. Effect of disk rotational speed on biological contactor efficiency. Journal Water Pollution Control Federation 51(11):2678-2690.

Fujie, Koichi, H.E. Bravo and H. Kubota. 1983. Operational design and power economy of a rotating biological contactor. Water Research 17(9):1152-1162.
Gilbert, W.G., J.F. Wheeler, and A. McGregor. 1986. Energy usage of rotation biological contactor facilities. Journal of the Water pollution Control Federation 58(1):47-51.

Grady, C. P. L. and H. C. Lim. 1980. Biological Wastewater Treatment, Marcel Dekker, Inc., New York, NY.

Grady, C. P. 1982. Modeling of biological fixed films: state-of-the art review. pp. 344-404. IN: Procedings of the First International Conference on Fixed-Film Biological Processes. Kingsland, Ohio.

Gujer, W. and M. Boller. 1986. Design of a nitrifying tertiary trickling filter based on theoretical concepts. Water Research 20(11):1353-1362.

Hochheimer, J.N., and F. W. Wheaton. 1991. Understanding biofilters, practical microbiology for ammonia removal in aquaculture. pp. 57-79. IN: Engineering Aspects of Intensive Aquaculture. Publication Number NRAES-49. Northeast Regional Agricultural engineering Service. Cornell University. Ithaca, New York.

Hochheimer, J. N. and F. W. Wheaton. 1997. Intensive culture of striped bass. Chapter 5, pp. 127-165. IN: Striped Bass and *Morone* Culture. Elesevier Science, Amsterdam, The Netherlands.

Hochheimer, J.N. 1990 Trickling filter model for closed system aquaculture. Unpublished Ph.D. dissertation. University of Maryland, College Park, MD.

Horrigan, S. G., A. F. Carlucci, and P. M. Williams. 1981. Light inhibitions of nitrification in sea-surface films. Journal of Marine Research 39(3):557-565.

Jansen, Jes la Cour, P. Harremoes and M. Henze. 1995. Treatments plants for nitrification. pp. 195-228. IN: Henze, M., P. Harremoes, J.C. Jansen, and E. Arvin. 1995. Wastewater Treatment: Biological and Chemical Processes. Springer-Verlag, New York.

Jones, R. D. and R. Y. Morita. 1985. Low temperature growth and whole cell kinetics of a marine ammonium oxidizer. Marine Ecology Progress Series 21:239-243.

Kawai, A., Y. Yoshida, and M. Kimata. 1965. Biochemical studies on the bacteria in the aquarium with a circulation system - II. Nitrifying activity of the filter sand. Bulletin of the Japanese Society of Scientific Fisheries 31(1):65-71.

Miller, G.E. and G.E. Libey. 1985. Evaluation of three biological filters suitable for aquacultural applications. Journal World Mariculture Society 16:158-168.

Odal, Shingo, K. Fujie, and H. Kubota. 1981. Effect of rotation speed on reaction rate on a rotating biological contactor. Journal Fermentation Technology 59(3):227-234.

Olson, R. J. 1981. Differential photo inhibition of marine nitrifying bacteria, Journal Marine Research 39(2):227-238.

Painter, H. A. 1970. A review of literature on inorganic nitrogen metabolism in microorganisms. Water Research 4:393-450.

Paloini, A.E. 1985. COD removal rates in RBC systems. Effluent and Water Treatment Journal 25(10):355-358.

Parker, N. E., M. A. Suttle and K. Fitzmayer. 1984. Total gas pressure and oxygen and nitrogen saturation in warm water ponds. Aquacultural Engineering 3(2):91-102.

Roberts, J. 1985. Mathematical models for trickling filter process. IN: Mathematical Models in biological Wastewater Treatment. S. E. Jorgensen and M. J. Gromiec, Editors. Elsevier, Amsterdam.

Speece, R. E., N. Nirmalakhandan, and Y. Lee. 1988. Design for high purity oxygen absorption and nitrogen stripping for fish culture. Aquacultural Engineering 7(3):201-210.

Spotte, S. 1979. Seawater Aquariums, John Wiley & Sons, New York, NY.

Summerfelt, S. T., and J.L. Cleasby. 1996. A review of hydraulics in fluidized-bed biological filters. Transactions of the American Society of Agricultural Engineers 39(3):1161-1173.

Timmons, MB. 1997. Professor, Cornell University. Ithaca, New York. Personal Communication.

USEPA (United States Environmental Protection Agency). 1975. Process Design Manual for Nitrogen Control. Office of Technology Transfer, United States Environmental Protection Agency, Washington, DC.

Water Pollution Control Federation. 1983. Nutrient control. Manual of practice No. FD-7. Washington, D.C.

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

Wortman, B. 1990. Effect of temperature on biodrum nitrification. Unpublished M.S. thesis. University of Maryland. College Park, MD.

Weng, C. And A.H. Molof. 1974. Nitrification in the biological fixed-film rotating disk system. Journal Water Pollution Control Federation 46:1674.

Westerman, P.W.A., T.M. Losordo and M.L. Wildhaber. 1993. Evaluation of various biofilters in an intensive recirculating fish production facility. pp. 326-334. IN:Techniques for Modern Aquaculture. American Society of Agricultural Engineers, St. Joseph, Michigan.

Wheaton, F. W., J.N. Hochheimer, G.E. Kaiser. 1991. Fixed film nitrification filters for aquaculture. pp. 272-303. IN: Aquaculture and Water Quality. D.E. Brune and J.R. Tomasso, editors. Advances in World Aquaculture, Vol. 3. World Aquaculture Society, Baton Rouge, Louisiana.

Wheaton, F. W. , J.N. Hochheimer, G. E. Kaiser, R. F. Malone, M.J. Krones, G.S. Libey, and C. C. Easter. 1994. Nitrification filter design methods. pp. 127-171. IN: Aquaculture Water Reuse Systems: Engineering Design and Management. M.B. Timmons and T. M. Losordo, Editors. Volume 27, Developments in Aquaculture and Fisheries Science. Elsevier. New York.

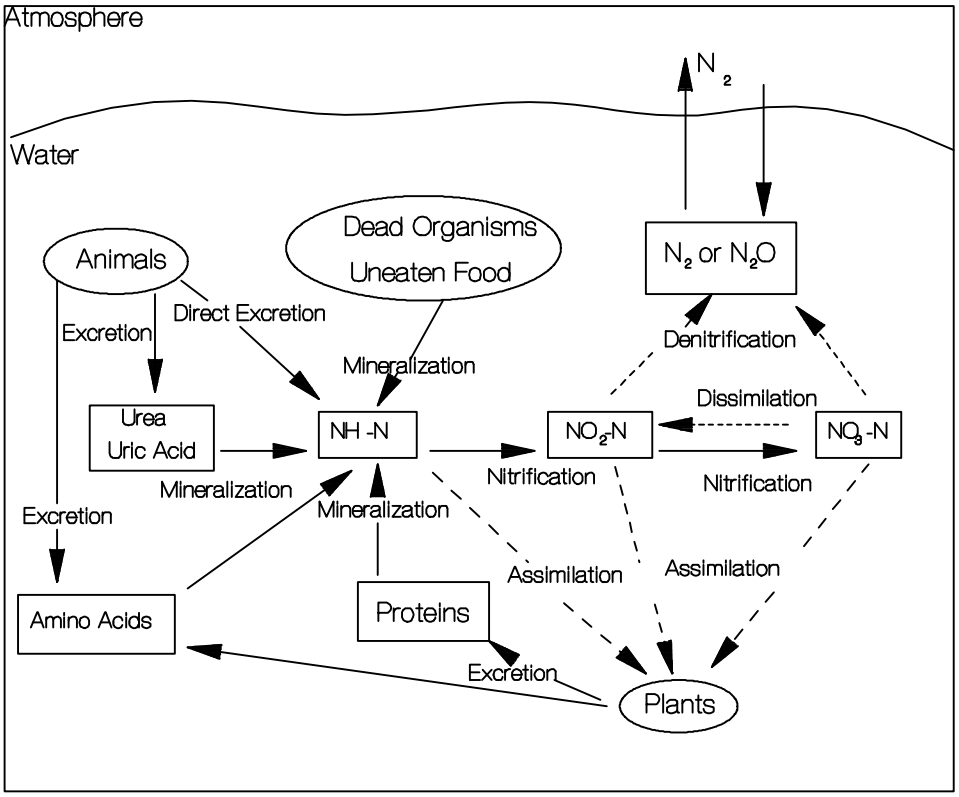

Figure 1. The nitrogen cycle in closed system aquaculture (Spotte, 1979)

Sizing and Management of Floating Bead Bioclarifiers

Ronald F. Malone
Chevron USA Professor

Lance E. Beecher
Research Associate

Aurelio A. DeLosReyes, Jr.
Postdoctoral Research Associate

Department of Civil and Environmental Engineering
Louisiana State University

Introduction

The use of bead filters with floating plastic media dates back to the mid-1970's when they were utilized as biofilters to support high density rearing of food and game fish in Idaho (Cooley 1979). Although successful, adoption of these early air-washed bead filters at other sites was limited. In the late 1980's, work performed at Louisiana State University demonstrated that a hydraulically washed bead filter was capable of providing solids control (clarification) and biofiltration for a high-density catfish rearing system (Wimberly 1990). Development of mechanically washed units (Malone 1992, 1995), which were compact and simple to operate, overcame many of the operational difficulties experienced by earlier designs. Shortly thereafter, the "bubble-washed" or "hourglass" configuration (Malone 1993) was developed and tested for use on outdoor ornamental or garden ponds. Since 1989, bead filters have been tested on food fish holding systems (such as tilapia, catfish, striped bass, and trout), along with a wide variety of specialized applications (including ornamental fish, alligators, crawfish, crabs, and oysters). This paper summarizes what is known about the use of floating bead filters, with particular emphasis on their biofiltration capabilities and their proper management for combined solids capture and nitrification.

Bead Filter Types and Operation

To meet the definition of floating bead filter used here, the unit selected must display two modes of operation: 1) a fixed-bed filtration mode employing upward flowing water, and 2) a fluidized backwashing mode. The floating bead filter must also utilize a media consisting of small spherical 3 - 5 mm plastic beads that float and do not contain any additives that may kill bacteria or be harmful to fish. The media should display a specific surface area of about 1150 m^2/m^3 (350 ft^2/ft^3) assuring that sufficient surface area exists for biofilm development. Rod-shaped beads with a maximum dimension of 3-mm (1/8-inch) are acceptable, but flattened disk-shaped beads are not. Porosity of the bed must exceed 35 percent. While filters of the principal author’s design are illustrated here, the fundamental processes occurring in all bead filters are the same. However, cost, reliability, and ease of operation vary among designs.

Bead filters are classified as "Expandable Granular Biofilters." They are designed to function as a physical filtration device (or clarifier) by removing solids, while simultaneously encouraging the growth of desirable bacteria that remove dissolved wastes from the water through biofiltration processes. The granular nature of the bead bed allows it to be cleaned to release solids and excessive biofloc, while providing large amounts of surface area for the nitrifying bacteria to attach. This permits large amounts of wastes to be treated using a relatively compact filter. Bead filters capture solids through four identifiable mechanisms, which include straining, settling, interception, and adsorption (Ahmed 1996; Drennan et al. 1995; Malone et al. 1993). They perform well in the control of suspended solids across a broad spectrum of sizes with nearly 50 percent of fine solids in the 5-10 micron range, being removed in a single pass (Ahmed 1996). Although inherently excellent clarifiers, the market for the bead filters is largely generated by their ability as biofilters.

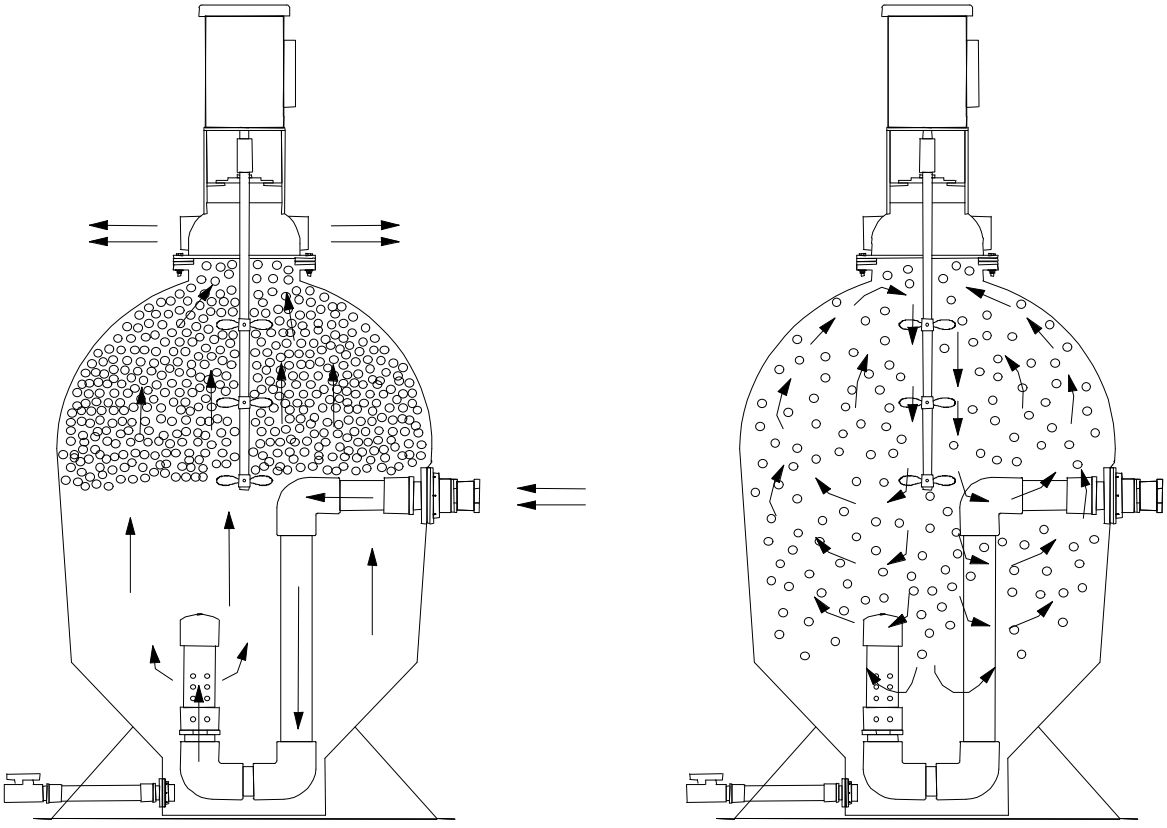

Figure 1. The Propeller-Washed Bioclarifiers are aggressively washed once a day or once every other day (Malone, 1992).

Bead filters differ primarily in their mode of backwashing. The means of backwashing is critical since it should be done in a way that the removal of accumulated solids in the system would not adversely impact the biofilm in the bead bed. Perhaps the most common type is the propeller-washed bioclarifier, consisting of a filtration bed of floating

plastic beads that are intermittently washed by motor-driven embedded propellers. The propeller-washed bioclarifiers are operated in the filtration mode most of the time (Figure 1). As recirculating water passes through the bed, suspended solids are captured and the biofiltration processes are active. Backwashing or cleaning of the bead bed is accomplished by turning off the pump and/or closing the inlet valve and then activating the mixing motor and propellers. The objective of the backwashing step is to release solids and excessive biofloc trapped between the beads. This is accomplished by the hydraulic shear forces induced by the propellers as the beads are thrust downward into the expansion zone and by contact between the beads as they swirl. The propeller-washed bead filters are designed to input a lot of cleaning energy in a short period of time. Excessive washing just damages the biofiltration performance without benefiting clarification. Once the bed has been expanded and agitated for several seconds, the mixing motor is turned off and the settling mode of operation is initiated. Typically, the filter is left idle for 3 - 5 minutes. The beads float upward reforming the filtration bed, while the sludge is concentrated in the settling cone. The final mode of operation is sludge removal. Settling is very effective and it is not necessary to drain the filter completely. Commonly, the sludge drain line is equipped with a clear segment of pipe, which allows the clarity of the discharged water to be observed. As soon as the draining water appears to be as clear as the rearing tank's water, the sludge valve is closed. This approach greatly reduces water loss without impacting filter performance.

While propeller-washed units dominate large-scale operations with units of up to 2.8 m^3 (100 ft^3), bubble-washed units are most frequently employed for small-scale systems. Bubble-washed bead filters, typically less than 0.28 m^3 (10 ft^3), are designed to be self-washing when drained. The discharge of the filter is equipped with a valve (or check valve) that prevents the back-flow of air into the filter when the sludge (or drain) valve on the bottom is opened. This causes a vacuum to form within the filter housing. An air inlet valve, located on the side of the filter just below the washing throat, is opened so that air can be sucked into the filter as it drains (Figure 2). This constriction is critical because as the water leaves the filtration head, the beads are fluidized downward, and pass through the narrow throat where they are scrubbed further by the rising bubbles. The washing process is complete once the filter is drained and all the beads have dropped into the expansion chamber. Readjusting the valves and refilling the filter with the recirculation pump starts the next filtration cycle. In contrast to propeller-washed units, bubble-washed bead filters lose the entire water volume contained in the filter during backwashing.

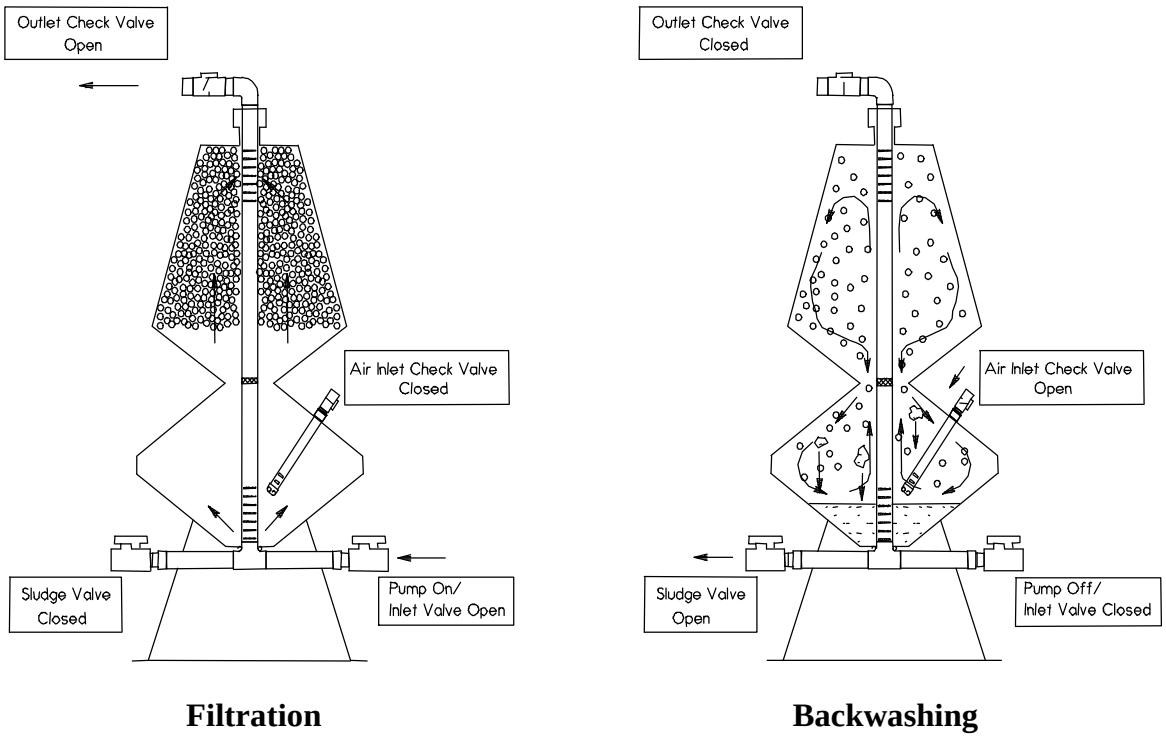

Figure 2. Bubble-washed bead filters use a constricted throat (Malone 1993) to intensify air washing. These units work best when backwashed frequently.

Management for Nitrification

The bead bioclarifier management plan for high nitrification rates assures that: 1) water quality conditions favorable for nitrification exist, 2) an appropriate mass of nitrifying bacteria reside in the filter, and 3) critical nutrients are rapidly transported to the bacteria (Figure 3). All three issues must be addressed to assure that the bead bioclarifier displays a high rate of nitrification.

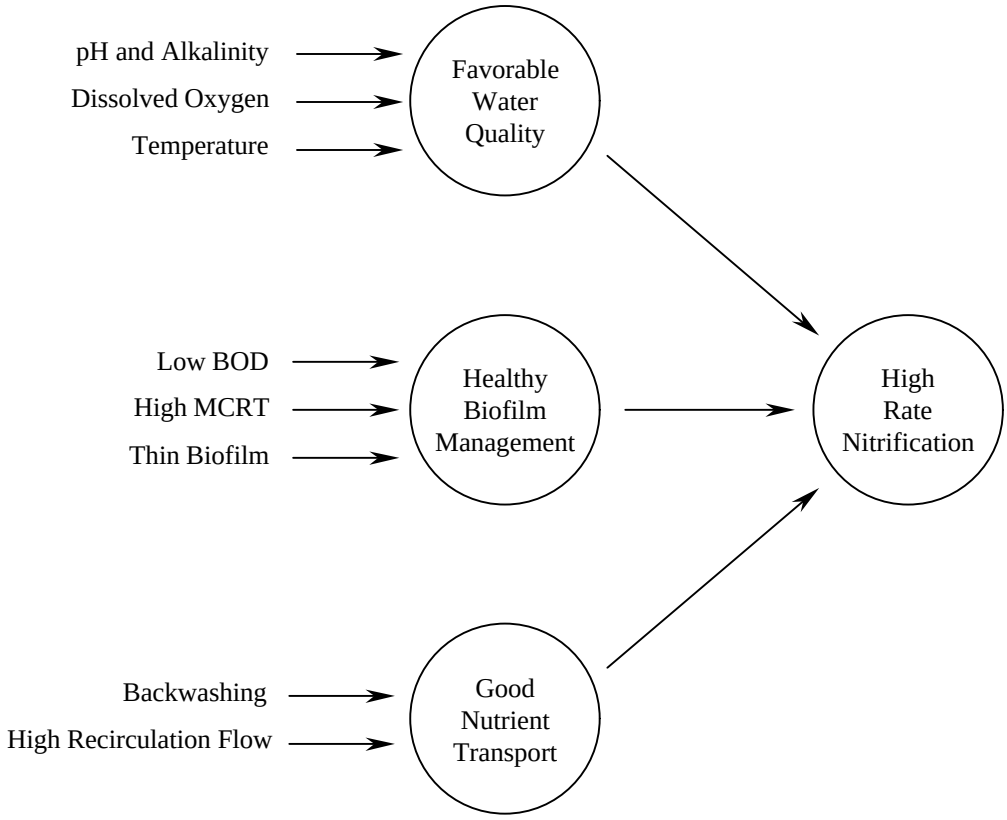

Figure 3. Good nitrification rates are easy to achieve with a floating bead bioclarifier provided attention is given to the filter's management.

Water Quality

Temperature. Floating bead filters have been used for a wide variety of applications ranging from the warmwater production of tilapia to coldwater production of salmon and trout. However, only the warmwater (20-30^{o} C) applications have been extensively documented. Conversion rates in coldwater (10-20^{o} C) applications are generally assumed to be lower, but there is little evidence to substantiate this inference. Additional studies are needed in this area. Typically, the authors use the ornamental criteria for coldwater growout applications.

Oxygen. The most important water quality parameter is oxygen. Nitrification in bead bioclarifiers is not substantially influenced by oxygen as long as the oxygen level in the filter effluent is above 3.0 mg/L. When the dissolved oxygen level coming out of the filter is above 3.0 mg/L, it is generally safe to assume that most of the bed has enough

oxygen to keep the nitrifying bacteria working at top speed. By the time effluent oxygen levels drop to about 2.0 mg/L, portions of the bead bed will be impaired by low oxygen. Below this level first *Nitrobacter*, then *Nitrosomonas* will slow down as the rate of oxygen diffusion into the bacterial biofilm begins to limit the nitrification process. High rate nitrification requires that the entire bed be kept working at maximum speed. The oxygen supply is controlled primarily by the flow rate through the filter.

Alkalinity. For all practical purposes, bicarbonate ions define total alkalinity levels in a recirculating system. The nitrification process consumes alkalinity at a rate of about 6-7 mg $CaCO_3$ per milligram of TAN converted (EPA 1975). This consumption of alkalinity must be addressed by water exchange or by direct chemical addition. The chemical preferred by the authors for this purpose is sodium bicarbonate ($NaHCO_3$), more commonly known as baking soda. Sodium bicarbonate has many desirable properties, including a high water solubility (>100,000 mg/L at 25 ^oC), and a low potential for overdosing since it takes a relatively large amount to raise the pH substantially. Sodium bicarbonate is safe to both fish and humans, and is widely available commercially in bulk quantities.

The amount of sodium bicarbonate required will be virtually the same regardless of the target alkalinity value selected. Procedures for calculating dosages have been clearly defined (Loyless and Malone 1997). If water exchange is minimal, the bicarbonate dosage requirement can be as high as 0.24 kg/kg feed (0.24 lb/lb feed). Generally a moderate sized system will require the addition of sodium bicarbonate every two to three days. Optimum nitrification is normally associated with alkalinity levels above 100 mg $CaCO_3$/L, but systems can be operated at lower levels within the bounds of the criteria because of safety factors applied to the nitrification capacities of the floating bead bioclarifiers.

pH and Carbon Dioxide. Bicarbonate additions not only increase the bicarbonate supply but also help raise the pH, benefiting the nitrification process. Inhibition of the nitrifying bacteria under growout conditions is noticeable once pH drops below 7.0 (Loyless and Malone 1997; Allain 1988; Paz 1984; Siegrist and Gujer 1987). When peak performance is demanded from a floating bead bioclarifier, a pH value in the range of 7.5-8.0 is normally recommended.

The ratio of bicarbonate ions over dissolved carbon dioxide concentration controls the system's pH. As the system is loaded, the carbon dioxide produced by the fish and bacteria rises. This carbon dioxide accumulation will cause a drop in the pH. At the same time, the nitrification process consumes bicarbonate ions. The combination of high dissolved carbon dioxide and low bicarbonates can create a radically low pH in the range of 4-5, which severely inhibits nitrifying bacteria. If the pH remains low after the alkalinity has been adjusted, then the system has high carbon dioxide levels. The CO_2 levels are controlled by the stripping rate of the aeration and degasification devices. Carbon dioxide is not normally an issue in systems employing blown air for aeration (Loyless 1995), but must be watched carefully in systems operating in enclosed buildings

and in systems using pure oxygen since oxygen injection equipment usually do not provide adequate gas exchange to strip carbon dioxide. Design guidelines for carbon dioxide stripping towers have been established (Grace and Piedrahita 1994; Colt and Bouck 1984).

As recommended, the authors adjust majority of the bead bioclarifier high-density growout systems they operate to a pH in the range of 7.5-8.0 with alkalinity falling between 150 to 200 mg $CaCO_3$/L. High carbon dioxide levels encountered (5-15 mg/L) often force values on the lower end of the pH range (7.5) and the upper end of the alkalinity range (200 mg $CaCO_3$/L) to be targeted (Loyless and Malone 1997). Selection of a target pH above 8.0 is generally avoided due to the increasing prevalence of the toxic unionized molecular form of ammonia (NH_3) at higher pH values (Huguenin and Colt 1989).

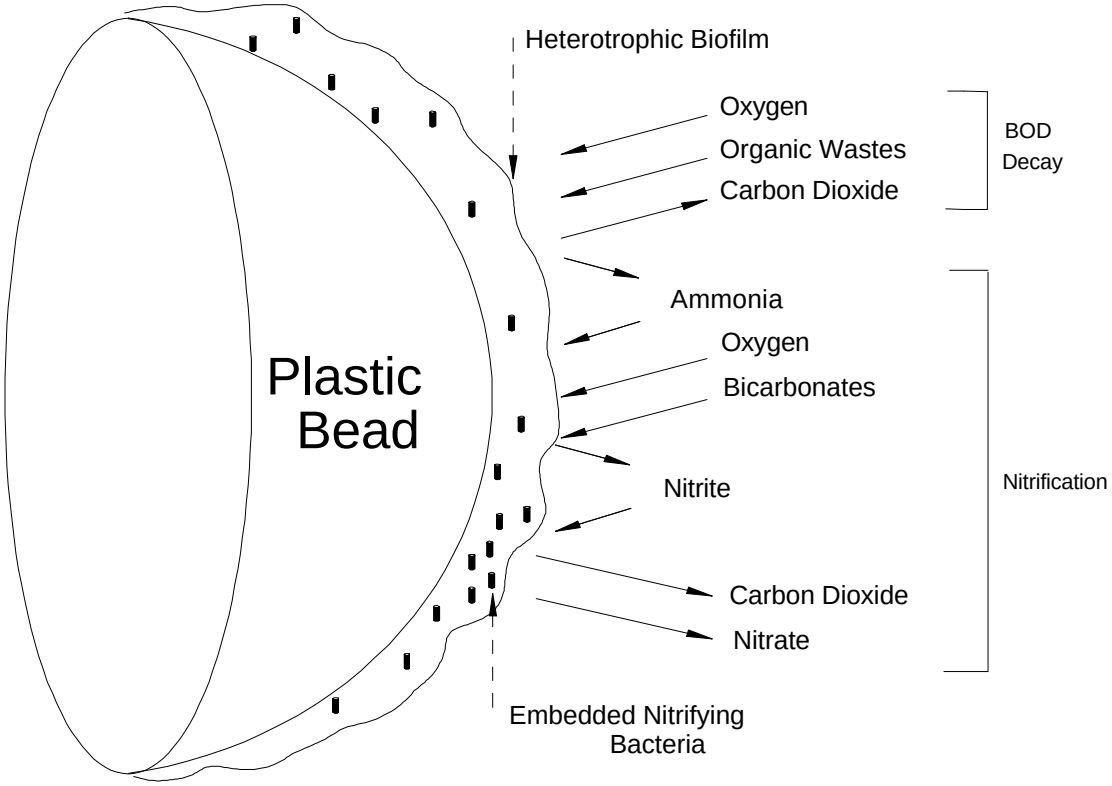

Figure 4. Each bead is coated with a heterophic bacterial film containing a population of nitrifying bacteria. There are approximately 20 million beads in a cubic meter (550,000 per cubic foot) with a surface area of about 1150 m^2/m^3.

Biofilm Management

In the biofiltration mode, bead bioclarifiers operate as fixed-film reactors. Each bead (Figure 4) becomes coated with a thin film of bacteria that extracts nourishment from the recirculating water as it passes through the bead bed. There are two general classifications of bacteria that are of particular interest: heterotrophic and nitrifying. The two bacteria co-exist in the filter, and understanding their impact on each other as well as on the filter is critical.

The group of heterotrophic bacteria encompasses a great number of genera/species, which share the common characteristic of extracting nourishment from the breakdown (decay) of organic matter. About 60 percent of the organic matter consumed is converted to bacterial biomass, whereas, the balance (40 percent) is converted to carbon dioxide, water, ammonia, and other chemicals. Biochemical oxygen demand (BOD) is largely an indirect measure of the biodegradable organic material in water. If the BOD in the water being treated is very high (> 20 mg -O_2/l) and conditions are favorable, the heterotrophs will quickly dominate the bead bed because of their rapid growth (doubling their population every ten to fifteen minutes). This allows for quick overpopulation of heterotrophs, which consume large amount of dissolved oxygen, and compete for space with slower growing nitrifying bacteria.

Nitrifying bacteria are specialists, extracting energy for growth from the chemical conversion of ammonia to nitrite and nitrite to nitrate. Nitrate is a stable end product that is generally non-toxic, unlike ammonia and nitrite. Nitrifying bacteria are generally assumed to be composed of two genera (*Nitrosomonas* and *Nitrobacter*), although recent studies indicate that other genera are involved (Hovanec et al. 1998; Hovanec and DeLong 1996). They are very slow growing and sensitive to a wide variety of water quality factors. It is not surprising that most bead filters used for biofiltration are managed to optimize conditions for nitrification.

Acclimation. Development of a biofilm layer on the media is required for biofiltration. Initially the biofilter has no bacteria and the culture must be started. The process of growing the initial bacterial culture in the biofilter or adjusting an established culture to a change in loading is called acclimation. The best way to acclimate a recirculating system with a biofilter is to just add a few hardy fish, turtles, or molluscs to the system and start to feed them. The heterotrophic bacteria will grow rapidly and quickly attach themselves to the beads. The nitrifying bacteria, however, are very slow reproducers and may require almost thirty days (2 - 3 weeks is more typical) to establish themselves.

Figure 5 illustrates the classical pattern of TAN and nitrite concentrations observed during filter acclimation with animals (Manthe and Malone 1987). The process starts with an increase in TAN concentration. You will know that the first group of nitrifiers responsible for ammonia conversion to nitrite are present in large numbers when the ammonia excreted by the fish stops accumulating and suddenly (within 36 hours) drops to near zero level. At the same time there will be a sudden rise in nitrite level, followed

by a gradual increase which will continue until suddenly the second group of bacteria, *Nitrobacter*, catch up with their new food supply and the nitrite concentrations plummet. The filter is now considered acclimated to a light loading. This initial stage of acclimation is critical because during this period, populations of bacteria which can effectively attack the specific wastes produced by the animals become established and these bacterial populations adjust to operate under the water quality conditions and temperature regime found in your system. This unique culture of bacteria will remain in the biofilter for years if it is just treated with a little common sense.

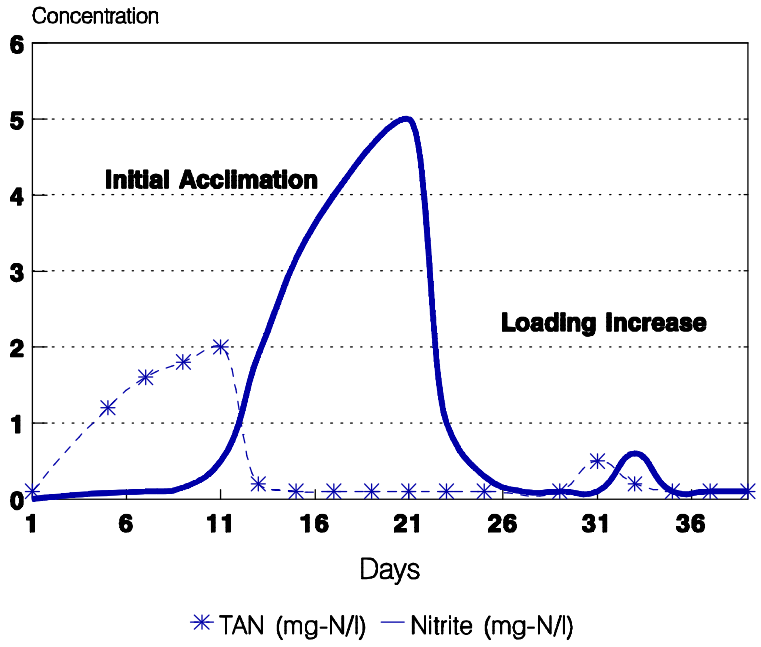

Figure 5. Acclimation of a bead filter with fish should take about three weeks. The nitrite peak is always higher than ammonia peak.

Backwashing. Organic solids not only breakdown to produce ammonia, but also encourage rapid heterotrophic bacteria growths that compete with the nitrifiers for space and potentially limiting nutrients such as oxygen (Zhang et al. 1995; Matsuda et. al. 1988). This phenomenon is not unique to bead filters; studies by Figueroa and Silverstein (1992) on rotating biological contactors, and Bovendeur et al. (1990) on trickling filters both associate increasing BOD levels with declining rates of nitrification. Backwashing removes the bulk of the heterotrophic bacteria, the captured TSS which form a major portion of their food source, and unfortunately, a lot of nitrifying bacteria. If the interval between backwashing is too short, the nitrifying bacteria will not have time to re-establish their population and a gradual washout occurs, dramatically limiting the nitrification capacity of the filter. The latter situation is avoided by managing the "Mean Cell Residence Time" (or MCRT) of the biofilm in the filter. A floating bead bioclarifiers optimum nitrification performance occurs when sludge accumulation

concerns are properly balanced with MCRT concerns by adjusting the backwashing frequency.

Once the basic water quality issues are addressed, backwashing becomes the principal management tool. It can be used to dramatically improve nitrification. This aspect of bead bioclarifier management has been extensively studied experimentally, and evaluated by a mathematical model (Golz 1997). These studies indicate that two classes of bioclarifiers exist. Hydraulic and air washed units fall into the "gently washed" category. These filters display reduced biofilm abrasion during backwashing and must be washed at a high frequency. Optimum performance for heavily loaded filters occurs when the filters are washed several times daily (Sastry 1996; Wimberly 1990). Conversely, propeller-washed and paddle-washed filters inflict a relatively heavy biofilm damage during backwashing, and are considered "aggressively washed." They must be backwashed infrequently, usually every other day, to allow growth of biofilm, thus, avoiding MCRT problems (Chitta 1993; Malone et al. 1993). Ironically, both classes of filters display nearly the same peak conversion rate of about 350-450 g TAN/m^3-day (10–13 g TAN/ft^3 -day).

Flow rate. Another important issue in optimizing a biofilters operation is nutrient transport. Assurance that the bacteria within the biofilm are always presented with sufficient levels of TAN and oxygen to keep the nitrification process proceeding at a high rate is very important. Maintaining good water quality in the rearing tank does not assure that the bacteria, which are residing in the depth of a packed bed of beads, are seeing these same conditions. Good nutrient transport requires rapid mixing first between the rearing tank and the biofilter, and secondly, even flow distribution within the bead bed. Biofilms actively consume (or convert) essential nutrients, which rapidly deplete supplies in the water immediately adjacent to them. Water must flow quickly and evenly through the bead bed to assure that the depleted waters are uniformly and rapidly displaced. If excessive solids or biofloc clogs a section of the bed then the nitrifying bacteria become inactive, lowering the performance of the bed as a whole.

The water recirculation rate of about 50 Lpm/kg feed-day (6 gpm/lb feed-day) is recommended to assure good oxygen delivery to the bioclarifier. This recirculation rate also assures adequate circulation between the rearing tank and bioclarifier, which assures tank TAN build-ups due to poor mixing are avoided. Extensive observations (Sastry 1996; Chitta 1993; Wimberly 1990) show that the oxygen consumed in filtration (or OCF, Manthe and Malone 1987) is around 165 g/kg feed (75 g/lb-feed**).** Experience has shown that filters operated with effluent oxygen levels below 2 mg/L are clearly not operating at their maximum capacity because of oxygen delivery problems (Golz 1997; Manthe et. al. 1988). When oxygen levels in the holding tank are maintained between 5 and 6 mg/L, the recirculation criterion includes a safety factor of about 50 percent.

In the authors' experience, the biofiltration capacity of bioclarifiers always improves with flow rate. The filters operate best with high flow rate and low removal efficiencies. Removal efficiencies can be calculated from Equation 1:

$$E = \frac{TAN_I - TAN_E}{TAN_I}(100) \quad (1)$$

where: E = Single pass removal efficiency (percent)
TAN_I = Influent TAN concentration (mg N/L)
TAN_E = TAN concentration in the effluent from the filter (mg N/L)

TAN removal efficiencies in the range of 25 percent are usually targeted when high nitrification performance is demanded. If the efficiency of a bed exceeds 50 percent, significant additional nitrification capacity can be realized by increasing flow rates. This raises the mean TAN concentration in the bed improving the gradient of diffusion into the biofilm, which results in improved conversions.

Ideally, recirculating systems should be designed so that back pressure on the pumps is low, minimizing energy requirements. The operational back pressure of floating bead bioclarifiers is dependent on the backwash frequency and feed loading rate. Bead filters that are “aggressively washed” (propeller- and paddle-washed) should be operated with a pump capable of delivering the required flow at pressures of about 69 kPa (10 psi) to accommodate extended backwash intervals of 1-2 days. “Gently washed” filters (air injected and hydraulic) can be matched with a lower head pump (34.5 kPa [5 psi]) if they are operated under a high frequency washing regime (>2 backwashes per day).

Management Strategy

A good biofilm management plan must address the needs of the nitrifying bacteria in terms of water quality, nutrient transport, and, biofilm harvesting. Table 1 summarizes the operational ranges for three operational levels. Since the water quality aspects are easily addressed, biofilm management efforts focus on backwashing and its impact on nitrifying bacterial biomass levels and flow rate through the bed. Periodic monitoring of the biofilters performance facilitates optimization in situations where peak nitrification performance is demanded. A drop in oxygen in the bioclarifier effluent usually evidences problems with water circulation rates. If the dissolved oxygen level coming out of the filter is less than 2 mg/L then the performance of the bioclarifier is (or on the verge of being) severely impaired by oxygen limitation. If the tank dissolved oxygen levels are above that prescribed in Table 1, then the problem rests with the pump or the backwashing regime of the filter. If the problem is not corrected the bioclarifier will first generate nitrite, then TAN conversion rates will fall, and in extreme cases, off-flavor problems with the fish will occur. Monitoring is not normally required when filters are operated at or below their recommended design capacities. These filters are normally set at a convenient backwash interval (once a day is problem is not corrected the bioclarifier will first generate nitrite, then TAN conversion rates will fall, and in extreme cases, off-

flavor problems with the fish will occur. Monitoring is not normally required when filters are operated at or below their recommended design capacities. These filters are normally set at a convenient backwash interval (once a day is generally good for all types) and only pH is monitored every other day or so to detect alkalinity exhaustion.

Table 1. The operational ranges for the management factors controlling nitrification in floating bead bioclarifiers are known.

Management Factor		Operational Range		
		Broodstock	Ornamental	Growout
Temperature, ^{o}C		20 - 30	20 - 30	20 - 30
Effluent Oxygen, mg/L		> 2.0	> 2.0	> 2.0
Feed Loading, kg/m^3-day [lb/ft^3-day]		< 4 < 0.25	< 8 < 0.5	< 16 < 1.0
Flow rate, Lpm/m^3 [gpm/ft^3]		$\geq$ 400 [$\geq$ 3]	$\geq$ 400 [$\geq$ 3]	$\geq$ 800 [$\geq$ 6]
Alkalinity, mg $CaCO_3$/L		> 50	> 80	> 100
pH		6.5 - 8.0	6.8 - 7.0	7.0 - 8.0
Backwash Interval, days	Aggressively washed	1 - 7	1 - 3	1 - 2
	Gently washed	1 - 3	1 - 2	0.5 - 1

Sizing Rationale

The primary method for the sizing of floating bead bioclarifiers is based on a volumetric organic loading rate. The ultimate source of organics in a recirculating system is the feed; therefore the sizing criterion is expressed in terms of weight of feed applied daily per cubic meter of beads (kg/m^3-day). This criterion assumes:

1) the filter is being employed as a bioclarifier,
2) organic loading is the principal factor controlling nitrification conditions within the bioclarifier,
3) the organic/nitrogen ratio is relatively consistent across a wide spectrum of feeds,
4) the filter is managed to sustain nitrification.

The criterion of 16 kg/m^3-day (1 lb/ft^3-day) has been widely tested and has proven to be robust in the commercial sector. At this feeding level, the filters can reliably provide

solids capture, BOD reduction, and nitrification while sustaining water quality conditions suitable for the growout of most food fish species. TAN and nitrite levels can be expected to remain well below 1 mg N/L. At this level the filter's biofiltration function is not stressed. This allows sufficient reserve nitrification capacity to tolerate the range of feed protein contents, and management strategies encountered in most commercial applications. Reduction of the criterion to 8 kg/m^3-day (0.5 lb/ft^3-day) allows the reliable maintenance of water quality conditions suitable for more sensitive species; particularly ornamental goldfish, koi, and tropical fish where fin quality, coloration, and appearance are critical to marketing objectives. Finally, a loading guideline of 4 kg/m^3-day (0.25 lb/ft^3-day) is used for breeding and broodstock maintenance programs where pristine conditions are justified by the value of the stock. Peak carrying capacities for the various bead filter models discussed in this paper vary from 24-32 kg/m^3-day (1.5-2.0 lb/ft^3-day) when filled with standard spherical beads. However, at these higher organic loading rates backwashing of the filters must be knowledgeably tuned to avoid problems (Sastry 1996; Chitta 1993; Wimberly 1990).

Another approach to sizing bead bioclarifiers is in terms of volumetric nitrification capacity (Malone et al. 1993). This criterion is based on the observation that a wide spectrum of floating bead (and other) biofilters are found to display areal conversion rates with a magnitude of about 300 mg TAN/m^2-day (28 mg TAN/ft^2-day) in recirculating systems with TAN and nitrite levels between 0.5 and 1.0 mg N/L. The authors suspect that this plateau of performance reflects TAN diffusion constraints as the biofilm thickens in response to increased loading (Harremoes 1982). Below a TAN concentration of about 1.0 mg N/L, laboratory evidence and empirical observations indicate the conversion rate declines with the TAN concentration (Chitta 1993). However, the relationship is complex and is impacted by the bead filter design, backwash frequency, the bed's porosity, flow rate, and a variety of other parameters. Thus, over, the years the authors have tended to simplify the process by utilizing volumetric conversions that are related to the areal conversions through the media's specific surface area (typically 1150-1475

Table 2. Volumetric TAN conversion rates (VTR) used to size floating bead bioclarifiers for warmwater applications.

Loading Regime	Feed Loading kg/m^3-day [lb/ft^3-day]	Expected Concentration mg N/L		VTR g TAN/m^3-day [g TAN/ft^3-day]
		TAN	NO_2-N	
Broodstock (very light)	< 4 [< 0.25]	<0.1	<0.1	70 [2]
Ornamental (moderate)	< 8 [< 0.5]	<0.3	<0.3	180 [5]
Growout (heavy)	<16 [< 1.0]	<1.0	<1.0	350 [10]

m^2/m^3 [350-450 ft^2/ft^3]) without going through the painstaking process of estimating specific surface area. The result is a conservative design table (Table 2). This table can be used in conjunction with Equation 2 to estimate the size of the bead filter:

$$V = (1-I_s)(E_{TAN})\ W / VTR \qquad (2)$$

where: V = volume of the bead bed in m^3 (or ft^3)
I_s = *In situ* nitrification fraction (unitless)
E_{TAN} = TAN excretion rate in g TAN/kg feed (or g TAN/lb feed)
W = Feed rate in kg/day (or lb/day)
VTR = Volumetric TAN conversion rate in g TAN/m^3-day (or g TAN/ft^3-day)

The *in situ* nitrification fraction recognizes the effect of nitrification occurring on the sidewalls of tanks, and in particular, the systems piping (Mia 1996). A value of $I_s = 0.3$ is conservatively estimated, although values in excess of fifty percent are frequently observed. The TAN excretion rate is normally assumed to be around 30 g/kg (13.6 g/lb) for a 35 percent protein feed typically used to support warmwater fish (Malone et al. 1990; Wimberly 1990). This value may be proportionally increased when a high protein feed is employed.

Three levels of performance for the bead bioclarifiers have been defined (Malone and DeLosReyes 1997): 1) broodstock (lightly loaded, very clean), 2) ornamental (moderate organic loading, clean), and 3) food fish growout (heavy organic loading, tolerable water quality). The volumetric conversion rates are assumed to decline with substrate concentration, as would be the case with TAN diffusion limitation. The values can be expected to hold for fresh and saltwater applications where the temperature is maintained between 20 and 30 ^oC.

The design values given are conservative with the indicated values easily achievable when the filters are managed to sustain nitrification. However, the bead filter nitrification performance can vary widely (Figure 6). Peak conversion rates are almost always associated with careful management (Sastry 1996; Chitta 1993; Wimberly 1990). Bead filters primarily operated for clarification display nitrification performance that are largely supplemental (DeLosReyes and Lawson 1996; DeLosReyes 1995; MP&L 1991).

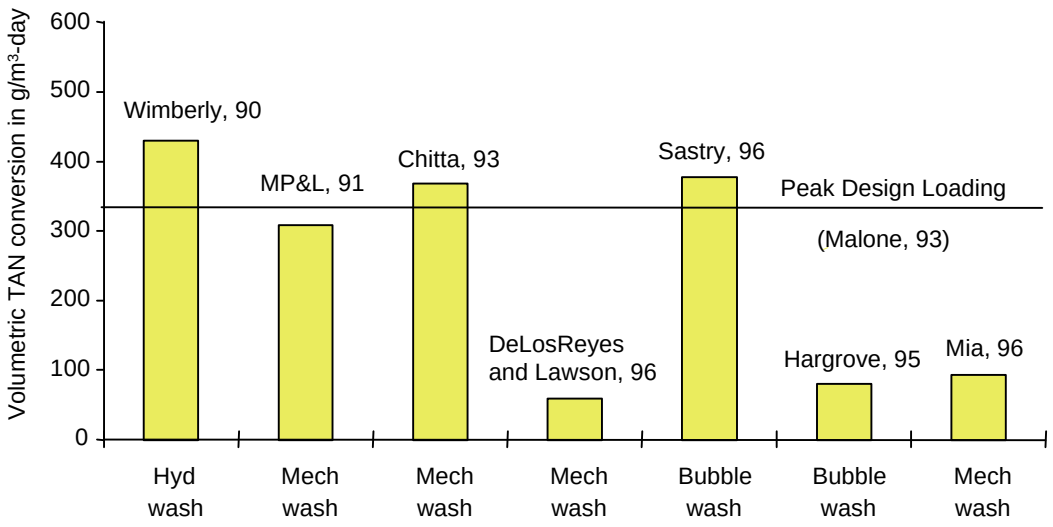

Figure 6. Bead filter performance can vary dramatically with loading and management.

Performance Evaluation

The volumetric TAN conversion rate (VTR), the volumetric nitrite conversion rate (VNR), and the volumetric oxygen consumption rate of the biofilter (OCF) can be used as principal parameters for evaluation and comparison of biofilter performance. The volumetric TAN conversion rate can be obtained by using Equation 3:

$$VTR = K_c (TAN_I - TAN_E) Q / V \quad (3)$$

where: VTR = volumetric TAN conversion rate in g TAN/m^3-day (or g TAN/ft^3- day)
Q = flow rate through the filter in liters per minute or Lpm (or gpm)
K_c = conversion factor of 1.44 (or 5.45)
TAN_I, TAN_E, and V are as defined previously.

The actual level of nitrification occurring in the filter may be higher since TAN is a by-product of heterotrophic breakdown of nitrogen rich organic compounds and biofloc. Despite its limitations, VTR allows the relationship between design and management parameters to be more closely examined.

The volumetric nitrite conversion rate (VNR in g NO_2-N/m^3-day or g NO_2-N/ft^3-day) is defined by Equation 4:

$$VNR = VTR + K_c (NO_{2I} - NO_{2E}) Q / V \quad (4)$$

where: NO_{2I} is the influent nitrite concentration in mg N/L
NO_{2E} is the effluent nitrite concentration in mg N/L
VTR, K_c, Q, and V are as defined previously.

As this equation illustrates, the readings of influent and effluent nitrite must be combined with the volumetric ammonia conversion rate to determine the level of *Nitrobacter* activity since nitrite is being produced as the ammonia is converted within the bed. Because of this phenomenon, the apparent nitrite removal efficiency may be near zero (i.e. influent and effluent values are nearly identical), although the filter may be vigorously processing nitrite to nitrate.

The volumetric oxygen consumption rate (OCF in g O_2/m^3-day) is very helpful in the management of bead filters. It indicates the total amount of bacterial activity within the filter, and can be obtained using Equation 5:

$$OCF = K_c (DO_I - DO_E) Q / V \quad (5)$$

where: DO_I is the influent dissolved oxygen concentration in mg O_2/L
DO_E is the dissolved oxygen concentration in mg O_2/L in the filter effluent
K_c, Q, and V are as defined previously.

OCF measures the combined respiration of the nitrifying bacteria, the heterotrophic bacteria extracting soluble BOD from the water column, and the heterotrophic bacteria responsible for the breakdown of solids (sludge) held in the filter. The apparent oxygen consumption rate of the nitrifying bacteria (OCN in g O_2/m^3-day or g O_2/ft^3-day) can be computed directly from the volumetric conversion rates for nitrification using Equation 6 since we can estimate the amount of oxygen required for nitrification from chemical equations:

$$OCN = (3.47\ VTR + 1.09\ VNR)\ 0.97 \quad (6)$$

The factor 0.97 (unitless) corrects for oxygen assimilation during bacterial growth. The volumetric oxygen consumption rate that can be attributed to heterotrophic activity (OCH in g O_2/m^3-day or g O_2/ft^3-day) can then be calculated by difference (Equation 7):

$$OCH = OCF - OCN \quad (7)$$

The ratio of OCN to OCF expressed as a percentage is a valuable indicator of the efficiency of a backwashing protocol. A high OCN percentage (>50 percent) indicates that the nitrifying population is relatively high, i.e. the heterotrophic bacterial population has been successfully controlled without excessive loss of the nitrifying population. The OCN percentage tends to drop under lightly loaded regimes as the backwashing interval is extended allowing for more complete digestion of accumulated sludges. The nitrification capacity, however, is not adversely impacted as substrate (TAN) availability, not biofilm diffusion characteristics, limit the conversion process.

Table 3 presents operational ranges for selected performance parameters that can be expected for the three loading regimes, following the management guidelines suggested in Table 1. These data must be interpreted carefully. The range of values collected reflects changes in organic loading regimes as impacted by backwash frequency and the highly variable *in situ* nitrification. In most cases, the filters were backwashed only as often as required for the water quality objectives for TAN and nitrite to be met. Thus, a lightly loaded broodstock filter can display an OCF as high as a growout filter that is generally washed frequently. VTR and VNR values are generally held below the filter's peak capabilities by competition from bacteria growing on the tank and pipe walls. These nitrifying bacteria populations tend to flourish, benefiting from the low BOD found in the water column. Optimum filter performance generally occurs when the OCN percentage for a given loading regime is highest. Monitoring the OCN percentages while varying the backwashing frequency will generally allow the best backwash frequency to be quickly identified.

Table 3. Typical values for the performance parameters under conditions derived from operational filters. Values derived principally from Wimberly (1990) and Sastry (1996).

Performance Parameter	Units	**Typical Operational Value**		
		Broodstock	Ornamental	Growout
VTR	g TAN/m^3-day	35-105	70-180	140-350
	g TAN/ft^3-day	1-3	2-5	4-10
VNR	g N/m^3-day	35-105	70-180	140-350
	g N/ft^3-day	1-3	2-5	4-10
OCF	kg O_2/m^3-day	0.7-2.5	1.4-2.5	2.5-3.0
	g O_2/ft^3-day	20-70	40-70	70-85
OCN/OCF	%	25-35	25-35	45-55
OCH/OCF	%	65-75	65-75	45-55

Discussion

In this paper, we have summarized research findings and empirical evidence based upon technology that has remained essentially unchanged since the last summary of capabilities of bead bioclarifiers in Malone et al. (1993). The technology has moved from the research laboratory into the commercial sector where floating bead filters are

enjoying a reasonable degree of acceptance as bioclarifiers (Lutz 1997). Hundreds of commercial scale units are now being used. The authors have limited this paper to a description of the performance characteristics typical of these commercial units, although we are forced to substantiate many of the statements with research results since commercial scale performance is often poorly documented. However, at least three aspects of ongoing research are worthy of note here, as these findings will have significant impact on commercial use and performance patterns in the upcoming decade.

First, the nitrification performance plateau described in this paper reflects recognition of: 1) water quality conditions that facilitate high rate nitrification, 2) establishment of basic operational strategies with respect to flow rates and backwashing, and 3) the use of 3-5 mm spherical polyethylene beads. The primary objective of ongoing research efforts is to design beads to improve our ability to hold substantial amounts of nitrifying bacteria, while providing protection for biofilm during backwashing. Additionally, the increased porosity provides more volume to store bacteria postponing the loss of hydraulic conductivity that impairs the bed's performance as oxygen transport drops. This strategy appears to be successful with ongoing experimental units displaying net conversion rates approaching double those at equivalent loading previously documented (Beecher et al. 1997).

Secondly, the recent computer model analysis (Golz 1997; Golz et al. 1996) has suggested the use of a "high frequency" backwashing strategy, which has proved itself valuable at least for "gently washed" filters. This strategy recognizes that eventually, as organic (feed) loading rates are increased in response to improved nitrification performance, accumulation of solids will adversely impact the filters. In the extreme, optimization of nitrification will dictate a minimization of solids accumulation within the filter. This can be accomplished with the modified beads whose high degree of biofilm protection permits backwash intervals of only an hour or two. At this high frequency the residual free (non-biofilm) solids in the filter drop to such a low level that all the secondary impacts of solids accumulated from the clarification function are eliminated. The heterotrophic food supply is eliminated, freeing a larger portion of the biofilm for the nitrifiers. Figure 7 illustrates the impact of high frequency backwashing on the peak performance (to date) of an experimental bubble-washed unit. Not only did this unit substantially raise the peak performance plateau established by earlier floating bead filter researchers (Sastry 1996; Chitta 1995; Wimberly 1990), it also entered the volumetric conversion realm of fluidized beds (Thomasson 1991) while functioning as a bioclarifier. The usefulness of the strategy is currently being tested with the aggressive propeller washed filters while new self washing pneumatic hull designs are being refined to eliminate the need for automation of the backwashing sequence.

Finally, the increased porosity of the modified media in combination with the high frequency washing strategy has permitted the operation of floating bead bioclarifiers with airlift pumps (DeLosReyes et al. 1997). Both experimental and commercial prototypes have been tested and shown to be capable of delivering high flow rates (30-37.8 Lpm (8-

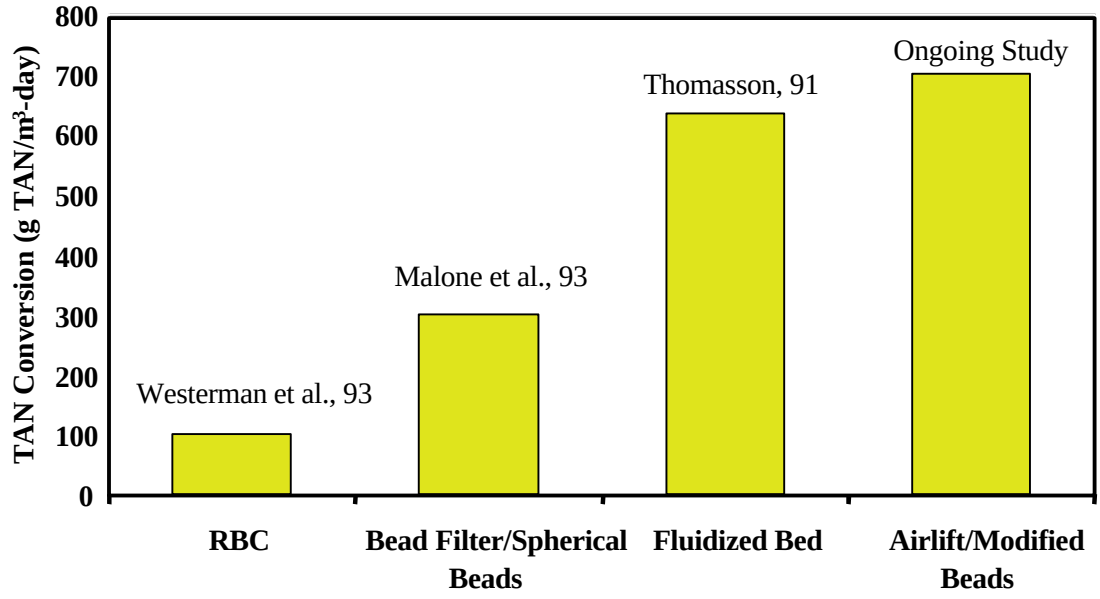

Figure 7. Volumetric TAN conversion rates of selected biofilters. Use of modified media and high frequency backwashing has shown promising results in ongoing experimental work on airlift bead bioclarifiers.

10 gpm)) with total headloss of about 30 cm (12 inches). Ongoing research in this area is specifically directed at the development of commercial scale prototypes.

Summary

Recognizing that the prime factor inhibiting widespread adoption of recirculating technologies is economics, the authors have advocated the use of floating bead filters as "bioclarifiers" (providing both solids capture and biofiltration) to permit the simplification of recirculating systems. Experimental and commercial use of the units has allowed extensive evaluation and establishment of reliable warmwater sizing guidelines for broodstock, ornamental, and growout conditions. The nitrification capabilities are controlled to a large extent by management. In this regard, particular emphasis is placed on pH and alkalinity control, oxygen delivery through adequate flow, and appropriate backwashing. The floating bead bioclarifiers using spherical beads have performance plateau around 350-450 g TAN/m^3-day (10–13 g TAN/ft^3 -day). Ongoing research into the use of modified beads, high frequency backwashing, and airlift recirculation indicate that dramatic changes in both the appearance and performance levels of floating bead bioclarifiers can be expected in the next few years.

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References

Ahmed, H. U. 1996. *The Effects of Fluxrate and Solids Accumulation on Small Size Particle Accumulation in Expandable Granular Bead Filters*. Master's Thesis, Louisiana State University, Baton Rouge, Louisiana, 74 pp.

Allain, P. A. 1988. *Ion Shifts and pH Management in High Density Shedding Systems for Blue Crabs (Callinectus sapidus) and Red Swamp Crawfish (Procambarus clarkii)*. Master's Thesis, Louisiana State University, Baton Rouge, Louisiana, 77 pp.

Beecher, L. E., A. A. DeLosReyes Jr., and D. G. Drennan, and R. F. Malone. 1997. Alternative Media for Enhanced Nitrification Rates in Propeller-Washed Bead Filters. *In*: Timmons M. B. and T. Losordo (Eds.). *Advances in Aquacultural Engineering*. Aquacultural Engineering Society Proceedings III, ISTA IV, Orlando, FL, November 9-12, 1997. NRAES, New York, pp. 263-275.

Bovendeur, J., A. B. Zwaga, B. G. J. Lobee, and J. H. Blom. 1990. Fixed-biofilm Reactors in Aquacultural Water Recycle Systems: Effect of Organic Matter Elimination on Nitrification Kinetics. *Water Research* 24(2) 207-213.

Chitta, B. S. 1993. *Effects of Backwash Frequency on the Nitrification in Plastic Bead Media Biofilters Used in Recirculating Finfish Culture Systems*. Master's Thesis, Louisiana State University, Baton Rouge, Louisiana, 129 pp.

Colt, J. E. and G. Bouck. 1984. Design of Packed Columns for Degassing. *Aquacultural Engineering* 3: 53-75.

Cooley, P. E. 1979. *Nitrification of Fish-Hatchery Reuse Water Utilizing Low-Density Polyethylene Beads as a Fixed-Film Media Type*. M. S. Thesis, University of Idaho, Moscow, Idaho, 53 pp.

DeLosReyes, A. A. Jr. 1995. *Solid Removal Schemes and the Rotating Biological Contactor in Recirculating Fish Culture Systems*. Ph.D. Dissertation, Louisiana State University, Baton Rouge, Louisiana, 193 pp.

DeLosReyes, A. A. Jr. and T. B. Lawson. 1996. Combination of a Bead Filter And Rotating Biological Contactor In A Recirculating Fish Culture System. *Aquacultural Engineering* 15(1): 27-39.

DeLosReyes, A. A. Jr., L. E. Beecher, and R. F. Malone. 1997. Design Rationale for an Air-Driven Recirculating Systems Employing a Bubble-Washed Bead Filter. *In*: Timmons M. B. and T. Losordo (Eds.). *Advances in Aquacultural Engineering*. Aquacultural Engineering Society Proceedings III, ISTA IV, Orlando, FL, November 9-12, 1997. NRAES, New York, pp. 287-294.

Drennan, D., W. Golz, Helal Ahmed, and R. F. Malone. 1995. Clarification Abilities of Floating Bead Filters Used in Recirculating Aquaculture Systems. *In*: *Aquaculture Engineering and Waste Management*, Proceedings from the Aquaculture Exposition VIII and Aquaculture Mid-Atlantic Conference, Washington, D.C., June 24-28, pp. 256-267.

EPA. 1975. Process Design Manual for Nitrogen Control. Technology Transfer Publication, U. S. Environmental Protection Agency.

Figueroa, L. A. and J. Silverstein. 1992. The Effect of Particulate Organic Matter on Biofilm Nitrification. *Water Environ. Res*. 64:728-733.

Golz, W. 1997. *A Theoretical Model of Nitrification in Floating-Bead Filters*. Master's Thesis, Louisiana State University, Baton Rouge Louisiana, 72 pp.

Golz W., K. A. Rusch, and R. F. Malone. 1996. Developing Backwash Protocols for Floating Bead Filters: A Model of Solids-Loading and Biofilm-Retention Effects on Nitrification. In: Libey, G. S. and M. B. Timmons (Eds.). *Successes and Failures in Commercial Recirculating Aquaculture.* Aquacultural Engineering Society Proceedings II, NRAES, New York, Volume 1, pp.196-205.

Grace, G. R. and R. H. Piedrahita. 1994. Carbon Dioxide Control. *In:* Timmons M. B. and T. M. Losordo (Eds.). *Aquaculture Water Reuse Systems: Engineering Design and Management*. Elsevier Science Publishing Company, Amsterdam, The Netherlands, pp. 209-234

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

Hovanec, T. A. and E. F. DeLong. 1996. Comparative Analysis of Nitrifying Bacteria Associated with Freshwater and Marine Aquaria. *Applied and Environmental Microbiology* 62(8): 2888-2896.

Hovanec, T. A., L. T. Taylor, A. Blakis, and E. F. DeLong. 1998. Nitrospira-Like Bacteria Associated with Nitrite Oxidation in Freshwater Aquaria. *Applied and Environmental Microbiology* 64(1): 258-264.

Huguenin, J. E., and J. Colt. 1989. *Design and Operating Guide for Aquaculture Seawater Systems.* Elsevier, New York.

Loyless, J. C. and R. F. Malone. 1997. A Sodium Bicarbonate Dosing Methodology for pH Management in Freshwater-Recirculating Aquaculture Systems. *The Progressive Fish-Culturist* 59:198-205.

Loyless, J. C. 1995. *A Feasibility Study of Using Air-lift Pumps for Aeration, Degasification, and Water Movement in a Recirculating Aquaculture System*. Master's Thesis, Louisiana State University, Baton Rouge, Louisiana, 116 pp.

Lutz , C. G. 1997. Greenhouse Tilapia Production in Louisiana. *Aquaculture Magazine* 23(6): 28-29.

Malone, R. F. 1995. Floating Media Biofilter, U.S. Patent No. 5,445,740. August 29, 1995.

Malone, R. F. 1993. Floating Media Hourglass Biofilter, U.S. Patent No. 5,232,586, August 3, 1993.

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

Malone, R. F. and A. A. DeLosReyes, Jr. 1997. Categories of Recirculating Aquaculture Systems. *In*: M. B. Timmons and T. Losordo (Eds.). *Advances in Aquacultural Engineering*. Aquacultural Engineering Society Proceedings III, ISTA IV, Orlando, FL, November 9-12, NRAES New York, pp. 197-208.

Malone, R. F., K. A. Rusch, and D. G. Burden. 1990. Kemp's Ridley Sea Turtle Waste Characterization Study: Precursor to a Recirculating Holding System Design. *Journal of the World Aquaculture Society* 21(2): 137-144.

Malone, R. F., B. S. Chitta, and D. G. Drennan. 1993. Optimizing Nitrification in Bead Filters for Warmwater Recirculating Aquaculture Systems. *In:* Wang, J. K. (Ed.). *Techniques for Modern Aquaculture*. American Society of Agricultural Engineers, Michigan, pp. 315-325.

Manthe, D. P., and R. F. Malone. 1987. Chemical Addition for Accelerated Biological Filter Acclimation in Closed Blue Crab Shedding Systems. *Aquacultural Engineering* 6: 227-236.

Manthe, D. P., R. F. Malone, and S. Kumar. 1988. Submerged Rock Filter Evaluation Using an Oxygen Consumption Criterion for Closed Recirculating System. *Aquacultural Engineering* 7: 97-111.

Matsuda, A. T. Ide, & S. Fuji. 1988. Behavior of Nitrogen and Phosphorus During Batch Aerobic Digestion of Waste Activated Sludge - Continuous Aeration and Intermittent Aeration by Control of DO. *Water Research* 22:1495-1501

Mia, R. I. 1996. *In Situ Nitrification in High Density Recirculating Aquaculture Production Systems*. Master's Thesis, Louisiana State University, Baton Rouge, Louisiana, 90 pp.

MP&L. 1991. Unpublished data from the Mississippi Power and Light's Demonstration Facility, Greenville, Mississippi.

Paz, J. D. 1984. *The Effects of Borderline Alkalinity on Nitrification in Natural Water Systems.* Ph.D. Dissertation, Polytechnic Institute of New York, New York, 125 pp.

Sastry, B. N. 1996. *A Comparison of Nitrification Capacity in Bead and Tubular Plastic Media*. Master's Thesis, Louisiana State University, Baton Rouge, Louisiana, 123 pp.

Siegrist, H., and W. Gujer. 1987. Demonstration of Mass Transfer and pH Effects in a Nitrifying Biofilm. *Water Research* 21(12): 1481-1487.

Thomasson, M. P. 1991. *Nitrification in Fluidized Bed Sand Filters for Use in Recirculating Aquaculture Systems*. Master's Thesis, Louisiana State University, Baton Rouge, Louisiana, 122 pp.

Westerman, P. W., T. M. Losordo, and M. L. Wildhaber. 1993. Evaluation of Various Biofilters in an Intensive Recirculating Fish Production Facility. *In:* Wang, J. K. (Ed.). *Techniques for Modern Aquaculture*. American Society of Agricultural Engineers, Michigan, pp. 326-334.

Wimberly, D. M. 1990. *Development and Evaluation of a Low-Density Media Biofiltration Unit for Use in Recirculating Fish Culture Systems*. Master's Thesis, Louisiana State University, Baton Rouge, Louisiana, 162 pp.

Zhang, T. C., Y. C. Fu, & P. L. Bishop 1995. Competition for Substrate and Space in Biofilms. *Water Environmental Resources* 67:992-1003.

Application of Fluidized-Sand Biofilters to Aquaculture

Michael B. Timmons, Professor
Agricultural and Biological Engineering
Cornell University
Ithaca, NY 14853

Steven T. Summerfelt, Research Engineer
Conservation Fund Freshwater Institute
PO Box 1746
Shepherdstown, WV 25443

Introduction

There is considerable debate as to the most appropriate biological filter technology for intensive aquaculture applications. The four major filter types used are: fluidized sand beds, trickling filters, rotating biological contactors, and floating bead filters. This paper focuses on the authors' experiences applying fluidized-sand biofilters (FSB) to indoor fish systems in both cool and warm water environments.

All biological filters are designed for the same function of oxidizing ammonia and nitrite to the fully oxidized form of nitrate. This phenomena is well described by Timmons and Losordo (1994). For simplicity, the production of total ammonia nitrogen (TAN) may be assumed to be 3% by weight of the fish feed being fed per day. So, if 100 kg of feed are fed per day to a particular tank system, then there will be approximately 3 kg per day of TAN produced. The biological filter must be designed to fully oxidize this TAN production, else the TAN concentrations in the tank will rise above design levels.

Given a defined rate of TAN production, a suitable biological filter must be designed. From a practical perspective biofilter selection is less critical in small production systems, i.e., systems that feed at rates below 50 kg of feed per day, than for larger farming systems. In small systems, biofilters can be over-designed and the added cost is generally not of critical importance to the overall economic success of the venture. Smaller operations are not competing in the wholesale market, but in niche markets and providing service or other product attributes that allows premium pricing to the seller/grower.

Conversely, for large production systems feeding 100 kg or more per day into a single fish culture tank system, economic designs for the biological filter become much more critical. We feel that the primary benefit of the FSB system is the ability to scale the biofilter to the system needs without paying large economic penalties as the FSB is scaled to an appropriate size. It is conceivable that a several hundred ton per yr facility might be operated on one to three biofilters for the entire facility. Once designed properly, in terms of sand size and degree of bed fluidization, the cross sectional area of the FSB system can be increased until the necessary volume of sand is obtained to oxidize the TAN load that is imposed.

General Design Approach

FSB can be designed following the steps listed below:

1) determine the TAN load
2) determine the sand volume required to match TAN load
3) select the design depth of sand bed
4) select the sand size in relation to flow rate available or desired
5) design the water delivery system

TAN Load

As previously discussed, the TAN production may be assumed to be 3% of the daily fish feeding rate:

$$\text{TAN (kg/day)} = 0.03 \times F \qquad (1)$$

where F = fish feeding rate per day, kg/d

Sand Volume and Nitrification Rates

Nitrification rates and efficiencies within FSB's have been measured as a function of sand size (Table 1) using replicated 10 cm (Cornell University) and 15 cm (Freshwater Institue) FSB reactors. The sands were fluidized at fixed velocities that were set to achieve bed expansions of 50% with clean sand (e.g., a bed that is 1 meter in static depth would be 1.5 meters in depth when expanded). At Cornell University, inlet concentrations were 0.6 to 0.7 mg/L TAN, temperature 26 C, pH 7.3, 6 to 7 mg/L DO, and TSS < 10 mg/L. At the Freshwater Institute, inlet concentrations were 0.5 to 0.6 mg/L TAN, temperature 15 C, pH 7.3 to 7.5, 10 to 11 mg/L DO, nitrite 0.04 to 0.06 mg/L, and TSS < 10 mg/L (Tsukuda et al., 1997).

Note that the TAN removal rates in Table 1 are expressed on a unit volume basis, instead of on a surface area basis. Low density medias such as RBC's and trickling towers provide nitrification rates proportionate to surface area provided by the media, but research suggests that nitrification rates in granular medias are much more closely related to volume of media than surface area provided by the media. The large surface area provided by small sands provides no advantage in terms of nitrification rate.

The fine sand biofilters, however, have demonstrated a much higher TAN removal efficiency (90% removal) when compared to the large sand biofilters (10-17%). The high removal efficiencies found with the finer sands are in part due to the low velocities required to fluidize fine sands (see section 2.3 below). The low velocities require larger FSB and longer hydraulic retention times than would a biofilter treating the same flow but with a larger sand.

Fine sands are not recommended for use in warm-water systems, however, due to possible difficulties controlling excessive biosolids growth. Control of biosolids growth is still an

issue when fine sands are used in cold-water applications, but biosolids growth is manageable at the colder temperatures.

The results (Table 1) show that FSB biofilters used at warmer temperatures (i.e., 2.7 kg/d/m^3 @ 26°C) remove ammonia at a much higher rates than at cooler temperatures (i.e., 0.5-1.5 kg/d/m^3 @ 15°C). Based upon these numbers, it seems safe to use a nitrification rate of 1.0 kg TAN/day/m^3 static bed as a design value for warm water systems and a rate of 0.7 kg TAN/day/m^3 clean static sand for cold water systems (15°C).

The required sand volume can be calculated given the TAN load and the TAN removal rates per unit volume of sand.

Table 1. Average TAN removal rates and efficiencies measured across cold-water (15°C) and warm-water (26°C) biofilters as a function of sand size.

Sand retained between sieve mesh sizes	40/70	20/40	18/30
Cold-Water (15°C) Systems (Tsukuda et al., 1997)			
TAN removal rate, kg/d/m^3 clean static bed	1.5	0.51	0.51
TAN removal rate, kg/d/m^3 expanded bed	0.41	0.35	0.35
TAN removal efficiency, % each pass	90	10	10
Warm-Water (25°C) Systems (M. B. Timmons, unpublished data)			
TAN removal rate, kg/d/m^3 clean static bed	NR	2.7	2.7
TAN removal rate, kg/d/m^3 expanded bed	NR	2.7	2.7
TAN removal efficiency, % each pass	NR	17	17

NR = not recommended due to excessive biofilm growth

Design Depth of Sand Bed

Choice of sand depth is related primarily to the physical constraints of the building ceiling height, whether the sand filter can be partially submerged into the ground, and whether any additional elevation is required to gravity flow back to the culture tank through aeration and/or oxygenation unit processes. In in any case, the design should produce as much sand volume as practically possible. The larger the sand volume and therefore lower TAN load per unit volume of sand -- provides a factor of safety for the overall design. It also minimizes the biological film growth per unit particle of sand, which will minimize sand bed growth and management problems related to changing expanded sand bed depths.

When fine sands are used, growth of biofilm can increase the expanded bed depth to the point that sand will be flushed from the reactor into the rest of the system. Therefore, fine sand biofilters are usually designed for eventual expansion of the biofilm-coated sand bed to achieve 200 to 300% of initial static sand depth (e.g., if a clean sand depth is 1 m before fluidization, after 50% fluidization the clean sand will be 1.5 m, once a thick biofilm grows in this bed the total sand expansion may be 200%, or around 3 m of total depth).

Sand Size in Relation to Flow Rate Available

Selecting the sand size relates to the previous discussion on sand depth. The overall design for the fish system will include calculating flow rates to maintain target levels for the various water quality parameters, primarily oxygen, ammonia, and carbon dioxide. Depending upon the type of oxygenation and CO_2 stripping units being used, large variations in required design flow rates can result. Thus, often the design of the different components of the overall system is manipulated until the flow rates for the different water quality parameters being controlled by the different system components are somewhat in balance.

Selecting the sand size and bed expansion (see section 2.3.1) determines the velocity through the biofilter. Selection of a finer sand will requires lower water velocities than selection of a larger sand. Because of the different water velocity requirements, at a given flow rate through the biofilter, the finer sands will result in a bigger biofilter than if a larger sand had been selected.

Suppliers of graded filter sands usually report the effective size (D_{10}) and uniformity coefficient (UC) of their sand. Fish farmers purchase sand for FSB's by specifying either an effective size or some range of sieve sizes (Table 2), in addition to an acceptable uniformity coefficient for the sand (where smaller UC's result in sands with less variation in particle diameter). Specifying sand that is sized through a 20/40 mesh means that the largest sand passes a 20 mesh sieve (D_{eq} = 0.841 mm) and the smallest sands are retained on a 40 mesh sieve and are larger than 0.42 mm. There is always some small percentage of sands that can be considered "dust" but this should be just 1 to 3% of the total mass of

Table 2. The opening size of each U.S. sieve series designation number (as reported by Perry and Chilton, 1973).

Sieve Designation Number†	Size of Opening, (mm)	Sieve Designation Number†	Size of Opening, (mm)
4	4.76	35	0.500
5	4.00	40	0.420
6	3.36	45	0.354
7	2.83	50	0.297
8	2.38	60	0.250
10	2.00	70	0.210
12	1.68	80	0.177
14	1.41	100	0.149
16	1.19	120	0.125
18	1.00	140	0.105
20	0.841	170	0.088
25	0.707	200	0.074
30	0.595	230	0.063

† Number of meshes per inch.

sand. Upon start-up of a new FSB, the bed will flush the small sand from the system for several days. Several culture tank flushings are required to clear the fines from the system, which should be accomplished before fish are added to the system.

The "effective size" (D_{10}) is defined as the opening size which will pass only the smallest 10%, by weight, of the granular sample. The D_{10} provides an estimate of the smallest sand in the sample and is the size used to estimate the maximum expansion at a given water velocity (AWWA, 1989).

The "uniformity coefficient" (UC) is a quantitative measure of the variation in particle size of a given media and is defined as the ratio of D_{60} to D_{10}.

The D_{90} is the sieve size for which 90% of the grains by weight are smaller. The D_{90} provides an estimate of the largest sand in the sample and is the value used during design to calculate the water velocity required to fluidize even the largest sand to some minimal expansion, e.g. 20%. One should be sure to check that the design fluidization values meets this minimum requirement. The D_{90} can be estimated from the effective size (D_{10}) and the uniformity coefficient (Cleasby, 1990):

$$D_{90} = D_{10} \cdot \left(10^{1.67 \cdot \log (UC)}\right) \tag{2}$$

The "mean size" (D_{50}) is the sieve size for which approximately 50% of the grains by weight are smaller. The D_{50} provides an estimate of the average size of the sand in the sample and is the value used during design to estimate the average bed expansion at a given superficial velocity. Using the uniformity coefficient of the sand, you can approximate the mean sand size (J. Cleasby, Iowa State University, pers. comm.):

$$D_{50} = D_{10} \cdot \left(10^{0.83 \cdot \log (UC)}\right) \tag{3}$$

If desired, the specific surface area of the static sand bed (S_b) can be approximated from the static bed void fraction (ε 0.45) and the sphericity of the sand (Ψ 0.75):

$$S_b = \frac{6 \cdot (1 - \varepsilon)}{\Psi \cdot D_{50}} \tag{4}$$

Expansion

A failure of the FSB to properly expand can result in severe problems. Under-fluidization of the FSB will result in bed channeling with water leaving the biofilter untreated. Degrees of poor fluidization will result in the larger sands moving to the bottom of the bed and becoming static. Such areas then are apt to become anaerobic or anoxic resulting in denitrification and other undesirable water chemistry changes, e.g. sulfide gas production.

Our recommendation is to choose an overall clean sand expansion of around 50% when designing the FSB system. Sand expansion is a function of water temperature and of

several sand characteristics (Summerfelt and Cleasby, 1996). Water velocity requirements increase with increasing expansion and increasing sand size (Table 3).

The influence of water temperature on viscosity, as well as variations in sand characteristics from different quarries, can create some error in estimating expansion velocity requirements for a given sand (illustrated in Table 3). Therefore, the numbers in Table 2 and as reported by Summerfelt and Cleasby (1996) should only be used for preliminary design estimates; a hydraulic test on a sample of the sand selected should be completed to determine the actual expansion velocities on a case by case basis. Construct hydraulic testing columns with care given to how the flow is distributed and use at least 1 m sand depth. Our experience with these test columns has shown that their flow distribution mechanism has a large impact on fluidization velocities in small columns (10-15 cm diameter) using shallow sand depths.

Table 3. Water velocities required to expand sand 20, 50, 100, and 150 percent with effective sizes (D_{10}) of 0.24, 0.45, 0.60, and 0.80 mm. Two velocities are reported in each cell, #1/#2: the first velocity is an average of measurements made during fluidization tests in 10 cm diameter columns where flow was distributed under a mesh screen used to support the sand bed (tests were at 15°C); the second velocity is the velocity that was predicted for the mean sand size (D_{50}), taken from the table shown in Summerfelt (1996).

	Sands Tested			
Retaining sieve mesh sizes	40/70	30/50	20/40	18/30
Effective size (i.e., D_{10}), mm	0.24	0.45	0.60	0.80
Uniformity coefficient	1.8	1.4	1.4	1.3
D_{50}, mm	0.37	0.59	0.79	0.99
Velocity requirements, cm/s				
20% expansion	0.5/0.4	0.7/0.9	0.8/1.4	1.3/1.9
50% expansion	1.0/0.8	1.3/1.5	1.9/2.2	2.7/2.9
100% expansion	1.4/1.4	2.0/2.4	3.1/3.3	4.6/4.2
150% expansion	1.9/1.9	2.7/3.1	4.1/4.2	5.9/5.2

Biosolids Management

The down side of the smaller sand sizes is flocculation growth in the FSB bed, especially as water temperature increases (Thomasson, 1991; Monoghan et al., 1996; Timmons, unpublished data). Biofilm growth on sand decreases this particles' effective density, which causes these particles to migrate toward the top of the biofilter column and increases the total biofilter expansion. Heterotrophic growth continuously occurs and also tends to trap sand particles in its growth. This biological growth plus the dead bacteria from biofilms also migrates to the top of the sand column. The smaller sands can be trapped in this flocculant material and remain at the top of the sand column. When the biosolid-coated sands remain at the top of the column, the shearing of biofilm material from the sand particles that occurs at the bottom of the bed in the vicinity of the orifice on the horizontal laterals does not occur. If the FSB reactor vessel has translucent walls, one can easily observe the interface between the fluidized sand and the flocculant layer. Using small sands, $D_{10} < 0.42$ mm (sieve size #40), the growth of the flocculant layer in warm

water systems can grow out of control unless actively managed. The FSB operator must have a regular routine of removing the flocculant layer, else the whole sand bed could become engulfed. Removing the flocculant layer will also generally require replacement of sand, especially when fine sands are used. Replacement of sand has an inherent disadvantage beyond the obvious problems (sand cost and labor) in that the fine material with the new sand will foul the water column. Depending upon the fish species and the rigor of the required water quality, pre-flushing of the new sand may be required.

Cornell University research experience using sands that have $D_{10} > 0.42$ under warm water conditions has been that the beds have no appreciable collection of flocculant material at the top of the FSB. This is a large advantage in terms of simplifying management. The disadvantage is that the larger sands require higher fluidization velocities and this reduces the size of the biofilter.

For nearly 10 years, the Freshwater Institute has consistently managed fine sand (D_{10} = 0.20-0.25 mm) FSB's in several recirculating rainbow trout culture systems. The FSB's operated reliably during this period, with TAN removal efficiencies typically ranging from 70-90%. Biosolids growth in the FSB's was usually controlled by siphoning biosolids from the top of FSB's as their beds reach a maximum depth, and then replacing lost sand as needed (Bullock et al., 1993; Heinen et al., 1996; Summerfelt and Cleasby, 1996; Tsukuda et al., 1997). The Freshwater Institute has also controlled biofilm thickness by shearing the biofilm in-vessel, using a pump to transport the flocculant particles from the top of the biofilter to the bottom of the bed, where shear forces are greatest (Figure 2). The biofilm stripper effectively maintained bed expansion at a fixed level and reduced biofilm thickness without trading-out sand (unpublished data).

In the Freshwater Institute's cold-water recycle system, Tsukuda et al. (1997) found that biosolids did not accumulate within expanded beds using sands with effective sizes of 0.60 and 0.80 mm. However, biosolids did sometimes collect in a distinct layer above the expanded sand layer. Siphoning the biosolids layer was simple with these larger sands, because the expansion depth of these sands remained fairly constant and the biosolids could be removed relatively free from sand. Because the biosolids layer is expanded, it is also fluid, which greatly reduces sand loss and the need to replace old sand with new sand (when larger sands are used). Therefore, a siphon withdrawing flow and biosolids from one point in the biofilter can remove all fluidized biosolids at depths above this level (for all sand sizes).

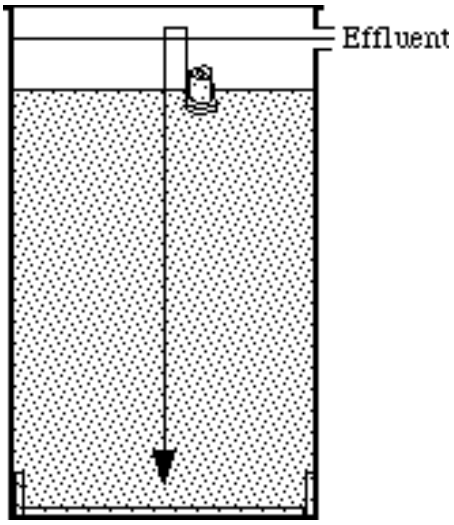


Figure 1. A small magenetic-drive submersible pump was able to withstand the abrasive effects of the sand being pumped and helped to strip biofilm and control bed depth (Freshwater Institute, unpublished data).

FSB Water Delivery System

Designing the FSB's water delivery system involves appropriate sizing of the plenums, pipes, and orifices to promote equal fluidization of the sand bed. Generally this involves using a collector plenum that receives water from the pumps and then assigning a given number of laterals to this plenum to provide an effective lateral grid across the bed floor. Maximum spacing of laterals center to center is in the 15 to 30 cm (6 to 12 inch) range. The basic idea is that the cross sectional area of the pipe "upstream" of a group of laterals or orifices should be 2 times the additive area of all the laterals or orifices. This results in the pressure gradient being dominated by the loss across the reduction section so that each reduced section (e.g. lateral or orifice) will see approximately the same pressure regardless of its location along the lateral (orifice case) or lateral attached to a collection plenum/manifold. This approach ensures that the flow will be approximately the same to each of the laterals from the manifold/plenum or from a lateral to each orifice.

We try to maintain the following area ratios:

$1.5 < \text{manifold/laterals} < 3$
$2.0 < \text{lateral/orifices} < 4$

As a rule of thumb, we have found that flow distribution manifolds designed according to these criteria have a total pressure requirement (in m or ft of water pressure) equal to three times the sand depth plus the height that the water must be lifted from the floor sump to the top of the biofilter. Therefore, total pumping head requirements range from 5 to 11 m (16 to 36 ft), depending mostly on the depth of the biofilter column. For precise pressure estimates, the hydraulic gradeline should be calculated across the piping from the pump sump to the water depth in the FSB column.

Summerfelt (1996) and Summerfelt et al. (1996) give a more detailed discussion on how to estimate pressure losses through the FSB system and how to size laterals and plenums/manifolds. An example of how a spreadsheet can be used to size the pipe manifold, pipe laterals, and distribution orifices is provided (Table 4).

Table 4. Example of the use of a spreadsheet to calculate the size of the FSB manifold, pipe laterals, and distribution orifices. Values that are in BOLD are user assigned numbers and (except for the expansion velocity) must be entered in a somewhat iterative procedure until the criteria in italics are met.

expansion velocity, cm/s	**2.6**	
expansion velocity, ft/min	5.1	
Flow Rate, gpm/ft^2	38.0	
Diameter of sand tank, inches	72	
Diameter of sand tank, ft	6.0	
Area of floor cross section, ft^2	28.3	
Total flow, gpm	1,074	
Lateral pipe diam, inches	**3.00**	
Lateral pipe area, sq inches	7.07	
Lateral Length, avg, feet	4.80	note: for round tank, avg L as 86% of tank D
Lateral spacing, inches	**12**	
Number of laterals (calc)	7.0	
Number of Laterals assigned	**6**	i.e. you have to have a unit number
Orifice diameter, inches	**0.563**	
Orifice area, in^2	0.248	
Orifice spacing (='s lat space) inches	12	
Number of orifices/lateral	11.6	assigned in pairs
Total # of orifices assigned per lateral	**12**	
Orifice area per lateral, in^2	2.99	
Total Orifice area, ft^2	0.124	
Pipe area : orifice area	2.4	*recommended between 2 to 4*
manifold area : lateral pipe area	**1.5**	*recommended area ratio is 1.5 to 3.0*
Min Req Manifold diameter, inch	9.0	(using one header pipe for all the laterals)
Min Req Manifold diameter, inch	6.4	(using two header pipes for all the laterals)
Orifice Coefficient (0.6 to 1.0)	0.8	
Velocity across orifice, ft/s	24.1	
Head loss across the orifices	9.0	*This number needs to be > sand depth*

Sizing of Orifices

Orifice size is selected to fluidize the bed by assuming the pressure loss across the orifice is greater or equal to the static sand depth:

$$C\ V^2/2g > \text{Depth} \quad (5)$$

where C is the orifice coefficient (decimal), g is the acceleration due to gravity (m^2/s), and Depth is the static sand depth (m). Orifice coefficient values are in the range of 0.6 to 0.8. While the choice of this orifice coefficient does not appear to be terribly significant, since the known value in the equation will be the flow rate, small changes in C can significantly affect the estimate of the pressure loss term. The severe negative consequence of using a C value that is too low is that the bed may not fluidize in all regions. We have used a C value of 0.7 in past designs, or used a C value of 0.6 and selected an orifice size to create a pressure loss 1.2-1.5 fold greater than the sand depth.

Design Example

Design a FSB for a warm water system using a 20/40 sized sand for a feeding rate of 100 kg feed/day and for a 20/40 sand.

TAN Load.

TAN Load = 100 kg/feed/day x 0.03 kg TAN/kg feed = 3 kg TAN/day

Volume Sand (V)

V= (3 kg TAN/day) / 1 kg TAN/day/m^3 = 3 m^3 sand required

Depth of Sand (h)

h = 2 m (unexpanded; placed on floor level; selected based upon ceiling height and elevation required to gravity flow back to culture tank through aeration column and/or oxygenation column)

Cross sectional Area of Bed (A)

A = V/h = 3 m^3/2 m = 1.5 m^2

Fluidization Velocity

For a 20/40 sand, Table 2 can be used to estimate D_{10} (40 mesh) = 0.42 mm
If the sand has a UC of 1.5, equations 2 and 3 can be used to estimate D_{90} and D_{50},

D_{90} = 0.42 mm $10^{1.67 \times \log 1.5}$ = 0.83 mm
D_{50} = 0.42 mm $10^{0.83 \times \log 1.5}$ = 0.59 mm

An estimate of the velocity required to expand the bed 50% can be looked up in Table 3: i.e., a 20/40 sand with D50 = 0.59 mm will fluidize 50% at around D_{10} = 1.3-1.9 cm/s. To check if the largest sands will expand, the velocity required to expand a 0.83 mm sand was looked up in Table 3 and was estimated to be anywhere from 1.0-2.2 cm/s. The design example will use 2.0 cm/s.

Flow Rate

Given the expansion velocity of 2.0 cm/s and the cross sectional area of the bed (1.5 m^2), the required flow rate is

$$Q = Vel \times A = 0.020 \text{ m/s} \times 1.5 \text{ m}^2 = 0.039 \text{ m}^3\text{/s} \times 60\text{s/min} = 1.8 \text{ m}^3\text{/min (474 gpm)}$$

From a mass balance on oxygen, the required flow rate (1.8 m^3/min) in this example would have to carry around 10 mg/L of available oxygen to meet the average respiration needs of the fish consuming 100 kg feed/day. Therefore, to achieve the available oxygen needs the flow would have to pass through an oxygenation unit to achieve on average a 15-17 mg/L of dissolved oxgyen before entering the culture tank.

In a cold-water application, sands with D_{10} of around 0.25 mm are selected because of the high nitrification efficiency that can be achieved. These biofilters are usually oversized, because the sand selection locks in a velocity of near 1.0 cm/s and the recirculating flowrate is controlled by the oxygen requirements of the fish.

Plugging Concerns

A major concern in operating FSB's is the plugging of the lateral systems with sand. This can happen if the check valves in line with the pumps and biofilter malfunction and do not close at the time pumps are shut down. Check valve failure should be rare, but if it ever does happen, water will siphon out of the biofilter to the pump sump (which is at a lower elevation) and carry sand into the pipe laterals until the pipes are plugged. To prevent this, top of the line swing check valves should be used to reduce the chance of failure. In the event of a failure, laterals can be unplugged, usually in a matter of hours, with the use of clean-outs on the pipe laterals at the top of the biofilter (Summerfelt et al., 1996). One must be prepared for plugging of the pipe laterals if the biofilter is to be restored to operating mode.

Currently, we have been successful in preventing this siphoning action by placing swing check valves above or below the pump (s). We have used heavy, brass or PVC swing check valves with a solid rubber swing flap or a brass swing flap and a well machined seat ledge. These valves are still working effectively after several years of operation. Others have used siphon breaks to let air into the manifold above the biofilter when pumps are shut down, which allows the water in the pipe manifold to drain into the biofilter and prevents siphoning. The Freshwater Institute has tried a siphon break such as this and found it to be undesirable, because the air filling the laterals and manifold is forced through the sand bed when pumping is restored. The large slug of air passing through the biofilter has caused significant volumes of sand and biosolids to dump out of the biofilter. This phenomena is probably much more of a problem with the finer sands and the associated mass of biosolids that accumulate in these biofilters. Similarly, any air leak into the pumping and lateral system will also promote sand loss from a bed and prevent proper operation, especially when smaller sands are used.

Some allowance should be made to prevent airlocks when check valves are employed. Simply, take a look at the system and if there is the potential for air lock to occur, make provisions to allow for air to escape from the pump. This is a typical problem if the

pumps are submerged and the collection sump is drained once a pump is turned off. When the sump area is refilled, the pipe above the pump will have trapped air and will prevent the pump from moving enough water to open the check valve.

Cost Basis

As stated in the beginning of this paper, our main argument for using FSB's is that their cost per unit of TAN treated is low compared to competing technologies. This is only true for larger systems. Cayuga Aqua Ventures, LLC (CAV) has recently installed 1.83 m diameter by 4.6 m high FSB systems using a static depth of sand of 2.14 m. Volume of sand in these systems is 5.61 m^3. Using the TAN nitrification rate of 1 kg TAN/day/m^3 indicates a nitrification rate of 5.61 kg TAN per day per FSB. Using equation to calculate the fish feeding load that could be imposed is 187 kg feed per day. These are rather impressive numbers. Performance at CAV lends credibility to these estimates to date. CAV uses a design value of 100 kg feed per day for these systems.

Costs provided by CAV for their FSB tank systems are approximately $2,200 for the fiberglass reactor vessel, $800 for a fiberglass plenum/manifold, plus the costs of pipe and valves. CAV estimates approximately 50 man hours to fully plumb a system to the fish tank unit.

Summerfelt and Wade (1997) report that two FSB biofilters built recently at two commercial fish farms treating flows of 1.5 to 2.3 m^3/min cost around $6000, including biofilter vessel, sand, piping, valves, shipping, and labor for installation. These prices were around five times less than the estimated cost for a installing a trickling filter of similar TAN removal capacity (Summerfelt and Wade, 1997).

References

American Water Works Association (AWWA). 1989. ANSI/AWWA B100-89. AWWA standard for filtering material. AWWA, Denver, Colo.

Bullock, G., J.A. Hankins, J. Heinen, C. Starliper, and J. Teska. 1993. *Qualitative and quantitative bacteriological studies on a fluidized sand biofilter used in a semiclosed trout culture system.* Biological Report 17, U.S. Fish and Wildlife Service, Washington, D.C.

Cleasby, J. L. 1990. Filtration. In: F. W. Pontius (ed). *Water Quality and Treatment,* American Water Works Association. Fourth Edition, McGraw-Hill Inc., New York, NY.

Heinen, J. M., J. A. Hankins, A. L. Weber, and B. J. Watten. 1996. A semiclosed recirculating-water system for high-density culture of rainbow trout. *The Progressive Fish-Culturist* 58:11-22.

Monaghan, T. J., Delos Reyes, A. A., Jeansonne, T. M., and Malone, R. F. 1996. Effects of media size on nitrification in fluidized sand filters. Page 110 In *Aquaculture America '96 Book of Abstracts.* World Aquaculture Society, Baton Rouge, Louisiana.

Perry, R. H. and C. H. Chilton. 1973. *Chemical Engineers Handbook*, Fifth Edition, McGraw-Hill Book Company, New York, New York.

Timmons, M.B. and T.S. Losordo. 1994. Aquaculture water reuse systems: engineering design and management. Elsevier Science, The Netherlands.

Summerfelt, S.T. 1996. Engineering design of a water reuse system. Pages 277-309 *in* R. C. Summerfelt, ed. *Walleye Culture Manual*. NRAC Culture Series 101. Central Regional Aquaculture Center Publication Center, Iowa State University, Ames, IA.

Summerfelt, S. T. and J.L. Cleasby. 1996. A review of hydraulics in fluidized-bed biological filters. ASAE Transactions 39:1161-1173.

Summerfelt, S.T., E.M. Wade. 1997. Recent advances in water treatment processes to intensify fish production in large recirculating systems. Pages 350-368 In: M. B. Timmons and T. M. Losordo (eds.) *Recent Advances in Aquacultural Engineering* (Proceedings). Northeast Regional Agricultural Engineering Service, Ithaca, NY

Summerfelt, S.T., J.A. Hankins, M.D. Durant, and J.N. Goldman. 1996. Removing obstructions: Modified pipe-lateral flow distribution mechanism reduces backflow in fluidized-sand biofilters. *Water Environment & Technology*, 8(11):39-43.

Thomasson, M. P. 1991. Nitrification in fluidized-bed sand filters for use in recirculating aquaculture systems. Master's thesis, Louisiana State University, Baton Rouge, Louisiana.

Tsukuda, S.M., J.A. Hankins, C.P. Marshall, S.T. Summerfelt, G.L. Bullock and T.K. Sawyer. 1997. Effects of sand size on fluidized-bed biofilter performance in cold water systems. Pages 368-380 In: M. B. Timmons and T. M. Losordo (eds.) *Recent Advances in Aquacultural Engineering* (Proceedings). Northeast Regional Agricultural Engineering
Service, Ithaca, NY.

Comparative Performance of Biofilm Reactor Types: Application of Steady-State Biofilm Kinetics

Barnaby J. Watten
U.S. Geological Survey
Leetown Science Center
1700 Leetown Road
Kearneysville, WV 25430

Michael B. Timmons and Brian J. Vinci
Cornell University
Agricultural and Biological Engineering
302 Riley-Robb Hall
Ithaca, NY 14853-5701

Steven T. Summerfelt
Freshwater Institute
P.O. Box 1746
Sheperdstown, West Virginia 25443

Ammonia toxicity in aquaculture is often circumvented through application of fixed biofilm reactors designed to encourage nitrification. Nitrification incorporates two consecutive reaction steps:

$$NH_3\text{-}N \xrightarrow[R_1]{} NO_2\text{-}N \xrightarrow[R_2]{} NO_3\text{-}N$$

Where R_1 represents the rate of formation of NO_2-N, and R_2 represents the rate of formation of the end product NO_3-N. Reactors used are classified hydraulically as plug-flow, plug-flow with dispersion, and mixed flow. Reactor type must be chosen carefully given its effect on treatment system performance as measured by cost, maintenance, required volume and changes in water chemistry. To characterize these effects and to demonstrate their design implications, we developed algorithms that predict performance given reactor type, inlet conditions and certain steady-state biofilm kinetic parameters. Model use focused on intermediate product formation (R_1) and conversion (R_2) given the sensitivity of fish to NO_2-N at relatively low concentrations. For example, our RBC and fluidized bed data base indicated R1 and R2 could be expressed as first order reactions. Hence

$$R_1 = d[NH_3\text{-}N]/dt = -k_1[NH_3\text{-}N]$$

$$R_2 = d[NO_2\text{-}N]/dt = -k_2[NO_2\text{-}N],$$

and the corresponding rate of change of intermediate product is:

$$R_1 - R_2 = k_1[NH_3\text{-}N] - k_2[NO_3\text{-}N]$$

where k_1 and k_2 are rate constants. Integration allows the rates above to be expressed in terms of the initial concentration of the reactants NH_3-N and NO_2-N; i.e. for plug flow reactors effluent concentrations are:

$$[NH_3 - N] = C_A e^{-k_1 t}$$

$$[NO_2 - N] = \frac{k_1 C_A}{k_2 - k_1}\left[e^{-k_1 t} - e^{-k_2 t}\right] + C_B e^{-k_2 t}$$

The end product of nitrification is then:

$$[NO_3 - N] = C_A\left[1 + \frac{k_1 e^{-k_2 t}}{k_2 - k_1} - \frac{k_2 e^{-k_1 t}}{k_2 - k_1}\right] + C_B\left(1 - e^{-k_2 t}\right) + Cc$$

Where C_A, C_B and C_c are inlet concentrations of NH_3-N, NO_2-N and NO_3-N, respectively; and t is water residence time in the reactor. Algorithms for the mixed flow reactor were established in a similar manner allowing reactor types to be compared with identical k_1, k_2, and t values. For example, Figure 1 shows NH_3-N removal in a plug flow reactor exceeds that provided by the mixed flow type but plug-flow operation results in higher effluent NO_2-N at a fixed NH_3-N removal efficiency (Figure 2). Figure 3 and 4 shows the net change in NO_2-N (plug-flow reactor) is related to the product k_1t, the inlet concentration ratio NO_2-N/NH_3-N, and the ratio k_2/k_1. We also applied a model based on the kinetics of steady-state biofilms to estimate the k_1 and k_2 coefficients needed in the conversion algorithyms. The biofilm model equates the energy derived from substrate utilization and the energy required for cell maintenance which includes the effect of shear stress on biomass loss. The model provides an estimate of biofilm thickness and substrate fluxes when substrate concentrations exceed a calculated minimum concentration required to maintain a biofilm. The biofilm model can be used to quantify the effects of changes in media density on fluidized bed reactor performance.

Figure 1.

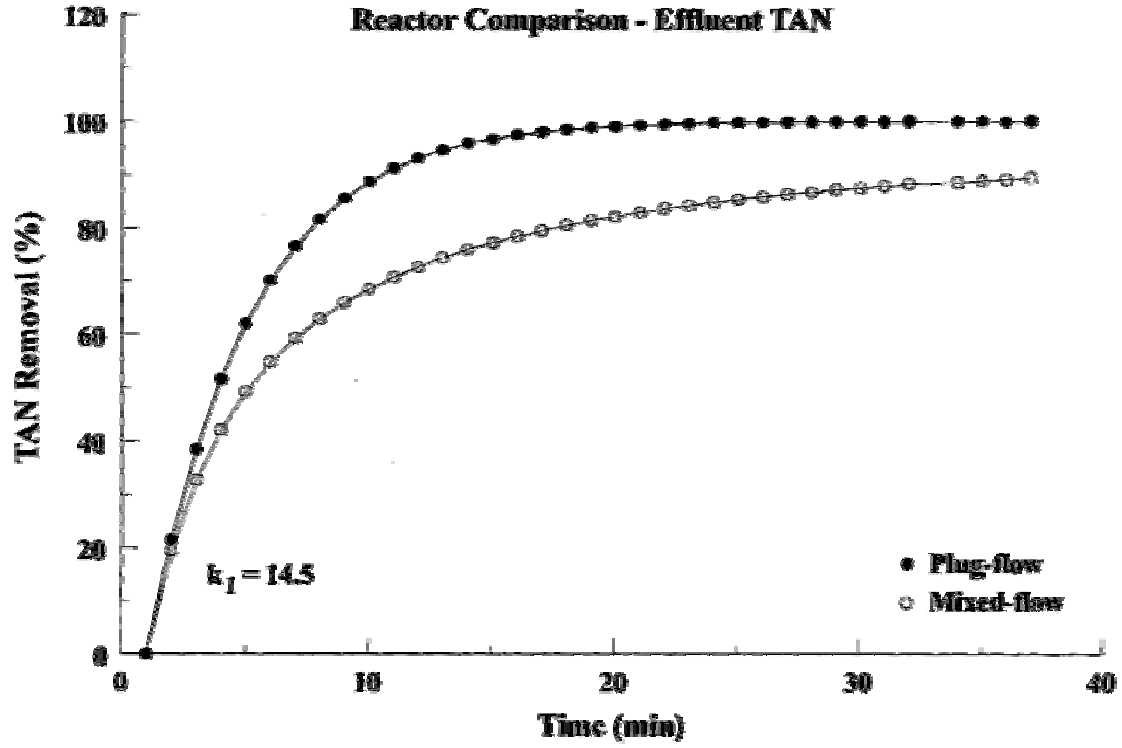


Figure 2.

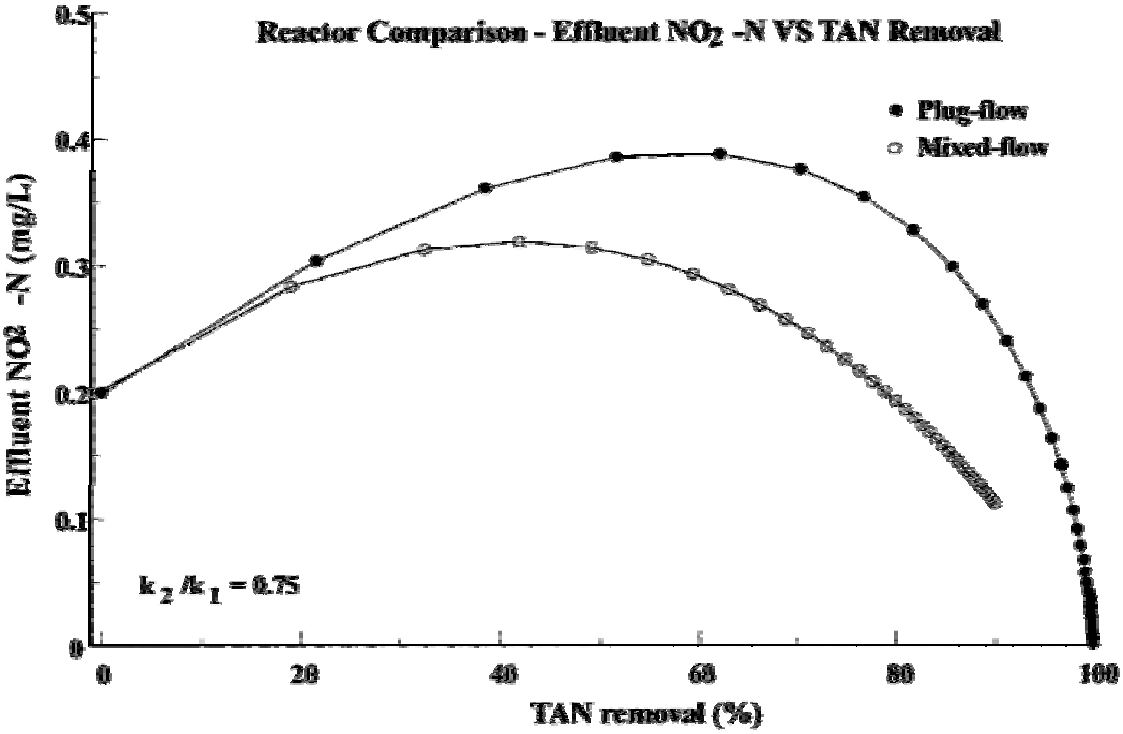


Figure 3.

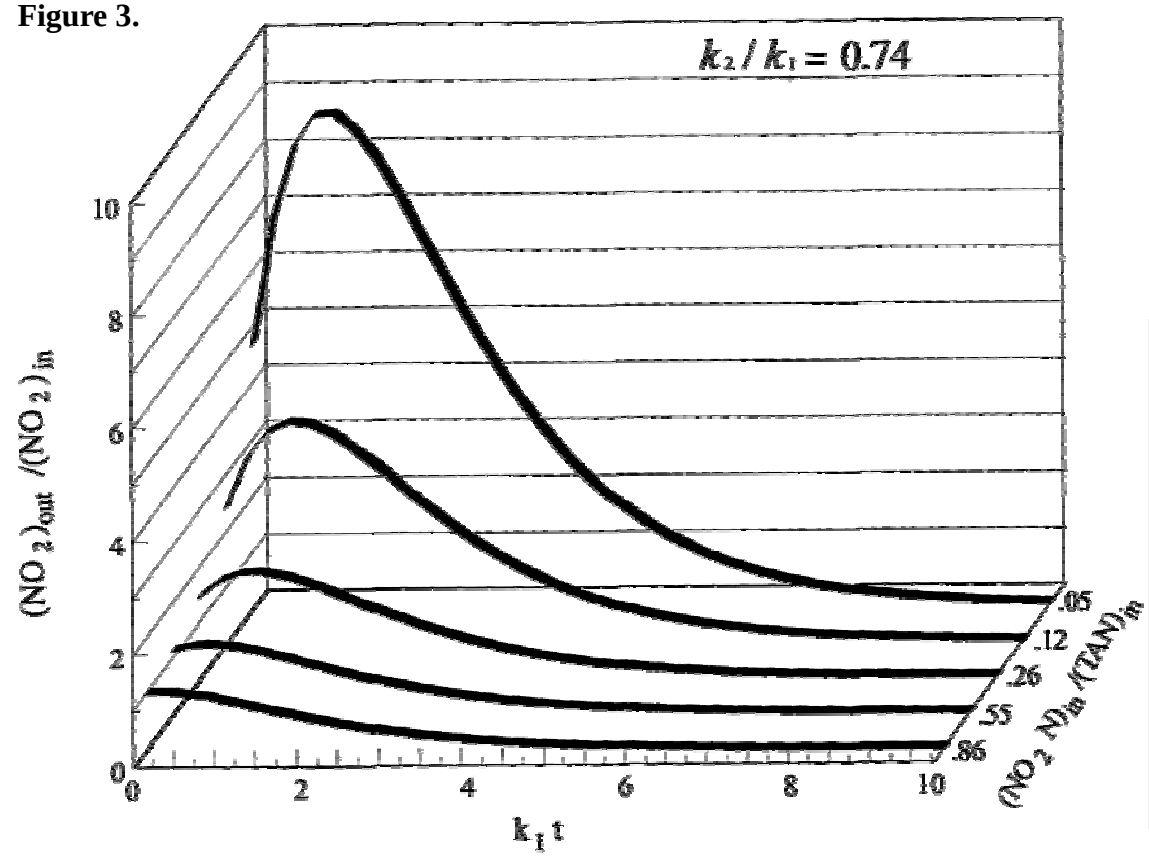
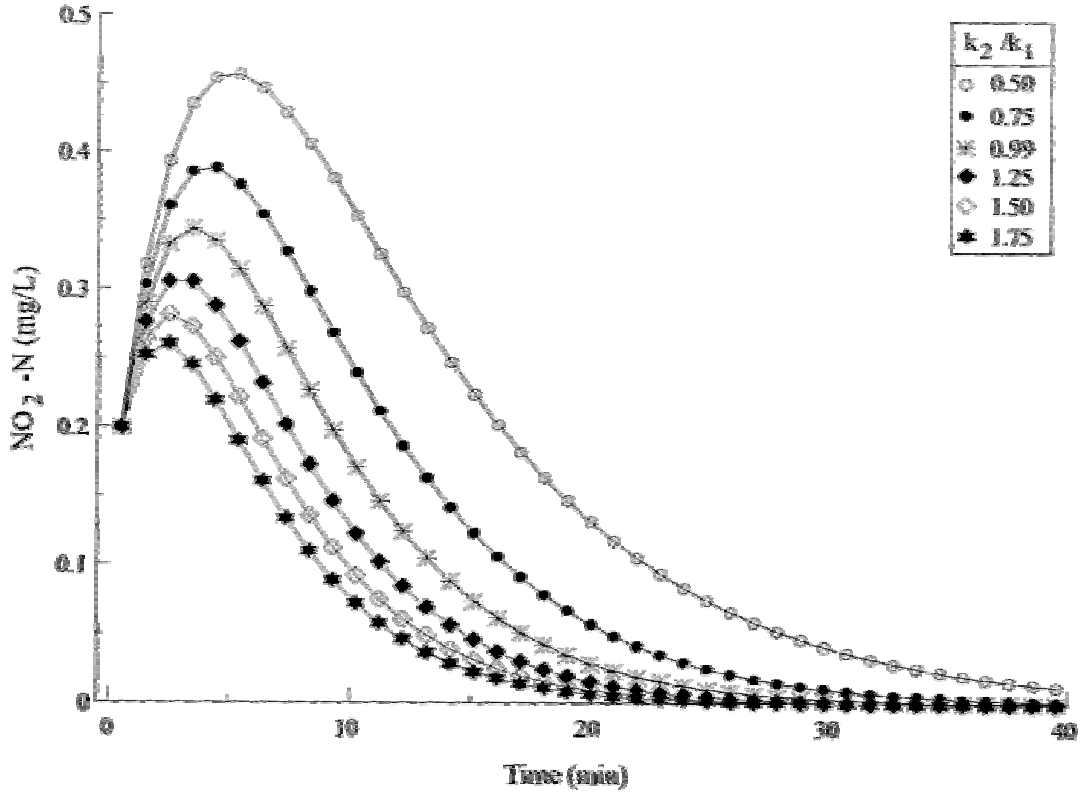


Figure 4.



Immediate and Stable Nitrification in Biofilters by Microbial Manipulations

Ami Horowitz, Ph.D.
Technical Director

Sarah Horowitz, Ph.D.
President

Aquaculture and Environmental Microbiology
UPAH Tech, Inc.
3666 Stoer Rd.
Shaker Heights, Ohio 44122

Abstract

Microbial populations of autotrophic nitrifying microorganisms were developed, that can withstand the inhibitory factors present in the aquaculture environment and function well in their presence. These microorganisms were further developed to possess high affinity to biofilter solid supports, such as plastic beads, enabling us to attach them to the supports in a strong and long-lasting way. By pre-coating these microorganisms onto biofilter solid supports, the start up time needed to operate nitrifying filters was practically eliminated. The activity of the microbially pre-coated biofilter solid supports was better than that of un-coated, conventional biofilter solid supports, and lasted for many months in fresh and seawater rearing facilities.

Introduction

Nitrifying biofilters are crucial to maintaining low levels of ammonia and nitrite in commercial recirculating aquaculture. The common start up time for new nitrifying biofilters is several weeks (Rosati, 1997) and the outcome is not predictable. The build up of microbial populations in these biofilters is left to chance, allowing the heterotrophic microorganisms a free hand at the expense of the more sensitive nitrifying autotrophs. This paper demonstrates how these hurdles can be overcome by microbial manipulations.

Results and Discussion

Nitrifying microorganisms are sensitive to inhibitors present in the water of aquaculture facilities (Horowitz and Horowitz, 1997). In order to obtain better nitrification, we isolated mixed nitrifying microbial populations from various natural environments and aquaculture facilities, and subjected them to microbial enrichment and selections procedures. The resulting mixed microbial populations performed better than the original microbial cultures when introduced to a green water tilapia recirculating system (in the case of fresh water nitrifying microbial populations) (Horowitz and Horowitz, 1997) or to a recirculating superintensive shrimp production system (in the case of seawater nitrifying microbial populations).

The majority of the nitrification activity in aquaculture production systems is performed by nitrifying microorganisms which are attached to a solid surface (e.g., the biofilter solid support surface), rather than free swimming microorganisms (Horowitz and Horowitz, 1997). Using microbial enrichment and selection techniques, we developed nitrifying microorganisms that possess high affinity to plastic biofilter solid supports, such as beads. This enabled us to attach these microorganisms to the biofilter support surface in a strong and long-lasting way. We pre-coated biofilter solid support with the new mixed population of nitrifying bacteria and studied the effect of the bacterial pre-coating on the potential of biofilter nitrification activity. The pre-coated solid support was incubated inside a tilapia tank in a recirculating green water system and was compared to a biofilter solid support that was not coated. During the test period, the 10,000-liter round tank occupied 250 Kg of fish that were fed about 1% of biomass per day (Horowitz et al., 1997). After a month, the microbially pre-coated and un-coated solid supports were tested for ammonia and nitrite removal potential. A significantly higher nitrification ability was exhibited by the microbially pre-coated solid support. Both ammonia and nitrite removal onset was immediate, and the ammonia and nitrite removal rates were much faster with the microbially pre-coated solid support than with the un-coated solid support. The un-coated solid support suffered a long lag time before ammonia and nitrite removal was detectable, apparently due to interference by heterotrophic microorganisms and organic matter (Figure 1).

The nitrification activity of microbially pre-coated solid support that had been incubated inside a tilapia tank in a recirculating green water system for five months was compared to the nitrification activity of fresh microbially coated solid support that had not been exposed to the fish tank. After the five months of exposure to the recirculating green water tilapia system, the microbially pre-coated solid support maintained its enhanced activity for removing ammonia and nitrite, and was as good as freshly prepared pre-coated solid support (Figure 2). Thus, microbial pre-coating of the nitrifying biofilter solid support has a long lasting positive effect on the biofilter nitrification potential and performance.

Ammonia removal was rapid and identical in the pre-coated and un-coated solid supports after 9 months of incubation. However, for nitrite removal, the advantage of pre-coating the biofilter solid support with the specialized nitrifying bacteria over the conventional approach of using un-coated beads was noticeable even after 9 months of continuous incubation in a tilapia green water system. Nitrite removal activity of the 9 months old microbially pre-coated solid support was immediate and fast, whereas the nitrite removal activity of the 9 month old solid support which was not pre-coated was slower, and suffered a long lag time before it started to show activity (Figure 3). Thus, microbial pre-coating enables the more sensitive nitrifiers (those that convert nitrite to nitrate) to maintain their stronghold on the biofilter solid support surface and withstand competition by other microorganisms.

The nitrifying microbial populations that were developed for fresh water systems and coated onto biofilter supports are presently applied in several tilapia recirculating production systems. The nitrifying microorganisms which were developed to perform well in a salt water shrimp culture environment were successfully coated onto biofilter

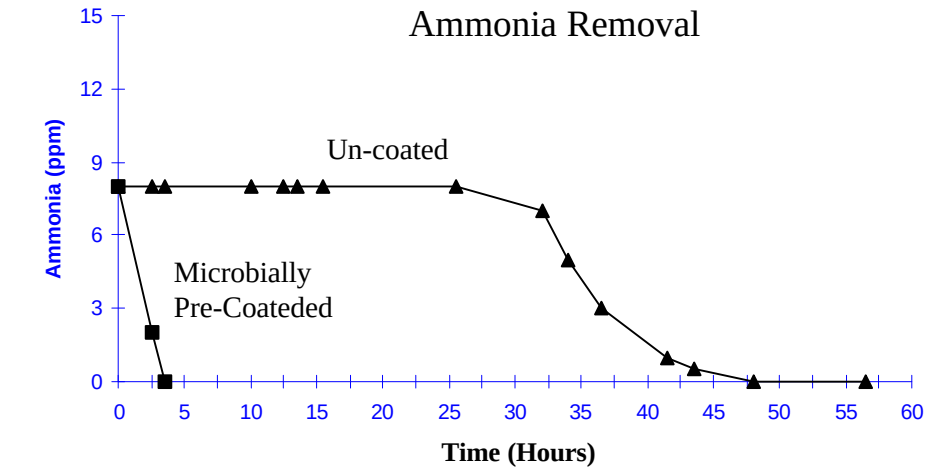
solid support, and are currently being used in recirculating intensive shrimp production system biofilters. By pre-coating these microorganisms onto biofilter solid supports, we were able to reduce the start up time needed to operate nitrifying filters to one to two days. Thus, there is no down time, and no risk of initial ammonia and nitrite accumulation for the fish and shrimp.

References

Horowitz, A., and S. Horowitz. 1997. What to watch for in biofilters? Advances in Aquacultural Engineering, M. B. Timmons and T. Losordo (Eds.), p. 295-308.

Horowitz, A., M.G. Pursley, S.B. Young, and S. Horowitz. 1997. Effect of biofilter configuration and solid support media pre-loading with nitrifying bacteria on tilapia in a recirculating green water system. Advances in Aquacultural Engineering, M. B. Timmons and T. Losordo (Eds.), p. 322-332.

Rosati, R. 1997. Practical guide for sizing and selecting biofilters. Aquaculture Engineering Society Fundamentals Workshop. Aquaculture '98, Las Vegas, NV.



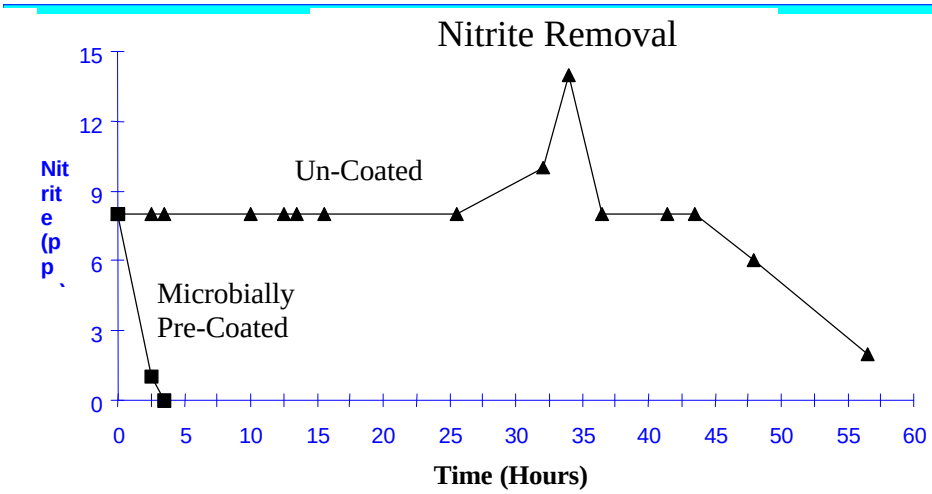


Figure 1. Removal of ammonia and nitrite by microbially pre-coated and un-coated solid supports that had been incubated inside a tilapia green water recirculating system for one month.

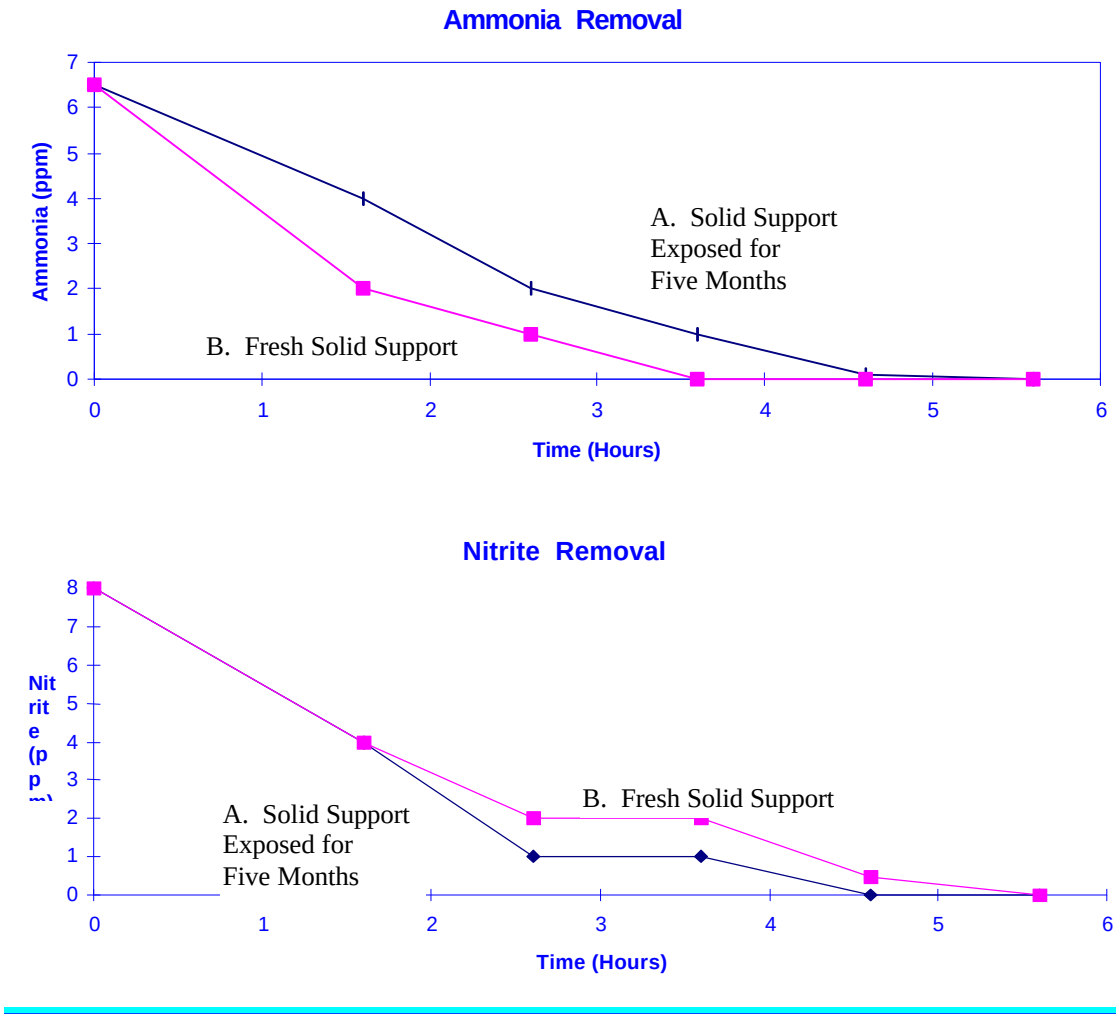


Figure 2. Removal of ammonia and nitrite by microbially pre-coated solid support which had been incubated inside a tilapia tank in a green water recirculating system for five months (A), and by fresh microbially coated solid support which has not been exposed to the fish tank (B).

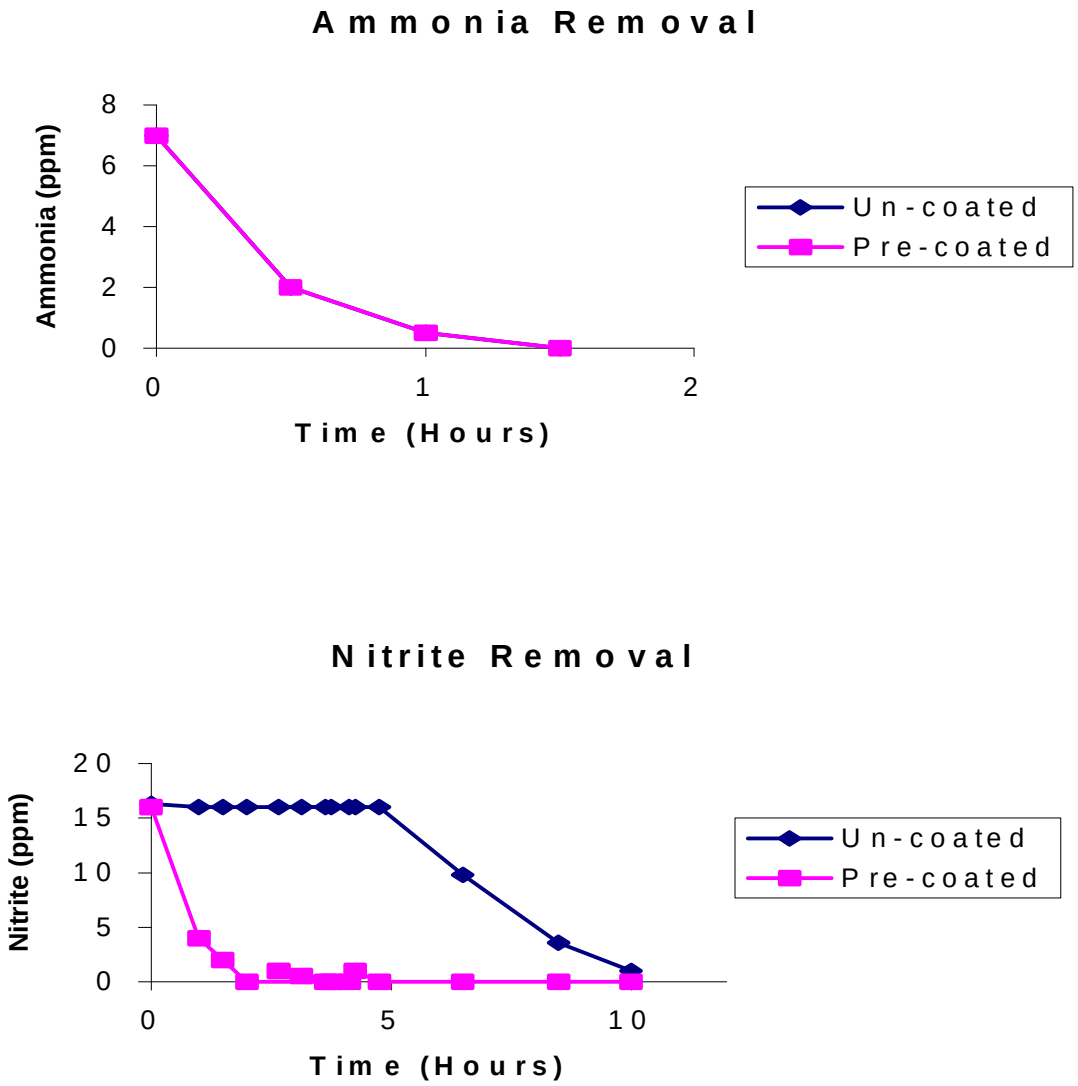


Figure 3. Removal of ammonia and nitrite by microbially pre-coated and un-coated solid supports that had been incubated inside a tilapia green water recirculating system for nine months.

Design of an Emergency Aeration System for Intensive Aquaculture Raceway Systems

M. Bower, S. Diaz, M. Ellis, B. Marchionini, and A. Sheldon
Biological Resources Engineering Department
University of Maryland College Park

This report summarizes the results of a Senior Design Project in the Biological Resources Engineering Department at the University of Maryland to design an emergency aeration system for intensive raceway production of trout. The system was tested by simulating both loading characteristics and fish respiration in a raceway, through a series of simulations using a deoxygenating slurry delivery system. The deoxygenating agents, sodium sulfite and cobalt chloride, in the slurry were delivered into a 2 m^3 holding tank as an impulse, step input and as a continuous input to simulate various test scenarios. The deoxygenating system was tested by monitoring dissolved oxygen concentrations in the holding tank for the three different inputs as a function of time. Several different aeration systems were designed and along with several commercial systems, subjected to the three loading scenarios. Dissolved oxygen levels were monitored for an extended period to ensure that the aeration system was able to maintain acceptable levels during a prolonged emergency scenario.

Development and Evaluation of a Feedback Control System for Dynamic Control of Dissolved Oxygen in Intensive Recirculating Aquaculture Systems

James M. Ebeling
Biological Resources Engineering Department
University of Maryland
jebeling@wam.umd.edu

Abstract

Over the past hundred years, agriculture productivity in the United States has reached record levels through mechanization, intensification, and automation. The short history of aquaculture has also seen a similar trend of increasing production levels in both open pond systems and indoors intensive recirculation systems. Improved monitoring and control of these production systems will yield a reduction in the risk of catastrophic losses and stress, in effluents and their potential environmental impact, in cost of production by maximizing yield per dollar of capital, and most importantly an overall improvement in product quality.

Low concentration of dissolved oxygen is the major variable limiting production in intensive aquaculture systems. With production densities approaching one pound of fish per gallon of water, supplemental oxygen is required to maintain optimal growing conditions. The high cost of on-site generation or transportation and storage of liquid oxygen makes it critical, for economic reasons, that pure oxygen be used in the most efficient manner possible. The ability to adjust oxygen concentration to meet constantly changing oxygen demand should have a significant impact on the overall economics of pure oxygen use. An improved understanding and control of dissolved oxygen in intensive systems will yield:

1) real time control of oxygen levels in the production tank,
2) elimination of high and low oxygen levels following feeding and other disturbances, thus reducing opportunities for stress induced diseases,
3) a quicker response to the faster changes in water quality as systems are pushed closer to their carrying capacity limits, and
4) automation of a critical process to reduce labor requirements and management responsibility.

This is one aspect of an overall research program to apply modern control system analysis to intensive aquaculture recirculating system, design and develop control algorithms and systems for optimizing water quality parameters and automate routine functions. This project developed a negative feedback control system for dissolved oxygen in intensive recirculating aquaculture systems. Control algorithms were developed and computer simulated for maintaining the dissolved oxygen levels in the

system at a specified set point, given a range of system loading and disturbances. System disturbances, influencing dissolved oxygen levels from baseline metabolism, include such dynamic changes as intensive feeding activity of the fish and stress response to some external stimuli. Several prototype control systems of varying design and cost were constructed from “off-the-shelf” components and installed on a research recirculation system at the Department of Biological Resources Engineering, UMCP. These systems were then evaluated based on their performance, i.e. ability to maintain a given oxygen set point under several fish stocking densities and system disturbances, and their overall economic savings compared to current systems used in the industry.

Growth of Mercenaria Seed Clams in a Recirculating Nursery System Utilizing Computer-Control and Fluidization Technology

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Timothy J. Pfeiffer
Aquaculture Systems Research Unit
USDA-Agricultural Research Service
University of Arkansas at Pine Bluff
Box 4912
Pine Bluff, AR 71611 USA

Kelly A. Rusch
Louisiana State University
Dept of Civil and Envir. Engineering
Baton Rouge, LA 70803 USA

Commercial hatcheries have utilized a number of approaches for taking young seed clams (1 mm) and growing them through the nursery phase to a size suitable for field planting (10 mm). The current developmental design of the recirculating seed clam system implemented at the University of Georgia Shellfish Aquaculture Laboratory consists of six clear cylindrical upweller units (5 cm in diameter, 76 cm in height); a 400 L feed reservoir; a solids separator; a bead filter (0.03 m^3); and utilized during the summer a 0.375 kW (0.5 hp) chiller unit. A 0.075 kW (0.1 hp) magnetic drive centrifugal pump provided a system water flow rate of 40 Lpm. A high water flow velocity (3.6 $\pm$ 0.2 Lpm) was maintained in each of the upweller units to fluidize the seed mass. Fluidizing the seed mass allows for the high density culture of seed by providing a more uniform distribution of food, and transporting waste material away from the seed mass. Culture density in each upweller unit reached approximately 5.5 g whole wet weight clam per cm^2 and greatest biomass growth rates (0.06 d^{-1}) were observed when an effective daily ration of approximately 2% dry weight of algae per g whole wet weight of clam was provided. Feeding of the seed clams from an algal storage reservoir and backflushing of the system bead filter were computer-controlled.

Fish Health Management of Recirculating Systems

David Crosby
Aquaculture Disease Specialist
Virginia State University
PO Box 9081
Petersburg, VA 23806

Fish producers today have very limited choices for controlling fish disease problems. Many bacteria that infect fish are now resistant to the only two FDA approved antibiotics. A fish health prevention/maintenance program is presently recognized as a required management practice by fish producers to lessen the risk of diseases,. This management approach has become particularly critical for producers using recirculating systems. Fish health prevention/maintenance programs require many elements including quarantine procedures, examination and monitoring for fish pathogens, and prophylactic treatments for parasites. Developing fish health prevention protocols are necessary to reduce likelihood of bacterial and parasitic diseases.

The vital key in the prevention of disease outbreaks is water quality. It well known that recirculating systems water quality can go from good to bad very quickly. When this happens dreaded disease problems often appear and seem to never go a way. Monitoring and managing of water quality parameters on a daily schedule is necessary management practice for recirculating facilities.

The greatest disease problem facing tilapia producers is Streptococcus (Strep). Strep is a rapidly emerging disease in the aquaculture industry using recirculating systems. Since control of this problem is difficult (Requiring extra label use of antibiotics in many cases), a fish health management plan is needed for reducing this problem.

Fish (Tilapia especially) should be checked and monitored for Strep at recirculating facilities. CNA agar with 5% sheep blood can be used for isolating Strep from fish. This media is well suited for screening for gram positives such as Strep. The brain (nervous tissue) or the intestines are used as the inoculum from fish. Plates are incubated for no more than 96 hrs. Suspicious colonies are gram strained. Those colonies that are gram positive cocci in pairs or short chain are further screened using the catalase test. Strep is catalase negative while many other gram positive cocci such as Staphylococcus are catalase positive. The number of fish (60 fish in most cases) in samples should follow American Fisheries Society Fish Health Section Blue Book procedures for pathogen detection.

It is critical that producers have a quarantine procedure in place for new fish arrivals at their facilities. It is at this stage where fish are checked for potential diseases problems. Fish are examined for parasites and Strep before moving them to production tanks. Also fish should

be treated prophylactically for external parasites before moving regardless of parasite intensity of fish.

Once in production tanks, fish should be sampled two to three times during the course of production to monitor for parasites and for Strep. A planned management program adds to expense of the operation but is preferred to having to shut down completely and disinfect an entire system.

Since fish are going to be sacrificed for a Strep check, they should be also examined for parasites especially Gyrodactylus (Skin Flukes). Research indicates that Strep infections are more likely to occur when trauma occurs to the epidermis. A heavy infection of skin flukes attaching to skin could create such epidermal trauma. Prophylactic treatments for external parasites must be part of a management plan. When fish are moved or transferred to new tanks a standard practice should include treatment for external parasites.

When developing a production facility, a fish health management plan should be use in guiding its construction. A producer should have a production system that is easily treatable and manageable for diseases. A facility that uses a few large units would most likely have difficulty in treatment and management of a problem leading to a shut down for disinfecting the system, thus having no production for months. If a disease gets into a facility with many small units, individual systems can be isolated and disinfected without total facility shut down.

Application of Industrial Monitoring and Control for an Experimental Carbon Dioxide Stripper in a Recirculating Aquaculture System

Scott M. Tsukuda
The Conservation Fund's
University of Maryland
Shepherdstown, WV

James M. Ebeling
Bioresources Engineering
Leetown Science Center
College Park, MD

Barnaby J. Watten
U.S. Geological Service
Leetown Science Cengter
Kearneysville, WV

Personal computers and generic industrial control software and modular hardware are well suited for monitoring, data collection, and control in aquaculture applications. This mix of components was chosen for the following reasons: PC's provide cost advantages over PLC based solutions, industrial control software offers a rich feature set, and modular hardware provides flexibility for input and output signals. This strategy was used to organize data from multiple locations in a recirculating aquaculture system using an experimental carbon dioxide stripper.

A standard desktop 486/66 computer running Windows 95 with an available serial port for RS-232 communication was used for this application. As the computer was placed in an office environment rather than in the laboratory it did not need to be an industrial version. Personal computers are relatively inexpensive machines that can be used simultaneously with other applications such as word processing in non-critical process monitoring and control environments. Additional stability is gained by dedicating the machine to data acquisition and control.

The user interface must be operator-friendly to facilitate data monitoring, collection, and analysis. We used Genesis by Iconics, a PC based industrial software package with scripting features for control algorithms and a customizable user interface. For our application, the software was used to collect and display data for the user in the office. Monitored data are summarized in a main display with additional detail such as pulsed-bed biofilter status, CO2 scrubber status and tank status, being easily selected by pressing a button for the appropriate display. Historical trending is also displayed for review of trends in collected data. Data from a CO2 gas phase monitor is used to control a three stage-carbon dioxide scrubber coupled with a pure oxygen contactor. The monitor uses an infrared detector to generate an analog signal proportional to CO2 readings over the range of 0 to 10% by volume.

The application uses modular Dutec® hardware for in-lab collection of data and control. The system uses a single RS-232 cable to send data to the office where the computer is located. It can be further expanded by using additional base units and the RS-422 or RS-485 communication protocols. The Dutec model used in this project consists of a base unit with 16 Analog or Digital signals and an expansion unit, allowing an additional 16 Digital I/O points. Individual modules were purchased according to the type of signal. The cost of this unit is leveraged most effectively when all 16 input/output locations are used.

Water level, dissolved oxygen, temperature, and flow rate are among the standard types of data collected. The flexibility of the Dutec base unit allowed us to purchase individual modules for each type of input or output provided which included DC digital inputs and outputs, AC analog voltage inputs, and AC analog current inputs.

As the electronics industry is constantly advancing, newer products offering additional features and simplicity of use continue to arrive on the market. This system will be able to adapt to changes in available products. Because a generic design was used, the sensors, the hardware, and software can be interchanged or replaced if necessary. At the same time, as configured, this system has adequate monitoring and control features for most aquaculture applications.

The Use of Commercial Probiotics in the Production of Marine Shrimp Under No Water Exchange

Tzachi M. Samocha, Ph.D.
Associate Professor
Shrimp Mariculture Project
TX Agricultural Experiment Station
Texas A&M University System
4301 Waldron Rd.
Corpus Christi, Texas 78418

Addison L. Lawrence, Ph.D.
Professor
Shrimp Mariculture Project
TX Agricultural Experiment Station
Texas A&M University System
1300 Port Street
Port Aransas, Texas 78373

Ami Horowitz, Ph.D., Technical Director
Sarah Horowitz, Ph.D., President
Aquaculture and Environmental Microbiology
UPAH Tech, Inc.
3666 Stoer Rd.
Shaker Heights, Ohio 44122

Abstract

The effect of a commercial bacterial supplement (probiotics) on the high density production of *Penaeus setiferus* in an outdoor tank system with no water exchange was studied, using high (45%) and low (20%) protein diets and high aeration. At $\alpha = 0.05$, the 3 months study revealed no significant difference between tanks treated with the commercial bacterial supplement and those that were not, for the mean shrimp survival, shrimp final yield, and shrimp final weight. However, some differences were significant at $\alpha = 0.08$. The probiotics treatment had no effect on the nitrogen cycle in the tanks. The commercial bacterial supplement was further tested for its microbial activity on shrimp sludge. No major differences were noted in respiration and chemical oxygen demand (COD) of the treated and control sludge samples. However, at the end of the test, biological oxygen demand (BOD) in the treated sludge was lower than that of the untreated sludge. Thus, based on this work and other studies of probiotics' use in aquaculture, it appears that commercial bacterial supplements might have some advantage, but more studies are necessary to answer this issue unequivocally.

Introduction

The issue of using commercial bacterial supplements (in liquid or powder forms) to benefit aquaculture production systems is controversial (Jory, 1998). This work describes the potential of probiotics use in high-density *P. setiferus* production systems with no water exchange.

Results and Discussion

A commercial bacterial supplement (BioStart HB-1&HB-2) was added according to the manufacturer (AMS) instructions to 6 test tanks of *P. setiferus* grown in an outdoor high-density production system with no water exchange and with high aeration. Three of the tanks were fed with a high (45%) protein diet, and the other three were fed with low (20%) protein diet. The 6 control growout tanks were fed as above, but did not receive the probiotics supplement. The volume of each tank was 10 m^3 and each tank had a 15 cm layer of clay soil on the bottom. The 3 months study revealed no significant difference between tanks treated with the commercial bacterial supplement and those that were not, at both the 45% and 20% protein diets, for the mean shrimp survival, shrimp final yield, and shrimp final weight at $\alpha = 0.05$ (Table 1). At $\alpha = 0.08$, however, some significant differences were noted.

Table 1. Mean shrimp survival, final yield, and final weight of *P. setiferus* treated or untreated with a commercial bacterial supplement (BS) after 3 months in an outdoor tank system with no water exchange. Feeding was with a 45% or 20% protein diet.

Treatment	Survival* % $\pm$ STD	Final Yield* Kg/m^2	Final Weight** g $\pm$ STD
45% + BS	84.1 $\pm$ 19.67	0.380	9.03 $\pm$ 1.681[a]
45% - No BS	94.1 $\pm$ 4.14	0.482	10.24 $\pm$ 1.247[b]
20% + BS	72.5 $\pm$ 24.61	0.413	11.40 $\pm$ 1.458[c]
20% - No BS	85.9 $\pm$ 7.14	0.431	10.03 $\pm$ 1.040[d]

* There is no significant difference at $\alpha = 0.05$ between treatments in each of these columns.

** There is no significant difference at $\alpha = 0.05$ between treatments in this column, except between a and c (a<c). At $\alpha = 0.08$ there is a significant difference in this column between the following treatments: a and b (a<b) and c and d (c>d).

The levels of ammonia, nitrite and nitrate in the experimental tanks were followed with time. There were no striking differences in the pattern of ammonia, nitrite, and nitrate accumulation with time in the tanks that received the commercial bacterial supplement and the untreated controls. The build-up of ammonia and nitrite levels with time was, however, related to the protein level in the feed. As expected, the tanks fed with the 45% protein diet had significantly higher levels of ammonia and nitrite than those fed with the 20% protein diet. Accumulation and removal rates of ammonia and nitrite were not affected by the probiotics addition.

The commercial bacterial supplement was further tested for its microbial activity on shrimp sludge. Shrimp sludge was incubated in the lab with and without probiotics addition. No major differences were noted in the respiration and COD between the

treated and untreated sludge. However, at the end of the test, BOD in the treated sludge was lower than that of the untreated sludge (Table 2).

Table 2. The effect of a commercial bacterial supplement (BS) on shrimp sludge – a lab study.

Treatment	Respiration mg O_2/l/hr after 1 day	COD ppm t=0	COD ppm 2 days	COD reduction in 2 days	BOD ppm t=0	BOD ppm t=2	BOD reduction in 2 days
Control sludge	2.02	89	77	13%	4288	3520	18%
Sludge + BS	2.20	87	79	9%	3104	352	89%

Based on the present work and on other studies of probiotics' use in aquaculture (Jory 1998; Queiroz and Boyd, 1998), it appears that there is no clear benefit for adding such bacterial supplements. However, some of the differences that were noted, especially BOD removal and shrimp size with low protein feed, need to be studied further. Similarly, Queiroz and Boyd (1998) found higher yield of catfish (at $\alpha = 0.10$) with the addition of another commercial probiotics product. Better understanding of the reasons for such effects may lead to better probiotics products in the future.

References

Jory, D.E. 1998. Use of probiotics in Penaeid shrimp growout. Aquaculture Magazine, 24(1): 62-67.

Queiroz, J.F., and C.E. Boyd. 1998. Effects of a bacterial inoculum in channel catfish ponds. J. World Aquaculture Soc., 29(1): 67-73.

A Study of Selected Fish Feed Binders: Effect on Generated Waste Quantity and Quality

Fredrick W. Wheaton and Sahdev Singh
Bio. Resources Engineering Dept.
University of Maryland
College Park, MD 20742

John N. Hochheimer
Piketon Research and Extension Center
The Ohio State University

Joseph H. Soares, Jr
Animal Sciences Department
University of Maryland
College Park, MD 20742

Abstract

Aquacultural effluent and the potential for pollution from it are some of the most widely discussed problems both within and outside of the aquaculture industry. Recirculating aquacultural systems are intensely stocked with fish, feeding on high-protein feed, to obtain production values that offset the high capital and operating costs of such systems. In these systems, approximately 3% of the fish feed is discharged as ammonia and 30 to 60% as solid waste rich in organic matter content. Such a wastewater has a potential to degrade the receiving water quality quite significantly.

This paper presents the results obtained from a series of experiments, conducted at the University of Maryland (UMCP) with hybrid striped bass, on the study of the effect of selected binders on generated waste quality and quantity. It also compares the UMCP findings with those from similar studies at Cornell University, Illinois State University Normal, and Louisiana State University Baton Rouge, using trout, tilapia, and catfish, respectively. The main objective of the presented study was to determine the effect of adding binding supplements to fish feed on fish growth, culture water quality, and culture system filter performance.

At UMCP, initially nine preliminary diets, including a control, were tested and then four, including a control, were further investigated. Treatment diets for preliminary studies were prepared using carageenan, sodium alginate, wheat gluten, guar gum, nutra-binder, bentonite, lignin sulfonate, and Pel-Plus as supplemented binders in a basal feed. Preliminary trials used all binders at a relatively high percent (10%) of the feed. Wheat gluten or nutra-binder (NB) at 5%, lignin sulfonate (LS) at 3%, and bentonite (B) at 5% of the feed were the three binders that were used for more detailed tests (secondary trials) after preliminary trials.

At UMCP, hybrid striped bass showed poor acceptance of feeds containing 10% guar gum, sodium alginate, wheat gluten, or bentonite as binder material. Weekly measured water quality parameters and solids characteristics (particle size distribution) during the secondary trials were not significantly different among the four treatments at the 5% level of significance. Somewhat similar results were reported by investigators at the other universities.

Effect of Chemotherapeutants on Nitrification in Fluidized-Bed Biofilters in a Recycle Rainbow Trout Culture System

M. F. Schwartz, G. L. Bullock, S. T. Summerfelt, J. A. Hankins and J. A. Mathias
The Conservation Fund's Freshwater Institute, Shepherdstown, WV

Two commonly used fish therapeutants, formaldehyde and benzalkonium chloride (roccal), were evaluated for their effect on the nitrification efficiency of fluidized-bed biofilters. The therapeutants were added at conventional concentrations to two small-scale (2200 L) recirculating trout culture systems which each contained six fluidized bed biofilters operating in parallel. Biofilter efficiency was measured, before and after treatments, by determining ammonia and nitrite removal efficiencies at ambient conditions, and when challenged with a spike of ammonium chloride at a concentration four times that of the ambient TAN. Three formalin treatments in recycle mode ranging from 167-300 ppm had no significant effect on biofilter efficiency. Four roccal treatments from 1-2 ppm were conducted; three bath treatments and one recycle treatment. None of the roccal treatments had an immediate effect on ambient nitrification but a delayed drop in biofilter efficiency was observed after 3-5 days with no effect detectable after 7 days in all of the treatments. There was no catastrophic drop in nitrification efficiency caused by any of the treatments. In only one instance did roccal treatment have a deleterious effect on nitrification when the biofilters were challenged.

Water Quality Limitation of Fixed Film Biofilters for Recirculating Aquaculture Systems

Shulin Chen and Songming Zhu
Department of Biological Systems Engineering
Washington State University
Pullman, WA 99164

The existing research, demonstration, and commercial recirculating aquaculture systems almost exclusively use fixed film biofilters for nitrification. The main advantage of this type of process is that the biofilm can maintain a relatively long cell detention time that accommodates the low growth rate of nitrifying bacteria. Due to the nature of the diffusion controlled process that occurs in a biofilter, the nitrification rate of a fixed film filter is directly related to ammonia concentration in the system. Based on biofilm theory, this paper presents a mathematical model for ammonia removal in a recirculating system, considering mass transfer through a biofilm, feeding rate, and volume and water exchange rates of the system.

Minimum concentration of substrate

A biofilm is a layer-like aggregation of microorganisms attached to a solid surface. Because the mass transfer in a fixed film process is diffusion controlled, there must be a minimum substrate concentration maintained in the bulk water to create a concentration gradient that drives the nutrient transport across the biofilm. In addition, a stable nitrifier population requires an adequate concentration of substrate. According to Rittmann and McCarty (1980), the minimum substrate concentration, S_{min} ($g.m^{-3}$), can be calculated as:

$$S_{\min} = K_S \frac{b}{Yk - b}$$

Where, K_S= half-velocity coefficient ($g.m^{-3}$); b = specific bacterial decay (day^{-1}); Y= yield of bacterial mass per unit of substrate mass ($g.g^{-1}$); k= maximum specific rate of substrate use ($g.g^{-1}.day^{-1}$).

Substrate flux into a biofilm

The flux of substrate into a biofilm (or removal rate), J ($g.m^{-2}.day^{-1}$), can be estimated by the formula of Atkinson and Davis (1974):

$$J = h\, kL_f X_f \frac{S_S}{K_S + S_S}$$

Where, η = the effectiveness factor; L_f = thickness of the steady-state biofilm (m); S_S = interfacial concentration of substrate ($g.m^{-3}$); X_f =bacterial density in biofilm ($g.m^{-3}$). A detailed description of this calculation can be found in Rittmann and McCarty (1980 and 1981).

Recirculating aquaculture systems

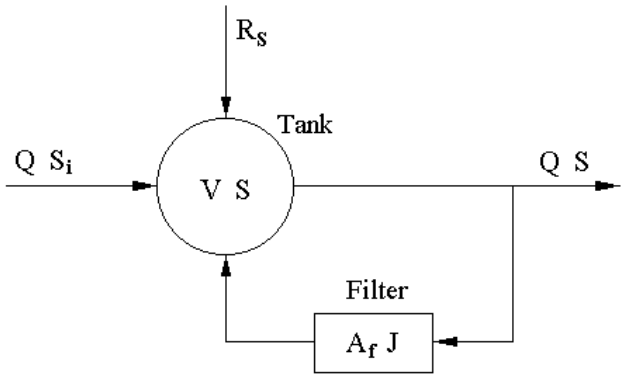


Fig. 1 Substrate transfers in a recirculating aquaculture system.

For a recirculating aquaculture system (Fig. 1), the equation of substrate mass balance can be expressed as:

$$V\frac{dS}{dt} = R_S R_f W_f - A_f J - (S - S_i)Q$$

Where, A_f = biofilm area (m^2); Q = water exchange rate ($m^3.day^{-1}$); R_f =daily fish feeding rate (g feed (g fish)$^{-1}$.day^{-1}); R_S = the ratio of substrate production to feed mass (g ammonia (g feed)$^{-1}$); S = substrate concentration ($g.m^{-3}$); S_i = substrate concentration of inflow water ($g.m^{-3}$); t = time (day); V = water volume (m^{-3}); W_f = total fish weight (kg).

Simulation and discussion

The ammonia nitrification process is simulated here since it's important for recirculating aquaculture. The relationship between ammonia concentration and removal rate for different values of biofilm parameters is indicated in Fig. 2. Due to energetic and kinetic constraints, a minimum concentration is needed to support a steady-state biofilm. Below the minimum concentration, biofilm growth occurs at a negative rate and the monolayer of bacteria gradually disappears. As a result, no steady-state-biofilm will exist, and substrate flux will be zero. When ammonia concentration is above the minimum level, but not very high, the relationship between ammonia concentration and removal rate is approximately linear (Fig. 2).

For recirculating aquaculture, simulation is carried out for a system with a 20 m^3 water tank and a PBF-10 filter (Aquaculture System Technologies, LLC, 1996). The biofilm area of the filter is about 370 m^2. It is assumed that the ammonia concentration of the inflow and the initial water bulk are zero, the daily fish feeding rate is 2.5%, and the ratio of ammonia production to feed mass is 3%. The values of the biofilm parameters are assumed to be the same as curve 2 in Fig. 2. Thus the minimum ammonia concentration is 0.286 $g.m^{-3}$, and the relationships between ammonia concentration, removal rate, and

fish biomass can be determined (Figs. 3 and 4). It is clear that different minimum amounts of fish mass must be maintained for different water exchange rates, to keep the ammonia concentration above the minimum level. Otherwise, the ammonia removal rate will be zero (Fig. 4) and the biofilm will be destroyed. When fish mass is more than the limit, both ammonia concentration and its removal rate have an approximately linear relationship with fish mass (Figs. 3 and 4).

This paper gives the initial results of the water quality limits of fixed film biofilters used for recirculating aquaculture systems. Note that BOD_5 interaction has not been considered, and the kinetic parameters used in the simulation are based on non-aquaculture wastewater. Additionally, further studies are needed, especially on experiments for parameter calibration, model validation, and effective application.

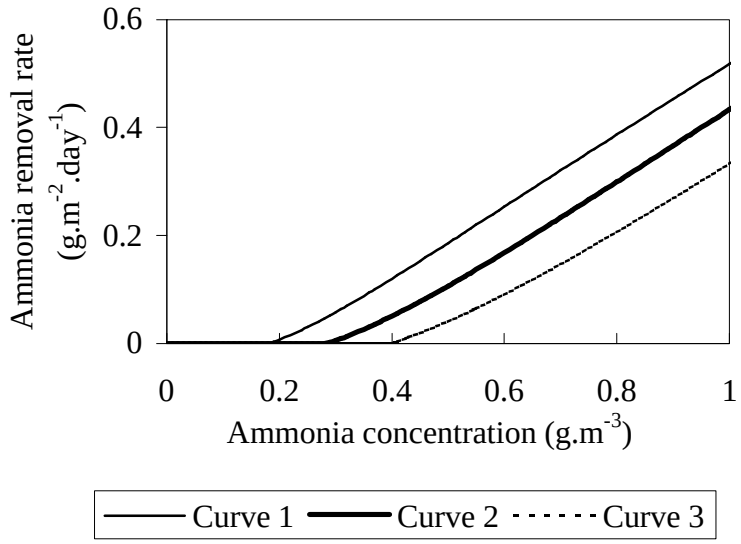


Fig. 2 Relationship between ammonia concentration and removal rate.

Curve 1: K_S=2 g.m^{-3}, k=2.0 g.g^{-1}.day^{-1}, Y=0.3 g.g^{-1}, S_{min}=0.182 g.m^{-3} ;
Curve 2: K_S=2 g.m^{-3}, k=2.0 g.g^{-1}.day^{-1}, Y=0.2 g.g^{-1}, S_{min}=0.286 g.m^{-3} ;
Curve 3: K_S=2 g.m^{-3}, k=1.5 g.g^{-1}.day^{-1}, Y=0.2 g.g^{-1}, S_{min}=0.40 g.m^{-3} .

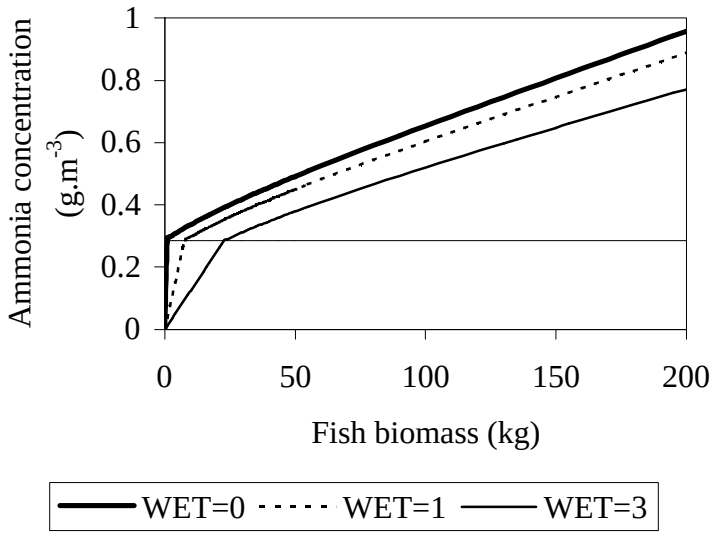


Fig. 3 Computed result of ammonia concentration vs fish mass. WER is water exchange times of the total water volume per day.

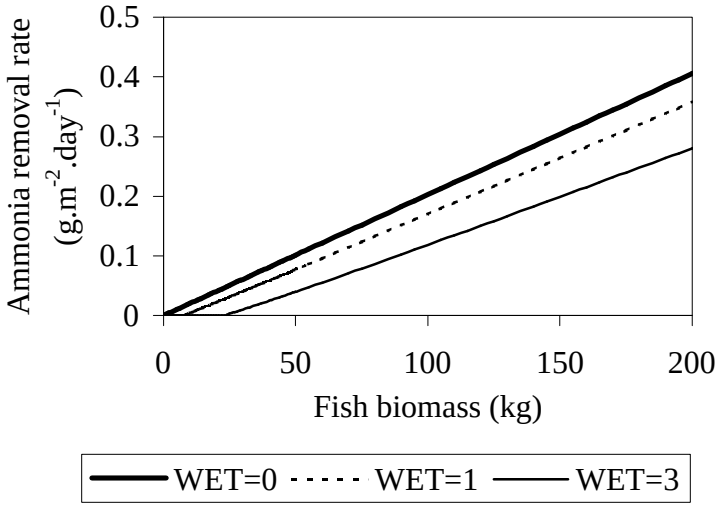


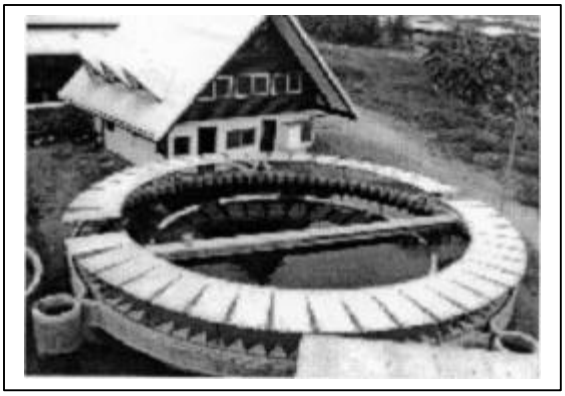
Fig. 4 Computed result of ammonia removal rate vs fish mass.

Aquaculture Engineering Design of Tilapia Breeding System in a Freshwater Recirculating System

Teodore U. Abalos
Institute of Fisheries Policy and Development Studies
College of Fisheries, University of the Philippines in the Visayas
Miag-ao, Iloilo 5023, Philippines

Abstract

Production of male tilapia individuals using hormone has become increasingly popular in the aquaculture industry and its effect on human health has not been known. Tilapia male individual grows faster compared to females. Tilapia fry supply in the Philippines is mainly dependent from hatcheries in fishponds, cages and fishpens in lakes and reservoirs. With the rapid expanding growout production system, degradation of the natural bodies of water and the absence of management policies will pose a problem for many aquaculture operations. There is therefore a need to design tilapia breading system that is environment friendly and poses no problems on consumers in the design and construction of pilot tilapia breeding system in a freshwater recirculating system in Cabuyao, Laguna, Philippines.



The breeding system is composed of male compartment, female compartment, fry compartment and fry collector with a total area of 154m^2. Five hundred (500) female (*Oreochromis niloticus*) and one hundred (100) male (O.areus) tilapia breeders were stocked in their respective compartment to evaluate the system performance in the production of male tilapia fry. The average recorded hybrid fry production in five (5) months was 29,008 per month. Based on five hundred (500) female breeders used, the average production rate was 58 fry/female/mo ranging from 37.4 to 74.2 fry/female/month. The production per unit area of the system equivalent to 188.4 fry/m^2/mo ranging from 121.5 to 241.0 fry/ m^2/mo. The percentage of hybrid male production based on random sampling (n=300) from first batch progeny grown in growout rearing tank for three (3) months was 91%.

System redesigning and re-engineering using alternative source of energy could be considered potential household backyard fry production component to supply the fry requirement of the industry. Further design improvement and development of the system for cost effective and efficient fry collection to minimize handling should be given emphasis.

Keywords: tilapia, breeding system, aquaculture engineering, and recirculating system

Performance of a Prototype Zeolite Recirculating Aquaculture System

Harry Westers
Aquaculture Bioengineering Corporation
Rives Junction, MI 49277

Joseph T. Fuss
Aquatic Systems Engineering
Middlebury Center, PA 16935

Abstract

A prototype demonstration recirculation aquaculture system (RAS), using a natural zeolite to remove ammonia, successfully supported and grew coho salmon (O*ncorhynchus kisutch)* for a period of one year.

The system has a total water volume of 4.0 m^3 (4000 l) distributed over two 1.3 m^3 volume circular rearing tanks, one 1.0 m^3 volume settling tank and a 0.4 m^3 volume zeolite column. Less than 5% of the water volume was replaced daily.

The rearing tanks are equipped with a split drain system. Solids are removed using one third of the 120 lpm recirculating flow, and are discharged via a central bottom drain into the settling tank. From there, the solid free water is pumped through the zeolite while injected with high purity oxygen and ozone.

Two thirds of the flow exits the tanks by way of a central standpipe, located at mid-depth. This flow is pumped to the top of a 2.0 m tall oxygen/ozone enhanced packed column for oxygenation and degassing of carbon dioxide. Adding ozone to the column retards biofouling of the packing media.

Temperature and dissolved oxygen were monitored continuously, total ammonia nitrogen and pH were recorded daily, total alkalinity and carbon dioxide were measured occasionally. Temperatures were maintained at an average of 12.0^{0}C, dissolved oxygen near or slightly above saturation. Because the normal pH of the water is 7.8, initially the pH was adjusted to maintain un-ionized ammonia at recommended concentrations. However it was soon noticed the salmon tolerated higher concentrations and pH adjustments were discontinued. Despite high ambient un-ionized ammonia concentrations of up to 0.35 mg/l, fish showed no signs of ammonia toxicity (gill hyperplasia) but ceased feeding at concentrations in excess of 0.20 mg/l.

The pH ranged from 7.8 to 8.0 most of the time and carbon dioxide registered from 30 to 35 mg/l. Total alkalinity increased, over time, from 300 mg/l to up to 1000 mg/l. Both calcium and sodium concentrations increased as a result of zeolite regeneration with a solution of NaCl and NaOH.

Ozone prevented biofouling of the zeolite and simultaneously disinfected the recirculation flow. Water clarity, most of the time, approached drinking water quality. Neither fungal nor bacterial infections were observed, even on fish with eroded snouts.

Fish growth was normal, feed conversion averaged one. The system was designed to support a maximum biomass of 208 kg (80 kg/m3), it maxed out at a biomass of 260 kg of 60 g fish and a daily feed input of 2.8 kg. This biomass represented a rearing density of 100 kg/m^3 (6.25 lbs/ft^3), a loading of 2.2 kg/lpm (18 lbs/gpm). Zeolite was regenerated with a 2.0 percent salt solution elevated to a pH of 11.5 to 12.5 with sodium hydroxide, allowing degassing of the ammonia.

Zeolite columns were regularly regenerated after feeding 10 kg of food. By that time the ambient total ammonia nitrogen concentration was at 20 mg/l, un-ionized ammonia between 0.25 and 0.35 mg/l.

Bacterial floc routinely obstructed rearing tank discharge plumbing and became especially bothersome once daily feed exceeded 1.5 kg. Some improvements were made in design to simplify clean-out, and routine maintenance readily controlled the floc.

The system's design and operating parameters were applied to estimate the costs of large systems. To construct a similar system at commercial scale costs approximately $3.00 per pound annual production capacity. Production costs (utilities, feed, chemicals, annual labor) for a coldwater species range from $1.50 to $1.75 per pound.

The results obtained thus far with the coho salmon are very encouraging and further studies are warranted.

An Integrated Recirculating System for the Production of Oysters and Shrimp

Jaw-Kai Wang
Biosystems Engineering Department
University of Hawaii at Manoa
Honolulu, Hawaii 96822
Jawkai@hawaii.edu

Aquatic animal production can be increased beyond what has been achieved without requiring additional water resources or causing additional environmental impact. In traditional culture the carrying capacity of a production system is ultimately limited by the accumulation of toxic metabolites such as NH_3 and CO_2 in the water. To increase the carrying capacity of a traditional aquaculture production system beyond its normal limit, specific waste treatment processes must be added to remove the limiting metabolites. Temperate zone re-circulating aquaculture production systems are examples of what can be done to increase the carrying capacity of an aquaculture system. However, these waste treatment measures add costs to the production, and commercially successful examples are rare. Two recent developments, the Partitioned Aquaculture System being developed at the Clemson University and the Integrated Recirculating Systems being developed at the University of Hawaii, are both noteworthy since they have gone beyond the present day waste management technology and demonstrated that cost competitiveness is possible for integrated recirculating systems.

The University of Hawaii Integrated Recirculating Oyster/Shrimp Production Systems is based upon a simple concept. In an oyster/shrimp system, the excess nutrients in the shrimp tank can be used to produce marine algae, such as *Chaetoceros* sp. The algal water is then pumped to a fluidized packed oyster column where oysters are suspended individually in a stream of high velocity water from the shrimp tank. The oyster feed on the algae, thereby eliminating oyster food cost while reducing the excessive nutrient load caused by the incomplete utilization of the shrimp feed. After the algae has been removed by the oysters, the water is returned to the shrimp tank to be reused. Up to 95% of sustained water reuse has been achieved. The normal range of water reuse is 80% to 90%. The system rests on two patents: The first is the Fluidized Bed technology, and the second is a pending patent application on the controlled production of marine algae in an open system. The ability to control the algal species is important to the success of the system, since it must be the right food for the oysters. By controlling the nutrient input to the tank, and by making sure there are sufficient oysters to remove algae continuously, a desired dominate algal species in the system can be maintained.

A venture capital group has licensed the patents from the University of Hawaii and created the Kona Bay Oyster & Shrimp Company in 1997. The company has begun its operation and is expected to reach full production in late 1998. Flat-bottomed, twenty-eight foot diameter round shrimp tanks with center drains are used. Circular water

motion is maintained in the tanks to remove all settleable solid. The oyster columns, 18 inches in diameter and 6 feet in height, can contain about 3,000 55-gram size American Cup oysters.

This paper shows the design steps for such a system.

Design and Construction of a Commercial Biosecure, Closed, Recirculating Shrimp Production System

Phillip G. Lee and Philip E. Turk
Aquatic Feeds and Filters, Inc.
PO Box 3131
Galveston, TX 77552

Addison L. Lawrence
Texas A&M Experiment Station
Shrimp Mariculture Research
Port Aransas, TX 78373-1450

Production losses from disease (i.e. viruses) have had a serious negative economic impact on marine shrimp farming world-wide. The need for specific-pathogen-free (SPF) broodstock that are either geographically or environmentally isolated from common diseases has become a priority. The latter is more difficult to accomplish because of possible sources of contamination from influent sea water, shared facilities and shared personnel. The establishment of commercial, environmentally isolated broodstock also necessitates the use of totally, closed recirculating water filtration systems to contain the costs of water replacement due to declining pH and nitrate accumulation.

A commercial biosecure facility composed of 4-100 mt raceway systems has been designed and constructed. The system is composed of 4-3.3 m W X 33 m L X 1.3 m D concrete raceways housed in greenhouses. Each raceway has a central concrete partition and a 1.6 m deep settling basin at one end. All effluent water is drawn from a screen standpipe located in the middle of the settling basin. Filtered water is returned to the surface of the raceways along the central partition at 1-2 m intervals. In addition, a cleaning system consisting of notched 5 cm polyvinyl chloride (PVC) pipe located along the lateral walls and medial partition suspends uneaten feed and particulates off the bottom. Two of the raceways have a combination upflow bead (2.2 m^3)/fluidized sand (1.44 m^3) biofilter system supplied with water from a 2-1 hp pumps (200 lpm). The other two raceways have a reciprocating biofilter (8.9 m^3) supplied by an airlift pump (500 lpm). All four raceways have protein skimmers and activated charcoal filters. The tanks and filters were all new construction and artificial sea salts were used to establish and maintain the salinity (5-25 ppt). Each raceway was designed to produce >100 shrimp m^{-2} for a total of 40,000 biosecure adult broodstock. In addition, the raceways were used to acclimate 5.5 X 10^6 postlarvae before they were stocked into production ponds.

This research project was supported by Woods Brothers Shrimp Farm, Gila Bend, AZ and a State of Texas Higher Education Coordinating Board Technology Development and Transfer grant (# 004952-079).

Procedure for Analyzing the Technical and Economic Risk of a Recirculating Aquaculture System

Kerry W. Tudor and Patrick D. O'Rourke
Illinois State University
Department of Agriculture
Campus Box 5020
Normal, IL 61790-5020

Since the economic results of investing in an aquaculture enterprise are unknown at the time the investment decision is made, it is advisable to analyze the uncertainty and risk of such an investment. This is particularly true when the potential aquaculture enterprise includes a recirculating system. Recirculating systems require large capital expenditures for building and equipment which give rise to high fixed costs and high operating leverage. If the investment is financed with a combination of debt and equity, the investors have both operating leverage and financial leverage, and increases in either tend to increase business risk. In addition, a recirculating aquaculture system is subject to technical failures which can result in critical death losses of fish.

One method of analyzing the risk of a potential investment is to simulate outcomes on paper prior to committing funds to physical facilities. The accuracy and usefulness of the simulation depends, of course, on the accuracy of underlying assumptions and the comprehensiveness of the model of the firm which is being simulated. Data collected from established recirculating systems contributes to the accuracy of underlying assumptions, and computer software allows simulation of fairly complex models of an aquaculture enterprise.

A spreadsheet was developed based upon the commercial-scale recirculating system in operation at Illinois State University. It contained information pertaining to revenue, capital costs, and operating costs including feed costs, feed conversion ratios, and fish growth rates. Output cells in the spreadsheet included net income, net present value, modified internal rate of return, breakeven volume measured in dollars of sales, and breakeven volume measured in kilograms sold. Estimates of probability density functions for various sources of technical and economic risk and uncertainty were derived from best available information and incorporated into the spreadsheet. A commercial risk analysis software which was based upon the Latin Hypercube sampling technique was used to quantify the impacts of various sources of risk and uncertainty on profitability and acceptability of the investment. Sources of risk and uncertainty were ranked from most important to least important based upon standardized beta coefficients which were generated by the risk analysis software.

Modeled sources of risk and uncertainty related to physical structures were annual repair and maintenance expenditures and salvage value. Information on salvage value was critical to computation of net present value and modified internal rate of return. Modeled

sources of risk and uncertainty related to fish growth were stocking weight of fingerlings, survival rate, and average daily rate of gain. Other modeled sources of risk and uncertainty included per unit feed cost and feed conversion ratio; per unit prices of LP gas, water, oxygen, and electricity; the hourly labor wage rate; and the price of fingerlings.

Based upon the assumption that 46,800 fingerlings weighing approximately 20 g would be stocked each year, a simulation based upon 1000 iterations generated the following results for output cells:

Output	Mean	Standard deviation
Net profit after taxes (annual)	$15,430	$5,554
Net present value (WACC = 14%)	($25,633)	$19,536
Modified internal rate of return	10.0%	0.3%
Breakeven volume – dollar sales	$88,309	$4,127.87
Breakeven volume – kilograms sold	21,145	1,665

The net present value (NPV) and modified internal rate of return (MIRR) analyses were based upon a five-year planning horizon. Results indicated that under the assumptions utilized, the investment could be expected to generate a 10 percent rate of return after taxes. The minimum MIRR generated by the simulation was –1.5 percent, and the maximum value generated was 18.6 percent.

Sensitivity analysis was utilized to determine which sources of risk and uncertainty had the strongest impacts on output values. The five most important sources of variation for net profit after taxes, NPV, and MIRR, were price per kg of fish at harvest, average daily rate of gain, feed conversion ratio, survival rate, and cost of feed per kg. The five most important sources of variation for breakeven volume measured in dollars of sales were feed conversion ratio, price per kg of fish at harvest, hourly labor wage rate, cost of feed per kg, and fish survival rate. The five most important sources of variation for breakeven volume measured in kg of sales were price per kg of fish at harvest, feed conversion ratio, hourly labor wage rate, cost of feed per kg, and fish survival rate.

With slight modifications, the spreadsheet can be used to analyze recirculating aquaculture systems of different scale and technical makeup. Although the analysis described above did not include a true bioeconomic model, it is possible to incorporate such information into a spreadsheet at the cost of greater complexity.

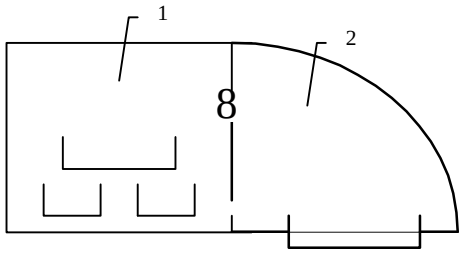
The Effect of Biological Air Purifying System With Aquatic Animal-Plant Integrated Greenhouse

S. R. Cui Professor
X. W. Miao Professor
B. H. Shao Engineer
L. X. Fu Associate Professor

Zhejiang Agricultural University, P.R. China

Abstract

Two different types of the joint greenhouses for aquatic animal-plant integrated system were developed at the Agricultural Bioenvironment Engineering Institute of Zhejiang Agricultural University in China. The joint greenhouses form mutual compensation system for exchange CO_2 and O_2 between the aquatic-animal housing and plant-growing greenhouse. The CO_2 from aquatic-animal housing can be used as a nutrient resource for the vegetable or flower inside the greenhouse, the plant photosynthesis product O_2 will purify and improve the air quality inside the animal housing. One of these is the back to back' interaction greenhouses (Fig1),

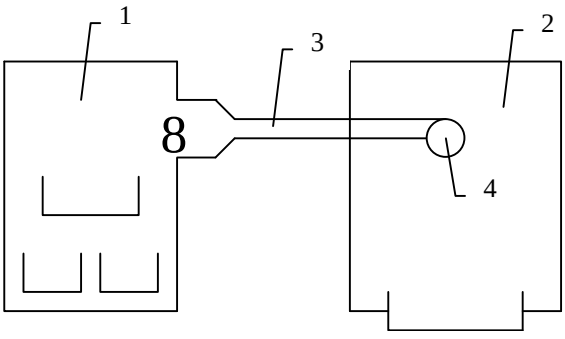


1. aquactic-animal housing
2. plant greenhouse

Fig. 1 Back to back interaction greenhouses system.

Dynamic variations of CO_2 concentration related to operating time was measured and measured with LI 6200 portable photosynthesis system.

Another type of interaction greenhouses is connected with pipe (Fig. 2)



1. aquactic-animal housing
2. plant greenhouse
3. Conecting pipe
4. Pipe with holes

[1]Fig. 2 The interaction greenhouses system with pipe connecting

Dynamic variation of CO_2 concentration was measured too and the simulated equation of CO_2 variation of aquatic-animal housing is as follows:

$$C_1 = 807 + 260e^{-\frac{t}{5000}}$$

C_1: The CO_2 concentration variation related to time of aquatic-animal housing

The simulated equation of plant greenhouse is as follows:

C2: The CO_2 concentration variation related to time of plant greenhouse

$$C_1 = 681 + 358e^{-\frac{t}{5000}} - 632e^{-\frac{t}{1366}}$$

The data indicates that when the ventilation system for air exchange is operating, for the back to back' interaction greenhouse system, the mean value of concentration of CO_2 inside plant greenhouse increase from 320ppm to 780ppm after half hour and the mean value of concentration for CO_2 inside aquatic-animal housing decrease from 1100ppm to 800ppm. For the interaction greenhouses system with pipe connecting, the mean value of concentration of CO_2 inside plant greenhouse increase from 360ppm to 780ppm and the mean value of concentration for CO_2 inside aquatic-animal housing decrease form 1100ppm to 700ppm after one hour, the measure results indicated that the hole position of the distribution pipe is one of the important factors of CO_2 concentration equality, this paper proposed some principles of air exchange system design.

The measure results indicated that the interaction greenhouses of aquatic-animal plant integrated system is an efficient method for purifying air in aquatic-animal housing, and it is used for not only purifying air, but also water recycling and economizing energy.

Integrating Hydroponic Plant Production with Recirculating System Aquaculture: Some Factors to Consider

James E. Rakocy
University of the Virgin Islands
Agricultural Experiment Station

Abstract

Aquaponics, the integration of hydroponic plant production with recirculating fish culture systems, is gaining in popularity among hobbyists and receiving attention in the commercial sector. Although the number of commercial operations is still small, at least two large suppliers of aquaculture and/or hydroponic equipment have introduced aquaponic systems in their catalogs and many schools are beginning to include aquaponics as a learning tool in their science curricula. With such high interest, aquaponics appears to be on the threshold of increased technological development and greater application.

Growing plants with fish requires some modification of the design criteria and operational procedures of standard recirculating aquaculture systems. Properly-designed aquaponic systems emphasize plant production, which receives about 90% of the culture area and generates 65-70% of the revenues. Raising and marketing plants require skills that may be unfamiliar to aquaculturists. Other factors to consider in adopting aquaponic technology involve:

Aeration. Plant roots require DO levels that are comparable those required by fish. DO levels should remain high throughout the hydroponic component and throughout the each mass of densely packed roots. Formation of anaerobic zones will cause root death. Rapid water exchange, intermittent dewatering or induced turbulence are used to ensure adequate root aeration.

Solids removal. Rapid removal of settleable and suspended solids from an aquaponic system reduces mineralization, which may decrease some dissolved nutrients to levels that limit plant growth. Aquaponic systems vary from daily or weekly removal of solids to no solids removal at all in some configurations.

Biofiltration. In a properly-designed aquaponic system with a high ratio of plant growing area to surface area for fish production, the hydroponic component should provide sufficient biofiltration, through direct ammonia uptake by the plants and nitrification on the submerged surface areas, so that a separate biofilter is not required. This is an important economic justification for integrating hydroponic plant production in a recirculating aquaculture system. Hydroponic technique. The hydroponic methods that are generally used in aquaponic systems are raft culture, nutrient film technique and ebb and flow systems with either gravel or sand

substrates. Each method requires special design considerations and has advantages or disadvantages depending on the plant crop being cultured.

Suitable fish species. Tilapia are the most common species cultured in aquaponic systems. Not all fish species are suitable. For example, hybrid stripped bass cannot tolerate the elevated levels of potassium, produced through supplementation, that are desirable for rapid plant growth.

Planting densities. It is essential that plants be given adequate space for robust growth. If plant densities are too high, plants will elongate, reducing market value, and lack of air circulation and moisture buildup will foster disease and pests outbreaks.

Stocking rates. Since aquaponic systems emphasize plant production and the fish rearing component is relatively small compared to the plants, stocking fish at extremely high densities and using pure oxygen systems are not practical or cost effective. Fish should be stocked at less than 100/cubic meter to increase individual growth rates, lower feed conversion and promote health.

Water source and exchange. The nutrients or contaminants in source water can impact plant growth and should be assessed. Water exchange should be minimized to maximize nutrient retention time in the system.

Base and nutrient addition. Potassium and calcium bases will neutralize acidity as well as supplement essential plant nutrients that are low in fish waste.

Nutrient and gas recovery. Hydroponic plants can not only recover waste nutrients from fish but also carbon dioxide that is sparged from the culture water and contained by a greenhouse.

Nitrate control. A partially anaerobic solids zone can be used to regulate nitrate levels by adjusting the cleaning frequency. High nitrate levels favor vegetative growth in plants while low nitrate levels promote fruiting.

pH. pH should be maintained at 7.0 for optimum nutrient availability while maintaining adequate nitrification.

Temperature. The optimum temperature is 24C for vegetable production and 30C for tilapia production. If possible, the temperature should be adjusted to favor plant production.

Solar radiation. Commercial hydroponic plants grow best in intense sunlight and should not be shaded. Low wintertime solar radiation in temperate regions will prolong plant production cycles.

Pest and disease control. The need for pest and disease control will be reduced by providing growing conditions that minimize stress. Biological control methods are appropriate for

aquaponic systems. Pesticides and antibiotics should not be used.

Monitoring. At prescribed stocking, planting and feeding rates, aquaponic systems require a minimum amount of monitoring. Only pH is monitored on a regular basis.

Management and labor. Aquaponic systems should be designed for ease of management and should minimize labor requirements. For profitability it is important to reduce operating expenses.

Ground Limestone as a Biofilter Media for Hybrid Stripped Bass Culture

Dale C. HoneyfieldDale C. Honeyfield
U.S. Geological SurveyU.S. Geological Survey
Biological Resources Division
Research and Development Laboratory
R.D. # 4, Box 63
Wellsboro, Pennsylvania 16901Wellsboro, Pennsylvania 16901

A pilot study was conducted to evaluate ground limestone as a biofilter media in a recirculation system containing hybrid striped bass. Two replicated biofilter configurations were used. Each system consisted of a rearing tank (122 cm diameter, 67 cm deep), settling tank (102 cm square, 21 cm deep), biofilter and re-oxygenation column (76 cm height, 15 cm diameter) containing Koch rings. The biofilter type and media consisted of either a pulsed bed near buoyancy plastic bead filter or a pulsed bed ground limestone filter. In the pulsed bed limestone and plastic bead filters, 3 kg of media were placed in each column (16 cm diameter, 260 cm height) with two columns per system. A timer-relay-ball valve assembly directed 22.7 l/min of water flow through each column at 30 s intervals. The intermittent flow expanded and mixed the bed during the 30 s of flow, shearing excess microbial growth. Estimated surface area per square meter of each material was, near buoyancy beads, 960 m^2/m^3 and limestone, 4000 m^2/m^3. Temperature was set at 23 C, water hardness adjusted to 150 ppm with calcium chloride and pH maintained above 6.5 by the addition of saturated lime solution. Each unit was stocked with 24 kg of hybrid striped bass averaging 397 g/fish. Fish were hand fed commercial feed twice daily to satiation. The effect of limestone on rearing unit pH and alkalinity was measured independently from ammonia removal rates to avoid any effect pH change may have on ammonia removal rates. During this period lime dosing was shut off. Ammonia removal rate based on a surface area basis was higher for the pulsed bead filter. When ammonia removal rate was calculated as per unit of bed volume, ammonia removal rate was higher for the limestone filter and the efficiency of ammonia removal was also higher with the limestone biofilter. Biological oxygen demand (BOD) measured in the rearing unit was similar between the two filter types. Suspended solids were higher in the limestone treatment. In addition to providing surface area for nitrifying organisms, limestone increased both pH and alkalinity of the water (Table 1). Use of limestone as a biofilter media resulted in less saturated lime solution used to maintain rearing water pH. After 2 months of operation, the limestone bed height had dropped by half and new limestone was added to return bed volume to original depth. Limestone media provided a reasonable nitrification rate and improved both pH and alkalinity but the rearing tank water remained cloudy or milky throughout the study. This turbidity was not due to the addition of lime to control pH. After three months, fish mortality (1-2 fish/day) was observed only in tanks with limestone filters. Microscopic inspection of the gill tissue from dead fish revealed small imbedded limestone fragments. In addition, it was discovered that limestone silt had trapped organic matter at a low point in the system and the trapped material became anaerobic. In summary, limestone supported reasonable nitrification rates, elevated pH and increased alkalinity of the water. The disadvantages include higher suspended solids,

gill irritation and a propensity of limestone silt to trap organic material resulting noxious undesirable by-products. Limestone has potential as a biological filter media if suspended limestone is removed and care is taken to avoid anaerobic pockets within the system.

Table 1. Biofilter ammonia removal and rearing tank water chemistry.

Measurement	Plastic Bead	Limestone
NH_4 removal rate, g/m^2 day (surface area)	0.41	0.30
NH_4 removal rate, g/m^3 day (bed volume)	0.39	1.47
NH_4 removal efficiency %	29.3	35.0
Rearing tank NH_4, ppm	0.38	0.46
Rearing tank BOD, ppm	4.02	4.26
Rearing tank Suspended Solids, ppm	2.4	6.0
Water pH and alkalinity with no base addition (CaOH)		
Rearing tank pH	5.6	6.2
Rearing tank alkalinity, ppm	27.9	49.0

Evaluation of an Aerated Floating Plastic Media Biofilter Within a Recirculating System Used to Produce Food-Size Yellow Perch

Justin M. Balter and Craig Hall
Shandaggan Farm, Inc.
68 Limber Road
Jackson Center, PA 16133

Glenn Snapp
Water Management Technologies
PO Box 66125
Baton Route, LA 70896

Steven T. Summerfelt
Conversation Fund Freshwater Institute
PO Box 1746
Shepherdstown, WV 25443

A moving bed biofilter consisting of aerated floating plastic media (Purac Engineering, Inc.) was installed within a recirculating system used to produce food-size yellow perch. The ridged tubular media is 7-8 mm long, 10 mm in diameter, and provides roughly 98% void space as a loose-packed bed. The manufacturer reports that the media has roughly 333 m^2/m^3 of surface area when filling 67% of reactor volume. For the present study, about 4.25 m^3 (150 ft^3) of "unwetted" media was divided into two equal portions and added to the two biofilter vessels (each 3.66 m diameter and 1.22 m sidewall depth). Each biofilter was operated at 0.95 m (3.1 ft) water depth. About 40.5 L/s (640gpm) of water flow and 20.4 L/s (43 cfm) of diffused air was split between the two biofilters, producing a volumetric gas:liquid ration of 0.5:1. The water flow was first filtered through a single drum filter (Hydrotech) operated with 40 μm sieve panels before entering the two biofilters.

The alkalinity of the system was maintained at or above 60 mg/L (as calcium carbonate) by addition of sodium hydroxide.

The water pH and TSS concentration were taken at least once per week. Ammonia, nitrite, oxygen, and carbon dioxide concentrations were measured before and after each biofilter three times a week. This data was used to determine the ammonia, nitrite, and carbon dioxide removal rates and efficiencies across the moving bed biofilter. The arial and volumetric nitrification rate of the media are reported and compared to rates reported for other aquaculture biofilter types.

Prediction and NMR Determination of Fluid Film Thickness and Velocity Distribution in Nitrifying Trickling Filters

Valdis Krumins

Biological trickling filters are a common means of removing nutrients from recirculating aquaculture systems. Design of trickling filters is currently based on rules-of-thumb or empirical loading curves. More rigorous design requires accurate descriptions of such parameters as wetted area, mass transport into and out of the biofilm, and shear forces on the film. These parameters are functions of fluid flow rates; therefore, knowing the distribution of flow rates would enable one to model these phenomena for the entire filter. Such models could then be used to optimize several aspects of trickling filter design, including packing media and irrigation rates.

In this work, nuclear magnetic resonance (NMR) is used to image miniature nitrifying trickling filters. The NMR images show the location of biofilm, packing media, and water, and also display the velocity of any flow. Therefore, they can be used to compute the distribution of flow rates and to observe any changes in flow patterns caused by the presence of an active biofilm.

In addition, a probability density function (PDF) of local flow rates is independently developed using only the irrigation rate, packing material properties, and fluid properties as inputs. Preliminary results show good agreement between the NMR images and the theoretically determined PDF.

The Chilean Aquaculture Industry and the Role Played By the Universidad Catolica Del Norte in Its Development

German E. Merino
Universidad Catolica del Norte, Depto
de Acuacultura
P.O.Box 117
Coquimbo, Chile

Introduction

The explosive growth of the aquaculture industry in Chile over the past decade would not have been possible if the country did not have good natural conditions with extensive coastal areas, lakes and rivers that provide optimal environmental conditions for farming, and also the existence of technicians and professionals able to meet the industry demands. Also, the existence of high quality fishmeal produced in Chile provides enough feed production to support the aquaculture activities.

The present work presents a brief description of the most important aspects of the species farmed in the country and the role played by the Universidad Católica del Norte in its development.

Farming of Endemic Species

Scallop farming.

In 1983 Trench et al. and in 1984 Disalvo et al. published the main data and technique culture about scallop farming, used until now. With the development of the "Centro de Acuacultura e Investigaciones Marinas" , a donation from the Japan International Cooperation Agency (JICA) to the Universidad Católica del Norte in 1985 began the development at commercial scale of scallops, and the hatchery techniques, natural larvae collection and ongrowing systems were then established (Illanes, 1988; DiSalvo, 1988).

There are currently 27 companies operating with investments of more than US$40 million that produced 8264 t in 1995 (SERNAP, 1995), this production placing Chile as the third scallop producer in the world. The average nominal price over the past 5 years has been US$13.3 per kg increasing to US$ 14.9 per kg in the case of fresh chilled scallops (Anonymous, 1997).

Seaweed Farming

Main seaweed farmed is *Gracilaria* spp., and there are 268 authorized seaweed farming concessions, with a 75% of them located in the south and 12% in the north of Chile.

The regular FOB prices were for dry seaweed US$1.9 per kg, but at present due to the unstable Asian markets and a strong decrease in the prices stopped all the exportation and only a few farms continue operating, to supply the local demand only.

Farming of Exotic Species

Salmon and trout farming

The introduction of salmonid species in Chile with the idea of running ocean-ranching began between 1905 and 1910 with the importation of 400.000 eggs of S*almo salar, Salmo trutta* and *Oncorhynchus mykiss*. Between 1924 and 1930 200.000 eggs of *Oncorhynchus tschawytscha,* 114.000 eggs of *O.nerka*, and 225.000 eggs of *O. kisutch* were introduced. Only in 1979 the first cage culture system was established and in 1986 it reached a production level of 900 t (Uribe, 1988), and 141.400 t in 1995 (Anonymous, 1997). Due to the location of the Universidad Católica del Norte in northern Chile, no research has been done in salmonids, that are grown in southern Chile.

There are currently 90 companies in Chile oriented to salmon and trout farming, which have approximately 361 farming authorized concessions, and the salmonids produced are exported to Japan (59.6%), USA (29.8%) and the European Community (6.3%).

Red abalone farming

The red abalone *Haliotis rufesces* was introduced by Universidad Católica del Norte (UCN) and Fundación Chile from California in the'80. There are at least 4 abalone ongrowing farming centers and three hatcheries, one of them belonging to the UCN.

Due to the Chilean Aquaculture and Fisheries Law, this species must be cultured in inland tank systems in northern Chile, but in the South its culture in long-lines systems was permitted.

Japanese abalone

This species was introduced by Universidad Católica del Norte and JICA at the end of the '80, and after six years of an important amount of research in artificial feed, aquaculture engineering, cost analysis, and other bioengineering factors, the UCN and JICA designed hatchery facilities with a production capacity of 500.000 seed per year, and all this seed will be ongrowing by a national corporation with an investment capital of US$1.000.000.

Oyster Farming

The first assay of Pacific Oyster (*Crassostrea gigas*) culture was made in 1978 by the Universidad Católica del Norte and Fundación Chile, with 20.000 seeds (2 at 5 mm) imported from Moss Landing (California, USA) (Hauer, 1988). From that starting point it has moved geographically to the South of Chile, where more than 94% of the total production is located (Anonymous, 1997).

There are 15 companies (individuals and trade unions) authorized to farm oysters, with a production of 1.130 t in 1995. Prices for fresh-chilled oysters have shown a rising trend reaching US$ 6.9 per kg in 1995 and US$ 5.97 per kg for frozen oyster (SERNAP, 1995).

One of the main problems in oyster farming is the impossibility of natural breeding because of the low temperature of the Chilean seawater, therefore, its production is highly dependent on the existence of two hatcheries located in Coquimbo area, which can produce the required seeds.

Australian prawn farming

The Australian prawn *Cherax tenuimanus* was introduced at the end of 80 decade by UCN and national business people, and reactivate this activity just this year through National Grant, that permit the development of a model farm and hatchery facility with the idea of start a new interesting aquacultural business.

Potential Species in Study at UCN

From 1986 the UCN was studying the biological aspects of three local species with a great potential for national and international markets, and they are:

Chilean flounder: (*Paralichtys* sp.) until now the UCN has achieved spawning in captivity and culture of larvae descending from these broodstock. Right now the UCN will be increasing the laboratory capacity to pilot scale with an initial production of 3.000 juveniles for 1998 and 10.000 juveniles for 1999. During this program studies on hatchery and nursery facilities, transport , and sea cage design systems will be carried out.

Chilean river prawn: *Cryphiops caementarius* is the only Paleomonidae specie in Chile, and since 1983 the UCN was studying adequate technology to develop a hatchery and ongrowing facility system (Rivera et al., 1983). Only last year Morales (1997) defined the hatchery techniques to reach the postlarval stages in only 45 days. With these results the possibility to continue building new ongrowing earth ponds will be open, because up to this discovery the only source for prawn seeds were the rivers, which provide limited supplies.

Sea urchin farming: the UCN has currently developed the technology for farming and stocking the local edible sea urchin (*Loxechinus albus*), with a larval phase of 20 days and survival rates over 30%, then they are cultured on policarbonate plates in race-way tanks of 10 t until 4 mm, then they are relocated in baskest hanging from long-line systems and ongrowimg there during 8 months (20 mm), and then they are ready for stocking natural areas (Guisado et al., 1988).

References

Anonymous, 1997. Compendium of Chilean aquaculture 1997. Technopress S.A., Editorial Antartica S.A., Santiago, Chile. 260 pp.

DiSalvo, L., E. Alarcón, E. Martinez & E. Uribe, 1984. Progress in mass culture of *Chlamys (Argopecten) purpurata*. Rev. Chilena de Hist. Nat., 57 :35-45.

DiSalvo, L., 1988. Cultivos de ostiones en sistemas de hatchery. In : Producción de larvas y juveniles de especies marinas, 111-112 pp. E. Uribe (Ed.). Universidad Católica del Norte, Coquimbo, Chile.

Guisado, C., V. Alcazar & J. Castilla, 1988. . In : Producción de larvas y juveniles de especies marinas, 113-114 pp. E. Uribe (Ed.). Universidad Católica del Norte, Coquimbo, Chile.

Hauer, R., 1988. Experiencia de producción de ostras individuales *Crassostrea gigas* en laboratorio. In : Producción de larvas y juveniles de especies marinas,115-118 pp. E. Uribe (Ed.). Universidad Católica del Norte, Coquimbo, Chile.

Illanes, J., 1988. Experiencias de captación de larvas de ostión (*Argopecten purpuratus*) en Chile, IV Región. In : Producción de larvas y juveniles de especies marinas,53-58 pp. E. Uribe (Ed.). Universidad Católica del Norte, Coquimbo, Chile.

Morales, M., 1997.

Rivera, M., P. Schmiede & J. Meruane, 1983. Desarrollo larval del camarón de rio del Norte *Cryphiops caementarius* (Molina, 1782) (Crustacea: Palaemonidae), en condiciones de laboratorio. 315-334. In: Symposium Internacional Avances y Perspectivas de la Acuacultura en Chile. H. Fuentes, J. Castillo & L. DiSalvo (Eds.). Universidad del Norte, Coquimbo, Chile, Septiembre 1983.

SERNAP, 1995. Anuario Estadístico

Trench, B., E. Alarcón & L. DiSalvo, 1983. Comparative evaluation of potential growout systems for *Chlamys purpurata* in Herradura Bay, Coquimbo, Chile. 255-268. In: Symposium Internacional Avances y Perspectivas de la Acuacultura en Chile. H. Fuentes, J. Castillo & L. DiSalvo (Eds.). Universidad del Norte, Coquimbo, Chile, Septiembre 1983.

Uribe, J., 1988. Producción de alevines de salmón. In : Producción de larvas y juveniles de especies marinas, 67-94 pp. E. Uribe (Ed.). Universidad Católica del Norte, Coquimbo, Chile.

Determination of the Primary Ammonia Removal Design Criteria for Biological Filters

Catherine A. Wells and Fredrick W. Wheaton
Department of Biological Resources Engineering
University of Maryland at College Park
cwells@wam.umd.edu

Abstract

Ammonia is one of the primary waste materials produced by aquatic species, which is toxic even in low concentrations. Therefore, recirculating aquaculture systems must have some means for ammonia removal. Thus, systems commonly employ nitrifying biological filters to convert ammonia to nitrite and thence to nitrate which is itself non-toxic even in high concentrations. This research investigated whether the ammonia inflow concentration or the ammonia loading rate to the biofilter was the more important biofilter design parameter. Five ammonia inflow concentrations and ammonia loading rates ranging from 0.5 – 8.57 mg NH_4/L and 109 – 2456 mg NH_4/L per filter per day were applied to each of five identical biodrums through a series of ten tests. Total ammonia removal and nitrite levels were monitored in each of the filters. Ammonia inflow concentrations, ammonia loading rates, temperature pH, dissolved oxygen levels and flow rates to the filters were measured and controlled by a semi-automated, synthetic wastewater system. Regression analysis yielded a relationship of ammonia removal versus ammonia inflow concentration and ammonia loading rate (R^2 = 0.93), implying that the interplay of both parameters is necessary in predicting biofilter performance. Percentage ammonia removed was inversely and linearly proportional to loading rate. Increase in flow rate, hence loading rate, under low ammonia concentration in the system, increased mass ammonia removal.

In-Situ Passive Waste Removal in Circular Fish Culture Tanks

J. D. McMillian, F. W. Wheaton, and J.M. Ebeling
Biological Resources Engineering Dept.
University of Maryland

A new approach to removing solid waste from recirculating aquaculture tanks was demonstrated, that incorporated part of the culture tank as the settling basin. Tank modifications included a dual standpipe design with an orifice to control discharge and the addition of a false bottom plate to form a quiescent area for solids settling. Three false bottoms and one control (no false bottom) unit were designed and tested in a 1 m^3 circular domed bottom fiberglass tank. The hypothesis that the standpipe/false bottom modifications would remove more waste from the water column than the standpipe modifications alone was tested. The hydraulic characteristics of the tank were determined using tracer studies. A completely mixed flow was present with only one of the false bottom designs, while the other three designs approached completely mixed flow. In addition, the effect of the false bottom designs on the velocity profile of the tank was analyzed. Results of the solids analysis indicated that this method of solids filtration is commensurate with other solid filtration units currently used in production scale recirculation systems. However the benefit from using the false bottoms in terms of solids removal (the reduced floor space) was outweighed by the extra amount of labor needed to maintain the tank and reduced tank circulation. Based on all the results, the control tank design is suitable for production if used as a primary filter for solids removal.

A Prototype Tilapia/Hydroponic Greenhouse Recirculating Production System for Institutional Application

Scott H. Newton and Jimmy Mullins
Virginia State University
Petersburg, VA 23806

George S. Libey and Mark Kidd
VA Polytechnic Institute and State Univ.
Blacksburg, VA 24061

A recirculating tank production system was designed and tested during 1996 and 1997 under greenhouse conditions for the dual purpose of producing fish and plants for human consumption, principally for application in correctional institutions. Overall size of the rectangular tank was 7.3 m long by 2.4 m wide by 1.5 m tall with a total water capacity of approximately 18,190 l in the entire system. The tank was divided into three sections: a fish growing section (approx. 13,650 l), a sump area, and a biofilter area - each approximately 2,270 l. The fish growth area was subdivided using **2** inch diameter plastic mesh material into a primary area (9,100 l) and a secondary area (4,550 l). Water circulation was provided by 5-two inch airlift pumps located between the sump area and the biofilter. Water flowed by gravity from the biofilter, through the grow tank, and through the sump area, returning to the air lift pumps. The circulation rate was approximately 475 lpm.

Similar tank systems were installed in greenhouses located in Blacksburg and Petersburg, Virginia in 1996. The first season of testing was May - November and included mechanical aspects of tank system physical functions, water use and water quality monitoring, fish production and management, and use of effluents for irrigation and plant production. Prior to the 1997 season, several engineering design changes were made to the tank systems, as well as biological management changes for fish and plant production.

Application goals for this greenhouse recirculating system were to produce a crop of food fish every 5 to 6 months and to reuse the effluent water for both irrigation and nutrient inputs into greenhouse vegetable crops. Tilapia was used in theses tests, while vegetables tested included tomatoes and lettuce. Lettuce was grown using a nutrient film technique system over the fish tanks.

Fish Production

Nine hundred tilapia, average weight 45 g, were stocked in the primary production area during May, 1997. Fish were fed frequently by hand and with automatic (belt) feeders. Water temperature, dissolved oxygen, and pH were monitored twice daily. Approximately 2.35 kgs of sodium bicarbonate was added weekly to maintain alkalinity and pH levels in the system. Routine maintenance involved weekly sump flushes. Fish harvests were made during October. For 1997 the combined production data from Virginia Tech and VSU was: fish survival was 92% with an average size 439 g. Total fish harvested was approximately 340

kgs with an additional harvest of 700 fish that averaged 124 g each from the secondary growth area.

Effluent Uses

After 2 months of operation, approximately 3,500 l of water had been released and replaced with new water to maintain water quality levels for fish production. Tank water was tested for plant application as a nutrient source. Lettuce grown using tank effluent performed similarly to control lettuce until it reached 110 g size, when signs of nutrient deficiencies were indicated. Nutrients in the effluent water were inadequate at the fish densities tested for commercial vegetable hydroponic systems.

The Speedy Text to Identify Optimal Growth Temperature for Aquatic-Animals

X. W. Miao, Professor
K. X. Zheng, Professor
X. Y. Jing, Research Assistant
Zhejiang Agricultural University, P.R. China

Abstract

The optimal growth temperature for aquatic animals, such as soft-shell turtles is affected by natural environment or experimental conditions. When water temperature is controlled above 24°C~25°C, the soft-shell turtles look very active. However, if the temperature falls below 20°C, the food intake of soft-shell turtles is significantly reduced. The heating facilities are widely applied to cultivate soft-shell turtles to dispel the hibernation period and the turtles will gain weight all year long. We have studied effect of water temperature on turtle's heart rate. The aim of this study is to provide certain scientific guidelines for fast cultivation of soft-shell turtle by investigating effects of water temperature on its physiological mechanism.

Experiments were performed in the ponds of the Agricultural Bio-environment Engineering Institute, Zhejiang Agricultural University. The experimental subjects were healthy adult soft-shell turtles. Their weight varied in the range of 226~280g (253g on average). Heart rate was taken by pH-38 ECG recorder. Water bath was applied to control water temperature, recording electrodes were well attached to four legs of turtle. To simulate temperature change in Zhejiang Province, the test temperature was controlled between 5~35°C and measurement was taken every 5°C.

In order to find consistent experimental conditions, Turtle was used to characterize an adaptive process of heart rate to the change of water temperature.

Experiment data show that the heart rate was constant after the turtle was kept in water for 40 minutes with the temperature unchanged. The heart rate was measured as a subject had been kept in one chosen temperature for 40 minutes.

The variation of heart rate as a function of temperature is plotted in Figure 1.

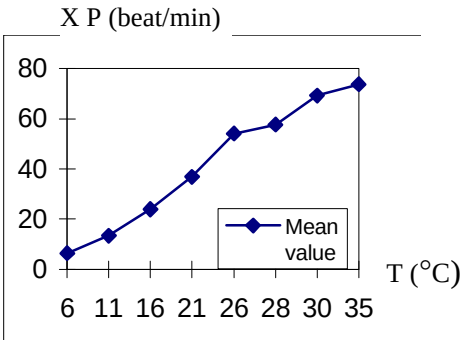


Fig. 1 The relationship between the mean value of heart rate and the water temperature

The results provide the clear evidence of that the heart rate of turtle is linearly proportional to water temperature:

$P=-16.429+2.737T$ $(P>0)$

$R^2=0.9812$

P: heart rate (beat/min)

T: water temperature (°C)

Mean value:

$$\bar{X}=\frac{\Sigma_{Xi}}{n}$$

The relationship between the sum of square and temperature is showed in fig 2.

The relationship between the coefficient of variation and temperature is shown in fig 3.

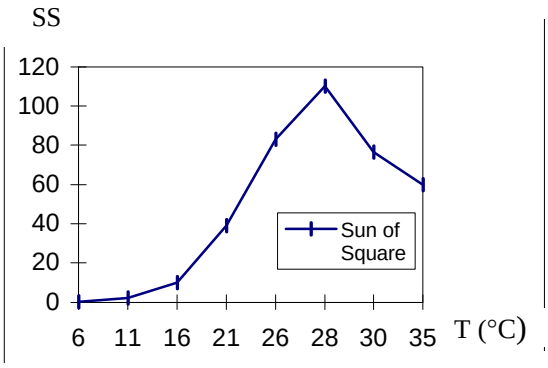


Fig. 2 The relationship between the sum of square and water
SS—Sum of Square
$SS=\Sigma(Xi-X)^2$

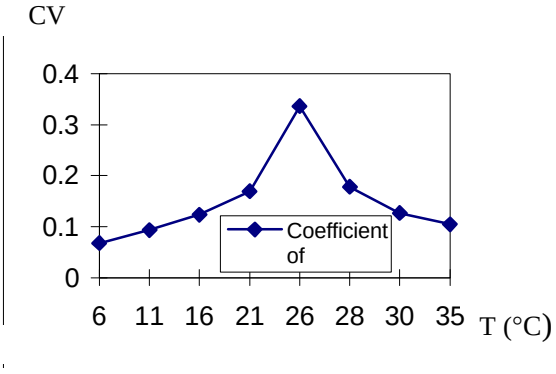


Fig. 3 The relationship between the coefficient of variation and water temperature
CV—Coefficient of Variation

$$CV=\frac{\sum (Xi-\bar{X})}{\bar{X}}$$

Through statistical analysis the heart rate of aquatic-animal is closely indicates a protective mechanism for animals to adjust their breath, digestion, and metabolism to the change of environment.

Nevertheless, the data indicates that heart rates have the largest variation as the temperature falls between 26°C to 30°C. Many experiments suggest that this temperature range is optimal for soft-shell turtle growth in the sense of large food intake, rapid weight gain. In this optimal growth temperature range, the heart rate variation of soft-shell turtle

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is not only affected by physiological mechanism, but also its behavior such as food intake and activity. That is a part of the reason to comply with the large variation of results in that temperature range, the conclusion is as follows:

Heart rate is linearly proportional to the temperature:

$$P=-16.492+2.737T \ (P>0)$$

In the optimal growth range (26°C ~30°C), relationship of heart rate and temperature presents large variations. Similar to homoiothermous animals, heart rate is affected by animal behavior in the optimal growth temperature.

To identify the optimal growth temperature for other types of poikilothermal animals, our study suggested measure heart rate as a function of temperature. The temperature that shows large variation may serve as the best cultivated range. Heart rate-temperature measurement will provide a quantitative method to quickly identify the optimal growth range for poikilothermal animals.