2006

Nitrogen and phosphorus removal in biological aerated filters (BAFs)

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Recommended Citation

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Nitrogen and phosphorus removal in biological aerated filters (BAFs)

by

Jeonghyub Ha

A dissertation submitted to the graduate faculty

in partial fulfillment of the requirements for the degree of

DOCTOR OF PHILOSOPHY

Major: Civil Engineering (Environmental Engineering)

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2006

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For the Major Program

To

my lovely wife Sung Hee,

my daughters Ji Won and Ji Woo

and

my parents

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ACKNOWLEDGMENTS

This dissertation would have been impossible without the support of many people. I would like to express my appreciation and gratitude to those who gave me invaluable advice and help. First and foremost, I am grateful to Dr.Ong, my major professor, who has contributed so much on guiding me during all these years. He always points out new and important directions for research and development. I very appreciate his proof readings and corrections for my dissertation in real-time fashion.

I am thankful to my committee member, Dr.Gu, Dr.Loynachan, Dr.Moorman and Dr.Sung, for their guiding of this research.

Most importantly, I am very grateful to my parents for the love and support in all my academic endeavors. Their belief in me helped me to reach my goals. I would like to express special thanks to my wife, Sung Hee, for her prayers, endless love and encouragement. She is the most precious gift that God gave to me. Finally, I am very grateful to God, who stands by me and helps me to go through all difficulties.

ABSTRACT

Three downflow pilot-scale biological aerated filters (BAFs) using different media, gravel, lava rock and plastic rings were operated to study their effectiveness as an add-on system or polishing process for aerated lagoons. Of the three media tested, gravel with a diameter of 5 mm was found to provide more than 96% nitrification at hydraulic retention time (HRT) as low as 0.5 hour with 200% recirculation. Although lava rock has a larger specific surface area than gravel, it did not performed as well and broke easily into fine particles. Experimental results showed that polishing BAF with gravel can be operated at 1 hour HRT without recirculation with nitrification as high as 83% and nitrification was as high as 96% with 200% recirculation at 1-hour HRT. Experiments with different COD/NH_3-N ratios (0.1 to 8) indicate that as COD/NH_3-N ratio increased, nitrification decreased linearly.

At a HRT of 2 hours and a wastewater temperature of 6.5 °C, most of the ammonia was nitrified in a gravel BAF. However, at a HRT of 1 hour, the ammonia removal (%) at a water temperature of 6.5 °C was halved at 54.3 (%) while sCOD removal (91.9%) was approximately close to that for a water temperature of 24 °C. By recirculating the effluent back into the BAF, ammonia removals (%) improved from 54.3 ± 2.7% to 76 ± 3.8% with 1 hour HRT for 100% recirculation at 6.5 °C. With 200% recirculation, ammonia removal improved further to 92 ± 1.5%. C/N ratio had a negative impact on nitrification with less than 5% ammonia removal for a C/N ratio of 8 at 6.5 °C. Ammonia loading reached a maximum of 0.36 kg NH_3-N/m^3-day at 6.5 °C while the maximum ammonia loading was 0.63 kg NH_3-N/m^3-day at 24 °C.

A pilot-scale partially aerated BAF was operated with an anaerobic, anoxic and oxic zone at a temperature of 24 °C for the removal of nitrogen at various hydraulic loading rates and recirculation rates. With 300% recirculation at a HRT greater than 3 hours, the total nitrogen removed was approximately 80%. sCOD removals and nitrification at HRT of 3 hours with recirculation rates of 200 and 300% was more than 96%. Based on the experimental results, the maximum ammonia masses removed were approximately 0.15, 0.19 and 0.21 kg NH_3-N/m^3-day for 100, 200 and 300% recirculation ratios, respectively.

By adding two alternating attached growth columns (diameter = 75 mm) after the partially aerated BAF, the phosphorus removal of more than 96% were obtained for influent phosphorus concentrations of 8 mg P/L and 16 mg P/L. The columns operated alternatively as aerobic or anaerobic reactors at a HRT of 6 hours over a one day cycle to stimulate phosphorus release and uptake without interruption. The effluent phosphorus concentration under aerobic condition was less than 2.0 mg P/L for as long as 35 hours. The experimental results demonstrated that by using two alternating biofilm filters in conjunction with a partially aerated BAF, phosphorus removal can be achieved.

CHAPTER 1. INTRODUCTION AND OBJECTIVES

INTRODUCTION

Eutrophication of rivers and lakes and the hypoxia condition in the Gulf of Mexico are generally attributed to nitrogen and phosphorus from surface runoff and for wastewater discharge. To correct some of these discharges, effluent ammonia, nitrogen and phosphorus limits have been added or being considered for the National Pollution Discharge Elimination System (NPDES) permits of wastewater facilities. Large cities typically have the necessary financial means to upgrade their facilities to meet these standards. In the Midwest and in rural areas, small communities typically do not have the financial means and manpower to manage biological nutrient removal facilities. Aerated lagoons are used widely in small communities for the treatment of domestic wastewater and are not designed or are unable to meet the effluent ammonia and total nitrogen concentrations. This is very true for the aerated lagoons in late winter and early spring months where ammonia levels often exceed the NPDES permit limitation. In addition to ammonia and total nitrogen discharge, US EPA and state regulating agencies are studying recommendations on the implementation of phosphorus water quality criteria for various streams and rivers in Iowa. Implementation of the water quality criteria would mean that wastewater treatment plants would eventually have to meet a regulated phosphorus limit for their effluent.

Biological aerated filters (BAFs) may be suitable as an add-on treatment process for aerated lagoons due to fact that this process maintains high concentration of active biomass on the support media. In fact, BAFs are not only suitable as an add-on process, but can be used for secondary treatment. BAFs are becoming increasingly popular in Europe due to their compactness. In the US, several wastewater treatment plants are using BAFs as secondary treatment (Belgiorno et al., 2003; Hamoda et al., 2004; M'Coy, 1997; Peladan et al., 1996; Pujol, 2000). In a typical aerated lagoon system, the BAF can be located immediately after the second cell of the aerated lagoon or after the first cell as an add-on system. This would minimize modification and cost to the present aerated lagoons. BAFs are relatively compact, easy to operate and may be more efficient in carbonaceous and ammonia removal than activated sludge systems (Bigot et al., 1999; Hodkinson et al., 1999; M'Coy, 1997; Mendoza-Espinosa and Stephenson, 1999; Wheale and Cooper-Smith, 1995). A typical BAF set up consists of a submerged, granular media with air injected at the bottom of the filter. The position of air injected can be varied to create various anaerobic/oxic zones.

Several researches have tested the BAF for ammonia and nitrogen removal (Lee et al., 2005; Rogaller and Bourbigot, 1990; Safferman et al., 2003; Tay et al., 2003). Several full-scale systems have reported the operating conditions and effectiveness of BAFs (Chen et al., 2000; Peladan et al., 1996; Pujol, 2000; Tschui et al., 1994). However, information on the performance of the BAF under various operating and environmental conditions are still lacking. For example, not much is known about the performance of BAF at low temperatures (Fdz-Polanco et al., 1994). The impact of different organic and hydraulic loading rates, and recirculation rates on nitrification needs further investigation (Boller et al., 1994). Most media provided by companies are proprietary. Use of commonly available media such as sand would encourage wider use of BAFs (Kent et al., 1996; Moore et al., 2001). An area that shows promise and needs further investigations are BAFs modified with anoxic and oxic zones for the removal of nitrogen within a single column or creating various anaerobic, anoxic and oxic zones within a single column for removal of nitrogen and phosphorus. The use of a single media bed for removal of nitrogen and phosphorus will minimize space use and the capital cost of the plant. Not much work has been done in the application of BAF for nitrogen and/or phosphorus removal.

Dissertation Objectives

The objectives of dissertation as follows:

1. investigate nitrification in a BAF for different media types and various operating parameters as an add-on system,
2. investigate the impact of different temperatures (6.5, 13 and 24 °C) and recirculation of the effluent on nitrification,
3. investigate the use of various oxic, anoxic and anaerobic zones within the BAF to treat wastewater as a secondary treatment for nitrogen removal. Operating conditions studied include recirculation rates, hydraulic and organic and nitrogen loading rates.
4. investigate phosphorus removal from the effluent of partially aerated BAF using alternating attached growth aerobic and anaerobic filters.

Dissertation Organization

The dissertation is organized into seven chapters with the Introduction and Objectives in Chapter 1 and a literature review on BAF and removal of nutrient in Chapter 2. Chapter 3 is focused on meeting objective number 1 where the nitrification performance of BAF was examined for different media types, at various COD and nitrogen loading rates, hydraulic loading rates and recirculation at 13 °C. Chapter 4 investigates objective No. 2 where the temperature effects and recirculation of treated effluent on nitrification are presented.

Chapter 5 presents the results of a partially aerated BAF system for the nitrification and denitrification of wastewater as a secondary treatment process under various hydraulic, organic, nitrogen and recirculation conditions. Chapter 6 investigates phosphorus removal using a partially aerated BAF coupled with two attached growth filters that alternate from aerobic to anaerobic filters and vice versa. Finally, Chapter 7 provided the general conclusions of the dissertation.

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CHAPTER 2. LITERATURE REVIEW

BACKGROUND

In this chapter, background information pertinent to this research is provided, focusing on a review of the following topics: (1) types of Biological Aerated Filter (BAF); (2) media types for BAFs; (3) nitrification in BAFs, emphasizing chemical oxidation demand (COD) and ammonia loadings, hydraulic retention time (HRT) and temperature effects on biotransformation of ammonia; (4) denitrification in BAFs; (5) phosphorus removal by phosphorus-accumulating organisms (PAOs); and (6) simultaneous nitrification, denitrification and phophorus removal.

2.1 Types of Biological Aerated Filters (BAFs)

Many wastewater treatment plants are required to meet increasingly stringent effluent standards. BAFs are well suited as an add-on or as an upgrade process since BAF retains an active biomass on the support media and may have smaller reactor volumes. BAFs can be built modularly and the system can be automated (M'Coy, 1997). BAFs have a solid media which acts as a support media for growth of microorganisms and are fully submerged with air injected into the reactor (Mendoza-Espinosa and Stephenson, 1999).

High organic loadings of 2.5 kg BOD_5/m^3-day have been reported as compared to 0.06 kg BOD_5/m^3-day for activated sludge plants (Smith et al., 1990). Peladan et al. (1997) have tested BAF systems with a loading rate as high as 18 kg COD/m^3-day. At these hydraulic loading rates, the biomass concentrations in BAFs can be 8 to 9 times greater than in activated sludge plants (Boller et al., 1994). In addition, HRT as low as 10 minutes did not seem to influence the reactor performance as reported by Pujol et al. (1998). The high water velocities positively influence and improve the mass of substrate transfer between aqueous phase and biofilms (Pujol et al., 1998).

BAFs can be operated either in an upflow or downflow mode (Figure 1a and b). Downflow systems with countercurrent air flow have the advantage of efficient mass transfer of oxygen to biofilm in the reactor (Mendoza-Espinosa and Stephenson, 1999). Downflow BAFs are effective when carbonaceous matter and ammonia removal are required in the reactor without oxygen limitation. In downflow BAF (Figure 1a), nitrifying microorganisms are typically found at the bottom of the reactor and therefore will not suffer oxygen limitation. Upflow systems with co-current air and wastewater flow can handle higher influent flowrates than downflow systems (Figure 1b). In upflow systems, odor problems are reduced since the wastewater is fed from the bottom. The BAF is a flexible system whereby different zones can be achieved. For example, an upflow BAF can have an oxic and anoxic zone to allow for nitrification and denitrification (Figure 1c). The anoxic zone removes soluble organic matter and nitrate, while the aerated zone oxidizes the remaining organic matter and ammonia. For example, the Biostyr technology marketed by Veolia Water was installed as an upgrade for several treatment plants in Denmark with a combination of nitrification-denitrification and filtration in a single reactor (Borregaard, 1997). The effluent concentrations obtained by the Biostyr system were 8 mg TN/L and a suspended solid content below 10 mg/L.

2.2 Media Types for Biological Aerated Filter

Selection of a suitable BAF medium is important in the design and operation of the process in order to meet strict discharge limits. The BAF media should be resistant to attrition, have appropriate specific weight, high specific surface area, and chemically stable

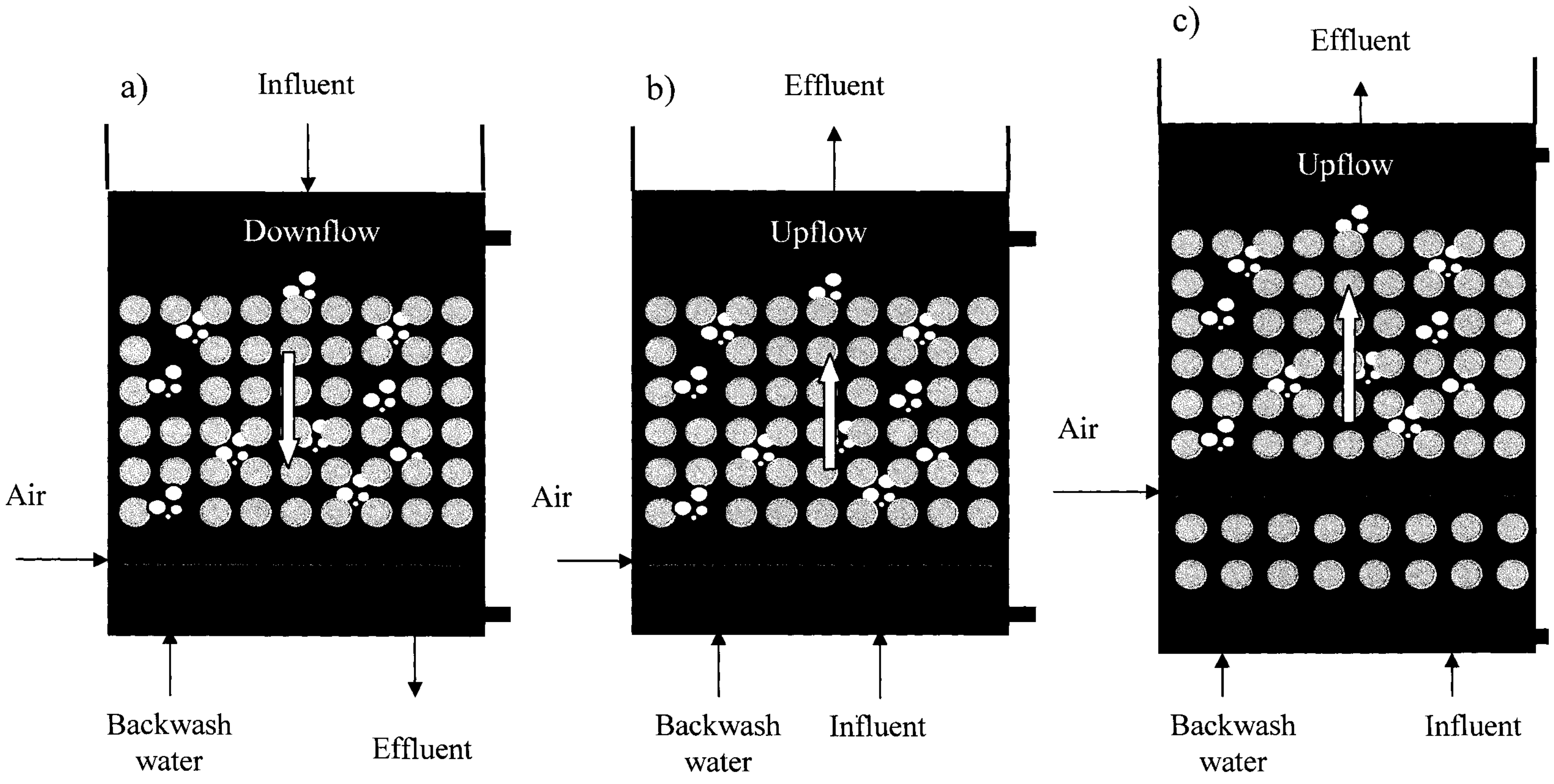

Figure 1. Schematic of (a) downflow and (b) upflow BAFs and (c) upflow BAF with oxic and anoxic zones.

(Kent et al., 1996). The size of a BAF media also affects the process performance and should be carefully selected for different applications (Mendoza-Espinosa and Stephenson, 1999). Media can be sunken type for downflow and upflow configurations, or floating type for upflow systems (Mann et al., 1998; Mendoza-Espinosa and Stephenson, 1999).

The Biocarbone process, a downflow BAF, uses activated carbon in the original design, but is currently using a 3- to 5-mm fired clay material (Metcalf and Eddy, 2003). For the Biofor process, an upflow submerged aerobic attached growth process, a 2- to 4-mm expanded clay material is used (Metcalf and Eddy, 2003). The Biostyr process, an upflow process, uses 2- to 4-mm polystyrene beads that have a specific density less than water (Metcalf and Eddy, 2003).

To examine the effects of media types on nitrogen removal under various conditions, Kent et al. (1996) characterized and tested seven different media (EFG (fireclay), Starlight, Molochite, old expended shale, new expended shale, Lytag, Arlita) using the British Effluent and Water Association (BEWA) standard. They found that specific surface areas for EFG (fireclay), Starlight, Molochite, old expended shale, new expended shale, Lytag, Arlita were 1.18, 2.16, 0.9, 0.83, 0.47, 3.89 and 3.98 m^2/cm^2, respectively but all media used were more prone to attrition than is expected in the BEWA standard. In a pilot study by Moore et al. (2001), two media sizes, 1.5 - 3.5 and 2.5 - 4.5 mm, made out of formed clay called Starlight C were tested in pilot-scale BAFs. The BAF with 3.5 mm media had a mean depth of 1.7 m, whereas the larger media 4.5 mm had a mean depth of 2.1 m. In this study, the smaller media gave better tCOD removal efficiency (83% as compared to 77%) than the larger media due to the larger surface area per unit volume.

Mann et al. (1999) compared a floating media (size 3 mm) made of poly-propylene and a sunken media (3 mm) made of a mixture of poly-propylene (60%) and calcium carbonate (40%) under identical hydraulic conditions. From an initial start-up loading of 0.486 kg $m^{-3}d^{-1}$ for suspended solids (SS) and 0.586 kg/m^3-day for soluble COD, the removal efficiencies dropped to 50 % when the loadings increased to 1.397 kg/m^3-day (for SS) and 1.403 kg/m^3-day (for sCOD). Results from this study showed that the floating media performed better at the higher flowrates of 1.4 kg/m^3-day under shock loading conditions than sunken media.

Farabegoli et al. (2003) evaluated the performance of four media types: glass (5 mm), plastic (6 mm), pozzolan (2 - 6 mm) and expanded clay - Arlita (5 - 8 mm) in a pilot-scale BAF in terms of organic matter and suspended solids removal. The experimental results showed that the plastic spheres and Arlita media were the optimal materials meeting both mechanical and biological requirements. The COD removal efficiency with the plastic media varied between 30% and 70% whereas 67% removal efficiency was obtained for the Arlita media.

Based on literature, a variety of materials have been used. However, many of the materials are manufactured and proprietary. Use of easily obtained materials such as gravel have been tested but have not been widely used. It will be an advantage if BAF can use easily obtained materials such as sand.

2.3 Nitrification in Biological Aerated Filter

2.3.1 Nitrification Principles

Autotrophic nitrifying microorganisms grow relatively slowly and are more sensitive toward environmental conditions than heterotroph microorganisms. As such the amount of degradable organic matter present has an impact on the growth of autotrophic nitrifying microorganisms.

Nitrification can be written as follows (Rittmann and McCarty, 2002):

Reaction by *Nitrosomonas*:

$$2NH_4^+ + 3O_2 \text{ --------}\rangle\ 2NO_2^- + 2H_2O + 4H^+ + \text{New Cells}$$

Reaction by *Nitribacter*

$$2NO_2^- + O_2 \text{ --------}\rangle\ 2\,NO_3^- + \text{New Cells}$$

The total reaction with respiration and cell synthesis of nitrifiers can be written as:

$$NH_4^+ + 1.83O_2 + 1.98\,HCO_3^- \text{ ----}\rangle\ 0.98\,NO_3^- + 0.021C_5H_7O_2N + 1.88H_2CO_3 + 1.04\,H_2O$$

As written above, 1 g of NH_3-N requires 4.33 g of O_2 and 7.14 g of alkalinity with the production of 0.15 g of cells.

Many environmental factors affect the performance of nitrifying biofilms in a BAF. In addition to the hydraulics and nutrient/substrate transport (diffusion) into the biofilm, the reactor configuration and the air-liquid mass transfer have an impact on nitrification. Besides the physical factors, nitrification are affected by the concentrations of dissolved nutrients and substrates (COD, NH_4^+, NO_2^-, NO_3^- and O_2), alkalinity (HCO_3^-) and pH, the concentration of toxic substances, and the relative diversity of microbial species (heterotrophs, *Nitrosomonas, Nitrobactor*), biomass density and biofilm thickness.

2.3.2 Hydraulic and Ammonia Loading on BAF Performance

BAF systems have been shown to operate successfully at higher hydraulic and organic loading rates than activated sludge systems (Belgiorno et al., 2003; Hamoda et al., 2004; Mendoza-Espinosa and Stephenson, 1999; Rogalla et al., 1994; Pelandan et al., 1996). In addition, BAF systems are effective in nitrogen and COD removals for a small footprint (Rogalla and Bourbigot, 1990; Rodgers et al., 2004; Westerman et al., 2000). Removals of organic carbon and ammonia under various conditions in BAFs from the literature are summarized in Table 1.

A two-year evaluation of a full-size biological aerated filter wastewater treatment unit with a nominal flow capacity of 1,893 m^3/day (0.5 mgd) was tested (Stensel et al., 1988). In this study, 80 – 88% nitrification was achieved in a combined BOD removal and nitrification system. Equations were developed to predict oxygen consumption and sludge production as a function of influent soluble BOD and suspended solids to influent total BOD ratios. The oxygen consumption per unit of BOD applied is low ranging from 0.43 to 0.80 kg O_2/kg BOD when compared to conventional activated sludge which ranged from 0.8 to 1.0 kg O_2/kg BOD.

Aerated biofilters for nitrification and effluent polishing were tested to assess process feasibility to upgrade conventional activated sludge treatment (Paffonia et al., 1990). Biodegradable carbonaceous (72 mg COD/L) and suspended matter (36 mg/L) were

Table 1. Organic carbon and ammonia removal in biological aerated filters under nitrification condition.

Configulation	Influent organic carbon (mg C/L)	Influent Ammonia (mg N/L)	HRT (h)	Ammonia loading rate (kg/m^3 day)	COD loading rate (kg/m^3 day)	Linear velocity (m/h)	Media type	Media Size (mm)	NH_4^+-N removal (%)	COD removal (%)	Reference
Downflow	92 - 144 (TBOD)	13 - 27		0.31 - 0.48 (TKN)	3.5 - 4 (BOD)	2.4	Clay	3.4 - 4.4	74 - 80	85	Stensel et al., 1988
Downflow	72	10 - 35	1	0.6	3 (BOD)	2.5 - 4	Sand	3 - 6	95	65	Paffoni et al., 1990
Downflow	120 (BOD)	-	-	-	3 - 5 (BOD)	17	Anthracite	-	-	83 (BOD)	Adachi and Fuchu, 1991
Upflow and downflow	255 (COD)	55 (TKN)	-	0.36	3.5 - 11.9	2.7 - 8.2	Expanded clay	2.7 - 6	68 - 80 (TKN)	69.8 - 78.8	Canler and Perret, 1994
Upflow	-	-	-	1.5	10	6 - 10	Clay	3 - 6	-	-	Pujol et al., 1994
Upflow and downflow	400	16	-	0.4 - 1.5	-	3 - 10	Granular Slate	2.5 - 3.5	-	-	Boller et al., 1994
Downflow	244 - 590 (COD)	25 (TKN)	-	0.4	2 (BOD)	2.5	Sand	-	80	90	Rogaller et al., 1994
Upflow	-	14 - 55	-	2.7	-	30	Clay	-	90	-	Peladan et al., 1996
Upflow	75	24	0.2	1	2	4 - 10	Clay	-	51	94	Pujol et al., 1998

completely removed at filtration velocities of 2.5 – 4 m/h. An influent concentration of 20 mg NH_3-N/L was totally converted with an empty bed contact time of 1 hour. The Arrhenius temperature coefficient for nitrification was estimated to be 1.05. Bachwash frequency was less than once per day with a maximum of 5% of filter flowrate used for backwashing.

The first full-scale Biocarbone aerated filter was used to treat the wastewater of Soissons (France) with a population of 40,000 (Rogalla and Sibony, 1992). Results showed that the carbonaceous removal rates were up to 4 kg BOD/m^3-day while nitrification rates were around 0.6 kg N/m^3-day. Sludge yields ranged consistently around 0.8 kg SS/kg BOD removed. Oxygen transfer efficiency varied from 7% to 15%. The secondary effluent quality at two hours detention time was below 5 mg N/L and 10 mg BOD/L.

High rate aerated biofilters for plant upgrades were developed by Rogalla et al. (1994) by using a synthetic floating material for enhanced performance, simplified backwash and a combination of anoxic and aerobic zones in a single reactor. Effluent quality in their studies was less than 10 mg/L for all components. Full-scale results of the upflow floating filter met the European directive for nitrogen and phosphorus removal which were 6 and 2 mg/L, respectively. Canler and Perret (1994) assessed the impact of operating parameters on the treatment efficiency of BAFs. Results obtained indicated that the effluent quality for COD was less than 90 mg/L at applied loads of 7 kg COD/m^3-day and hydraulic loads from 0.8 to 4.6 m/h have no negative influence on effluent TSS quality with TSS less than 30 mg/L for an influent TSS of 182 mg/L. Three different aerated pilot scale biofilters (Biocarbon, Biostyr, Biopur) were operated as tertiary nitrification systems (Tschui et al., 1994). Results showed that nitrification depended on the water velocities as well as air velocities. Operating the BAFs at higher velocities of air (30 m/h) and water (14 m/h) increased the nitrification rate but increased the headloss. The parameters affecting biofilm reactors were identified by Boller et al. (1994). In this study, a comparison between the different biofilm processes and activated sludge systems indicated that biofilm systems require smaller reactor volumes than suspended growth systems. Zhang and Bishop (1994) showed that higher fluid velocities across the surface of biofilm reduced the resistance of chemical components transfer into biofilms. In addition, Debus et al. (1994) observed that for higher fluid velocities degradation of aromatic compounds by a biofilm reactor improved.

Peladan et al. (1996) conducted a pilot study using Biofor system for water velocities between 5 and 20 m/h. The pilot plant was able to nitrify 2.7 kg NH_3-N/m^3-day at 14 °C. The effluent ammonia concentration remained under 4 mg/L at a water velocity of 10 m/h and an influent ammonia concentration of 35 mg/L. The study also investigated nitrification at velocities as high as 30 m/h and they found that an increase in water velocity has a positive effect on nitrification.

For nitrification in an upflow biofiltration reactor, Pujol et al. (1998) investigated the impact of water velocity and applied loading rate on nitrification and found that extremely short HRT of 10 minutes did not influence the reactor performance for nitrification. As indicated earlier, high water velocities may positively influence and improve the mass transfer between aqueous phase and biofilms. Husovitz (1998) reported that when the hydraulic loading was changed from 5 to 16 m/h while holding the COD and ammonia loadings constant, the ammonia removal increased, even though contact time decreased.

The influence of organic and ammonia loading on nitrifier activity and nitrification performance for a two-stage BAF was studied during the winter time by Gilmore et al. (1999). In this investigation, overall nitrification efficiency for winter conditions was greater than 90% when ammonia loads were 0.6 kg N/m^3-day. The zones of ammonia oxidizing activity progressed along the length of the columns as organic and ammonia loadings to the system increased. The oligonucleotide probe data indicated that this shift is due to higher BOD loads to the second stage which supported higher growth of heterotrophs. The experimental results showed that chemical measurement of nitrogen species corresponded well with ammonia oxidizing bacteria (AOB) activity levels based on rRNA hybridization with a group-specific oligonucleotide probe. In addition, the relative levels of AOB activity appeared to decrease as higher BOD loading rates were applied to the reactor due to an increase in the fraction of heterotroph microorganisms. The BAF system performed best and showed highest AOB activity levels based on rRNA hybridization probe when low average BOD loadings below 1.4 kg/m^3-day were applied to the BAF.

The effect of the suspended and attached biomass on nitrification in a two submerged biofilters in series was studied by Villaverde et al. (2000). In this study, the HRT in both filters was adjusted to 2 hours. Superficial velocities of 1 m/h for liquid and 8 m/h for the air were maintained throughout all the experiments. Filter backwash was conducted when the observed head loss through the filter was above 1.2 m of the column. The activity of attached and suspended biomass was determined by closed respirometry assays which showed higher activity for the attached biomass than the suspended biomass.

Tertiary nitrification at different temperatures and COD loads in an upflow biofilter with a floating media was studied by Payraudeau et al. (2000). The experimental results showed that the biofilter achieved very low, steady state, effluent ammonia concentrations (less than 2 mg/L) with different applied nitrogen loads of 0.3 to 2.7 kg TKN/m^3-day even at low temperature of 14 °C and applied carbonaceous loads of 9 kg COD/m^3-day.

At high carbonaceous concentrations, the rapid growth rate of heterotroph microorganisms results in limited autotrophic bacteria growth (Nogueira et al., 2002). Nogueira and his worker quantified the composition and dynamics of microbial consortia by fluorescence in situ hybridization (FISH) with rRNA-targeted oligonucleotide probe and found that addition of acetate resulted in a growth of heterotroph microorganisms which decreased nitrification.

A recent experiment by Lshikawa et al. (2002) performed with a new dual-medium aerated filter showed that water velocity of 4.6 m/h resulted in ammonia effluent concentration of 2.1 mg/L when the influent NH_3-N concentration of influent was 20 mg/L, at a temperature of 15 °C, and a total media depth of 3 m (depth of upper filter layer - 2 m; depth of lower filter layer - 1 m).

The effect of substrate surface loading rates of 1.2, 0.6 and 0.3 g TOC/m^2-day on biofilm growth and structure by chemical, biochemical and microscopic methods was further investigated by Wijeyekoon et al. (2004). The results showed that nitrification was suppressed under substrate loading of 1.2 g TOC/m^2-day. Under the low loading of 0.3 g TOC/m^2-day, most of the active organisms were present at the base of the biofilm layer while high loading rates showed distinct stratification of active and inactive cells with active cells present close to the top and center of the biofilm layer.

As reviewed above, BAFs play a significant role on effective nitrogen and COD removals for a small footprint and short HRTs. In addition, BAF systems could be operated with a loading rate as high as 18 kg COD/m^3-day and 2.7 kg NH_3-N/m^3-day compared to 0.12 kg COD/m^3-day and 0.2 kg NH_3-N/m^3-day for activated sludge plants. Therefore, BAFs are effective systems for ammonia and COD removals and suitable for upgrading conventional wastewater treatment plants.

2.3.3 C/N ratio on nitrification

Carbon:nitrogen (C/N) ratio of the wastewater has been found to impact the distribution of both nitrifiers and heterotrophs population through the biological filters (Fdz-Polanco et al., 2000; Li et al., 2002). An understanding of the impact of C/N ratio on nitrogen removal from wastewater is imperative for optimizing biofilm reactor.

Cheng and Chen (1994) employed a three-phase fludized-bed bioreactor packed with granular activated carbon to study the nitrification of high-stregth ammonium (500 mg N/L) wastewater. In this study, various COD/NH_4^+-N ratios (0 to 1) were applied to the fluidized bed reactor using various concentrations of sucrose and with a loading of 2.0 kg NH_4^+-N/m^3-day. The dissolved oxygen levels were maintained at 3.0 mg/L. The experimental results clearly indicated that the addition of organic carbon inhibit the bioactivity of ammonium oxidizer and nitrite oxidizer.

Oligonucleotide probes targeted to 16S rRNA of the nitrifiers were used by Ohashi et al. (1995) to characterize microbial structure changes in the biofilms due to organic loading changes. The experimental results indicated that for C/N ratios (weight/weight) of 0.18, 0.37, 0.52 and 1.5, the production of nitrifiers decreased with increasing C/N ratio and the biofilm remaining after backwashing also contained a higher fraction of nitrifiers as compared to the backwash solids stripped from the outer layer of biofilm.

Fdz-Polanco et al. (2000) described the changes in biofilm density and specific activities of carbon within a nitrifying upflow BAF as a function of the C/N ratio in the wastewater. Their results indicated that the COD/NH_3-N limit without losing nitrification was about four. For COD/NH_4^+-N ratios higher than 4, there were two different zones within the filter, the first zone located at the entrance where organic matter and ammonia entering the filter were removed at rates of 3.85 kg TOC/m^3-day and 0.19 kg N/m^3 day, respectively and the second zone with removal rates of 0.42 kg TOC/m^3-day and 0.96 kg N/m^3-day.

The effect of C/N ratio (weight/weight) on the spatial distributions of ammonia-oxidizing bacteria and their activity was investigated by Satoh et al. (2000) using microelectrodes and fluorescent in situ hybridization (FISH) technique. The C/N ratios tested were 0, 1 and 2, respectively. In this study, interspecies competition for O_2 between ammonia-oxidizing bacteria and heterotrophic bacteria was experimentally evaluated and the researchers found that an increase in the C/N ratio resulted in a decrease in the specific NH_4^+ oxidation rates in the outer part of biofilm. This experimental results clearly provided insights on the stratified spatial distributions of the microbial ecology and population dynamics of nitrifying bacteria in the biofilm.

In another recent study, Ballinger et al. (2002) observed the effect of influent C/N ratio (from 2 to 5) on the structure of β-proteobacterial autotrophic AOB by DGGE analysis of 16S rRNA gene fragments amplified using a range of AOB-selective primers. At a C/N ratio of 2, nitrification was measured at approximately 1.0 mg NH_4^+-N/g dry wt of biomass/h and at a C/N ratio of 5, there was a 50% reduction in nitrification rates. Quantitative FISH analysis indicated that β-proteobacterial AOB were present at approximately 10^8 cells/mL of biomass with a C/N ratio of 2 while β-proteobacterial AOB were not detected by FISH for a C/N ratio of 5.

According to the above literature reviews, nitrification was suppressed under high COD loading conditions. For effective nitrogen removal in the BAF systems, the COD/NH_3-N ratio should be less than 4. For C/N ratios higher than 4, nitrification was inhibited by heterotrophic microorganisms.

2.3.4 Nitrification Models and Rates

Simple input and output models have been developed for BAFs (Sanz et al., 1996). These models assumed carbonaceous and nitrification kinetics are either zero or first order reactions. For example, Behrendt (1999) found that a first-order model was able to describe the ammonia concentration changes within a bench-scale BAF with a HRT of 1.2 hour. Another example is the use of zero-order kinetics by Mann et al. (1998) to describe the ammonia concentration changes in a BAF with polypropylene media (diameter of 2.4 - 2.7 mm and length 4 - 6 mm) and a flow rate of 0.2 m^3/min. A novel mathematical model for biofilm processes was developed (Vayenas and Lyberatos, 1994) based on material balances on the biofilm and bulk liquid. The model was originally developed for trickling filter. The biofilm was assumed to be planar, homogeneous and the reactions at steady state with the expected profiles of substrates in the biofilm as shown in Figure 2. Using a control volume and the diffusion equation as the starting point, Vayenas and Lyberatos derived an expression for the ammonia concentration gradient as a function of the ammonia concentration CS:

$$dCS/dz = (2\ \mu_{max}\ \chi /\ D_S Y_S)^{0.5}\ F(CS, CS_B, CO_B)$$

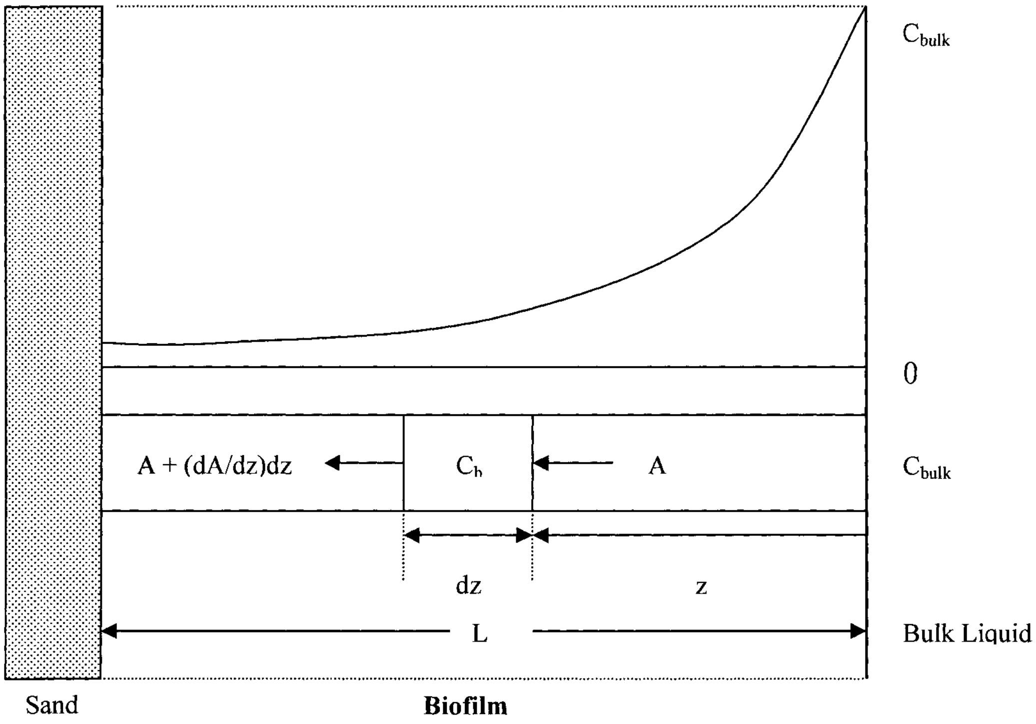

Figure 2. Substrate profile in biofilm.

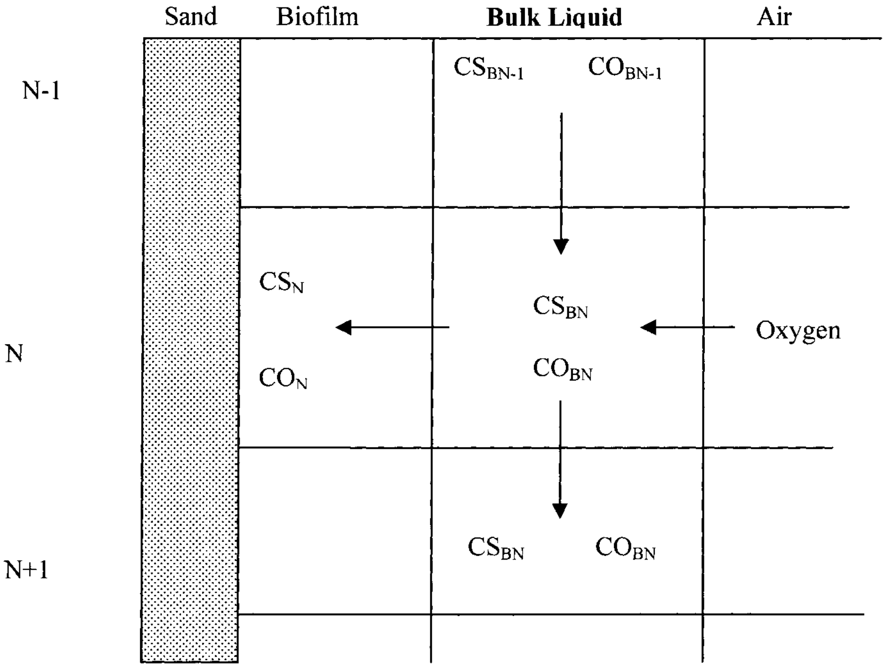

Figure 3. Basic processes used in the mathematical model.

where F(CS, CS_B, CO_B) is a function of the ammonia concentration (CS) and the bulk liquid oxygen (CO_B) and ammonia (CS_B) concentrations, μ_{max} is maximum specific growth rate (1/d), χ is biodensity (mg/cm^3), Ds is diffusion coefficient of ammonia (cm^2/day) and Ys is ammonia effectiveness factor (g-biomass/g-NH_3-N).

The expression F(CS, CS_B, CO_B) was obtained by using Figure 3 for the transport of substrates into the aqueous/biofilm phases and by conducting a mass balance for ammonia in the bulk liquid. This expression is given by:

$$F = \{CS - T/D \log[(E+DCS)/E] - J/I \log [(K + ICS)/K]\}$$

where $T = K_A(B - AK_A)$, $D = B + Ko - AK_A$, $E = K_AD$, $J = Ko(B + Ko)$, $I = AD$, $K = (B + Ko)^2 - AK_A(B + Ko)$, $A = Y_SD_S/Y_OD_O$, $B = CO_B - ACS_B$, Q is volumetric flow rate (m^3/d), A is media specific area (m^2/m^3), V is volume of reactor (m^3) and ε is porosity. Yo is oxygen yield factor (g-biomass/g-O_2), Do is diffusion coefficient of oxygen (cm^2/day), CS is ammonia concentration (mg/L), CO is oxygen concentration (mg/L), K_A is ammonia saturation concentration (mg/cm^3) and Ko is oxygen saturation constant (mg/cm^3).

To relate the nitrification rates at different temperatures, Tschui et al., (1994) proposed the following relationship:

$$r_v, NH_3\text{-}N(T) = r_v, NH_3\text{-}N(T{=}10\ °C)\cdot \exp[k_T\cdot(T-10\ °C)]$$

where r_v, NH_3-N(T) is the nitrification rate at a wastewater temperature T [g/m^3·day], r_v, NH_3-N(T=10°C) is the nitrification rate at a wastewater temperature T=10°C [g/m^3·day], T is the wastewater temperature [°C] and k_T is the temperature coefficient [1/°C] = 0.03/°C.

2.4 Denitrification in biological filter

Heterotrophic denitrifying organisms produce nitrate reductase in the reduction of nitrate to nitrogen gas, the final product of denitrification. Simultaneously, NO_3^--N is assimilated and utilized for cell synthesis with nitrate acting as the electron acceptor. Denitrification is strongly dependent on the availability of carbon.

In denitrification, the sequential reduction of nitrate reaction by heterotroph microorganisms is as follow:

$$NO_3^- \dashrightarrow NO_2^- \dashrightarrow NO \dashrightarrow N_2O \dashrightarrow N_2$$

Rittmann and McCarty (2002) provided the following balanced reaction for denitrification with methanol as a carbon source:

$$6\ NO_3^- + 5\ CH_3OH \text{--------}\rangle\ 3\ N_2 + 5\ CO_2 + 7\ H_2O + 6\ OH^-$$

The overall reaction with respiration and synthesis of denitrifier (cell formula = $C_5H_7O_2N$) can be written as:

$$NO_3^- + 1.08\ CH_3OH + 0.24\ H_2CO_3 \text{----}\rangle\ 0.06C_5H_7O_2N + HCO_{3-} + 1.68\ H_2O + 0.47\ N_2$$

For the denitrification of 1 g of NO_3^--N, denitrifiers consume 2.47 g of methanol, produce 3.57 g of alkalinity and synthesis 0.45 g of cell.

2.4.1 Denitrification Studies Using Biological Filters

Biological filter systems have been shown to operate successfully at higher nitrate loading rates and organic loading rates than suspended growth systems (Abu-Ghararah, 1996; Nogueira et al., 1998; Rabah and Dahab, 2004). Experimental studies in the literature for the packed bed, submerged filters and fluidized beds on denitrification under various operating and design conditions are discussed and are summarized in Table 2.

A biological aerated filter with oxic and anoxic conditions for the complete nitrogen removal was used by Rogaller and Bourbigot (1990). They showed that the minimum BOD/N ratio needed was 3 with maximum removal of nitrogen for BOD/N of 6. Denitrification efficiency was found to be directly proportional to the recirculation ratio. With a recycling rate of 300%, a total nitrogen removal of 75% was obtained. To maintain the efficiency at 75%, applied flux to anoxic zone should not exceed 1.5 kg NO_3^--N/m^3-day. Complete degradation of organic carbon was obtained at loading of 10 kg COD/m^3-day with nitrification and denitrification.

The long- and short-term effects of hydraulic loading on a pilot packed tower bioreactor for the denitrification of groundwater were investigated by de Mendonca et al. (1992). From the experimental results it was shown that at a hydraulic loading rate of 3.63 m/h, denitrification was maintained in the first 48 hours after air scour, while several days were required to restore the biofilm denitrification performance when the hydraulic loading was doubled. Hwang et al. (1994) performed several batch tests and pilot-scale studies with submerged columns on biological denitrification using isopropanol as the carbon source. In the pilot-scale study using submerged columns, at an influent nitrate-N concentration of 20

Table 2. Organic carbon and nitrate removal in biological filters under denitrification condition.

Configuration	Influent organic carbon (mg C/L)	Influent nitrate (mg N/L)	HRT (h)	Nitrate loading rate (kg/m^3 /day)	COD loading rate (kg/m^3 /day)	Linear velocity (m/h)	Media type	Media Size (mm)	NO$_3^-$-N removal (%)	COD removal (%)	Reference
Upflow	-	11 - 28	-	2.5	-	16 - 21	Bead	0.42-0.55	-	-	Hermanowicz and Cheng, 1990
Upflow	570	-	2	1.5	10	7.1	Light weight bead	-	75	89	Rogaller and Bourbigot, 1990
Upflow and downflow	255 (COD)	55 (TKN)	-	0.36	3.5 - 11.9	2.7 - 8.2	Expanded clay	2.7-6	68 - 80 (TKN)	69.8 - 78.8	Canler and Perret, 1994
Upflow	-	-	-	>4	10	14	Clay	3-6	-	-	Pujol et al., 1994
Downflow	1,000 (isopropanol)	22	-	0.7 - 1.2		4.2	Anthracite	3	95	-	Hwang et al., 1994
Downflow	220 (COD)	-	-	1	3	-	Plastic polymer packing	-	90	-	Ryhiner et al., 1994
Upflow	1,250	75	6	-	5	-	Glass ring	15 - 25	86	95	Chui et al., 1996
Downflow	-	-	-	1	4.5	-	Clay and sand	1 - 2	-	-	Koch and Siegrist, 1997
Upflow	60 (sCOD)	-	0.1	6.4	3 - 9.1	36	Sand	0.46	95	-	Semon et al., 1997
Upflow	434	-	-	0.6 - 1.2	20	13	Expanded clay	3-6	68	93	Pujol and Tarallo, 2000
Upflow	2,840 - 4,080 (COD)	868.1 - 1,022.7	3.8	5.68 - 6.69	18.59 - 26.71	18	Glass raschig ring	26 x 25	97.5 - 100	99.4	Ong et al., 2002

mg/L and isopropanol concentration of 40 mg/L, more than 95% denitrification efficiency was observed. A maximum nitrogen removal rate of 1.2 kg NOx-N/m^3-day was obtained and sludge biofilm production was 1.38 kg VSS /kg NOx-N reduced. Dahab and Kalagiri (1996) used a cyclically-operated fixed-film bio-denitrification process to remove nitrate from drinking water and investigated the ability of a two-stage and a single-stage system to remove nitrate and residual organics from treated water. The two-stage systems were found to be more effective in reducing nitrates, nitrites and soluble organics in the wastewater than a single-stage system. Chui et al. (1996) studied the removal efficiency of nitrogen and organic matter in an anoxic/aerobic upflow fixed-bed filter. Without recirculation of effluent to the anoxic zone, 41% to 86% of the nitrogen was removed for an influent concentration of 250 mg N/L and a HRT of 6 – 24 hours. Vredenbregt et al. (1997) used pilot-plant scale fluidized-bed reactors for both biological nitrification and denitrification and found that a stable removal rate of 90 – 95% could be maintained.

Using a ultra-compact biofilm reactor (UCBR), Ong et al. (2002) showed that the volumetric nitrate-N conversion rate increased steadily from less than 0.5 kg/m^3-day to about 2.5 kg/m^3-day for first stage of the reactor with the conversion rate increasing rapidly to 10.5 kg/m^3-day in the second stage. In this study, the specific growth rate of the denitrifiers (μ) and the observed biomass growth yield coefficient (Y_{obs}) in the reactor varied from1.5 to 4.2 1/d and 0.2 to 0.65 g biomass/g NO_3^--N, respectively.

A comparison of combined and separated biological aerated filter (BAF) performance for pre-denitrification/nitrification of municipal wastewater by Rother et al. (2002) indicated that the single reactor system potentially offers substantial savings in investment costs and space requirements for a large-scale treatment plant as both reactor systems did not show significant differences in the treatment of organic carbon and nitrogen.

Some of the most recent work by Ong et al. (2002) and Lee et al. (2004) illustrates further progress in denitrification using packed bed (PB) columns for high rate nitrogen and carbon removals. Ong et al. (2002) reported that the removal rates of the anoxic-oxic packed bed system were 7.41 kg N/m^3-day and 28 kg COD/m^3-day. The N and COD removal efficiencies of the anoxic-oxic PB system were in the range of 97.5 - 100% and 98.6 - 99.4%, respectively. Also, Farabegoli et al. (2003) reported that denitrification rate was 2.4 kg/m^3-day for water temperature of 25 °C in a tertiary up-flow sand filter. Further investigation by Lee et al. (2004) showed that the highest achievable TN and COD removal rates were 47.2 g N/m^2-day and 158.0 g COD/m^2-day and a dual-stage PB system was capable of achieving TN and COD removal efficiencies greater than 99% and 98%, respectively.

Based on literature, packed bed columns can be operated with a loading rate as high as 7.41 kg N/m^3-day for denitrification and 28 kg COD/m^3-day for organic carbon removal. In addition, hydraulic retention time as low as 0.1 h did not impact the denitrification efficiency.

2.4.2 Denitrification Microbial Ecology

Heterotroph microorganisms that are responsible for denitrification includes *Achromobactor, Alcaligens, Arthrobacter, Agrobacterium,Bacillus, Corynebacterium, Chromobacterium, Escherichia, Flavobacterium, Hypomicrobium, Micrococcus, Moraxella, Neisseria, Paracoccus, Propionibacterium, Pseudomonas, Rhizobium, Rhodopsedomonos, Spirillum, Vibrio, Halobacterium,* and *Methanomonas* (Gaudy and Gaudy, 1988). These microorganisms are ubiquitous in wastewater and they use organic carbon such as organic matters, methanol, ethanol, and acetic acid as a carbon source. Alcohols produced by fermentation and volitale fatty acid (VFA) can be used as a carbon source for denitrification.

2.4.3 Factors affecting denitrification in biological filter

The temperature of the wastewater affects the performance of denitrifying biofilms. Denitrification can occur over the temperature range of 35 °C – 55 °C but the reaction rate slows down for low temperature. Lee et al. (2004) suggested that attached growth processes are less sensitive to temperature than suspended growth processes. Attached growth processes are minimally affected by dissolved oxygen concentration of 1 - 2 mg O_2/L whereas the suspended growth may be inhibited at concentration of 0.5 mg O_2/L. The optimum pH for denitrification suggested by Halling and Jorgensen (1993) is between 7.0 - 7.5. The hydraulic loadings and nutrient transport conditions to the biofilm surface, reactor configuration, and specific reactor condition are also important for denitrifying reactors. As discussed earlier, the concentration of electron donor (NO_3^-) and C/N ratio affects the denitrification performance. In the case of attached growth, they can be operated at low C/N ratio than suspended growth as the longer SRTs result in less required carbon for cell synthesis and a reduced C/N ratio (Halling and Jorgensen, 1993).

2.5 Oxidation-Reduction Potential (ORP)

Oxidation-reduction potential (ORP) may be used to monitor and characterize conditions in unaerated bioreactors (Koch and Oldham, 1985). Redox potential is a measurement of activity of electrons involved in oxidation-reduction reactions within an aqueous environment (Peddie et al., 1989). Koch and Oldham (1985) indicated that ORP may provide a very meaningful indication of reactor performance to monitor bacterial activity and may be used as basis for optimization for nitrogen removal although the indicated relationship between redox conditions and P release is not always consistent. In a study by Peddie et al. (1989), the ORP in bench-scale and pilot-scale was monitored within the aerobic sludge digesters undergoing alternating aerated and nonaerated conditions. The experimental results showed that real-time changes in ORP were related to changes in system chemistry and biological activity and corresponded to the presence of the appropriate electron acceptors. The reproducibility of the ORP profiles and sensitivity of measured potential appeared to make ORP an ideal parameter to control and optimize in bioreactor systems.

Enhanced nutrient removal by ORP-controlled aeration in laboratory-scale extended aeration treatment system using microelectrodes was investigated by Lo et al. (1994). In this study, the achievable COD, TKN, NH_3-N and TN removal efficiencies were 96%, 97%, 98% and 93%, respectively. Excellent phosphorus removal of 99.5 ± 0.1% was obtained under an ORP set point range of 70 - 180 mV.

Bishop and Yu (1999) used microelectrode techniques to study the stratification of microbial processes and the associated redox potential changes in biofilms. The redox potential profile changes were found to be correlated to shifts in microbial processes in the biofilms. A recent study by Li and Bishop (2004) on oxidation-reduction potential changes in aeration tanks and microprofiles of activated sludge floc provided a good relationship between ORP and nutrient removal along the length of the aeration tanks. The ORP values increased dramatically as organic matter was removed along the aeration tanks while nitrification occurred at higher ORP values (380 - 420 mV).

2.6 Phosphorus removal

In a bioreactor, phosphorus removal requires alternating anaerobic and aerobic stages for phosphorus release and uptake, respectively (Jeon et al., 2000; Park et al., 2001; Wang et al., 2001). In biological phosphorus removal, phosphorus-accumulating organisms (PAOs) store VFA in their cells as poly-β-hydroxy-butyrate (PHB) and release phosphorus under anaerobic condition. Under aerobic condition, PAOs take up phosphorus and utilized PHB stored in the cell. Uptake of phosphorus is usually higher than the release of phosphorus under anaerobic conditions resulting in a net uptake from the wastewater. Under anoxic conditions, luxury uptake of phosphorus by denitrifying phosphorus-accumulating organisms (dPAOs) using PHB and NO_3^--N may also occur (see Figure 4).

Proposed metabolic pathway for enhanced biological phosphorus removal (EBPR) is shown in Figure 5 with glucose as the sole carbon source (Jeon and Park, 2000). In this study, phosphorus-accumulating organisms (PAOs) could use glucose as a source for the anaerobic polyhydroxyalkanoate (PHA) synthesis. As shown in Figure 5, glucose was converted to lactic acid by lactate-producing organisms (LPOs) before acetate-using PAOs used the glucose as an energy source for the anaerobic PHA synthesis from acetate and the lactate-using PAOs anaerobically synthesized PHA from lactate at the expense of polyphosphate.

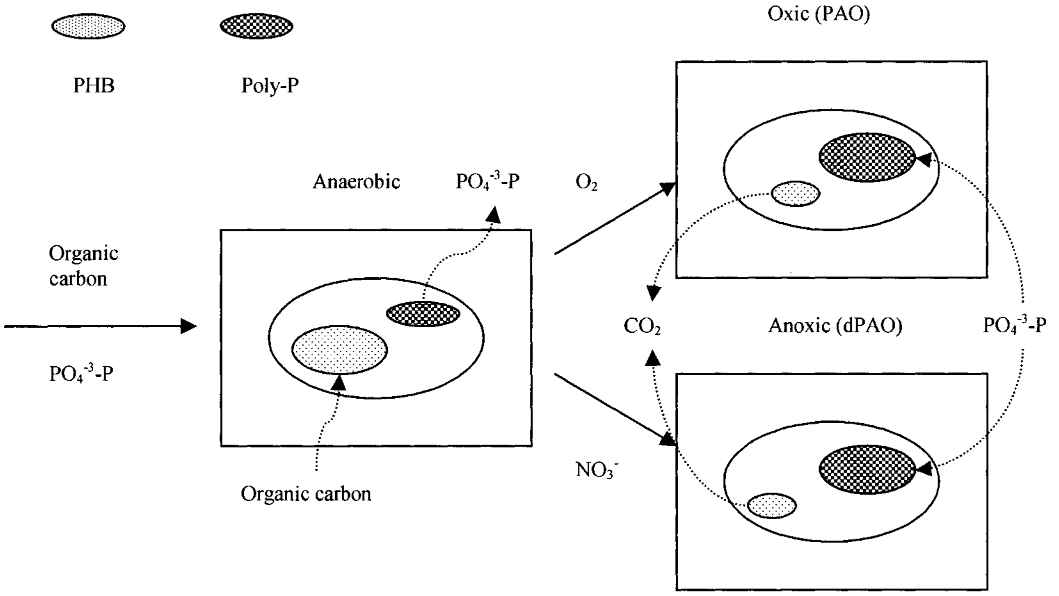

Figure 4. Phosphorus removal mechanism of phosphorus accumulating organisms (PHB - poly-β-hydroxy-butyrate, PAOs - phosphorus- accumulating organisms and dPAO - denitrifying phosphorus-accumulating organisms).

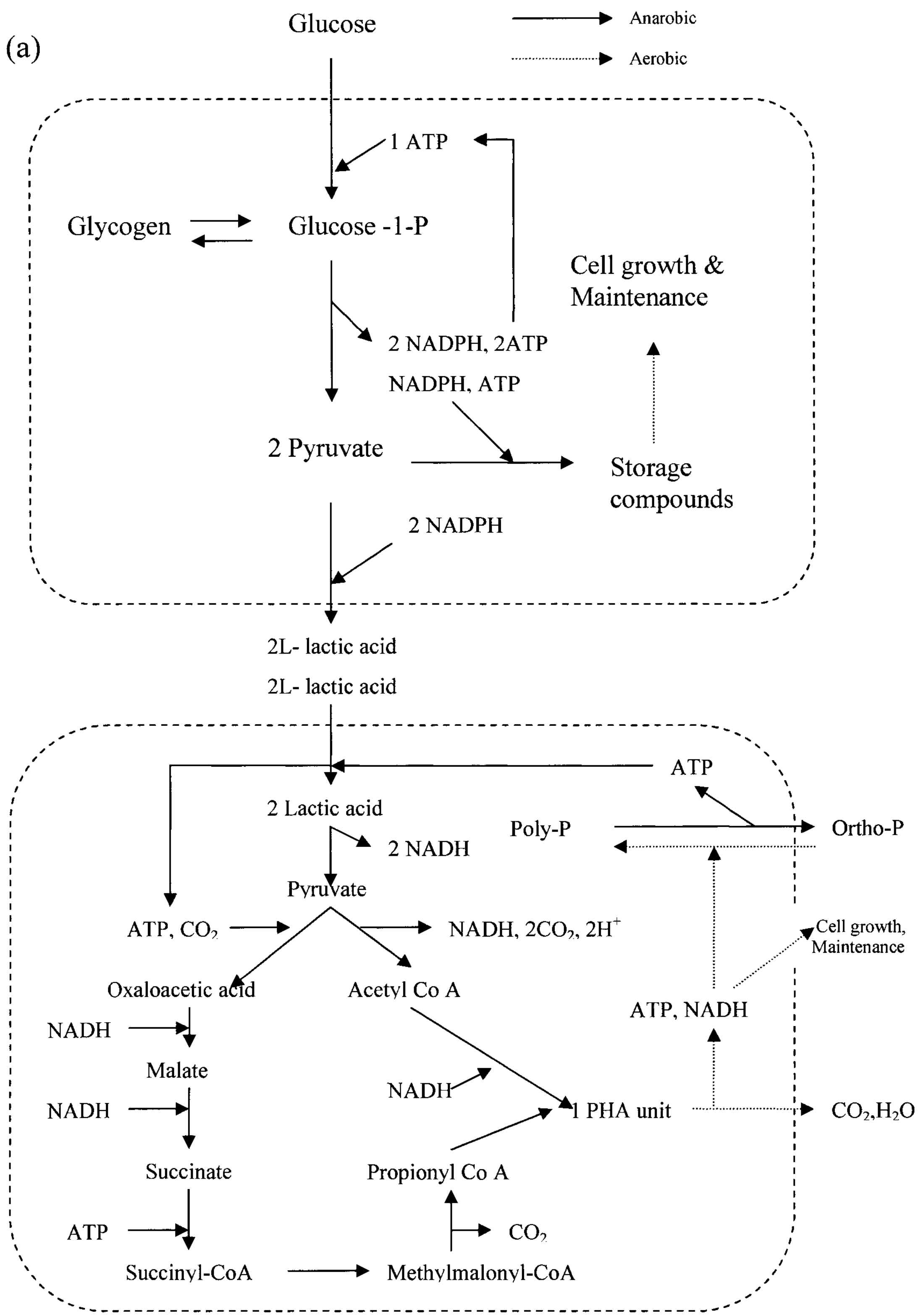

Figure 5. Methabolic pathway of glucose in EPBR (Jeon and Park, 2000).

2.6.1 Phosphorus Removal Studies

Unlike other conventional nutrient removal system where multiple tanks are required and long HRT is used, the partially aerated biological filter containing anaerobic, anoxic and aerobic conditions can achieve COD removal and ammonia oxidation as well as denitrification and suspended solid removal with short HRT due to the high biomass within a single reactor system (Chui et al., 1996; Gonçalves et al., 1992; Rogallar et al., 1900; Safferman et al., 2003; Tay et al., 2003). However, until recently, phosphorus removal in biofilm reactor had not been widely reported. Few investigations have studied phosphorus removal in biological filter systems (Chui et al., 1996; Gonçalves et al., 1992; Tay et al., 2003).

Gonçalves et al. (1992) demonstrated first that high phosphorus uptake could be achieved at the short HRT using alternating anaerobic-aerobic continuous biofilm reactor. Nitrification was complete from 50 mg N/L of ammonia to below 1 mg N/L and phosphorus was reduced from 14.3 mg P/L to below 1 mg P/L. Biological phosphorus removal in fludized bed biological reactor (FBBR) was investigated by Rovatti et al. (1995).
In this investigation, the feasibility of excess phosphorus uptake using a fludized bed was tested. A thin biofilm which has high surface/volume ratio was proved particularly effective for phosphorus removal. Only when strict anaerobic conditions were reached, effective phosphorus uptake was ensured. In this study, COD uptake reached 87.1% and phosphorus removal was 50.2%. Choi et al. (1996) conducted phosphorus removal experiments in a single reactor combining anaerobic and aerobic conditions. In this investigation, phosphorus removal efficiencies were 92% for total phosphorus (TP) and 90% for PO_4-P. For an influent TKN/TP of 2.5, 3.6 and 5.1, removal efficiencies for TP were 89%, 68% and 56%,

respectively. In the case of SRT, high removal efficiencies were observed with an SRT of 10 days. A recent work by Tay et al. (2003) showed that 72% phosphorus removal was obtained at the optimum COD:N:P ratio in a single upflow fixed-bed filter provided with anaerobic, anoxic, and oxic conditions. The experimental results revealed that phosphorus removal efficiency was affected more by its own concentration than that of COD and N concentrations.

2.7 Simultaneous nitrification, denitrification and phophorus removal

Simultaneous nitrification-denitrification (SND) is when aerobic-anoxic conditions coexist in the same reactor and within the biological flocs where nitrification takes place in the aerobic outer layer of the flocs and denitrification occurs in the anoxic kernel of the flocs with low dissolved oxygen (DO) concentration (Pochana and Keller, 1999). It is also possible for phosphorus uptake to occur since oxygen diffusional limitation creates anaerobic kernels where phosphorus release occurs (see Figure 6). Successful simultaneous nitrogen and phosphorus removal (SNDPR) has been reported in sequencing batch reactors (SBR) using anaerobic-aerobic cycling with low dissolved oxygen (DO) concentration, where phosphorus release occurs in the anaerobic phase and phosphorus uptake and simultaneous nitrification-denitrification occurs in the aerobic phase (Zeng et al., 2003; Keller et al., 1997; Pastorelli et al., 1999).

Two bench-scale sequencing batch reactors were operated to examine SND by Münch et al. (1996). The reactors were operated in parallel, with a HRT of 18 h, and a SRT of 15 d in temperature of 18 – 22 °C. They were able to reduce the total nitrogen by 74%. The influence of DO on the nitrification rate was described by a Monod kinetic model with a high oxygen half-saturation coefficient for autotrophic nitrifiers ($K_{O.A}$) of 4.5 mg/L. They also indicated the importance of measuring nitrite in the effluent to monitor the reactor performance so that aerobic denitrification was not overestimated. Using COD:N ratios of approximately 5:1 and 10:1 with an intermittently aerated and decanted single reactor, Yoo et al. (1999) found that the optimal maximum DO concentration for nitrogen removal was determined to be around 2.0 - 2.5 mg/L. The controlling factors for simultaneous nitrification and denitrification (SND) in a 2-stage, intermittent aeration process, designed for nitrogen and phosphorus removal were investigated by Zhao et al. (1999). In this investigation, acetate addition into the anaerobic zone can improve both nitrification and

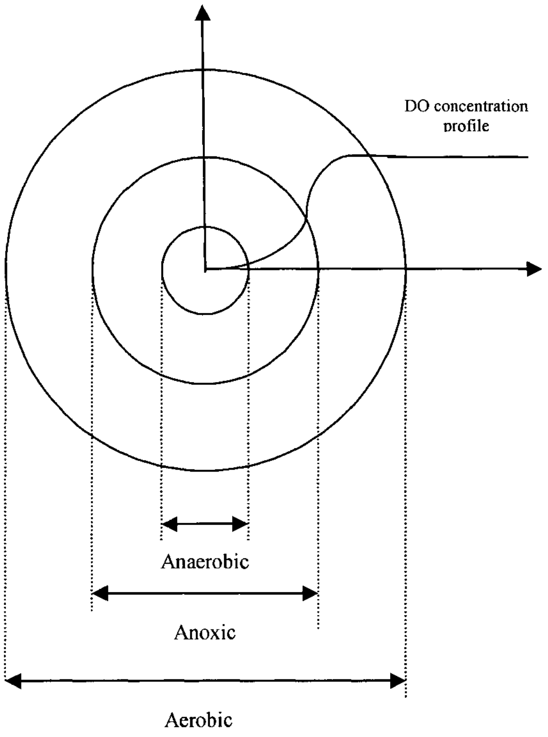

Figure 6. Dissolved oxygen in a large biomass floc at low bulk concentration (Zeng et al., 2003).

denitrification in the reactor. The NO_x removal due to SND reached up to 50% of the influent total nitrogen (TN) under low DO and intermittent aeration conditions.

The possibility of achieving biological phosphorus and nitrogen removal in a sequencing batch reactor was investigated by Garzón-Zúñiga and Gonzalez-Martínez (1996) using 36 hours duration of 10 hours anaerobic, 20 hours aerobic, 3 hours anoxic and 3 hours aerobic. The experimental results showed the removals of COD, phosphates and ammonia nitrogen of 89 ± 1%, 75 ± 15%, and 87 ± 10%, respectively for an initial COD of 205 mg/L, TN of 24 mg/L and PO_4^{-3}-P of 7.8 mg/L. Zeng et al. (2003) used a lab-scale sequencing batch reactor (SBR) to investigate simultaneous nitrogen and phosphorus removal. The reactor was constantly mixed with a magnetic stirrer (250 rpm) and operated with a cycle time of 4.8 h in temperature between 18° to 22° C. During the aerobic stage, the DO level was between 0.45 and 0.55 mg/L. These experimental results showed that nitrogen removal was via nitrite, not nitrate and denitrifying glycogen-accumulating organisms (dGAOs) rather than denitrifying polyphosphate-accumulating organisms (dPAOs) were responsible for the denitrification activity.

Bertanza (1997) studied the possibility of simultaneous nitrification and denitrification for high N removal efficiencies in extended aeration activated sludge plants, without alternating to anoxic and aerobic phases in the aeration tank. In this study, it was determined that high nitrogen removal efficiencies were achieved with F/M values of 0.15-0.2 kg BOD/kg MLSS-d with an ORP between 120 – 180 mV. Pochana and Keller (1999) performed experiments on SND in activated sludge systems and observed that up to 95% of total nitrogen removal was possible.

Under low DO conditions, a loss of inorganic nitrogen of up to 90% was observed in a rotating biological contactor (Helmer and Kunst, 1998) treating wastewater with TN of 250 mg/L. Using in-situ hybridization with fluorescence-labeled, rRNA-targeted oligonucleotide probes, they found heterotroph microorganisms of *Thiosphaera pantotropha* and *Nitrosomonas sp.,* which are capable of simultaneous nitrification and denitrification.

Sen and Dentel (1998) performed experiment for simultaneous nitrification denitrification (SND) in a fludized bed reactor (FBR) without physical separation of the aerobic and anoxic zones. The upflow velocity in the reactor was >0.79 m/min (48 m/h). The removal efficiencies of NO^-_3-N, NH_3-N and COD for the influent concentrations of 59.4, 47.0 and 809.4 mg/L were 50%, 97% and 83%, respectively.

A biofilm process Siporax^{TM} using a porous Raschig-ring type filter for simultaneous removal of ammonia and nitrate was examined by Menoud et al. (1999). They demonstrated the feasibility of SND inside a single reactor containing Siporax^{TM} Raschig packing. Maximum nitrification and denitrification capacities of 0.6 and 0.83 g N/L-day were observed, respectively. A new biological aerated filter (BAF) for nitrogen removal based on simultaneous nitrification and denitrification was developed by Puznava et al. (2001). At low dissolved oxygen concentration from 0.5 to 3 mg O_2/L, the biofilm was not fully penetrated with oxygen and denitrification was carried out in a large part of the biofilm in the filter. The total nitrogen residual was 20 mg TN/L with an elimination efficiency of 60 to 70% at applied TN loads of 0.1 up to 1.2 kg TN/m^3-day.

Hamamoto et al. (1997) developed a technology to simultaneously remove nitrogen and phosphorus under aerobic and anaerobic conditions in a single reactor. A steel plate tank, 1.5 m wide x 5.6 m long x 2.5 m high was used as the reactor. In their investigation, influent

BOD values ranged between 85 and 410 mg/L while the average effluent BOD was 2.3 mg/L. When average influent TN and TP concentrations were 46 mg/L and 7.3 mg P/L at flow rates between 339 and 477 m^3/day, average nitrogen and phosphorus removal rates in the full-scale plant were 96% and 93%.

2.8 Summary

BAF systems are attractive in term of cost and space needed. High substrate loadings of 3.5 – 11.9 kg COD/m^3-day and 2.7 kg NH_3-N/m^3-day have been reported as compared to 0.12 kg COD/m^3-day and 0.2 kg NH_3-N/m^3-day for activated sludge plants. In addition, HRT as low as 10 min were reported to have no effect on the BAF performance. Furthermore, the active biofilm on the media in the BAF may be suitable to accumulate temperature-sensitive nitrifier for low temperature operation. However, there are not many BAF systems in USA and not much work has been done on the performance of BAFs at low temperatures. In conventional BAF systems, the investigations have focused on removal of ammonia and organic carbons in wastewater. However, as the discharge limits for wastewater treatment plants shifts from removing ammonia and carbon sources to removal of nitrogen, phophorus as well as carbon, the BAF can be modified to effectively remove nutrients and organic carbon. The impact of various operating parameters such as organic and hydraulic loading rates and recirculation ratios need to be investigated along with the impact of low temperature and the effect of temperatures on nitrification. Modifying BAF systems by creating anaerobic, anoxic and oxic zones or by adding alternating aerobic and anaerobic columns for removals of nitrogen, COD and phosphorus need further investigation.

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CHAPTER 3. IMPACT OF MEDIA TYPE AND VARIOUS OPERATING PARAMETERS ON NITRIFICATION IN POLISHING BIOLOGICAL AERATED FILTERS (BAFS)

A paper to be submitted to Water Research

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ABSTRACT

Three BAFs made of PVC pipe with a diameter of 75 mm were constructed and operated at 13 °C. Each BAF has a media depth of 1.7 m. Media used were 5-mm gravel, 5-mm lava rock, and 12.5-mm diameter by 15-mm long plastic rings. Influent sCOD and ammonia concentrations in the feedwater were approximately 50 mg/L and 25 mg/L, respectively, simulating the effluent from aerated lagoons. Ammonia removals at hydraulic retention time (HRT) of 2 hour for gravel, lava rock, and plastic ring were 98.5, 98.9, and 97.8%, respectively. With 100 and 200% recirculation at 1-hr HRT, ammonia removals (%) improved from 90.1 to 96, 76.5 to 90, and 65.3% to 79.5% for gravel, lava rock, and plastic ring, respectively. Experiments with different COD/NH_3-N (0.1 to 8) ratios indicate that as COD/NH_3-N ratio increased, ammonia removal decreased linearly. Based on ammonia and sCOD loading studies at different hydraulic loading rates, an optimum ammonia loading applied of 0.6 kg NH_3-N/m^3-day was determined for all three BAFs with different media types. The zero-order biotransformation rates in gravel media were significantly similar at about 14 mg NH_3-

N/ L-hr for all three different HRTs tested. Overall biotranformation rates in gravel media at different HRTs were found to be higher than the biotransformation rates for lava rock and plastic ring media types. The experimental results demonstrate that BAF can be used as an add-on system to aerated lagoons or as a secondary treatment to meet ammonia discharge limits.

Keywards: nitrification, biological aerated filter, recirculation, NPDES

INTRODUCTION

In the Midwest of the United States, aerated lagoons are used widely by small communities for the treatment of domestic wastewater. Their wide spread use is credited to their relatively low operational costs and low maintenance requirements. Most of these systems were designed for 5-day carbonaceous biochemical oxygen demand ($CBOD_5$) and suspended solids removal and are not designed for ammonia removal. Over the past several years, effluent ammonia limits have been added to the National Pollution Discharge Elimination System (NPDES) permits of these facilities. These aerated lagoons, however, are unable to meet the effluent ammonia concentrations during the late winter and early spring months - often exceeding the NPDES permit requirements. The cause of this increase in effluent ammonia concentration is due to a decrease in nitrification activity over the winter months.

A survey of ten aerated lagoons in Iowa by Ong and co-workers indicated that some aerated lagoons performed better than other aerated lagoons under the same climatic conditions (Van Dyke et al., 2003). Ong and co-workers concluded that the better performance of these facilities may be due to the longer retention time of the aerated lagoons. Providing longer retention time for aerated lagoons is not economical since a large area will be needed. If detention time plays an important role then recirculation of the wastewater may improve nitrification. Just as important as the retention time is the maintenance of a large biomass of nitrifiers even if the nitrification rates are slow at low temperatures. With aerated lagoons, maintaining a large biomass is not practical because of the simple construction and hydraulics of the lagoons. A possible solution that combines both variables is the biological aerated filter (BAF). BAFs may be added as a treatment process after the aerated lagoons or used directly as a secondary treatment. BAFs are becoming increasingly popular in Europe due to its compactness (M'Coy, 1997; Peladan et al., 1996; Pujol, 2000).

BAFs have a relatively low capital cost investment, easy operation and are more efficient than activated sludge systems (M'Coy, 1997; Bigot et al., 1999; Hodkinson et al., 1999; Mendoza-Espinosa and Stephenson, 1999; Wheale and Cooper-Smith, 1995). BAF consists of a submerged, granular media that treat carbonaceous and nitrogeneous matter using biomass fixed to the media and capturing the suspended solids in the media. Over recent years, different configurations were developed to improve the treatment efficiency of BAFs. BAFs may be designed as downflow (Paffoni et al., 1990) or upflow systems (Rogalla et al., 1992). Pilot-scale studies by Peladan et al. (1996) seemed to suggest that BAFs might be effective in removing ammonia when the wastewater is recirculated. Using an upflow BAF, 90% ammonia removal was observed at an ammonia loading rate of 4 kg/m^3-day, a hydraulic loading rate of 30 m^3/m^2-h and at a temperature of 17.5 °C (Peladan et al., 1996). The hydraulic loading rate used by Peladan et al. (1996) was about 4 times higher than the typical hydraulic loading rates for BAFs.

In a typical aerated lagoon system, the BAF can be located immediately as an add-on system after the second cell of the aerated lagoon. This would minimize modification and cost to the aerated lagoons. The objectives of this study were to investigate the application of BAFs with three different media types for nitrification under various operating conditions and to provide a preliminary assessment of its feasibility for small communities to meet their NPDES limits. The media selected must be easily obtained and readily available. The proposed research is focused on conducting BAF studies at various operating parameters such as chemical oxygen demand (COD) and nitrogen loading rates, hydraulic loading rates, COD/NH_3-N ratio, and effluent recirculation to ascertain its effectiveness at operating temperatures of 13 °C.

MATERIALS AND METHODS

Three down-flow pilot-scale BAFs made of PVC pipe as shown in Figure 1 were set up. Each BAF has a diameter of 75 mm to minimize wall effects and a media depth of 1.7 m. For each BAF, fine gravel, lava rock, and plastic ring material were used as packing materials. The size of plastic ring to be used was 12.5 mm diameter and 15 mm long and was made from long PVC pipes. Gravel and lava rock used had an average diameter of 5 mm (between US Standard Sieve No. 2 (5.36 mm) and No. 4 (4.36 mm)). The porosities for gravel, lava lock, and plastic ring were estimated to be 0.36 ± 0.01, 0.53 ± 0.02, and 0.76 ± 0.01, respectively. The specific surface areas were measured to be 0.93×10^6, 3.81×10^6 and 3.67×10^2 m^2/m^3 for gravel, lava rock and plastic rings, respectively.

Three Masterflex Model 75530-70 pumps were used to feed synthetic wastewater to the three BAFs while another three Cole-Parmer 7553-50 pumps were used for recirculation of the treated effluent through the filters. An Emerson Model S55JXDRL-2565 pump was used for backwashing. Air was introduced into the BAF using filtered in-house compressed air via air diffusers placed at the bottom of the BAF. The air flow rate was measured using a Gilmont ball flow meter (Barrington, IL). Sampling points were located throughout the length of the BAF column as indicated in Figure 1. Two Fisher Scientific Model 9105 coolers were used to maintain the temperature of feed solutions and the synthetic wastewater on the top of BAFs at 13 ±1 °C.

To assess the effectiveness of BAFs under varying conditions, a synthetic wastewater was used as feed wastewater (see Table 1). The synthetic wastewater contained organic carbon, essential nutrients, ammonia, and minerals to simulate the characteristics of treated wastewater from aerated lagoons. COD and ammonia concentrations of the feed water were approximately 50 mg/L and 25 mg/L, respectively.

The BAFs were seeded using activated sludge from the aeration tank of a municipal wastewater treatment plant in Iowa. The three BAFs were operated initially for a period of time at room temperature of 24 °C until the soluble chemical oxygen demand (sCOD) and ammonia removal were above 90% before the operating conditions at 13 ± 1 °C were varied to assess the BAF performance.

Experiments were conducted for three different hydraulic retention times (HRTs), 0.5, 1, and 2 hours. Nominal flow rates were 95, 190, and 380 L/day leading to hydraulic loading rates from 1 to 4 m^3/m^2-hr. The operational characteristics of the BAFs are summarized in Table 2. Recirculation rates of 100 and 200% of the influent rates were tested. Measurement throughout the study included influent and effluent concentration of sCOD, Total Kjeldahl Nitrogen (TKN), alkalinity, ammonia, nitrate, and nitrite. Other parameters that were measured include pH, temperature, DO, and flow rates. Samples used for the analyses were one-hour composite samples. Analyses were conducted according to Standard Methods (APHA, 2002).

RESULTS AND DISCUSSION

BAF Operational Performance

The performances of the three BAFs for three different media are shown in Figures 2 to 4. The BAFs were operated for more than 250 days on a continuous basis but only 150 days of operations are shown in Figures 2 to 4. The BAFs were initially operated at 2-hour HRT and at 24 °C and then at 13 ± 1 °C. At this temperature, the BAF was operated at 2-hour, 1-hour, and 0.5-hour HRT with and without recirculation at 100 or 200%. Linear velocities within the columns for 2-hour HRT, 1-hour HRT, 0.5-hour HRT, and 1-hour HRT plus 100% recirculation and 1 hour HRT plus 200 % recirculation were 0.85, 1.7, 3.4, 3.4, and 5.1 m/hr, respectively. Backwashing was performed every two days for 0.5-hour HRT, every four days for 1-hour HRT, and once per week for 2-hour HRT and for flows with 100% and 200% recirculation. For each operating condition, the BAFs were operated over 2 to 3 weeks period until steady state conditions were assumed. Steady state conditions were assumed when 3 consecutive effluent COD concentrations did not vary more than 10 %. The suspended solids for BAF were measured initially for 2-hours HRT and were found to be 6 ± 1.7 mg/L. Subsequent SS measurements were not conducted.

With a HRT of 2 hours at a temperature of 24 °C and 13 °C (day 0 to 55), the sCOD removals were greater than 95% and the effluent NH_3-N concentration were close to zero. The results show that when HRT was lowered (increase in hydraulic loading rate), nitrification decreased accordingly. As shown in Figures 2 to 4, the nitrification efficiency improved with recirculation. The impact of various operating parameters on nitrification are summarized and discussed below.

Impact of Media Type and HRT Without Recirculation on Removal Efficiencies

The average removal percentages (with 95 % confidence intervals) of NH_3-N, sCOD, TKN, and alkalinity for different media types and HRTs without recirculation are presented in Figure 5. For HRT of 2 hours, more than 97% of ammonia and 92% of sCOD were removed for all three media types. As a direct evidence of nitrification, more than 83% of the influent alkalinity was consumed. The average ratios of alkalinity consumed and ammonia oxidized were 6.7, 7.5, and 7.6 mg $CaCO_3$/mg NH_3-N for gravel, lava rock, and plastic ring media, which are reasonably close to the theoretical value of 7.14 mg $CaCO_3$/mg NH_3-N. TKN removals were observed to be about 97% for all three media types at 2 hours HRT. The average effluent ammonia, nitrate and nitrite concentration under different HRTs and media types are presented in Tables 3, 4, and 5.

For HRT of 1 hour, approximately 83% of ammonia and 89% of sCOD were removed in the BAF with gravel media, while removal percentages in BAF with the lava rock and plastic ring were 65% ammonia and 88% sCOD and 59% ammonia and 86% sCOD, respectively. In the BAF with gravel media, approximately 75% alkalinity and 84.5% TKN removal were obtained while approximately 58 % and 52% alkalinity removal and 69.6% and 63.4% TKN removal were obtained in lava rock and plastic ring BAFs, respectively.

For HRT of 0.5 hour without recirculation, the percent removals for ammonia for all three media types were less than 40%. The percent of alkalinity and TKN consumed was also below 40% and 44%. However, sCOD removals for all three BAFs were above 80%. The impact of HRT on ammonia removal was obvious – there was a reduction in nitrification for a corresponding lower HRT.

For a wastewater temperature of 13 °C, an HRT of 2 hours without recirculation will be needed for all three media to obtain an ammonia concentration of less than 1 mg/L for an influent ammonia of 25 mg N/L and a sCOD of 50 mg/L. Similarly both lava rock and plastic ring can provide the same removal at 2-hour HRT. But all three media types cannot achieve 1 mg N/L effluent ammonia at an HRT of 1 hour without recirculation. However, of the three media used, gravel gave the lowest mean effluent concentrations of ammonia, alkalinity, and sCOD. A possible reason for better performance with gravel is the high specific surface area of gravel resulting in large biomass on media. In addition, it was found that the lava rock breaks into smaller particles easily.

Impact of Recirculation on Ammonia and sCOD Removal

The impacts of 100% and 200% recirculations of effluent wastewater on ammonia and sCOD removal are presented in Figure 6. Recirculation experiments were conducted at 1-hour HRT since most ammonia was removed at an HRT of 2 hours. The results showed that with 100% recirculation at 1 hour HRT and 13 ± 1 °C, ammonia removals improved from 83.3 ± 6.51 to 90.1 ± 1.4%, from 65.3 ± 7.92 to 76.5 ± 3.2%, and from 58.8 ± 3.85 to 65.3 ± 1.2% for gravel, lava rock and plastic ring, respectively. With 200% recirculation, ammonia removal improved further to 96 ± 0.5%, 90 ± 0.4% and 79.5 ± 1.8% for gravel, lava rock and plastic ring, respectively. Also, sCOD removal with 100% and 200% recirculation at 1-hour HRT and 13 °C improved to 92.6 ± 4.3and 93.6 ± 2.2%, 88.5 ± 3.6and 92.6 ± 1.6%, and 87.4 ± 2.7and 91.7 ± 1.7% for gravel, lava rock and plastic ring, respectively.

Several researchers observed moderate improvement in nitrification efficiency for an increase in hydraulic loading (Tschui et al., 1994; Peladan et al., 1996; Husovitz, 1998) which may be attributed to enhanced mass transfer of substrate into the biofilms while maintaining the sCOD and ammonia mass loading constant. Another possible explanation is that the higher recirculation rates result in the dilution of ammonia and carbon substrate and therefore allowing for transformation of ammonia over a shorter HRT. A simple mass balance will indicate that higher ammonia removal was not totally due to dilution. At an HRT of 1 hour and without recirculation for the gravel, the effluent ammonia concentration was approximately 4 mg N/L. With an influent ammonia concentration of 25 mg N/L, a 100 % recirculation will result in an influent concentration after mixing of 14.5 mg N/L. With a recirculation of 100%, the HRT within the column is now 0.5 hour. Assuming the ammonia removal rate at 0.5 hour HRT without recirculation applies, i.e., about 40% ammonia removal, the estimated effluent ammonia concentration for 1 hour HRT and 100% recirculation should be 14.5 mg N/L x 0.6 = 8.7 mg N/L. Since the actual effluent ammonia concentration was 2.5 mg N/L which is lower than the estimated 8.7 mg N/L, it is possible that recirculation had an impact on nitrification within the column even though the water linear velocity has doubled through the gravel media.

Changes in Ammonia and Nitrate Concentrations with Media

The changes in ammonia and nitrate concentration within the bioreactor at different HRTs and media types are presented in Figures 7 and 8. Three different ammonia and COD hydraulic loadings were applied by changing the HRTs while maintaining influent COD concentration of 50 mg/L and ammonia concentration of 25 mg N/L.

For all three media types at an HRT of 2 hours (Figure 7a), complete nitrification of ammonia was observed at a depth of 1.4 m. The appearance of nitrate was in good agreement with the disappearance of ammonia (see Figures 7 and 8). Of the three media types, gravel showed the best ammonia removal.

At HRTs of 1 and 0.5 hour, significant differences in the ammonia removal were observed for the three media types. Ammonia effluent concentration was 4.3 ± 1.4 mg/L for gravel, 8.9 ± 1.9 mg/L for lava lock, and 10.8 ± 1.3 mg/L for plastic ring at 1 hour HRT. For an HRT of 0.5 hour, negligible nitrification was observed within the first 0.7 m of all three columns (see Figure 8c). This may be due to the increase of hydraulic loading rates resulting in insufficient bed volume available for nitrification.

Effect of COD/NH_3-N (C/N) Ratio on Nitrification

The effects of substrate COD/NH_3-N (C/N) ratio on the ammonia removal (%) for six different C/N ratios are presented in Figure 9. As the C/N ratio increased, the ammonia removal (%) and ammonia mass removed per unit volume (%) decreased linearly. Correspondingly, the effluent ammonia and COD concentrations increased as C/N ratio increased (see Figure 9a and 9b). Of the three media types, the gravel reactor showed the best percentage of ammonia removal efficiency and ammonia mass removed at different C/N ratios.

As shown in Figure 9a and 9b, BAFs operating at low C/N ratio (under the ratio of C/N = 2) gave ammonia removal (%) of approximately 83, 63, and 60% (see Figure 9) and the ammonia mass removed were approximately 0.5, 0.39, and 0.35 kg NH_3-N/m^3-day for gravel, lava rock, and plastic ring, respectively (see Figure 9b). At high C/N ratio (above the ratio of C/N = 8), ammonia removal (%) was 32, 26, and 20 % and the ammonia mass removed was 0.18, 0.15, and 0.12 kg NH_3-N/m^3-day for gravel, lava rock, and plastic ring, respectively (see Figure 9).

The experimental results imply that for low organic loading, the nitrification activity in the filter was not restrained but was limited at higher COD concentrations. A possible reason is the competition in the oxygen uptake between the heterotrophs and nitrifier where nitrifiers inside the biofilm are impacted by the rapid growth of heterotrophs in the presence of higher COD which may retard the oxygen contact frequency of ammonium oxidizing bacteria or nitrifying bacteria (Satoh et al., 2000).

Impact of Ammonia and sCOD Loading

For the various experimental runs, the ammonia and sCOD loading rates can be plotted against the percent ammonia removed, effluent ammonia concentration, and mass of ammonia removed as shown in Figures 10a, 10b, and 10c. The percent ammonia removal (%) for all three media decreased as the ammonia and sCOD loading increased (see Figure 10a) while the effluent ammonia concentrations for all three media increased (see Figure 10b). The ammonia mass removed for all three media increased until about 0.6 kg NH_3-N/m^3-day of applied ammonia loading before the mass ammonia removed decreased for increasing ammonia loading.

There were significant differences in ammonia mass removed amongst the three media types as the applied ammonia loading increased. As explained earlier, a possible reason for the decrease in mass of ammonia removed may be due to the competition between the heterotrophs and autotrophs. These observations suggest that higher hydraulic sCOD mass loading produced a competitive effect on both the ammonium oxidation and the nitrite oxidation. This is in agreement with the study by Gilmore et al. (1999) where high $CBOD_5$ mass loads resulted in an increase in effluent ammonia concentration. Nogueira et al. (2002) speculated that the formation of heterotrophic microorganism layer on the top of nitrifying biofilm in a continuous circulating bed reactor limited the nitrifiers oxygen supply and growth.

Overall Ammonia Biotransformation Rates in BAFs

The ammonia biotransformation rates, k, may be used to describe nitrification occurring within the BAF and can be used in the preliminary sizing of BAF systems. Studies conducted by Boller and Gujer (1986) indicated that the overall ammonia biotransformation rate within the BAF systems may be expressed as a zero-order reaction with respect to the ammonia concentration. Using a simple input-output model and under steady state conditions, the influent and effluent concentration is related to the zero-order reaction rate by

$$\mathbf{C - C_o = -k\,t \quad = \quad -k\,h/v} \qquad \mathbf{(1)}$$

where C_o is the influent ammonia concentration (mg/L), C is the actual ammonia concentration (mg/L), k is the ammonia biotransformation rate (mg NH_3-N/L-hr); and t is time of reaction (hrs), h is the depth of the sand media (m) and v is hydraulic loading rate (m/hr). A plot of C versus t provides a straight line curve with a slope equal to k and an intercept equal to influent ammonia concentration as shown in Figures 11, 12 and 13 for gravel, lava rock and plastic ring, respectively .

The estimated zero-order biotransformation rates along with their 95% CI are summarized in Table 6. The overall biotranformation rates in gravel at different HRTs were higher than the rates for lava rock and plastic ring media types indicating that gravel may be a suitable material for low-cost BAFs. For the gravel, the zero-order biotransformation rates were statistically similar for all three HRTs tested (2, 1, and 0.5 hour). In the case of lava rock media, the transformation rates were statistically similar for 2-hour and 1-hour HRT and 1-hour and 0.5-hour HRT but 2-hour HRT was different for 0.5-hour HRT. The biotransformation rates for the plastic rings were different at different HRTs. Statistically, the zero-order biotransformation rates for lava rock and gravel were similar for 1-hour and 2-hour HRT.

CONCLUSIONS

Experimental results for three BAFs using different media showed BAF may be used as an add-on system to aerated lagoons or as a secondary treatment to meet ammonia discharge limits. Of the media tested, gravel was found to provide more than 96% nitrification at HRTs as low as 0.5 hr with 200% recirculation. Lava rock media is not a suitable media as the media breaks and is not resistant to attrition when backwash is performed. In the case of plastic rings, the lower specific surface area resulted in lower nitrification rates as compared to the other two media types.

Experimental results showed that with 100 and 200% recirculation at 1 hr HRT and 13 ± 1 °C, ammonia removals (%) improved to 90.1 ± 1.4 and 96 ± 0.5%, 76.5 ± 3.2 and 90

± 0.4%, and 65.3 ± 7.92 and 79.5 ± 3.2% for gravel, lava rock and plastic ring, respectively. COD removal (%) with 100 and 200% recirculation at 1 hr HRT and 13 °C improved to 92.6 ± 4.3 and 93.6 ± 2.2%, 88.5 ± 3.6 and 92.6 ± 1.6%, and 87.4 ± 2.7 and 91.7 ± 1.7%, respectively. By varying the COD/NH_3-N (C/N) ratios, experimental results indicate that ammonia removal decreased when the C/N ratios increased. This may be due to interspecies competition for substrates between autotroph and heterotroph microorganisms.

Based on the ammonia and COD loading, an optimum ammonia loading of 0.6 kg NH_3-N/m^3-day was determined for all three different media types. The zero-order biotranformation rates in gravel were significantly similar at about 14 mg NH_3-N/L-hr for all three HRTs tested.

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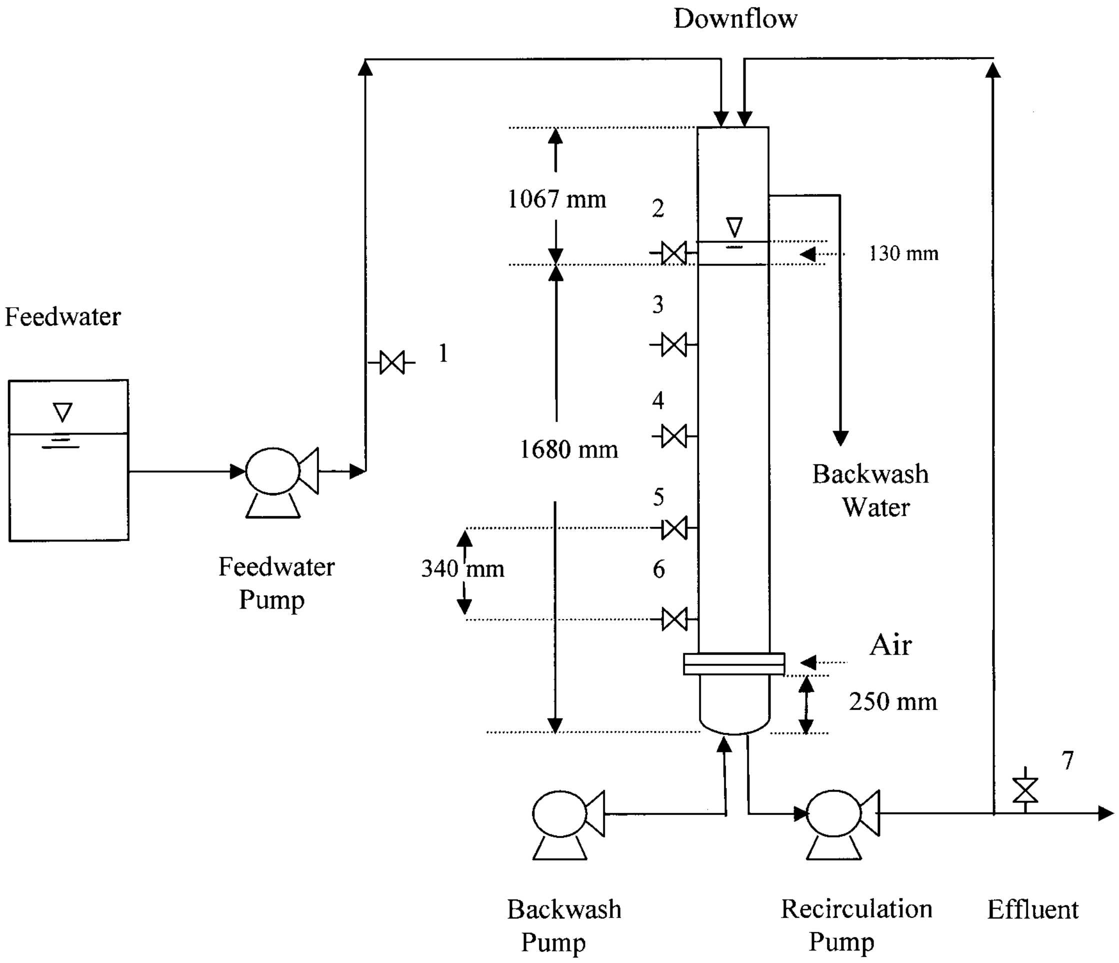


Figure 1. Schematic diagram of BAF system

Table 1. Composition of synthetic wastewater.

Ingredient	**Concentration (mg/L)**
Calcium Chloride ($CaCl_2$)	20
Magnesium Sulfate ($MgSO_4$)	3
Ferric Chloride ($FeCl_3$)	2
Sodium Biphosphate (NaH_2PO_4)	50
Sodium Bicarbonate ($NaHCO_3$)	200*
Potassium Chloride (KCl)	4
Sodium Acetate (CH_3COONa)	26
Nutrient Broth	15
Ammonium Chloride (NH_4Cl)	95
Isomil	**

* Alkalinity is reported in mg/L of total alkalinity as calcium carbonate.

** 500 mL solution in 400 L of synthetic wastewater. Stock solution consists of 22 mL liquid Isomil in 500 mL of distilled water.

Table 2. Experimental operating condition for biological aerated filters.

Parameter	**Value**
Diameter of BAF	75 mm
Media Depth	1.7 m
Flow Rates	95, 190 and 380 L/day
Hydraulic Retention Times	0.5, 1 and 2 hr
Recirculation ratios	100%, 200%
Backwash Flow Rate	3.33 L/min
Air Flow Rate	300 mL/min
Temperature in Reactor	13 ± 1 °C
Influent NH_3-N	25 mg/L
Influent COD	50 mg/L
Influent Alkalinity	200 mg/L as $CaCO_3$
Influent Total Kjeldahl Nitrogen	30 mg N/L
Influent pH	8.0 ± 0.1

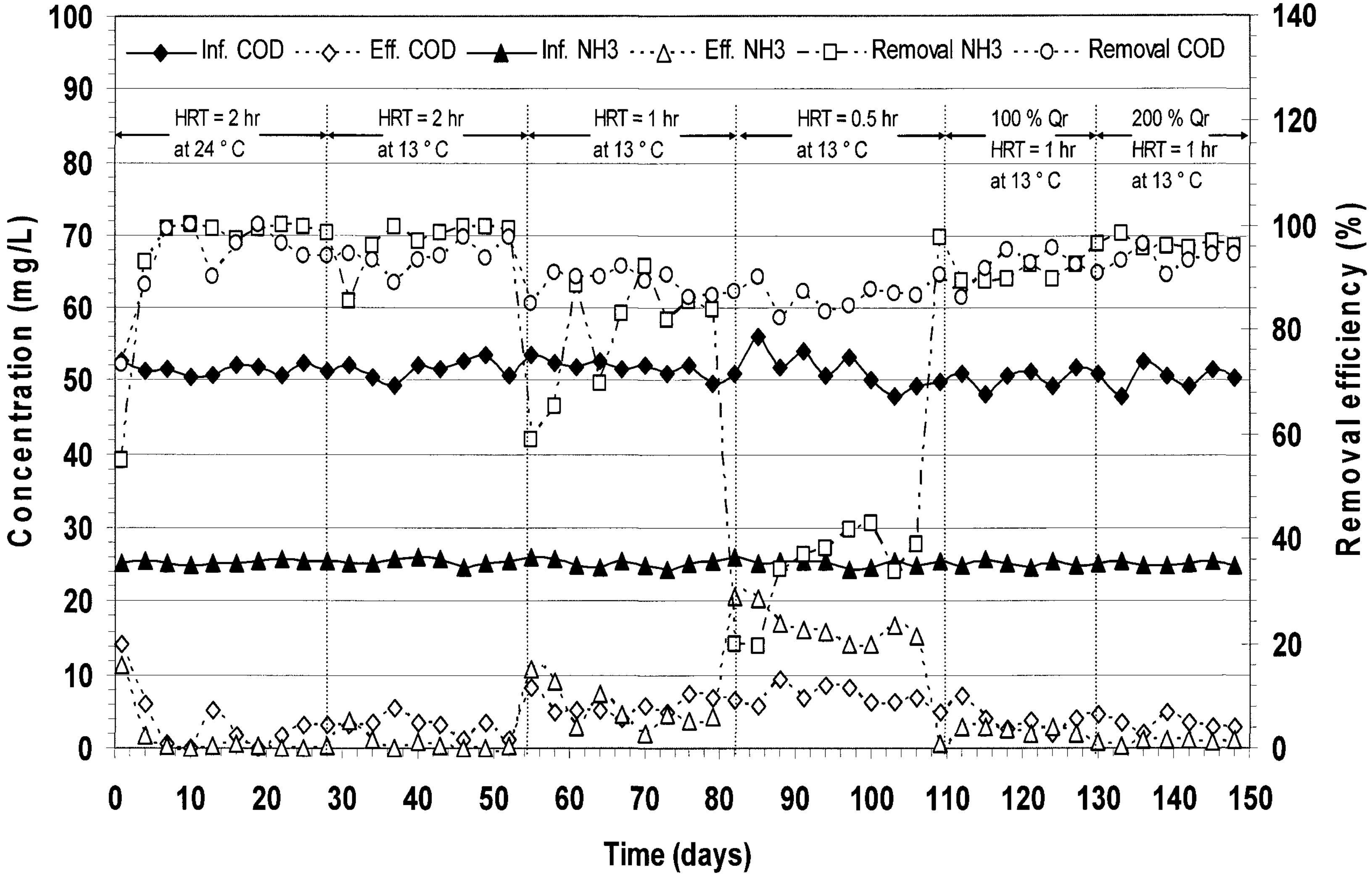


Figure 2. BAF performance for gravel media at different HRTs and recirculation (Qr).

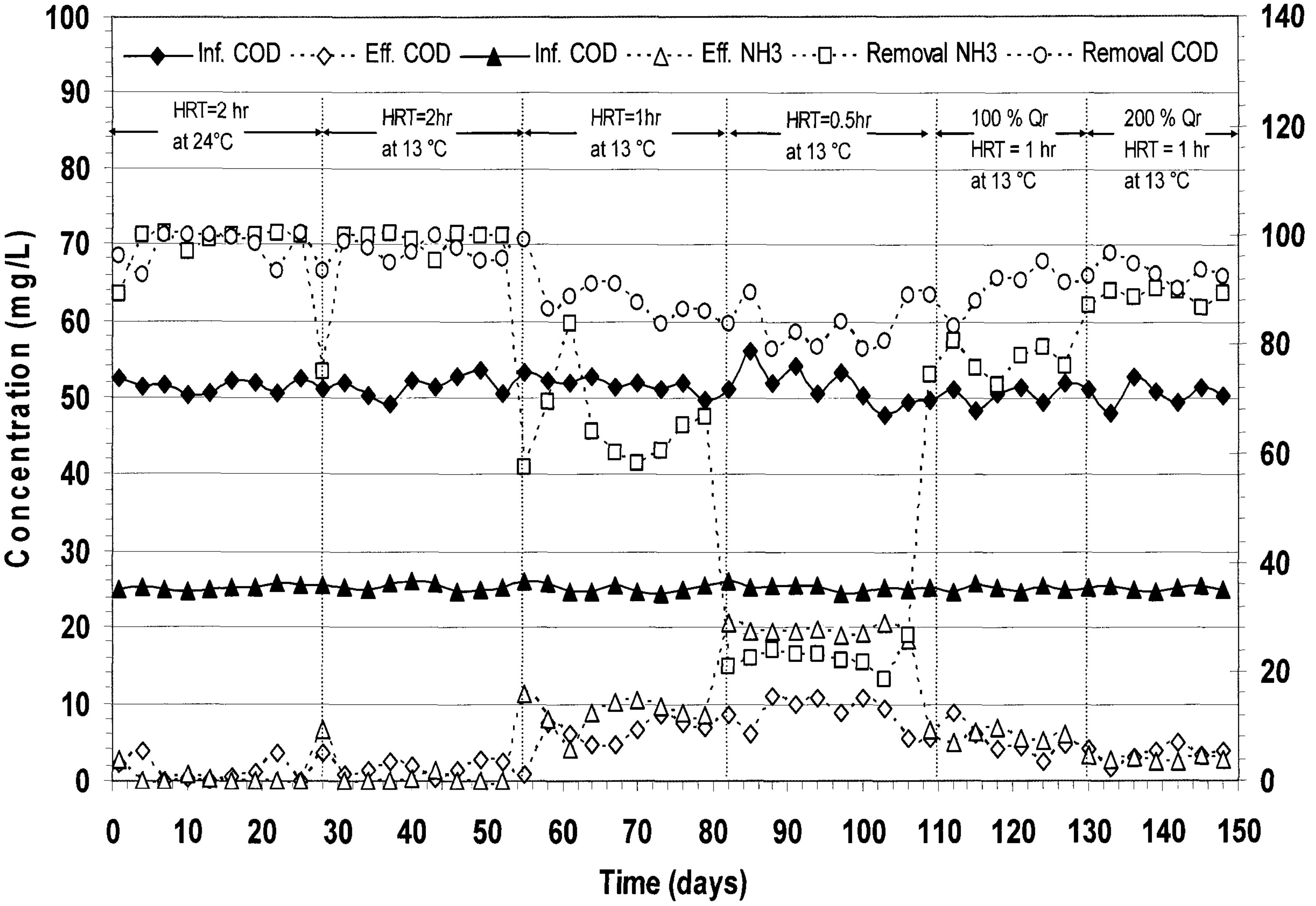


Figure 3. BAF performance for lava rock media at different HRTs and recirculation (Qr).

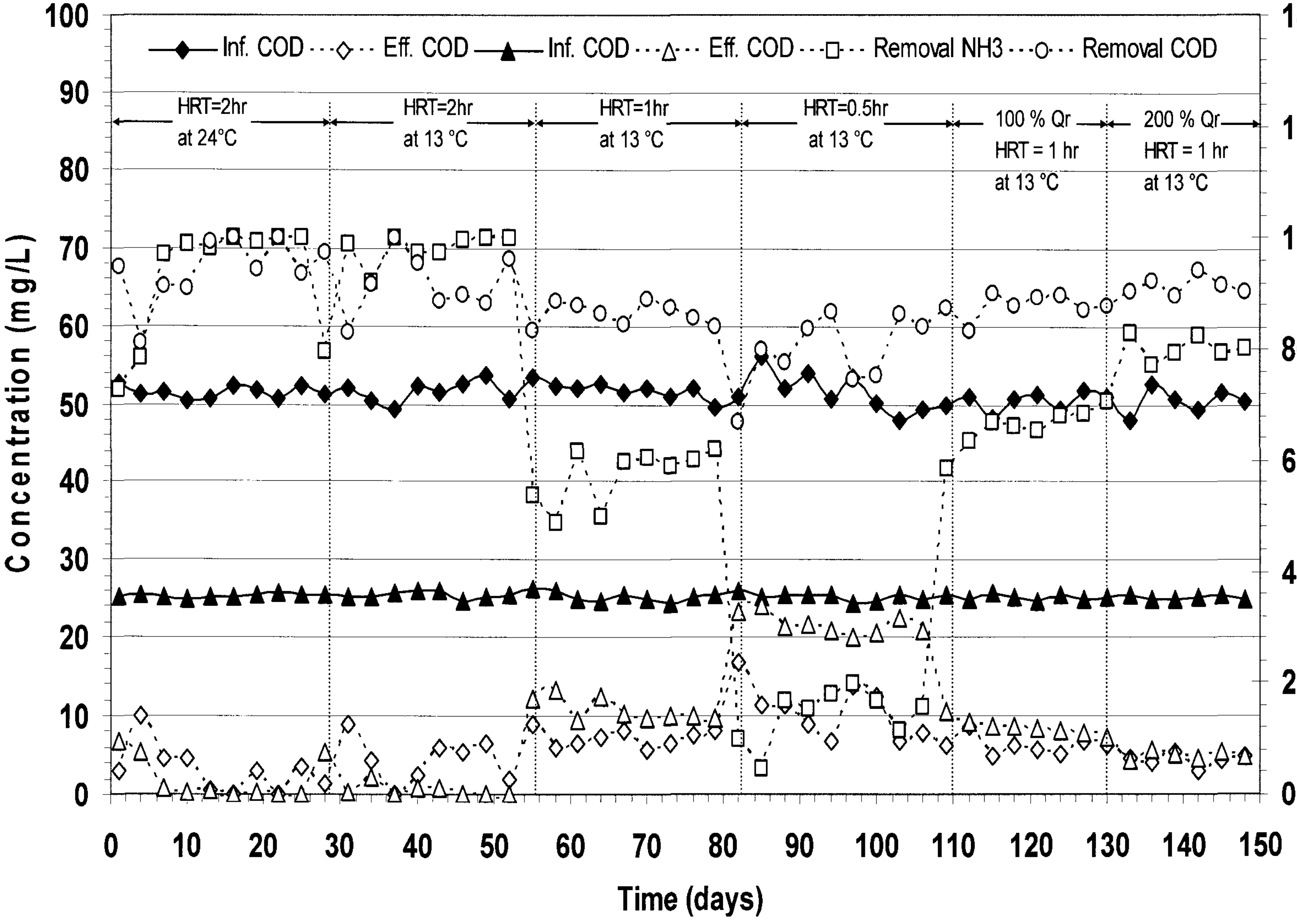


Figure 4. BAF performance for plastic ring media at different HRTs and recirculation (Qr).

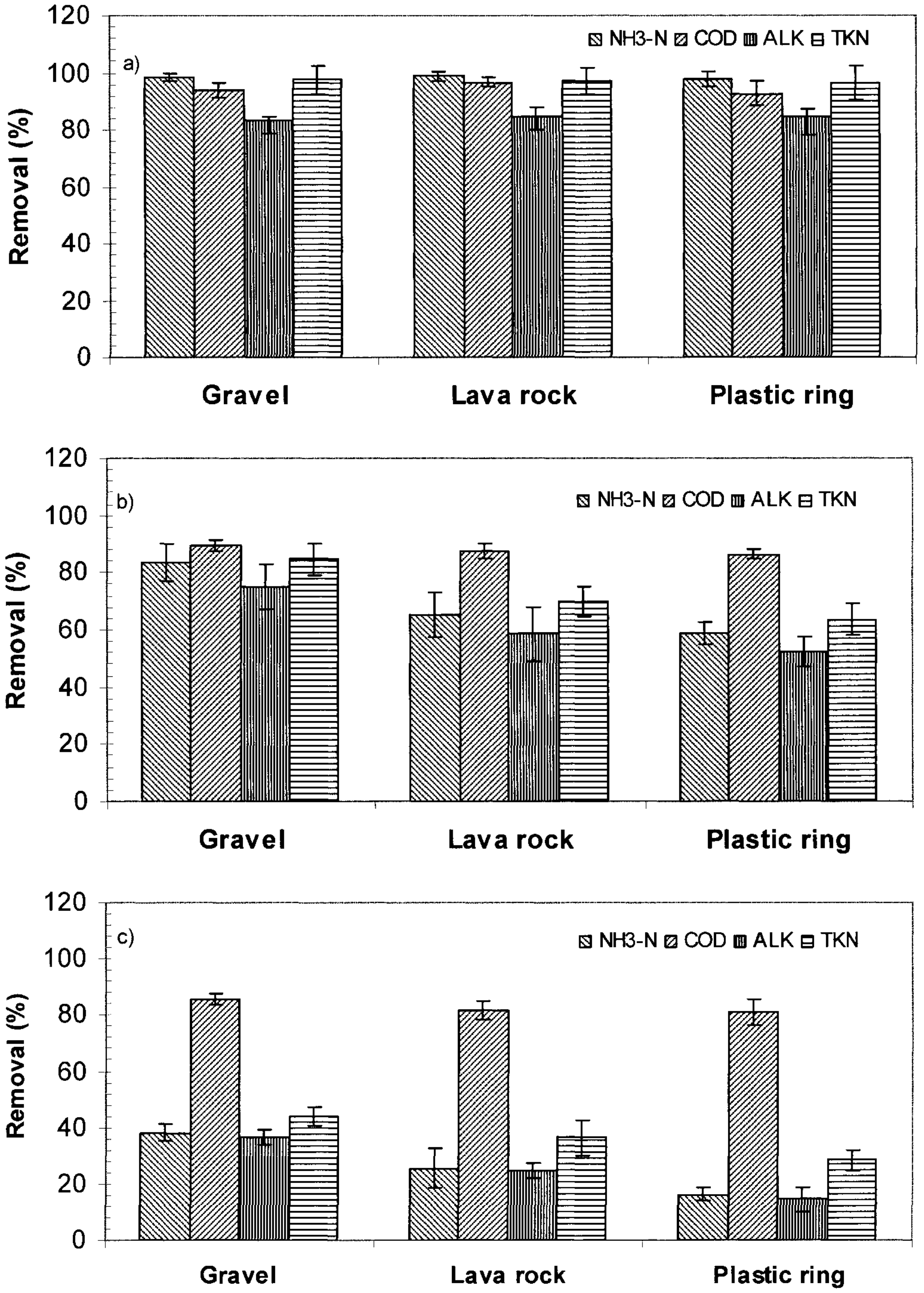


Figure 5. Effect of media type and HRT on ammonia, COD, TKN and alkalinity removal at temperature of 13 °C: (a) HRT = 2 hr, (b) HRT = 1 hr, and (c) HRT = 0.5 hr (without recirculation).

Table 3. Influent and effluent concentrations of BAFs at 2 hours HRT.

	Gravel		Lava rock		Plastic ring	
Parameter[a]	**Influent**	**Effluent**	**Influent**	**Effluent**	**Influent**	**Effluent**
sCOD (mg/L)	51.5 ± 1.2	3.1 ± 1.2	51.5 ± 1.2	1.4 ± 1.2	51.5 ± 1.2	4.1 ± 2.7
TKN (mg/L)	31.2 ± 1.3	0.8 ± 1.2	31.2 ± 1.3	0.9± 1.2	31.2 ± 1.33	1.1 ± 0.9
NH_3-N (mg/L)	25.4 ± 0.4	0.3 ± 0.2	25.4 ± 0.4	0.4 ± 0.5	25.4 ± 0.4	0.6 ± 0.7
Akalinity (mg/L)	200.4 ± 5.4	32.4 ± 5.1	200.4 ± 5.4	33.6 ± 12.8	200.4 ± 5.4	34.0 ± 11.1
Temperature (°C)	12.3 ± 0.5	14.2 ± 0.2	12.4 ± 0.4	14.7 ± 0.3	13.8 ± 0.3	14.6 ± 0.3
pH	8.1 ± 0.1	7.3 ± 0.2	8.1 ± 0.1	7.1 ± 0.1	8.1 ± 0.1	7.0 ± 0.1
Dissolved oxygen (mg/L)	-	4.4 ± 0.3	-	4.6 ± 0.4	-	4.3 ± 0.6
NO_3^--N (mg/L)	0.5 ± 0.3	21.2 ± 1.3	0.5 ± 0.3	20.5 ± 1.6	0.5 ± 0.3	23 ± 1.4
NO_2^--N (mg/L)	0.2 ± 0.1	0.4 ± 0.2	0.2 ± 0.1	0.5 ± 0.4	0.2 ± 0.1	0.5 ± 0.1
Water flow (L/day)	95	--	95	--	95	--
Air flow (mL/min)	300	--	300	--	300	--
Backwash flow (L/min)	3.33	--	3.33	--	3.33	--

[a] All parameter values are average of 9 data points.

Table 4. Influent and effluent concentrations of BAFs at 1 hour HRT.

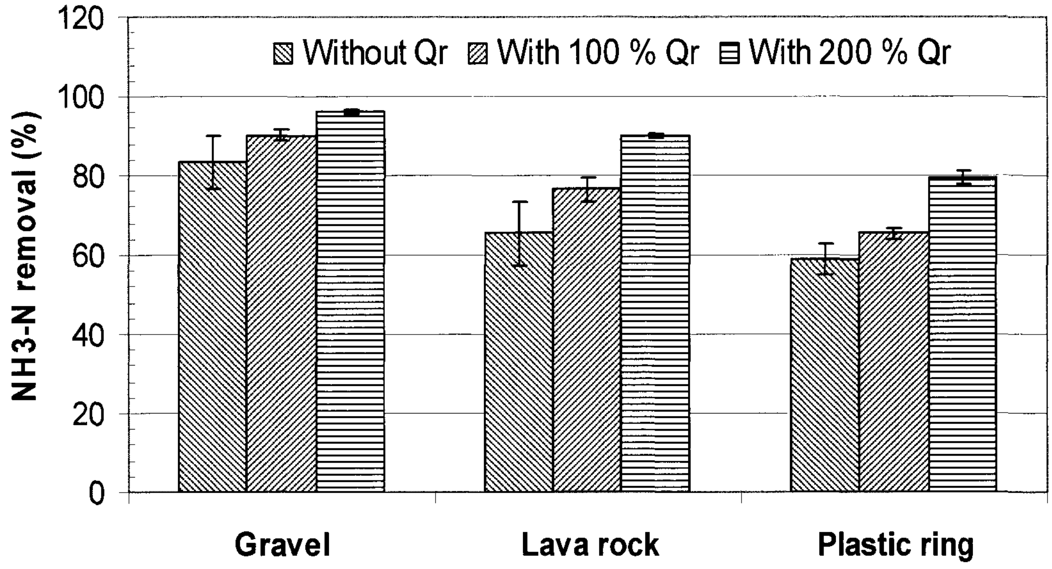
	Gravel		Lava rock		Plastic ring	
Parameter	**Influent**	**Effluent**	**Influent**	**Effluent**	**Influent**	**Effluent**
sCOD (mg/L)	51.8 ± 1.0	5.4 ± 1.7	51.8 ± 1.0	5.9 ± 2.2	51.8 ± 1.0	7.2 ± 1.0
TKN (mg/L)	30.9 ± 1.3	4.8 ± 0.7	30.9 ± 1.3	9.4 ± 0.8	30.9 ± 1.3	11.3 ± 0.8
NH_3-N (mg/L)	25.0 ± 0.6	4.3 ± 1.4	25.0 ± 0.6	8.9 ± 1.9	25.0 ± 0.6	10.8 ± 1.3
Akalinity (mg/L)	203.3 ± 5.8	50.7 ± 14.0	203.3 ± 5.8	77.6 ± 10.1	203.3 ± 5.8	93.8 ± 10.0
Temperature (°C)	13.6 ± 0.6	15.3 ± 0.6	13.6 ± 0.7	15.3 ± 0.7	13.9 ± 0.5	15.3 ± 0.5
pH	7.9 ± 0.1	7.1 ± 0.2	7.9 ± 0.1	7.0 ± 0.2	7.9 ± 0.1	7.2 ± 0.2
Dissolved oxygen (mg/L)	-	4.2 ± 1.1	-	4.3 ± 0.6	-	4.4 ± 0.7
NO_3^--N (mg/L)	0.6 ± 0.3	18.5 ± 2.0	0.6 ± 0.3	14.4 ± 2.5	0.6 ± 0.3	13.8 ± 2.0
NO_2^- -N (mg/L)	0.2 ± 0.1	0.5 ± 0.3	0.2 ± 0.1	0.7 ± 0.2	0.2 ± 0.1	0.6 ± 0.2
Water flow (L/day)	190	--	190	--	190	--
Air flow (mL/min)	300	--	300	--	300	--
Backwash flow (L/min)	3.33	--	3.33	--	3.33	--

[a] All parameter values are average of 9 data points.

Table 5. Influent and effluent concentrations of BAFs at 0.5 hour HRT

	Gravel		**Lava rock**		**Plastic ring**	
Parameter	**Influent**	**Effluent**	**Influent**	**Effluent**	**Influent**	**Effluent**
sCOD (mg/L)	51.6 ± 2.4	7.2 ± 1.2	51.6 ± 2.4	8.9 ± 1.8	51.6 ± 2.4	10.7 ± 3.2
TKN (mg/L)	30.6 ± 1.5	17.2± 0.6	30.6 ± 1.5	19.5± 2.2	17.2 ± 1.5	21.9± 1.2
NH_3-N (mg/L)	25.2 ± 0.5	16.7 ± 2.3	25.2 ± 0.5	19.0 ± 1.9	25.2 ± 0.5	21.4 ± 1.8
Akalinity (mg/L)	201.8 ± 6.3	136.4 ± 18.4	201.8 ± 6.3	153.6 ± 6.7	201.8 ± 6.3	176.0 ± 12.3
Temperature (°C)	13.3 ± 0.5	14.3 ± 0.5	13.2 ± 0.5	14.2 ± 0.5	13.5 ± 0.3	14.5 ± 0.5
pH	8.0 ± 0.1	7.0 ± 0.2	8.0 ± 0.1	7.0 ± 0.2	8.0 ± 0.1	7.2 ± 0.2
Dissolved oxygen (mg/L)	-	4.1 ± 0.3	-	4.4 ± 5.4	-	4.2 ± 0.6
NO_3^--N (mg/L)	0.4 ± 0.2	7.2 ± 1.3	0.4 ± 0.2	5.8 ± 0.9	0.4 ± 0.2	4.3 ± 1.2
NO_2^- -N (mg/L)	0.2 ± 0.1	0.5 ± 0.2	0.2 ± 0.1	0.8 ± 0.3	0.2 ± 0.1	0.7 ± 0.2
Water flow (L/day)	380	--	380	--	380	--
Air flow (mL/min)	300	--	300	--	300	--
Backwash flow (L/min)	3.33	--	3.33	--	3.33	--

[a] All parameter values are average of 9 data points.



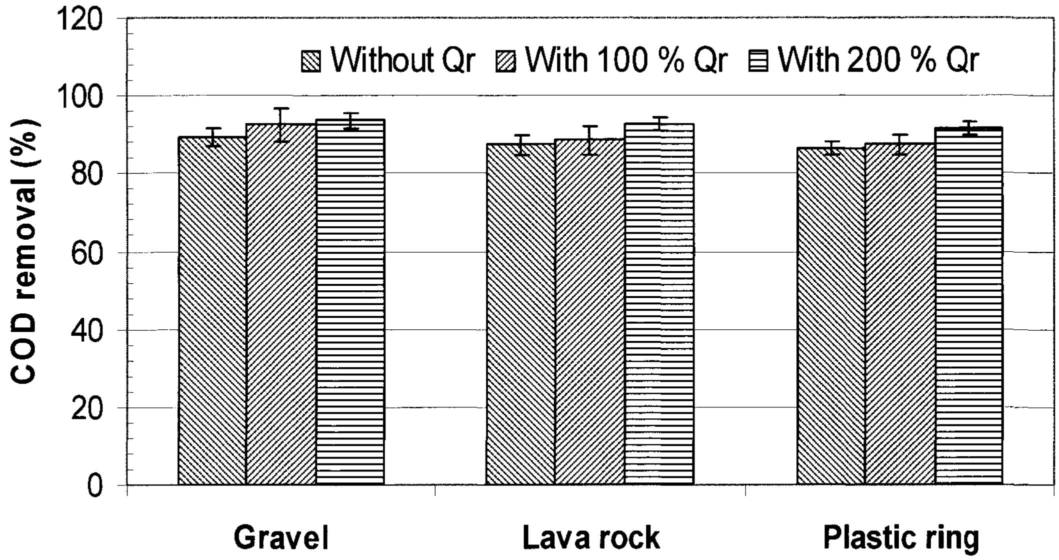


Figure 6. Ammonia and COD removals with 100 and 200 % recirculation (Qr) (HRT = 1 hour without recirculation).

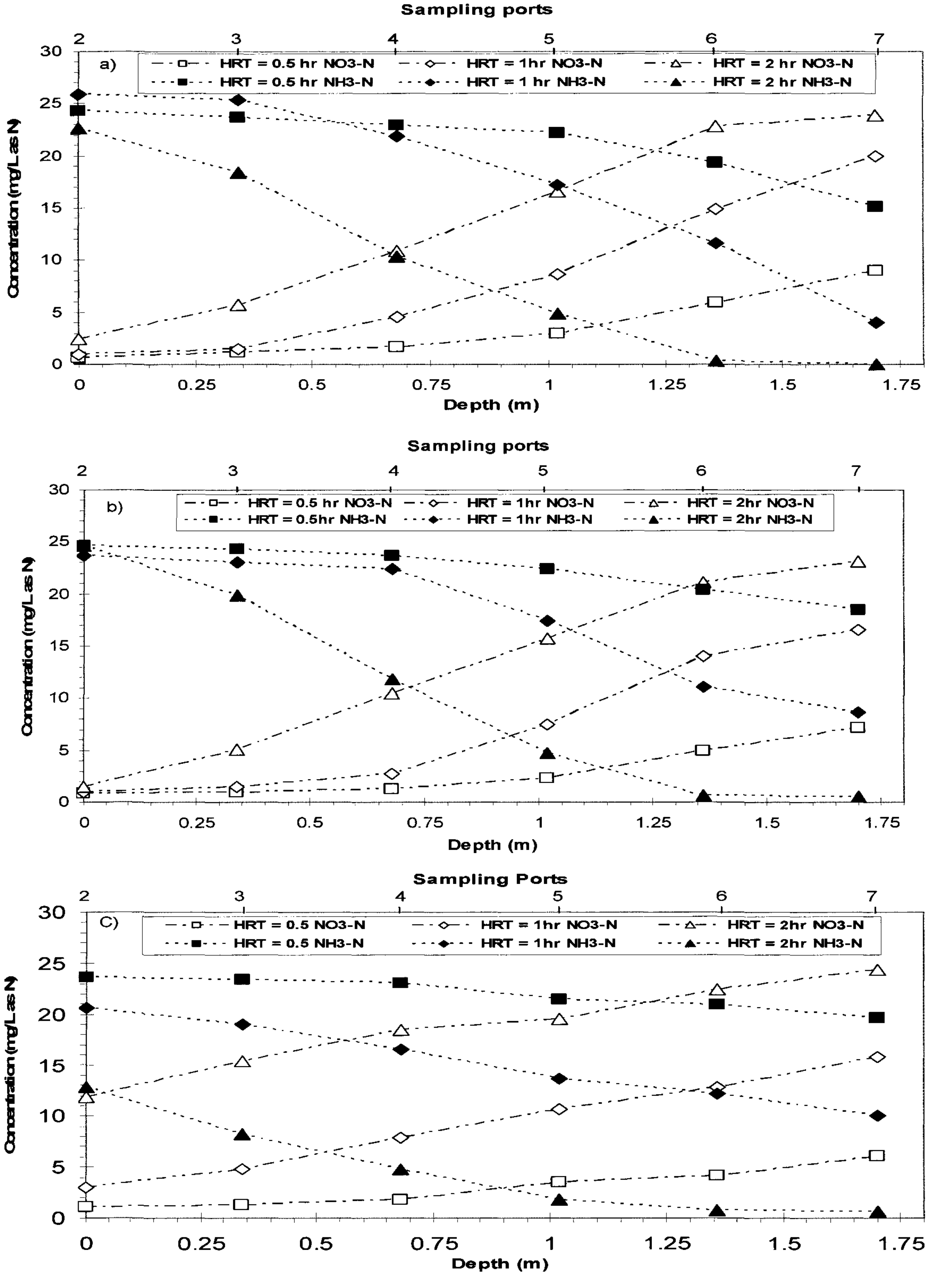


Figure 7. Ammonia and nitrate concentrations change within BAF column: (a) gravel bio-reactor, (b) lava rock bio-recator, and (c) plastic ring bio-reactor (without recirculation).

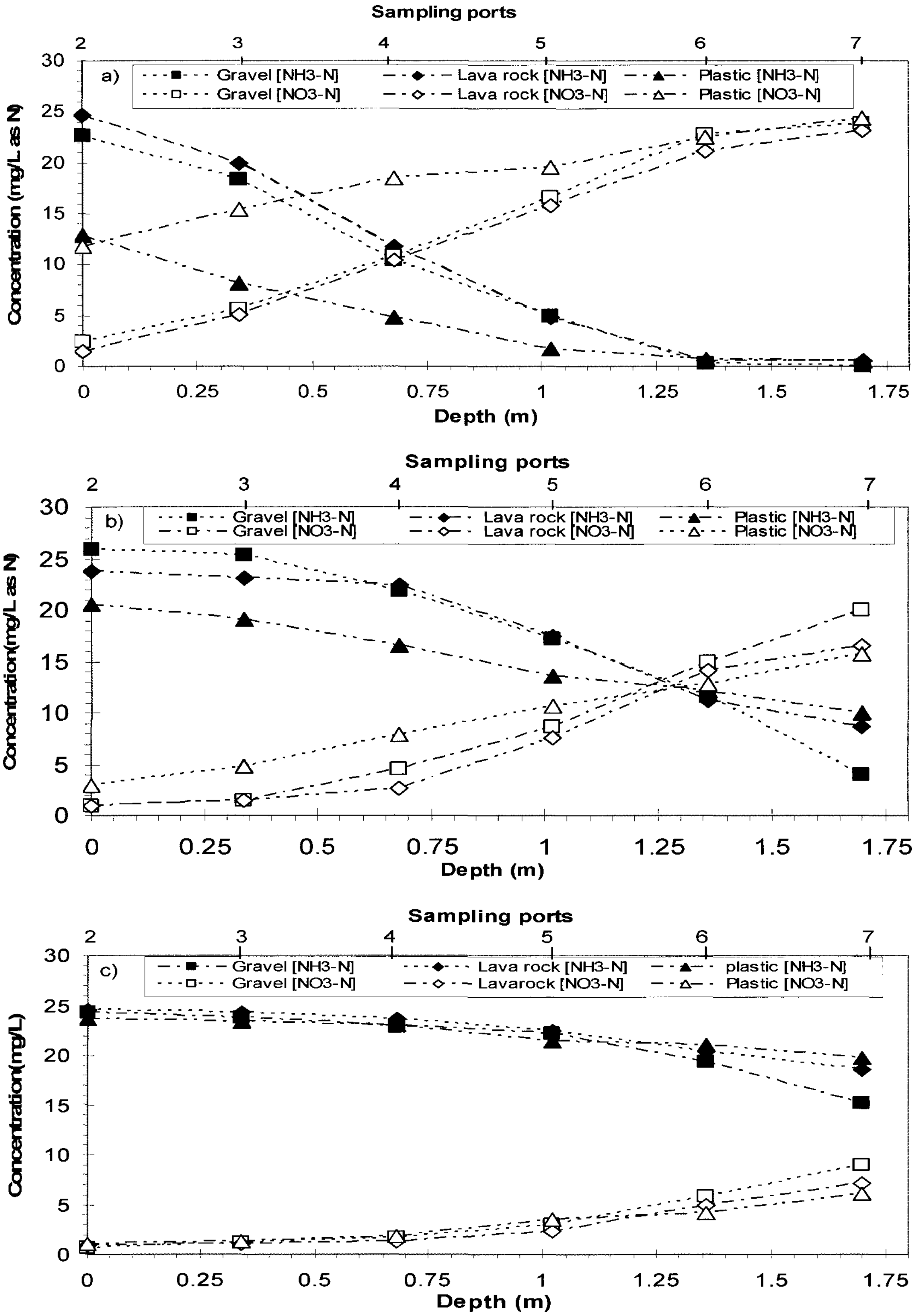


Figure 8. Ammonia and nitrate concentrations change at different HRTs: (a) 2 hours, (b) 1 hour, and (c) 0.5 hour (without recirculation).

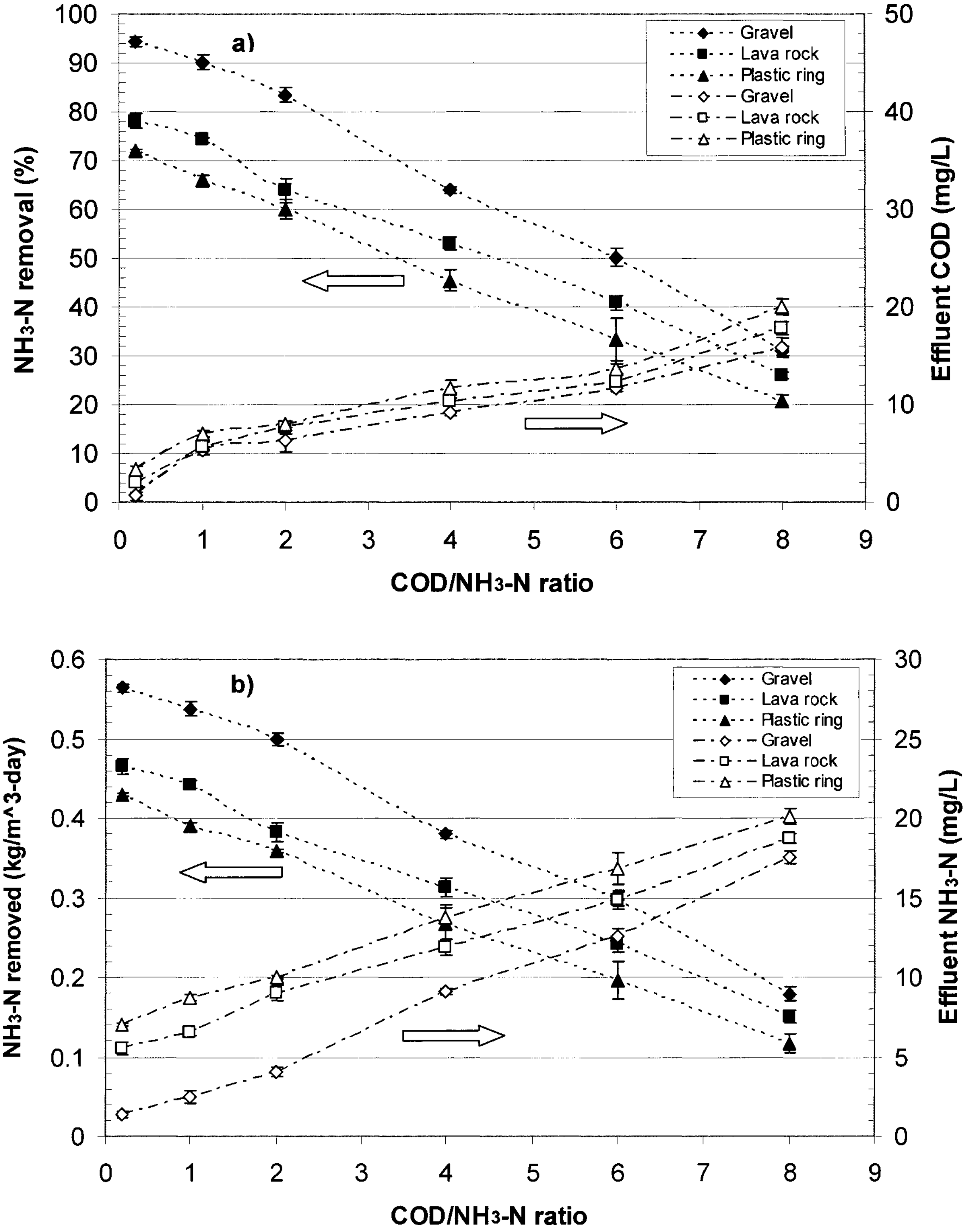


Figure 9. Effect of COD/NH$_3$-N ratio at 13 ± 1 ° C: (a) ammonia removal (%) and effluent COD concentration and (b) ammonia mass removed and effluent ammonia concentration (HRT = 1 hr).

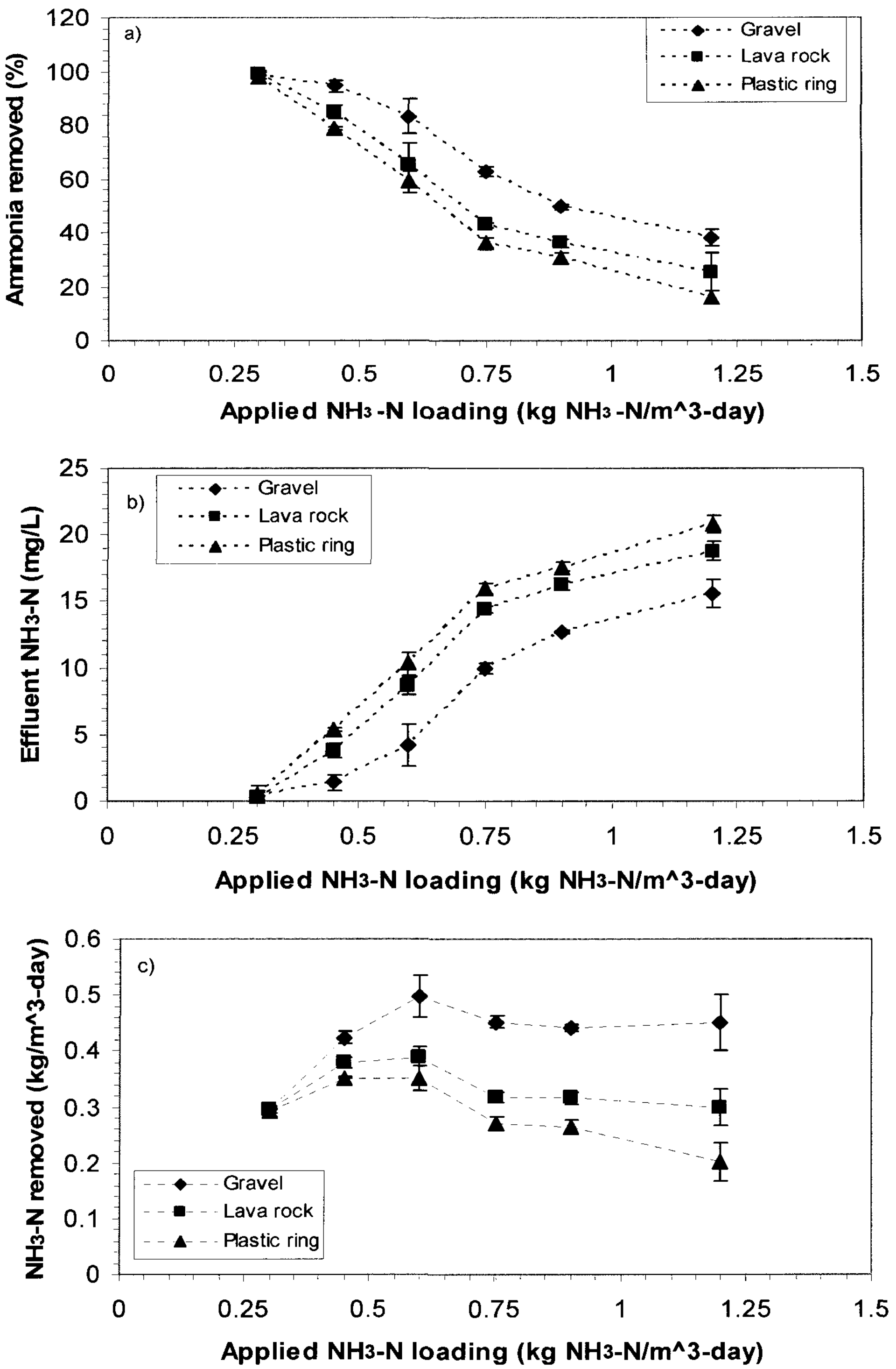


Figure 10. Effect of ammonia and COD loadings on ammonia removal at temperature of 13 ° C: a) ammonia percent removal with different ammonia mass loads, b) effluent ammonia concentration with different ammonia mass loads, and c) ammonia mass removed with different ammonia mass loads.

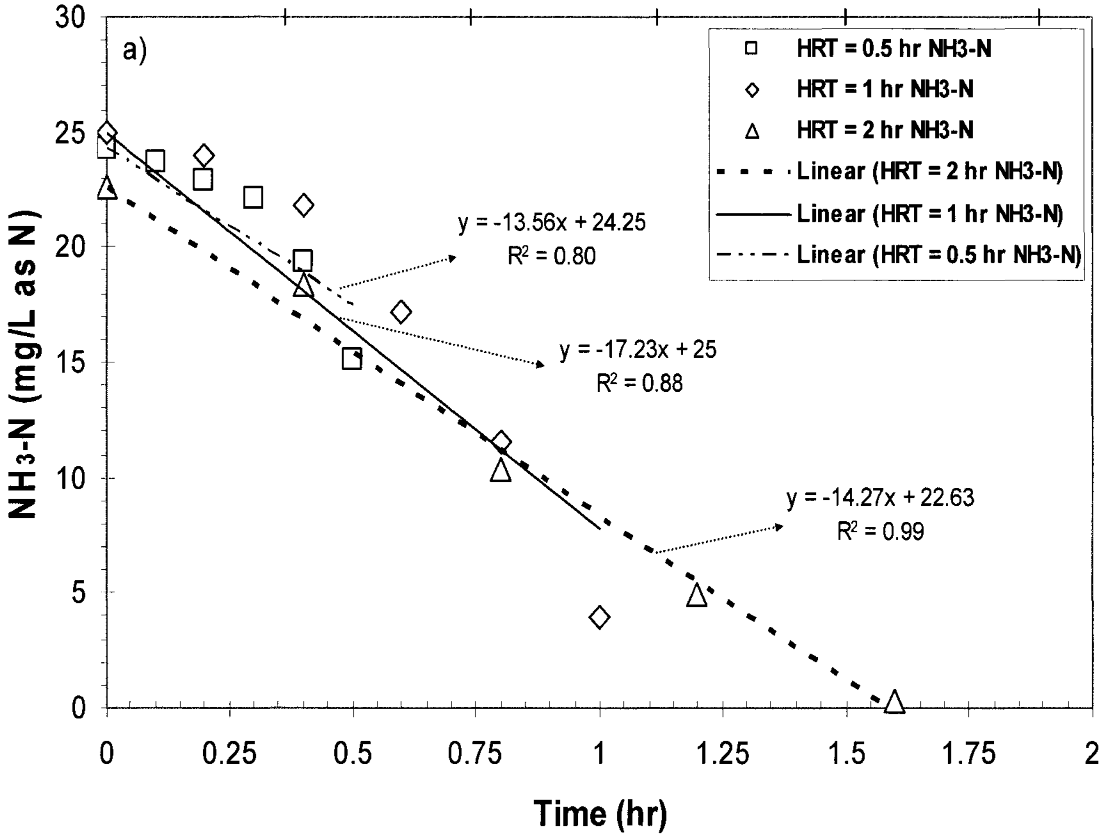


Figure 11. Zero-order plot of experimental data at different HRTs and 13 ± 1°C in gravel media.

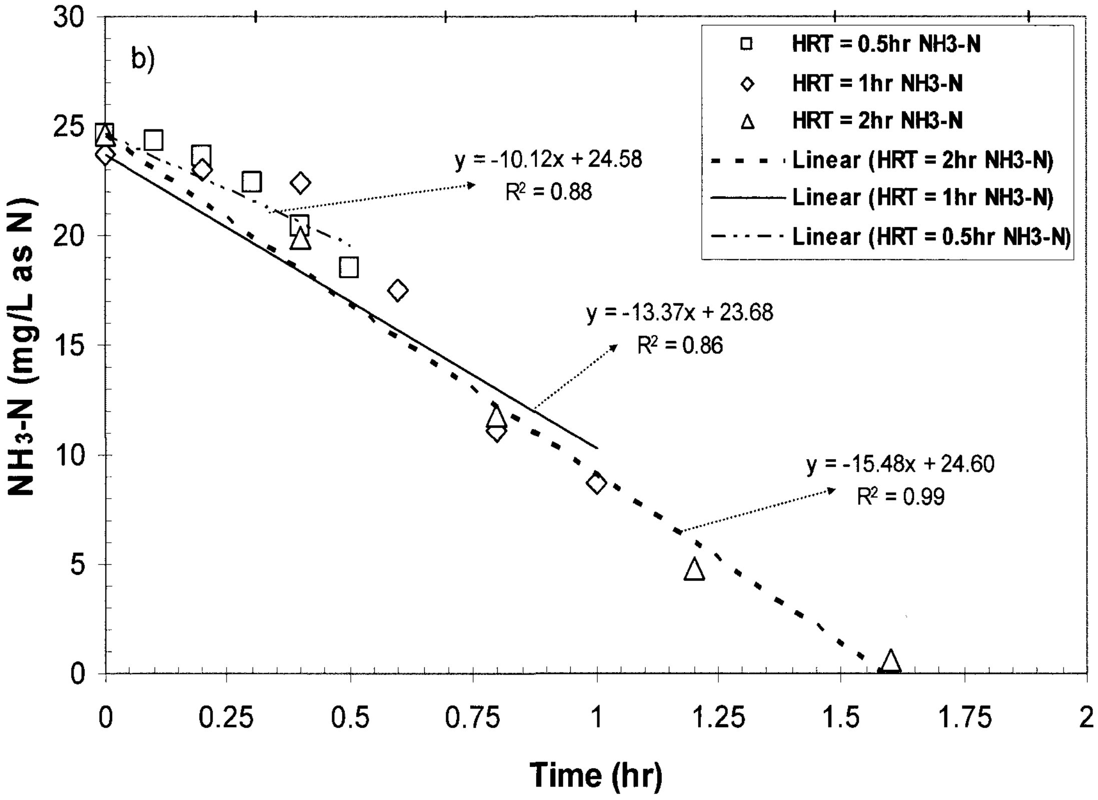


Figure 12. Zero-order plot of experimental data at different HRTs and 13 ± 1°C in lava rock media.

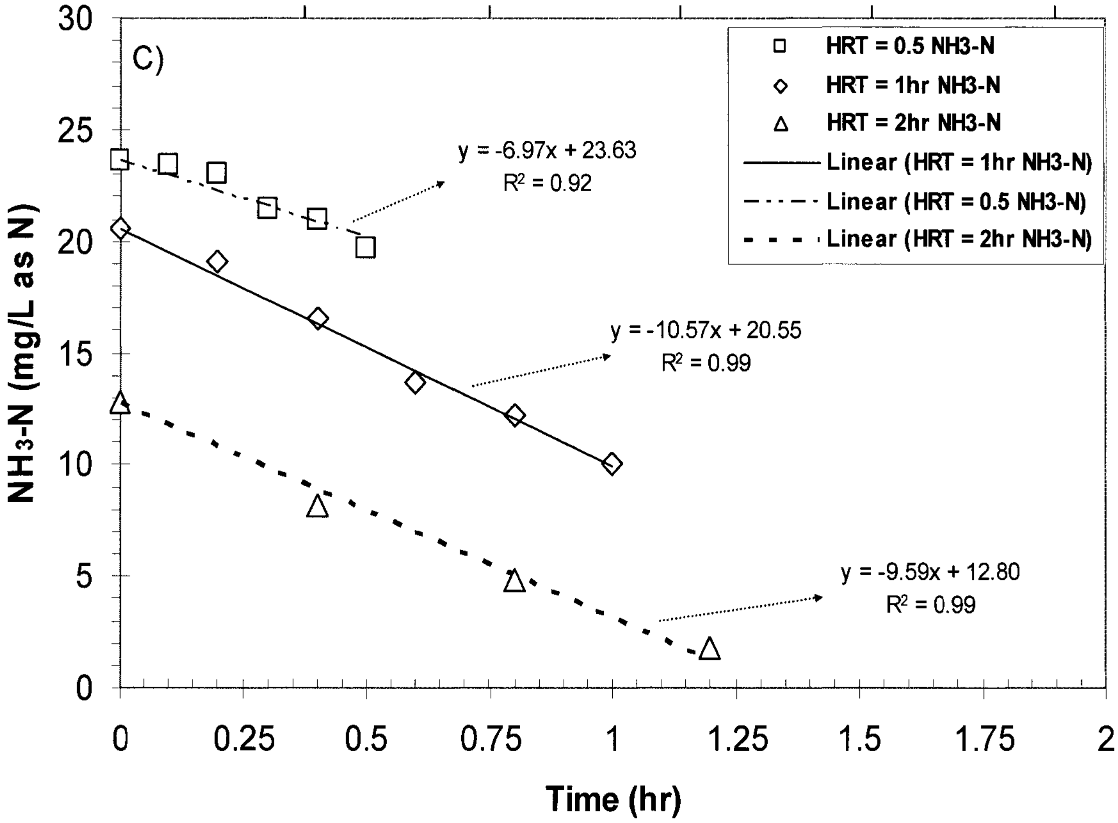


Figure 13. Zero-order plot of experimental data at different HRTs and 13 ± 1°C in plastic ring media.

Table 6. Estimation of zero-order ammonia bio-transformation rate in BAF for various HRTs and media types at 13 ± 1 °C.

Run	**Gravel**		**Lava rock**		**Plastic ring**	
condition	**k (mg NH_3-N/L-hr)[a]**	**R^2**	**k (mg NH_3-N/L-hr)[a]**	**R^2**	**k (mg NH_3-N/L-hr)[a]**	**R^2**
HRT=2hr	14.27 ± 0.83	0.99	15.43 ± 0.94	0.99	9.59 ± 0.92	0.99
HRT=1hr	17.23 ± 3.06	0.88	13.37 ± 2.64	0.86	10.57 ± 0.49	0.99
HRT=0.5hr	13.56 ± 3.62	0.80	10.12 ± 1.76	0.88	6.97 ± 0.99	0.92

[a] 95% confidence interval

CHAPTER 4. TEMPERATURE EFFECTS ON NITRIFICATION IN POLISHING BIOLOGICAL AERATED FILTERS (BAFS)

A paper submitted to WEFTEC 2006

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ABSTRACT

The effect of temperature and COD/NH_3-N ratios were investigated using a 75 mm diameter biological aerated filter with a gravel media depth of 1.7 m. Influent sCOD and ammonia concentrations in the feedwater were approximately 50 mg/L and 25 mg/L. At a hydraulic retention time (HRT) of 2 hours and a wastewater temperature of 6.5 °C, most of the ammonia was nitrified. However, at a HRT of 1 hour, the ammonia removal (%) at a water temperature of 6.5 °C was halved at 54.3 (%) while sCOD removal (91.9%) was approximately close to that for a water temperature of 24 °C. By recirculating the effluent back into the BAF, ammonia removals (%) improved from 54.3 ± 2.7% to 76 ± 3.8% for 100% recirculation and 1 hour HRT at 6.5 °C. With 200% recirculation, ammonia removal improved further to 92 ± 1.5%. C/N ratio has a negative impact on nitrification with less than 5% ammonia removal for a C/N ratio of 8 at 6.5 °C. Ammonia loading reached a maximum of 0.36 kg NH_3-N/m^3-day at 6.5 °C while the maximum ammonia loading removed was 0.63 kg NH_3-N/m^3-day at 24 °C. The ammonium concentrations throughout the BAF column can be modeled using a simple zero-order kinetic equation and a correlation relating the zero-order rate with the wastewater temperature and HRTs was developed. The experimental results indicated that gravel BAF at HRT of 1 hour with 100% or 200% recirculation can be used as an add-on technology for nitrification under cold weather conditions. Without recirculation, the experimental results demonstrated that the BAF could be operated at an HRT of 2 hour to ensure nitrification at 6.5 °C.

Keywards: nitrification, biological aerated filter, recirculation, temperature effect, NPDES

1. INTRODUCTION

Aerated lagoons are used widely by small communities for the treatment of domestic wastewater. Their relatively low operational costs and low maintenance requirements have resulted in widespread use in the Midwest. Over the last decade, discharge limits of certain pollutants for these facilities have changed or have been added even though some of these systems are not designed to treat these pollutants. An example is the inclusion of effluent ammonia limits to the National Pollution Discharge Elimination System (NPDES) permits of these facilities. Since many of these aerated lagoons are shallow ponds and will ice over during the winter months especially in the Midwest, these facilities are unable to meet the effluent ammonia concentrations during these winter months. Many of these facilities are considering modifying the hydraulics of the lagoons, include an add-on treatment process or conduct stream mixing studies to better estimate their NPDES discharge limits.

BAFs are well suited as an add-on upgrade process since BAF systems have been shown to operate successfully at higher hydraulic and organic loading rates than activated sludge system (Rogalla et al., 1992; Tschui et al., 1994; Peladen et al., 1996; Pujol et al., 1998; Bigot et al., 1999; Hodkinson., 1999). Peladan et al. (1996) have reported removal loading rates of up to 2.7 kg NH_3-N/m^3-day and 18 kg COD/m^3-day in BAFs. At these hydraulic loading rates, the biomass concentrations in BAFs can be 8 to 9 times greater than in activated sludge plants (Boller et al., 1994). The high biomass in the media may be effective for ammonia removal for low temperature wastewater. Furthermore, studies have shown that the BAF systems (Biocarcon, Biostyr^{TM}, Biopur) are relatively compact, have a relatively low capital cost investment, easy operation and are more efficient than activated sludge systems (Paffonia et al., 1990; Rogalla and Sibony, 1992; Tschui et al., 1994).

Wastewater in aerated lagoon systems at different seasons can experience wide temperature changes from as low as 5 °C in winter, to as high as 25 °C in the summer giving a strong need to determine influence of temperature on the treatment process. Temperature influences a variety of biofilm processes in BAFs from attachment and detachment of the biofilm, lysis of cells, and microbial activity. These effects complicate the determination of the temperature effects on biochemical processes. Although the effect of temperature on nitrification using BAFs has been studied (Fdz-polanco et al., 1994; Choi et al., 1998), a structured study on the effect of temperature on nitrification for different HRTs is needed.

In this study, a down-flow pilot-scale sand BAF is operated as an add-on system to aerated lagoons treating synthetic wastewater simulating the effluent of aerated lagoons. As indicated earlier, due to seasonal variations, aerated lagoons can experience wide temperature swings that can affect final effluent quality. The objective of this study is to investigate the effects of different temperatures on the performance of BAF especially nitrification. Other variables investigated include COD and nitrogen loading rates and recirculation on nitrification at various temperatures.

MATERIALS AND METHODS

A down-flow pilot-scale BAF made of PVC pipe as shown in Figure 1 was set up. The BAF had a diameter of 75 mm and a media depth of 1.7 m. The media used was gravel with an average diameter of 5 mm (between US Standard Sieve No. 2 (5.36 mm) and No. 4 (4.36 mm)). The porosity for gravel was estimated to be 0.36 ± 0.01 while the specific surface area of gravel was 0.93×10^6 m^2/m^3.

To assess the effectiveness of BAF under varying conditions, a synthetic wastewater was used as feed wastewater. The synthetic wastewater contained 20 mg/L of $CaCl_2$, 3 mg/L of $MgSO_4$, 2 mg/L of $FeCl_2$, 50 mg/L of NaH_2PO_4, 200 mg/L of $NaHCO_3$ as calcium carbonate, 4 mg/L of KCl, 26 mg/L of CH_3COONa, 15 mg/L (COD = 10 mg/L) of nutrient broth, 22 mL of Isomil (COD = 20 mg/L) in 400 L of synthetic wastewater and 95 mg/L of NH_4Cl. sCOD and ammonia concentrations of the feed water were approximately 50 ± 2.1 mg/L and 25 ± 1.3 mg/L, respectively.

The gravel BAF was seeded using activated sludge from the aeration tank of a municipal wastewater treatment plant in Iowa. The gravel BAF was operated initially for a period of time at room temperature of 24 °C until steady state removals for sCOD and ammonia were above 90%. The operating temperatures were changed to 13 and 6.5 °C to assess the BAF performance.

The operational characteristics of the BAFs are summarized in Table 1. The BAF was operated at three hydraulic retention times (HRTs) and three different temperatures. Influent and effluent samples were collected throughout the study and analyzed for sCOD, Total Kjeldahl Nitrogen (TKN), alkalinity, ammonia, nitrate, nitrite, pH, temperature and DO according to Standard Methods (APHA, 2002). Samples used for the analyses were one-hour composite samples.

RESULTS AND DISCUSSION

BAF operational performance and head loss

Representative performances of the gravel BAF at temperature of 24 and 13 °C are shown in Figure 2. Average sCOD and ammonia concentrations of the feed water were approximately 50 mg/L and 25 mg/L, respectively. The BAFs were initially operated at a temperature of 24 ± 1 °C and then at the 13 ± 1 and 6.5 ± 1 °C. A HRT of 2 hours was used initially and then changed to 1 hour HRT followed by 0.5 hour HRT at the same temperature. With a HRT of 2 hours, 1 hour and 0.5 hour, the linear velocities in the BAF were 0.85, 1.7 and 3.4 m/hr, respectively.

The sCOD removals were greater than 95% and the effluent NH_3-N concentration were close to zero (removal efficiency of 99%) for a HRT of 2 hours at a temperature of 13 °C. Nitrification and sCOD removal efficiency were very stable at a HRT of 2 hours. Experimental results showed that when HRT was 0.5 hours, effluent ammonia concentrations were approximately 15 mg N/L, which corresponded to a removal efficiency of 38% but sCOD removals were greater than 85%.

Headloss in the gravel BAF for three different HRTs at temperature of 13 °C were measured as shown in Figure 3. Regular backwash was needed to remove excess biomass within the column to prevent clogging. The headloss for a clean column was approximately 16 cm. At a HRT of 0.5 hour, the headloss reached 70 cm in 50 hours. For a HRT of 1 hour, it took 88 hours to reach 70 cm. However, at a HRT of 2 hours, the headloss increased slowly and was about 34 cm after 140 hours as shown in Figure 3. Based on the headloss study, backwashing was performed every two days for 0.5 hour HRT, every four days for 1 hour HRT, and once per week for a HRT of 2 hours. The backwash rate used was 10 m/hr which is typically in the range of backwashing of the sand filters (7 to 15 m/hr). As a comparison, Mendoza-Espinosa and Stephenson (1999) indicated that a conventional downflow BAF needs to be backwashed every 12 hours.

If the backwashing is done correctly, Bacquet et al. (1991) reported that reactor performance was not reduced since only 30% of the biomass was removed by backwashing. Similarly, Visvanathan and Nhien (1995) observed that effluent quality was not impacted if backwashing process is undertaken correctly.

Ammonia and nitrate concentration profiles with and without aeration at 2 hours HRT

Experiments with and without aeration at temperature of 13 °C and HRT of 2 hours were performed to investigate the impact of aeration on ammonia removal. Airflow rate used was 300 mL/min. The ammonia and nitrate concentration profile are presented in Figure 4. With aeration, the appearance of nitrate was in agreement with the disappearance of ammonia. Approximately 24 mg N/L of ammonia was converted to nitrate while the rest of nitrogen probably were assimilated into biomass. The effluent ammonia concentration was almost zero (i.e., 0.3 ± 0.2 mg/L). Without aeration, ammonia conversion was limited with an effluent ammonia concentration of 17.7 ± 0.6 mg/L. Ammonia removal in BAF was enhanced with aeration due to the enhanced oxygen mass transfer from the bulk water to biofilms on the media.

Effect of temperature at different hydraulic retention times

Ammonia removal efficiencies at different temperatures of 6.5, 13 and 24 °C and HRTs of 2 hours, 1 hour and 0.5 hour are presented in Figure 5. For a HRT of 2 hours and for temperatures as low as 6.5 °C, more than 95% of ammonia were removed. For a HRT of 1 hour, the percent ammonia removed (%) at temperature of 24, 13 and 6.5 °C for sand media were 97.9 ± 1.2, 83.3 ± 6.5 and 54.3 ± 1.7%, while at a HRT of 0.5 hour, ammonia removal was 53 ± 3.6, 38 ± 3.2 and 29.9 ± 1.8%, respectively. The experimental results showed that a BAF operated at a HRT of 2 hour without recirculation can be used as an add-on treatment technology to nitrify the ammonia at a temperature as low as 6.5 °C. This is consistent with the study conducted by McHarness et al. (1975) where the same degree of nitrification was maintained by increasing the HRT when the temperature of the wastewater was reduced.

Ammonia, nitrite and nitrate concentration within the column at different HRTs and temperatures

Ammonia concentrations within the column at different HRTs and wastewater temperatures are presented in Figure 6. For all three wastewater temperatures at HRT of 2 hours, complete nitrification of ammonia was observed within the column. At HRT of 2 hours, ammonia was completely removed within the depth of 0.7, 1.4 and 1.7 m at wastewater temperatures of 24 °, 13 ° and 6.5 °C, respectively (see Figure 6a). At a HRT of 1 hour, complete ammonia removal was observed for 24 °C with an ammonia effluent concentration of 0.01 mg N/L, while the effluent ammonia concentrations were 4 and 12 mg N/L at wastewater temperatures of 13 °and 6.5 °C, respectively (see Figure 6b). However, for HRT of 0.5 hour, about 40% and 50% nitrification occurred for a wastewater water temperature of 13 °C and 24 °C but with less than 30% removal for 6.5 °C (see Figure 6c).

Also plotted in Figure 6, are the nitrate and nitrite concentrations. For all cases, the appearance of nitrate was in good agreement with the disappearance of ammonia while the nitrite concentrations at 24 °C reached about 5 mg N/L within the column but the effluent nitrite were found to be less than 1 mg N/L. The nitrite concentrations within the column at 6.5 °C were less than 2 mg N/L with an effluent nitrite concentrations of less than 1 mg N/L (see Figure 6c).

Overall Ammonia Biotransformation Rates at different temperatures in BAFs

The change in ammonia concentration with depth can be modeled using a zero-order reaction as proposed by Boller and Gujer (1986). The overall ammonia biotransformation rate, k within the BAF systems may be expressed as:

$$C - C_o = -k\,t \quad = \quad -k\,h/v \qquad (1)$$

where C_o is the influent ammonia concentration (mg/L), C is the actual ammonia concentration (mg/L), k is the zero order ammonia biotransformation rate (mg NH_3-N/L-hr); and t is time of reaction (hrs), h is the depth of the sand media (m) and v is hydraulic loading rate (m/hr). A plot of C versus t provide a straight line curve with a slope equal to k and an intercept of influent ammonia concentration (see Figure 7). Table 2 summarized the overall biotransformation rate, k for three different HRTs and at different temperature of 24, 13 and 6.5°C. The zero-order ammonia biotransformation rate increased as temperature increased (see Table 2) with rates at HRT of 2 hours in the BAF of 32.06 ± 4.59, 16.19 ±

1.48 and 12.97 ± 0.78 mg NH_3-N/L-hr at temperature of 24, 13 and 6.5 °C, respectively. At temperature of 24 °C, the biotransformation rates at 2-hour HRT was statistically similar for 1-hour HRT but 2-hour and 1-hour HRT were statistically different for 0.5-hour HRT. For the temperature of 13 °C, the zero-order biotransformation rates at different HRTs were statistically similar. At temperature of 6.5 °C, the biotransformation rates were statistically similar for 2-hour and 1-hour HRT but 2-hour was different for 0.5 hour HRT. Statistically, the zero-order biotransformation rates were different at various temperatures.

The biotransformation rates were regressed against the HRTs and temperature to provide a correlation that predicts k values for different HRTs and temperatures as shown in below:

$$k_{HRT,T} = 16.97 + 0.139[1 - HRT] - [-1.63(HRT)^2 + 4.96\,(HRT) - 1.72][13 - T] \qquad (2)$$

The predicted rates using the above equation are presented in Table 3. The model predicted well the biotransformation rates at 24 °C and 13 °C but did not predict well the biotransformation rates at 6.5 °C.

Ammonia flux to biofilm at different temperature in BAF

When the column is at steady state, the substrate flux into biofilms is similar to the flux out of the bulk water. The ammonia flux to biofilm in BAF is described using the following steady-state equation (Manem and Rittmann, 1990).

$$J = Q (C_o - C)/aV \tag{3}$$

where Q is the flow rate (m^3/hour), C_o is the influent ammonia concentration (mg/L), C is the effluent ammonia concentration (mg/L), V is the volume of control (m^3), and a is the specific surface area of media (m^2/m^3). In equation 2, aV is the biofilm surface area (m^2).

A plot for the overall ammonia flux rates to biofilms at different HRTs and temperatures are shown in Figure 8 and Table 4. At a HRT of 2 hours, the flux rates in BAF were similar at temperature of 24, 13 and 6.5 °C. At a HRT of 1 hour, the flux rates at temperature of 24, 13 and 6.5 °C for sand media was 0.59, 0.54 and 0.35 g/m^2-day, respectively. At a HRT of 0.5 hour, the flux rates at temperature of 24, 13 and 6.5 °C for sand media was 0.72, 0.51 and 0.40 g/m^2-day, respectively. As shown in Figure 8 and Table 4, the ammonia flux rates to biofilms increased as flow rates and temperature increased.

Effect of recirculation on nitrification and sCOD removal

As shown in Chapter 3, recirculation of the effluent helped to improve ammonia removal. In this study, experiments at temperature of 6.5 °C and 13 °C with HRT of 1 hour were performed with 100% and 200% recirculations to investigate the effect of recirculation on ammonia and sCOD removals (Figure 9a and 9b). With 100% recirculation at 1 hour HRT and 13 ± 1 °C, ammonia removals (%) improved from 83.3 ± 6.5 to 90.1 ± 1.4% while with 200% recirculation, ammonia removal improved to 96 ± 4.3% ± from 89.2 ± 2.1%. sCOD removal (%) with 100% and 200% recirculations at 1-hour HRT and 13 °C improved to 92.6 ± 4.3 and 93.6 ± 2.2%.

With the water temperature at 6.5 °C, recirculation of the effluent resulted in higher ammonia removal. At 100 % recirculation at 1-hour HRT, ammonia removals (%) improved from 54.3 ± 2.7 to 76 ± 3.8%, and with 200% recirculation, ammonia removal improved to 92 ± 1.5% from 76 ± 3.8%. sCOD removal (%) with 100% and 200% recirculation at 1-hour HRT and 6.5 °C improved to 88 ± 2.8 and 89 ± 4.3%. The results showed that even at low temperature (6.5 °C), recirculation of effluent can effectively improve ammonia removal even at 1-hour HRT. One possible reason is that the increased water velocity by recirculation may have resulted in enhanced mass transfer of oxygen, substrate and ammonia into biofilms (Tschui et al., 1994; Peladen et al., 1996; Pujol et al., 1998).

Impact of C/N ratio on ammonia and COD removal at different temperatures

The effects of substrate C/N (sCOD/NH_3-N) ratio on the ammonia removal (%) for six different C/N ratios at different temperatures of 6.5, 13 and 24 °C are presented in Figure 10. In this study, the influent ammonia concentration was fixed at 25 mg as N/L while influent sCOD concentrations were varied to investigate the effect of C/N ratio. Ammonia removal for C/N=0.2 to C/N=8 showed that C/N ratio had an impact on the nitrification of ammonia at different temperatures. Ammonia removal efficiency at 24 °C was close to 100% for a C/N ratio of 0.2 but was about 50% for C/N ratio of 8. At lower temperature of 6.5 °C, ammonia removal efficiency were significantly lower than at 24 °C or 13 °C and was close to negligible for a C/N ratio of 8. Ammonia percent removal or mass removal appeared to decrease linearly with the C/N ratio. Based on the results in Figure 10, the gravel BAF at the temperature of 24 and 13 °C with 1-hour HRT was effective in the ammonia removal with approximately more than 91.9% and 83.5% removal for C/N ratio less than 2.

Boller et al. (1997) observed that the efficiency of the reactors decreased between 30 and 50% when the COD loading increased in tertiary nitrifying BAFs. In addition, Ohashi et al. (1995) indicated that the nitrifier activity in the filter was not restrained at low C/N ratio but higher COD concentrations limited the nitrification activity probably due to rapid heterotroph microorganisms growth covering the outlayer of biofilms on the media.

sCOD and ammonia loadings on ammonia removal at different temperatures

Results of six different ammonia and sCOD loadings rates at 6.5, 13 and 24 °C are summarized in Figure 11. The percent ammonia removal for the different temperatures decreased as the ammonia and sCOD loadings increased and temperature decreased (see Figure 11a). Similar mass of ammonia removed per day for all three temperatures were observed at an ammonia loading of 0.3 kg NH_3-N/m^3-day (see Figure 11c), while significant divergent in ammonia mass removed among the three temperatures were observed as the applied ammonia and sCOD loadings increased. For a temperature of 24° C, the mass of ammonia removed reached an asymptotic of approximately 0.63 kg NH_3-N/m^3-day for ammonia mass loading greater than 0.9 kg NH_3-N/m^3-day. However, in the case of 13 °C and 6.5 °C, the ammonia mass removed reached an asymptotic value for ammonia loading greater than 0.6 kg NH_3-N/m^3-day. Mass of ammonia removed for 6.5 °C was found to be 0.1 kg NH_3-N/m^3-day which was lower than that of 13° C. (see Figure 11c). The above results are similar to that reported by Paffoni et al. (1990) where the ammonia mean loading rates ranged from 0.5 to 0.75 kg NH_3-N/m^3-day for temperature between 12 °and 20 °C.

Based on six different hydraulic ammonia and sCOD loading studies, the maximum loading removed was 0.36, 0.5 and 0.63 kg NH_3-N/m^3-day at 6.5, 13 and 24 °C, respectively (see Figure 11c). Of the three different temperatures, the temperature of 24 °C and HRT of 0.5 hour showed the best percentage of ammonia removal efficiency and mass removed throughout the ammonia and sCOD loadings. In this study, the experimental results indicated that even at a temperature of 13 °C, the ammonia removal in BAFs are still effective until an applied ammonia loading of 0.6 kg NH_3-N/m^3-day.

CONCLUSIONS

Experimental results indicated that gravel BAF at HRT of 1 hour with 100% or 200% recirculation of the treated effluent can be used as an add-on technology in meeting NPDES nitrogen limits for cold weather conditions. Without recirculation, the experimental results showed that the sand BAF could be operated at a HRT of 2 hours to ensure nitrification at temperature of 6.5 °C. With 100% recirculation at 1-hour HRT and temperature of 6.5 ± 1 °C, ammonia removals (%) improved from 54.3 to 76%. With 200% recirculation, ammonia removal improved to 92%. However, the C/N ratios have a negative impact on ammonia removal with less than 5% ammonia removal for a C/N ratio of 8 at 6.5 °C. Both temperatures and C/N ratio have an impact on nitrification in BAFs. The maximum ammonia loading at 6.5 °C was 0.36 kg NH_3-N/m^3-day while the ammonia loading at 24 °C was 0.63 kg NH_3-N/m^3-day. The biotransformation rates were estimated using zero-order kinetic rates. A correlation equation was developed that predicted biotransformation rates as a function of temperature and HRTs.

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Table 1. Experimental operating condition for biological aerated filters.

Parameter	**Value**
Diameter of BAF	75 mm
Media Depth	1.7 m
Flow Rates	95, 190, 380 L/day
Hydraulic Retention Times	0.5, 1, and 2 hr
Recirculation ratios	100%, 200%
Backwash Flow Rate	0.75 L/min
Air Flow Rate	300 mL/min
Temperature in Reactor	6.5, 13 and 24 °C
Influent NH_3-N	25 mg N/L
Influent COD	50 mg/L
Influent Alkalinity	200 mg/L as $CaCO_3$
Influent Total Nitrogen	30 mg N/L
Influent pH	8.0 ± 0.1

Table 2. Estimation of overall ammonia biotransformation rates in biofilm for various hydraulic retention times at different temperatures.

Run Conditions	Gravel Media					
	24 °C		13 °C		6.5 °C	
	k (mg NH_3-N/L-hr)[a]	R^2	k (mg NH_3-N/L-hr)[a]	R^2	k (mg NH_3-N/L-hr)[a]	R^2
HRT = 2 hrs	32.06 ± 4.59	0.98	16.19 ± 1.48	0.96	12.97 ± 0.78	0.81
HRT = 1 hr	33.51 ± 3.05	0.96	16.97 ± 3.23	0.86	10.38 ± 2.71	0.77
HRT = 0.5 hr	20.45 ± 4.81	0.81	16.39 ± 4.02	0.82	9.07 ± 1.59	0.85

[a] 95% confidence interval

Table 3. Prediction of ammonia biotransformation rates for various hydraulic retention times at different temperatures using the generalized equation.

Run Conditions	Gravel Media		
	24 °C	13 °C	6.5 °C
	k (mg NH_3-N/L-hr)[a]	k (mg NH_3-N/L-hr)[a]	k (mg NH_3-N/L-hr)[a]
HRT = 2 hrs	35.88	16.83	6.19
HRT = 1 hr	34.68	16.97	6.51
HRT = 0.5 hr	20.75	17.04	14.62

Table 4. Ammonia flux to biofilm at different HRTs and temperatures in BAF.

	Flux, J (g/m^2-day)		
	HRT = 2 hr	HRT = 1 hr	HRT = 0.5 hr
24 °C	0.32 ± 0.01	0.59 ± 0.01	0.72 ± 0.02
13 °C	0.32 ± 0.01	0.54 ± 0.01	0.51 ± 0.03
6.5 °C	0.32 ± 0.01	0.35 ± 0.01	0.40 ± 0.02

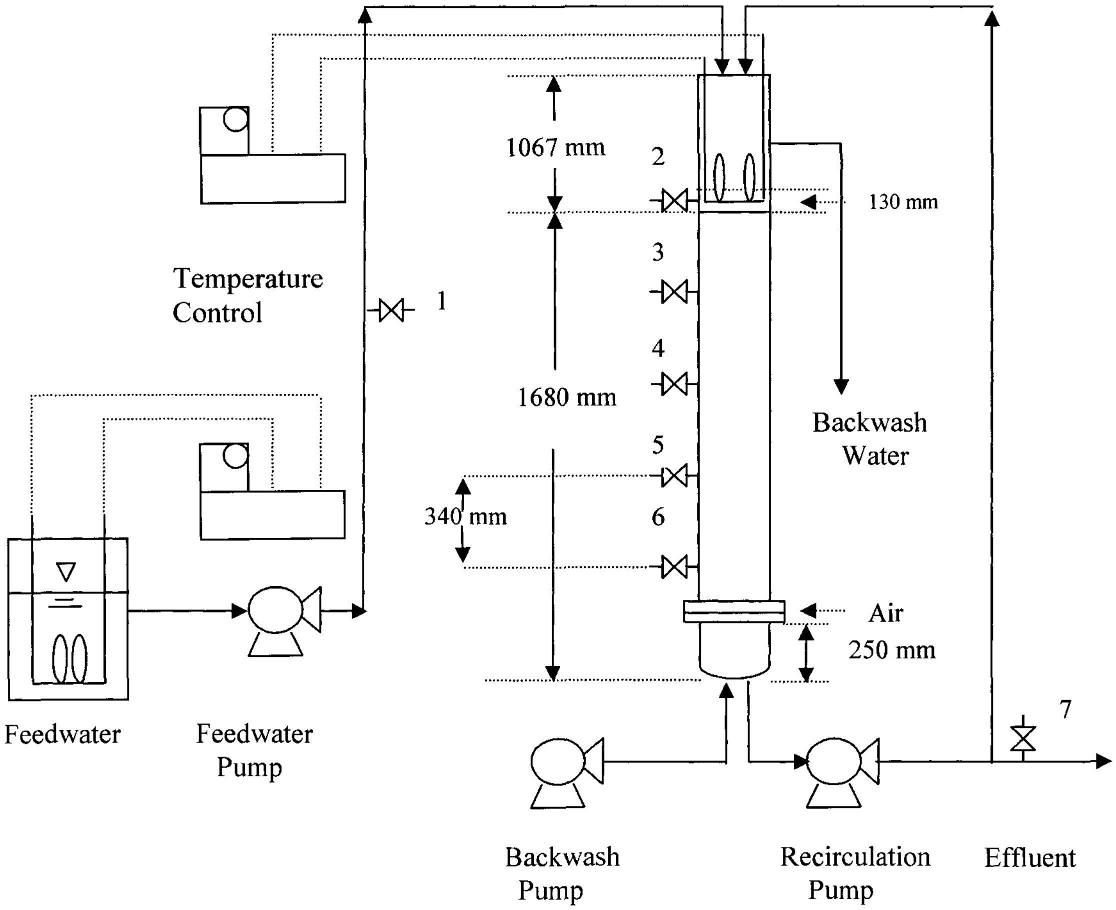


Figure 1. Schematic diagram of a BAF system.

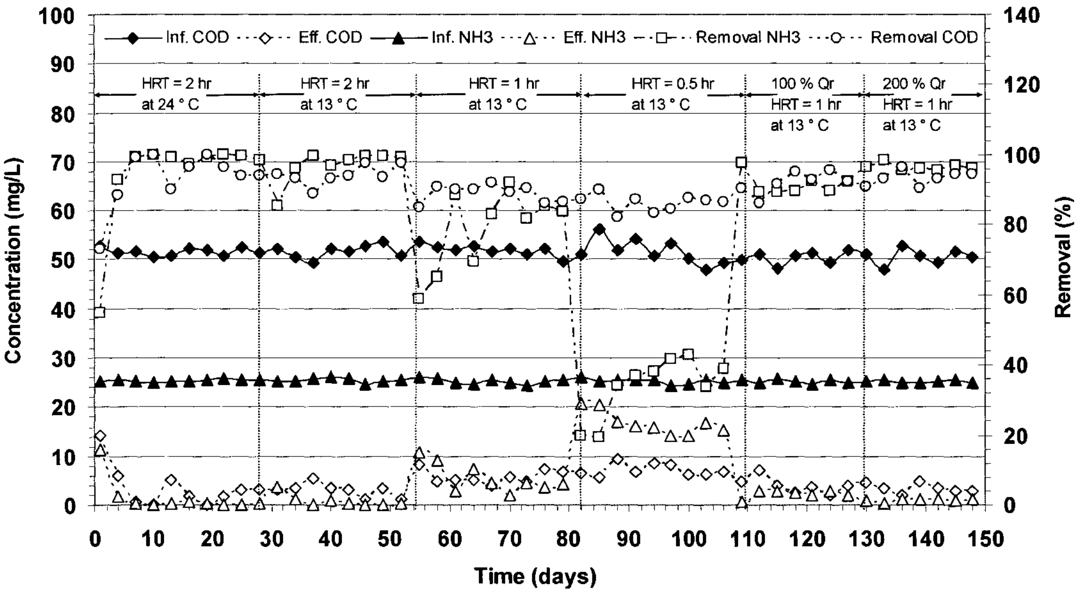


Figure 2. Gravel BAF performance at different HRTs and recirculation ratios at temperature of 13 °C.

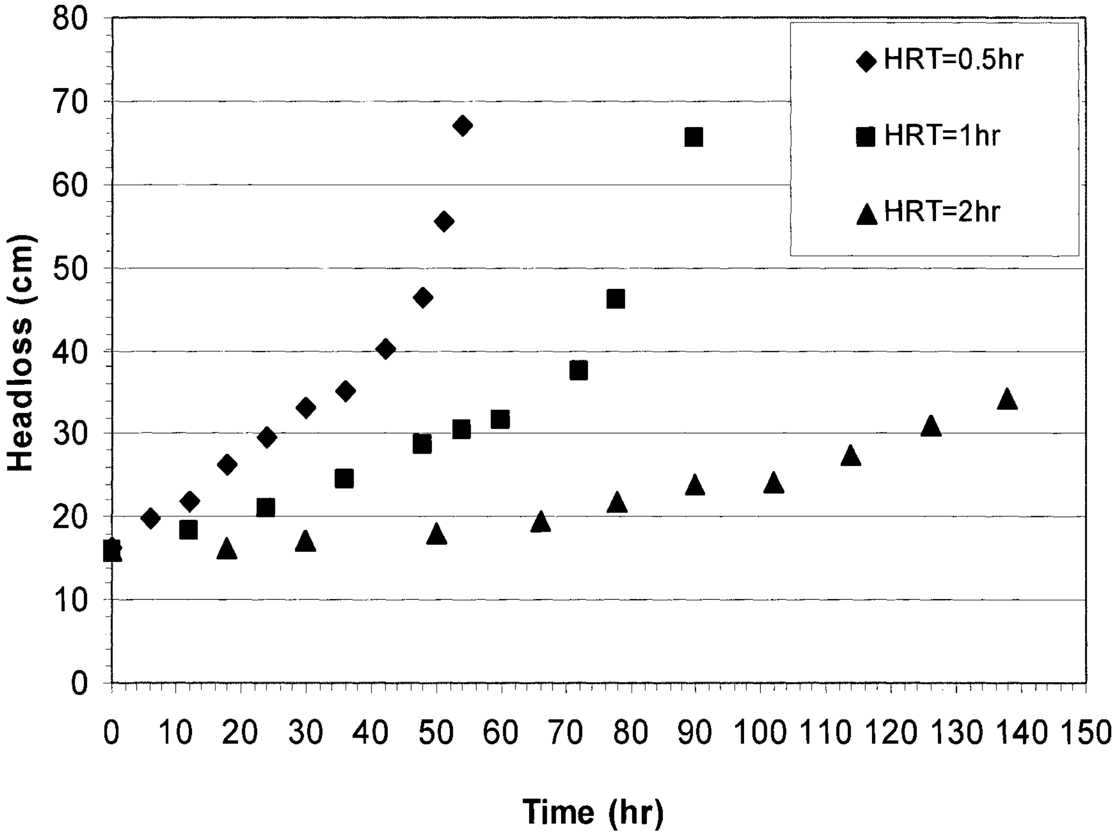


Figure 3. Headloss changes of gravel reactor at different HRTs and temperature of 13 °C.

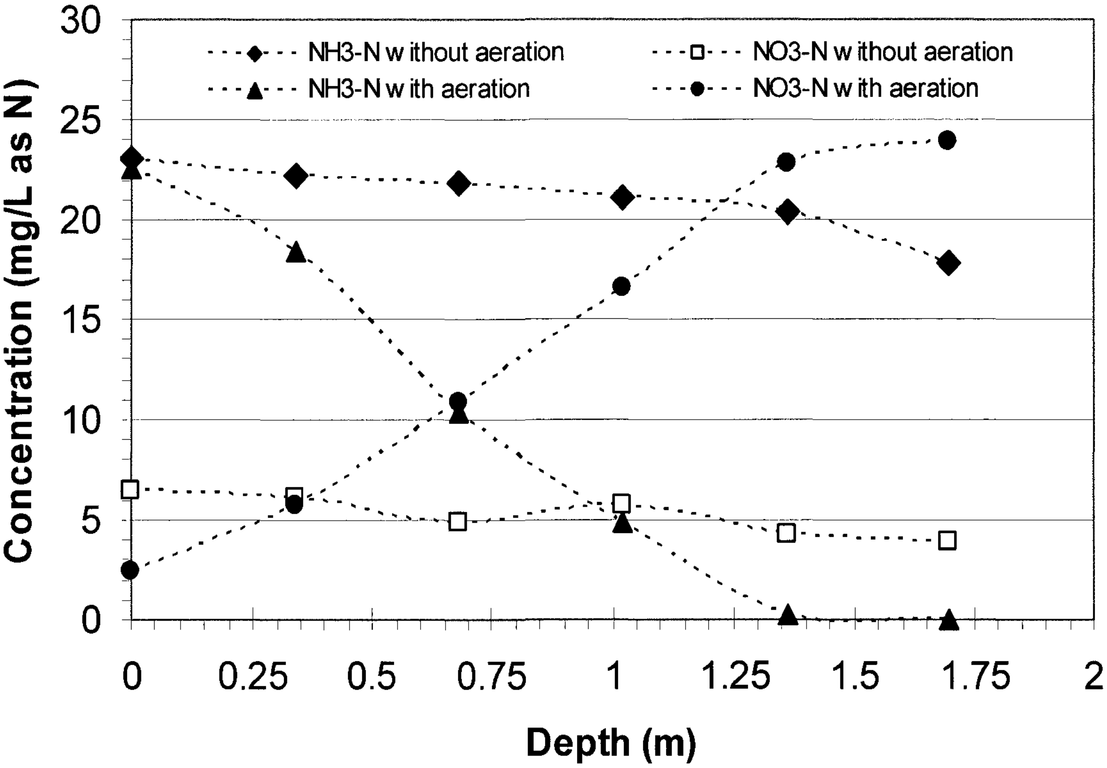


Figure 4. Ammonia and nitrate concentration profile with and without aeration at HRT of 2 hours and temperature of 13 °C.

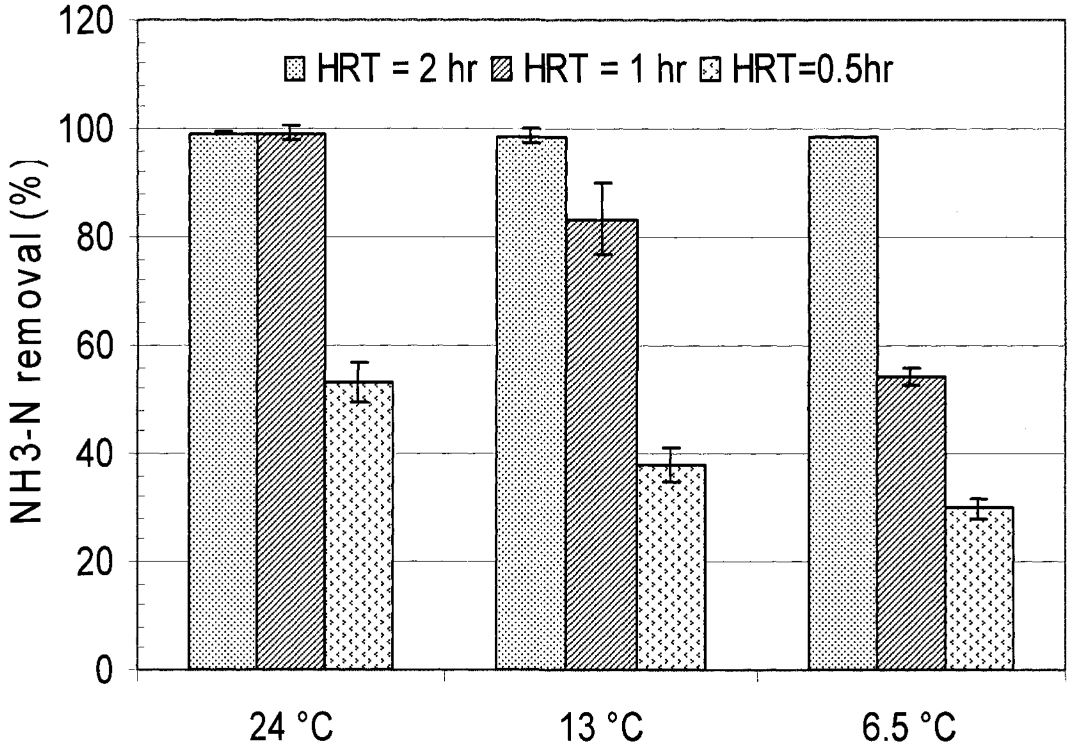


Figure 5. Effect of temperature of 6.5, 13 and 24 °C on ammonia removal in gravel BAF.

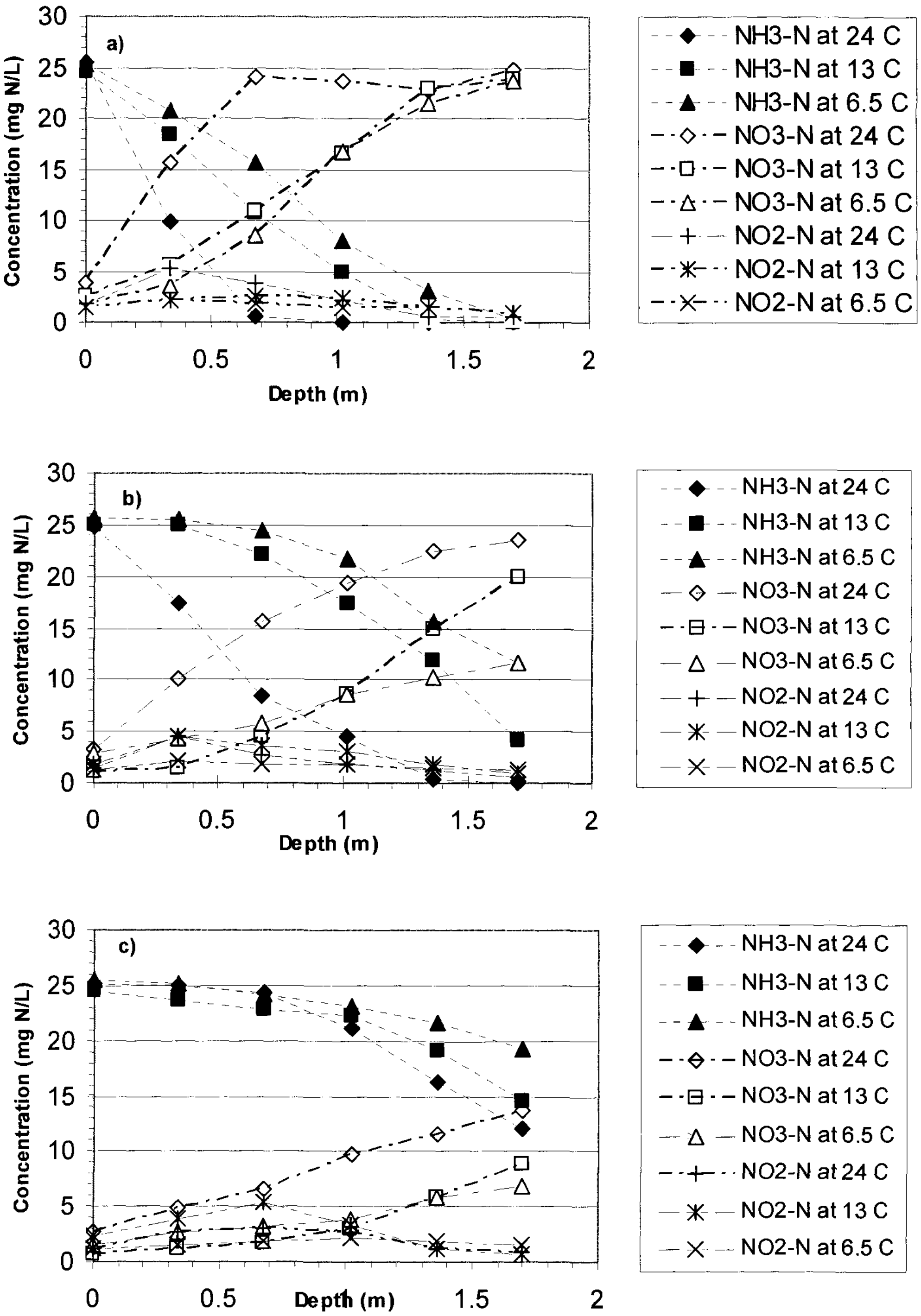


Figure 6. Ammonia, nitrate and nitrite concentration profiles at different temperatures and HRTs: (a) 2 hours, (b) 1 hour, and (c) 0.5 hour.

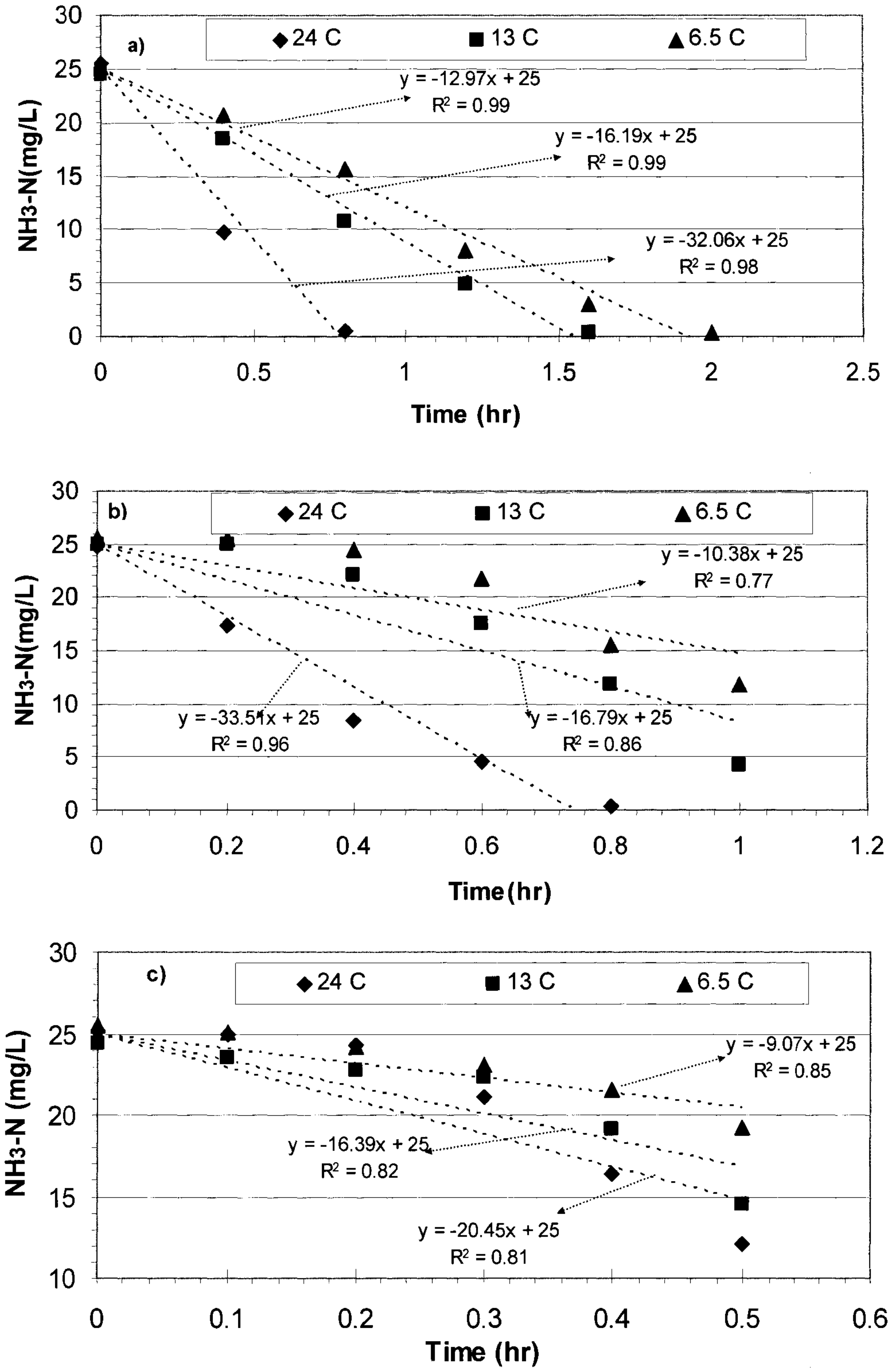


Figure 7. Kinetic plot of experimental data at different HRTs and temperatures in gravel media: (a) HRT = 2 hours, (b) HRT = 1 hour and (c) HRT = 0.5 hour.

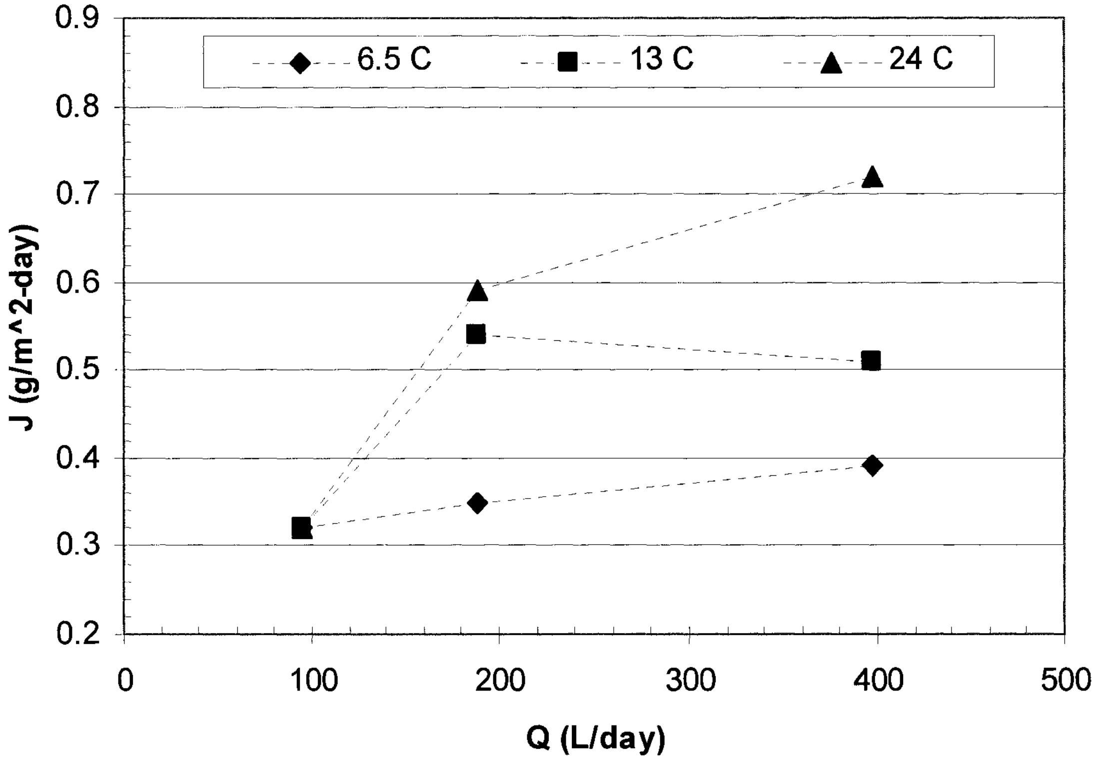
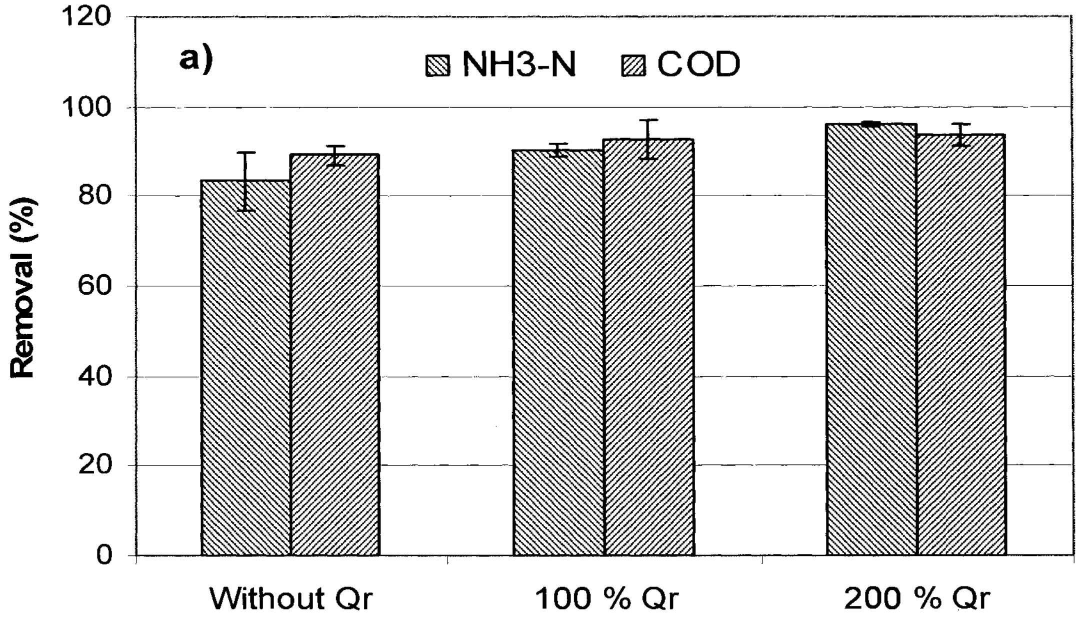


Figure 8. Ammonia flux to biofilm with different flow rates at different temperature of 6.5, 13 and 24 °C in gravel BAF.



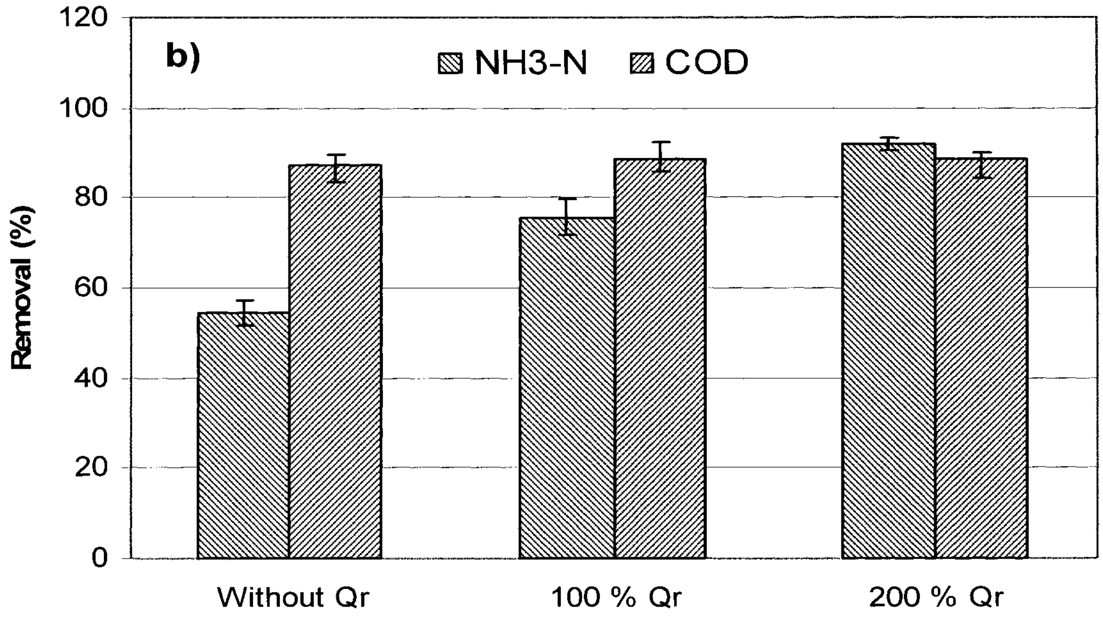


Figure 9. Ammonia and COD Removals with 100 and 200 % recirculation at the temperatures of (a) 13 °and (b) 6.5 °C with 1 hour HRT.

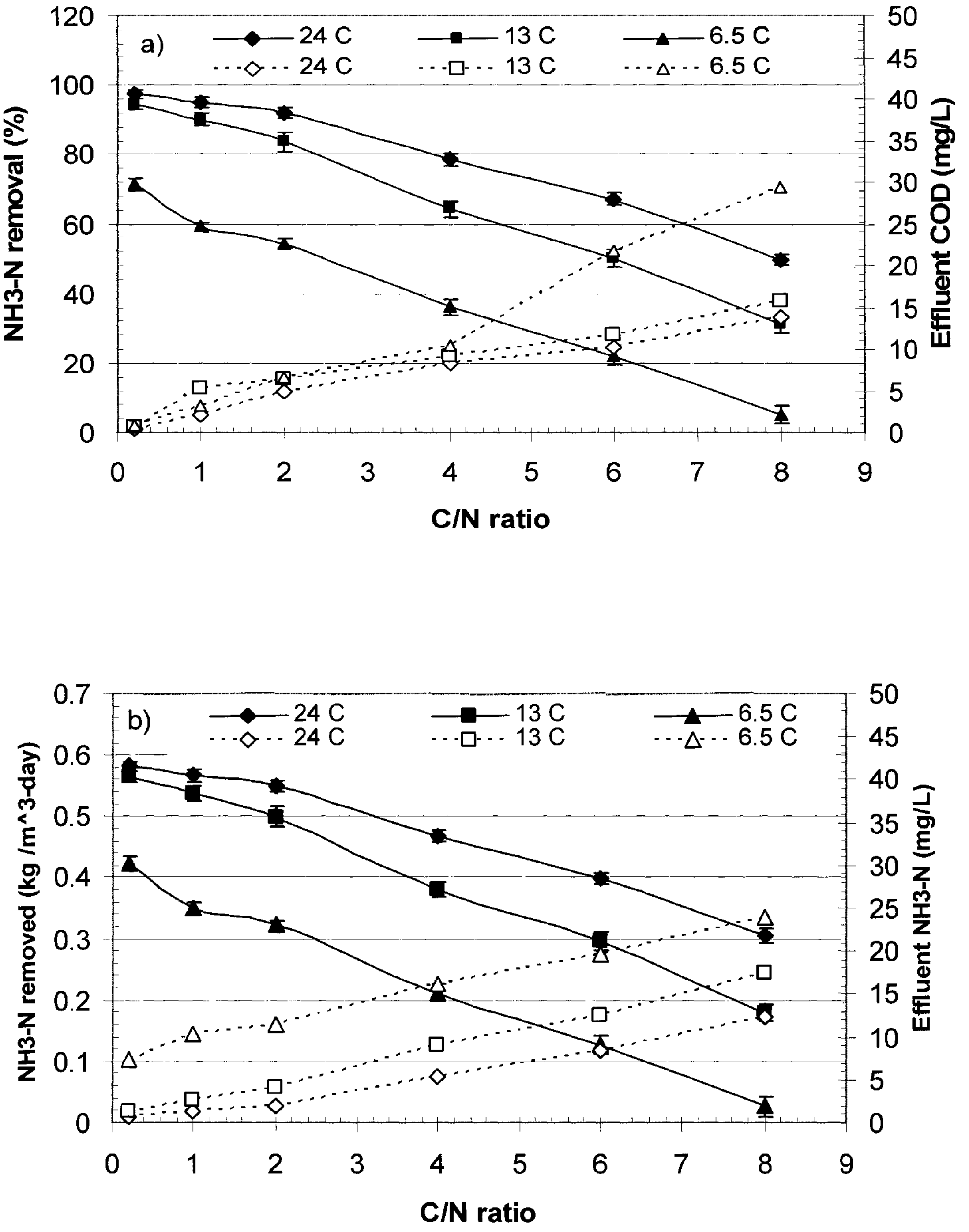


Figure 10. Effect of COD/NH_3-N ratio on (a) ammonia removal (%) and (b) ammonia mass removed at different temperatures in gravel reactor. (HRT = 1hr)

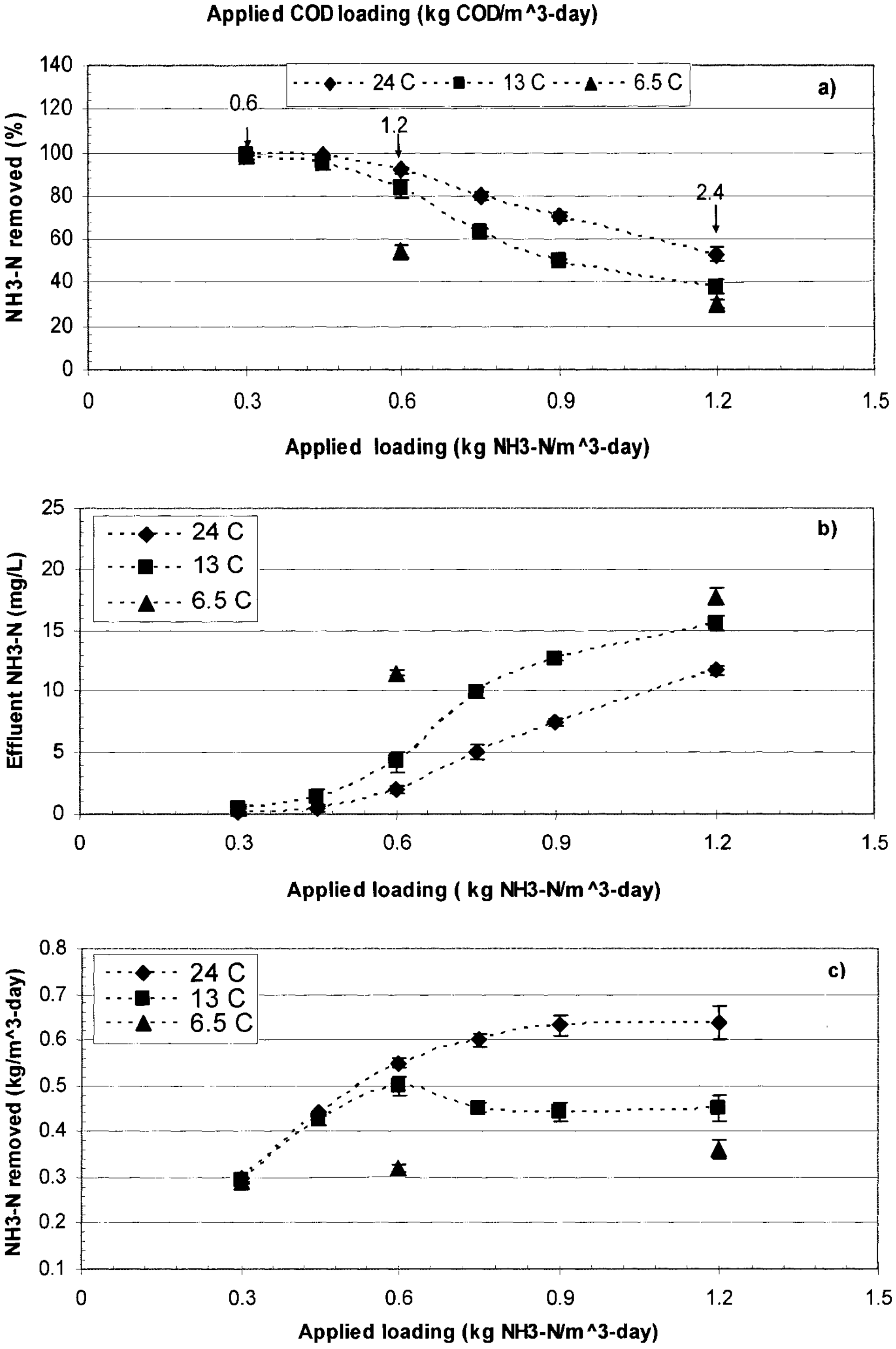


Figure 11. Effect of ammonia mass and SCOD loads at different temperature of 6.5, 13 and 24 °C in gravel BAF: (a) ammonia removal (%), (b) effluent ammonia concentration and (c) ammonia mass removed.

CHAPTER 5. NITRIFICATION AND DENITRIFICATION IN PARTIALLY AERATED BIOLOGICAL AERATED FITER (BAF)

A paper accepted for publication by Water Science and Technology

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ABSTRACT

A 104 mm (4-inch) diameter pilot-scale biological aerated filter (BAF) with a media depth of 2.5 m (8.3 feet) was operated with an anaerobic, anoxic and oxic zone at 23 ± 1 °C. The media for the anaerobic and anoxic zones was 10 mm diameter gravel while the media for oxic zone was 5 mm diameter gravel. The influent sCOD and total nitrogen concentrations in the feedwater were approximately 250 mg/L and 35 mg N/L, respectively. sCOD removal at optimum hydraulic retention time (HRT) of 3 hours with recirculation rates of 100, 200 and 300% in the column was more than 96%. Nitrification was found to be more than 96% for 3 hours HRT at 200% and 300% recirculation. Total nitrogen removal was consistently above 80% for 4 and 6 hours HRT at 300% recirculation. For 3 hours HRT and 300% recirculation, total nitrogen removal was approximately 79%. Denitrification was found to follow a half-order reaction kinetics while nitrification followed a zero-order reaction kinetics. The ammonia loading rates for maximum ammonia removed were 0.15 and 0.19 kg NH_3-N/m^3-day for 100% and 200% recirculation, respectively. The experimental results demonstrated that the BAF can be operated at an HRT of 3 hours with 200 - 300% recirculation rates with more than 96% removal of sCOD and ammonia, and at least 75% removal of total nitrogen.

Keywords Nitrification; Denitrification; Biological aerated filter; Hydraulic loading rate; Recirculation

INTRODUCTION

In recent years, various state regulatory agencies in the Midwest of United States have added or are considering adding nitrogen and phosphorus effluent limits to the National Pollution Discharge Elimination System (NPDES) permits of small community wastewater treatment facilities (0.04 m^3/s or 1 mgd) to address the increasing problem of euthrophication in the Midwest and hypoxia in the Gulf of Mexico. Many of these small communities use aerated lagoons for the treatment of wastewater which are not designed to remove nitrogen and phosphorus. To address the NPDES requirement, many communities are exploring modifications to their aerated lagoons, incorporating an add-on treatment unit to the aerated lagoons or by using a stand alone secondary treatment unit for nitrogen removal.

Biological removal of nitrogen requires an anoxic stage for the denitrification of nitrates and nitrites where nitrate or nitrites are reduced to nitrogen gas under anoxic conditions. In conventional suspended growth activated sludge systems, nitrogen removal is accomplished using two separate tanks: an anoxic and an aerobic tank (Metcalf and Eddy, 2003). An alternative approach for nitrogen removal is to use continuous upflow biofiltration such as a modified biological aerated filter (BAF) (Rogalla et al., 1992; Boller et al., 1994; Pujol et al., 1994., Peladan et al., 1996; Ong et al., 2002). In a modified BAF, the anoxic zone can be created by introducing the wastewater in an upflow mode at the bottom of the media filter creating an anaerobic/anoxic zone and by injecting air about midway of the column creating an oxic zone from the aeration point to the outlet of the filter. Recycling of the nitrified effluent back into the anoxic zone is essential in the removal of nitrogen (Metcalf and Eddy, 2003). By creating the two zones within a column, the BAF can be made relatively compact as compared to a suspended growth activated sludge system (Rogalla et al., 1900; Chui et al., 1996; Sanz et al., 1996; Sen and Dentel, 1998; Tay et al., 2003). For example, Chui et al. (1996) found that without recirculation of the effluent, between 41% and 86% of the total nitrogen was removed for an influent concentration of 250 mg N/L and a volumetric loading of up to 1 kg N/m^3-day. Another advantage with a partially aerated BAF is that ammonia oxidation as well as denitrification can be achieved within a short hydraulic retention time (HRT)(less than 6 hours) (Rogalla et al., 1900; Sanz et al., 1999; Tay et al., 2003).

In this study, a partially aerated BAF system using two media sizes: a larger size media for the anaerobic and anoxic zones and a smaller size media for the oxic zone, is investigated as a secondary treatment and for nitrogen removal. Although some work has been done by other researchers (Rogalla et al., 1990 and Sanz et al., 1996), further indepth study is needed on the impact of various operating conditions and recirculation on the total nitrogen and soluble chemical oxygen demand (sCOD) removal. The objective of this study is to investigate the impact of hydraulic loading rates and recirculation rates on total nitrogen and sCOD removal for a partially aerated BAF.

Materials and Methods

A pilot-scale partially aerated BAF was set up with an anaerobic, anoxic and oxic zone as shown in Figure 1. This partially aerated filter has a diameter of 104 mm (4 inches) to minimize wall effect and a media depth of 2.5 m. The average diameter of the gravel used in the anaerobic and anoxic zone was 10 mm while in the oxic zone, the average diameter of the gravel was 5 mm. The measured porosities for 5 mm and 10 mm gravel were 0.36 ± 0.01 and 0.43 ± 0.02, respectively. Measured specific surface areas for the 5 mm gravel and 10 mm gravel were 0.93 and 0.62 x 10^6 m^2/m^3, respectively.

The oxic zone was created by supplying air at about midway of the column while the anoxic zone was created by recycling the treated effluent to the bottom of anoxic zone as shown in Figure 1. Experiments were initially conducted with a total HRT of 8 hours with 1.5 hours HRT for the anaerobic zone, 2.5 hours HRT for the anoxic zone and 4 hours HRT for the oxic zone. Other total HRTs tested include 2, 3, 4 and 6 hours. Recirculation of the treated effluent at 100%, 200% and 300% of the influent flow rate were investigated for total nitrogen and sCOD removal. The partially aerated BAF was seeded using activated sludge from the aeration tank of a municipal wastewater treatment plant. The partially aerated BAF was operated for a period of time (about 60 days) at a total HRT of 8 hours until steady state removals for sCOD, ammonia and nitrate were above 90 % before the operating conditions were varied.

The composition of the synthetic wastewater used as feed wastewater is shown in Table 1. The synthetic wastewater contained organic carbon, essential nutrients, ammonia, and minerals to simulate the characteristics of raw domestic wastewater. sCOD and total nitrogen concentrations of the feed water were approximately 250 mg/L and 35 mg N/L, respectively.

The nominal flow rates used were 57, 75, 113, 151, and 226 L/day (see Table 2) giving hydraulic loading rates ranging from 0.3 to 1.2 m^3/m^2-hr. The air flow rate was kept constant at 2,500 mL/min maintaining a 5 mg/L of dissolved oxygen (DO) concentration in the oxic zone. Measurement throughout the study included influent and effluent concentration of sCOD, total nitrogen, phosphorus, alkalinity, ammonia, nitrate and nitrite. Other parameters that were measured include pH, ORP, temperature, DO, head losses and flow rates. One-hour composite samples were used for the analyses. Analyses were conducted according to Standard Methods (APHA, 2002).

RESULTS and DISCUSSION

Ammonia and nitrate concentrations within the BAF without recirculation

Profiles of ammonia and nitrate concentrations within the BAF column at HRT of 8 hrs without recirculation are presented in Figure 2. In the anaerobic and anoxic zones, ammonia concentrations remained fairly constant or increased slightly due to ammonification of organic nitrogenous compounds. In the oxic zone, the disappearance of ammonia was in good agreement with the appearance of nitrate (see Figure 2). As shown in Figure 2, approximately 22 mg N/L of ammonia was converted to nitrate while the rest most probably were assimilated into the biomass. Without recirculation of effluent to the anoxic zone, total nitrogen removal was approximately 35% mainly due to assimilation of nitrogen in the biomass.

Effect of recirculation on sCOD, ammonia and nitrate removals

The impact of 100%, 200% and 300% recirculation of the effluent on sCOD, ammonia and nitrate removal while maintaining a system HRT of 8 hours are presented in Figure 3a, 3b and 3c, respectively. Influent sCOD, ammonia and total nitrogen concentrations in the feedwater were approximately 250 mg/L, 25 mg N/L and 35 mg N/L, respectively. About 30% of the sCOD were removed in the anaerobic zone followed by partial dilution of the recirculating effluent in the anoxic zone as shown in Figure 3a. Ammonia was completely removed for 100%, 200% and 300% recirculation (see Figure 3b). Recirculation of effluent into the anoxic zone resulted in an increase in nitrate concentration (between 3 to 6 mg N/L) but this nitrate concentration was reduced to less than 2 mg N/L indicating that denitrification was occurring within this zone (see Figure 3c). With more recirculation, a lower nitrate concentration was obtained in the anoxic zone and nitrate concentrations in the effluent were 9.6, 5.4 and 4.4 mg N/L for 100%, 200% and 300% recirculation, respectively.

In summary, the experimental results showed that with 100% recirculation, approximately 97 ± 0.75%, 100 ± 0.1% and 56 ± 2.1% of sCOD, ammonia and nitrate were removed, respectively. With 200 % recirculation, approximately 98%, 100% and 75% of sCOD, ammonia and nitrate were removed, respectively. With 300% recirculation, the removal rates increased to approximately 98%, 100% and 80% of sCOD, ammonia and nitrate, respectively. Total nitrogen removal with 100%, 200% and 300% recirculation were 68 ± 2.1%, 81 ± 1.8% and 83 ± 1.4%, respectively.

Effect of HRTs on sCOD, ammonia and nitrate removal

Figure 4 shows the average percentage removal (with 95% confidence intervals) for sCOD, ammonia-N and total nitrogen at 2, 3, 4 and 6 hours HRT and 100%, 200% and 300%

recirculation rates. sCOD removal remained high at more than 96% for 3 hours HRT and for all recirculation but had only 92% removal at 2 hours HRT at 300% recirculation. Similarly, for ammonia, complete nitrification was achieved at 4 and 6 hours HRT for all recirculations. But, for 2 and 3 hours HRTs and 100% recirculation, ammonia removal was 69% and 42%, respectively. However, with 200% and 300% recirculation, nitrification improved to 96 ± 2.1% at 3 hours HRT while nitrification improved to 63 ± 1.2% and 72 ±1.3% for 2 hours HRT, respectively. The total nitrogen removed were the same for 4 and 6 hours HRTs for 100%, 200% and 300% recirculation. Total nitrogen removal was highest at 82 ±1.1% for either 4 or 6 hours HRT and 300% recirculation. At a shorter HRT of 3 hours and 300% recirculation, the total nitrogen removed reduced slightly to about 79 ± 0.7% which was statistically different from the removal rate at 4 hours HRT and 300% recirculation. At 2 hours HRT, total nitrogen removal was about 63 ± 2.8% due to the lack of sufficient bed volume and contact time within the column. The experimental results showed that with 200 or 300% recirculation, it is possible to achieve more than 90% ammonia and sCOD removal and more than 75% nitrogen removal even at an HRT as low as 3 hours.

Nitrification and denitrification kinetics within the column

Denitrification biotransformation rates, k, may be used to describe denitrification that is occurring within the BAF. Under steady state conditions and using a simple input-output model, the overall nitrate biotransformation rates within the BAF system was assumed by Hultman et al. (1994) to be approximately equal to a half-order reaction with respect to nitrate concentration, i.e.,:

$$dC/dt = -k\ C^{0.5} \qquad (1)$$

This equation yields the following:

$$C^{0.5} - C_o^{0.5} = -0.5\ k\theta \qquad (2)$$

where C_o is the influent nitrate concentration (mg N/L), C is the nitrate concentration (mg N/L), k is the half order nitrate biotransformation rate ($[mg/L]^{0.5}/hr$); and θ is the empty bed hydraulic retention time (hr). Equation 2 may be rewritten as:

$$C^{0.5} - C_o^{0.5} = -0.5\ k\ h/v \qquad (3)$$

$$\text{or}\ \ [C/C_o]^{0.5} = 1 - \alpha\ h/[C_o]^{0.5}$$

where h is the bed height (m), v is hydraulic loading rate (m/hr) and α is defined as the reaction rate per hydraulic load (k/2 v). A plot of $[C/C_o]^{0.5}$ versus $h/[C_o]^{0.5}$ provides a straight line curve with a slope equal to α and an intercept of 1. The half order biotransformation rate, k can be determined from the α value.

In the case of nitrification, some studies have indicated that the overall ammonia biotransformation rate within the BAF systems may be expressed as a zero order reaction with respect to the ammonia concentration (Boller and Gujer, 1986). The equation is as follows:

$$C - C_o = -\ k\ t \quad = \quad -\ k\ h/v \qquad (4)$$

where C_o is the influent ammonia concentration (mg N/L), C is the actual ammonia concentration (mg N/L), k is the zero order ammonia biotransformation rate (mg NH_3-N/L-hr); and t is time of reaction (hrs), h is the depth of the sand media (m) and v is hydraulic loading rate (m/hr). A plot of C versus t provides a straight line curve with a slope equal to k and an intercept of influent ammonia concentration.

Representative plots for ammonia and nitrate biotransformation rates at different recirculation rates and for a HRT of 8 hours and temperature of 23 °C are presented in Figures 5 and 6. The overall biotransformation rates for nitrification and denitrification along with 95% CI are summarized in Table 3. In these kinetic studies, it was observed that the optimal recirculation rate at HRT of 8 hrs and 24 °C was at 300% recirculation ratio. With 300% recirculation rate, the nitrate and ammonia transformation rate were 22.68 ± 1.19 $[mg/L]^{0.5}$/hr and 11.66 ± 2.23 mg NH_3-N/L-hr, respectively. In addition, comparison of the biotransformation rates under anoxic and oxic conditions at different recirculation ratios showed that as recirculation ratio increased, the rate of ammonia and nitrate transformation increased due to fact that high rate flow may allow enhanced mass transfer of substrates and dilution of the wastewater (Tschui et al., 1994; Peladan., 1996; Pujol., 2000).

pH, ORP, nitrite and phosphorus changes within the column

A set of representative data of pH, ORP, nitrite and phosphorus changes within the column for 100% recirculation at HRT of 8 hours are presented in Figure 7a and 7b. The experimental results showed that the ORP in the anoxic zone was about -200 to -300 mV while the ORP value in the oxic zone was about 172 mV (see Figure 7a). In addition, Figure 7a shows that under anoxic zone, pH value increased to 7.9 due to production of alkalinity (hydroxyl ion) while under oxic zone, pH decreased to 7.3 due to consumption of alkalinity during nitrification. The effluent nitrite concentrations were monitored and found to be less than 1 mg N/L in the reactor (see Figure 7b).

The phosphorus within the system were measured to assess whether biological phosphorus removal may be occurring within the reactor since the BAF with anaerobic, anoxic and oxic zones along with recirculation may favor biological phosphorus removal. After six months of running the reactor, biological phosphorus removal was not observed to be occurring (see Figure 7b). A probable reason is that phosphorus accumulating organisms in the anaerobic zone of the attached growth BAF system was unable to adequately move to the anoxic/oxic zone where phosphorus can be luxuriously taken up (Jeon et al. (2001)). Another probable reason is that phosphorus removal may be limited when nitrate concentrations are relatively high (8 -15 mg N/L) in the reactor system (Metcalf and Eddy, 2003).

NH_3-N, sCOD and TN mass removed at different loading rates

Another approach in presenting the results is to plot the mass removed for different ammonia and sCOD loading rates as shown in Figure 8. As in Figure 8a, with 100% and 200% recirculation, the mass of ammonia removed increased until approximately 0.15 and 0.19 kg NH_3-N/m^3-day of applied loading rates before it started to decrease. Based on the experimental results, the maximum ammonia masses removed were approximately 0.15 and 0.19 kg NH_3-N/m^3-day for 100% and 200% recirculation, respectively. It can be assumed that the maximum ammonia mass removed at 300% recirculation is greater than 0.21 kg NH_3-N/m^3-day. In the case of sCOD, removal of sCOD were directly proportional to the sCOD loading rates indicating that within the loading rates tested, sCOD removal was not impacted (Figure 8b). A slight divergence was observed at a sCOD loading of 1.97 kg/m^3-day but this was statistically insignificant.

Total nitrogen removed increased with TN loading rates but level off at about 0.27 kg TN/m^3-day (see Figure 8c) for 100% and 200% recirculation. With 300% recirculation, the BAF was able to handle a higher total nitrogen loading rate but seems to be tapering with increasing loading rates.

CONCLUSIONS

Experimental results showed that the partially aerated BAF may be used as a treatment process for nitrogen and sCOD removal to meet discharge requirements. The experimental results showed that nitrate and total nitrogen removal efficiencies increased as recirculation rates increased. The experimental results demonstrated that the BAF can be operated at an HRT of 3 hours with 200 - 300% recirculation rates with more than 96 % removal of sCOD and ammonia and at least 75% removal of total nitrogen. Denitrification was modeled using a half-order reaction while nitrification was modeled using a zero-order reaction. The kinetic rates for nitrification and denitrification showed that as HRT decreased, the rate of ammonia and nitrate transformation increased. The ammonia loading rates for maximum ammonia removed were 0.15 and 0.19 kg NH_3-N/m^3-day for 100% and 200% recirculation, respectively. Of the three recirculation rates, the 300% recirculation showed the best mass removal of nitrate and TN. This study demonstrates that a partially aerated

BAF can be a compact system operating at a low HRT that can be used for small communities in treatment of their wastewater for nitrogen removal.

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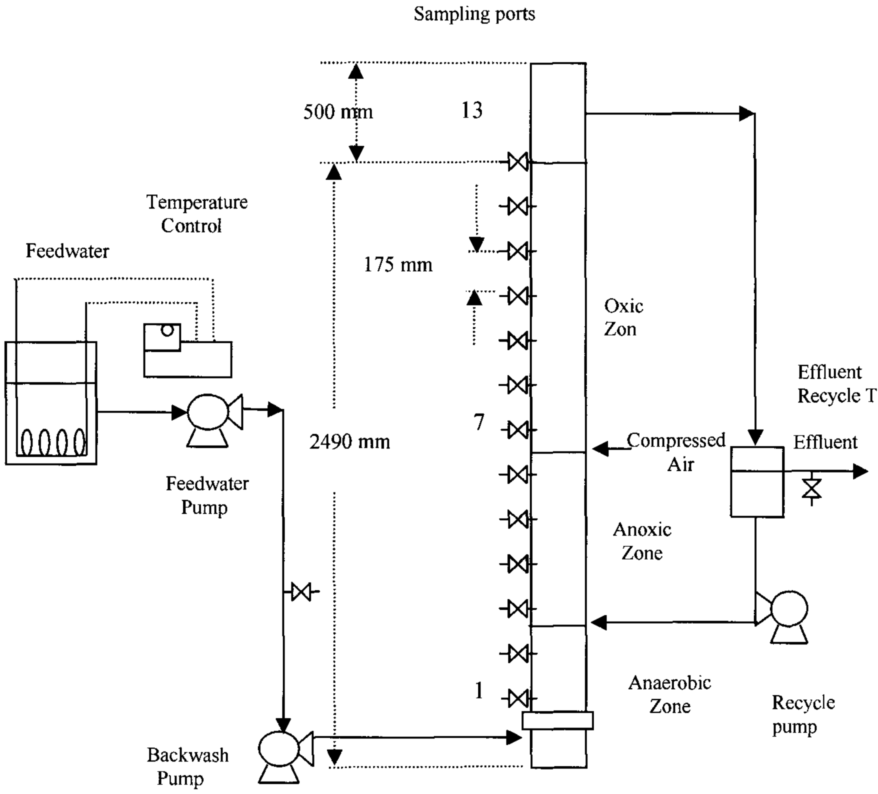

Figure 1. Flow diagram of BAF system with anaerobic, anoxic and oxic zones.

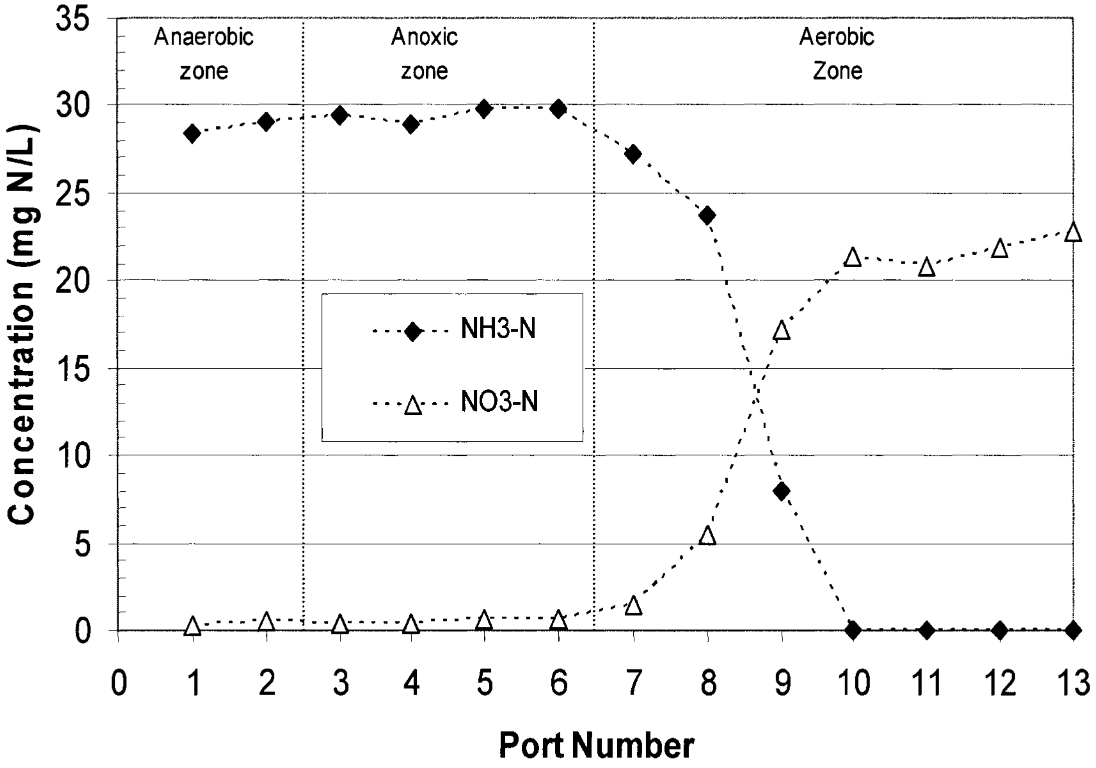

Figure 2. Ammonia and nitrate concentrations within the column without recirculation at HRT of 8 hrs (Total depth of column – 2.49 m, Distance between two sampling ports 175 mm).

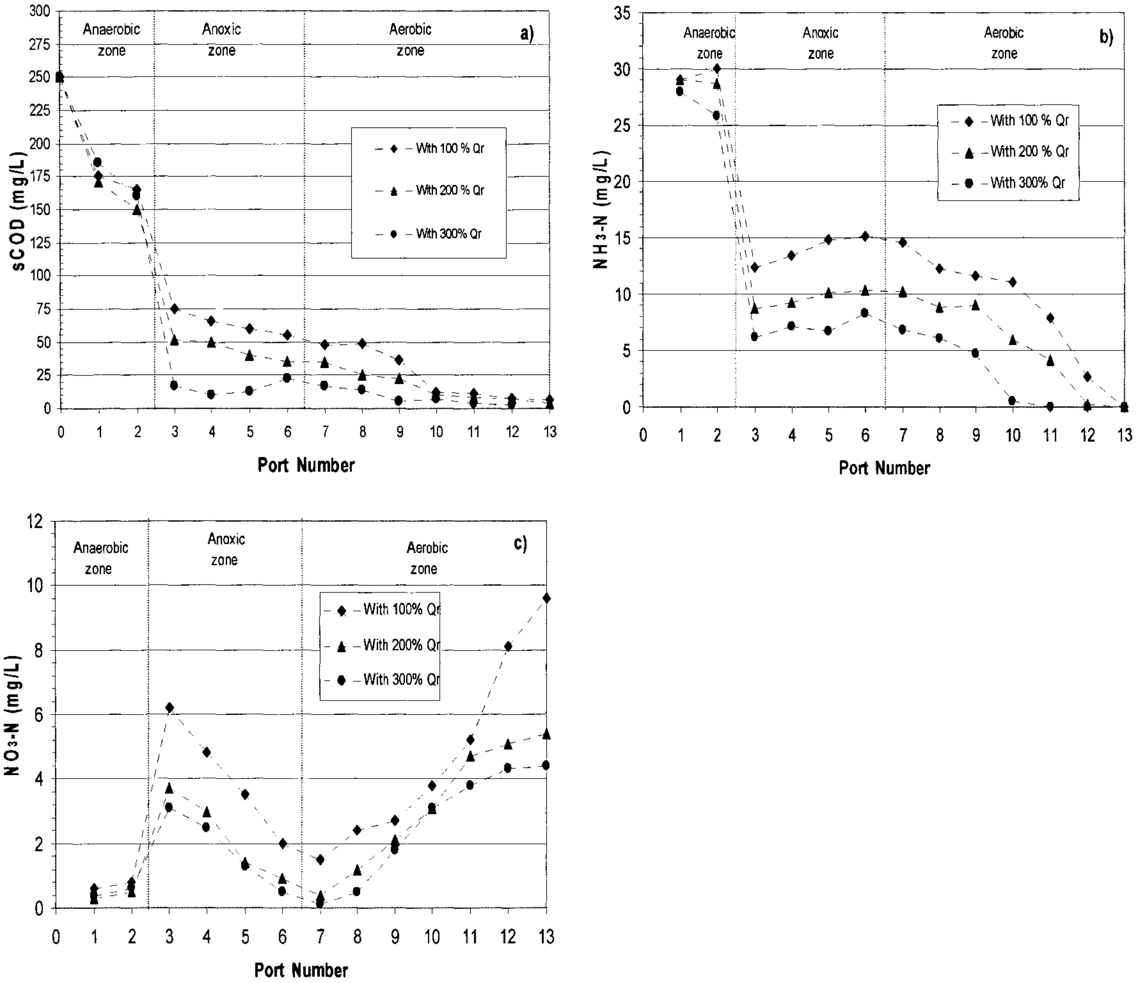

Figure 3. (a) sCOD, (b) ammonia and (c) nitrate concentration within the column with 100 %, 200 % and 300 % recirculation (Qr)(Total depth of column – 2490 mm, Distance between two sampling ports 175 mm). (HRT = 8 hrs)

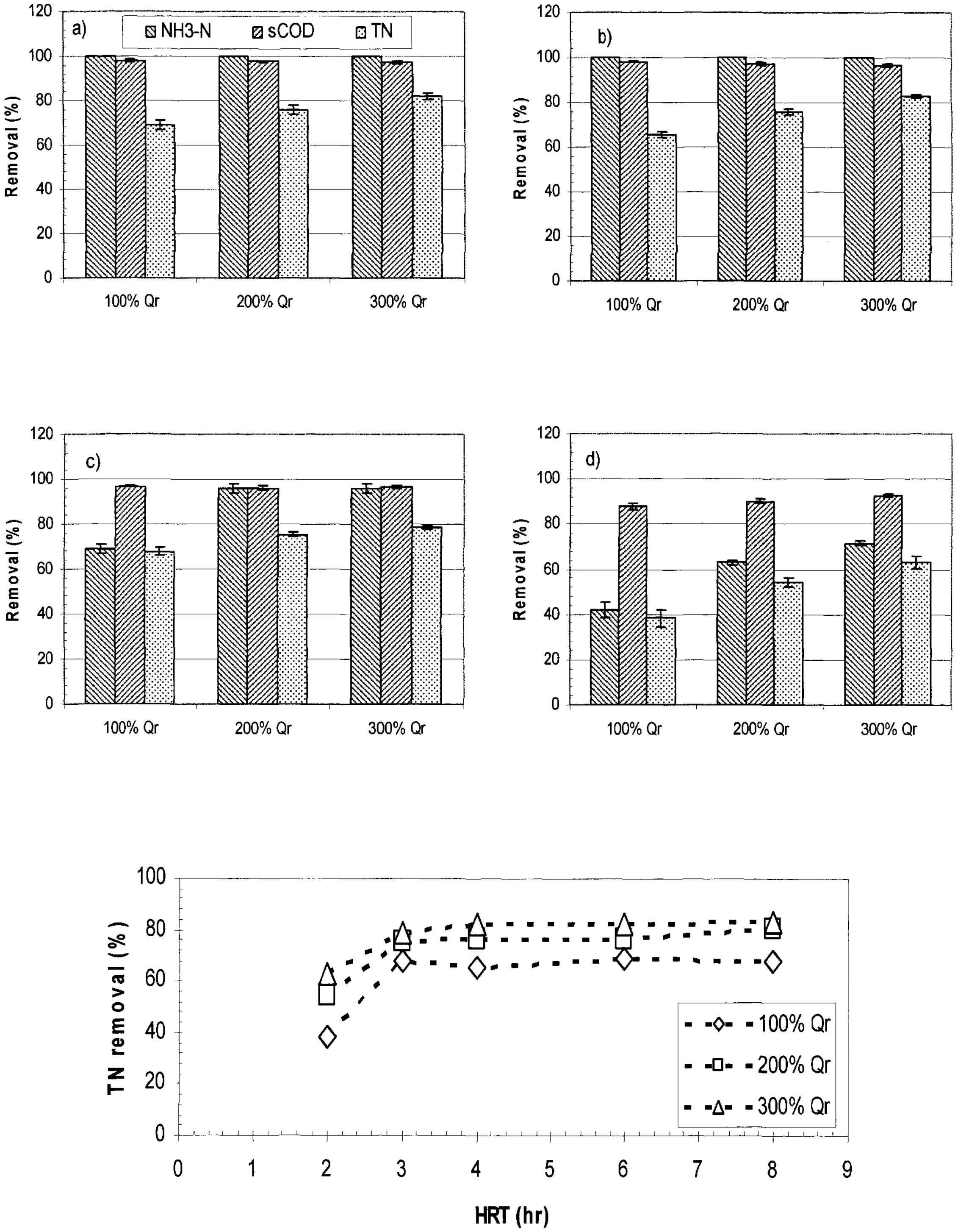

Figure 4. Effect of HRT and recirculation rate on ammonia, sCOD, and total nitrogen (TN) removal in the column: (a) HRT = 6hrs, (b) HRT = 4 hrs, (c) HRT = 3 hrs and (d) HRT = 2 hrs.

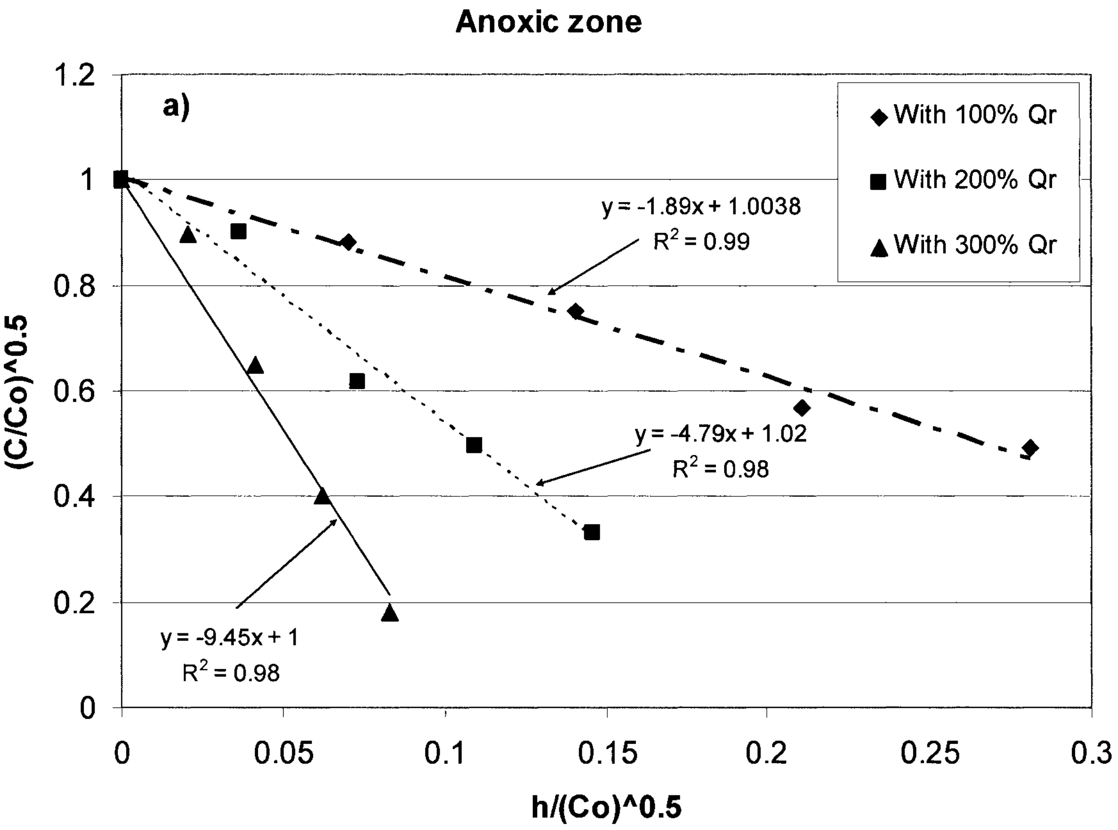

Figure 5. Kinetic plot of experimental data on denitrification at HRT of 8 hours and 24 ± 1°C with different recirculation ratios.

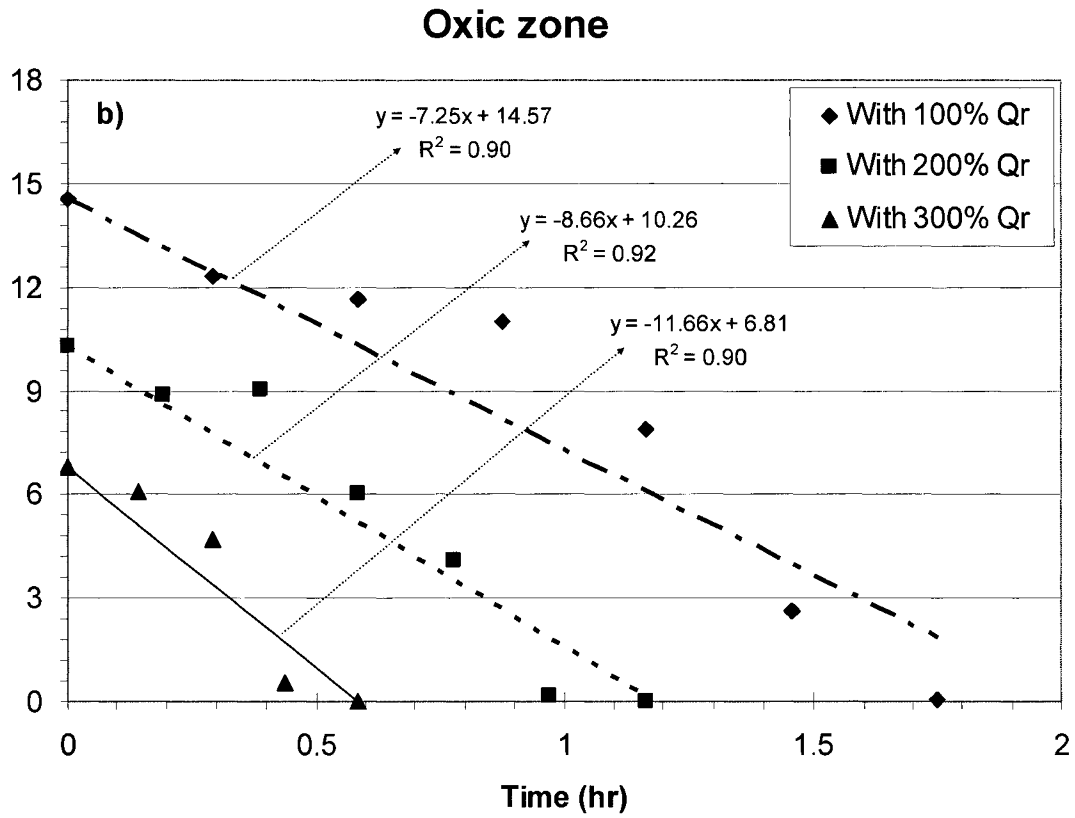

Figure 6. Kinetic plot of experimental data on nitrification at HRT of 8 hours and 24 ± 1°C with different recirculation ratios.

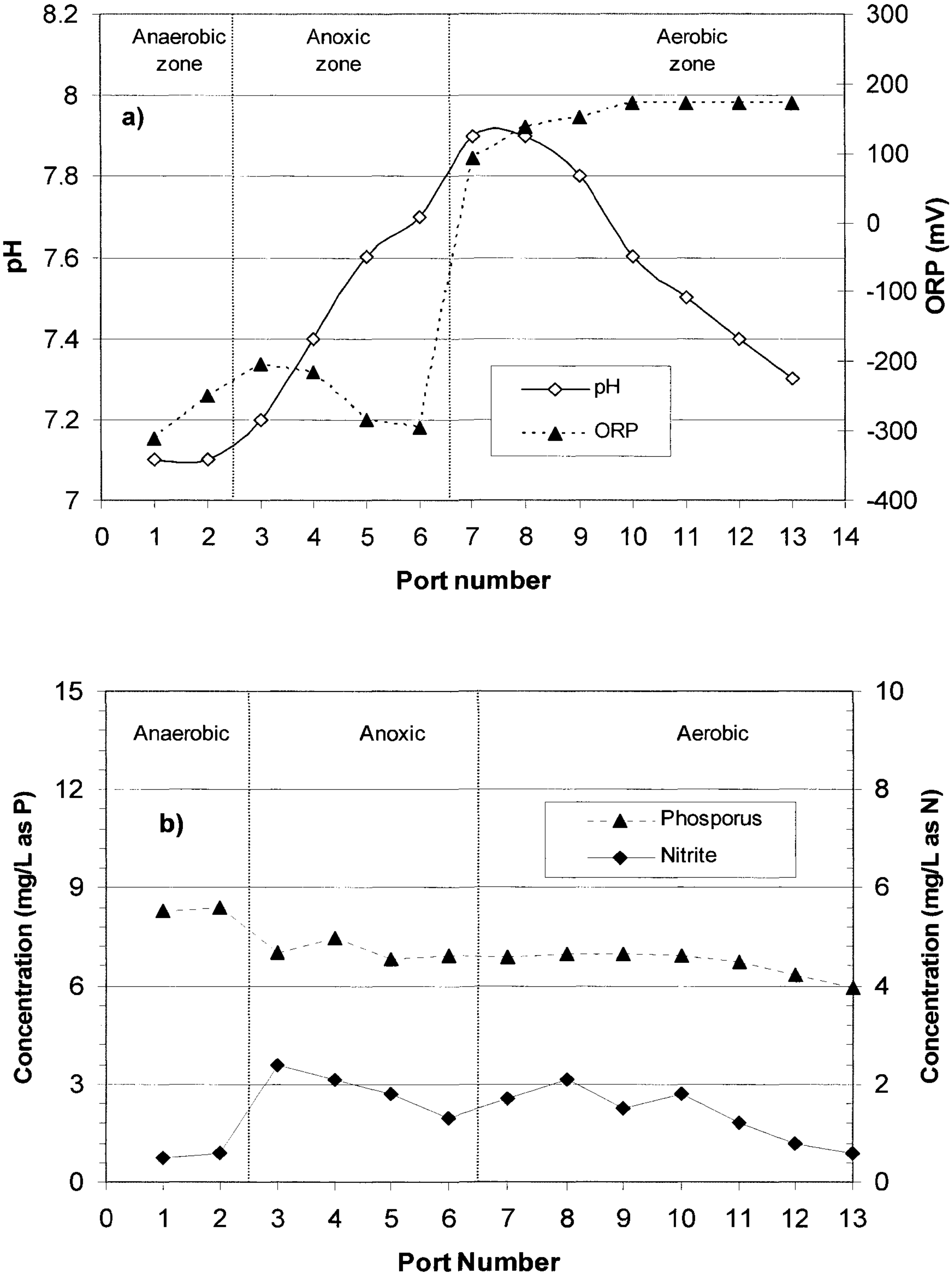

Figure 7. (a) ORP and pH, and (b) nitrite and phosphorus changes under anaerobic, anoxic and oxic conditions within the column at 8 hours HRT with 100% recirculation.

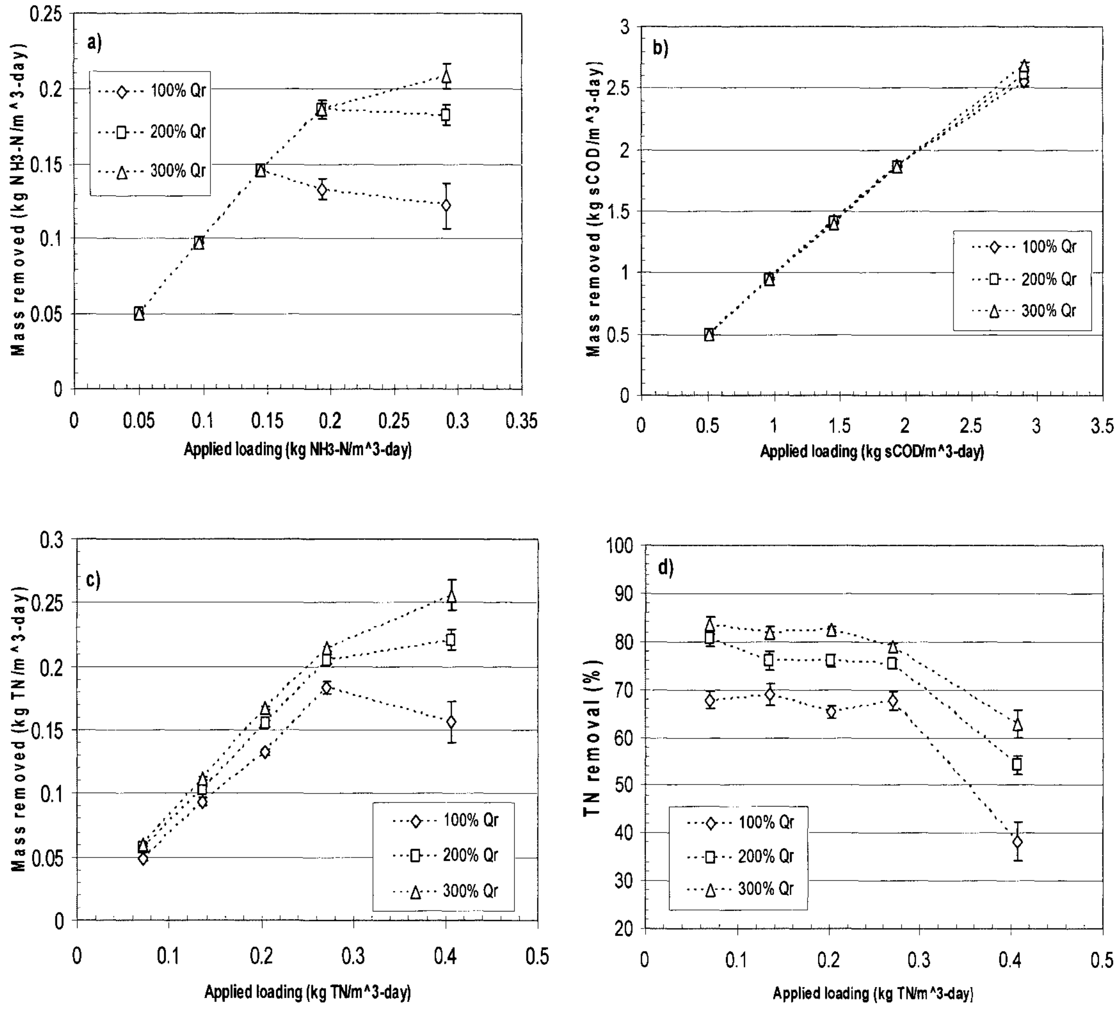

Figure 8. Effect of (a) sCOD, (b) ammonia and (c) total nitrogen loading rates for mass removed and (d) TN removal (%).

Table 1. Composition of synthetic wastewater.

Ingredient	Concentration (mg/L)
Calcium Chloride ($CaCl_2$)	40
Magnesium Sulfate ($MgSO_4$)	6
Ferric Chloride ($FeCl_3$)	4
Sodium Biphosphate (NaH_2PO_4)	31
Sodium Bicarbonate ($NaHCO_3$)	100*
Potassium Chloride (KCl)	4
Sodium Acetate (CH_3COONa)	282 (sCOD = 220 mg/L)
Nutrient Broth	15 (sCOD =10 mg/L)
Ammonium Chloride (NH_4Cl)	95
Isomil (baby liquid food)	22 mL in 400 L of wastewater (sCOD = 20 mg/L)

* Reported as mg/L as calcium carbonate

Table 2. Experimental operating condition for biological aerated filters.

Parameter	Value
Diameter of BAF	104 mm (4 inches)
Media Depth	2.5 m (8.3 feet)
Influent PO_4^{-3}-P	8 mg as P/L
Influent NH_3-N	25 mg/L
Influent sCOD	250 mg/L
Influent Alkalinity	100 mg/L as $CaCO_3$
Influent Total Nitrogen	35 mg/L
Flow rates	57, 75, 113, 151, 226 L/day
Hydraulic Retention Times	2, 3, 4, 6 and 8 hr
Recirculation ratios	100%, 200%, 300%
Influent pH	7.0
Backwash Flow rate	3.33 L/min
Air Flow Rate	2.5 L/min
Temperature (Influent)	23 ± 1 °C

Table 3. Estimation of overall ammonia and nitrate biotransformation rates in partially aerated BAF for hydraulic retention time of 8 hours at 24 ± 1 °C with different recirculation rates (Qr).

Operation condition	**Anoxic zone**			**Oxic zone**	
	Half order			**Zero order**	
	α (1/m)[a]	k([mg/L]$^{0.5}$/hr[a]	R^2	k(mg NH_3-N/L-hr)[a]	R^2
100% Qr	1.89 ± 0.11	2.27 ± 0.13	0.99	7.25 ± 1.13	0.90
200% Qr	4.79 ± 0.54	8.62 ± 0.97	0.98	8.66 ± 1.13	0.92
300% Qr	9.45 ± 0.79	22.68 ± 1.19	0.98	11.66 ± 2.23	0.90

[a] 95% confidence interval

CHAPTER 6. PHOSPHORUS REMOVAL USING ATTACHED GROWTH ALTERNATING FILTER

INTRODUCTION

Enhanced biological phosphorus removal is an increasingly popular technology for removing phosphorus (P) in wastewater. In biological phosphorus removal (Jeon et al., 2000; Park et al., 2001; Wang et al., 2001), phosphorus-accumulating microorganisms (PAOs) are capable of uptaking volatile fatty acids (VFAs) under the anaerobic condition into their cell forming polyhydroxybutylate (PHB) at the expense of the intercellular adenosinetriphosphate (ATP), and at the same time release phosphorus. Under aerobic condition, PAOs used the PHB as a carbon source and at the same time uptake phosphorus from the wastewater resulting in a net gain in phosphorus uptake. Under the anoxic condition, denitrifying phosphorus-accumulating organisms (dPAOs) may uptake phosphorus by using PHB and nitrate as electron acceptor in wastewater.

Among the various biofilm technologies, biological aerated filter (BAF) is considered as a system incapable of enhanced biological phosphorus removal (EPBR). Franci Gonçalves and Rogalla (1992) have investigated phosphorus removal using fixed biological bed up-flow reactors. They showed that it is possible to grow PAOs in fixed biofilm by alternating anaerobic and aerobic conditions. In addition, successful nitrogen and phosphorus removal at high strength wastewater was investigated by Tay et al. (2003) using a single fixed-bed filter with anaerobic, anoxic and aerobic zones and recirculating effluent of treated and partially treated wastewater. This new partially aerated BAF system may be added as a treatment process after the primary clarifier. Studies have shown that these kinds of modified BAF systems are relatively compact, have a relatively low capital cost investment, easy operation and are more efficient than activated sludge systems (Chui et al., 1996; Rogalla et al., 1990; Tay et al., 2003).

However, not much work has been done in the application of BAF with oxic and anoxic or anaerobic conditions for effective nitrogen and phosphorus removal. BAF modified by creating anoxic and oxic zones for the removal of nitrogen or creating various anaerobic, anoxic and oxic zones for removal of nitrogen and phosphorus needs further investigation for complete nitrogen and phosphorus removal (Boller et al., 1994).

Unlike other conventional nutrient removal systems where large space and high hydraulic retention time (HRT) are required, the modified BAF by adding alternating anaerobic and aerobic columns may achieve COD removal and ammonia oxidation as well as denitrification, phosphorus and suspended solid removal with a small footprint and short HRT due to high active biomass (Gonçalves and Rogalla, 1992; Chui et al., 1996; Rogalla et al., 1990; Tay et al., 2003). Chapter 5 showed that the partially aerated filter can be used as a treatment process for nitrogen and sCOD removal to meet discharge limits. In this chapter, the major objective of this study is to investigate phosphorus removal by adding attached growth alternating aerobic and anaerobic filters for phosphorus removal.

MATERIALS AND METHODS

Two attached growth columns (diameter = 75 mm) were added after the partially aerated BAF for the phosphorus removal (see Figure 1). The media used for these two

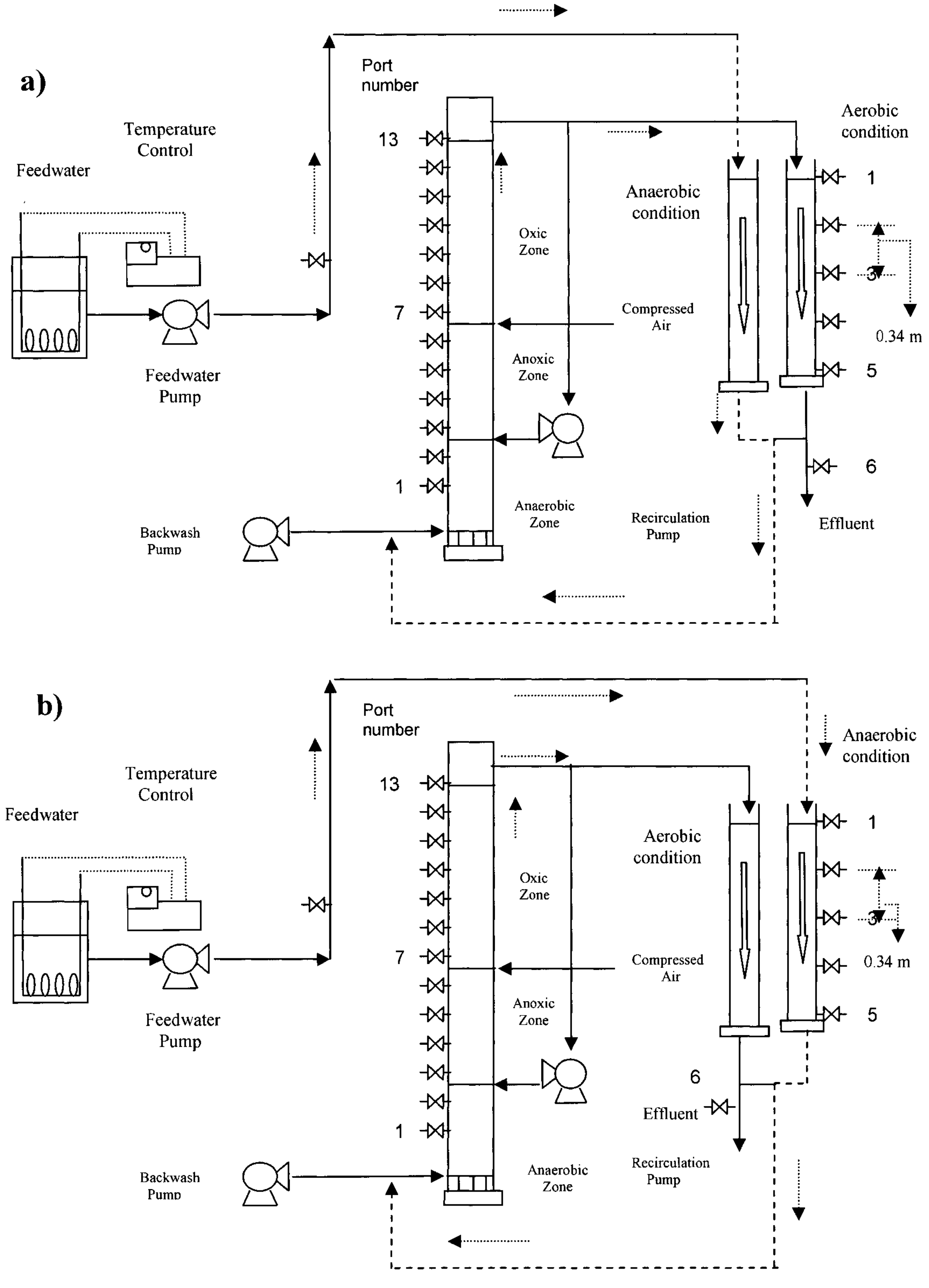

Figure 1. Flow diagram of partially aerated BAF system with oxic, anoxic and anaerobic zones and (a) 1st (anaerobic condition) and 2nd (aerobic condition) and (b) 1st (aerobic condition) and 2nd (anaerobic condition) alternating attached growth columns.

columns was gravel with a mean diameter of 5 mm. The operation of the combined system is illustrated in the flow diagrams in Figure 1. Under typical operating conditions, one column was operated under oxic conditions for the phophorus removal while the other column was operated under anaerobic conditions to simulate PHB accumulation in the microbial cells. The alternating attached growth reactors were operated at a hydraulic retention time of 6 hours with one day cycle of aerobic and anaerobic conditions to stimulate phosphorus release and uptake without interruption. As shown in Figure 1(a), the raw influent sCOD of 500 mg/L and 8 mg P/L was pumped to the column under anaerobic conditions where PAOs accumulate PHBs in the cell. The effluent from anaerobic column was fed into the partially aerated BAF for nitrogen and carbonaceous removals. The effluent from the partially aerated BAF was then sent to the second column where it was aerated to remove phosphorus from the wastewater.

The system was operated over a length of time until the phosphorus removal in the second column tapered off. At this point, the second column received the influent wastewater, reaching anaerobic condition, while the first column aerated to receive the effluent from the partially aerated BAF. The two columns were alternated from anaerobic to aerobic to accumulate phosphorus. Phosphorus was removed from the system by backwashing the column and removing the phosphorus rich biomass. The system was operated over a period of 6 months before PAOs were enriched within the two columns.

The alternating filters were seeded using activated sludge from the aeration tank of a municipal wastewater treatment plant in Iowa. To assess the effectiveness of BAF under varying conditions, a synthetic wastewater was used as feed wastewater as shown in Table 1.

Measurement throughout the study included ortho-phosphates, ammonia, nitrate and nitrite concentrations. Samples used for the analyses were one-hour composite samples. Analyses were conducted according to Standard Methods (APHA, 2002).

Table 1. Composition of synthetic wastewater

Ingredient	Concentration (mg/L)
Calcium Chloride ($CaCl_2$)	40
Magnesium Sulfate ($MgSO_4$)	6
Ferric Chloride ($FeCl_3$)	4
Sodium Biphosphate (NaH_2PO_4)	31
Sodium Bicarbonate ($NaHCO_3$)	100*
Potassium Chloride (KCl)	8
Sodium Acetate (CH_3COONa)	564 (COD=440)
Nutrient Broth	30 (COD=20)
Ammonium Chloride (NH_4Cl)	95
Isomil	(COD=40)**

* Alkalinity is reported in mg/L of total alkalinity as calcium carbonate.

** 500 mL solution in 400 L of synthetic wastewater. Stock solution consists of 44 ml liquid Isomil in 500 mL of distilled water.

RESULTS AND DISCUSSION

Phosporus removal in the alternating columns

Figure 2 shows the effluent phosphate concentrations in both columns over a 48 hours cycle. For the aerobic column, the incoming phosphorus concentration of 5.8 mg P/L was removed rapidly to less than 1 mg P/L until approximately 6 hours of run time. The phosphorus concentration in the effluent then started to increase slowly to about 2.1 mg P/L in 45 hours. For the anaerobic column, phosphorus was released reaching a concentration

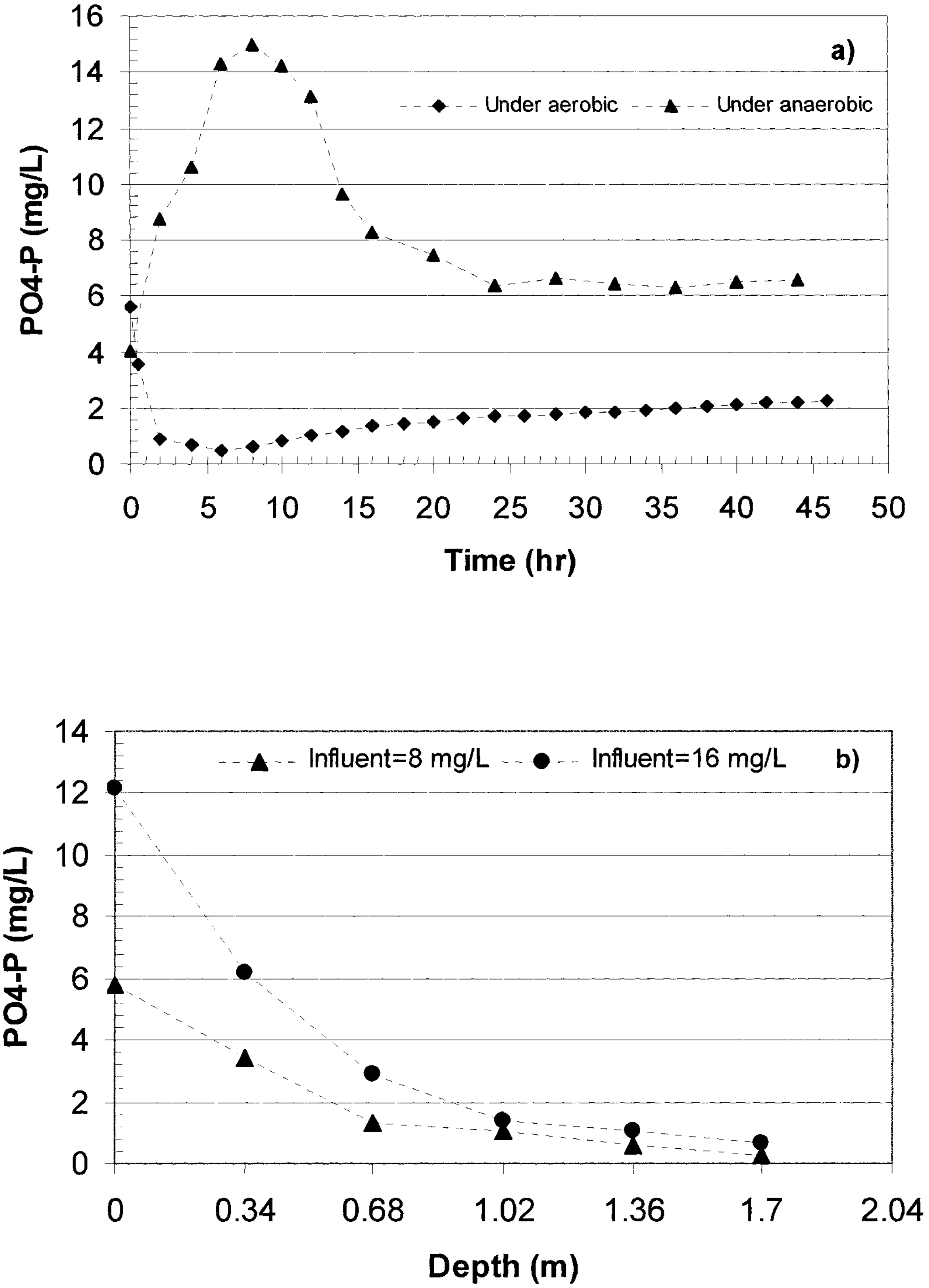

Figure 2. Phosphate changes at HRT = 6 hours in alternative attached growth reactor systems: (a) under aerobic and anaerobic conditions with time and (b) under aerobic condition with media depth.

of 15 mg P/L up to 8 hours of run time. The phosphorus concentration then decreased accordingly to about 6.2 mg P/L (see Figure 2a). Since flows through the aerobic and anaerobic columns were equal, the results showed that there was a net accumulation of phosphorus within the aerobic column by the biofilm. To further demonstrate phosphorus removal, the phosphorus concentrations within aerobic column were measured at different positions within the column. As shown in Figure 2b, more than 97% and 96% phosphorus in the aerobic column was removed with influent phosphorus concentrations were 8 mg P/L and 16 mg P/L, respectively. Figure 2b shows that with a HRT of 6 hours for the aerobic column, the influent phosphorus concentration of 16 mg P/L were equally removed as compared to the lower concentration of 8 mg P/L. Typical phosphorus concentration in municipal wastewater varies from 8 – 12 mg P/L. The results of the experiments was in line with the studies performed by Gonçalves and Rogalla (1992) investigating the feasibility of biofiltration technology for phosphorus removal. They found that high phosphorus uptake (93%) was achieved by PAOs at the short hydraulic retention time of 1 hour in an upflow granular aerated filter. Their upflow granular filter using floating media consists of two reactors in series but with alternating anaerobic and aerobic conditions. Their results showed that nitrification was complete with conversion of 50 mg N/L of ammonia to below 1 mg N/L and phosphorus was reduced from 14.3 mg P/L to below 1 mg P/L while denitrification efficiency (less than 20%) was poor in both filters.

CONCLUSIONS

A new continuous biological phosphorus removal system by putting two filters in series and alternating conditions from anaerobic to aerobic and vice versa was found to maintain high phosphorus removal. The experimental results showed that as high as 96% phosphorus removal in the aerobic column can be removed when different influent phosphorus concentrations (8 mg P/L and 16 mg P/L) were provided to the BAF systems. The effluent phosphorus concentrations in aerobic column were less than 2 mg P/L for as long as 35 hours.

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CHAPTER 7. GENERAL CONCLUSIONS AND FUTURE STUDIES

Three down-flow pilot-scale BAFs made of PVC pipe with a diameter of 75 mm were constructed and operated at temperature of 13 °C. Gravel, lava lock and plastic ring material were used separately as packing materials for each BAF. Experimental results for three BAFs using different media showed that BAF may be used as an add-on system to aerated lagoons or as a secondary treatment to meet ammonia discharge limits. The experimental results showed that on the basis of ammonia removal efficiency as well as overall effluent quality (SCOD, alkalinity, ammonia, nitrate and TKN), gravel was the better media for supporting nitrification at HRTs as low as 0.5 hour with recirculation. A comparison of C/N ratio (C/N=0.2 to C/N=8) indicated that the C/N ratio had an impact on ammonia removal at 13 °C. With an increase in C/N ratio, the ammonia removal (%) decreased. In addition, the experimental results showed that with recirculation at 1-hour HRT and 13 ± 1 °C, ammonia removals were improved. A probable optimum operating condition for the BAF with sand media is at 1 hour HRT with 100 – 200% recirculation at 13 °C.

The effects of different temperatures on nitrification were investigated. With recirculation, the ammonia removal efficiency was improved. At low C/N ratio (under the ratio of C/N = 2) with 1-hour HRT, ammonia removal (%) at 6.5 °C was 54%. However, C/N ratio has a negative effect with less than 5% ammonia removal for C/N ratio of 8. The maximum ammonia loading at 6.5 °C was 0.36 kg NH_3-N/m^3-day while the loading at 24 °C was 0.63 kg NH_3-N/m^3-day. Based on zero-order kinetic model, the biotransformation rates at HRT of 1 hour for temperatures of 6.5°, 13° and 24 °C were 10.38 ± 2.71, 16.97 ± 3.23 and 33.51 ± 3.05 mg NH_3-N/L-hr, respectively. The experimental results indicated that gravel BAF at HRT of 1 hour with recirculation have significant potential as an add-on technology in meeting NPDES nitrogen limits for cold weather conditions. Without recirculation, the experimental results demonstrated that the BAF could be operated at an HRT of 2 hours to ensure nitrification at 6.5 °C.

Nitrogen removal was demonstrated using a 104 mm (4-inch) diameter pilot-scale biological aerated filter (BAF) with a media depth of 2.5 m (8.3 feet) and operated with an anaerobic, anoxic and oxic zone at temperature of 23 °C. The experimental results demonstrated that the BAF can be operated at an HRT of 3 hours with 200 - 300% recirculation rates with more than 96% removal of sCOD and ammonia and at least 75% removal of total nitrogen. The experimental results showed that nitrate and total nitrogen removal efficiencies increased as recirculation rates increased. Using zero-order kinetics for nitrification and half-order kinetics for denitrification, the kinetic values on nitrification and denitrification indicated that as recirculation rates increased, the rates of ammonia and nitrate transformation increased. Of the three recirculation rates, the 300% recirculation with an HRT of 3 hours showed the best mass removal of nitrate and TN. This study demonstrates that a partially aerated BAF can be operated at a low HRT as a compact system that can be used for small communities in treatment of their wastewater for nitrogen and COD removal.

Alternating anaerobic and aerobic columns were added after the partially aerated BAF for the phosphorus removal. The alternating attached growth reactors were operated at HRT of 6 hours. By alternating the conditions from anaerobic to aerobic and vice versa, more than 97% and 96% phosphorus was removed when the influent phosphorus concentrations were 8 mg P/L and 16 mg P/L. The results also showed that the effluent phosphorus concentration from the aerobic filter was less than 2.0 mg P/L for as long as 35 hours.

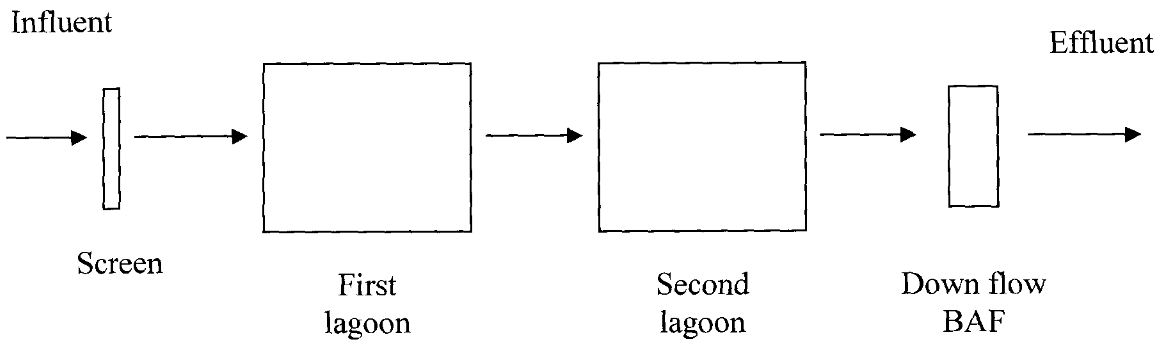
Recommended Future Work

Based on the results of this study, the following are recommended for future research.

1. Develop a more advanced mathematical model to understand the complicated reaction phenomena in the reactor system to estimate the nitrification, denitrification and phosphorus removals.
2. Study the effect of COD:N:P ratio to find optimum C/N/P ratios for the effective nutrient removal.
3. Research the temperature effect on the performance of enhanced biological phosphorus removal (EPBR).
4. Further investigate the impact of the volume ratios of oxic, anoxic and anaerobic zones for effective nitrogen and phosphorus removal
5. Research the characteristic and fates of soluble microbial products (SMP) in BAF systems at various operating parameters.
6 Investigate the optimum oxygen concentration, HRT and loading rates for simultaneous nitrification, denitrification and phosphorus removal in the BAF systems.
7. Apply molecular microbiology technique (FISH or PCR) to understand and identify microbial activity and distribution of hetrotrophs and autotrophs in the reactor systems.

APPENDIX A. ENGINEERING APPLICATION OF BAFS

BAF after second aerated lagoon



Engineering Design

The purpose of the engineering design is to provide a process design of a BAF system for the nitrification of effluent wastewater from aerated lagoons. The design will be based on winter month conditions - a worst case scenario. The flow rate and wastewater concentrations after the second aerated lagoon were assumed as follows:

Flow rate = 1 mgd = 3785 m^3/day

<u>Influent</u> BOD_5 = 25 mg/L

TSS = 50 mg/L

TKN = 30 mg/L

The BAF design basis are as follows:

Temperature = 6.5 °C

<u>Effluent</u> BOD $\leq$ 5 mg/L

TSS $\leq$10 mg/L

NH_3-N $\leq$ 1.5 mg/L

Based on the research work conducted, the following information was obtained for the design of a BAF as a polishing unit for an aerated lagoon. Without recirculation at 13 °C, ammonia removals at 1 hour and 2 hours HRT were 84% and 98%, respectively. With 100% and 200% recirculation at HRT of 1 hr, ammonia removal improved accordingly to 90% and 96%, respectively.

At 6.5 °C, ammonia removals were 95% and 55% for 2 hours and 1 hour HRT, respectively, without recirculation (see Figure 5 of page 101). With 200% at 1 hour HRT, ammonia removal was about 92% at 6.5 °C.

To consistently achieve more than 90% ammonia removal , a HRT of at least 1 hour with 200% recirculation is needed at 6.5 °C. This information was used as the basis of the design.

At 1 hour HRT without recirculation, the zero-order kinetic rate for nitrification was approximately 17 mg NH_3-N/L-hr at 13 °C and 10.4 mg NH_3-N/L-hr at 6.5 °C (see Table 2, page 96 of dissertation).

1. For a HRT of 1 hour, the hydraulic loading rate (HLR) based on the 75 mm pilot-scale plant was 1.7 m/hr (see page 84 of dissertation). The surface area is given as:

   $A = [3785\ m^3/day] * [1day/24hr] / [1.7\ m/hr]$

   $A = 92.8\ m^2$

   Assume 2 square BAFs, the length and width of the BAF were estimated to be $(92.8/2)^{0.5} = 6.8$ m. Select 7m,

   Total area = $7 * 7 * 2 = 98\ m^2$

   New HLR = $3785/98*24 = 1.6$ m/hr

2. The bed depth for ammonia removal was estimated using the zero-order kinetic rate of 10.4 mg NH_3-N/L-hr at 6.5 °C or 0.25 kg NH_3-N/m^3-day and an effluent ammonia concentration of 1.5 mg/L.

   Using $C - Co = - kt$, If $C = 1.5$ mg/L, then with 100% recirculation, Co is given as

   $$[(1)(1.5) + (1)(30)]/2 = 15.75\ mg/L$$

   $$t = - (1.5 - 15.75)/10.4 = 1.37\ hrs$$

   $$\text{Media depth} = 1.37 * 1.6\ m/hr$$

   $$= 2.2\ m$$

With 200% recirculation, an effluent ammonia concentration of less than 1.5 mg/L can be obtained at 6.5 °C. Obviously, at higher temperatures, better ammonia removal will be obtained and recirculation can be decreased accordingly.

Total height of BAF = 2.2 m + 0.7 m + 0.4 m + 0.3 m = 3.6 m

where 0.7 m is equal to water level above the media (based on headloss research, see page 84 of dissertation), 0.4 m is for freeboard and 0.3 m for the gravel support.

3. Daily sludge production

$Y_{obs} = 0.4$ kg VSS / kg $BOD_{5,removed}$ (Rittmann and McCarty, 2001)

$BOD_{5,removed}$ (kg/day) = [3785 m^3/day] * [25 – 5]mg/L * [$10^3/10^6$]

= 75.7 kg/day

$VSS_{produced} = 75.7 * 0.4 = 30.28$ kg/day

Daily Total Solids Production (kg/day)

= 0.3 * [75.7 kg/day] / 0.6 where (VSS / SS = 0.6)

= 50.47 kg/day

4. Air flow rate

- Oxygen demand for BOD removal

$BOD_{L,removed}$ (kg/day) = [3785 m^3/day] * [25 – 5]mg/L * [$10^3/10^6$]/0.6

= 126 kg/day where $BOD_5/BOD_L = 0.6$

For carbonaceous oxygen demand

Since the BAF is backwashed everyday, VSS_{wasted} is assumed to be equal to $VSS_{produced}$.

[O_2 as kg/day] = [$BOD_{L,removed}$] – 1.42 [VSS_{wasted}]

= 126 kg/day – 1.42 [30.28 kg VSS/day]

= 83 kg O_2/day

- Oxygen demand for nitrification

$[O_2 \text{ as kg/day}] = 4.57\ [No - Ne]$

$No = [\text{Influent TKN}] - [\text{Nitrogen used in cell synthesis}] = 113.55 \text{ kg/day} - 3.8 \text{ kg/day}$

$= 109.8 \text{ kg/day}$

where nitrogen used for Cell synthesis is given by (BOD:N = 100:5) = BOD* [5/100]

$= 75.7 \text{ kg/day} * 0.05 = 3.8 \text{ kg/day}$

$Ne = [\text{Average effluent } NH_3\text{-N}] = 1.5 \text{ mg/L} * [3785 \text{ m}^3\text{/day}] * [10^3/10^6]$

$= 5.7 \text{ kg/day}$

$[O_2] = 4.57\ [109.8 \text{ kg/day} - 5.7 \text{ kg/day}] = 475.7 \text{ kg/day}$

$[O_2]_T = 83 + 475.7 = 558.7 \text{ kg } O_2\text{/day}$

Using

Air density = 1.2 kg /m^3 (Metcalf and Eddy, 2003)

Oxygen transfer rate = 8%

Oxygen in air = 21%

Total volume of air needed

$= 558.7 \text{ kg/day} / 1.2 \text{ kg/m}^3 * \text{air}/0.21\ O_2 * 1/0.08 * \text{day}/24\text{hr}$

$= 1155 \text{ m}^3\text{/hr}$

5. Backwash flowrate

A backwash rate of LV = 15 m/hr was used in the study for a duration of 10 minutes every days.

The amount of water for backwash = $15 \text{ m/hr} * 10 \text{ min} * 7*7*2 \text{ m}^2 * 1 \text{ hr} / 60 \text{ min}$

$= 245 \text{ m}^3$

This is equivalent to (245/3785 * 100) or 6.5% of the total wastewater treated.

6. Summary for BAF design with 100% Qr at 13 °C and HRT of 1hr.

Number of BAFs:	2 units
Size:	W 7m x L 7m x H 3.6m
Surface area:	98 m^2
Bed depth:	2.2 m
Bed volume:	216 m^3
HRT:	1.37 hr
HLR:	1.6 m/hr
Qr:	100 - 200%

References

Rittmann, B. E. and McCarty, P. L. (2001). Environmental Biotechnology. McGraw-Hill Company, Inc., New York, USA.

Metcalf and Eddy. (2003).Wastewater Engineering, Treatment and Reuse, 4th edition. McGraw Hill Inc., New York.

APPENDIX B. EXPERIMENTAL DATA IN CHAPTER 3

Impact of Various Operating Parameters on Nitrification in Polishing Biological Aerated Filters (BAFs)

Sand BAF performance

Day	Influent COD (mg/L)	Effluent COD (mg/L)	Influent NH_3-N (mg/L)	Effluent NH_3-N (mg/L)	Removal (%) NH_3-N	Removal (%) COD
1	52.6	14.29	25.1	11.4	54.58	72.83
4	51.3	6	25.4	1.8	92.91	88.30
7	51.6	0.54	25.1	0.2	99.20	98.95
10	50.4	0.1	24.8	0	100	99.80
13	50.6	5.1	25.1	0.2	99.20	89.92
16	52.2	1.8	25.2	0.7	97.22	96.55
19	51.8	0.1	25.4	0.2	99.21	99.80
22	50.6	1.8	25.8	0	100	96.44
25	52.4	3.1	25.6	0.1	99.60	94.08
28	51.2	3	25.6	0.4	98.43	94.14
31	52	3	25.2	3.7	85.31	94.23
34	50.4	3.5	25.1	1	96.01	93.05
37	49.3	5.5	25.8	0.1	99.61	88.84
40	52.2	3.5	26	0.8	96.92	93.29
43	51.5	3.1	25.9	0.4	98.45	93.98
46	52.7	1.2	24.6	0.1	99.59	97.72
49	53.6	3.5	25.1	0.1	99.60	93.47
52	50.6	1.2	25.4	0.2	99.21	97.62
55	53.4	8.1	26.2	10.8	58.78	84.84
58	52.3	4.8	25.9	9.1	64.86	90.81
61	51.9	5.2	24.8	2.9	88.33	90
64	52.7	5.2	24.6	7.5	69.54	90.15
67	51.5	4	25.5	4.4	82.71	92.27
70	52	5.6	24.8	2	91.92	89.22
73	51	4.8	24.5	4.5	81.65	90.62
76	52	7.3	25.1	3.7	85.36	86
79	49.6	6.8	25.6	4.2	83.62	86.36
82	51	6.5	26	20.8	20	87.31
85	56	5.6	25.2	20.3	19.43	90
88	51.9	9.4	25.6	16.9	34	81.92
91	54	6.9	25.5	16.1	36.95	87.24
94	50.6	8.5	25.6	15.9	37.92	83.27
97	53.3	8.3	24.4	14.2	41.84	84.41
100	50.2	6.3	24.6	14.1	42.71	87.52
103	47.9	6.3	25.4	16.8	33.93	86.81
106	49.4	6.7	24.9	15.2	39	86.47
109	49.8	4.8	25.4	0.65	97.44	90.36

112	51	7.1	24.8	2.7	89.11	86.07
115	48.3	4	25.8	2.8	89.14	91.71
118	50.6	2.5	25.2	2.6	89.68	95.05
121	51.3	3.7	24.6	1.9	92.27	92.78
124	49.4	2.1	25.6	2.7	89.45	95.74
127	51.8	3.9	25	1.9	92.41	92.47
130	51	4.6	25.2	0.9	96.42	90.98
133	48	3.3	25.6	0.4	98.43	93.12
136	52.7	1.9	25	1.1	95.63	96.39
139	50.8	4.9	24.8	1	95.96	90.35
142	49.4	3.4	25.2	1.1	95.63	93.11
145	51.5	2.9	25.6	0.8	96.87	94.36
148	50.4	2.9	25	1	96	94.24

Lava rock reactor performance

Day	Influent COD (mg/L)	Effluent COD (mg/L)	Influent NH_3-N (mg/L)	Effluent NH_3-N (mg/L)	Removal (%) NH_3-N	Removal (%) COD
1	52.6	2.2	25.1	2.8	88.84	95.81
4	51.3	3.9	25.4	0.1	99.60	92.39
7	51.6	0.1	25.1	0	100	99.80
10	50.4	0.2	24.8	0.9	96.37	99.60
13	50.6	0.1	25.1	0.3	98.80	99.80
16	52.2	0.5	25.2	0.1	99.60	99.04
19	51.8	1	25.4	0.1	99.60	98.06
22	50.6	3.5	25.8	0	100	93.08
25	52.4	0.1	25.6	0.1	99.60	99.80
28	51.2	3.5	25.6	6.5	74.60	93.16
31	52	0.7	25.2	0.1	99.60	98.65
34	50.4	1.4	25.1	0.1	99.60	97.22
37	49.3	2.6	25.8	0	100	94.72
40	52.2	1.8	26	0.3	98.84	96.55
43	51.5	0.2	25.9	1.3	94.98	99.61
46	52.7	1.4	24.6	0	100	97.34
49	53.6	2.7	25.1	0.1	99.60	94.96
52	50.6	2.4	25.4	0.1	99.60	95.25
55	53.4	0.7	26.2	11.2	57.32	98.72
58	52.3	7.3	25.9	8	69.15	86
61	51.9	6	24.8	4.1	83.52	88.47
64	52.7	4.8	24.6	8.9	63.84	90.94
67	51.5	4.8	25.5	10.2	60	90.75
70	52	6.5	24.8	10.4	58.12	87.52

73	51	8.5	24.5	9.7	60.48	83.34
76	52	7.3	25.1	8.8	65	86
79	49.6	7	25.6	8.6	66.43	85.97
82	51	8.5	26	20.6	20.84	83.33
85	56	6	25.2	19.6	22.21	89.34
88	51.9	11	25.6	19.5	23.83	78.82
91	54	9.8	25.5	19.6	23.17	81.91
94	50.6	10.6	25.6	19.7	23	79.14
97	53.3	8.7	24.4	19	22	83.76
100	50.2	10.6	24.6	19.3	21.54	78.93
103	47.9	9.4	25.4	20.7	18.55	80.42
106	49.4	5.6	24.9	18.3	26.53	88.72
109	49.8	5.6	25.4	6.5	74.40	88.75
112	51	8.7	24.8	4.9	80.24	82.94
115	48.3	6	25.8	6.4	75.19	87.57
118	50.6	4	25.2	7	72.22	92.09
121	51.3	4.4	24.6	5.5	77.64	91.42
124	49.4	2.5	25.6	5.3	79.29	94.93
127	51.8	4.6	25	6.1	75.63	91.11
130	51	4	25.2	3.3	86.90	92.15
133	48	1.7	25.6	2.7	89.45	96.45
136	52.7	2.9	25	2.9	88.43	94.49
139	50.8	3.8	24.8	2.5	89.91	92.51
142	49.4	4.9	25.2	2.6	89.68	90.08
145	51.5	3.4	25.6	3.4	86.71	93.39
148	50.4	3.9	25	2.7	89.24	92.26

Plastic ring reactor performance

Day	Influent COD (mg/L)	Effluent COD (mg/L)	Influent NH_3-N (mg/L)	Effluent NH_3-N (mg/L)	Removal (%) NH_3-N	Removal (%) COD
1	52.6	3	25.1	6.9	72.51	94.29
4	51.3	9.9	25.4	5.5	78.34	80.70
7	51.6	4.7	25.1	0.8	96.81	90.89
10	50.4	4.7	24.8	0.3	98.79	90.67
13	50.6	0.54	25.1	0.5	98.01	98.93
16	52.2	0.1	25.2	0.1	99.60	99.80
19	51.8	3.1	25.4	0.2	99.21	94.01
22	50.6	0.1	25.8	0.1	99.61	99.80
25	52.4	3.5	25.6	0.1	99.61	93.32
28	51.2	1.4	25.6	5.3	79.29	97.26

31	52	8.9	25.2	0.3	98.81	82.88
34	50.4	4.3	25.1	2.1	91.63	91.46
37	49.3	0.1	25.8	0.1	99.61	99.79
40	52.2	2.4	26	0.7	97.31	95.40
43	51.5	6	25.9	0.7	97.29	88.34
46	52.7	5.5	24.6	0.1	99.59	89.56
49	53.6	6.5	25.1	0.1	99.60	87.87
52	50.6	2	25.4	0.1	99.61	96.04
55	53.4	9	26.2	12.2	53.42	83.12
58	52.3	6	25.9	13.3	48.65	88.53
61	51.9	6.5	24.8	9.6	61.37	87.57
64	52.7	7.3	24.6	12.4	49.62	86.11
67	51.5	8.1	25.5	10.3	59.63	84.32
70	52	5.8	24.8	9.8	60.51	88.88
73	51	6.5	24.5	10.1	58.83	87.33
76	52	7.6	25.1	10.1	59.85	85.44
79	49.6	8	25.6	9.8	61.72	83.92
82	51	16.9	26	23.4	10	66.91
85	56	11.5	25.2	24.1	4.42	79.54
88	51.9	11.5	25.6	21.3	16.87	77.43
91	54	9	25.5	21.6	15.34	83.37
94	50.6	6.9	25.6	21	18	86.43
97	53.3	13.7	24.4	20	19.75	74.31
100	50.2	12.5	24.6	20.5	16.78	75.12
103	47.9	6.7	25.4	22.5	11.42	86
106	49.4	7.9	24.9	21	15.73	84
109	49.8	6.3	25.4	10.6	58.26	87.34
112	51	8.7	24.8	9.1	63.31	82.94
115	48.3	4.8	25.8	8.6	66.66	90.06
118	50.6	6.3	25.2	8.6	65.87	87.54
121	51.3	5.6	24.6	8.5	65.44	89.08
124	49.4	5.2	25.6	8.2	67.96	89.47
127	51.8	6.8	25	7.9	68.42	86.87
130	51	6.3	25.2	7.4	70.63	87.64
133	48	4.6	25.6	4.4	82.81	90.41
136	52.7	4.2	25	5.7	77.24	92.03
139	50.8	5.4	24.8	5.1	79.43	89.37
142	49.4	2.9	25.2	4.5	82.14	94.12
145	51.5	4.4	25.6	5.3	79.29	91.45
148	50.4	4.9	25	5	80	90.27

Effects of media types and HRTs

HRT=2hr, Temp=13 °C

	NH_3-N (%)	COD (%)	ALK (%)	TKN (%)	Standard deviation ±			
Sand	98.5	93.9	83.3	97.44	1.35	2.81	1.33	4.87
Lava rock	98.9	96.5	84.8	97.12	1.65	1.6	2.79	4.81
Plastic ring	97.8	92.6	84.3	96.47	2.71	4.18	2.64	6.04

HRT=1hr, Temp=13 °C

	NH_3-N (%)	COD (%)	ALK (%)	TKN (%)	Standard deviation ±			
Sand	83.3	89.2	75.1	84.47	6.51	2.1	7.74	5.63
Lava rock	65.3	87.5	58.4	69.58	7.92	2.53	9.54	5.29
Plastic ring	58.8	86.2	52	63.43	3.85	1.66	5.22	5.59

HRT=0.5hr, Temp=13 °C

	NH_3-N (%)	COD (%)	ALK (%)	TKN (%)	Standard deviation ±			
Sand	38	85.3	36.5	43.79	3.19	2.02	2.75	3.13
Lava rock	25.5	81.6	24.4	36.27	7.17	3.33	2.78	6.17
Plastic ring	16.2	80.9	14.3	28.43	2.39	4.79	4.25	3.58

Effects of recirculation

Ammonia removal (Influent NH_3-N = 25 mg N/L)

	Without Qr NH_3-N (%)	With 100% Qr NH_3-N (%)	With 200% Qr NH_3-N (%)	Standard deviation ±		
Sand	83.3	90.1	96	6.51	1.4	0.5
Lava rock	65.3	76.5	90	7.92	3.2	0.4
Plastic ring	58.8	65.3	79.5	3.85	1.2	1.8

COD removal (Influent COD = 50 mg/L)

	Without Qr COD (%)	With 100% Qr COD (%)	With 200% Qr COD (%)	Standard deviation ±		
Sand	89.2	92.6	93.6	2.1	4.3	2.2
Lava rock	87.5	88.5	92.6	2.5	3.6	1.6
Plastic ring	86.2	87.4	91.7	1.7	2.7	1.7

Ammonia and nitrate profiles

HRT =0.5 hr

Depth (m)	**Sand**	**NH_3-N (mg/L)**	**NO^-_3-N (mg/L)**	**NH_3-N ±**	**NO^-_3-N ±**
0	1	24.26	0.60	0.19	0.50
0.34	2	23.70	1.15	0.13	0.25
0.68	3	22.93	1.70	0.21	0.01
1.02	4	22.15	3.00	0.19	0.20
1.36	5	19.35	5.90	0.28	0.20
1.7	6	15.15	8.95	0.65	0.15

Depth (m)	**Lava rock**	**NH_3-N (mg/L)**	**NO^-_3-N (mg/L)**	**NH_3-N ±**	**NO^-_3-N ±**
0	1	24.58	0.85	0.20	0.15
0.34	2	24.26	1.00	0.19	0.10
0.68	3	23.63	1.25	0.18	0.05
1.02	4	22.41	2.35	0.16	0.25
1.36	5	20.42	5.00	0.13	0.10
1.7	6	18.52	7.15	0.09	0.25

Depth (m)	**Plastic ring**	**NH_3-N** (mg/L)	**NO^-_3-N** (mg/L)	**NH_3-N** ±	**NO^-_3-N** ±
0	1	23.63	1.10	0.18	0.10
0.34	2	23.40	1.25	0.18	0.15
0.68	3	23.01	1.80	0.17	0.20
1.02	4	21.47	3.50	0.14	0.10
1.36	5	20.98	4.15	0.13	0.05
1.7	6	19.68	6.10	0.11	0.10

HRT = 1 hour

Depth	**Sand**	**NH_3-N** (mg/L)	**NO_3^--N** (mg/L)	**NH_3-N** ±	**NO_3^--N** ±
0	1	25.86	1.00	0.09	0.10
0.34	2	25.32	1.45	0.12	0.15
0.68	3	21.82	4.55	0.31	0.25
1.02	4	17.22	8.70	0.29	0.10
1.36	5	11.58	14.90	0.27	0
1.7	6	3.97	20.00	0.25	0.80

Depth	**Lava rock**	**NH_3-N** (mg/L)	**NO_3^--N** (mg/L)	**NH_3-N** ±	**NO_3^--N** ±
0	1	23.68	0.95	0.96	0.05
0.34	2	23.02	1.45	1.16	0.25
0.68	3	22.37	2.70	1.14	0.10
1.02	4	17.45	7.55	0.47	0.05
1.36	5	11.14	14.10	1.21	0.20
1.7	6	8.70	16.55	0.10	0.35

Depth	**Plastic ring**	**NH_3-N** (mg/L)	**NO_3^--N** (mg/L)	**NH_3-N** ±	**NO_3^--N** ±
0	1	20.55	3.00	0.06	0.20
0.34	2	19.04	4.80	0.03	0.10
0.68	3	16.53	7.90	0.05	1.00
1.02	4	13.64	10.70	0.11	0.10
1.36	5	12.15	12.80	0.07	0.80
1.7	6	10.07	15.75	0.21	0.45

HRT = 2hrs

Depth	Sand	NH_3-N (mg/L)	NO_3^--N (mg/L)	NH_3-N ±	NO_3^--N ±
0	1	22.63	2.47	0.87	0.73
0.34	2	18.40	5.67	0	0.43
0.68	3	10.40	10.87	0.35	0.63
1.02	4	4.90	16.63	0	1.07
1.36	5	0.30	22.83	0	0.37
1.7	6	0	23.90	0	0.70

Lava rock

Depth	Lava rock	NH_3-N (mg/L)	NO_3^--N (mg/L)	NH_3-N ±	NO_3^--N ±
0	1	24.6	1.50	0.60	0.15
0.34	2	19.9	5.10	0	0.50
0.68	3	11.8	10.50	0.50	0.40
1.02	4	4.80	15.77	0	0.83
1.36	5	0.60	21.13	0	0.37
1.7	6	0.50	23.17	0	0.48

Plastic ring

Depth	Plastic ring	NH_3-N (mg/L)	NO_3^--N (mg/L)	NH_3-N ±	NO_3^--N ±
0	1	12.8	11.80	0	0.40
0.34	2	8.20	15.37	0.20	0.68
0.68	3	4.80	18.47	0	0.53
1.02	4	1.77	19.57	0.08	0.58
1.36	5	0.70	22.43	0	0.72
1.7	6	0.60	24.33	0	0.37

Effects of C/N ratio

COD/NH_3-N	**NH3-N removal (%)**			**Standard deviation**		
C/N	Sand	Lava rock	Plastic ring	±	±	±
0.2	94.3	78.1	72.1	0.9	1.5	0.4
1	90.1	74.3	65.9	1.5	0.5	1.0
2	83.5	63.9	60.1	1.5	2.6	2.0
4	64.1	53.0	45.5	0.7	1.3	2.3
6	50.1	41.0	33.4	1.8	1.5	4.3
8	30.9	26.1	20.7	1.0	0.6	1.3

COD/NH_3-N	**Effluent COD concentration (mg/L)**					
C/N	Sand	Lava rock	Plastic ring	±	±	±
0.2	0.6	2.0	3.3	0.4	0.3	0.4
1	5.3	5.7	7.0	0.4	0.4	0.3
2	6.3	7.6	8.0	1.1	0.6	0.4
4	9.1	10.3	11.6	0.3	0.9	0.9
6	11.6	12.3	13.6	0.3	0.9	0.6
8	15.8	17.8	20.0	1.0	0.7	0.9

COD/NH_3-N	**Effluent NH3-N concentration (mg/L)**					
C/N	Sand	Lava rock	Plastic ring	±	±	±
0.2	1.4	5.5	7.0	0.2	0.4	0.1
1	2.5	6.5	8.7	0.4	0.2	0.2
2	4.1	9.0	10.0	0.3	0.5	0.1
4	9.1	11.9	13.8	0.2	0.5	0.8
6	12.6	14.9	16.8	0.5	0.4	1.0
8	17.5	18.7	20.1	0.4	0.3	0.5

COD/NH3-N	**NH_3-N kg/m^3-day removed**					
C/N	Sand	Lava rock	Plastic ring	±	±	±
0.2	0.56	0.47	0.43	0.005	0.010	0.002
1	0.54	0.44	0.39	0.010	0.005	0.005
2	0.50	0.38	0.36	0.007	0.012	0.002
4	0.38	0.31	0.27	0.005	0.012	0.019
6	0.30	0.24	0.20	0.012	0.010	0.024
8	0.18	0.15	0.12	0.010	0.007	0.012

Effect of Ammonia and COD Loadings

Applied BOD Loading	**Ammonia Removal (%)**					
kg COD/m^3/day	Sand	Lava rock	Plastic ring	±	±	±
0.6	98.5	98.9	97.8	1.35	1.65	2.71
0.9	94.5	84.8	78.6	2.30	2.30	0.50
1.2	83.3	65.3	58.8	6.51	7.92	3.85
1.5	63.0	42.9	36.3	1.80	1.00	2.10
1.8	49.8	36.1	30.5	0.90	1.40	1.95
2.4	38.0	25.5	16.2	3.19	7.17	2.39

Applied NH_3-N Loading	**NH_3-N removed (kg/m^3-day)**					
kg NH_3-N /m^3/day	Sand	Lava rock	Plastic ring	±	±	±
0.3	0.29	0.30	0.29	0.004	0.005	0.008
0.45	0.42	0.38	0.35	0.011	0.009	0.002
0.6	0.50	0.39	0.35	0.038	0.016	0.021
0.75	0.45	0.32	0.27	0.012	0.009	0.012
0.9	0.44	0.32	0.27	0.007	0.012	0.013
1.2	0.45	0.30	0.20	0.050	0.033	0.034

Applied NH_3-N Loading	**Ammonia Removal (%)**					
kg NH_3-N /m^3/day	Sand	Lava rock	Plastic ring	±	±	±
0.3	98.5	98.9	97.8	1.35	1.65	2.71
0.45	94.5	84.8	78.6	2.30	2.30	0.50
0.6	83.3	65.3	58.8	6.51	7.92	3.85
0.75	63.0	42.9	36.3	1.80	1.00	2.10
0.9	49.8	36.1	30.5	0.90	1.40	1.95
1.2	38.0	25.5	16.2	3.19	7.17	2.39

Applied NH_3-N Loading kg NH_3-N /m^3/day	**Effluent NH_3-N (mg/L)** Sand	Lava rock	Plastic ring	±	±	±
0.3	0.4	0.3	0.6	0.34	0.43	0.68
0.45	1.4	3.8	5.4	0.60	0.50	0.10
0.6	4.2	8.7	10.3	1.59	0.67	0.88
0.75	9.9	14.4	16.0	0.40	0.30	0.40
0.9	12.7	16.2	17.6	0.19	0.33	0.35
1.2	15.6	18.8	20.8	1.05	0.69	0.70

APPENDIX C. EXPERIMENTAL DATA IN CHAPTER 4

Temperature Effects on Nitrification in Polishing Biological Aerated Filters (BAFs)

Sand BAF performance

day	Influent COD (mg/L)	Effluent COD (mg/L)	Influent NH_3-N (mg/L)	Effluent NH_3-N (mg/L)	Removal (%) NH_3-N	Removal (%) COD
1	52.6	14.29	25.1	11.4	54.58	72.83
4	51.3	6	25.4	1.8	92.91	88.30
7	51.6	0.54	25.1	0.2	99.20	98.95
10	50.4	0.1	24.8	0	100	99.80
13	50.6	5.1	25.1	0.2	99.20	89.92
16	52.2	1.8	25.2	0.7	97.22	96.55
19	51.8	0.1	25.4	0.2	99.21	99.80
22	50.6	1.8	25.8	0	100	96.44
25	52.4	3.1	25.6	0.1	99.60	94.08
28	51.2	3	25.6	0.4	98.43	94.14
31	52	3	25.2	3.7	85.31	94.23
34	50.4	3.5	25.1	1	96.01	93.05
37	49.3	5.5	25.8	0.1	99.61	88.84
40	52.2	3.5	26	0.8	96.92	93.29
43	51.5	3.1	25.9	0.4	98.45	93.98
46	52.7	1.2	24.6	0.1	99.59	97.72
49	53.6	3.5	25.1	0.1	99.60	93.47
52	50.6	1.2	25.4	0.2	99.21	97.62
55	53.4	8.1	26.2	10.8	58.78	84.84
58	52.3	4.8	25.9	9.1	64.86	90.81
61	51.9	5.2	24.8	2.9	88.33	90
64	52.7	5.2	24.6	7.5	69.54	90.15
67	51.5	4	25.5	4.4	82.71	92.27
70	52	5.6	24.8	2	91.92	89.22
73	51	4.8	24.5	4.5	81.65	90.62
76	52	7.3	25.1	3.7	85.36	86
79	49.6	6.8	25.6	4.2	83.62	86.36
82	51	6.5	26	20.8	20	87.31
85	56	5.6	25.2	20.3	19.43	90
88	51.9	9.4	25.6	16.9	34	81.92
91	54	6.9	25.5	16.1	36.95	87.24
94	50.6	8.5	25.6	15.9	37.92	83.27
97	53.3	8.3	24.4	14.2	41.84	84.41
100	50.2	6.3	24.6	14.1	42.71	87.52
103	47.9	6.3	25.4	16.8	33.93	86.81
106	49.4	6.7	24.9	15.2	39	86.47
109	49.8	4.8	25.4	0.65	97.44	90.36
112	51	7.1	24.8	2.7	89.11	86.07
115	48.3	4	25.8	2.8	89.14	91.71
118	50.6	2.5	25.2	2.6	89.68	95.05

121	51.3	3.7	24.6	1.9	92.27	92.78
124	49.4	2.1	25.6	2.7	89.45	95.74
127	51.8	3.9	25	1.9	92.41	92.47
130	51	4.6	25.2	0.9	96.42	90.98
133	48	3.3	25.6	0.4	98.43	93.12
136	52.7	1.9	25	1.1	95.63	96.39
139	50.8	4.9	24.8	1	95.96	90.35
142	49.4	3.4	25.2	1.1	95.63	93.11
145	51.5	2.9	25.6	0.8	96.87	94.36
148	50.4	2.9	25	1	96	94.24

Effects of temperature

HRT=2hrs	**Removal (%) NH_3-N**	±	**HRT=1hr**	**Removal (%) NH_3-N**	±	**HRT=0.5hr**	**Removal (%) NH_3-N**	±
24 °C	99.2	0.10	**24 °C**	91.9	1.50	**24 °C**	53	3.60
13 °C	98.5	1.35	**13 °C**	83.3	6.50	**13 °C**	38	3.20
6.5 °C	98.4	0.20	**6.5 °C**	54.3	1.70	**6.5 °C**	29.9	1.80

Headloss changes

HRT=2hrs		**HRT = 1 hr**		**HRT=0.5**	
Time (hr)	(cm)	**Time (hr)**	(cm)	**Time(hr)**	(cm)
0	15.9	**0**	16.0	**0**	16.1
18	16.2	**12**	18.4	**6**	19.8
30	17.2	**24**	21.0	**12**	21.8
50	17.9	**36**	24.5	**18**	26.4
66	19.5	**48**	28.6	**24**	29.5
78	21.7	**54**	30.3	**30**	33.0
90	23.8	**60**	31.5	**36**	35.2
102	24.3	**72**	37.5	**42**	40.2
114	27.5	**78**	46.0	**48**	46.2
126	31.1	**90**	65.6	**51**	55.6
138	34.2			**54**	66.9

Ammonia and nitrate profiles with and without aeration

		Without aeration		With aeration	
		NH_3-N	NO_3^--N	NH_3-N	NO_3^--N
ports	**Depth(m)**	(mg/L)	(mg/L)	(mg/L)	(mg/L)
1	**0**	23.0	6.5	22.6	2.5
2	**0.34**	22.2	6.1	18.4	5.7
3	**0.68**	21.8	4.9	10.4	10.9
4	**1.02**	21.1	5.7	4.9	16.6
5	**1.36**	20.4	4.3	0.3	22.8
6	**1.7**	17.7	3.9	0	23.9

Changes in ammonia, nitrite and nitrate concentrations in the column

Temp =24 °C

(HRT=0.5)

Sand	**Time**	mV	NH_3-N (mg/L)	NO_3^--N (mg/L)	NO_2^--N (mg/L)
0	0	38.9	25.2	2.7	1.2
0.34	0.1	39.3	24.9	4.8	2.7
0.68	0.2	40.2	24.2	6.5	3.0
1.02	0.3	44.5	21.1	9.7	2.7
1.36	0.4	50.9	16.3	11.5	1.2
1.7	0.5	56.6	12.1	13.7	0.9

HRT=1hr

Sand	Time (hr)	mV	NH_3-N (mg/L)	NO_3^--N (mg/L)	NO_2^--N (mg/L)
0	0	34.1	24.7	3.3	1.5
0.34	0.2	44.2	17.3	10.1	4.5
0.68	0.4	60.2	8.5	15.6	2.7
1.02	0.6	78.9	4.5	19.4	1.8
1.36	0.8	119	0.3	22.5	1.2
1.7	1	167.5	0	23.6	0.6

(HRT=2hr)

Sand	**Time**	mV	NH_3-N (mg/L)	NO_3^--N (mg/L)	NO_2^--N (mg/L)
0	0	54.7	25.4	3.8	1.8
0.34	0.4	76.4	9.8	15.6	5.2
0.68	0.8	118.7	0.5	24.0	3.8
1.02	1.2	210	0	23.7	2.2
1.36	1.6	214	0	22.8	0.6
1.7	2	220	0	24.8	0.6

Changes in ammonia, nitrite and nitrate concentrations in the column

Temp =13 °C

(HRT=0.5)		NH_3-N	NO^-_3-N	NO^-_2-N
Sand	**Time**	(mg/L)	(mg/L)	(mg/L)
0	0	24.5	0.6	2.4
0.34	0.1	23.6	1.2	3.9
0.68	0.2	22.7	1.7	5.4
1.02	0.3	22.4	3.0	3.3
1.36	0.4	19.1	5.9	1.2
1.7	0.5	14.5	8.9	0.6

HRT=1hr		NH_3-N	NO^-_3-N	NO^-_2-N
Sand	Time (hr)	(mg/L)	(mg/L)	(mg/L)
0	0	25.0	1.0	1.8
0.34	0.2	25.0	1.4	4.5
0.68	0.4	22.1	4.6	3.6
1.02	0.6	17.5	8.7	3.1
1.36	0.8	11.9	14.9	1.8
1.7	1	4.2	20.0	0.9

HRT=2 hrs		NH_3-N	NO^-_3-N	NO^-_2-N
Sand	**Time**	(mg/L)	(mg/L)	(mg/L)
0	0	24.5	2.5	1.5
0.34	0.4	18.4	5.7	2.1
0.68	0.8	10.7	10.9	2.6
1.02	1.2	4.9	16.6	2.3
1.36	1.6	0.3	22.8	1.5
1.7	2	0	23.9	0.9

Changes in ammonia, nitrite and nitrate concentrations in the column

Temp =6.5 °C

HRT=0.5			**NH_3-N**	**NO^-_3-N**	**NO^-_2-N**
Sand	**Time**	mV	(mg/L)	(mg/L)	(mg/L)
0	0	32.8	25.4	1.5	1.2
0.34	0.1	33.3	25.1	2.6	1.5
0.68	0.2	34.6	24.2	3.2	1.8
1.02	0.3	36.1	23.1	3.9	2.1
1.36	0.4	38.2	21.6	5.8	1.8
1.7	0.5	41.4	19.3	6.9	1.5

HRT=1hr			**NH_3-N**	**NO^-_3-N**	**NO^-_2-N**
Sand	Time (hr)	mV	(mg/L)	(mg/L)	(mg/L)
0	0	39.9	25.7	2.8	1.2
0.34	0.2	40.2	25.5	4.4	2.1
0.68	0.4	41.6	24.5	5.7	1.8
1.02	0.6	45.4	21.8	8.6	1.8
1.36	0.8	54.1	15.6	10.3	1.5
1.7	1	59.5	11.7	11.7	1.2

(HRT=2)			**NH_3-N**	**NO^-_3-N**	**NO^-_2-N**
Sand	**Time**	mV	(mg/L)	(mg/L)	(mg/L)
0	0	38.9	25.2	1.9	1.5
0.34	0.4	45.0	20.7	3.7	2.1
0.68	0.8	51.8	15.7	8.5	1.8
1.02	1.2	62.2	8.0	16.8	1.5
1.36	1.6	101.3	3.0	21.4	1.5
1.7	2	129.0	0.3	23.6	0.9

Effects of recirculation ratios (influent NH_3-N = 25 mg/L, influent COD = 50 mg/L)

At 13 °C	R (%) NH_3-N	R (%) COD	SD ±	SD ±	**At 6.5 °C**	R (%) NH_3-N	R (%) COD	SD ±	SD ±
Without Qr	83.3	89.2	6.5	2.1	**Without Qr**	54.3	87.0	2.7	3.5
100 % Qr	90.1	92.6	1.4	4.3	**100 % Qr**	75.7	88.4	3.8	2.8
200 % Qr	96.0	93.6	0.5	2.2	**200 % Qr**	91.9	88.6	1.5	4.3

Effects of C/N ratios

COD/NH_3-N	**NH_3-N removal (%)**		**COD/NH_3-N**	**Effluent COD (mg/L)**	
C/N	Sand	±	**C/N**	Sand	±
0.2	97.6	0.7	**0.2**	0.4	0.1
1	95.0	1.6	**1**	2.2	0.3
2	91.9	1.5	**2**	5.0	0.8
4	78.4	1.8	**4**	8.4	0.7
6	67.0	1.7	**6**	10.3	0.4
8	49.6	1.6	**8**	13.8	0.7

COD/NH_3-N	**Effluent NH_3-N (mg/L)**			**NH_3-N kg/m^3-day removed**	
C/N	Sand	±	C/N	Sand	±
0.2	0.6	0.2	**0.2**	0.583	0.01
1	1.3	0.4	**1**	0.566	0.01
2	2.0	0.4	**2**	0.549	0.01
4	5.4	0.4	**4**	0.468	0.01
6	8.3	0.4	**6**	0.399	0.01
8	12.2	0.5	**8**	0.306	0.01

Effect of Temperature on mass removed

Loading		NH$_3$-N Removal (%)		SD	SD	SD
kg/m^3-day	24 °C	13 °C	6.5 °C	±	±	±
0.3	99.6	98.5	98.4	0.01	1.4	0.2
0.45	98.2	94.5		0.5	2.3	
0.6	91.9	83.3	54.3	1.5	4.5	2.7
0.75	80.1	63		1.9	1.8	
0.9	70.5	49.8		2.2	0.9	
1.2	53	38	29.9	3.6	3.2	1.8

Loading		Effluent NH$_3$-N Concentration (mg/L)				
kg/m^3-day	24 °C	13 °C	6.5 °C	±	±	±
0.3	0.1	0.4	0.5	0	0.3	0.1
0.45	0.4	1.4		0.1	0.6	
0.6	2.0	4.2	11.5	0.4	0.8	0.2
0.75	5.0	9.9		0.5	0.4	
0.9	7.4	12.7		0.6	0.2	
1.2	11.7	15.6	17.7	0.8	0.6	0.7

Loading	Ammonia kg NH$_3$-N/m^3-day removed					
kg/m^3-day	24 °C	13 °C	6.5 °C	±	±	±
0.3	0.299	0.294	0.290	0	0.01	0.01
0.45	0.441	0.424		0	0.01	
0.6	0.549	0.498	0.320	0.01	0.02	0.01
0.75	0.599	0.452		0.02	0.01	
0.9	0.632	0.441		0.02	0.02	
1.2	0.637	0.450	0.360	0.04	0.03	0.01

Ammonia flux to biofilm

Q		J (g/m^2-day)	
(L/day)	6.5 °C	13 °C	24 °C
95	0.32	0.32	0.32
189	0.35	0.54	0.59
397	0.39	0.51	0.72

APPENDIX D. EXPERIMENTAL DATA IN CHAPTER 5

Ammonia and nitrate profiles without recirculation (HRT = 8 hours)

Port Number	NH_3-N (mg/L)	NO_3^--N (mg/L)	NO_2^--N (mg/L)
p1	28.34	0.3	0.5
p2	28.99	0.5	0.6
p3	29.46	0.4	2.4
p4	28.90	0.4	2.1
p5	29.75	0.6	1.8
p6	29.85	0.7	1.3
p7	27.16	1.4	1.7
p8	23.72	5.4	2.1
p9	7.91	17.2	1.5
p10	0.05	21.4	1.8
p11	0.01	20.8	1.2
p12	0.01	21.8	0.8
p13	0.01	22.8	0.6

COD concentration within the column (HRT = 8 hrs)

	100% Qr		**200% Qr**		**300% Qr**	
Port #	COD (mg/L)	Absorbance	COD (mg/L)	Absorbance	COD (mg/L)	Absorbance
0	250		250		250	
1	175.25	0.082	170.25	0.080	185.25	0.086
2	165.25	0.078	150.25	0.072	160.25	0.076
3	75.25	0.042	51.25	0.148	16.67	0.231
4	65.25	0.038	50.00	0.151	10.00	0.247
5	60.25	0.036	40.42	0.174	12.92	0.240
6	55.25	0.034	35.83	0.185	22.50	0.217
7	47.75	0.031	34.58	0.188	17.08	0.230
8	48.33	0.155	25.42	0.210	13.75	0.238
9	36.67	0.183	22.92	0.216	5.83	0.257
10	12.50	0.241	10.00	0.247	7.50	0.253
11	10.83	0.245	8.33	0.251	3.75	0.262
12	7.08	0.254	7.50	0.253	2.92	0.264
13	6.25	0.256	3.33	0.263	2.50	0.265

Ammonia concentration within the column (HRT = 8 hours)

Port Number	with 100% Qr NH_3-N (mg/L)	ln (mV)	with 200% Qr NH_3-N (mg/L)	ln(mV)	with 300% Qr NH_3-N (mg/L)	ln(mV)
p1	29.00	3.68	29.03	3.67	28.01	3.70
p2	30.00	3.66	28.70	3.71	25.80	3.76
p3	12.34	4.10	8.67	4.23	6.17	4.29
p4	13.35	4.07	9.27	4.21	7.18	4.32
p5	14.79	4.02	10.16	4.19	6.67	4.36
p6	15.11	4.01	10.35	4.10	8.32	4.24
p7	14.57	4.03	10.26	4.10	6.81	4.35
p8	12.29	4.10	8.87	4.20	6.05	4.40
p9	11.64	4.13	9.04	4.19	4.70	4.50
p10	11.01	4.15	6.00	4.41	0.55	4.80
p11	7.87	4.25	4.12	4.54	0	5.24
p12	2.64	4.55	0.17	4.82	0	5.27
p13	0.05	5.00	0	5.28	0	5.29

Nitrate concentration within the column

Port Number	With 100% Qr Nitrate (mg/L)	With 200% Qr Nitrate(mg/L)	With 300% Qr Nitrate(mg/L)
p1	0.6	0.3	0.4
p2	0.8	0.5	0.6
p3	6.2	3.7	3.1
p4	4.8	3.0	2.5
p5	3.5	1.4	1.3
p6	2.0	0.9	0.5
p7	1.5	0.4	0.1
p8	2.4	1.2	0.5
p9	2.7	2.1	1.8
p10	3.8	3.1	3.1
p11	5.2	4.7	3.8
p12	8.1	5.1	4.3
p13	9.6	5.4	4.4

Effect of HRTs (Influent NH_3-N, COD, TN = 25, 250, 35 mg/L)

HRT = 6hr	% Removal NH_3-N	% Removal COD	% Removal TN	Effluent mg/L NH_3-N	Effluent mg/L COD	Effluent mg/L TN
100% Qr	100	98.2	68.9	0	4.44	10.9
200% Qr	100	97.7	76.1	0	5.69	8.37
300% Qr	100	97.0	82.1	0	7.5	6.27

HRT = 4hr	% Removal NH_3-N	% Removal COD	% Removal TN	Effluent mg/L NH_3-N	Effluent mg/L COD	Effluent mg/L TN
100% Qr	100	98.0	65.4	0	5.00	12.10
200% Qr	100	97.4	76.1	0	6.39	8.37
300% Qr	100	96.7	82.6	0	8.19	6.11

HRT = 3hr	% Removal NH_3-N	% Removal COD	% Removal TN	Effluent mg/L NH_3-N	Effluent mg/L COD	Effluent mg/L TN
100% Qr	68.8	96.8	67.8	7.83	7.92	11.27
200% Qr	96.0	96.2	75.3	0.99	9.58	8.63
300% Qr	96.0	96.3	78.8	0.98	9.17	7.43

HRT = 2hr	% Removal NH_3-N	% Removal COD	% Removal TN	Effluent mg/L NH_3-N	Effluent mg/L COD	Effluent mg/L TN
100% Qr	42.0	87.7	38.1	14.47	30.83	21.67
200% Qr	62.8	90.0	54.3	9.34	25.00	16
300% Qr	71.6	92.5	62.9	7.07	18.75	13

Kinetics (half order) on denitrification in anoxic zone

With Qr 100% h/(Co)^0.5 ([mg/L] ^0.5/hr)	**With Qr 200% h/(Co)^0.5 ([mg/L] ^0.5/hr)**	**With Qr 300% h/(Co)^0.5 ([mg/L] ^0.5/hr)**	**With Qr 100% (C/Co)^0.5**	**With Qr 200% (C/Co)^0.5**	**With Qr 300% (C/Co)^0.5**
0	0	0	1	1	1
0.07	0.04	0.02	0.88	0.90	0.90
0.14	0.07	0.04	0.75	0.62	0.65
0.21	0.11	0.06	0.57	0.49	0.40
0.28	0.15	0.08	0.49	0.33	0.18

Kinetics (zero order) on nitrification in oxic zone

	Time(hr)		**100% Qr**	**200% Qr**	**300% Qr**
100%	**200%**	**300%**	**NH_3-N (mg/l)**	**NH_3-N (mg/l)**	**NH_3-N (mg/l)**
0	0	0	14.57	10.26	6.81
0.29	0.19	0.15	12.29	8.87	6.05
0.58	0.39	0.29	11.64	9.04	4.70
0.88	0.58	0.44	11.01	6.00	0.55
1.17	0.78	0.58	7.87	4.12	0
1.46	0.97	0.73	2.64	0.17	0
1.75	1.17	0.88	0.05	0	0

Nitrite and phosphorus changes

Port #	**PO_4^{-3}-P(mg/L)**	**Absorbance**	**NO_2^--N(mg/L)**
p1	8.27	0.91	0.5
p2	8.38	0.92	0.6
p3	6.99	0.77	2.4
p4	7.44	0.82	2.1
p5	6.81	0.75	1.8
p6	6.92	0.76	1.3
p7	6.88	0.76	1.7
p8	6.98	0.77	2.1
p9	6.96	0.77	1.5
p10	6.92	0.76	1.8
p11	6.74	0.74	1.2
p12	6.35	0.70	0.8
p13	5.93	0.66	0.6

ORP and pH changes within the column

Port #	pH	ORP(mV)
p1	7.1	-312.2
p2	7.1	-250.6
p3	7.2	-204.6
p4	7.4	-214.5
p5	7.6	-284.1
p6	7.7	-294.6
p7	7.9	92.0
p8	7.9	138.0
p9	7.8	151.0
p10	7.6	171.2
p11	7.5	171.7
p12	7.4	172.0
p13	7.3	172.7

Ammonia mass removed

applied NH_3-N versus NH_3-N mass removed

applied load kg/m^3-day	**mass removed kg/m^3-day** 100%	200%	300%
0.05	0.05	0.05	0.05
0.10	0.10	0.10	0.10
0.15	0.15	0.15	0.15
0.19	0.13	0.18	0.19
0.29	0.12	0.18	0.21

Standard deviation

applied load kg/m^3-day	**Mass removed kg/m^3-day** 100%	200%	300%
0.05	0	0	0
0.10	0	0	0
0.15	0	0	0
0.19	0.01	0.01	0.01
0.29	0.02	0.01	0.01

sCOD mass removed within the column

Applied SCOD versus removed SCOD

Applied load kg/m^3-day	**Mass removed kg/m3-day** 100%	200%	300%
0.5052015	0.494	0.495	0.497
0.969267	0.954	0.947	0.940
1.452615	1.424	1.416	1.404
1.9372485	1.876	1.863	1.866
2.9065155	2.548	2.616	2.688

Standard Deviation

Applied load kg/m^3-day	**Mass removed kg/m^3-day** 100%	200%	300%
0.51	0	0	0
0.97	0	0	0
1.45	0	0	0
1.94	0	0	0
2.91	0.04	0.03	0.02

TN mass removed in the column

applied mass TN versus TN mass removed

Applied mass load kg/m^3-day	**Mass removed kg/m^3-day**			**TN removal(%)**		
	100%	200%	300%	100%	200%	300%
0.07	0.048	0.057	0.059	67.71	80.57	83.43
0.14	0.093	0.103	0.111	68.86	76.00	82.00
0.20	0.133	0.155	0.168	65.43	76.00	82.57
0.27	0.184	0.205	0.214	67.71	75.43	78.86
0.41	0.156	0.221	0.256	38.29	54.29	62.86

Standard Deviation

Applied mass load kg/m^3-day	**Mass removed kg/m^3-day**			**TN removal(%)**		
	100%	200%	300%	100%	200%	300%
0.07	0.001	0.001	0.001	1.86	1.63	1.83
0.14	0.003	0.003	0.001	2.26	1.97	1.11
0.20	0.003	0.003	0.001	1.29	1.31	0.60
0.27	0.005	0.003	0.002	1.94	1.06	0.74
0.41	0.016	0.008	0.012	4.00	2.00	2.86

APPENDIX E. EXPERIMENTAL DATA IN CHAPTER 6

Influent = 8 mg/L , HRT=6hrs

(Under aerobic condition)			(Under anaerobic condition)		
Time(hr)	**PO_4^{-3}-P (mg/L)**	**absorbance**	**Time(hr)**	**PO_4^{-3}-P (mg/L)**	**absorbance**
0	5.61	0.623	**0**	4.05	0.454
0.5	3.55	0.400	**2**	8.73	0.488
2	0.88	0.112	**4**	10.58	0.588
4	0.67	0.089	**6**	14.32	0.790
6	0.50	0.071	**8**	14.99	0.826
8	0.61	0.082	**10**	14.25	0.786
10	0.80	0.104	**12**	13.10	0.724
12	1.05	0.130	**14**	9.67	0.539
14	1.19	0.146	**16**	8.24	0.462
16	1.34	0.162	**20**	7.48	0.421
18	1.43	0.171	**24**	6.39	0.362
20	1.54	0.183	**28**	6.61	0.374
22	1.62	0.192	**32**	6.45	0.365
24	1.69	0.200	**36**	6.28	0.356
26	1.72	0.203	**40**	6.52	0.369
28	1.80	0.211	**44**	6.56	0.371
30	1.84	0.216			
32	1.88	0.220			
34	1.92	0.224			
36	1.98	0.231			
38	2.05	0.238			
40	2.11	0.245			
42	2.17	0.251			
44	2.21	0.256			
46	2.26	0.261			

2nd alternating Reactor PO_4^{-3}-P removal under aerobic condition

Influent = 8 mg/L depth(m)	PO^{-3}_4-P (mg/L)	Absorbance
0	5.76	0.645
0.34	3.43	0.386
0.68	1.33	0.153
1.02	1.04	0.120
1.36	0.61	0.072
1.7	0.24	0.031

Influent = 16 mg/L depth(m)	PO^{-3}_4-P (mg/L)	Absorbance
0	12.19	0.656
0.34	6.15	0.662
0.68	2.91	0.321
1.02	1.35	0.157
1.36	1.02	0.122
1.7	0.65	0.083