

Biological Filters: Trickling and RBC Design

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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 nitrite 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 bacteria 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 *nitrosomonas* 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 systems (urea, amino acids and uric acid) are 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. 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 referred 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:



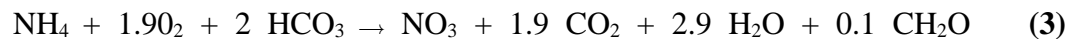
The reactions shown in Equations 1 and 2 release energy that is used by *Nitrosomonas* and *Nitrobacter* for cellular growth and maintenance. Oxygen serves as an electron acceptor and is the only electron acceptor that can be used by *Nitrosomonas* and *Nitrobacter* (Water Pollution Control Federation,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



where CH₂O 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 literally 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 (RBCs) 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. RBCs 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 fluidize 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 ($\text{m}^3/\text{m}^2\cdot\text{d}$). 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 $29 \text{ m}^3/\text{m}^2\cdot\text{d}$ (Norton Actifil). Roberts (1985) reported minimum hydraulic loadings of 32 to $55 \text{ m}^3/\text{m}^2\cdot\text{d}$ for random packed media (plastic pall rings). For design purposes, a MWR of $50 \text{ m}^3/\text{m}^2\cdot\text{d}$ 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 $\text{m}^3/\text{m}^2\cdot\text{d}$ for randomly packed plastic media (plastic pall rings). Grady and Lim (1980) reported a UIR range of 234 to 350 $\text{m}^3/\text{m}^2\cdot\text{d}$ (Dow Surfpac). A design UIR of 300 $\text{m}^3/\text{m}^2\cdot\text{d}$ 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 $\text{m}^3/\text{m}^2\cdot\text{d}$) 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 of ammonia and nitrite removal 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/m³; (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
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Type	plastic rings
Diameter	2.5 cm (1 inch)
Void Fraction	0.92
Specific Surface Area	220 m ² /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/m³, the total volume of water required is:

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

$$\text{Volume}_{\text{Water}} = 75.7 \text{ m}^3$$

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:

$$\text{Mass}_{\text{Feed}} = 9,072 \text{ kg of fish} \times 0.02 \text{ kg /kg of fish}$$

$$\text{Mass}_{\text{Feed}} = 181 \text{ kg}$$

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:

$$\text{Oxygen} = 0.25 \text{ kg O}_2 / \text{kg feed} \times 181 \text{ kg feed/day}$$

$$\text{Oxygen} = 45.3 \text{ kg O}_2 / \text{day}$$

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 , $\text{NH}_3\text{-N}$, or un-ionized ammonia) and ammonium (often reported as NH_4 , $\text{NH}_4\text{-N}$, 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:

$$\begin{aligned}\text{TAN} &= 181 \text{ kg feed/day} \times 0.03 \text{ kg TAN/kg feed} \\ \text{TAN} &= 5.4 \text{ kg TAN/day}\end{aligned}$$

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):

$$\begin{aligned}[\text{TAN}] &= (5.4 \text{ kg TAN/day} \times 1000 \text{ g/kg}) \div (24 \text{ hours/day} \times 75.7 \text{ m}^3 \times 2 \text{ exchanges/hr}) \\ [\text{TAN}] &= 1.5 \text{ mg/L}\end{aligned}$$

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/m² · 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:

$$\text{Area}_{\text{Filter}} = 5.4 \text{ kg TAN/d} \times 1000 \text{ g/kg} \div 1 \text{ g TAN/m}^2 \cdot \text{d}$$

$$\text{Area}_{\text{Filter}} = 5,400 \text{ m}^2$$

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:

$$\text{Volume}_{\text{Media}} = 5,400 \text{ m}^2 \div 220 \text{ m}^3/\text{m}^2$$

$$\text{Volume}_{\text{Media}} = 24.6 \text{ m}^3$$

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 $30 \text{ m}^3/\text{m}^2 \cdot \text{d}$ and a maximum hydraulic loading of $225 \text{ m}^3/\text{m}^2 \cdot \text{d}$ 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:

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

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

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

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

$$\text{Flow Rate}_{\text{Filter}} = 605.6 \text{ m}^3/\text{d}$$

The volume needed for each filter unit is:

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

$$\text{Volume}_{\text{Unit}} = 4.1 \text{ m}^3$$

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

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

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

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

$$\text{Diameter}^2 \geq (4 \times \text{Area}_{\text{Cross-Sectional}}) \div \pi$$

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

$$\text{Diameter} \geq 1.85 \text{ m}$$

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

$$\text{Height} = \text{Volume}_{\text{Unit}} \div \text{Area}_{\text{Cross-Sectional}}$$

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

$$\text{Height} = 1.31 \text{ m}$$

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 RBCs 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 RBCs.

Because an RBC rotates through both an aqueous 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 reemerging into the air. Similarly 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 RBCs. 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 RBCs 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 RBCs were used for COD (Chemical Oxygen Demand) removal, the removal of soluble 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 RBCs at 3 RPM (peripheral velocity of approximately 0.94 ft/sec). Gilbert et al. (1986) found in commercial installations most RBCs 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 RBCs rotational speed increases (Fujie et al., 1983; Gilbert et al., 1986). Gilbert et al. (1986) surveyed 29 sewage treatment plants that had RBCs. 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 RBCs 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 RBCs 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 consensus of those using RBCs 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 RBCs 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

RBCs operate such that some proportion 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 maximize 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/m²-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 prediction equation was:

$$S/S_0 = e^{(-K(\text{Stage Number})/W^n)}$$

Where,

S = Effluent ammonia Concentration (mg TAN/L)

S₀ = Influent ammonia concentration (mg TAN/L)

W = Mass loading of substrate (mass/area of biofilter) (g TAN/m² 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 m² 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/ m² 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 m² of surface area, a flow rate of 130 L/min and a hydraulic loading per tank cross-sectional area of 160 L/min-m². The hydraulic loading calculated by dividing the flow rate by the filter or specific surface area was 0.28 L/min-m². 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-m². 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/m³ of catfish the RBC removed 0.78 g N/m² /day; at a stocking density of 118 kg/m³ the RBC removal rate was 0.63 g N/m² /day; and at a stocking density of 57 kg/m³ the RBC removed 0.19 g N/m² /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 °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, theoretically, 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/m²-day and provided 67 ± 18 percent removal of TAN and 59 ± 11 percent removal of nitrite-nitrogen when operated at 27-28 °C. Wheaton et al. (1994) in their development of RBC designs used TAN removal rates in mg TAN/m²/day of: 379, 193, and 122 at 30, 25, and 15 °C, repetitively. In the way of contrast Jansen et al. (1995) cited nitrification rates of between 2 and 4.48 g N/m² (2000 to 4480 mg N/m²) for RBCs 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 °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/m²-day (this is higher than what Miller and Libey (1985) found).

$$\text{Ammonia production} = (\text{fish weight}) (\% \text{ body weight fed per day}) (\text{TAN produced per kg feed fed})$$

$$\text{Ammonia produced} = (9071 \text{ kg fish}) (0.02) (0.03)$$

$$= 5.44 \text{ kg ammonia produced per day}$$

$$\text{Specific surface area needed} = (\text{Ammonia produced/ day}) / (\text{Ammonia removal /m}^2 \text{ /day})$$

$$\begin{aligned}
&= (5.44 \text{ kg/day}) / (0.75 \text{ g TAN/ m}^2 \text{ /day}) \\
&= 7253 \text{ m}^2
\end{aligned}$$

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:

$$\begin{aligned}
\text{Area per plate} &= \pi (\text{Radius})^2 \text{ (two sides of plate)} \\
&= \pi (1.5\text{m})^2 (2) \\
&= 14.2 \text{ m}^2
\end{aligned}$$

The number of plates needed are:

$$\text{Number of Plates} = 7253 \text{ m}^2 / 14.2 \text{ m}^2$$

$$\text{Number of Plates} = 511 \text{ plates that are 3 m in diameter}$$

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

$$\begin{aligned}
\text{Filter length} &= (511 \text{ plates}) (1 \text{ cm per plate}) \\
&= 511 \text{ cm} = 5.11 \text{ m} = 201.1 \text{ inches} = 16.8 \text{ ft}
\end{aligned}$$

The volume occupied by the filter is:

$$\begin{aligned}
\text{Filter volume} &= \pi R^2 (\text{length}) \\
&= 36.12 \text{ m}^3
\end{aligned}$$

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:

$$\begin{aligned}
\text{Tank Volume} &= (36.12 \text{ m}^3) (2.14) \\
&= 77.3 \text{ m}^3
\end{aligned}$$

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

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

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-m², Westerman et al. (1993) reported a loading rate of 0.28 L/min-m², and Miller and Libey (1985) used a loading rate of approximately 0.8 L/min-m². In this example a loading rate of 0.2 L/min-m² will be used. Thus the flow rate will have to be:

$$\begin{aligned} \text{Flow rate through the filter} &= (0.2 \text{ L/min-m}^2) (7253 \text{ m}^2) \\ &= 1450 \text{ L/min} = 383 \text{ gal/min} \end{aligned}$$

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 RBCs 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.

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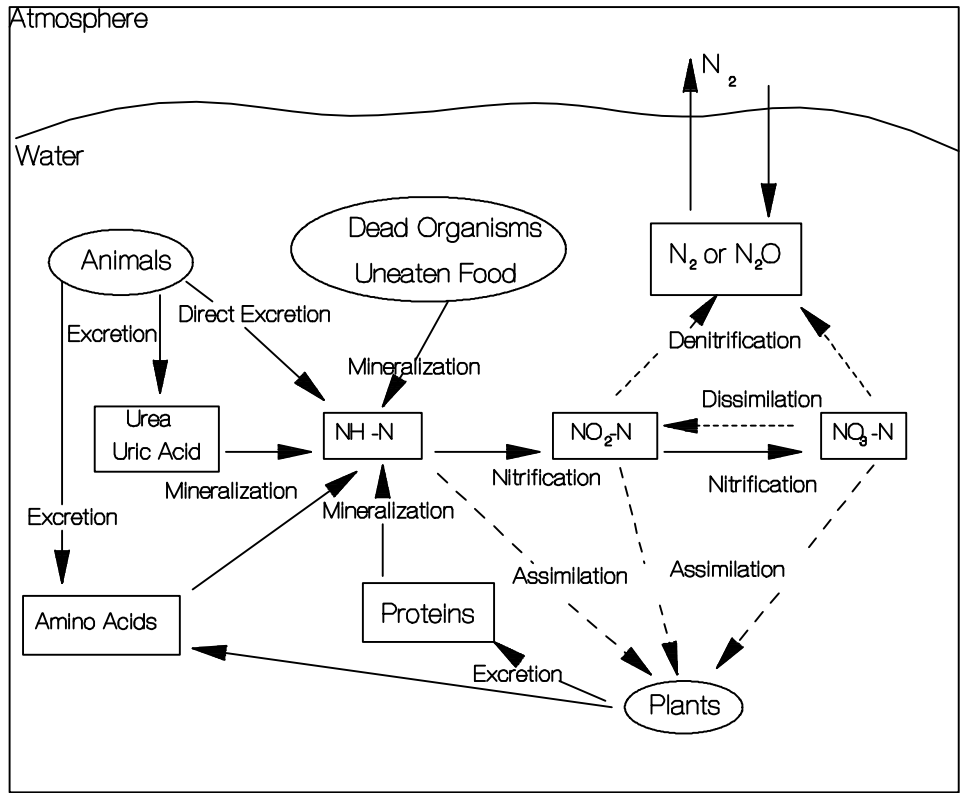


Figure 1. The nitrogen cycle in closed system aquaculture (Spotte, 1979)