ADVANCED WASTEWATER TREATMENT USING ACYLE-RESIN FIBER BIOMASS CARRIER

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(アクリル繊維性の微生物担体を用いた排水の高度処理に関する研究)

Doctoral Dissertation
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Abstract

Some problems remain unsolved for conventional suspended processes such as a large space requirement, complicated operation and excessive waste sludge production. The swim-bed technology involving a novel acryl fiber material, biofringe, is presented for effective treatment of organic wastewater. The biofringe material allows attachment of large amounts of biomass on a flexible matrix in a fixed position. By this approach, flexing of the matrix induced by wastewater flow creates a "swimming" motion that enhances mass transfer of nutrients to the attached growth (*i.e.*, biofilm). Thus, all the potential benefits of fixed-bed and fluidized-bed reactors are retained.

In this research, the performances of the swim-bed technology on high-rate and domestic wastewater are described. In addition, reductions of excess sludge in different process configurations are also reported.

The capacity of a single swim-bed process for high-rate wastewater treatment was investigated in the first part study. 7.7 L and 21.6 L reactors, packed with biofringe carriers were used in this part. They demonstrated effective treatment of organic wastewater with 80% COD removal efficiencies at volumetric loadings up to 12 $kg/m^3/d$ and hydraulic retention times as short as 3 h. As much as 133 g of biomass per meter of biofringe support matrix was retained or 13.3 g/L with respect to the biofringe retention zone. Only limited evidence for nitrification occurred at low COD loading rates (ca. 1.6 $kg/m^3/d$). In addition, filamentous growth was very heavy at the lower loading rates, but was avoidable at COD loadings of 8 $kg/m^3/d$ or greater. The levels of extracellular polymers--proteins in particular--in the biofilm were very high compared to levels reported for flocculent or granular sludges. While treatment in this study focused on industrial level applications, the possibility of using this technology in other treatment scenarios involving lower organic loadings was discussed.

The second part of this study focused mainly on the reduction of excess sludge in a 21.6 L single biofringe reactor. The research work was conducted by increasing the COD volumetric loading rates from 1 kg-$COD/m^3/d$ to 7 kg-$COD/m^3/d$ stepwisely. Sludge productions were reduced considerably at loadings from 1 kg-$COD/m^3/d$ to 5 kg-$COD/m^3/d$, during which period observed sludge yields and apparent sludge yields ranged from 0.14

kg-MLSS/kg-$COD_{removed}$ to 0.19 kg-MLSS/kg-$COD_{removed}$ and from 0.060 kg-TSS/kg-$COD_{removed}$ to 0.12 kg-TSS/kg-$COD_{removed}$, respectively. Although sharply increase values of observed sludge yield of 0.29 kg-MLSS/kg-$COD_{removed}$ and apparent sludge yield of 0.21 kg-TSS/kg-$COD_{removed}$ were observed at a loading of 7 kg-COD/m^3/d, they were still lower than those reported for conventional activated sludge processes. Species and densities of the protozoa and metazoa varied with the changes of loadings. *Ciliates* were the dominant specie at a loading of 1 kg-COD/m^3/d and *Rotifers* played the main role from loadings of 2 kg-COD/m^3/d to 5 kg-COD/m^3/d. The occurrence of abundant protozoa and metazoa could account for the significant sludge reduction. Sharply increased sludge yields at a loading of 7 kg-COD/m^3/d was associated with a shortage of protozoa and metazoa. The longer SRTs were thought to be a reason for the observed excess sludge reduction. The high levels of the extracellular polymers (EPS) varying from 64% to 84% contributed to the large amount of biomass attached on the biofirnge.

The performance of the swim-bed process with sludge recirculation for high-rate treatment of high strength wastewater was investigated in the third part of the study. The enhanced treatment capacities of this system were achieved by increasing the volumetric loading rates. The best performances were achieved at a loading of 6 kg-COD/m^3/d, with an average COD removal efficiency of 96.8%, nitrification efficiencies up to 79%, and an average total nitrogen removal efficiency up to 59.1%. Clear effluent with average SS of 32mg/L was achieved. Mixed liquor suspended solid (MLSS) concentrations in the reactor reached over 20g/L, while sludge volume indexes (SVI) was as low as 30. The MLVSS concentration was 93% of the MLSS. The high level of biomass maintained in the reactor would account for the exceptional treatment efficiencies. Moreover, microscopic observations revealed the granular-like floc formation in the reactor at a loading of 6 kg-COD/m^3/d, which could be responsible for the notable sludge settling ability. The anoxic zone contained in the granular-like floc should enhance the nitrogen removal. The biofringe carrier was considered to be the reason for the formation of the granular-like floc because it promoted the high amount of extracellular polymeric substances (EPS) contained in the biomass.

The forth part of the study explored the performance of the swim-bed technology with sludge recirculation for domestic wastewater treatment. The study was carried out by shortening the hydraulic retention time (HRT) to increase the volumetric loading rate. A conventional activated sludge (CAS) process was operated in parallel for comparison under loadings from 0.5 kg-BOD/m^3/d to 1.5 kg-BOD/m^3/d. The BF process showed better tolerance to organic loading shock and demonstrated superior contaminant removal efficiencies with stable performance compared to those of the CAS process. Sludge washout occurred frequently in the CAS reactor at loading rates up to 0.5 kg-BOD/m^3/d and the CAS process had to be restarted with new sludge after increaing the loading to 1.5 kg-BOD/m^3/d. Sludge production was reduced approximately 40% in the BF process at different loadings comparing to those of the CAS process. Much more protozoa and metazoa were contained in the BF process, which could be responsible for the lower sludge production and improved contaminant removal. The HRTs were further shortened in the BF process after the comparative tests were finished. Over 85% COD removal was achieved even when the HRT was as short as 1.2 h at a loading of 3 kg-BOD/m^3/d. However, pronounced sludge bulking made it difficult to conduct sludge recycle. Low strength influent with easily biodegradable compounds enhanced the growth of filamentous microorganism and *Zoogloea* spp., which further resulted in the sludge bulking. Strategies on operational conditions with short HRTs to abate sludge bulking were discussed.

The performance of the swim-bed process with sludge recirculation was investigated in the fifth part of the study using three kinds of low-strength wastewater. Among them two kinds were synthetic in the lab, one was diluted by the corn-steep liquor, and the other one was a mixture of peptone and meat. The third one was an industrial wastewater containing polyvinyl chloride (PVC). Treatment performances and sludge characteristics were compared under different HRTs. Experimental results showed that high removal efficiencies were achieved treating these three kinds of wastewater. However, sludge bulking could not be avoided treating those two synthetic wastewaters. On the other hand, with granular floc formatopm, excellent settling performances were achieved for the treatment of industrial wastewater. It was considered that the excessive filamentous and *Zoogloae* spp. growth could be inhibited by the difficult discomposing characteristics of the PVC.

In general, swim-bed technology demonstrated remarkable performance for high-rate treatment of high-strength wastewater, and treatment of low strength wastewater at a short HRT. Considerable excess sludge reductions could also be achieved under different operational conditions.

List of abbreviations*

BF	Biofringe
CAS	Conventional Activated Sludge
EPS	Extracelluar Polymers
HRT	Hydraulic Retention Time
NE	Nitrification Efficiency
SSB	Single Swim-Bed process
SRT	Sludge Retention Time
SSBR	Single Swim-Bed reactor with sludge Recycle
VLR	Volumetric Loading Rate

Excluding chemical formulae and common abbreviation such as COD, MLSS, etc.

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Chapter 1 Introduction

1.1 Suspended growth process

The principal biological processes used for wastewater treatment can be divided into two main categories: suspended growth and attached growth (or biofilm) processes. In suspended growth processes, the microorganisms responsible for treatment are maintained in liquid suspension by appropriate mixing methods. Many suspended growth processes used in municipal and industrial wastewater treatment are operated with a positive dissolved oxygen concentration (aerobic), but applications exist where suspended growth anaerobic reactors are used, such as high organic concentration industrial wastewaters and organic sludge. The most common suspended growth process used for municipal wastewater treatment is the activated-sludge process.

The activated sludge process was developed around 1913 at the Lawrence Experiment Station in Massachusetts by Clark and Gage (Metcalf and Eddy, 1930), and by Ardern and Lockett (1914) at the Manchester Sewage Works in Mancheseter. The activated sludge process was so named because it involved the production of an activated mass of microorganism capable of stabilizing a waste under aerobic conditions. In the aerobic tank, contact time is provided for mixing and aerating influent wastewater with the microbial suspension generally referred to as the mixed liquor suspended solids (MLSS) or mixed liquor volatile suspended solids (MLVSS). Mechanical equipment is used to provide the mixing and transfer of oxygen into the process. The mixed liquor then flows to clarifier where the microbial suspension is settled and thickened. The settled biomass, described as activated sludge because of the presence of active microorganisms, is returned to the aeration tank to continue biodegradation of the influent organic material. A portion of the thickened solids is removed daily or periodically as the excess biomass that would accumulate along with the non-biodegradable solids contained in the influent wastewater. If the accumulated solids are not removed, they will eventually find their way to the system effluent.

These treatment processes are typically referred to as primary treatment (solids settlement), secondary treatment (biological treatment of the supernatant and settled solids) and tertiary treatment (additional polishing stages such as lagoon, micro-filtration or disinfection).

However, conventional activated sludge processes are dealing with several disadvantages: large sludge production, limited sludge concentration, sometimes poor sedimentation, large sedimentation basins and relatively poor effluent quality (suspended solids) (Cornelissen, et al., 2003).

Especially, the excess sludge production of conventional activated processes is reported to be around 15-100 L/(kg-BOD removed), where over 95% of water is included (Cui, et al., 2004). Treatment and disposal of excess sludge in an activated sludge system requires tremendous costs, amounting to approximately half the entire operational cost for domestic wastewater treatment plants. Appropriate management of the excess sludge is an essential part of wastewater treatment. The disposal of excess sludge, however, has been one of the most expensive processes. It may be up to 60% of the total plant operation costs (Lay, et al., 1999).

1.2 Attached growth process in wastewater treatment

1.2.1 Comparison of attached growth process and suspended growth process

The attached growth process (biofilm) as another alternative of biological wastewater treatment methods has also a long history. In this process, a biofilm consisting of microorganisms, particulate material, and extra-cellular polymers is attached and covers the support packing material, which may be plastic, rock, or other material. The growth and substrate utilization kinetics described for the suspended growth process were related to the dissolved substrate concentration in the bulk liquid. For attached growth processes, substrate is consumed within a biofilm. Depending on the growth conditions and the hydrodynamics of the system, the biofilm thickness may range from 100 μm to 10 mm (WEF, 2000). A stagnant

liquid layer (diffusion layer) separates the biofilm from the bulk liquid that is flowing over the surface of the biofilm or is mixed outside of the fixed film. Substrates, oxygen and nutrients diffuse across the stagnant liquid layer to the biofilm, and the products of biodegradation from the biofilm enter the bulk liquid after diffusion across the stagnant film.

Attached growth processes can be grouped into three general classes: (1) no submerged attached growth processes, (2) suspended growth process with fixed-film packing, and (3) submerged attached growth aerobic processes.

Tricking filters with rock packing, as a nonsubmerged fixed-film biological reactor, have been a common, simple, and low-energy process used for secondary treatment since the early 1900s. The concept of a trickling filter grew from the use of contact filters in England in the late 1890s. In the 1950s, plastic packing began to replace rock in the United States. The use of plastic packing allowed the use of higher loading rates and taller filters with less land area, improved process efficiency, and reduced clogging. In the 1960s, practical designs were developed for rotating biological contactors (RBCs), which provided an alternative attached growth process where the packing is rotated in the wastewater treatment tank, versus pumping and applying the wastewater over static packing.

The placement of packing materials in the aeration tank of the activated-sludge dates back to the 1940s with the Hays and Griffith processes (WEF, 2000). Present-day designs use more engineered packings and include the use of packing materials that are suspended in the aeration tank with mixed liquor, fixed packing material placed in portions of the aeration tank, as well as submerged RBCs.

Submerged attached growth processes began in the 1970s and extended into the 1980s, as a new class of aerobic attached growth processes. It could be divided into upflow, downflow packed-bed reactors and fluidized-bed reactors that do not need secondary clarification. Their unique advantage is the small footprint with an area requirement that is a fraction (one-fifth to one-third) of that needed for activated-sludge tratment.

Biofilm reactors offer several advantages over conventional suspended growth systems. A well-known property of biofilm systems is their capacity to handle shock loads. Carrier materials with adsorption or ion exchange properties allow it to buffer high concentrations of toxic substrates which would otherwise exceed the tolerance of the bacteria (Martienssen, 2000). Still a puzzling feature of biofilms is their ability to degrade high concentrations of even toxic substrates. The microorganism detached from the packing material, with higher density and bigger granular, is easy to separate from the liquid. Furthermore, attached growth processes could treat low concentration wastewater. For conventional activated sludge system, if the influent BOD value keeps lower than 50-60 mg/L, it will affect the formation and growth of sludge flock. The attached growth processes, however, could degrade the wastewater with BOD_5 value as low as 20-30 mg/L to 5-10 mg/L. Also, attached growth processes are easy to manage and could cut the cost.

Furthermore, in the CAS process, the MLVSS can usually be kept at 1,500-3,000 mg/L in the aeration tank, and the sludge is in the completely suspending state. Consequently, if increasing loading rate or organic concentration further, it will cause the biomass washout or make it difficult for solid-liquid separation, which will result in the effluent quality deteriorating. In contrast, attached growth process can retain high biomass concentration since the microorganism is adsorbed on the surface of support material. The maximum MLVSS concentration can reach to 22,000-150,000 mg/L, which is 7-20 times higher than the CAS process. In the attached growth process, the biomass attach on the surface of support material, so their density of them is heavier than liquid. Furthermore, the close packing state makes it easy to separate from liquid. This characteristic is beneficial for biomass retaining, especially makes it possible to keep high microorganism concentration with low growth rate. For instance, the nitrifying bacterium is one kind of low growth rate bacteria. In the CAS process, high nitrifier concentration needs not only controlling the substrate concentration but also dominating the sludge retention time (SRT). While attached growth process can immobilize the biomass on the surface of support material, and supply enough growth time, which is in favor of the microorganism enrichment. In the attached growth process, biomass is usually enclosed by the support material or in compactness state. So when the biofilm

contacts the toxic pollutant in the wastewater, the strong resistance of close packing state or blocking effect of support material can abate the impaction of the toxic waste to the microorganism, thereby the operation stability is enhanced in maximum extent.

1.2.2 Fixed bed process

For the aerobic fixed-film processes, it is made up of packing, biofilm and liquid. The organic pollutants or ammonium is oxidized when the liquid flows past the biofilm formed on the surface of packing. Usually, oxygen is supplied by diffused aeration into the packing or by being predissolved into the influent wastewater. The type and size of packing is a major factor that affects the performance and operating characteristics of fixed bed process. For this process, excess solids from biomass growth and influent suspended solids are trapped in the system and must be periodically removed. Besides, it requires a backwashing system much like that used in a water filtration plant to flush out accumulated solids, usually on a daily basis.

The major advantages of fixed bed processes are their relatively small space requirement, the capacity to effectively treat dilute wastewater, no sludge settling issues as in activated-sludge process, and aesthetics. Their disadvantages include a more complicated system in terms of instrumentation and controls, limitations of economies of scale for application to larger facilities, and generally a higher capital cost than activated-sludge treatment.

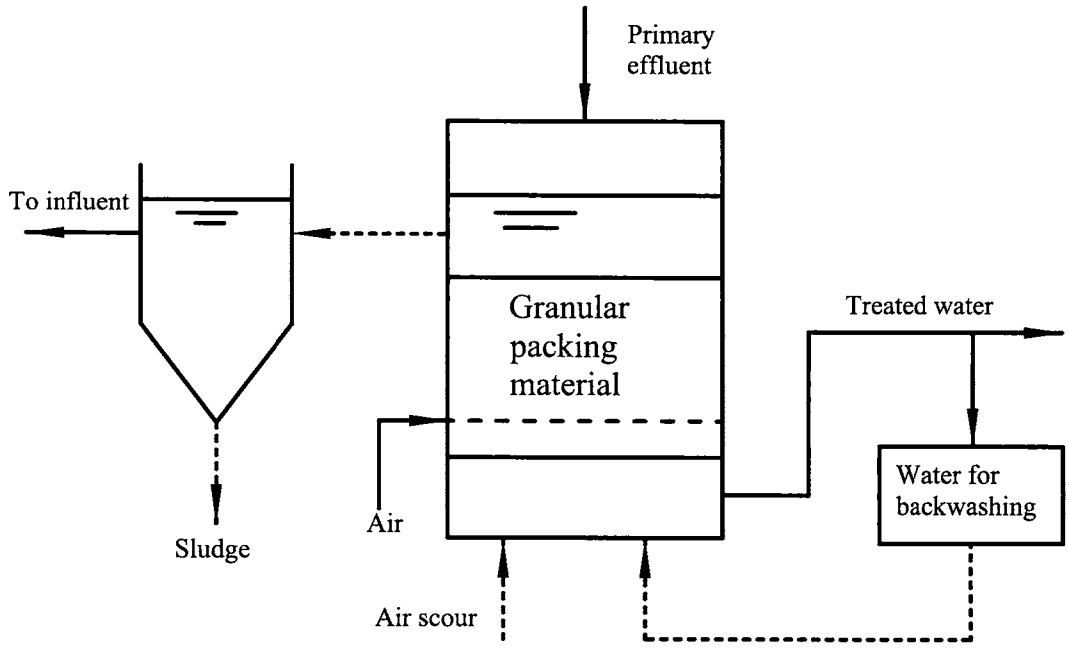


Fig. 1- 1 Schematic of Biocarbone down flow biological process

The Biocarbone process is a typical example of a downflow submerged attached growth process. Over 100 facilities have been constructed worldwide since the development of the process in France in the early 1980s. The process has also been termed the biological aerated filter (BAF) (Stensel., et al., 1988). In contrast, the Biofor process is an upflow submerged aerobic attached growth process. This process also spread widely in Europe and North America. Inlet nozzles distribute the influent wastewater up through the bed, and an air header provides process air across the bed area. Backwashing is typically done once per day with a waster flush rate of 10 to 30 m/h to expand the bed (Lazarova, et al., 2000). Fine screening of the wastewater is needed to protect the inlet nozzles. The Biofor process has been applied for BOD removal and nitrification, tertiary nitrification, and denitrification.

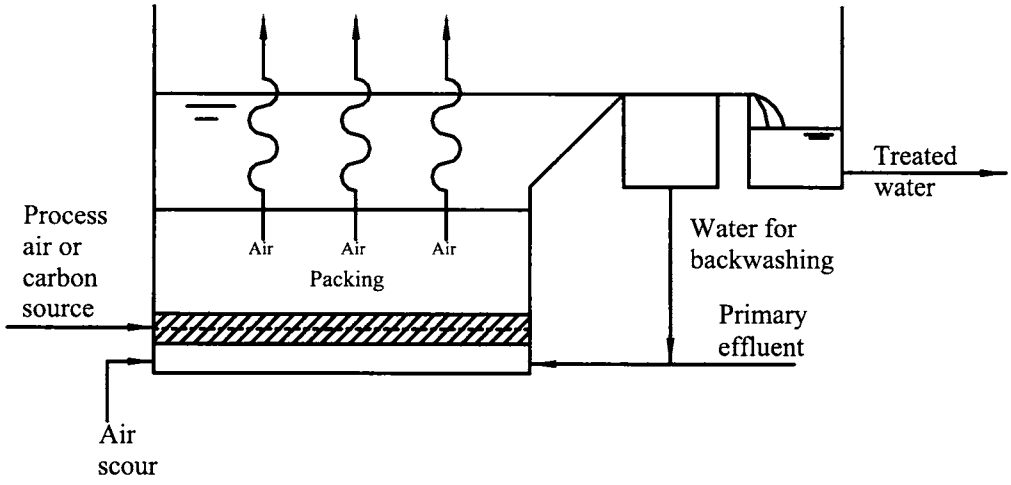


Fig. 1- 2 Schematic of Biofor upflow biological reactor

For anaerobic treatment, fixed bed processes also play very important roles. Generally, packing material placement may be in the entire depth or, for hybrid designs, only in the upper 50 to 70 percent. The most common packing materials are corrugated plastic cross-flow or tubular modules, similar to aerobic attached growth processes. Low upflow velocities are generally used to prevent washing out the biomass. Over time, solids and biomass will accumulate in the packing to cause plugging and flow short-circuiting. At this point, solids must be removed by flushing and draining the packing. This process could endure high COD loadings, with relatively small reactor volumes, and operational simplicity. However, the cost of the packing material and operational problems are the main limitations for this process. Besides, maintenance associated with solid accumulation and possible packing plugging could affect the performance of this process. This process is best suited for wastewaters with low suspended solids concentrations.

1.2.3 Fluidized bed process

Fluidized bed process is a new kind of wastewater treatment technologies and was developed in 1970s. The packing materials usually are sand and activated carbon. Since the biofilm increases in size, the packing becomes lighter and can fluidize with wastewater through the whole reactor. The packing of small granular have high value of specific surface area (2,000-3,000 m^2/m^3), which makes this system with high biomass concentration (10-14 g/L).

Moreover, the packing in fluidized state can provide more time of exposure with contaminants when wastewater flows from bottom to up or from left to right. Also the high bifilm activity can be obtained because of the friction and collision among the packing, which also enhances the mass transfer efficiency. As the packing fluidize ceaselessly, it is beneficial for preventing the clogging in this system. According to the different power supply, the fluidized bed processes can be divided into two-phase fluidized bed process with liquid stream as power supply, three-phase fluidized bed process with air stream as power supply, and mechanical fluidized bed process. It can also be categorized into aerobic and anaerobic fluidized bed processes in response to the oxygen involving or not.

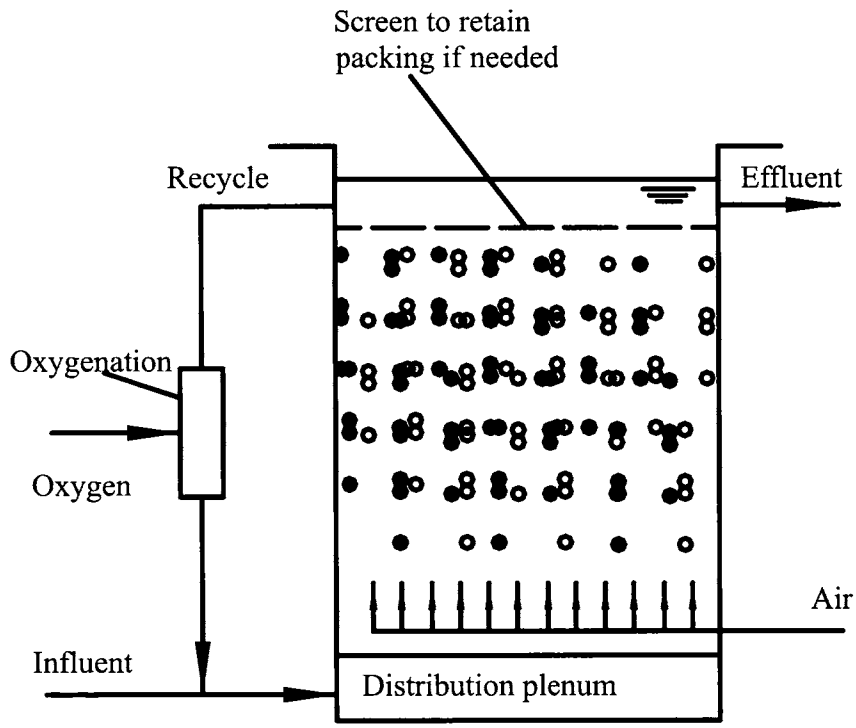


Fig. 1- 3 Schematic of fluidized bed biological reactor (FBBR)

For municipal wastewater treatment, fluidized bed processes have been applied for post-denitrification. Aerobic fluidized bed processes are frequently used to treat groundwater contaminated with hazardous substances. In these applications activated carbon is used for the packing to provide both carbon adsorption and biological degradation (Stutton, et al., 1994). The main advantages for fluidized bed processes in this application are (1) it provides

an extraordinarily long SRT for microorganisms necessary to degrade the xenobiotic and toxic compounds; (2) shock loads or non-biodegradable toxic compounds can be absorbed onto the activated carbon; (3) high-quality effluent is produced low in TSS and COD concentration; (4) the oxygenation method prevents stripping and emission of toxic organic compounds to the atmosphere; and (5) the system operation is simple and reliable.

For anaerobic process, fluidized bed processes need higher upflow liquid velocities of about 20 m/h to provide about 100 percent bed expansion. And effluent recycle is used to supply sufficient upflow velocity. Besides sand, other packing materials have been considered for use in anaerobic fluidized bed processes including diatomaceous earth, anion and cation exchange resin, and activated carbon (Wang, et al., 1984; Kindzierski, et al., 1992). In anaerobic conditions, activated carbon has been widely used for treating industrial and hazardous waste streams. The mean diameter of the granular activated carbon particles is 0.6 to 0.8 mm and upflow velocities of 20 to 24 m/h are used (Hickey, et al., 1991; Iza, 1991). Anaerobic fluidized bed processes can provide high biomass concentrations, relatively high organic loadings, high mass transfer characteristics, the ability to handle shock loads due to its mixing and dilution with recycle, and minimal space requirements. However, care must be taken in the inlet and outlet designs to assure good flow distribution. Also it requires high power to make the packing to fluidize and the cost of reactor packing is high, et al. (Wang, et al., 1986; Fox, et al., 1988).

1.2.4 Swim-bed technology

Swim-bed technology, involving the innovative acryl-resin fibers biomass carrier-Biofringe (NET Co., Ltd, BF), is a new concept for the wastewater treatment. The biofringe material composes of fringe yarns (diameter, *ca.* 3 mm) attached to a support filament as shown in Figure 1-4. The fringe yams are symmetrically attached, extending equal distances beyond each side the support filament, and twisted to give an even 3-dimsonal distribution.

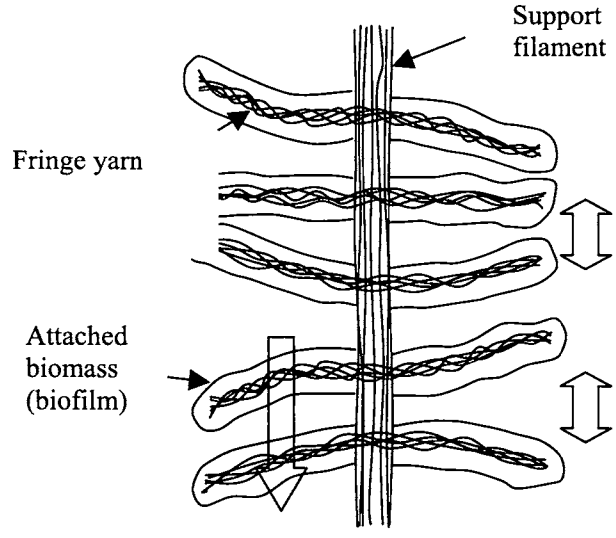


Fig. 1- 4 Configuration for biomass carrier

The staple fiber of the fringe yarns is a hydrophilic acrylic composite. The material has a rough texture with a porosity surface, which allows for a great amount of the sludge attached on it. BF with a flexible fringe yam matrix in a fix position is induced by water flow to flex, which causes a 'swimming' motion that enhances mass transfer of nutrients to the bioflm. So that, swim-bed technology combines the advantages of fix-bed and fluidized-bed processes. It also eliminates the head losses with absence of clogging and channeling, which cannot be easily avoided in a fix-bed process. The process could be continuously operated without dependence on hydrodynamic conditions to avoid settling or floating of the attachment medium or the requirement of screens or traps to prevent washout, which can be difficult to achieve in fluidized-bed process. Fig. 1-5 gives a section view of the biofringe material with biomass attachment. The anoxic zone and aerobic zone formed in the thick biofilm, which provided a possible condition for the simultaneous nitrification and denitrification. The large amount of sludge attachment would extend the sludge retention time (SRT) and encourage the much more occurrence of protozoa and metazoa. Apart from the high efficiencies on contaminant removal, the sludge reduction also should be achieved by applied this approach.

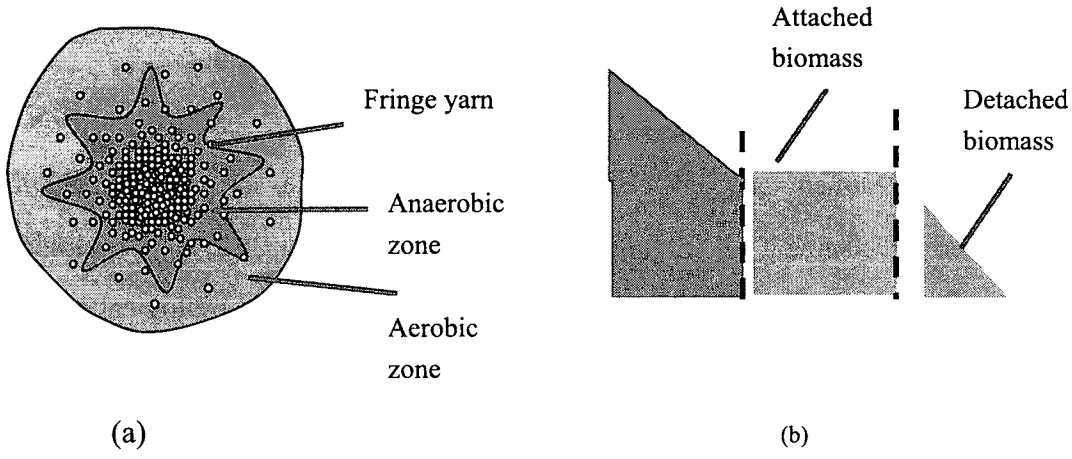


Fig. 1- 5 Sectional view of the fiber

Since characteristics of the BF material is light, durable, the simple reactor design with less space requirement and uncomplicated operation with low cost should be achieved by application the swim-bed to the practical wastewater treatment project. Additional, the potential remarkable treatment performance and considerable excess sludge reduction make swim-bed technology become a promising technology for wastewater treatment.

1.3 High organic loading rate wastewater treatment

High organic concentration wastewater usually has the following characteristics: (1) it has a very high COD concentration, above 2,000 mg/L generally, but the BOD concentration is fairly low and the ratio of BOD to COD is lower than 0.3 commonly; (2) toxic waste is contained in the high concentration wastewater, including aromatic compound, heterogeneous ring compound, sulfide and heavy metal ion, which are difficult to treat and very dangerous to human being; (3) it has high concentration chromaticity and bad foul smell; (4) strong acid or strong base are included.

Due to the biodegradation by biomass, high concentration wastewater can cause the water column oxygen deficit, which will result in the death of biologic creature and water quality deterioration. The toxicity of this kind of wastewater can accumulate in the water column and

soil for a long time, and it is possible that the toxic substance is absorbed by human being body through biological magnification and biological accumulation.

Among the treatment process for high concentration wastewater, biological process is the main alternative since it is economical, no secondary pollutant, etc.. The biological processes can be classified into aerobic biological process, anaerobic biological process, biofilm process, fermentation process, and so on.

Jeong, et al. obtained granular sludge with high concentration in a UASB reactor, and the size of the granular sludge ranged from 1 to 5 mm with good settling ability. Later high COD removal efficiency of 90% was achieved at high organic loading rate up to 18 kg-COD/m^3/d (Jeong, et al., 2005).

Sallis, et al. have conducted a split-feed anaerobic baffled reactor (SFABF) for granule development. Comparing the performance of SFABF and anaerobic baffled reactor (ABF), they found that the SFABR configuration could promote the formation of granules, with stable granulation being achieved in a relatively short period of time. Consequently, in the SFABR, 95% COD was removed at an organic loading rate of 10.5 kg-COD/m^3/d after only 70 days operation (Sallis and Uyanik, 2003).

Lee, et al. developed a novel bioreactor containing self-flocculated anaerobic granular sludge for high-performance hydrogen production from sucrose-based synthetic wastewater. This system could produce an optimal volumetric hydrogen production rate of 7.3 L/h/L and a maximal hydrogen yield of 3.03 mol H_2/mol sucrose when it was operated at a hydraulic retention time of 0.5 h. Packing of a small quantity of carrier matrices on the bottom of the upflow reactor significantly stimulated sludge granulation that can be accomplished within 100 h. So the bioreactor could start up with a low HRT of 4-8 h (corresponding to an organic loading rate of 2.5-5.0 g-COD/h/L) and enable stable operations at an extremely low HRT (up to 0.5 h) without washout of biomass. The ability to maintain high biomass concentration (maximum 26 g/L) at high organic loading rate highlighted the hydrogen production

efficiency (Lee, et al., 2004).

One *Chryseomonas luteola* strain was isolated from raw baker's yeast factory effluent as the dominant part of the microbial community and evaluated for its biodegradative activity, using the raw effluent as substrate. The strain was able to utilise the raw effluent and produce higher concentrations of energetically favourable metabolites and thereby, could contribute to the first degradation step in an anaerobic biological treatment process. A 3×4×3 factorial design indicated optimal degradation conditions in a specific environmental framework of 48 h incubation time, COD concentration of 30 g/l, pH of 6.0 and temperature of 35□. The *C. luteola* strain was thereafter used in a pre-degradation step followed by an anaerobic digestion step in a 5 L laboratory-scale hybrid digester. With the use of the pre-degraded effluent, significant improvements were found in the overall anaerobic digestion performance. These included increased COD (>15%) and TVFA (>50%) removals, especially propionic acid (88%) removal, as well as higher biogas yields (18%). The results also showed a prominent improvement in fatty acid utilisation and methanogenesis. The predegradation step resulted in better process control and increased stability of the system, even at relatively high organic loading rates (10 kg COD/m^3/d). When the raw effluent was not pre-treated (control bioreactor), no improvement in bioreactor efficiency was observed (Van Der Merwe-Botha and Britz, 1997)

An upflow anaerobic sludge blanket (UASB) technology has showed an excellent performance compared to other biological treatment methods when a high organic loading rate of domestic and industrial wastewaters was applied. Park, et al. (1998) applied one kind of algae from lake-sediments and synthetic activated ceramic as media on UASB performance. The algae and synthetic media were introduced to the UASB reactor in order to obtain the enhanced granulation, which resulted in the increase in the UASB performance. 1-3% higher methane content and 3-10% higher COD removal efficiency were obtained in the reactors with the media than in the reactor without the media. The respective gas production rates in three reactors 1, 2, and 3 were 0.15-0.36 m^3/kg COD/day, 0.24-0.54 m^3/kg

COD/day, and 0.24-0.56 m^3/kg COD/day. As organic loading rate increased, gas production rates increased.

For anaerobic processes, operating at high organic loading rates during start-up can lead to instability, accumulation of volatile fatty acids and low pH, such problems being exacerbated in reactors that exhibit plug-flow characteristics. Moreover, plug-flow conditions increase the exposure of biomass to any toxic components in the feed. Also the complicated operation makes it difficult to get the stable performance.

A new flexible fibre biofilm reactor was developed for the treatment of wastewater from fruit and vegetable processing plants by Yu, et al.. Experiments were carried out to evaluate the performance of the treatment process. Acclimatisation characteristics of the treatment process were also evaluated. The removal efficiencies for COD and BOD under different influent organic strengths were evaluated. Results indicated that over 90% of COD removal and 95% of BOD, removal could be achieved. The performance parameters were also compared with a conventional activated sludge process under similar conditions, operated in parallel. It was found that the biofilm reactor exhibited a number of advantages over the conventional reactor. These include (a) high organic loading rate, (b) long sludge retention times and low sludge discharge rate in the settling tank (about 10%), (c) elimination of the sludge recycle stream, and (d) no sludge bulking problem at high organic loading rates (Yu, et al., 2003).

Liu, et al (2005) used a SBR reactor for high organic loading rate synthetic wastewater treatment with aerobic granular sludge. This aerobic granular sludge could sustain the high organic loading rate to 4.0 kg-COD/m^3/d, with good settling ability (settling velocity 36 m/h).

Park, et al. has investigated the possibility of petrochemical wastewater treatment in an aerated submerged fixed-film reactor (ASFFR). This reactor demonstrated 91.8-96.6% removal efficiencies of soluble COD and exhibited efficient and stable performance at high organic loadings of 6.21 kg-COD/m^3/d (Park, et al., 1996). Holler and Trosh applied a membrane bioreactor for high organic loading rate urban wastewater treatment. The organic

loading rate has been varied in the range of 6-13 $kg/m^3/d$. With the characteristic of membrane bioreactor, the biomass concentration reached 10-22 g/L, above 95% of COD reduction was achieved (Holler and Trosch, 2001).

Biological treatment methods are usually effective and commonly used in the treatment of organic wastewater. However, there are a number of problems for conventional biological treatment methods in treating high concentration wastewater from industries such as food processing. For example, microorganisms in the aeration tank in an activated sludge process cannot survive under continuous series of shock loads and the process could become operationally unstable. The associated problem of sludge bulking often occurs when the organic loading to the treatment process is high.

1.4 Sludge reduction

Activated sludge system has been employed to treat a wide variety of wastewater, and over 90% of the municipal wastewater treatment plants use it as the core part of the treatment process. The basic function of a wastewater biological treatment process is to convert organics to carbon dioxide, water and bacterial cells. The cells can then be separated from the purified water and disposed of in a concentrated form called excess sludge. Assuming that activated sludge has a growth yield efficiency of 0.5 mg dry weight per mg biological oxygen demand (BOD), 1 kg BOD removed will generate 0.5 kg dry excess sludge. It must be realized that the excess sludge generated from the biological treatment process is a secondary solid waste that must be disposed of in a safe and cost-effective way. The ultimate disposal of excess sludge has been and continues to be one of the most expensive problems faced by wastewater utilities, e.g. the treatment of the excess sludge may account for 25% up to 65% of the total plant operation cost (Zhao and Kugel, 1997). So far, sludge production and disposal are entering a period of dramatic change, driven mainly by stringent environmental legislation. The requirements of new European Union laws on municipal wastewater treatment are creating a doubling in sludge production by the end of 2005, while at the same time placing additional restrictions on sludge disposal (Rocher, et al., 2001; Spinosa, 2001). Sludge

disposal to all the established outlets could become increasingly difficult or, in the case of sea disposal, will become illegal by this year. Environmental pressures on sludge recycling to land may lead to restrictions on applications in terms of nitrogen content and more stringent limits for metals in soils (EPA, 1999).

Various approaches relying on either one or a combination of physical, chemical and biological principles have been exploited for producing less sludge. While the underlying principles vary widely, the reduction endeavours have mainly targeted either of the following two approaches:

- Process changes to lower sludge production by the biological treatment system.
- Post treatment of excess waste activated sludge to reduce the amount for disposal.

In the former approach, the wastewater treatment plant design and/or operation is engineered to achieve minimized sludge yield (kg VSS produced/kg BOD stabilized). An extended aeration version of the CAS is one such example. On the other hand, excess WAS reduction through post treatment involves sludge oxidation by physical, chemical or biological means. A choice between the two strategies can be situation specific and could require diligent techno-economic evaluations.

1.4.1 Sludge production through processes changes

The CAS process can be modified to a low sludge producing process by making changes in the design and/or operating parameters. The growth rates of bacteria in long sludge-aged systems are generally believed to be lower than those found in short sludge-aged systems. This can be explained by the differences in the allocation of cellular energy for cell maintenance, as opposed to cellular reproduction. In the conventional approach for lower sludge production, the mixed liquor suspended solids (MLSS) are simply retained in the aeration basin under endogenous respiration for extended periods of time. One drawback of this approach is that cell solubilization proceeds slowly and thus long solids residence times are required. This limitation could be overcome by using a relatively modern approach in which the return activated sludge (RAS) is treated by physical and/or chemical means before

return to the aeration basin. Low and Chase (1999) demonstrated how sludge yield in a dispersed growth ASP could be reduced by as much as 44% by increasing the biomass concentration from 1.7 to 10 g/L while maintaining other operating conditions similar.

The extended aeration process is similar to the conventional plug-flow CAS except that it operates in the endogenous respiration mode, which requires long aeration time and a low organic loading (0.16-0.4 kg $BOD/m^3/d$ vs. 0.3-0.6 kg $BOD/m^3/d$ for the CAS process). Theoretically, no excess sludge should be produced in the extended aeration process as the growth rate of the new cells equals the decay rate of the existing cells (Corbitt, 1989). In practice, however, excess sludge is produced but in significantly lower quantities than that in the CAS process (Corbitt, 1989). A much longer cell residence time and low food/microorganism (F/M) ratio maintains the extended aeration culture in endogenous respiration producing less sludge. However, despite low organic loading, low F/M ratio, and longer hydraulic residence time (HRT) in the aeration basin, the extended aeration systems are typically 5-10% less efficient in BOD removal than the CAS process, which in certain cases could be undesirable in view of stringent environmental regulations. However, when treating pulp and paper wastewaters, extended aeration systems have been shown to remove up to 99% of incoming BOD (Ericsson, et al., 1999; Carlson, et al., 2000). The Springfield Water and Sewer Commission, operating a 67 mgd (million gallons per day) domestic wastewater treatment facility in Agawam, Massachusetts, converted their CAS process to the extended aeration process for reduced sludge production (Borgatti, et al., 2000). The extended aeration operation reduced sludge production from 715 to 504 dry ton/month-a 30% decrease. At the current landfill hauling and disposal contract cost of U.S. $310 per dry ton, savings of U.S.$785,000 per year were realized in addition to savings in sludge dewatering polymer and electricity costs. Moreover, sludge dewatering improved from 21% to 27% solids due mainly to improved P/S sludge ratio. The extended aeration sludge had lower odour potential and better stability because of low respiration rate (Borgatti, et al., 2000). The increase in aeration system electricity costs was estimated as U.S. $51,000 per year.

One approach to sludge reduction is to improve aeration to allow for a high MLSS

concentration in the aeration basin thus increasing the SRT. Oxygen transfer limitations of the conventional aeration equipment have often restricted the use of this approach as there must be sufficient dissolved oxygen in the system to meet the needs of the bacteria. Larger bacterial flocs, in particular, offer high resistance to oxygen transfer. In order to understand the direct effect of dissolved oxygen concentration in the bulk liquid on excess sludge production, Abbassi, et al. (2000) developed a mathematical model taking into account mass transfer of oxygen, biological reactions within flocs and the endogenous respiration process. Model predictions supported with lab experiments showed that the oxygen concentration in the bulk liquid has a significant effect on the amount of excess sludge production. They hypothesized that an increased oxygen concentration in the bulk liquid leads to adeeper diffusion of oxygen into the floc, which enlarges the aerobic volume inside the floc leading to a deficient situation regarding the organic substrate. According to Monod kinetics, a decreased substrate concentration puts more emphasis on cellular maintenance leading to an inferior growth rate and a lower sludge production. The formation of larger bacterial aggregates also negatively impacts sludge reduction in mixed cultures where prey and predators co-exist (Lee and Welander, 1996a).

In low sludge production (LSP) approach, the CAS is modified to a two-stage process to establish a microbial food chain that would result in reduced sludge production. The first stage, because of short solids residence time (3-5 h) and no sludge recycle, selectively promotes the dispersed growth of bacterial cells. Its primary function is to remove soluble BOD. The second stage is designed as a predator stage (long SRT) by selectively growing filter feeders (protozoa and rotifers) that prey on the single cell bacteria produced in the first stage and thus converting excess sludge into energy, water and carbon dioxide. The process principle is theoretically sound and is based on the fact that during biomass conversion from a lower trophic level (bacteria) to a higher one (protozoa and metazoa) energy is lost because maintenance and other physiological processes require energy. Decreased energy thus remains available for anabolic processes (biomass production) when higher life forms are present (Ratsak, et al., 1994). The loss of energy and biomass production has an inverse relationship. As the loss of energy approaches maximum, the biomass production approaches minimum

(Ratsak, 1994). A schematic of this low sludge process is shown in Fig. 4. Laboratory and field studies have verified that sludge production from a two trophic level process is typically less than one third of that from the conventional ASP (Asselin, et al., 2004; Stuart, et al., 2000). In ciliate proliferation studies, Ratsak, et al. (1994) found that the introduction of a second stage led to a 12-43% reduction in sludge yield compared with a single stage system. Lee and Welander (1996a, b) applied this concept to a synthetic wastewater (acetate and MeOH) and different pulp and paper industry wastewaters. The first stage was a laboratory completely stirred reactor for decomposition of dissolved organics by bacteria. The second stage was a biofilm reactor for the growth of predators. Sludge yield reductions ranging from 32% to 93% were obtained depending upon the type of biofilm carrier-material and the type of wastewater used. Microscopic examination of the biofilm confirmed the presence of a variety of species of both protozoa and metazoa, ranging from filter feeders (ciliates, rotifers) to more advanced predators (Lee and Welander, 1996a, b). Lee and Welander (1996b) investigated various designs of the predator stage, i.e., suspended growth and biofilm reactors, for pulp and paper sludge reduction. The sludge production decreased from 0.2 to 0.4 g-SS/g-COD removed for the conventional ASP to between 0.01 and 0.23 g-SS/g-COD removed for the two-stage process.

1.4.2 Sludge reduction through post treatment

Methods are available to reduce the quantity of generated sludge requiring ultimate disposal. Once produced, the sludge can be treated using a number of treatment technologies, such as heat application, chemical oxidation and digestion to reduce the amount requiring disposal.

Among heat treatment alternatives for sludge reduction, incineration is the most popular method. It completely evaporates water in sludge and effectively oxidizes organics at high temperatures to CO_2 and H_2O. In addition to being energy intensive and thus costly (Modell, et al., 1992), incineration generally suffers from operating problems in incinerators and bark boilers. The problems include sludge handling, bark and sludge mixture consistency variations and the downgraded boiler capacity because of high water content (Nichols, 1992).

Particulate and gaseous emissions requiring air pollution control equipment remain additional issues with sludge incineration. Incineration of sludge in the recovery boiler can, and has been practised in the pulp and paper industry (Harila and Kaila, 1995). The drawback is a reduction in black liquor treatment capacity of the recovery boiler.

Wet air oxidation (subcritical water oxidation) is a flameless oxidation method for the oxidation of mainly organic substances with air or other oxidizing agents at pressures of 2-20 MPa and temperatures of 150-370℃ (Zimmerman and Diddams, 1960). The process is similar in principle to supercritical water oxidation (SCWO) with the exception that the reaction mixture is kept below the critical point of water. While SCWO can achieve complete oxidation of the organic fraction of the sludge solids, wet air oxidation can achieve effective hydrolysis (>95% as COD) of the sludge organic compounds but incomplete oxidation (> 95% as COD) (Shanableh, 2000). Process efficiency depends upon operating parameters such as temperature, pressure, air supply and feed solids concentration. The extent and the rate of oxidation can be increased by elevating the reaction temperature within the range 120-370℃. Operating pressures between 1 and 27 Mpa can be used depending upon the degree of oxidation desired. As with incineration, an external oxygen supply is needed. Since thermal efficiency and process economics are strongly dependant on air input it is thus critically important that the optimum air requirement is determined. As an example, activated sludge with a heat value of 15,212 kJ/kg (6,540 BTU/lb) typically requires 5.14 kg air/kg of sludge for wet air oxidation (US EPA, 1979). Although wet oxidation does not require predewatering (as low as 1% solids sludge can be fed to the process), the process operating costs can be significantly reduced by increasing sludge consistency. In one study wet air oxidation costs decreased from $38 to $23 /tonne by increasing sludge solids from 3% to 6%. High sludge solids keep the oxidation process self-sustaining.

Aerobic or anaerobic sludge digestion is commonly practised in the municipal sector (Metcalf and Eddy Inc., 1991). These processes convert raw sludge into a less offensive form with regard to odour, rate of putrefaction and microorganism content in addition to mass reductions of 50-70% (Lee, et al., 1976). Anaerobic and aerobic digestion partially converts putrescible matter into liquid, dissolved solid and gaseousby-products with a significant

destruction of pathogens. Anaerobic digestion has been described as a potential method of reducing the quantity of pulp-mill WAS (Puhakka, et al., 1992a, b), but neither of the digestion methods are commonly used in the pulp and paper industry. The large amounts of generated sludge and the long retention times needed for its digestion are main reasons to limit its implementation. Recent advances in reducing the retention times are making these technologies more attractive to the pulp and paper industry. Excessive foaming is another problem in both aerobic and anaerobic digestion of sludge. Control methods are available but not always effective since the causes of foaming are not fully understood (Spinosa, et al., 1994).

1.5 Domestic wastewater treatment

There is a tremendous need to develop reliable technologies for the treatment of domestic wastewater. Such treatment systems must fulfill many requirements, such as simple design, use of non-sophisticated equipment, high treatment efficiency, and low operating and capital costs. In addition, consistent with population growth and increase in urbanization, the cost and availability of land is becoming a limiting factor, and "footprint size" is increasingly becoming important in the choice of a treatment system.

Anaerobic technologies should be considered for domestic wastewater treatment as an alternative to more conventional aerobic technologies in most developing countries for a variety of reasons. Anaerobic technologies already have been applied successfully for the treatment of a number of waste streams, including lowstrength wastewaters such as domestic wastewater, particularly under tropical conditions (Van Haandel and Catunda, 1997). Anaerobic treatment can be carried out with technically simple setups, at any scale, and at almost any place. It produces a small amount of excess, well stabilized sludge, and energy can be recovered in the form of biogas. The process can be carried out in both centralized and decentralized modes, and the latter application can lead to significant savings in investment costs of sewerage systems. However, while anaerobic processes have gained popularity over the past decade, skepticism related to their application for domestic wastewater treatment

remains widespread.

Within the spectrum of anaerobic treatment technologies, the upflow anaerobic sludge blanket (UASB) reactor offers great promise, especially in a developing country context. It is a robust high-rate reactor system, generally without moving parts, limiting both capital and operating costs. The reactor retains a high amount of biomass in the form of dense granules or aggregates of microorganisms. Furthermore, good contact between biomass and wastewater is ensured due to mixing as a result of recirculation and biogas production. However, when the volumetric loading rate is below 1-2 kg-COD/m^3/d, biogas production is limited (Lettinga, et al., 1993). The expanded granular sludge bed (EGSB) reactor, a modified version of the UASB reactor, has a much higher upflow velocity, which would enable greater mixing when loading rates are low, but would cause washout of fine biomass particles that typically form after prolonged treatment at low loading rates (Aiyuk and Verstraete, 2004).

Another promising newer technology is the membrane bioreactor (MBR), a process involving membrane filtration combined with biofor enhanced treatment processes capable of logical treatment. The advantages offered by MBRs over conventional activated sludge process (ASP) include a small footprint and reduced sludge production. Submerged MBRs take only half the land area of a conventional ASP, and sludge production is similarly approximately halved. Since sewage sludge disposal contributes significantly to overall operating costs, there are significant potential benefits in reducing its production,

Using the bench scale submerged membrane system, treatment properties and temeprature effects have been investigated by Kashino, et al. (1996). Consequently, a high BOD removal rate (about 98%) was achieved. Although NH_4-N loading was high, a high NH_4-N removal rate (about 95%) was also obtained above 13℃. Ueda and Hata found that membrane filtration was able to be continued for the first 371 days without membrane washing and was still stable after the washing; average removal ratios of BOD, TOC, SS, total nitrogen, total phosphorous and coliform bacteria were 99, 93, 100, 79, 74% and 6-log units, respectively (Ueda and Hata, 1999). Chu (2005) et al. used an oxygen-limited membrane bioreactor seeded with anaerobic granular sludge, investigating for concurrent removal of organic

substances and nitrogen from synthetic domestic wastewaters. The oxygen addition rates were controlled at 3-4 kg $O_2/m^3/d$. The total COD removal efficiencies of more than 94% were achieved throughout the whole operation period. N was removed through the simultaneous nitrification and denitrification process that took place in the granular sludge bed, with the removal efficiencies of 80-91% at a hydraulic retention time of 15 h.

1.6 Objectives and procedures of this study

1.6.1 Objectives

The objectives of this study were intended to investigate the performance of the swim-bed technology on wastewater treatment and the biological mechanism in swim-bed reactor. Two systems packing with BF material were used in this study, one with settling tank (SSB) and one not (SSBR).

The objectives were met by performing the following activities

- Investigating the treatment potential and evaluating excess sludge production using SSB process for organic wastewater treatment.
- Experimentally demonstrating performances of the SSBR process treating high-strength wastewater with high-rate and low strength wastewater with short HRTs.

1.6.2 Procedures

The investigation performed to achieve the above objectives was divided into five parts.

Part I: Experiments were conducted to investigate the treatment potential of SSB process primarily with respect to removal of organic carbon from wastewater. Conducting loading-rate studies at various influent organic carbon levels and wastewater flow rates performed in this study. In addition, the influence of flow velocity in the biomass retention matrix was investigated and the contents of extracellular polymers in the sludge were determined.

Part II: Treatment performance and sludge characteristics of SSB process were further investigated in this part study. Observed sludge yields and apparent sludge yields were estimated respectively. Affecting factors on EPS production and biomass attachment were also investigated. The mechanism of considerable sludge reduction was discussed as the key point.

Part III: The wastewater treatment by using SSBR process was considered to meet the strict discharge standard. Performances of this system on treating high-strength wastewater were evaluated under different loadings. Apart from the contaminant removal, the investigation also focused on the sludge characteristics.

Part IV: Treatment capacity of the SSBR process for treating low strength wastewater was studied in this part. The comparisons of the treatment performance and sludge production with conventional activated sludge process (CAS) were conducted in the first phase of this part. Operational strategies for short HRT with sludge bulking were characterized in the second phase.

Part V: The performances of SSBR process on treating three kinds of low-strength wastewater were evaluated. Relationship between the characteristics of the feed water and the treatment results was explored.

This dissertation was divided into seven chapters, including this introduction part, five parts reporting the results of the experiments, and one chapter for the conclusions and the recommendations.

1.7 References

Abbassi B., Dullstein S., and Rabiger N. (2000) Minimization of excess sludge production by increase of oxygen concentration in activated sludge flocs; experimental and theoretical approach, Water Res. 34 (1), 139-146.

Aiyuk S.E. and Verstraete W. (2004) Sedimentological evolution in an UASB treating SYNTHES, a new representative synthetic sewage, at low loading rates, Biores. Technol., 93(3), 269-78.

Ardern E., and W.T. Lockett (1914) Methanogenesis in Thermophilic Biogas Reactors, Antonie van Leeuwenhoek, vol. 67, 91-102, The Netherlands.

Asselin C., Chicoine K., Parisien A., Riffon R., Ouellet B., Palacek K., amd Luedtke H. (2004) Pilot testing and full-scale implementation of the low sludge production (LSP) process.

Borgatti D.R., Brooks J.A., Carney K.B., Krupa D.P., and Kruzel J.P. (2000) High biosolids costs? Reduce volume, starve the bugs! Proceedings of the WEF Biosolids 2000 Annual Conference and Exhibit, Boston.

Carlson B.-L., Ericsson, T., Lovblad R., Persson S., and Simon O. (2000) The reconstruction of an aerated lagoon to a long-term aerated activated sludge (LAS) plant at Sodra Cell, Monsteras Kraft Pulp Mill. TAPPI Int. Environ. Conf. Proc., Denver 1, 363-370.

Corbitt R.A. (1989) Standard Handbook of Environmental Engineering, McGraw-Hill Publishing Company, Toronto, Ont.

Cornelissen E.R., Janse W. and Koning J. (2003) Wastewater treatment with the internal MEMBIOR, Desalination, 146, 463-466.

Chu L.B., Zhang X.W., Li X.H., and Yang F.L. (2005) Simultaneous removal of organic substances and nitrogen using a membrane bioreactor seeded with anaerobic granular sludge under oxygen-limited conditions, Desalination, 172(3), 271-280.

Cui R., Jahng D.J. (2004) Nitrogen control in AO process with recirculation of solubilized excess sludge, Water Res., 38:1159-1172.

EPA, 1999. Biosolids generation, use, and dispodal in the United States. EPA530-R-99-009, Washington, DC.

Ericsson T., Lunding B., and Simon, O. (1999) Redesign of an aerated lagoon to a long-term aerated activated sludge system with anoxic selector and very low phosphorus discharge, TAPPI International

Environmental Conference Proceedings, Nashville, vol. 3., 989-994.

Fox P., M.T. Suidan, and J.T. Pfeffer (1988) Anaerobic Treatment of Biologically Inhibitory Wastewater, Journal Water Pollution Control Federation, 60, 86-92.

Harila P. and Kaila J. (1995) Proven technology for secondary sludge disposal in a recovery boiler, TAPPI Engineering Conference Proceedings, Dallas. p. 293.

Hickey R.F., W.M. Wu, M.C. Viega, and R. June (1991) Start-up Operation, Monitoring, and Control of High-Rate Anaerobic Treatment Systems, Water Science and Technology, 24(8), 207-256.

Holler S. and Trosch W. (2001) Treatment of urban wastewater in a membrane bioreactor at high organic loading rates, Journal of Biotechnology, 9(2), 95-101.

Iza J. (1991) Fluidized Bed Reactors for Anaerobic Wastewater Treatment, Water Scicence and Technology, 24(8), 109-132.

Jeong H.S., Kim Y.H., Yeom S.H., Song B.K., and Lee S.I. (2005) Facilitated UASB granule formation using organic-inorganic hybrid polymers, Process biochemistry, 40(1), 89-94.

Kishino H., Ishida H., Iwabu H., and Nakano I. (1996) Domestic wastewater reuse using a submerged membrane bioreactor, Desalination, 106(1-3), 115-119.

Kindzierski W.B., M.R. Gray, P.M. Fedorak, and S.E. Hrudey (1992) Activated Carbon and Synthetic Resins as Support Material for Methanogenic Phenol-Degrading Consortia-Comparison of Surface Characteristics and Initial Colonization, Water Environment Research, 64, 786-795.

Lay J.J., Lee Y.J., Noike T. (1999) Feasibility of biological hydrogen production from organic fraction of municipal solid waste, Water Res., 33:2579-2586.

Lazarova V., J. Perera M., and P. Shields (2000) Application of Aerated Biofilters for Production of High Quality Water for Industrial Reuse in West Berlin, Water Science and Technology, 41, 417-425.

Lee K.S., Wu J.F., Lo Y.C., Lin P.J., and Chang J.S. (2004) Anaerobic hydrogen production with an efficient carrier-induced granular sludge bed bioreactor, Biotechnology and Bioengeering, 87(5), 648-657.

Lee N.M., and Welander T., (1996a) Use of protozoa and metazoa for decreasing sludge production in aerobic wastewater treatment. Biotechnol. Lett. 18 (4), 429-434.

Lee N.M., and Welander T. (1996b) Reducing sludge production in aerobic wastewater treatment through manipulation of the ecosystem. Water Res. 30 (8), 1781-1790.

Lee E.G.-H., Mueller J.C., and Walden C.C. (1976) Ultimate disposal of biological sludges—a novel

concept. Pulp Pap. Can. 77 (6), 50-56.

Lettinga G., De Man A., Van der Last A.R.M., Wiegant W., Van Kippenberg K., Frijns J., and Van Buren J.C.L. (1993) Anaerobic treatment of domestic sewage and wastewaters. Water Sci. Technol. 27(9), 67-73.

Liu L.L., Wang Z.P., Yao J., Sun X.J., and Cai W.M. (2005) Investigation on the formation and kinetics of glucose-fed aerobic granular sludge. Enzyme and Microbial Technology, 36(5-6), 712-716

Low E.W., and Chase H.A. (1999) Reducing production of excess biomass during wastewater treatment. Water Res. 33 (5), 1119-1132.

Martienssen M. (2000) Simultaneous catalytic detoxification and biodegradation of organic peroxides during the biofilm process, Water Research, 34(16), 3917-3926.

Metcalf L., and H.P. Eddy (1930) Sewerage and Sewage Disposal, A Textbook, 2d. ed., McGraw-Hill, New York.

Metcalf and Eddy Inc. (1991) Wastewater Engineering—Treatment Disposal and Reuse, McGraw-Hill, Inc., New York, NY.

Modell M., Larson J., and Sobczynski F. (1992) Supercritical water oxidation of pulp mill sludges, TAPPI J. 76 (6), 195-202.

Nichols W. (1992) Solid waste options, Am. Papermaker 55 (10), 41-43.

Park T.J., H. Lee K., S. Kim D., and W. Kim C. (1996) Petrochemical wastewater treatment with aerated submerged fixed-film reactor (ASFFR) under high organic loading rate, Water Scicence and Technology, 34(10), 9-16.

Park J.E., J.O. Kim, W.B. Lee, S.T. Lee, and J.J. Lee (1997) UASB performance in presence of algae and synthetic media, Water Science and Technology, 36(12), 125-133.

Puhakka J.A., Alavakeri M., and Shieh W.K. (1992a) Anaerobic treatment of Kraft Pulp-Mill waste activated sludge: gas production and solids reduction, Bioresour. Technol. 39, 61-68.

Puhakka J.A., Alavakeri M., Shieh W.K. (1992b) Anaerobic treatment of Kraft Pulp-Mill waste activated sludge: sludge dewaterability and filtrate quality, Bioresour. Technol. 39, 69-75.

Ratsak C.H., Kooi B.W., and van Verseveld H.W. (1994) Biomass reduction and mineralization increase due to the ciliate Tetrahymena Pyriformis grazing on the bacterium Pseudomonas Fluorescens, Water Sci. Technol. 29 (7), 119-128.

Rocher M., Roux G., Goma G., Pilas-Begue A., Louvel L., and Rols, J.L. (2001) Excess sludge reduction

in activated sludge processes by integrating biomass alkaline heat treatment, Water Sci. Technol. 44, 437-444.

Saby S., Djafer M., and Chen G. (2003) Effect of low ORP in anoxic sludge zone on excess sludge production in oxic-settlinganoxic activated sludge process, Water Res. 37 (1), 11-20.

Sallis P.J. and Uyanik S. (2003) Granule development in a split-feed anaerobic baffled reactor, Bioresource Technology, 89(3), 255-265.

Shanableh A. (2000) Production of useful organic matter from sludge hydrothermal treatment, Water Res. 34 (3), 945-951.

Spinosa L. (2001) Evolution of sewage sludge regulations in Europe, Water Sci. Tehcnol. 44, 1-8.

Spinosa L., Kempa E.S., Okuno N., and Vesilind P.A. (1994) Global sludge management: a status report and perspectice, Water Sci. Technol. 30 (8), 73-80.

Stensel H.D., Brenner R.C., Lee K.M., Melcer H., and Rackness K. (1988) Biological Aeraated Filter Evaluation, Journal Environmental engineering, 14, 65.

Stuart P., Kenny R., and Sointio J. (2000) A critical review of the low sludge production process for the pulp and paper industry, Proceedings of the 86th Annual Meeting, PAPTAC, Montreal. B209-B216.

Stutton P.M., and P.N. Mishra (1994) Activated Carbon Based Biological Fluidized Beds fro Contaminated Water and Wastewater Treatment: A State-of-the Art Review, Water Science and Technology, 29, 309-315.

US EPA, 1979. Process Design Manual for Sludge Treatment and Disposal.

Van Der Merwe-Botha, and T.J. Britz (1997) Combined pre-degradation and anaerobic digestion for the treatment of a baker's yeast factory effluent, Water Science and Technology, 36(6-7), 295-301.

Van Haandel A., Catunda P.F.C. (1997) Application and perspectives of anaerobic waste water treatment in Latin America; Sustainable rural environment and energy network, 5th FAO/SKEN workshop, anaerobic conversion for environmental protection, sanitation and reuse of residues, 24-27 March.

Ueda K. and Hata K. (1999) Domestic wastewater treatment by a submerged membrane bioreactor with gravitational filtration, Water Research, 33(12), 2888-2892.

Wang Y.T., M.T. Suidan, and J.T. Pfeffer (1984) Anaerobic Activated Carbon Filter for the Degradation of Polycyclic N. Aromatic Compounds, Journal Water Pollution Control Federation, 56, 1247-1253.

Wang Y.T., M.T. Suidan, and B.E. Rittman (1986) Anaerobic Treatment of Phenol by an Expanded Bed Reactor, Journal Water Pollution Control Federation, 58, 227-233.

WEF (2000) Aerobic Fixed-Growth Reactors, A special publication prepared by the aerobic fixed-growth reactors task force, Water Environment Federation, Alexandria, VA.

Yu Q., Xu H., and Williams P. (2003) Development of flexible fibre biofilm reactor for treatment of food processing wastewater, Environmental Technology, 24(4), 429-434.

Zhao Q.L., and Kugel G. (1997) Thermophilic/mesophilic digestion of sewage sludge and organic waste, J. Environ. Sci. Health 31, 2211-2231.

Zimmerman F.J., Diddams D.G. (1960) The Zimmerman process and its applications in the pulp and paper industry, TAPPI J. 43 (8), 710-715.

Chapter 2 Evaluation of single swim-bed process for wastewater treatment

2.1 Introduction

For aerobic wastewater treatment, fixed-bed attached-growth processes (biofilters) offer some advantages over suspended-growth processes such as reduced sensitivity to toxicity, co-existence of aerobic and anoxic metabolic activities and compactness. Newly developed fluidized-bed (or moving-bed) attached-growth processes have further demonstrated elimination of head losses with absence of clogging and channeling, improved mass transfer and the potential for up-grading existing treatment plants without constructing new tanks (Lazarova and Manem, 1994; Pastorelli, *et al.*, 1997a). Fluidized-bed reactors have thus been of considerable interest for removal of organic compounds from wastewater in recent years (Pastorelli, et al., 1997b; Loudidou and Zouboulis, 2001). In addition, Lazarova and Manem (1996), using gas-lift technology, introduced the circulating floating-bed reactor, which demonstrated a synergy between hydrodynamic characteristics and biological treatment performance countering the negative influence of solid media hold-up that can occur in fluidized-bed processes.

In this paper, swim-bed technology involving the novel biofringe material is presented for high-rate treatment of organic wastewater. The biofringe material allows for attachment of large amounts of biomass on a flexible matrix in a fixed position. By this approach, flexing of the matrix induced by wastewater flow creates a "swimming" motion that enhances mass transfer of nutrients to the attached growth (*i.e.*, biofilm). Thus, all the potential benefits of fluidized-bed reactors stated above are retained without dependence on hydrodynamic conditions to avoid settling or floating of the attachment medium and without the requirement of screens or traps to prevent washout. It is our hypothesis that high treatment efficiencies will be obtained using this method, with simple reactor design and uncomplicated operation.

The objective of this study is to investigate the treatment potential of a biofringe process

primarily with respect to removal of organic carbon from wastewater. This objective was met by conducting loading-rate studies at various influent organic carbon levels and wastewater flow rates. In addition, the influence of wastewater velocity in the biomass retention matrix was investigated and the contents of extracellular polymers in the sludge were determined.

2.2 Materials and methods

2.2.1 Reactors and operating conditions

The two reactors used in this study were constructed of acryl resin, each having downdraft and updraft sections in a parallel upright arrangement as shown in Fig. 2-1. The mid-sized reactor had a cross-sectional plan with downdraft and updraft sections of 100×100 mm and 100×25 mm, respectively, and a height to effluent port of 630 mm for a total liquid volume of 7.7 L. The larger reactor had downdraft and updraft sections of 96×100 mm and 100×36 mm, respectively, and a height to effluent port of 1640 mm for a liquid volume of 21.6 L. Both reactors had clear zones of approximately 70 mm at the bottom and 30 mm at the top (below and above the biofringe reaction zone in the downdraft section).

To each reactor, influent was introduced deeply within the updraft section using a peristaltic pump. Air was also introduced near the base of the updraft section, which served to mix and oxygenate the wastewater while circulating it through the reactor. Tracer studies using polyvinyl alcohol beads (dia., 4 mm; s.g., 1.025) were used to establish a correlation between airflow rate (easily monitored using a gauge) and water flow velocity in the narrow updraft section, which was then used to estimate (by continuity) nominal average water velocities in the reaction zone of the downdraft section. Tracer studies using nitrate were conducted prior to biomass input to evaluate reactor flow behaviors. Even at relatively low airflow rates of 1 to 5 L/min (water velocities, ca. 5 to 10 cm/s), the breakthrough of continuous tracer input yielded Ct/Co values of 0.60 at the hydraulic retention time (HRT), indicating nearly complete-mix conditions (Ct/Co is 0.63 for complete mix). Over most of the study, a small plastic shield was placed over the effluent port, the use of which resulted in a slight reduction

of Ct/Co to about 0.5. The shield was not used during the final three weeks of operation for both reactors (see Results and Discussion).

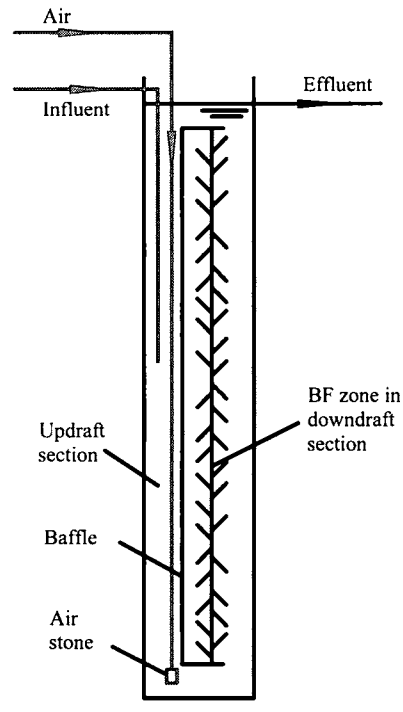


Fig. 2- 1 Cross-sectional schematic showing the basic configuration for the 7.7 L and 21.6 L reactors; See this chapter for detailed dimensions

The influent solution consisted of a fish paste and peptone mixture prepared as a stock solution at 40 and 60 g-COD/L, respectively, and diluted with tap water to obtain desired influent concentrations. The 5-d biochemical oxygen demand (BOD) was 74% of the chemical oxygen demand (COD) for influent solutions. A buffer solution consisting of $KHCO_3$ was also added to a final concentration of 125 mg/L (alkalinity addition, 62 mg $CaCO_3$/L). The tap water used for mixing the influent was of groundwater origins and contained 19 mg Na^+, 6 mg K^+, 20 mg Ca^{2+}, 7 mg Mg^{2+}, 3 mg NO_3^- and 24 mg SO_4^{2-} per liter. The naturally occurring alkalinity was 70 mg $CaCO_3$/L (increased to *ca.* 130 mg $CaCO_3$/L with addition of the buffer) and total hardness was 70 mg $CaCO_3$/L. Operation was conducted at room temperature (*ca.* 23°C) and relatively dark conditions were maintained (apart from sampling and inspection).

2.2.2 Seed sludge

The reactors were initially seeded using activated sludge from a lab-scale fill-and-draw batch reactor. The synthetic medium used for the development and maintenance of the seed sludge was essentially the same as that used in this chapter.

2.2.3 Extraction and analysis of extracellular polymers

Extraction of extracellular polymers from sludge was done by either the autoclaving or alkaline-washing method. For both methods, a 10-mL representative sample of settled unwashed sludge was collected and thick slimy components were minced by tearing or cutting. For the autoclaving method (after Bhatti, *et al.*, 1993), the sludge sample was messed up to 100 mL with tap water in a glass beaker and mixed by swirling. The sample was then autoclaved at 120℃ for 30 min. Bhatti, *et al.* (1993) determined this time to be optimal, with longer times causing lysis of cells resulting in erroneously high yields (including intracellular polymers). The sample was then cooled to room temperature with occasional mixing prior to centrifuging at 3,000×*g* for 30 min. 50 mL of the supernatant was then collected and mixed with about 20 mL of water prior to checking the pH. Though not necessary in these experiments, if the pH is found to vary outside of a neutral range (pH 6 to 8), adjustments should be made with dilute HCl and NaOH solutions. Finally, the solution was messed up to 100 mL with water.

For the alkaline-washing method, the sludge sample was messed up to 100 mL with tap water and a NaOH solution in a glass beaker to a final NaOH concentration of 2 N (*e.g.*, 67 mL of 3 N NaOH into 33 mL of diluted sample). The sample was then mixed by gentle stirring (*ca.* 100 rpm) for 2 h prior to centrifuging at 3,000×*g* for 30 min. 50 mL of the supernatant was then collected and mixed with about 20 mL of water prior to adjusting the pH. A 1+1 HCl solution was used to bring the pH down to a neutral value. Over 15 mL of the HCl solution was required, which was added slowly with stirring to avoid heating the sample. Finally, the solution was messed up to 100 mL with water. For both procedures, above, the extracted extracellular polymers in the final 100-mL solution represented the contribution of only

one-half (*i.e.*, 5 mL) the original sludge sample.

In the extracted solutions, proteins were measured using the method of Lowry *et al.* (1951) and carbohydrates by the method of Dubois *et al.* (1956). Nucleic acids (combined RNA and DNA) were estimated by the UV absorption method (Experimental Guidelines for Biotechnology, 1992) using the following equation:

$$\text{Nucleic acids (g/L)} = 30.98\, A \,/\, (10{,}000 \cdot (0.09)\, b) \qquad (2\text{-}1)$$

where, 30.98 is the gram molecular weight of phosphorous, A is the absorbance of the sample solution at 260 nm, 10,000 is the constant of proportionality (*absorbtivity*) of phosphorous in nucleic-acid form (average of the RNA and DNA components), 0.09 is the weight fraction of phosphorous in nucleic acids and b is the absorbance light path (1.0 cm in this study).

2.2.4 Other analytical methods

COD was measured by the closed reflux colorimetric method according to *Standard Methods* (5220 D; APHA *et al.*, 1995), with prior centrifugation of effluent samples at 1,000×g for 15 min to remove undissolved components. BOD was measured using a respirometer (BODTrack; Hach Co., Ltd., Loveland, CO). The pH level was measured by the electrometric method using a pH meter (IM-22P; TOA Electronics, Ltd., Tokyo, Japan). Dissolved oxygen (DO) was measured using a DO meter (OM-51; Horiba, Ltd., Kyoto, Japan).

Ammonium (NH_4^+) was quantified by the phenate method as described by Kanda (1995), with prior centrifugation of effluent samples at 1 000 × g for 15 min. Nitrite (NO_2^-), nitrate (NO_3^-) and sulfate (SO_4^{2-}) ions were measured using an ion analyzer (IA-100 system; TOA Electronics, Ltd., Tokyo, Japan), with pretreatment by a 0.45-μm syringe filter for effluent samples. Total nitrogen (total-N) was determined by the persulfate method according to *Standard Methods* (4500-N_{org} D; APHA *et al.*, 1995) with the digestion time extended to 60-min (which was found to be necessary to assure complete digestion of organic

compounds). Effluent Total-N was determined on well-settled samples, thus not reflecting nitrogen in the biomass or sludge (though not excluding the soluble organic component). By the persulfate method all nitrogen is oxidized to NO_3^-, which was measured using the UV spectrophotometric screening method according to *Standard Methods* (4500-NO_3^- B; APHA *et al.*, 1995) (the use of an ion analyzer was ineffective in this case due to interference from compounds in the persulfate solution). The suspended solids (SS) content was determined according to *Standard Methods* (2540 D; APHA *et al.*, 1995). The total sludge content was estimated as mixed-liquor suspended solids (MLSS) and biomass as mixed-liquor volatile suspended solids (MLVSS). For the determination of MLSS, a sludge sample of know volume was washed twice by centrifuging at 1,000×*g* for 15 min, decanting and resuspending in deionized water and then dried to a constant weight at 105℃ (with cooling under desiccation). MLVSS and mineral (ash) contents of MLSS samples were determined following ignition at 550□ for 1 h.

2.3 Results and discussion

2.3.1 Reactor startup and biomass attachment

For startup of the 7.7-L reactor, 15.4 g of activated sludge was placed in the reactor with tap water for an initial total sludge concentration of 2.0 g/L and airflow was set at 2 L/min to circulate the solution through the reaction zone at a velocity of 7 cm/sec. Attachment of sludge to the biofringe material (determined by the decrease of total sludge in solution) proceeded as shown in Fig. 2-2. The attachment of 9.7 g of sludge during a 30-h period amounted to 18.6 g/m of biofringe support filament for the 7.7-L reactor. For the 21.6-L reactor, an initial total sludge concentration of 1.64 g/L was introduced with approximately the same circulation conditions. Following 27 h, 13.0 g of sludge had attached, which amounted to only 8.4 g/m of support filament.

Following the sludge attachment periods, above, influent was started with an initial COD concentration of approximately 700 mg/L and influent flow rates were set at 0.4 L/h and 0.8 L/h for the 7.7-L and 21.6-L reactors, respectively. During the first week, the airflow was

maintained at a relatively low rate of 2 L/min and sludge solutions became clear within 2 d for the 7.7-L reactor and 4 d for the 21.6-L reactor. Following the first week, the reactors were considered acclimated and airflow rates were increased to 5 L/min and the influent flow rates were increased to 0.6 L/h and 1.7 L/h, for the 7.7-L and 21.6-L reactors, respectively.

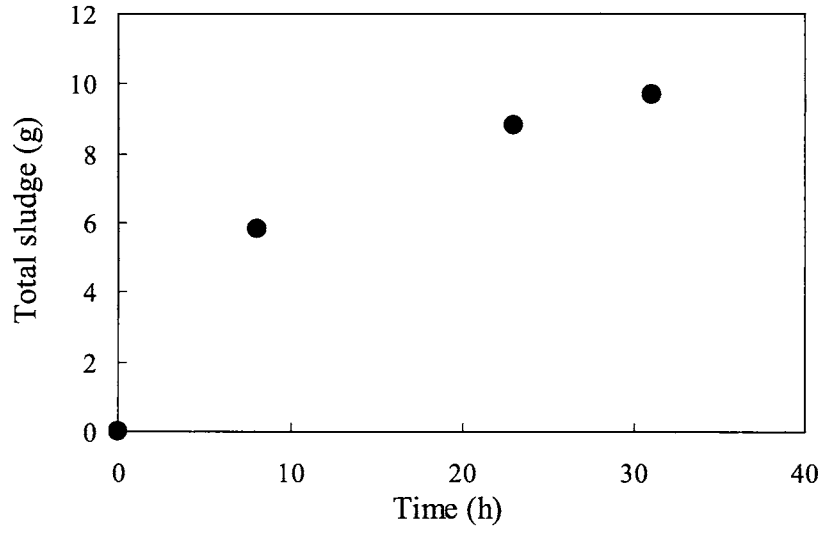


Fig. 2- 2 Time course of total sludge attachment to the BF material in the 7.7 L reactor

2.3.2 Treatment performance for COD removal

The degradation of aggregate organic constituents in the 7.7-L reactors was evaluated by COD losses as shown in Figure 2-3. Over the first 40 d of operation, the COD removal rates for the 7.7-L reactor (Figure 2-3) were very consistent; however, during this period pronounced changes in the appearance of the biofilm and the amount of SS in the reactor occurred. After 14 d of operation, a heavy flocculent growth appeared, producing an SS in the reactor of nearly 500 mg/L. The flocs consisted of large (3 to 5 mm diameter) snow flake-like particles and were easily retained using a small plastic shield over the effluent port as evidence by the low levels of SS in the effluent. During this period of flocculent growth, a microscopic assay of the sludge in the 7.7-L reactor revealed large numbers of protozoa in a non-filamentous biomass. Following day 24, however, the floc content greatly diminished and in the following days the attached growth became increasingly thick and pendulous. On day 32, a microscopic assay revealed an almost entirely filamentous growth with large

numbers of protozoa and by day 40 the 7.7-L reactor was nearly packed with sludge. On day 46, the biofringe material was drawn from the 7.7-L reactor and a 5.0-L settled bed volume of thick slimy sludge was collected with a total sludge concentration of 12.2 g/L (biomass, 11.4 g/L). Accounting for the original mass of attached sludge and SS lost in the effluent and a treatment efficiency of 80%, a rough estimate of 0.15 g biomass produced per g of COD removed (Y_{obs}) is made. The biofringe matrix, retaining only slight amounts of attached biomass, was then returned to the 7.7-L reactor and operation was restarted with twice the prior influent COD level in an effort to restrict growth of filamentous organisms and the water circulation velocity was doubled to enhance detachment and removal of excess growth. Within 8 d, the attached growth appeared to be at a normal level and testing was resumed (day 52, Fig. 2-3).

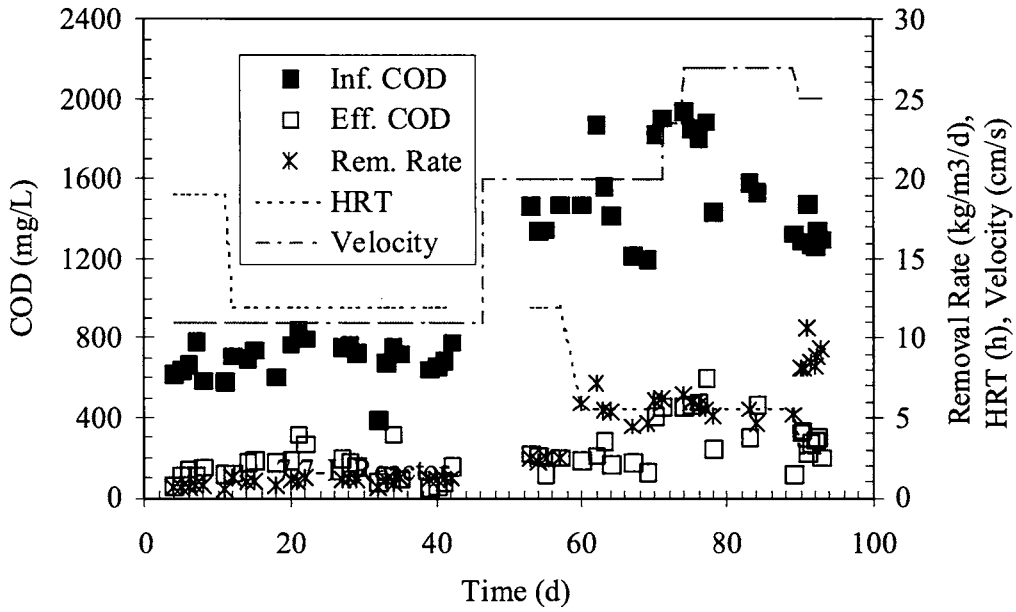


Fig. 2- 3 Time courses of COD concentrations and COD removal rates versus hydraulic retention time and water flow velocity in BF zone for the 7.7 L reactor

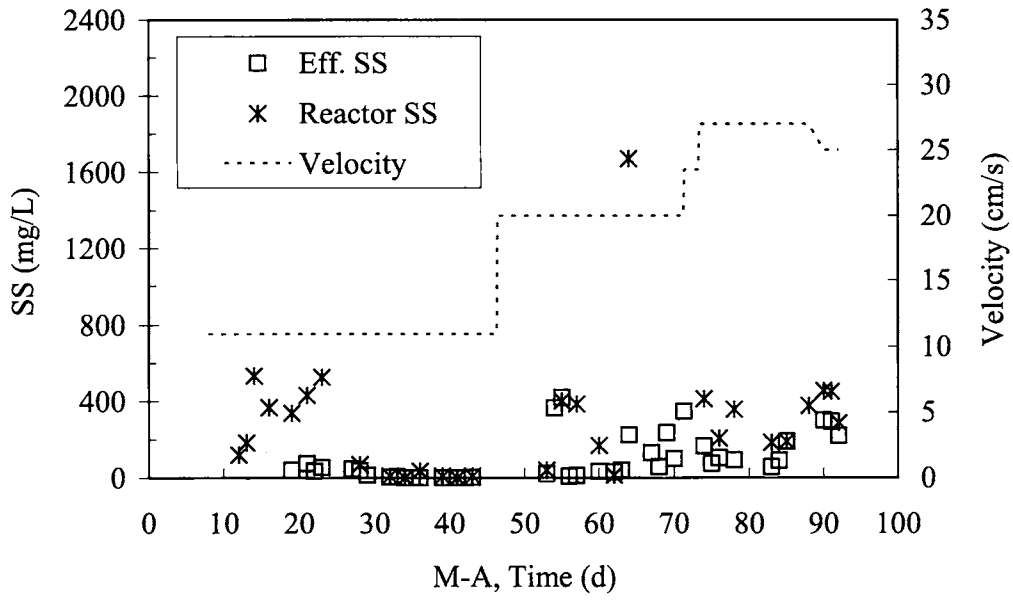


Fig. 2- 4 Time courses of SS level versus flow velocity in BF zone for the 7.7 L reactor

During a parallel period of operation, the 21.6-L reactor followed a similar pattern of performance with very consistent COD removal rates. Also, an early (day 12) microscopic assay revealed an abundance of protozoa with very little evidence of filamentous growth. By day 14, a heavy flocculent growth (*ca.* 4 mm diameter snowflake-like particles) appeared, producing a SS in the reactor of 200 mg/L (Fig. 2-5), which persisted for about 10 days. As with the other reactor, abatement of the flocculent biomass was linked with the appearance of a thick slimy biofilm and a microscopic assay on day 25 showed an almost entirely filamentous growth, though with fewer protozoa. With the taller 21.6-L reactor, however, crowding due to sludge accumulation did not occur and the slimy filamentous biomass appeared to function well by flexing with the water flow and allowing for effective substrate-biomass contact, especially with the increase in circulation velocity from 15 to 20 cm/s on day 44.

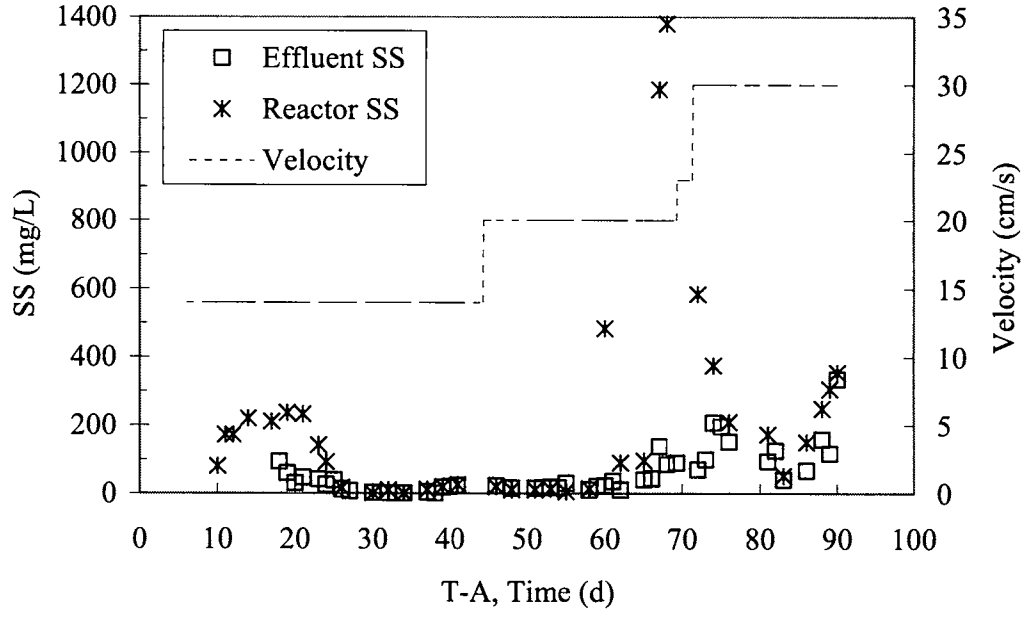


Fig. 2- 5 Time courses of SS level versus flow velocity in BF zone for the 21.6 L reactor

With subsequent increases in COD loading rates, removal rates increased in a linear manner over the entire range of testing for both reactors as shown in Fig. 2-6. Removal efficiencies were consistently at approximately 80% and the highest loading rates of 11 to 12 $kg/m^3/d$ (removal rates, 9 to 10 $kg/m^3/d$), at an HRT of only 3 h, were well within a range of high-rate industrial application. However, the SS contents of both reactors again increased, with extremely high flocculent SS levels appearing following day 60; thus, on day 70 the shields over the effluent ports were removed and the water circulation velocities were again increased. A free flow of solids in the narrow reactors was thus allowed and, as shown in Fig. 2-4 and 2-5, the reactor SS and effluent SS levels remained nearly the same for the remainder of the study.

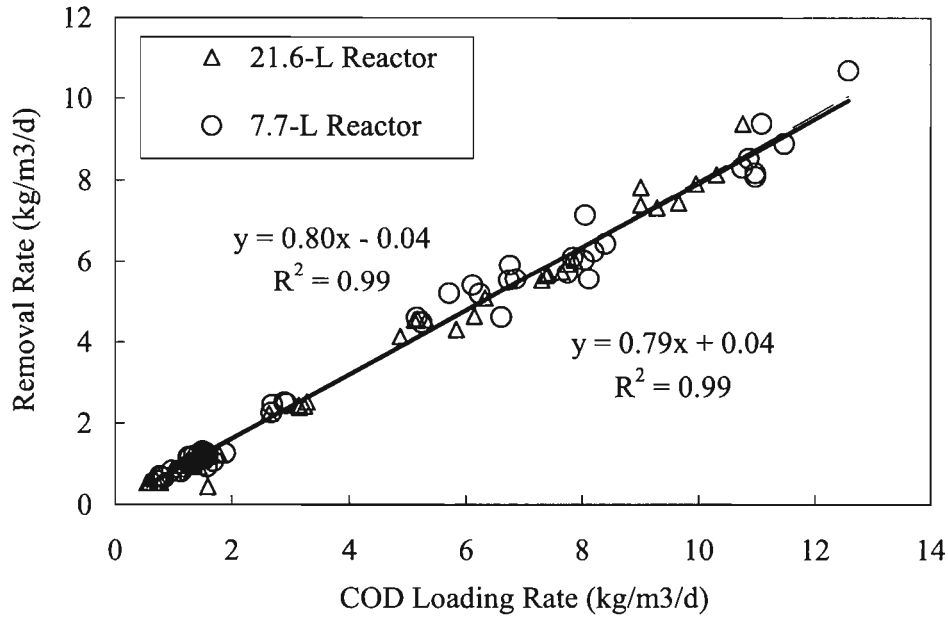
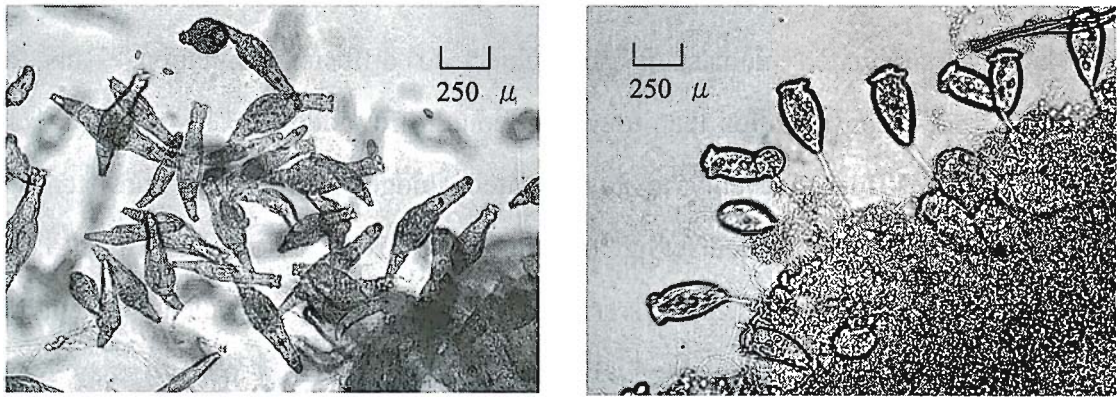


Fig. 2- 6 Linear relation between COD removal rate and loading rate



(a) (b)

Fig. 2- 7 Microscopic photographs of activated sludge on BF material showing protozoa:

(a) *Philodina* sp. (b) *Vorticella* sp.

At the end of the study, microscopic assays revealed a biomass with very few protozoa and only a few pockets of relatively short filamentous growth in the sludge of the 7.7-L reactor. The sludge of the 21.6-L reactor was basically the same with, however, slightly higher levels of protozoa and filament, perhaps because it had not been cleaned out and restarted during the study. Nonetheless, these results suggest that filamentous growth was avoidable at the higher loading rates (*ca.* 8 kg/m^3/d). The total settled-bed volume of sludge collected from the 7.7-L

reactor was 5.5-L, which was corroborative of the amount collected when the same reactor was emptied earlier in the study. The total biomass retrieved was 69.0 g dry weight (ash, 6.3%), which is 133 g/m of biofringe support filament or 13.3 g/L with respect to the reaction zone of the 7.7-L reactor. From the 21.6-L reactor, a 15.0-L bed of sludge was collected with a total biomass of 175.8 g (ash, 6.2%), which is 114 g/m of biofringe support or 11.4 g/L with respect to the reaction zone.

The occasional operational irregularities associated with changes in biomass solids during the study were influenced largely by the narrow reactor configurations. It is interesting, though, that COD removal efficiencies were not impacted by these variations, especially when high reactor SS levels occurred. Yu *et al.* (2001) demonstrated that non-attached biomass plays a larger treatment role than its portion of the total reactor biomass (including attached growth) would account for. This would especially be true for the easily degradable substrate in this study because suspended biomass is not subject to diffusion limitation. Thus, a less restraining process configuration than used here could offer additional benefit. For example, biofringe elements spaced evenly in an activated sludge basin would consistently have maximized substrate-biomass contact due to unhindered flexing action and be able to obtain a steady-state equilibrium between the sessile and planktonic (or flocculent) phenotypes. In addition, with lower organic loadings, filamentous biofilm (if it occurs) may not be a drawback in a treatment system with an attached-growth process of this nature that retains suspended sludge by filtration rather than sedimentation and return. These and other applications are avenues of further study.

2.3.3 Nitrogen transformations

A degradation of nitrogenous compounds across the 7.7-L and 21.6-L reactors is shown in Tables 2-1 and 2-2, respectively. Total-N was not determined for the sampling events with low COD loadings; by calculation, though, the influent total-N concentrations can be estimated to be approximately 70 mg N/L in those cases. The low levels of influent oxidized nitrogen (NO_2^-+NO_3^-) were due to the naturally occurring NO_3^- in the tap water used for

mixing, confirming that the influent solution was contributing nitrogen only in organic form with relatively low levels of NH_4^+ resulting from hydrolysis.

Table 2- 1 Nitrogen components in influent and effluent solutions for the 7.7 L reactor with respect to influent COD concentration and loading rate

Influent COD (mg/L)	COD load ($kg/m^3/d$)	Influent (mg-N/L)			Effluent (mg-N/L)		
		Total-N[a]	NH_4^+	NO_2^-+NO_3^-	Total-N[a]	NH_4^+	NO_2^-+NO_3^-
680	1.4	-	1.2	<1	-	25.4	22.8
640	1.3	-	8.8	<1	-	38.7	6.4
690	1.4	-	5.7	<1	-	8.5	12.6
780	1.5	-	1.3	<1	-	7.2	16.2
1570	6.7	154	21.5	<1	115	90.7	<1
1580	6.8	134	82.4	<1	99	96.6	<1

a) Total-N represents combined NO_2^-, NO_3^-, NH_4^+ and soluble organic-bound forms of nitrogen, but not biomass. Analyses were not performed on relative early term samples with low COD loadings; however, the influent values can be estimated to be ca. 70 mg-N/L

Evidence of nitrification activity only occurred under low COD loadings (1.6 $kg/m^3/d$ or less) and NO_2^- was the predominant form of oxidized nitrogen being produced. During the periods of high COD loadings when oxidized nitrogen did not accumulate in the effluents, the DO levels were 7.2 (s.d., 0.7) mg/L and 6.8 (s.d., 0.5) mg/L in the 7.7-L and 21.6-L reactors, respectively, indicating that enhanced growth of heterotrophs was inhibiting nitrification rather than oxygen limitation. In addition, effluent TN concentrations were approximately 25% below the influent levels, which could reflect nitrogen losses via cycling in aerobic and anoxic zones of the thick biofilm; however, it would more likely be due to assimilation considering the lack of evidence for nitrification.

Table 2- 2 Nitrogen components in influent and effluent solutions for the 21.6 L reactor with respect to influent COD concentration and loading rate

Influent COD (mg/L)	COD load ($kg/m^3/d$)	Influent (mg-N/L)			Effluent (mg-N/L)		
		Total-N[a]	NH_4^+	$NO_2^-+NO_3^-$	Total-N[a]	NH_4^+	$NO_2^-+NO_3^-$
730	1.5	-	2.2	<1	-	15.9	20.5
740	1.5	-	1.4	<1	-	22.6	18.4
740	1.5	-	2.0	<1	-	3.6	25.4
800	1.6	-	2.8	<1	-	5.3	35.7
750	3.2	75	20.4	<1	53	51.7	<1
1580	6.3	134	82.4	<1	107	101	<1

a) See footnote with Table 2-1

2.3.4 Changes in pH

Influent pH for both reactors was near neutral throughout the study: 7.2 (s.d., 0.2) and 7.1 (s.d., 0.2) for the 7.7-L and 21.6-L reactors, respectively. During the first four weeks of operation when the influent COD concentration was relatively low (ca., 700 mg/L), the change in pH across the 7.7-L reactor was insignificant (increase of 0.2 (s.d., 0.3; n, 18)). During the following three weeks, under the same testing conditions, there was a significant increase in pH of 0.8 (s.d., 0.2; n, 14), which may have indicated a maturation in the biomass community; however, treatment performance did not change with the exception that SS in the reactor was greatly decreased. After increasing the influent COD (to ca., 1 500 mg/L, Fig.2-3), a more pronounced increase in pH of 1.2 (s.d., 0.2; n, 32) was evidenced over the remaining seven weeks.

For the 21.6-L reactor, during the first two months of operation when the influent COD was low (ca. 700 mg/L), the pH change across the reactor was less stable (average pH increase of 0.2 (s.d., 0.6; n, 48). During the last week of this period, though, when the flow rate was doubled (HRT reduced from 12 to 6 hr) a steady increase in pH of 0.8 (s.d., 0.2; n, 5) occurred. While changes in pH would intrinsically be a function of substrate utilization on a concentration rather than a rate basis, this event coincided with a period of extremely high SS

accumulation in the reactor making interpretation of these observations difficult. Subsequently, with an increase in the influent COD concentration (to *ca.* 1 500 mg/L), a pH increase of 1.0 (s.d., 0.2; n, 22) was maintained for the remainder of the study coupled with stable COD removals, though with considerable variation in SS levels.

2.3.5 Characterization of extracellular polymers

Table 2- 3 Compositions of extracellular polymers in biofilm samples. All values are reported as percent (%) of dry biomass (i.e., MLVSS). Ash contents were approximately 6% for all samples

	7.7 L reactor, day 47 (filamentous)	7.7 L reactor, day 47 (filamentous)	21.6 L reactor, day 94 (non-filamentous)	21.6 L reactor, day 96[a] (non-filamentous)
Extraction method	Autoclaving	Alkaline-washing	Alkaline-washing	Alkaline-washing
Nucleic acids	6.7	7.7	13.3	12.1
Carbohydrates	8.5	9.0	10.1	9.5
Proteins	30.8	35.3	50.3	48.8
Total	46.0	52.0	73.7	70.4

a) The 7.7 L reactor had been opened and cleaned and restarted on day 47, thus a 49-d growth period is represented here.

Extractions of extracellular polymers from biofilms of relatively early term, extremely filamentous growths and late term, non-filamentous growths were performed and analyzed for nucleic acids, carbohydrates and proteins. As shown in Table 2-3, the results were very consistent for the early term, filamentous samples analyzed by different methods and also for the late term, non-filamentous samples drawn from different reactors. While carbohydrates did not change significantly among all the samples, nucleic acids and proteins were considerably higher, for the late term, non-filamentous samples. Because the 7.7-L reactor had been cleaned and restarted mid-way through the study (just following the 47-d sample), the extended time of enrichment or maturation of the sludges would not be the reason for the noted difference. Thus, the possibility that these differences are characteristic of the different types of growths dominating in the biofilms is considered.

Extracellular polymers are known to exist in most biologically active environments and to aid in microbial aggregation or adhesion (Harris and Mitchell, 1973) as with the attached growth on the flexible biofringe matrix. The total extracellular polymers in the attached growth of this study (46 to 73%, Table 3) are considerably higher than values reported for anaerobic granular sludges (10 to 15%, Fukuzaki, *et al.*, 1991; 12 to 24%, Bhatti, *et al.*, 1995), a denitrifying granular sludge (7.8%, Bhatti, *et al.*, 2001) and various activated sludges (1 to 10%, Kakii, *et al.*, 1989). However, the protein contents reported here for the filamentous samples are comparable to a range of 25 to 35% reported by Lazarova and Manem (1996) for the attached growth of a circulating floating bed reactor. Thus, it appears higher levels of extracellular polymers-and proteins in particular-are a characteristic of biofilms (*i.e.*, growth on an attachment medium) as compared to flocculent or even granular sludges.

2.4 Summary and conclusions

Swim-bed technology using the novel biofringe attachment material demonstrated effective treatment of organic wastewater with 80% COD removal efficiencies at volumetric loadings of up to 12 kg/m^3/d and hydraulic retention times as short as 3 h. The biofringe material allowed for attachment of large amounts of biomass in a matrix that flexes with the wastewater flow, thus providing a high degree of contaminant-biomass contact in a fully retainable biofilm without the hydrodynamic difficulties associated with floating-bed media or the clogging associated with fixed-bed media. As much as 133 g of biomass per meter of biofringe support matrix was retained or 13.3 g/L with respect to the biofringe retention zone. Limited evidence for nitrification only occurred at low loading rates (1.6 kg/m^3/d or lower) due to interference from heterotrophic growth at higher loadings. In addition, filamentous growth was very heavy at the lower loading rates, but appeared to be avoidable at COD loading rates of 8 kg/m^3/d or greater. Apart from difficulties associated with the narrow reactor configuration used in this study, though, it appeared that filamentous growth is not necessarily undesirable for this process and its occurrence did not impact treatment efficiency. While treatment focused on industrial level applications in this study, the possibility exists for using this technology for domestic or other levels treatment.

2.5 References

APHA, AWWA, and WEF: Standard Methods for the Examination of Water and Wastewater, 19th Ed. American Public Health Association, Washington, D. C. (1995).

Bhatti Z.I., Furukawa K., and Fujita M. (1993) Qualitative changes in upflow anaerobic sludge blanket granules during extracellular polymer extraction by autoclaving. Japanese Journal of Water Treatment Biology 29 (2), 41-49.

Bhatti Z.I., Furukawa K., and Fujita M. (1995) Comparative composition and characteristics of methanogenic granular sludges treating industrial wastes under different conditions. Journal of Fermentation and Bioengineering 79, 273-280.

Bhatti Z.I., Sumida K., Rouse J.D., and Furukawa K. (2001) Characterization of denitrifying granular sludge treating soft groundwater in an upflow sludge-blanket reactor. Journal of Bioscience and Bioengineering 91 (4), 373-377.

Dubois M., Gilles K.A., Hamilton J.K., Rebers P.A., and Smith F. (1956) Colorimetric method for determination of sugars and related substances. Analytical Chemistry 28 (3), 350-356.

Experimental Guidelines for Biotechnology: Society of Fermentation and Bioengineering, Osaka, Japan (1992). 98-99 (in Japanese).

Fukuzaki S., Chang Y., Nishio N., and Nagai S. (1991) Characteristics of granular methonogenic sludge grown on lactate in a UASB reactor. Journal of Fermentation and Bioengineering 72, 465-472.

Harris R.H. and Mitchell R. (1973) The role of polymers in microbial aggregation. Annual Review of Microbiology 27, 27-50.

Kakii K., Nakatani K., Shirakashi T., and Kuriyama M. (1989) Extracellular polymers in relation to settling properties of activated sludge. Journal of Fermentation and Bioengineering 68, 365-370.

Kanda J. (1995) Determination of Ammonium in Seawater Based on the Indophenol Reaction with o-Phenylphenol (OPP). Water Research 29, 2746-2750.

Lazarova V. and Manem J. (1994) Advances in biofilm aerobic reactors ensuring effective biofilm control. Water Science and Technology 29 (10/11), 319-327.

Lazarova V. and Manem J. (1996) An innovative process for waste water treatment: the circulating floating bed reactor. Water Science and Technology 34 (9), 89-99.

Loukidou M.X. and Zouboulis A.I. (2001). Comparison of two biological treatment processes using

attached-growth biomass for sanitary landfill leachate treatment. Environmental Pollution 111, 273-281.

Lowry O.H., Rosebrough N.J., Farr A.L., and Randall R. J. (1951) Protein measurement with the folin phenol reagent. Journal of Biological Chemistry 193, 265-275.

Pastorelli G., Andreottola G., Canziani R., Darriulat C., de Fraja Frangipane E., and Rozzi, A. (1997a) Organic carbon and nitrogen removal in moving-bed biofilm reactors. Water Science and Technology 35 (6), 91-99.

Pastorelli G., Andreottola G., Canziani R., de Fraja Frangipane E., De Pascalis F., Gurrieri G. and Rozzi A. (1997b) Pilot-plant experiments with moving-bed biofilm reactors. Water Science and Technology 36 (1), 43-50.

Yu H., Kim B.J., and Rittmann B. E. (2001) Contributions of biofilm versus suspended bacteria in an aerobic circulating-bed biofilm reactor. Water Science and Technology 43 (1), 303-310.

Chapter 3 Excess sludge reduction and sludge characteristics by using single swim-bed technology treating high-rate wastewater

3.1 Introduction

The activated sludge process is widely used for wastewater treatment. It has been developed for about one-hundred years, but many problems still exist. Especially the generation of a large amount of excess sludge that has to be wasted was a big problem. The cost for excess sludge treatment has been estimated to be 50-60% of the total expense of wastewater treatment plants. Accordingly, there is much interest in biological treatment methods with less excess sludge production.

Several strategies have been suggested for the purpose of sludge reduction. One is sludge disintegration. Cell lysis will release cell contents into the medium, thus providing an autochthonous substrate that contributes to the organic loading. Components of this substrate are reused in microbial metabolism while a portion of the carbon is liberated as products of respiration, which results in an overall reduction in biomass production. Since the biomass growth that subsequently occurs on this autochthonous substrate cannot be distinguished from growth on the original organic substrate, this strategy is also termed as cryptic growth (Mason, et al., 1986). Ozonation is another well known method of sludge disintegration. Yusui and Shibata (Yasui and Shibata, 1994) developed a process consisting of sludge ozonation and a biodegradation stages for treating municipal and industrial wastewaters. No excess sludge was withdrawn and no significant accumulation of inorganic solids occurred in the aeration tank when operated under the optimal rates. Kamiya and Hirotsuji (Kamiya and Hirotsuji, 1998) subsequently developed a new system combining both biological treatment and intermittent ozonation to reduce excess sludge production using less ozone, which also favorably controlled the sludge bulking. Results of their lab-scale experiments treating synthetic wastewater indicated that intermittent ozonation was preferred over continuous ozonation

because it reduced sludge production by 50% with only 30% of the ozone dose required for continuous ozonation and it resulted in improving the sludge settling characteristics (Egemen, et al., 1999; Egemen, et al., 2001; Deleris, et al., 2002; Ahn, et al., 2002). Thermal treatment is another sludge disintegration method. Furthermore, this process has also been developed in combination with thermal and chemical treatment for sludge reduction. In addition, it was found that alkaline treatment by NaOH addition combined with thermal treatment (pH10, 60□ for 20 min) was the most efficient sludge elimination process (Rocher, et al., 1999; Rocher, et al., 2001). However, most disintegration technologies have an unfavorable cost: benefit ratio and are not wildly regarded as economically viable (Weemaes and Verstraete, 1998).

The second technology for sludge reduction is uncoupling metabolism. Bacterial anabolism is coupled to catabolism of substrate through rate limiting respiration (Senez, 1962). However, uncoupled metabolism would occur if respiratory control did not exist and instead the biosynthetic processes was rate limiting. Therefore, excess free energy would be directed away from anabolism so that the production of biomass can be reduced. Uncoupled metabolism was observed under some conditions, such as in the presence of inhibitory compounds, heavy metals, excess energy uncoupling, abnormal temperatures, and limitation of nutrients (Liu and Tay, 2001; Stouthamer, 1979). Recently, many researchers have focused on sludge reduction induced by chemical uncoupling. Sludge production was reduced significantly, raging from 40% to 86.9%, by adding different chemical uncouplers to biological treatment (Chen, et al., 2000; Chen, et al., 2002; Yang, et al., 2003; Low, et al., 1998; Low, et al., 2000; Strand, et al., 1999;). However, the application of chemical uncouplers for sludge reduction may cause a reduction in COD removal, an increase in oxygen consumption and deterioration in activated sludge properties such as settling and dewatering. Most of chemical uncouplers tested are xenobiotic and potentially harmful to the environment. Thus, their application should be observed carefully and further research is needed on the environmental impacts of long-term use of chemical uncouplers (Wei, et al., 2003).

The third approach for sludge reduction is to maximum the sludge retention time while

increasing the numbers of predators within the biological treatment system. Microorganisms satisfy their maintenance energy requirement in preference to producing additional biomass, which has possible applications for sludge reduction during biological wastewater treatment. By increasing the biomass concentration, it would theoretically be possible to reach a situation in which amount of potential energy provided equals the maintenance demand. The complete sludge retention had little impact on wastewater treatment performance except a slight increase in the sludge inorganic fraction (Muller, et al., 1995).

The protozoa and metazoa can not only predate on the bacteria but also degrade the remaining substrate. During energy transfer from low to high trophic levels, energy is lost due to inefficiencies during biomass conversion. In the past, protozoa and metazoa were usually used as important indicators of process performances and efficiencies in biological wastewater treatment processes. Recently, many researchers have focused on sludge reduction induced by grazing on bacteria (Wei, et al., 2003). The minimization of sludge production by protozoa and metazoa predation on bacteria was investigated in two two-stage systems treating synthetic wastewaters, in which the second stage was a suspended-carrier biofilm reactor (Lee and Welander, 1996a; Welander and Lee, 1994). The sludge production in the predator stage was significantly decreased by 60-80% compared with that in the bacterial stage. The total sludge yield was 0.05 g-TSS/g-$COD_{removed}$ in the two-stage system fed acetic acid, whereas it was 0.17 g-TSS/g-$COD_{removed}$ in the other two-stage system fed methanol. Further study was carried out to investigate sludge reduction with this two-stage system treating different pulp and paper industry wastewaters (the second stage designed as activated sludge and biofilm reactors, respectively) (Lee and Welander, 1996b). Results of this study showed that the sludge yield (0.01-0.23 g-TSS/g-$COD_{removed}$) of the two-stage system was obviously lower than that (0.2-0.4 g-TSS/$COD_{removed}$) of the conventional activated sludge processes treating the same wastewater.

Swim-bed technology involving the novel acryl fiber material, biofringe (BF), is a new concept for the treatment of organic wastewater. The BF material allows for attachment of large amounts of biomass on a flexible matrix in a fixed position. By this approach, flexing of

the matrix induced by wastewater flow creates a "swimming" motion that enhances mass transfer of nutrients to the attached growth (*i.e.*, biofilm). Thus, all the potential benefits of fluidized-bed reactors are retained. In addition, it was our hypotheses that sludge production in a BF process would be reduced due to the formation of a longer food chain. The abundance of protozoa and metazoa in a BF process should enhance the sludge reduction. The objective of this study was to evaluate the performance of a single BF reactor for excess sludge reduction. The characteristics of biomass were investigated as well.

3.2 Materials and methods

3.2.1 Reactors and operating conditions

The reactor containing BF material used in this study was constructed by acryl resin, having downdraft and updraft sections in a parallel upright arrangement as shown in Fig. 3-1. It also had clear zones of approximately 70 mm at the bottom and 30 mm at the top (below and above the biofringe reaction zone in the downdraft section). The working volume was 21.6 L. Influent was introduced deeply within the updraft section using a peristaltic pump at a fixed flow rate of 2 L/h. Air was also introduced near the base of the updraft section, serving to mix and oxygenate the wastewater while circulating it through the reactor. The air flow rate was fixed at 10 L/min and was related to the highest water flow velocity, by a tracer study using polyvinyl alcohol beads (dia., 4 mm; s.g., 1.025) ,which were used to establish a correlation between airflow rate (easily monitored using a gauge) and water flow velocity in the narrow updraft section.

The influent stock solution consisted of a bonito fish meat and peptone, and was diluted with tap water to obtain desired influent concentrations. The 5-day biochemical oxygen demand (BOD) was 74% of the chemical oxygen demand (COD) for influent solutions. $KHCO_3$ buffer solution was also added to the influent to a concentration of 125 mg/L (alkalinity addition, 62 mg $CaCO_3$/L). The tap water used for mixing the influent contained 19 mg Na^+, 6 mg K^+, 20 mg Ca^{2+}, 7 mg Mg^{2+}, 3 mg NO_3^- and 24 mg SO_4^{2-} per liter. The naturally occurring alkalinity

was 70 mg $CaCO_3$/L (increased to *ca.* 130 mg $CaCO_3$/L with addition of the buffer) and total hardness was 70 mg $CaCO_3$/L. In order to keep the operational temperature of 25□, a heater was used in the wintertime and air-conditioning was applied in the summer time under relatively dark conditions.

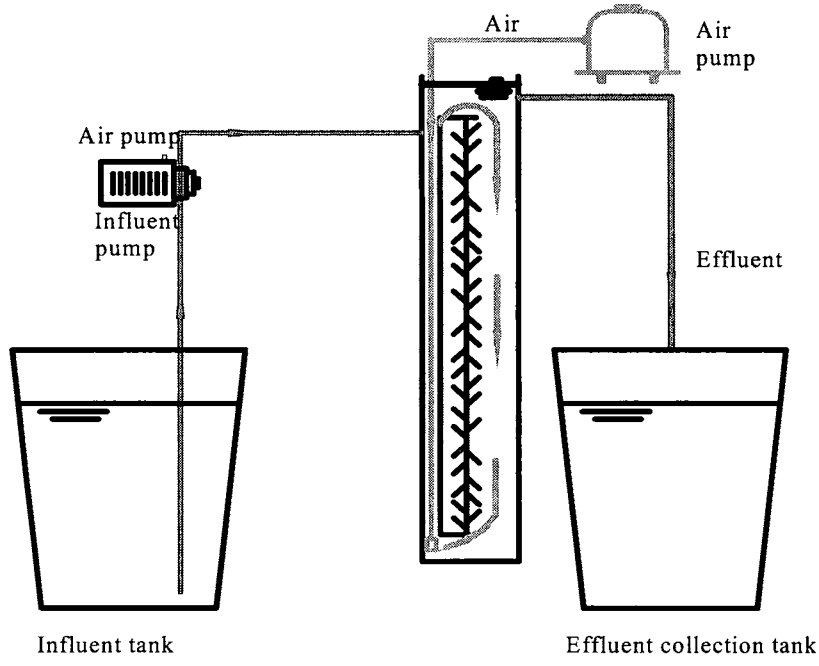


Fig. 3- 1 Schematic diagram of experimental apparatus

The experiments were divided into five runs corresponding to the applied COD volumetric loading rates (VLRs). Table 1 gives an overview of the main operational characteristics for the whole study. At the end of each run, the reactor was stopped and the BF material was removed from the reactor. All the biomass was then detached by hand and collected along with biomass gleaned from cleaning the interior of the reactor and the total amount was quantified. The experiment was then immediately re-started using the biomass detached from the BF as seed sludge. To quantify the amount of solids lost in the effluent, the entire effluent was collected and representative samples were for SS determinations.

Table 3- 1 Operational conditions

Run	I	II	III	IV		V	
HRT (h)	10.8	10.8	10.8	10.8	10.8	10.8	
COD loading rate (kg-COD/m^3/d)	1	2	3	4	5	6	7
Influent COD (mg/L)	450	900	1350	1800	2250	2700	3150
Duration (days)	30	20	16	6	16	8	21

3.2.2 Seed sludge

The reactor was initially seeded using activated sludge from a lab-scale fill-and-draw batch reactor. The synthetic medium used for the development and maintenance of the seed sludge was the same as that used in this study.

3.2.3 Analytical methods

Analytical methods are the same as the previous chapter (p.35)

Extracellular polymers (EPS) of biomass attached to biofringe were extracted by the alkaline-washing method (Rouse, et al., 2003). In the extracted solutions, proteins were measured using the method of Lowry *et al.* (1951) and carbohydrates by the method of Dubois *et al.* (1956). Nucleic acids (combined RNA and DNA) were estimated by the UV absorption method (Experimental Guidelines for Biotechnology, 1992).

3.2.4 Calculation methods

The observed sludge yields (Y_{obs}) were calculated according to the following equations for all runs.

$$Y_{obs} = \frac{g\text{-}X_{end} - g\text{-}X_{start}}{g\text{-}COD_{removed}} \tag{3-1}$$

$$g - X_{end} = X_{BF} + X_R + \sum_{i=1}^{n} X_i \tag{3-2}$$

The apparent effluent sludge yields (Y_e) basing on the effluent SS concentrations were calculated according to the following equation:

$$Y_e = \frac{\sum X_e}{g - COD_{removal}} \tag{3-3}$$

where the term $g\text{-}X_{start}$ is the total amount of seed biomass presented at the beginning of each run. $g\text{-}X_{end}$ represents the total amount of the biomass at the end of the experiment, which including the biomass attached on the biofringe(X_{BF}), dispersed bacteria contained in the cleaning water for the reactor(X_R) and the sum of the biomass(X_e) in the daily collected effluent. The term $g\text{-}COD_{removed}$ is the total amount of COD removed in each run.

Basing on the assumption that the attached biomass on the biofringe was a constant amount after the biofilm became mature, the values of sludge retention time (SRT) and food to microorganisms ratio (F/M) respectively were estimated as:

$$SRT = \frac{V \cdot X_a}{Q_W \cdot X_e} \tag{3-4}$$

$$F/M = \frac{Q_s \cdot C_s}{X_a \cdot V} \tag{3-5}$$

where V (L) is the reactor volume, X_a (mg/L) is the average MLSS concentration in reactor, Q (L/d) represents flow rate, X_e(mg/L) is the average effluent SS concentration, and C_s (mg/L) is the average influent COD concentration.

3.3 Results and discussion

3.3.1 General treatment

The degradation of organic constituents was evaluated by COD removals as shown in Fig. 3-2. The COD removals were maintained at a nearly constant level throughout the entire period of testing and over 80% of COD removal efficiencies were obtained even at VLRs of 7 kg-COD/m^3/d.

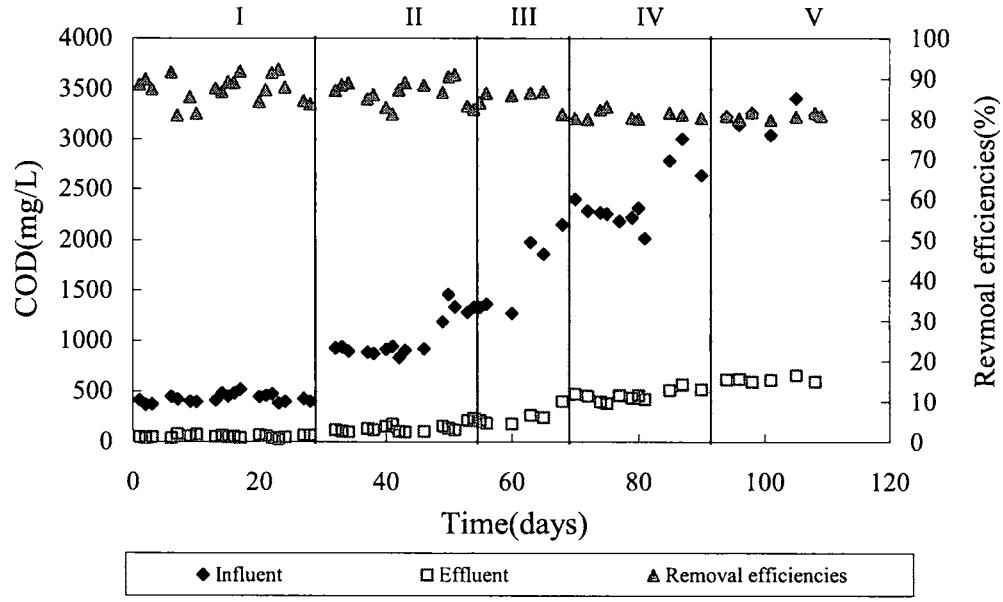


Fig. 3- 2 Time curses of COD concentrations

Fig. 3-3 and Fig. 3-4 show the daily changes in reactor MLSS and effluent SS. The MLSS concentrations varied greatly, these results show that the MLSS and effluent SS levels increased with an increase in VLR. They were not stable even under the same operational conditions. This implied that the reactor MLSS levels fluctuated greatly even within one day's operation. This would be closely related to attachment/detachment profile of biofile. Suspended solids in a BF reactor can be regarded as the fragments of biofilm (Rouse, et al., 2004). The average effluent SS varied from 10.3 mg/L to 640.9 mg/L with the increase in VLRs from 1 kg-COD/m^3/d to 7 kg-COD/m^3/d.

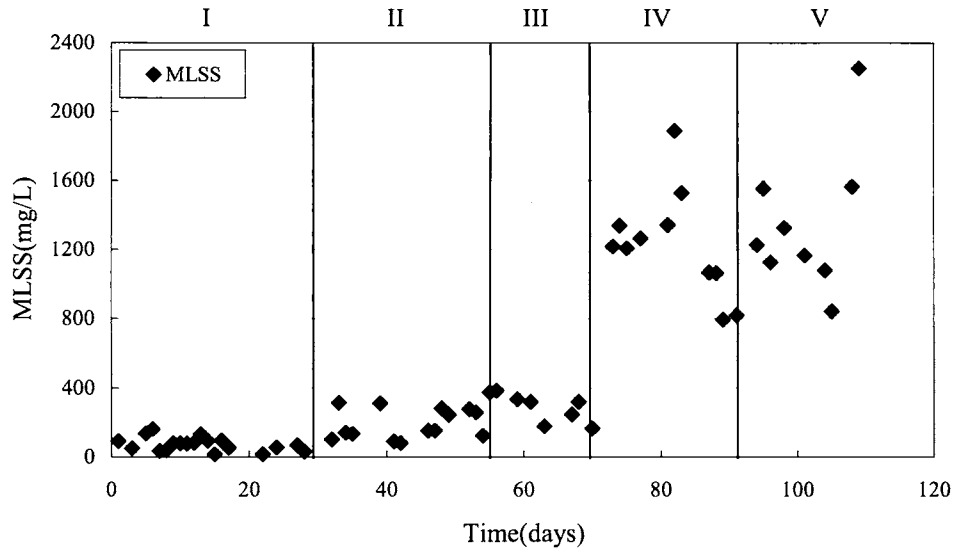


Fig. 3- 3 Time courses of reactor MLSS concentrations

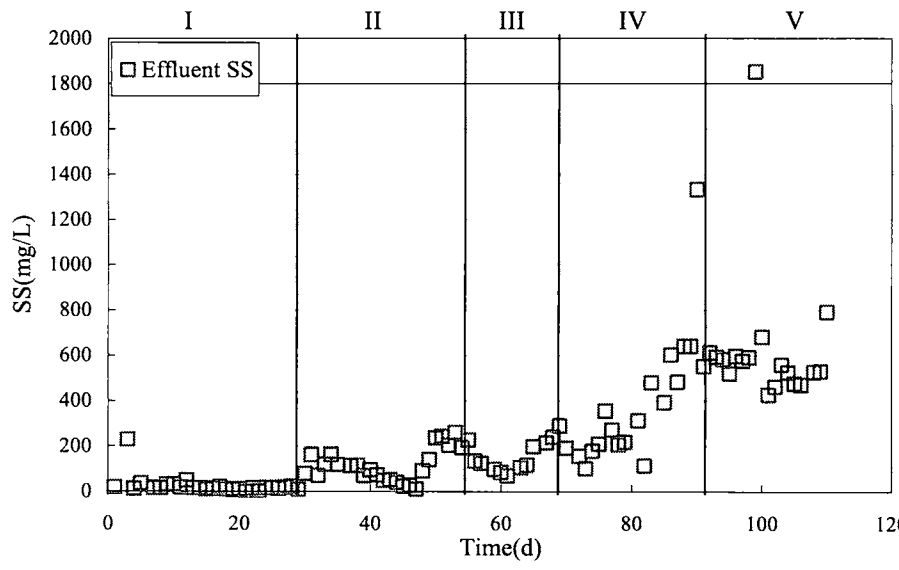


Fig. 3- 4 Time courses of effluent SS

3.3.2 Sludge production

The observed sludge yields (Y_{obs}) and apparent sludge yields for effluent SS (Y_e) were determined according to equations (3-1) and (3-3), respectively. Fig. 3-5 shows the values of Y_{obs} and Y_e in each run. The levels of observed sludge yields (Y_{obs}) ranged from 0.13 kg-MLSS/kg-$COD_{removed}$ to 0.29 kg-MLSS/kg-$COD_{removed}$ and the levels of apparent sludge

yields (Y_e) varied from 0.06 kg-MLSS/kg-$COD_{removed}$ to 0.21 kg-MLSS/kg-$COD_{removed}$ when the VLR increased from 1 kg-COD/m^3/d to 7 kg-COD/m^3/d. The sludge yields were reduced considerably in the first four runs. Although sludge yield increased a lot in Run V corresponding to a COD VLR of 7 kg-COD/m^3/d, they were still lower than that reported (30-50%) for typical activated sludge process. (Ghyoot and Verstraete, 1999)

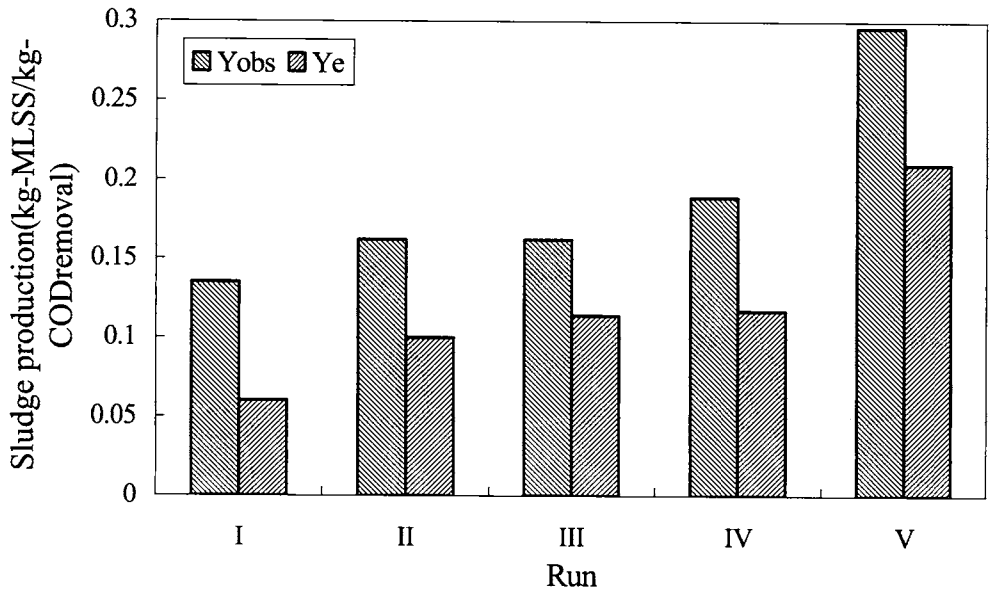


Fig. 3- 5 Changes of observed and apparent sludge yields

3.3.3 Sludge characteristics

3.3.3.1 Sludge retention time and F/M ratio

SRTs and F/M ratios for all runs were calculated by Equations (3-4) and (3-5), respectively (shown in Table 3-2). Although the lab-scale study was conducted with short run periods, the SRTs were longer than those of conventional activated sludge processes ranging from 3-6 days. The longer SRT benefited sludge reduction. The results of a pilot-scale side-stream MBR treating synthetic wastewater at SRTs ranging from 30 days to 2 days showed that both sludge yield and biomass viability generally increased with decreasing SRT (Cicek, et al., 2001). Little excess sludge production was obtained in a pilot cross-flow MBR plant with complete sludge retention. High reactor MLSS concentration of 40-50 g/L were obtained and only 6 % of influent carbon was assimilated.

The F/M ratios from Run II to Run IV were higher than optimal F/M ratios applied in conventional activated sludge processes. However, better removal efficiencies were still achieved in these three runs, which demonstrate that the swim-bed process is well suited for the wastewater treatment under high F/M ratios.

Table 3- 2 Summarization of the SRT values and the F/M ratios in each Run

Run	I	II	III	IV	V
SRT (d)	49	17	11	15	15
F/M (kg-COD/m^3/d)	0.33	0.69	0.97	0.73	0.38

3.3.3.2 Microorganisms

Table 3- 3 Enumeration results for protozoa and metazoan; (count/mL)

	filamentous microorganism	*Ciliate*	*Rotifer*	*Chaetonotus*	*Fflagellate*	*Dyplogaster*
I	+	1.6×10^3-2.0×10^5	2.5×10^2-6.3×10^2	1.5×10-1.0×10^2	6.3×10^2-3.0×10^3	1.3×10-2.0×10^2
II	-	7.9×10^2-1.3×10^3	1.6×10^3-3.2×10^3	2.5×10^2-3.2×10^2	-	2.5×10-1.0×10^2
III	-	-	6.3×10^4-1.3×10^5	-	-	-
IV	-	-	2.5×10^2-2.0×10^3	-	-	-
V	-	-	-	-	-	-

Frequent microscopic observations of sludge were performed. Table 3-3 gives an overview of the observation results. Many *Mstigophora* were observed in Run I. *Ciliophora*, which preyed on the free bacteria, appeared in both Run I and Run II (shown in Fig. 3-6). These two species of protozoa were abundant in the activated sludge in Run I and their dominance was closely related to clear effluent and good sludge settling ability. Others also observed that *Ciliophora* protozoa were generally the most abundant in protozoa communities in a wastewater treatment plant (Curds, et al., 1983). They are responsible for decreasing the

numbers of free bacteria and also play an important role in consuming dissolved organic matter (Ratsak, et al., 1996). Many researchers have reported that *Ciliophora* contributes to the reduction of excess sludge production. The growth of *Ciliophoras* in a two-stage pure culture chemostat system was associated with 12-43% sludge reduction (Ratsak, et al., 1994) and *Ciliophoras* and *Mstigophoras* contributed to sludge reduction in an aerobic treatment process (Lee and Welander, 1996). *Chaetonotus* was also found during the first two runs (shown in Fig.7 (a)) and an increase in number was observed in the second run. *Chaetonotus* usually use small protoza and filamentous algal as food and they are also known to be related for sludge reduction. *Rotatoria* occurred throughout the first four runs and their numbers increased with increases in VLRs from 1 kg-COD/m^3/d to 3 kg-COD/m^3/d (showed in Fig.7 (b)). Many *Rotatorias* were still found in Run IV although their number decreased compared to Run III. It has been suggested that *Rotatoria* play an important role in the activated sludge process (Gray, 1989). The growth of *Rotatoria* is mainly dependent on the amount of food available and they consume several times of their body weight every day (Clément, et al., 1991). This can be used to explain why the greatest abundance of *Rotatoria* was observed in Run III at a VLR of 3 kg-COD/m^3/d. The *Rotatoria* can graze on algal and particles in size ranging from 0.2-10μm according to the research work of Lapinski (2003). The total biomass of the treatment system in which *Rotatoria* grazed on suspended particles was subsequently reduced by ~10% (TSS) within 48h at the *Rotatoria* density used. Consistent with that, a preliminary experiment suggested that total solids removal was also significantly reduced by *Rotatoria* grazing. In this study, considerable sludge reduction was obtained due to the *Rotatoria* occurrence (Lapinski and Tunnacliffe, 2003).

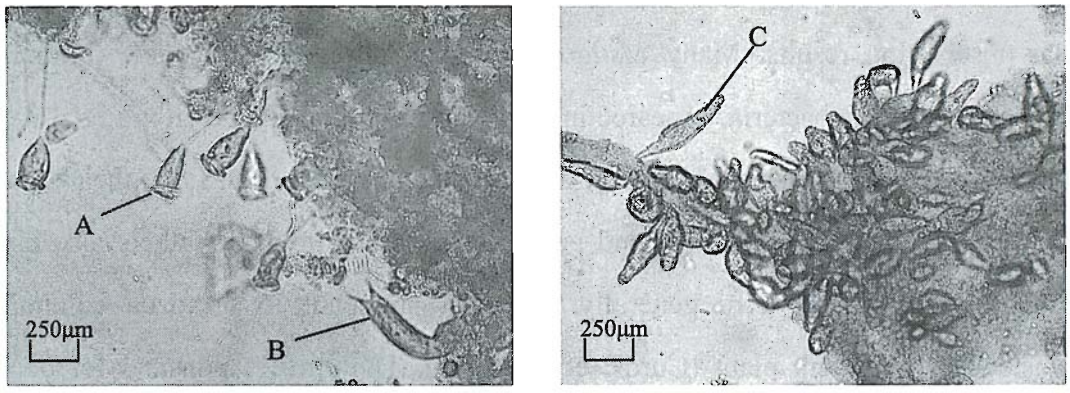


Fig. 3- 6 Photos of microorganism in reactor; A: *Ciliatge*, B: *Chaetonotus*, C: *Rotifer*

It was apparent that high organic removal efficiencies and sludge reductions were closely related to the occurrence of the protozoa and metazoa in large numbers. Their high diversity accounted for considerable sludge reduction in the first two runs. No increases in the observed sludge yields were observed with increases in VLRs from 2 kg-COD/m^3/d to 3 kg-COD/m^3/d. This result was due to the abundance of the *Rotatoria* at loadings of 3 kg-COD/m^3/d. Increases in both observed sludge and excess effluent sludge yields for Run V were associated with a decrease in protozoa and metazoa populations. Thus, it is not recommended to operate single swim-bed reactor at VLRs over 7 kg-COD/m^3/d.

3.3.4 Characterization of extracellular polymers

At the end of each run, the EPS of biofilm were analyzed. The results were shown in Table 3-4. The total EPS in the attached growth of this study were significant higher than reported values of various activated sludges (Kakii, et al., 1989). Protein was the main component of EPS, which was consisted with the results of Rouse et al. (Rouse, et al., 2004). The relative high level of EPS-protein was the characteristic for biofilm growth (i.e., growth on an attachment medium) as compared to the flocculent or granular sludge.

Table 3- 4 Compositions of extracellular polymers in biofilm samples for each Run

	Run I	Run II	Run III	Run IV	Run V
Proteins	66	58	49	48	62
Carbohydrate	11	8	12	8	8.7
Nucleic acids	9	10	12	8	11
Total	86	76	73	64	82

All values are reported as percent (%) of dry biomass (i.e. MLVSS)

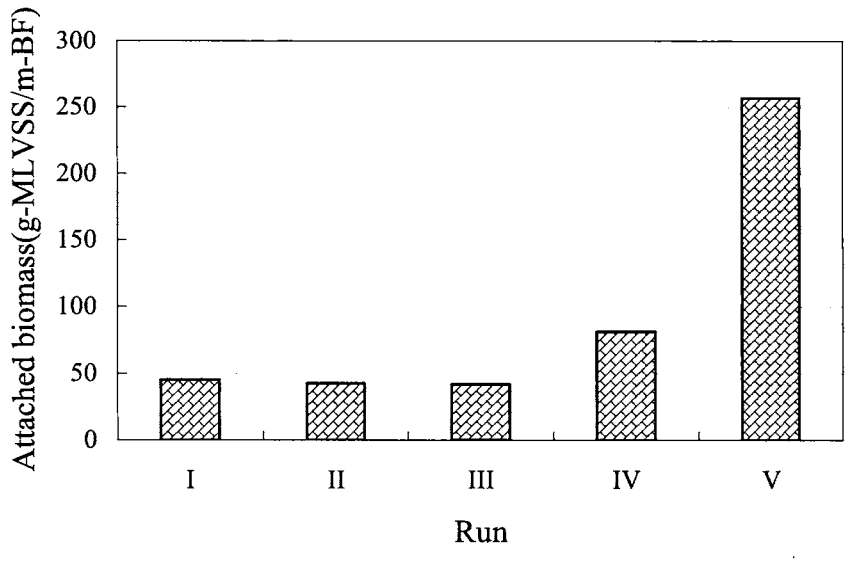


Fig. 3- 7 Attached-biomass on biofringe

In biofilm, EPS provides the framework, into which microbial cells are embedded. The attached biomass on BF material varied from 42.0 g-MVLSS/m-BF to 251 g-MLVSS/m-BF in this study. The high levels EPS obtained in the present study contributed to the high amounts of the biomass attached on BF.

EPS came from the natural secretions of bacteria, cell lysis and hydrolysis production (Zhang and Bishop, 2003). The ratios of EPS to the biomass first reduced in the first four runs by enhancing the VLRs and re-increased in Run V. The trend of EPS changes responding to the VLRs changes illustrated that the provided nutrient would affect the ratios of EPS to the biomass. Miqueleto, et al. also proved that the EPS production depended on the VLRs applied in the reactor (Miqueleto, et al., 2005). Zhang found the EPS of biofilm were degraded by their own producers and other microorganisms in the starvation phase (Zhang and Bishop, 2003). Li got the similar result when investigating the influence of starvation phase on the properties and the development of aerobic granules (Li, et al., 2006). On the contrary, the relative higher EPS concentrations were obtained in the Run I and Run V. The F/M ratios were lower than the other three runs. Further study was needed to make clear the affection factors to the EPS concentrations.

3.4 Conclusions

1. Good treatment performances were achieved using the swim-bed technology. COD removal efficiencies greater than 80% were obtained even at VLRs up to 7 kg-COD/m^3/d.

2. The BF material allowed for the attachment of large amounts of biomass. The attached biomass ranged from 42.0 g-MLVSS/m-BF to 251 g-MLVSS/m-BF. The large amounts of attached-growth extended the SRTs varying from 11 to 49 days and contributed to the abundance of metazoa and protozoa, which contributed to the significant sludge reduction. The observed sludge yields and effluent sludge yields ranged from 0.13 kg-MLSS/kg-$COD_{removed}$ to 0.29 kg-MLSS/kg-$COD_{removed}$ and from 0.06 kg-MLSS/kg-$COD_{removed}$ to 0.21 kg-MLSS/kg-$COD_{removed}$, respectively.

3. High levels of the EPS/biomass ratios were observed, which could enhance the attachment of the biomass on the BF matrix. These results demonstrated that the EPS/biomass ratios were affected by the VLRs.

3.5 References

Ahn K.H., Park K.Y., Maeng S.K., Hwang J.H., Lee J.W., Song K.G., Choi S. (2002) Ozonation of wastewater sludge for reduction and recycling, Water Sci. Technol., 46(10), 71-77.

APHA, AWWA and WEF (1995) Standard Methods for the Examination of Water and Wastewater, 19th Ed. American Public Health Association, Washington, D.C.

Chen G.H., Mo H.K., Saby S., Yip W.K., Liu Y. (2000) Minimization of activated sludge production by chemically stimulated energy spilling, Water Sci. Technol., 42(12), 189-200.

Chen G.H., Mo H.K., Liu Y. (2002) Utilization of a metabolic uncoupler, 3,30,40,5-tetrachlorosalicylanilide (TCS) to reduce sludge growth in activated sludge culture, Water Res., 36(8), 2077-2083.

Cicek N., Macomber J., Davel J., Suidan M.T., Audic J., Genestet P. (2001) Effect of solids retention time

on the performance and biological characteristics of a membrane bioreactor, Water Sci. Technol., 43(11), 43-50.

Clément P., Wurdak E., In: Harrison F.W., Ruppert E.E. (1991) Rotifera in microscopic anatomy of invertebrates, vol. 4, Aschelminthes. Chichester: Wiley-Liss & Sons, p. 219-289.

Curds C.R., Hawkes, H.A. (1983) Biological activeities and treatment processes. Ecological Aspecs of Used-Water Treatment, Academic Press, London Chapter 2.

Deleris S., Geaugey V., Camacho P., Debellefontaine H., Paul E. (2002) Minimization of sludge production in biological processes: an alternative solution for the problem of sludge disposal, Water Sci. Technol., 46(10), 63-70.

Dubois M., Gilles K.A., Hamilton J.K., Rebers P.A., Smith F. (1956) Colorimetric method for determination of sugars and related substances, Analytical Chemistry 28(3), 350-356.

Egemen E., Corpening J., Padilla J., Brennan R., Nirmalakhandan N. (1999) Evaluation of ozonation and cryptic growth for biosolids management in wastewater treatment, Water Sci. Technol., 39(10-11), 155-158.

Egemen E., Corpening J., Nirmalakhandan N. (2001) Evaluation of an ozonation system for reduced waste sludge generation, Water Sci. Technol., 44(2-3), 445-52.

Experimental Guidelines for Biotechnology (in Japanese): Society of Fermentation and Bioengineering, Osaka, Japan, pp.98-99 (1992).

Ghyoot W., Verstraete W. (1999) Reduced sludge production in a two-stage membrane-assisted bioreactor, Wat. Res., 34(1), 205-215.

Gray N.F. (1989) Biology of wastewater treatment. Oxford, New York, Tokyo: Oxford University Press.

Kakii K., Nakatani K., Shirakashi T., Kuriyama M. (1989) Extracellular polymers in relation to settling properties of activated sludge, J. Ferment. Bioeng., 68: 365-370.

Kamiya T., Hirotsuji J. (1998) New combined system of biological process and intermittent ozonation for advanced wastewater treatment, Water Sci. Technol., 38(8-9): 145-153.

Lapinski J., Tunnacliffe A. (2003) Reduction of suspended biomass in municipal wastewater using bdelloid rotifers, Water Res., 37, 2027-2034.

Lee N.M., Welander T. (1996a) Reducing sludge production in aerobic wastewater treatment through manipulation of the ecosystem, Water Res., 30(8), 1781-1790.

Lee N.M., Welander T. (1996b) Use of protozoa and metazoa for decreasing sludge production in aerobic

wastewater treatment, Biotechnol. Lett., 18(4), 429-434.

Lee N.M., Welander T. (1996) Reducing sludge production in aerobic wastewater treatment through manipulation of the ecosystem, Wat. Res., 30(8), 1781-1790.

Li Z.H., Kuba T., Kusuda T. (2006) The influent of starvation phase on the properties and the development of aerobic granules. Enzeme and microbial Technology, 38, 670-674.

Liu Y., Tay J.H. (2001) Strategy for minimization of excess sludge production from the activated sludge process. Biotechnol. Adv., 19(2), 97-107.

Low E.W., Chase H.A. (1998) The use of chemical uncouplers for reducing biomass production during biodegradation, Water Sci. Technol., 1998, 37(4-5), 399-402.

Low E.W., Chase H.A., Milner M.G., Curtis T.P. (2000) Uncoupling of metabolism to reduce biomass production in the activated sludge process, Water Res., 34(12), 3204-3212.

Lowry O.H., Rosebrough N.J., Farr A.L., Randall R.J. (1951) Protein measurement with the folin phenol reagent, J. Biol. Chem. 193, 265-275.

Mason C.A., Hamer G., Bryers J.D. (1986) The death and lysis of microorganism in environmental process, FEMS Microbiol Rev., 39, 373-401.

Miqueleto A.P., Rorigues J.A.D., Ratusznei S.M., Foresti E., Zaiat M. (2005) Treatment of easily degradable wastewater in a stirred anaerobic sequencing batch biofilm reactor, Water Res., 39, 2376-2384.

Muller E.B., Stouthamer A.H., van Verseveld H.W., Eikelboom D.H. (1995) Aerobic domestic waste water treatment in a pilot plant with complete sludge retention by cross-flow filtration, Water Res., 29(4), 1179-1189.

Ratsak C.H., Kooi B.W., van Verseveld H.W. (1994) Biomass reduction and mineralization increase due to the ciliate tetrahymena Pyriformis grazing on the bacterium Pseudomonas Fluorenscens, Water Sci. Technol., 29(7), 119-128.

Ratsak C.H., Maarsen K.A., Kooijman S.A.L. (1996) Effect of protozoa on carbon mineralization in activated sludge, Wat. Res., 30, 1-9.

Rocher M., Goma G., Begue A.P., Louvel L., Rols J.L. (1999) Towards a reduction in excess sludge production in activated sludge processes: biomass physicochemical treatment and biodegradation, Appl. Microbiol. Biotechnol., 51(6), 883-890.

Rocher M., Roux G., Goma G., Begue A.P., Louvel L., Rols J.L. (2001) Excess sludge reduction in

activated sludge processes by integrating biomass alkaline heat treatment, Water Sci. Technol. 44(2-3), 437-444.

Rouse J.D., Yazaki D., Cheng Y.J., Koyama T., Furukawa K. (2004) Swim-bed technology as an innovative attached-grouh process for high-rate wastewater treatment, Japanese of Water Treatment Biology, 40(3), 115-124.

Senez J.C. (1962) Some considerations on the energetics of bacterial growth, Bacteriol. Rev., 26, 95-107.

Stouthamer A.H. (1979) Correlation of growth yields. In: Quayle JR, editor. Microbial biochemistry, international reviewof biochemistry, vol. 21. Baltimore: University Park. p. 1-47.

Strand S.E., Harem G.N., Stensel H.D. (1999) Activated-sludge yield reduction using chemical uncouplers, Water Environ. Res., 71(4), 454-458.

Weemaes M.P.J., Verstraete W.H. (1998) Evaluation of current wet sludge disintegration techniques, J. Chem. Technol. Biotechnol., 73, 83-92.

Wei Y.S., Houten R.T.V., Borger A.R., Eikelboom D.H., Fan Y.B. (2003) Minimization of excess sludge production for biological wastewater treatment, Water Res., 37, 4453-4467.

Welander T., Lee N.M. (1994) Minimization of sludge production in aerobic treatment by use of predators, The Second International Symposium on Environmental Biotechnology, 4-6 July 1994, Brighton, UK.

Yang X.F., Xie M.L., Liu Y. (2003) Metabolic uncouplers reduce excess sludge production in an activated sludge process, Process Biochem., 38(9), 1373-1377.

Yasui H., Shibata M. (1994) An innovative approach to reduce excess sludge production in the activated sludge process, Water Sci. Technol., 30(9), 11-20.

Zhang X.L., Bishop P.L. (2003) Biodegradability of biofilm extracellular polymeric substances, Chemosphere, 50, 63-69.

Chapter 4 Performance of the swim-bed process with sludge recirculation for high-rate wastewater treatment

4.1 Introduction

Many cities around the world are looking for compact wastewater treatment alternatives since space for treatment plants is becoming scarce. Swim-bed technology involving the novel acryl fiber material, biofringe(BF), is presented here for high-rate treatment of organic wastewater. The BF material allows attachment of large amounts of biomass on a flexible matrix in a fixed position. By this approach, flexing of the matrix induced by wastewater flow creates a "swimming" motion that enhances mass transfer of nutrients to the attached growth (*i.e.*, biofilm).

The secondary sedimentation process plays an important role in the operation of suspended growth wastewater treatment processes. It combines the functions of (1) a thicken, to produce a thickened sludge for return to the aeration tank, (2) a clarifier, to produce a clean final effluent and (3) a storage tank to store sludge under peak loading conditions (Fuchs and Staudinger, 1998). Moreover, settling and compaction ability of activated sludge is crucial to the overall performance and efficiency of the treatment process as well as the quality of the receiving water body. Flocculation of activated sludge is an active process and depends on physical, chemical and biological factors. Activated sludge flocs are aggregates of suspended solids containing different groups of microorganisms and organic and inorganic particles embedded in a polymeric network of extracellular polymers (EPS) (Fround, et al., 1996; Biggs and Lant, 2000; Wilen, et al., 2001). It is recognized that the amount of EPS, surface properties (colloidal properties), floc size distribution, density and filament length are the major factors associated with activated sludge floc formation (Urbain, et al., 1993; Eikelboom, et al., 1983; Magara and Nambu, 1976; Andreadakis, 1993; Zita and Hermansson, 1997; Mikkelsen and Keiding, 2002).

The objective of this research is to investigate wastewater treatment performance using

swim-bed technology under high loading rates. Furthermore, simultaneous removals of COD and nitrogen are studied. Additionally, the ability to reduce excess sludge production even under high loading rates is investigated. These objectives were met by increasing the organic loading rates in a swim-bed reactor stepwise, at a hydraulic retention time (HRT) of 10.8 hours.

4.2 Materials and methods

4.2.1 Reactors and operating conditions

The reactor containing BF material used in this study was constructed of acryl resin, having downdraft and updraft sections in a parallel upright arrangement as shown in Fig. 4-1 It also had clear zones of approximately 70 mm at the bottom and 30 mm at the top (below and above the biofringe reaction zone in the downdraft section). The working volume was 10.8 L. Influent was introduced deeply within the updraft section using a peristaltic pump with a fixed flow rate of 1 L/h. Air was also introduced near the base of the updraft section, serving to mix and oxygenate the wastewater while circulating it through the reactor. The air flow rate was fixed at 10 L/min.

The settling tank was also made of acryl resin with a 2.5 L working volume and 0.017 m^2 of water surface area. The effluent from the outlet of the BF vessel was fed into the center of the settling tank by the gravity. The underflow was drawn from the central bottom of the settler and returned to the BF reactor at a 100% recycle rate.

The influent stock solution characteristics were the same as the in previous chapter (p.52).

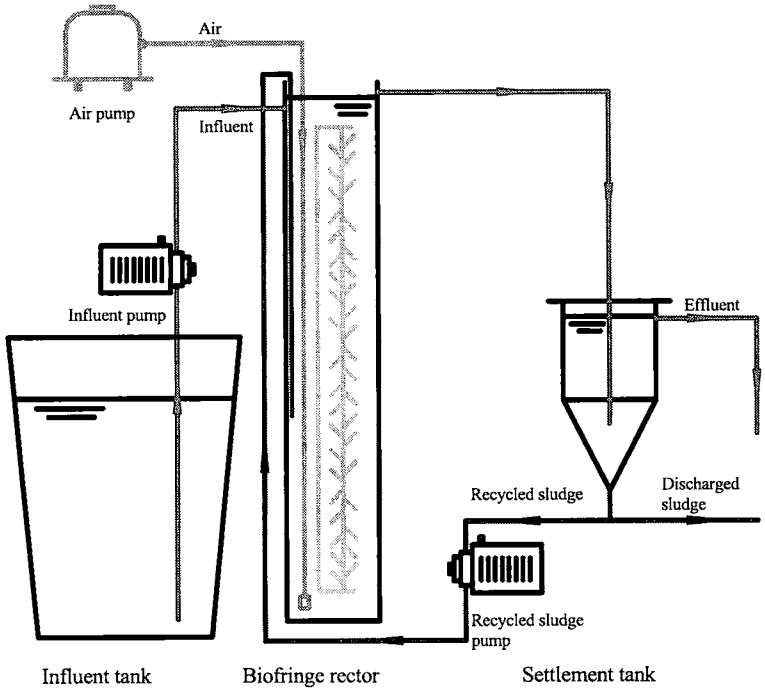


Fig. 4- 1 Schematic diagram of experimental apparatus

4.2.2 Seed sludge

The method for sludge seeding was the same as in the previous chapter (p.35).

4.2.3 Analytical methods

Analytical methods were the same as in the previous chapter (p.35).

EPS of suspended sludge and biomass attached to biofringe were extracted by the alkaline-washing method (Rouse et al., 2004). The EPS equation was described in Chapter 2 (p.34).

4.3 Results and discussion

4.3.1 General treatment performance

4.3.1.1 COD removal performance and changes in effluent SS

The degradation of organic components was evaluated by COD removals as shown in Fig.

4-2. The COD removal efficiencies increased slightly with increase in the volumetric loadings from the operation of 2 kg-COD/m^3/d. The highest average COD removals of 96.6% in the reactor and 96.8% in the effluent were obtained under a volumetric loading rate (VLR) of 6 kg-COD/m^3/d. Effluent COD removal efficiencies were slightly improved compared with that for the reactor.

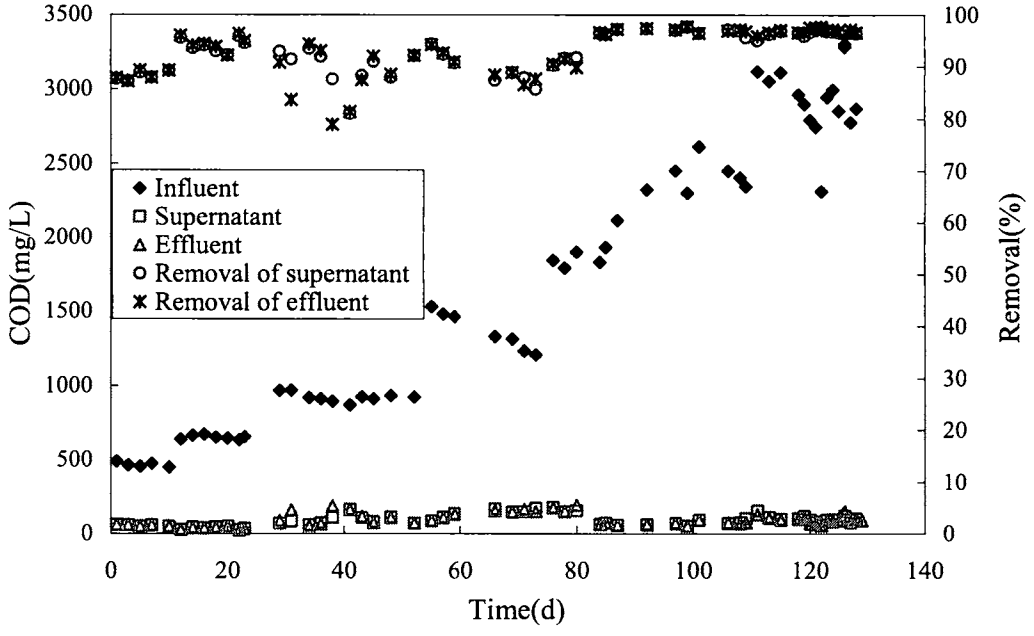


Fig. 4- 2 Time courses of COD concentrations and removal efficiencies

The effluent SS concentrations are shown in Fig. 4-3. On the 6th day, a chunk of sludge floated up in the settling tank resulting in the highest SS level of 131 mg/L; otherwise, the SS levels were always below 100 mg/L. It appeared that elevated loading rates had an influence on the effluent SS. Although SS levels became slightly higher at a loading rate of 6 kgOD/m^3/d, the average concentration was only 45.5 mg/L.

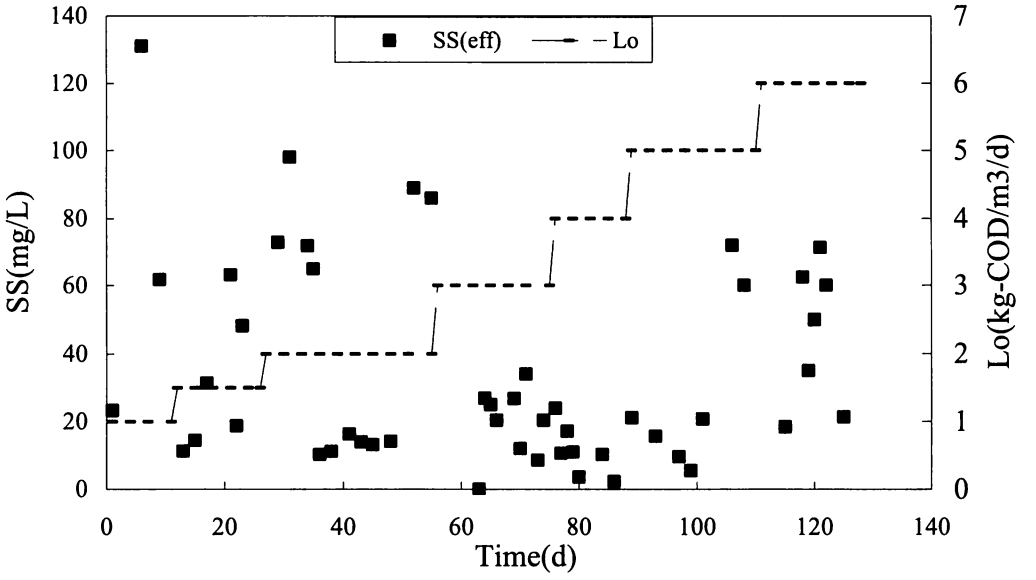


Fig. 4- 3 Time course of effluent SS versus COD loading rate; Lo: volumetric COD loading rate

4.3.1.2 T-N removal performance

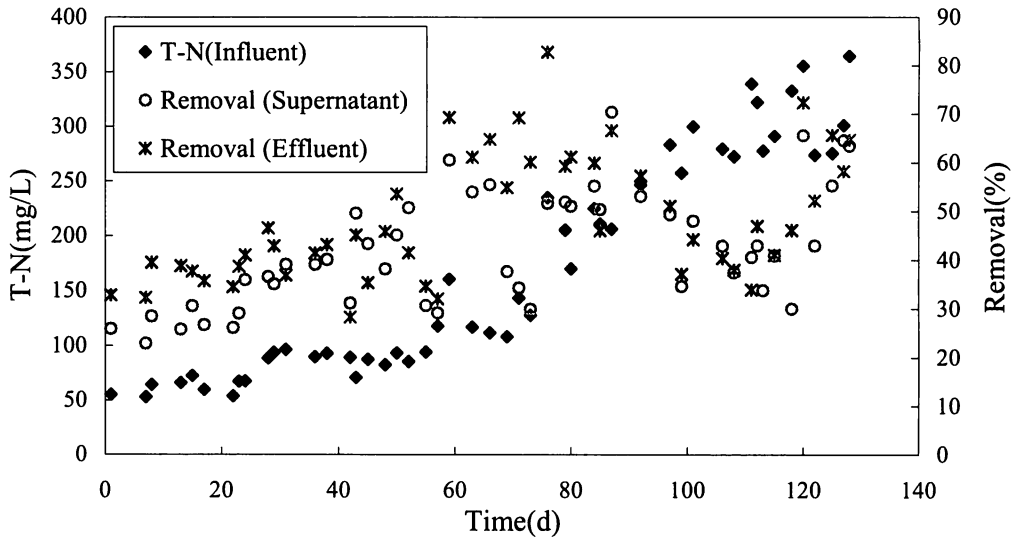


Fig. 4- 4 Time courses of influent T-N and removal efficiencies

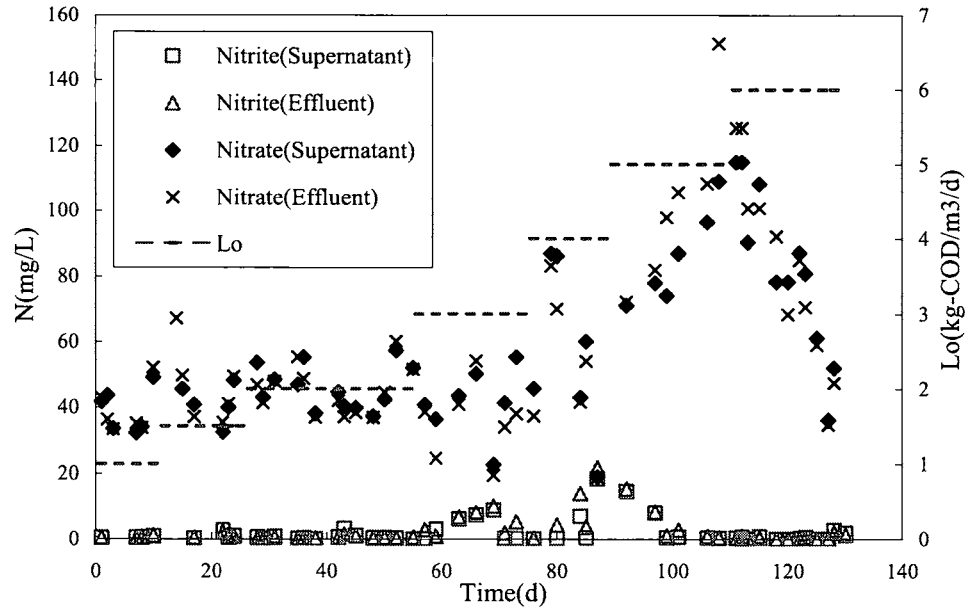


Fig. 4- 5 Time courses of nitrite and nitrate versus COD loadings

The influent ratio of COD to T-N was about 10:1. It was confirmed that the influent consisted of only organic components and low level of ammonium resulting from ammonification. Fig. 4-4 shows the time courses of T-N changes. The T-N removal efficiency first increased by increasing the organic loading rate from 1 kg-COD/m^3/d to 4 kg-COD/m^3/d, but decreased under operation at 5 kg-COD/m^3/d. When the organic loading rate was increased to 6 kg-COD/m^3/d, the T-N removal efficiencies increased and the highest T-N removal of 59.1% was obtained. NO_3-N also demonstrated an increasing trend with increase in the organic loading rate (as shown in Fig. 4-5).

The highest nitrate concentration was obtained on the 108th day under operation at the VLR of 5 kg-COD/m^3/d, while pH was the lowest at 4.42. However, nitrate reduced sharply under operation at the loading of 6 kg-COD/m^3/d. Fig. 4-6 shows the time courses of in NH_4-N concentrations; additionally, Fig. 4-7 shows the time courses of nitrification efficiencies. Nitrification efficiency (NE) was estimated according to the following equation:

$$NE=(1-EAM/ITN)\times 100\% \quad (4\text{-}1)$$

where NE is nitrification efficiency, EAM is effluent ammonium concentration and ITN is

influent total nitrogen concentration. As shown in Fig. 4-7, even though the ammonium accumulated in runs with loadings of 5 kg-COD/m^3/d and 6 kg-COD/m^3/d, average NE values of 78% and 79% were still achieved. It was illustrated that a more stable and better nitrification could be obtained in a combined process than in a single process (Rouse, et al., 2004). It was considered that the denitrification was inhibited under operation with low loading rates due to the lower SS levels and higher dissolved oxygen (DO) concentrations of around 7 mg/L. By increasing the influent loading rates, improved organic oxidation performance was observed with large amounts of oxygen consumption, which could contribute to the enhancement of denitrification and improvement of T-N removal efficiencies. Nevertheless, nitrogen removal efficiencies deteriorated under operation at 5 kg-COD/m^3/d due to the accumulation of nitrate. Furthermore, oxidation of ammonium occurred with a large consumption of alkalinity, resulting in lower pH values, which could also inhibit the activity of nitrite oxidizing bacteria.

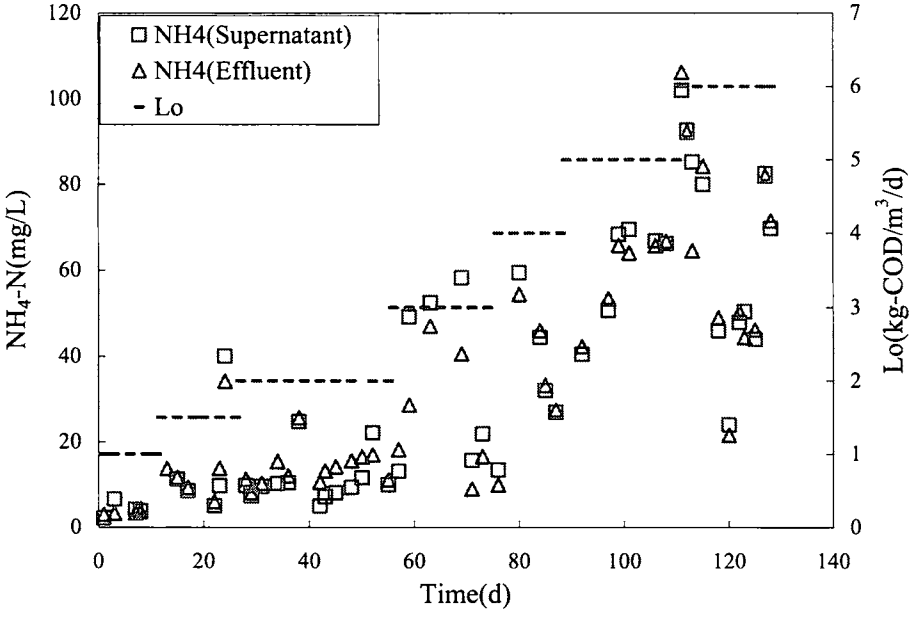


Fig. 4- 6 Time courses of NH_4-N concentrations

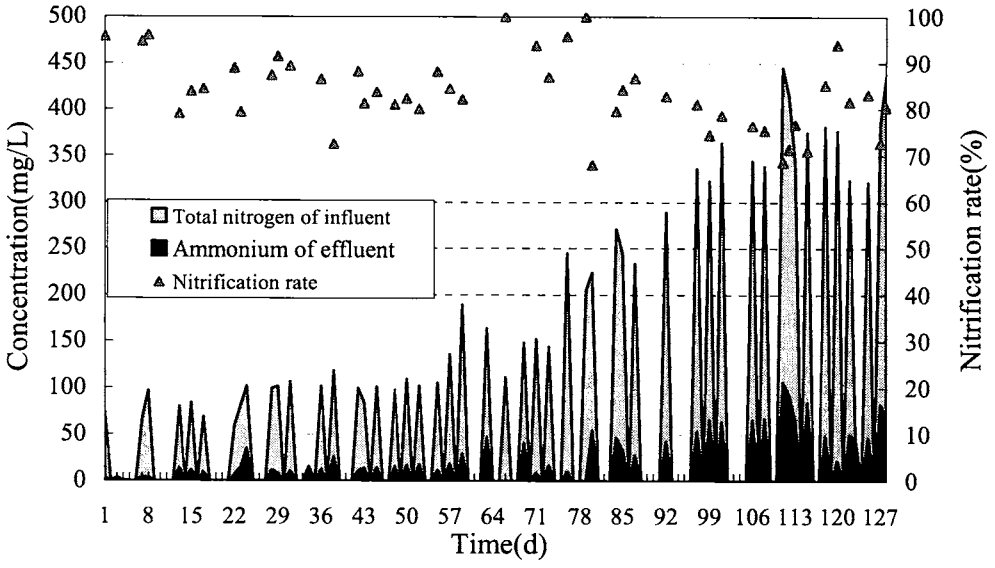


Fig. 4- 7 Time courses of nitrification efficiencies (NE)

4.3.2 Influence of suspended sludge on contaminant removal

An investigation was conducted by drawing out the BF material from the reactor and operating the reactor for four days under the VLR of 6 kg-COD/m^3/d. The effluent samples were continuously collected for four days and the average removal efficiencies were compared with that obtained in the previous run under the VLR of 6 kg-COD/m3/d with BF (shown in Table 4-1). Slightly reduced pollutant removal efficiencies were obtained by removing the BF material from the reactor and the nitrification might be inhibited according to the decreased values of NE. These results inferred that the suspended sludge in the BF reactor had a higher activity than that of conventional sludge.

Table 4- 1 Comparison of contaminant removal efficiencies

Process	Loading rates kg-COD/m^3/d	T-N removal rates (%)	NE (%)	COD removal rates (%)
With BF	6.0	59.1	79	96.8
Without BF	6.0	55.8	68	94.0

4.3.3 Biomass characteristics

MLSS concentrations increased with increase in COD VLRs and could maintain stably under the Run of one COD loading (see Fig. 4-8). Higher MLSS concentrations over 20 g/L were achieved at a COD loading rate of 6 kg-COD/m^3/d and the observed sludge yield (Y_{ob}) was calculated to be 0.002 g-MLSS/$gCOD_{removed}$ for this loading. This high level of activated sludge in the reactor ensured good treatment performance. Sludge loading rate was determined on the basis of organic loading rate and MLSS level in the reactor. As shown in Fig. 4-9, a relatively low value of about 0.3 kg-COD/g-MLSS/d was obtained at a volumetric loading rate of 6 kg-COD/m^3/d, which would explain why the best pollutant removal efficiencies were achieved under the volumetric loading rate of 6 kg-COD/m^3/d.

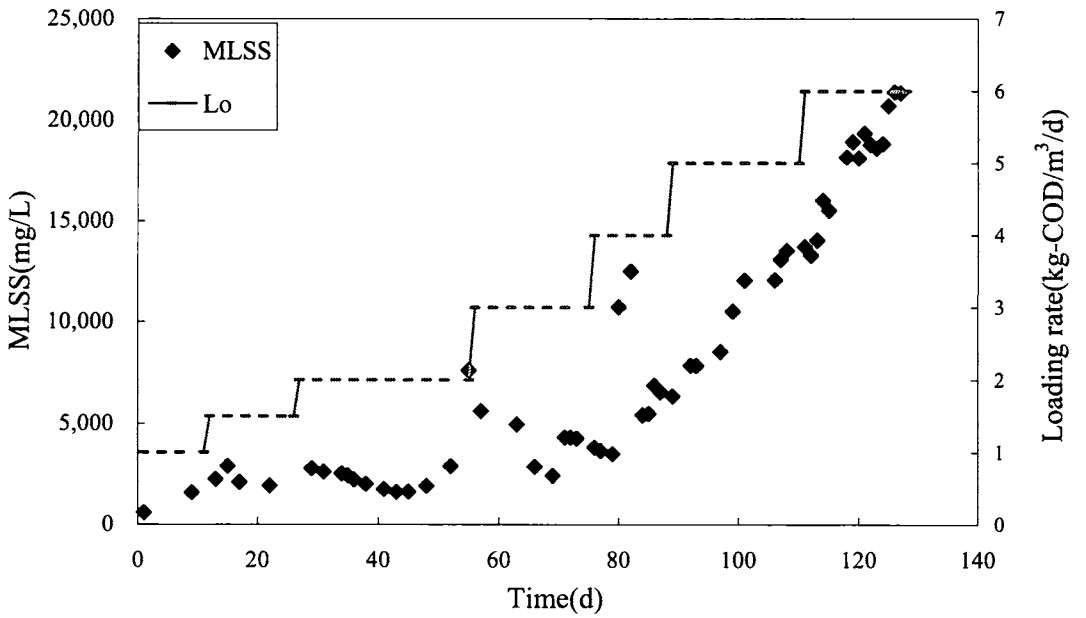


Fig. 4- 8 Time courses of MLSS concentrations

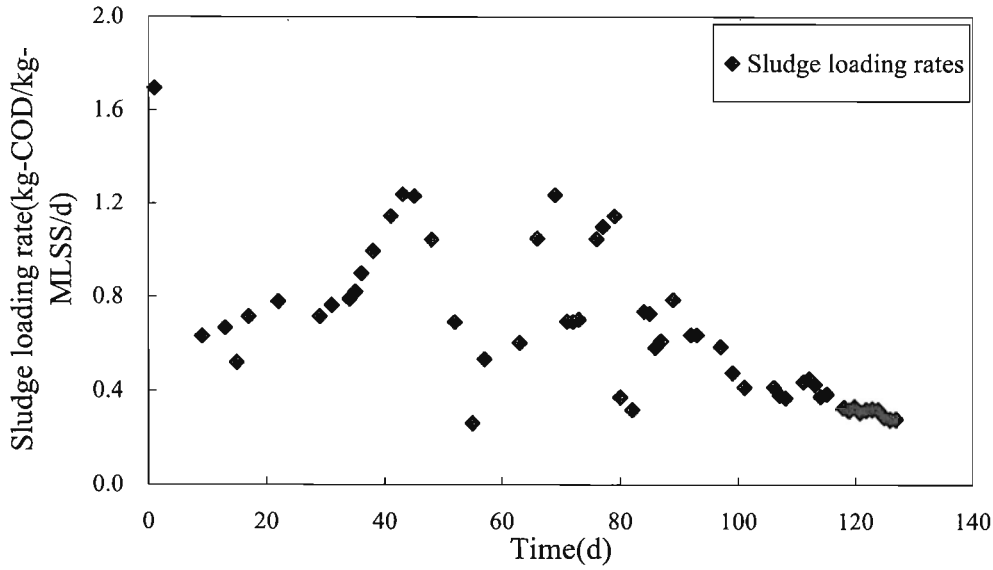
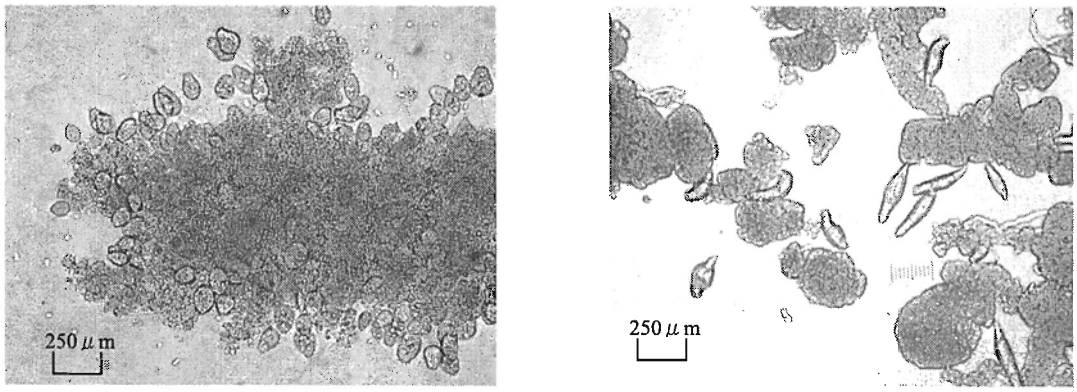


Fig. 4- 9 Time course of sludge loading rates



(a)after 36 days(×100)　　(b) after 121 days(×100)

Fig. 4- 10 Microscopic photographs of activated sludge

It is well known that the presence of protozoa and metazoa plays an important roles in keeping treated wastewater clear by consuming dispersed bacteria (Curds,1992; Ratsak, et al.,1996). Furthermore, the growth of microorganism higher in the food chain can also reduce the excess sludge production (Wouter, et al., 1999). Microscopic observations revealed the presence of many protozoa and mentozoa growing in the reactor's suspended solids (see Fig. 4-10).

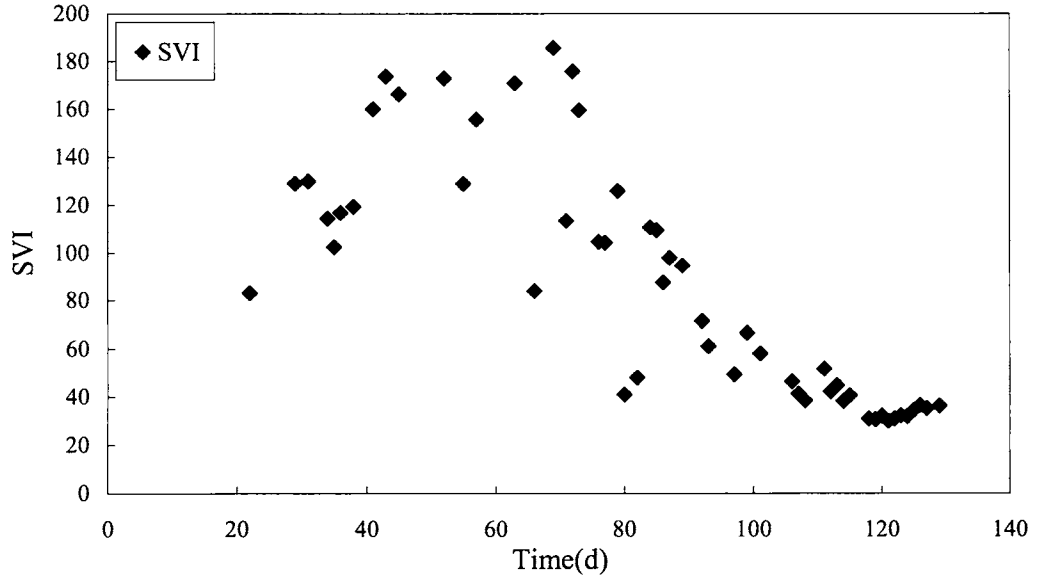


Fig. 4- 11 Time courses of SVI values

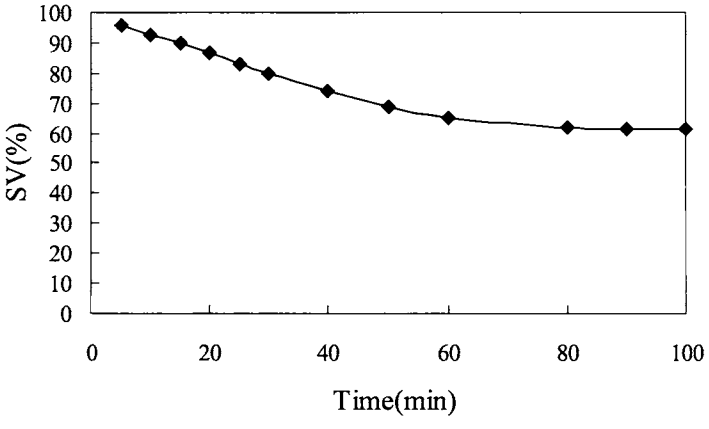


Fig. 4- 12 Sludge settling curve (MLSS 20 g/L)

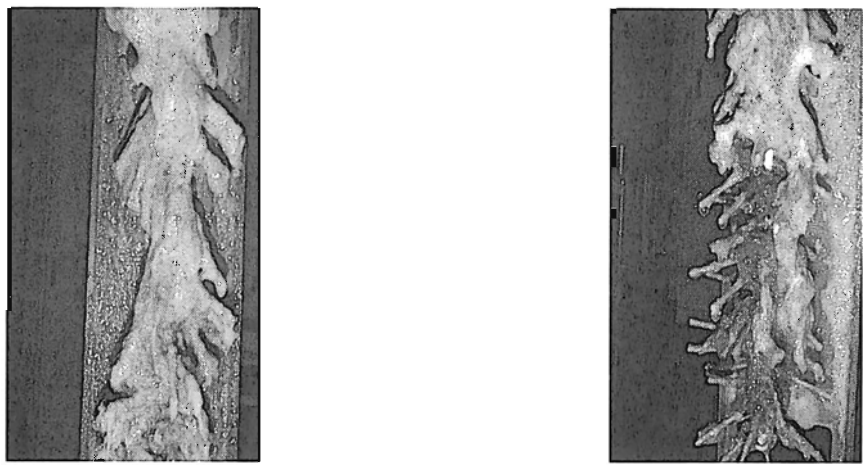
SVI values decreased with increases in the loading rate over the period showed in Fig. 4-11. The lowest SVI value of 30 was obtained at a volumetric loading rate of 6 kg-COD/m^3/d with the highest MLSS concentration of about 20 g/L, of which MLVSS accounted for 93%. Good sludge settling was also demonstrated by a settling test using one liter suspended solid taken from the reactor (Fig. 4-12). The lower SVI values attributed to the prominent compressibility and settling ability of the sludge. Good and stable setting characteristics were exhibited in the settling tank as well. The formation of spherical granular-like flocs as shown in Fig. 4-10 (b) was closely related to these good settling characteristics (Jin, et al., 2003). The flocs profiles were described on two size scales, macro-scale and micro-scale because of the complex structure of the aggregates (Wu, et al., 2002). The average floc size in the macro-scale obtained was estimated to be 1.23 mm by light microscope. The high level of EPS contained in the reactor's suspended sludge was thought to be beneficial to the aggregation of the small particles and the formation of

granular-like flocs. Table 4-2 shows the EPS characteristics of the suspended sludge. Granular-like flocs could provide a suitable micro-environment for the occurrence of simultaneous nitrification and denitrification reactions. Thus, the improved nitrogen removal rates at the volumetric loading rate of 6 kg-COD/m^3/d might have been due to the formation of the granular-like flocs.

Table 4- 2 Composition of EPS in suspended sludge under the loading of 6 kg-COD/m^3/d.

Protein	Carbohydrate	Nucleic acid	Total
32	10	8	50

- All values are percent (%) of MLSS



(a) upside (b) downside

Fig. 4- 13 Photographs of BF material drawn out of reactor

On the 132nd day, BF material was drawn out from the reactor and it was determined that there was 27.2 g biomass per meter of the biofringe matrix. The sludge color changed along the biofringe with light and dark patches (see Fig. 4-13), suggesting that aerobic zone were common in the upper part of the BF reactor, and anaerobic zone in the lower part. EPS were extracted from the biomass sampled from the upper and lower parts and the compositions are described in Table 4-3. The result shows that the concentrations of protein, carbohydrate and nucleic acids contained in the attached sludge of the upper part were all much higher than those contained in the lower part.

Table 4- 3 Composition of EPS extracted from biofilm

Position	Protein (%)	Carbohydrate (%)	Nucleic acid (%)	Total (%)
Upside	39.9	8.5	6.74	55.1
Downside	17.4	5.3	1.9	24.6

- All values are percent (%) of MLSS

4.4 Conclusions

A swim-bed treatment system, combining a reactor using the novel BF material and a settling tank, was evaluated and shown to be an effective method for treatment under high loading rate. On the basis of the results obtained, the following conclusions were obtained:

1. 97% COD and 59% total nitrogen could be removed by this combined process even at a volumetric loading rate of 6 kg-COD/m^3/d.

2. A high MLSS concentration with good settling characteristics was achieved. The highest MLSS level was over 20 g/L and the lowest SVI was 30. Due to the longer food chain in a BF reactor, sludge production was low and the observed sludge yield was only 0.002 $gMLSS/gCOD_{removal}$.

3. Granular-like flocs formed in the mixed liquor. The high-level of EPS in the suspended sludge appeared to relate to the exceptionally good settling characteristics of mixed liquor and the improved nitrogen removal efficiencies.

4. 27.2 g of biomass per meter of BF was retained. Aerobic and anaerobic zones were formed in the upper and lower parts, respectively, of the BF reactors.

4.5 References

Andreadakis A. (1993) Physical and chemical properties of activated sludge floc, Water Sci. Technol., 27(12), 1707-1714.

APHA, AWWA, and WEF: Standard Methods for the Examination of Water and Wastewater, 19th Ed. American Public Health Association, Washington, D. C. (1995).

Biggs C. and Lant P. (2000) Activated sludge flocculation: on-line determination of floc size and the effect of shear, Water Res., 34, 2542-2550.

Dubois M., Gilles K.A., Hamilton J.K., Rebers P.A., and Smith F. (1956) Colorimetric method for determination of sugars and related substances, Analytical Chemistry 28(3), 350-356.

Eikelboom D. and H. van Bijsen (1983) Microscopic sludge investigation manual, TNO Research Institute for Environmental Hygiene, The Netherlands.

Experimental Guidelines for Biotechnology (in Japanese): Society of Fermentation and Bioengineering, Osaka, Japan, pp.98-99 (1992).

Frolund B., Palmgren R., Keiding K., and Nielsen P. (1996) Extraction of extracellular polymers from activated sludge using a cation exchange resin, Water Res., 30, 1749-1758.

Fuchs A. and Staudinger G. (1998) Characterising the clarification of the supernatant of activated sludges, Wat. Res., 33(11), 2527-2534.

Jin B., Wilen M.-B., and Lant P. (2003) A comprehensive insight into floc characteristics and their impact on compressibility and settleability of activated sludge. Chemical Engineering Journal 95, 221-234.Curds C.R. (1992) Protozoa in the water industry. Cambridge University Press, Cambridge, U.K.

Lowry O.H., Rosebrough N.J., Farr A.L. and Randall R.J. (1951) Protein measurement with the folin phenol reagent, J. Biol. Chem., 193, 265-275.

Magara Y. and Nambu S. (1976) Biochemical and physical properties of an activated sludge on settling characteristics, Water Res., 10, 71-77.

Mikkelsen L. and Keiding K. (2002) Physico-chemical characteristics of full scale sewage sludges with implication to dewatering, Water Res. 36, 2451-2462.

Ratsak C.H, Maarsen K., and Kooijman A.L.M. (1996) Effects of protozoa on carbon mineralizatio in activated sludge, Wat. Res., 30, 1-12.

Rouse J.D., Yazaki D., Cheng Y.J., Koyama T., and Furukawa K. (2004) Swim-bed technology as an innovative attached-grouh process for high-rate wastewater treatment, Japanese of Water Treatment Biology, 40(3), 115-124.

Urbain V., Block J., and Manem J. (1993) Bioflocculation in activated sludge: an analytical approach, Water Res. 5, 829-838.

Wilen B., Jin B., and Lant P. (2001) The influence of key chemical constituents in activated sludge on

surface and flocculating properties, Water Res., 37(9), 2127-2139.

Wouter G. and Willy V. (1999) Reduced sludge production in a two-stage membrane-assisted bioreactor, Wat. Res., 34(1), 205-215.

Wu R., Lee D., Waite T., and Guan J. (2002) Multilevel structure of sludge floc, J. Colloid Interface Sci., 252, 383-392.

Zita A. and Hermansson M. (1997) Effects of bacterial cell surface structures and hydrophobicity on attatchment to activated sludge flocs, Appl. Environ. Microbiol., 63, 1168-1170.

Chapter 5 Treatment capacity of swim-bed technology for domestic wastewater

5.1 Introduction

Some problems remain unsolved for the conventional activated-sludge process such as large space requirement, complicated system operation and large excess sludge production. The expense for excess sludge treatment has been estimated to be half of the total cost of wastewater treatment plant operation (Lee, et al., 2005). Fixed-bed attached-growth processes offer some advantages over suspended-growth processes due to reduced sensitivity to toxicity, co-existence of aerobic and anoxic metabolic activities and compactness. While newly developed fluidized-bed (or moving-bed) attached-growth processes have further demonstrated elimination of head loss with absence of clogging and channeling, improved mass transfer and the potential for up-grading existing treatment plants without constructing new tanks (Lazarova and Manem, 1994; Pastorelli, et al., 1997a). Fluidized-bed reactors have thus been of considerable interest for the removal of organic compounds from wastewater in recent years (Pastorelli, et al., 1997b; Loukidou, et al., 2001). In addition, Lazarova and Manem (1996), using gas-lift technology, introduced the circulating floating-bed reactor, which demonstrated a synergy between hydrodynamic characteristics and biological treatment performance countering the negative influence of solid media hold-up that can occur in fluidized-bed processes.

Swim-bed technology involving the novel acryl fiber material, biofringe, is a new concept for the treatment of organic wastewater. The biofringe (BF) material allows for attachment of large amounts of biomass on a flexible matrix in a fixed position. By this approach, flexing of the matrix induced by wastewater flow creates a "swimming" motion that enhances mass transfer of nutrients to the attached growth (*i.e.*, biofilm). Thus, all the potential benefits of fixed-bed and fluidized-bed reactors stated above are retained.

Application of the metozoa and protozoa as the predators is one of the important strategies for excess sludge reduction. During carbon transfer from low to high trophic levels, energy is lost

during biomass conversion. Under optimal conditions, the total loss of energy will be maximal and the total biomass production will thus be minimal (Wei, et al., 2003). It was our hypothesis that the excess sludge production should be reduced in swim-bed process since there were many metazoa and protozoa contained in the attached-growth sludge (Rouse, et al., 2004).

The objective of this part is to test the capacity of swim-bed processes for domestic wastewater treatment. To achieve this objective, contaminant removal efficiencies, sludge production and biomass characteristics were evaluated experimentally. The influent BOD concentrations were maintained at 150 mg/L throughout the whole test.

5.2 Materials and methods

5.2.1 Reactors and operating conditions

Table 5- 1 Operational conditions of Phase I

	Run I			Run II		Run III	
VLR (kg-BOD/m^3/d)	0.5			0.5	1	1	1.5
Operational days (d)	1-71	71-84	84-91	91-104	104-148	148-154	155-208
Air flow rate (L/min)	10	8	10	10		10	
HRT (h)	7.2			7.2	3.6	3.6	2.4
Sludge recycle rate (%)	100			100		100	
Volume of settler(L)	2.5			2.5		2.5	
Surface loading rate (m/d)	2.1			2.1	4.2	4.2	6.2

The study was conducted in two phases. Table 5-1 and Table 5-2 gave an overview of operational conditions for the two phases, respectively. In phase I the performance of swim-bed process on treating domestic wastewater was compared with that of conventional

activated sludge process (CAS). The comparison study was conducted with three runs, corresponding to the loading changes. The air-flow rates were initially fixed at 10 L/min. For reducing the shear stress in the reactors to keep the biomass attach on the BF material, the air-flow rates were reduced to 8 L/min on day 71. Since the decrease of the air-flow rate caused effluent SS levels increase in the BF reactor, the air-flow rates were re-increased to 10 L/min on day 84.

Table 5- 2 Operational conditions of Phase II

	Run IV		Run V		Run VI	Run VII		Run VIII
VLR(kg-BOD/m^3/d)	1.5	2	2	3	2	2	2.5	2.5
Operational days (d)	121-217	218-240	241-247	248-258	259-280	281-285	286-294	295-304
Air flow rate (L/min)	10		10		10	10		10
HRT (h)	2.4	1.8	1.8	1.2	1.8	1.8	1.4	1.4
Sludge recycle rate (%)	100		100	50	100	50		50
Volume of settler (L)	5		5		5	5		-
Surface loading rate (m/d)	2.7	3.6	3.6	5.4	3.6	3.6	4.5	-

The CAS and BF processes were operated in parallel under the same conditions. Two identical acryl resin reactors with the same working volumes of 10.8 L were used, one with the BF material and one without (showed in Fig. 5-1). The filling ratio of BF material was 50%. The designs of the reactors were same as the previous study. The settling tanks with 2.5 L working volumes and 0.017 m^2 of water surface areas were used in Phase I. The effluents from the bioreactors were fed into the settling tanks by gravity. The settled sludge was drawn from the central-bottom of the settlers and returned to the bioreactors with 100% recycle rates.

In order to estimate sludge production, total sludge for the system was measured at the end of each Run. For the CAS process, total sludge was determined from both in the reactor and

settling tank; while for the BF process, it also included the sludge attached on the support material. After sludge quantification, the two reactors were re-started immediately, using the sludge taken from previous Run of each reactor. Volumetric loading rates were then increased to a higher level after allowing several days for adaptation. Effluents of both processes were collected every day and the sludge discharged with the effluents was determined.

In Run III, almost all of the sludge washed out from the CAS reactor when increased the volumetric loading rate to 1.5 kg-BOD/m^3/d, so that the reactor was restarted again by using the new seeding sludge.

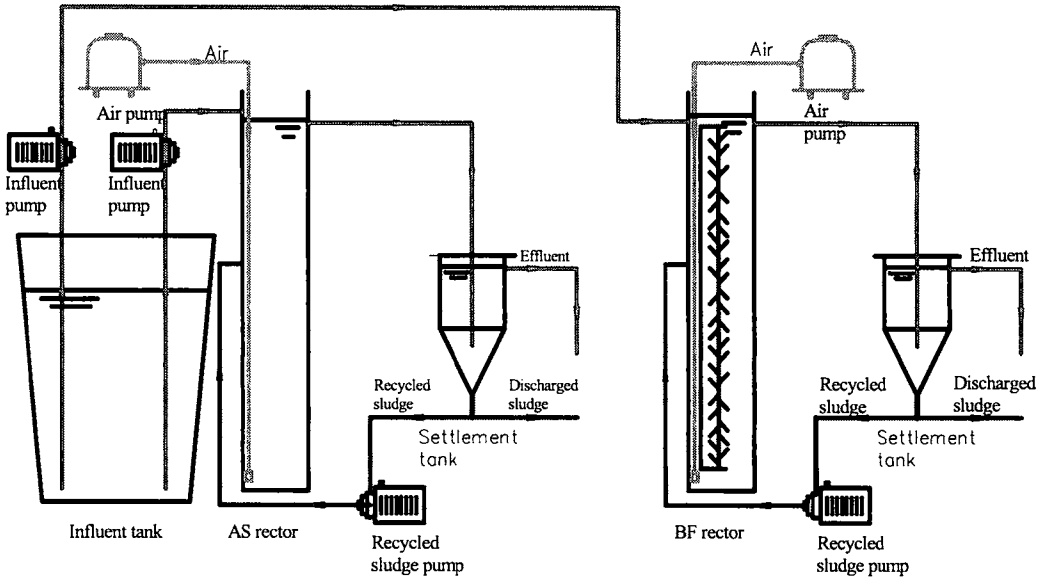


Fig. 5- 1 Schematic diagram of experimental system

Since it was difficult to operate CAS process at the loadings up to 1.5 kg-BOD/m^3/d, the CAS was stopped and BF process was continued to be operated in Phase II. In order to meet the increase of influent flow rate, a bigger settling tank with volume of 5 L and surface area of 0.04 m^2 was used. Five runs (Run IV-Run V) made up of Phase II. The experiments first were conducted by increasing the loadings from 2 kg-BOD/m^3/d (Run IV) to 3 kg-BOD/m^3/d (Run V) stepwisely. Sharply increased sludge production was obtained in Run IV (with loading of 2 kg-BOD/m^3/d) and surprising change of flocs were observed in Run V (with loading of 3

kg-BOD/m^3/d), In order to make clear the biological mechanism above, the study was repeated at the loading of 2 kg-BOD/m^3/d in Run VI. Increase the loading to 2.5 kg-BOD/m^3/d, the performances of BF process with settling tank and without settling tank were compared in Run VII and Run VIII, respectively.

Table 5- 3 Characteristics of influent

Component	Concentration (mg/L)	Component	Concentration (mg/L)
Na^+	19	COD	200
K^+	6	BOD	150
Mg^+	7	T-N	11
Ca^+	20	T-P	4
NO_3^-	3	Alkalinity	70
SO_4^{2-}	24		

Corn-steep liquor was diluted by the tap water to make up of the influent. The original solution contained about 667 g-COD/L. The 5-day biochemical oxygen demand (BOD) was 72% of the chemical oxygen demand (COD) for influent solutions. Table 5-3 gave a summarization of the characteristics of the feed water. To maintain the operational temperature of 25℃, heaters were used in the wintertime and an air-conditioner was applied in the summer time. The experiment was conducted under relatively dark conditions. Samples were taken every two days.

5.2.2 Startup

For startup of the two reactors, 12.4 g of activated sludge from a lab-scale fill-and-draw batch reactor were placed into each reactor with tap water. The initial MLSS concentration was 1,150 mg/L. Influent feeding started after allowing 30 hours for sludge attachment on BF material (Rouse, et al., 2004).

5.2.3 Analytical methods

5.2.3.1 Analytical method

All the analytical methods were the same as the previous chapter (p.35). The compounds of EPS were determined according to the method mentioned in the previous parts (p.34).

5.2.3.2 Determination of Specific Oxygen Uptake Rate

Sampled the mixed liquor from the reactors (CAS and BF) and harvested by centrifuge at 18,700g for 10min. After discarding the supernatant, the pellets were used for the determination of their SOURs. SOURs were measured for substrate used in this study. Solutions saturated with DO were prepared at an MLVSS of 140-150 mg/L in 150 mL flasks. The DO concentration was monitored under continuous mixing (100-130 rpm). The oxygen uptake rate (SOUR) was determined according the following equations:

$$OUR = -\frac{d[O_2]}{dt} \tag{5-1}$$

$$SOUR = \frac{OUR}{MLVSS} \tag{5-2}$$

where $[O_2]$ is the oxygen concentration (mg-O_2/L), t is time (h), and MLVSS is the volatile MLSS concentration of mixed liquor (g/L).

5.2.3.3 Determination of the observed sludge yield

The observed sludge yeild (Y_{obs}) was calculated according to the following equations:

$$Y_{obs} = \frac{g\text{-}X_{end} - g\text{-}X_{start}}{g\text{-}COD_{removed}} \tag{5-3}$$

$$g\text{-}X_{end} = X_{ST} + X_R + X_E + X_w \tag{5-4}$$

where the term $g\text{-}X_{start}$ is the total amount of seed sludge presented at the initial stage of each Run. The term $g\text{-}X_{end}$ is the sum of the sludge at the end of every Run. This includes the sludge in the reactor (X_R) (in the BF process, X_R is the sum of the suspended solids and the sludge attached on BF material), the sludge in the settling tank (X_{ST}) and the sludge in the effluent (X_E). The term $g\text{-}COD_{removed}$ means the total COD removed in each Run. The term X_w

presents the withdrawn sludge.

5.3 Results and discussion

5.3.1 Degradation performance

The degradations of organic constituents were evaluated by COD removals. The changes of effluent COD concentrations and COD removals of the two processes in Phase I were showed in Fig. 5-2 and Fig. 5-3, respectively. Super and stable COD removal efficiencies were achieved in BF process throughout the whole test with average removal efficiency of 92.5%. The loading shock nearly have no effect on COD removals. Highest COD removal efficiencies were achieved in Run III with respect to the average removal of 94.7% and effluent COD concentration was as low as 10.3 mg/L.

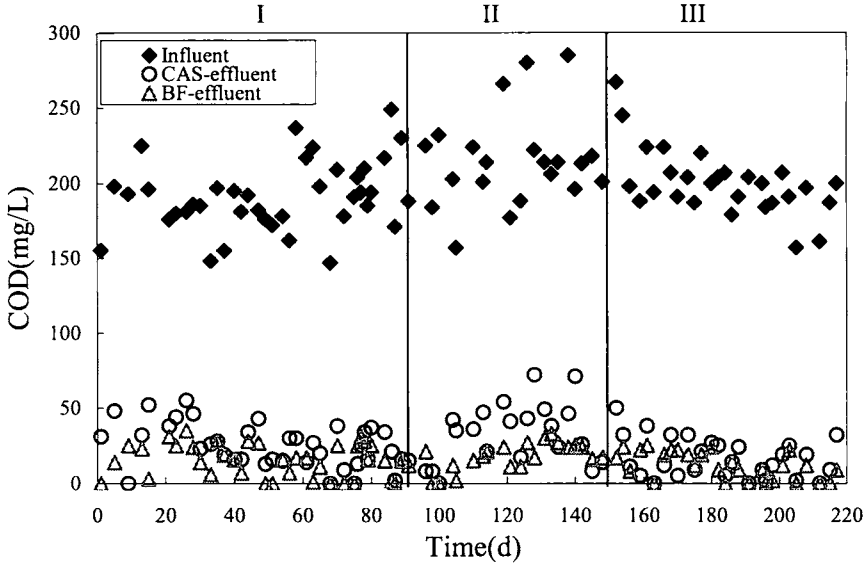


Fig. 5- 2 Time courses of COD changes in phase I

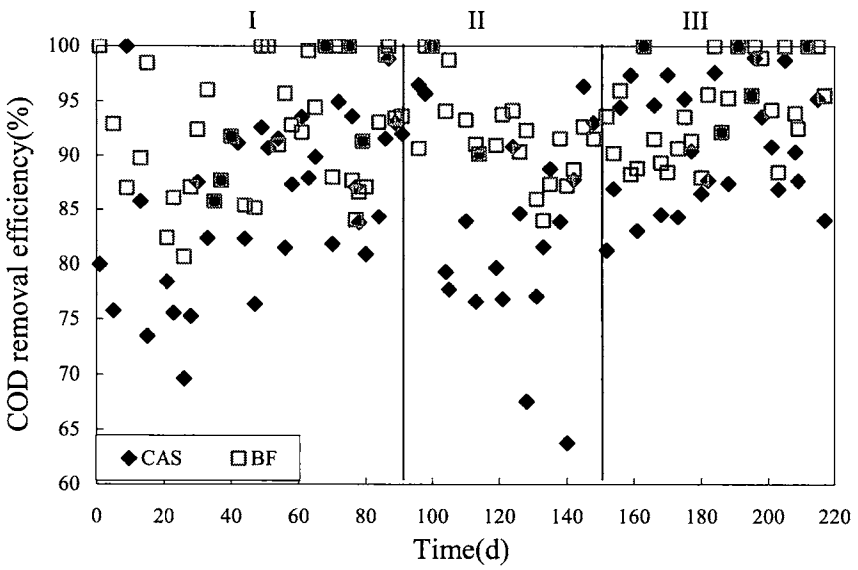


Fig. 5- 3 Time courses of COD removal efficiencies in phase I

Satisfactory and stable effluent COD concentrations were achieved in the CAS process in Run I with an average COD removal efficiency of 86%. They became unstable in Run II. First, dropped sharply below 80% after increased the loadings to 1.0 kg-BOD/m^3/d on day 104, then re-increased to 90% on day 114, but the performances were not stable afterward. The average removal efficiency was calculated to be only 83% for this Run. Increase of COD removal efficiencies with average removal of 87.8% were achieved in CAS process in Run III, which attributed to the new seed sludge.

5.3.2 The effluent suspended solids

Fig. 5-4 showed the daily changes of effluent SS in the CAS and BF processes during Phase I. In general, clear effluents with lower SS were achieved in both processes during Run I, except the result of CAS on day 40 that was as high as 205 mg/L. The considerable high SS obtained related to a sudden change in the sludge particle size distribution in CAS process. The particle size distribution measured on day 37 ranging from 0.67 μm to 8.82 μm, while those on day 41 ranging from 2.98 μm to 394 μm. It was observed on days 40 that some small particles floated on the surface of the CAS's settling tank. Apart from the formation of the big size floc by flocculation, the considerable small size particles should have washed out

together with the effluent, which caused the high effluent SS concentrations.

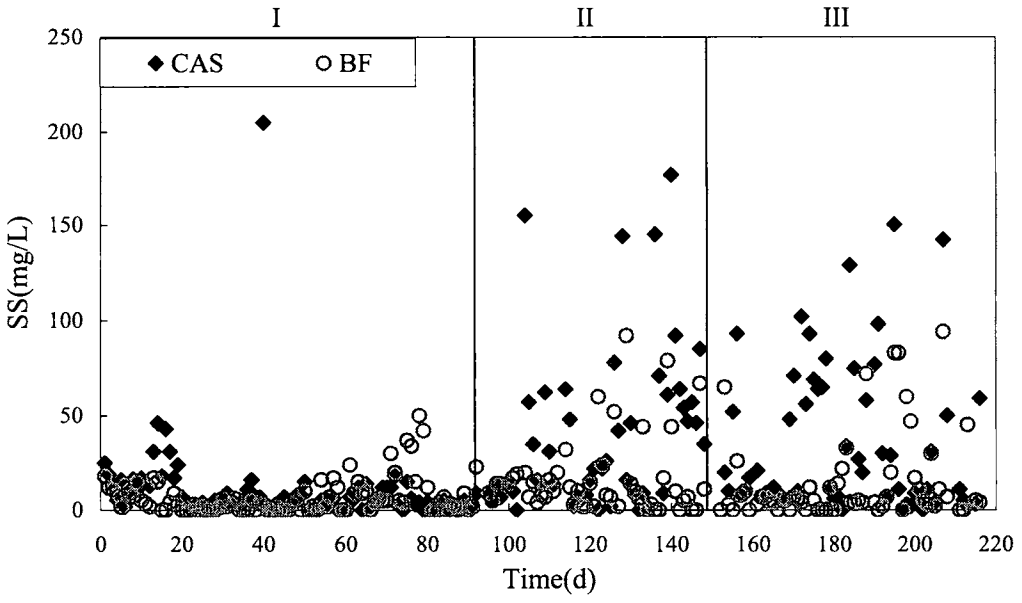


Fig. 5- 4 Time courses of effluent SS in phase I

Reducing airflow rate from 10 L/min to 8 L/min caused an obvious increase in effluent SS for the BF process, which were higher than those of the CAS process. There were also sudden changes in particle size distribution for the BF process responding to the increase of SS levels. It was considered that the reduction of air-flow rate caused this change due to an increase in non-flocculating microorganisms concentrations (Liu, et al., 2005), which resulted in higher effluent SS levels.

The sludge washout in Run II and III caused the unstable effluent SS in both processes. Significant higher average effluent SS of CAS process were obtained both in Run II and Run III, corresponding to the results of 49 mg/L and 99 mg/L, respectively, while those of BF process for Run II and Run III were 15 mg/L and 16 mg/L. The much more frequent sludge washout should account for the higher effluent SS in CAS process.

5.3.3 Nitrogen transformation

Influent nitrogen mainly consisted of organic compounds with respect of T-N concentration

of 13 mg/L. Fig. 5-5, 5-6, and 5-7 showed the daily changes in ammonium, nitrite and nitrate concentrations in both CAS and BF process during Phase I, respectively. The organic nitrogen first was transferred to ammonia-nitrogen by ammonification. Nitrification is carried out in two steps, the conversion of ammonium oxidizing and nitrate oxidizing steps (Shin, et al., 2005). The bacteria responsible for nitrification were autotrophic bacteria. Nitrifying activity was a function of temperature, pH, ammonia concentrations and bacteria concentrations of nitrifiers (Hoilijoki, et al., 1999). It was observed that the reactor MLSS concentrations and the organic loadings have obvious effects on the ammonification and nitrification efficiencies in the present study. The complete ammonifications and higher nitrification efficiencies were achieved both in BF and CAS processes in the first two Runs. The effluent ammonia concentrations decreased to lower than 1 mg/L and kept at those levels after initial 11 days. Nitrate comprised of the main part of the total effluent nitrogen. Increased the VLR to 1.5 kg-BOD/m^3/d on day 155 in Run III, the washout of the suspended solids in each reactor caused obvious increases in effluent ammonium concentrations for both processes. Those observed on day 156 and day 159 in CAS were 1.57 mg/L and 1.86 mg/L, respectively, while that on day 159 of BF process was 2.15 mg/L and the incomplete ammonifictations were obtained in the following 15 days in BF process. The increase BF effluent nitrate concentration of 4.18 mg/L on day 177 indicated the recovery of nitrification performance. The MLSS concentration of that day increased to 1,198 mg/L. However, it was observed that the sharp increase in MLSS concentration had a negative affect on nitrification efficiency on day 180. The sharp increase in MLSS concentration to 3,056 mg/L responded to effluent nitrate concentrations decrease to 0.96 mg/L. The biomass detachment from BF material caused the increase in MLSS concentration at that day, so that it was considered the detached biomass could inhibit the nitrifying bacteria activity. However, the transitory inhibition disappeared on day 182.

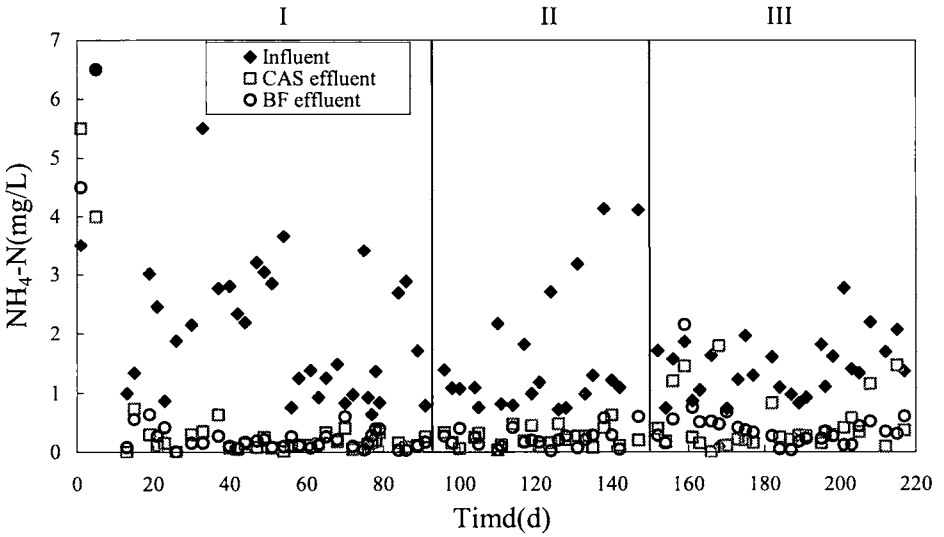


Fig. 5- 5 Time courses of ammonium concentrations of CAS and BF processes in phase I

CAS process demonstrated unstable nitrification performances in Run III due to the sludge washout. Nevertheless good nitrification efficiencies were achieved in CAS process by seeding new sludge in Run III. However, when the VLR in CAS was elevated to 1.5 kg-BOD/m^3/d on day 168, the effluent ammonium concentrations re-increase to 1.79 mg/L. Although the nitrification efficiencies recovered soon on day 170 with effluent ammonium concentration of 0.11 mg/L and high nitrate concentration of 7.8 mg/L, the sludge washout on day 172 and 194 resulted in incomplete ammonification and worse nitrification efficiencies in CAS again.

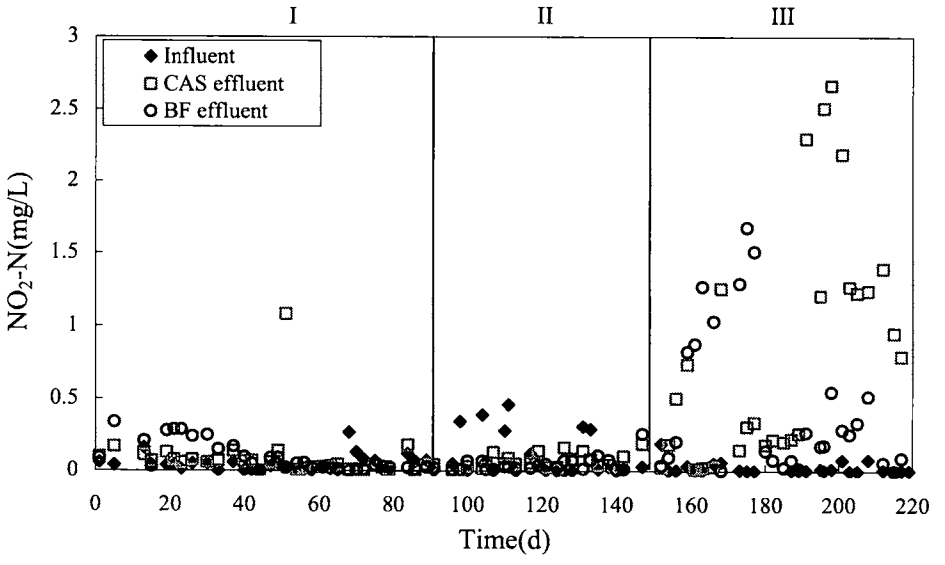


Fig. 5- 6 Time courses of nitrite concentrations for CAS and BF processes in phase I

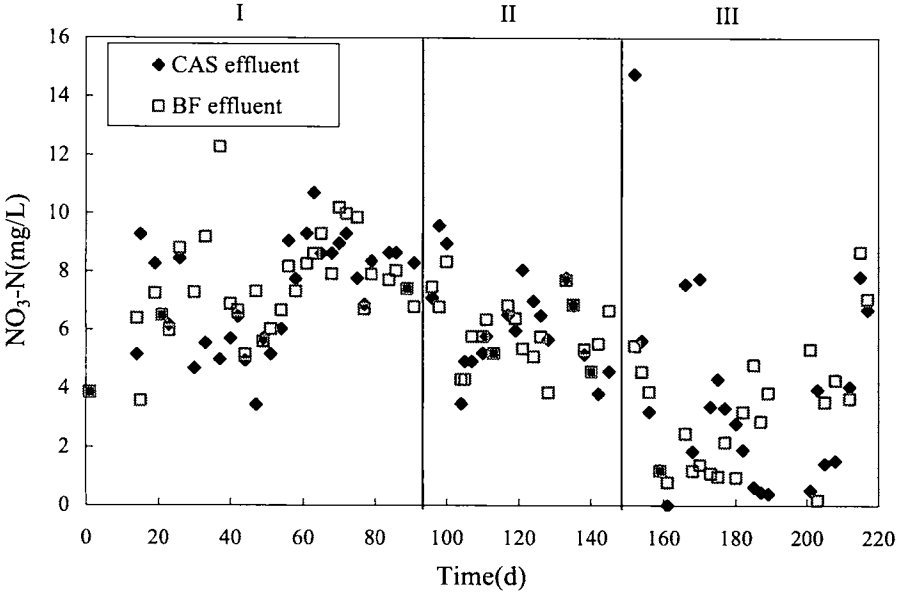


Fig. 5- 7 Time courses of nitrate concentrations for CAS and BF processes in phase I

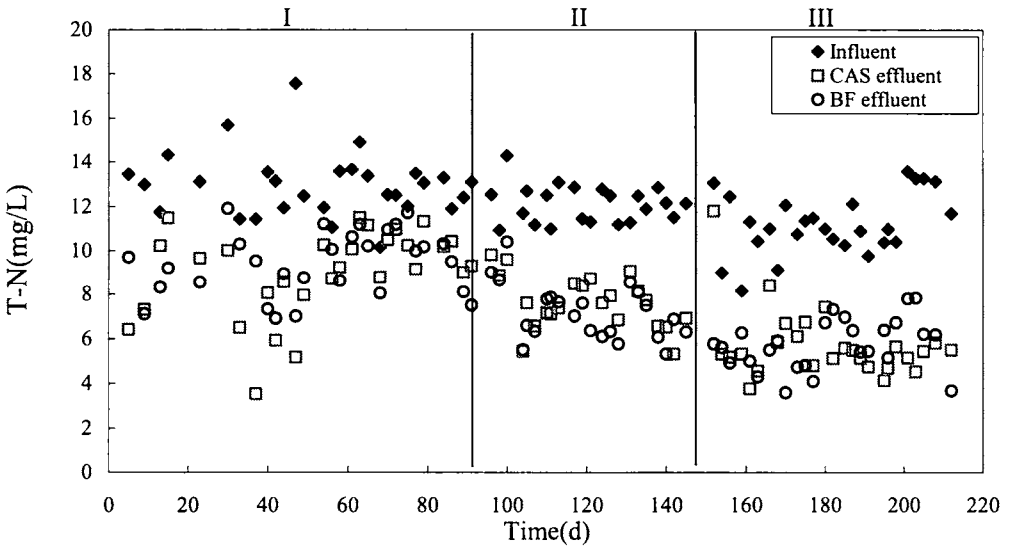


Fig. 5- 8 Time courses of T-N concentrations for CAS and BF processes in phase I

Almost similar nitrogen removal performances were obtained for both processes in Run I, as shown in Fig. 5-8. Significant higher average total nitrogen removal efficiency of 44% was obtained for the BF process in Run II comparing with that of CAS process. The average T-N removal in CAS was 34%. In Run II, these higher T-N removal efficiencies for the BF process can be explained by the creation of an anoxic denitrification zone inside the attached biofilm of the BF biomass carrier. The thickness of the biofilm increased by elevated the volumetric loadings, which should account for the enhanced nitrogen removal capacity. The highest average removal in BF of 57% was achieved in Run III. It was surprised that the average nitrogen removal in Run III for CAS process was around 50%, which should related to the big size flocs developed in CAS process. The anoxic zone should have been contained in them.

5.3.4 Sludge production

Fig. 5-9 showed the changes of the observed sludge yields of CAS and BF processes in Phase I. The sludge yields increased obviously by elevating the VLRs from 0.5 kg-BOD/m^3/d to 1 kg-BOD/m^3/d in both processes. The slight increase of sludge production for CAS in Run III related to the using of the new seed sludge. Comparing with CAS process, sludge reduction ratios in BF process for each loading were calculated to be 35%, 50% and 37% for Run I, II

and III, respectively.

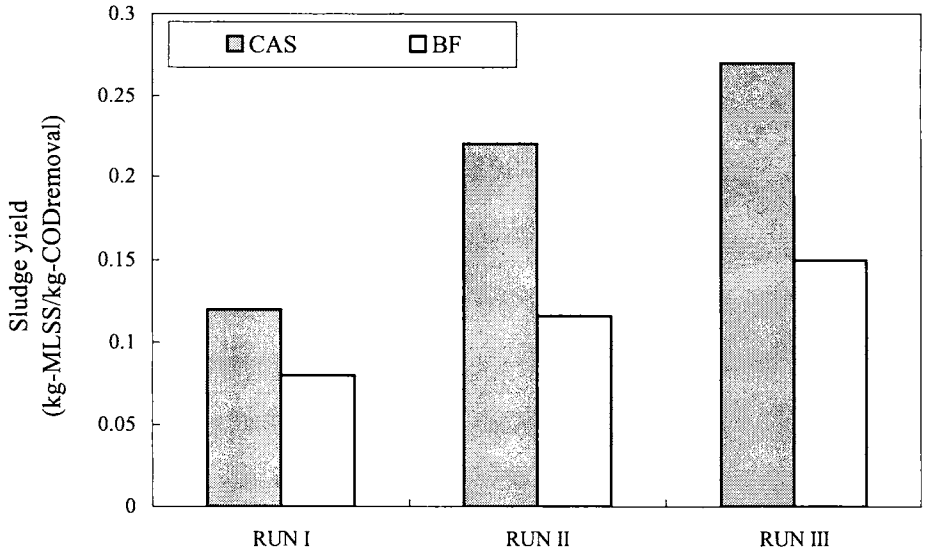


Fig. 5- 9 Sludge yields of CAS and BF processes in phase I

5.3.5 Biomass growth and settling ability

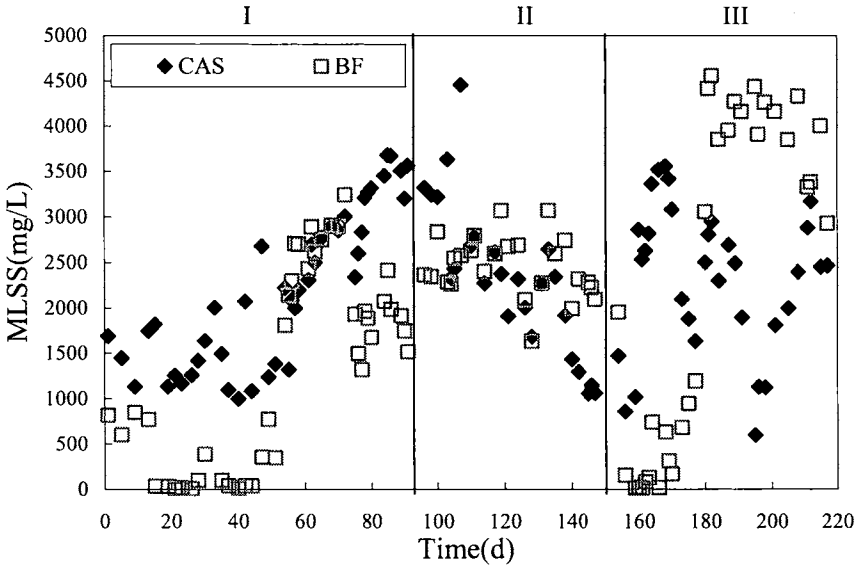


Fig. 5- 10 Time courses of MLSS in CAS and BF processes during Phase I

Fig. 5-10 and 5-11 showed the reactor MLSS and SVI values changing by the time. The accumulated biomass in CAS's settling tank effected the measured MLSS concentrations of reactor at the initial time. Biomass growth from initial 1,150 mg/L to final 3,562 mg/L was

achieved in CAS process in Run I. As soon as increasing the VLR to1 kg-BOD/m^3/d in Run II, around 10 g-TSS washed out from the CAS process, causing the reactor MLSS dropped from 3,634 mg/L to 2,326 mg/L. The accumulated sludge filled in the settling tank of CAS process as well. In order to maintain enough effective biomass for treatment in CAS process, most of the sludge in settling tank was returned to its reactor manually on day 107, resulting in an increase in MLSS concentrations to 4,456 mg/L. However, the biomass could not be kept in CAS reactor well due to the worse settling ability (SVI values varied around 200). MLSS concentration reduced to 2,686 mg/L again on day 110. The sludge bulking in CAS should account for the frequent sludge washout since day 126. The MLSS concentration in CAS reactor dropped to 1,064 mg/L at the end of Run II.

Increasing the VLR from 1 kg-BOD/m^3/d to 1.5 kg-BOD/m^3/d on day 155 in Run III, sludge washout occurred in the reactor and MLSS concentration dropped to 860 mg/L. Due to the shortage of the biomass, the CAS process had to be restarted again on day 169 by seeding 30.9 g-TSS, corresponding to the reactor MLSS concentration of 2,858 mg/L. The SVI value of the new seed sludge was as low as 79. When VLR was elevated to 1.5 kg-BOD/m^3/d in CAS process on day 167 again, the CAS settling tank was full of sludge soon (showed in Fig. 5-12) and the SVI value increased from 74 of day 165 to 118 of day 168. 18.7 g-TSS was withdrawn from the CAS reactor on day 170. The effluent SS decrease to 10 mg/L was obtained on day 171. Elevated effluent SS of 102 mg/L was observed on day 172, indicating that the sludge washout occurred again in CAS reactor. The reactor MLSS concentration on day 173 dropped to 2,100 mg/L and on day 195 was as low as 600 mg/L, which all resulted from the sludge washout. Sludge bulking occurred on day 208 and the SVI value was as high as 292 mg/L in CAS reactor.

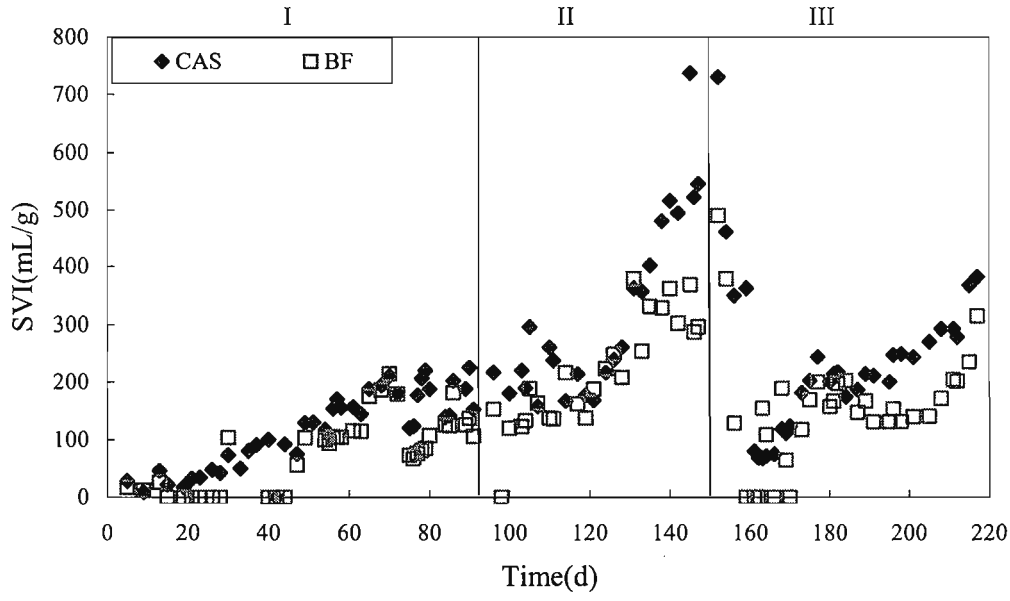


Fig. 5- 11 Time courses of SVI values in CAS and BF processes during Phase I

Large activated sludge flocs developed in the BF process during the initial 46 days of operation when average reactor MLSS concentration was as low as 100 mg/L. On day 47 reactor MLSS concentrations increased suddenly, which were related to the detachment of immobilized activated sludge on the BF material. The reason for this sludge detachment was the decrease in flow velocity caused by the excess accumulation of attached biomass on BF. The increase of suspended biomass enhanced their capacity to compete the nutrient with the attached-growth biomass, which caused the continuous increase of suspended biomass and decrease of attached-growth biomass. In order to attach-immobilize activated sludge on the BF again, the airflow rates were reduced to 8 L/min. Subsequently, the decrease in reactor MLSS concentration implied that the biomass re-attached on the BF successfully. Sludge bulking occurred in the BF process after day 134 in Run II as well, but low effluent SS concentrations were still obtained for the BF process.

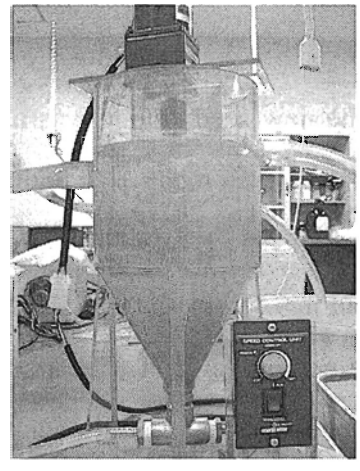


Fig. 5- 12 Photo of CAS settling taken on day 167 in phase I

By increasing the volumetric loading rate to 1.5 kg-BOD/m^3/d, almost all of the suspended solids washed out from the BF reactor and the MLSS concentration dropped to 14 mg/L. The new suspended biomass growth with the MLSS concentrations over 4000 mg/L was obtained in BF again in Run III Sludge bulking was observed in BF process on day 215, which resulted in the sludge washout on day 216 with the reduction of MLSS to 2,928 mg/L.

5.3.6 Sludge activity

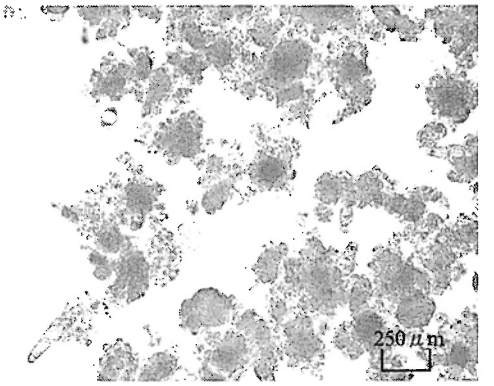
Table 5- 4 Evaluation of SOURs for Phase I and Phase II

	Days	SOUR (g-O_2/kg-MLSS/h)	
		CAS	BF
Phase I	90	3.44	3.78
	147	8.93	13.50
	196	9.20	18.72
Phase II	246	-	26.94
	273	-	40.00

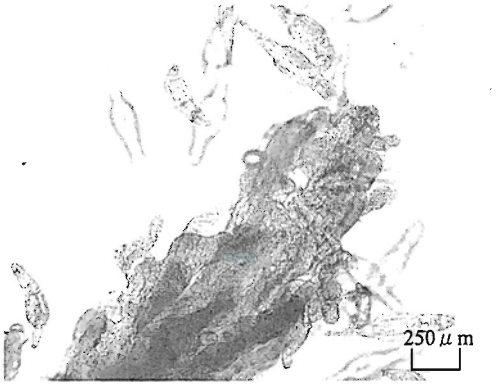
The biomass activities were evaluated by the specific oxygen uptake rate (SOUR). Table 5-4 gave a summary of SOURs values obtained from suspended sludge of the two processes. The SOURs for the suspended sludge of BF were higher than those of CAS. This result illustrated that the suspended sludge had higher activity than that of CAS, which should account for the better removal performance in BF process.

5.3.7 Microbial communities

Seed biomass consisted of granular-like flocs containing *Rotatoria*. After few days of operation, the microorganisms and floc shape changed significantly in the two processes (Fig. 5-13(a)). Filamentous microorganisms were found on day 39 and their numbers increased sharply in the CAS process, while they occurred in the BF process at the end Run I and their growth increased following day 134. The numbers of filamentous microorganisms in the CAS process were much greater than those in the BF process at the end Run II (Fig. 5-13(b)).

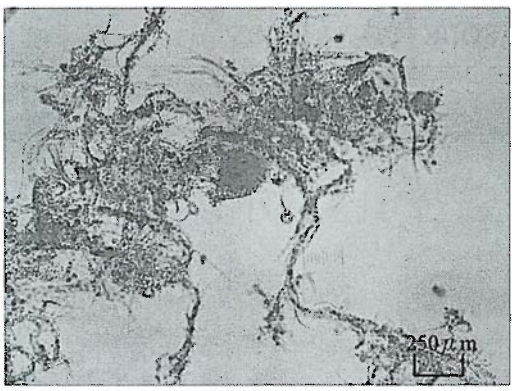


CAS

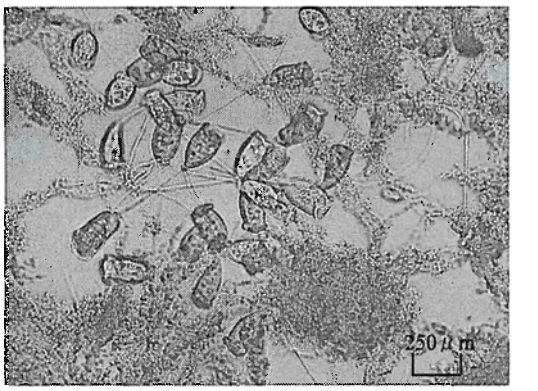


BF

(a) Taken on day 25 in the first Run

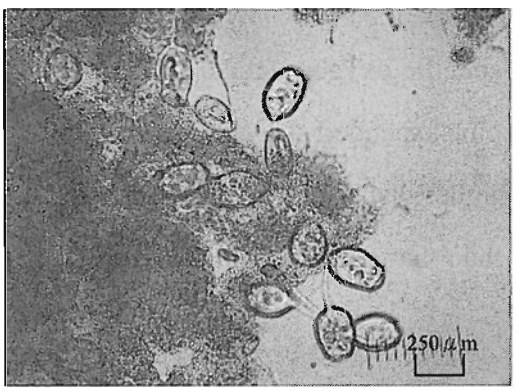


CAS

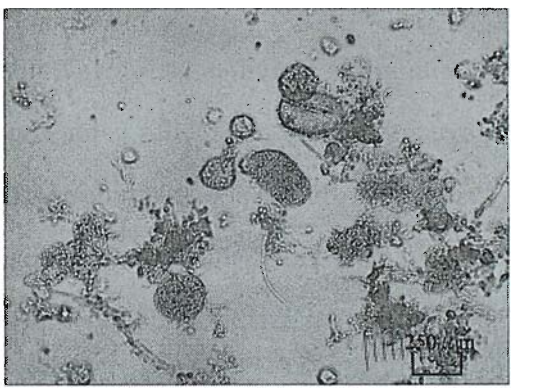


BF

(b) Taken at the end of the second Run



(c) BF sludge taken on day 160



(d) new seed sludge for CAS in Run III

Fig. 5- 13 Microscopic photos for CAS and BF in phase I

The filamentous microorganisms are thought to promote the sludge bulking and to have a negative effect on sludge reduction (Ghyoot, et al., 1999). In Run III, the bulking sludge washed out from both of the reactors after increasing the volumetric loading rate. Afterwards, new suspended flocs developed in the BF process. Many protozoa were with the new floc at

the intial time (see Fig. 5-13(c)). The new seed sludge with very few protozoa and metazoa (Fig. 5-13(d)) was used to restart the CAS; however, the filamentous microorganisms occurred in the both processes at the end of Run III.

Table 5- 5 Enumeration of protozoa and metazoa during Phase I

		Run I		Run II		Run III	
		CAS	BF	CAS	BF	CAS	BF
Bacteria	Filamentous microorganism	+ → ++	+	+++	++	++	+
Protozoa	*Mstigophora*	-	0-5.5	-	-	0	5-8.2
	Ciliophora	2.3-2.8	2.4-3.9	1.8-3.7	4.5-4.9	3.2-3.9	4.4-4.6
	Rotatoria	1.6-2.9	2.8-3.4	1.3-2.7	3.4-3.8	3.1-3.9	1.9-4.6
Metazoa	*Chaetonotus*	2.4-2.8	2.8-3.2	0-2.1	2.2-2.7	2.5-3.0	0-4.2
	Arthropoda	-	0-3	-	-	-	-

Note: Results are given as log (count/mL); +: few, ++: many, +++: dominant

The data showed in Table 5-5 revealed that the occurrences of protozoa and metazoa in the BF process was much more than that of the CAS process during the entire comparison study. Especially when BOD volumetric loading rates increased from 0.5 kg-BOD/m^3/d to 1 kg-BOD/m^3/d on day 103, the numbers of protozoa and metazoa in the CAS process reduced sharply and only *Ciliophora* and *Rotatoria* were fond with numbers of 63 and 10 ml^{-1}, respectively. On the contrary, the increasing numbers of them were observed in the BF process. Moreover, a large number of *Mstigophora* were found in the BF process sludge both in Run I and Run II, while there were no occurrences of them in the CAS process. The growth of *Ciliophora* in a two-stage pure culture chemostat system was associated with 12-43% sludge reduction (Ratsak, et al., 1994). Lee also found that *Ciliophora* and *Mstigophora* contributed to sludge reduction in an aerobic treatment process (Lee, et al., 1996). Lapinski (2003) studied the reduction of suspended biomass in municipal wastewater using *Rotatoria*. *Rotatoria* have been shown to remove wastewater particles from suspension,

partially by consumption and partially by improving the settling ability. They have a beneficial effect on biomass yield and clarity of treated wastewater effluent. After all, many more of protozoa and metazoa occurred in the BF process than that of the CAS process should contribute to the better contamination removal and considerable sludge reduction.

5.3.8 MLSS and SS under short HRT conditions

Fig. 5-14 showed the variation of MLSS and SS concentrations under the shorter HRT in Phase II. When the first phase study finished, all the sludge contained in the BF process (including the suspended and attached biomass) was used as the seed sludge for study of Phase II. Over 4,000 mg/L MLSS were kept in BF process at the initial time of Run IV and effluent SS were lower than 40 mg/L. Sludge bulking caused sludge washout on day 233 and 235, respectively, which resulted in sharply increase of effluent SS levels that accessed 100 mg/L with decrease of MLSS concentrations of 1,758 mg/L. It was difficult to maintain the suspended solids in the reactor at the loading of 3 kg-BOD/m^3/d in Run V due to the serous bulking occurrence. The flocs with big size and lower density floated in the reactor, which make the sludge recycle become difficult and most of the sludge lost in the effluent instead of returning to the reactor. Except the initial time and sudden increase on day 257, MLSS concentrations never suppressed 500 mg/L. The sudden increase on day 257 associated with the biofilm detached from BF. In order to explore the optimized operational conditions under short HRT with sludge bulking, the reactor was restarted to repeat the study at a loading of 2 kg-BOD/m^3/d in Run VI. There was no recovery of the flocs from the bulking situation with the loading reduction. Even the biofilm detached process led to an obvious increase to 1,486 mg/L, it was still difficult to achieve the high MLSS concentrations as RUU IV. The effluent is not clear and contained some floated floc with big size. Increasing the loading to 2.5 kg-BOD/m^3/d in Run VII, MLSS concentrations ranged around 1,000 mg/L. Continuous operation of the BF reactor without settling tank in Run VIII led to a slight decrease in MLSS concentrations. Unclear effluents with variable SS were obtained since Run VI.

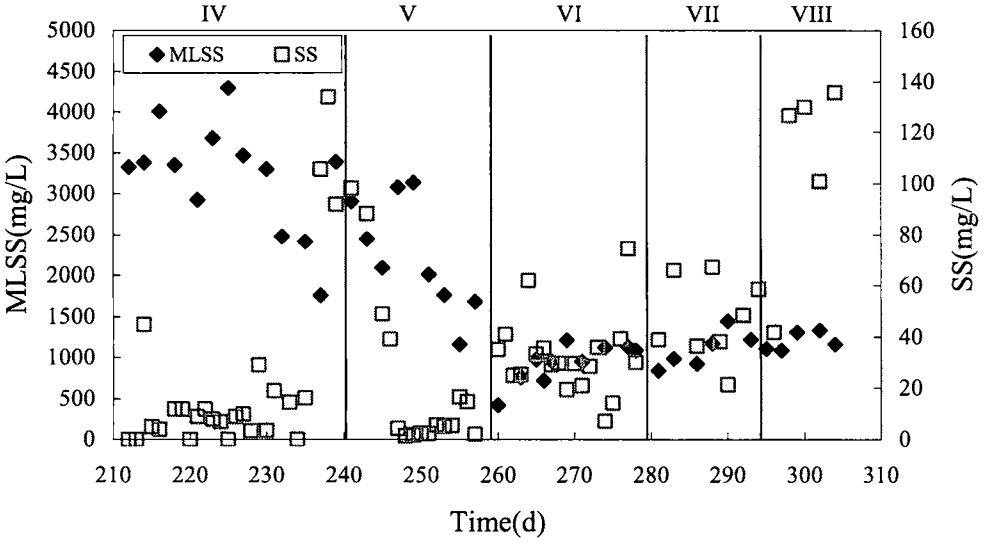


Fig. 5- 14 Time courses of rector MLSS and effluent SS during phase II

5.3.9 Treatment performance under short HRT conditions

The BF process with sludge recycles still showed high contaminant treatment capacity with average COD removal over 85% even with shorter HRT of 1.2h, corresponding to the loadings of 3 kg-BOD/m^3/d (showed in Fig. 5-15). The operation of Run VIII without sludge recycle resulted in slight decrease in the removal with average COD removal efficiency of 82%. The ammonification and nitrification efficiencies were observed to be associated with the suspended solids concentrations. Complete ammonification and higher nitrification efficiencies were obtained only in Run IV (showed in Fig. 5-16). Over 50% of the nitrogen removals (showed in Fig. 5-17) were achieved throughout the whole test, which attributed to a large amount of sludge attached on the BF material.

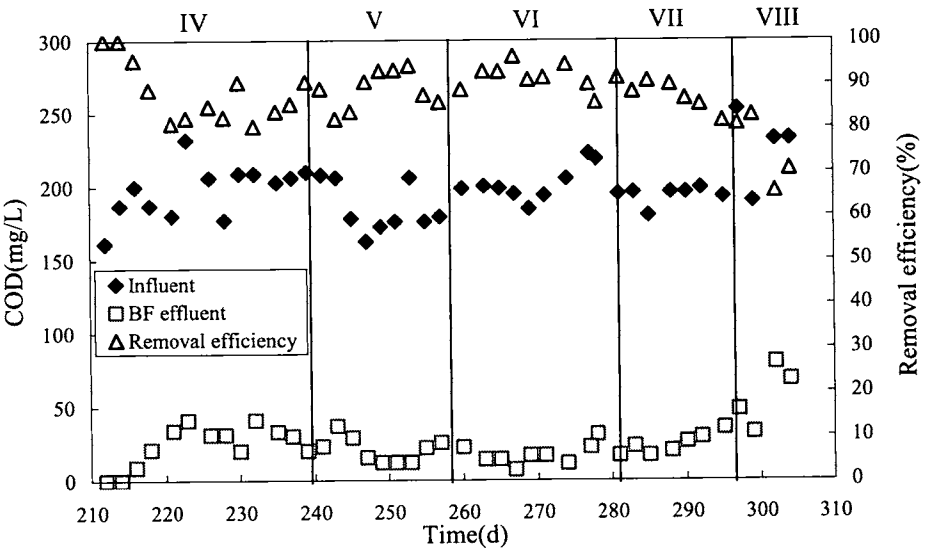


Fig. 5- 15 Time courses of COD concentrations during phase II

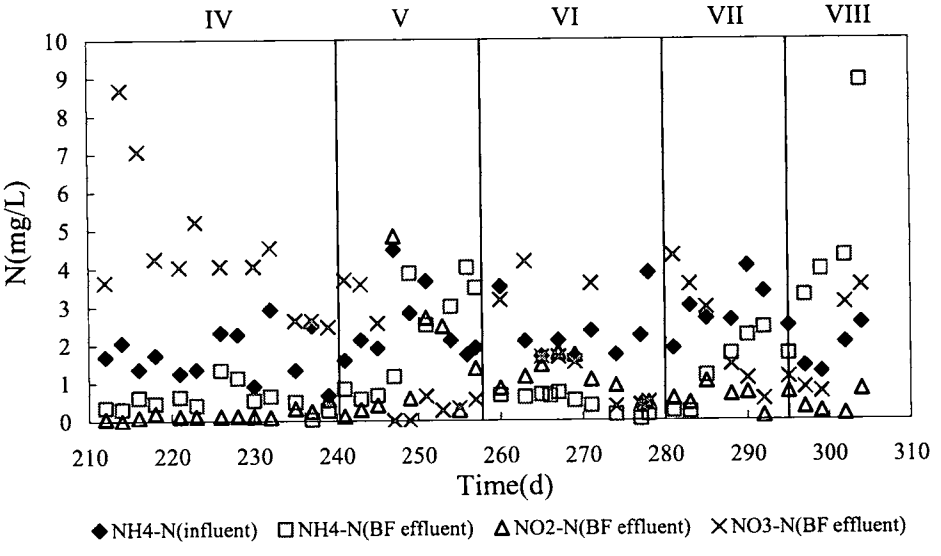


Fig. 5- 126 Time courses of ammonium, nitrite, and nitrate concentrations during phase II

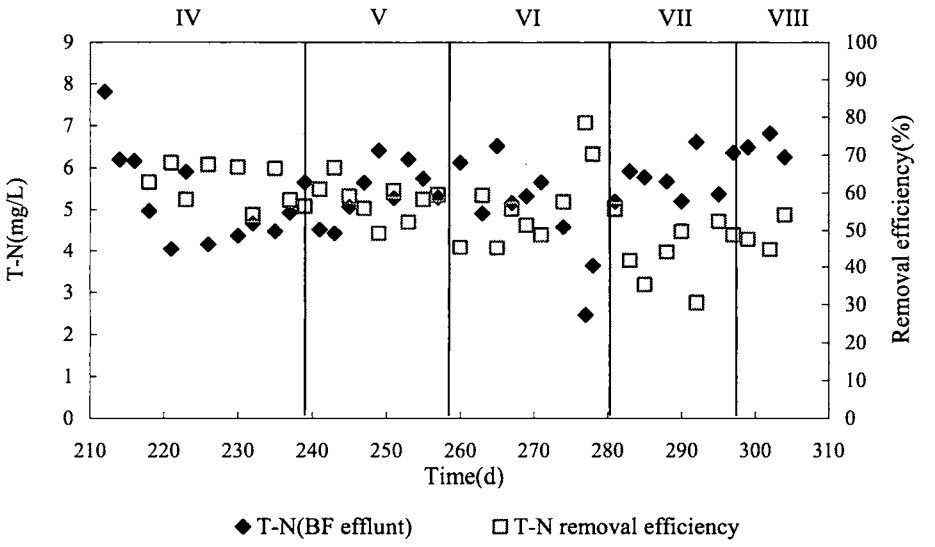
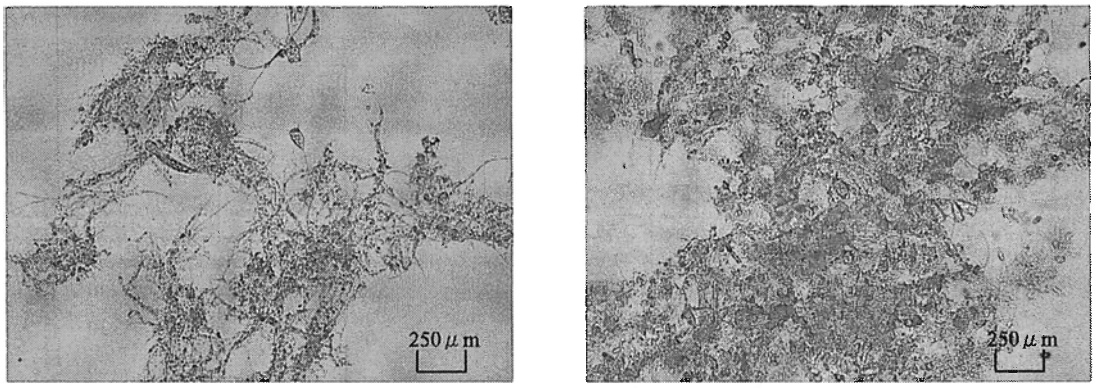


Fig. 5- 17 Time courses of TN effluent concentration and removal during phase II

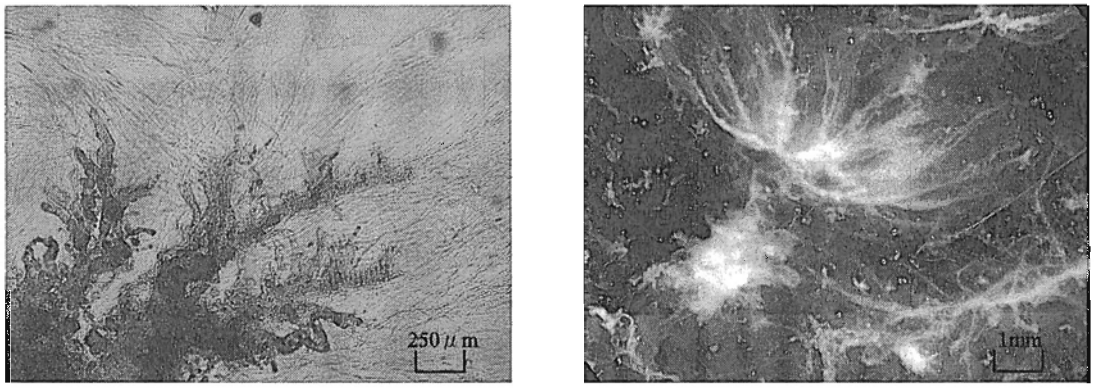
5.3.10 Sludge bulking

The CAS process is limited by the ability to separate the biomass from the effluent (Tixier, et al., 2003). Sludge bulking in the present research work caused the sludge washout and unclear effluent. Sludge bulking is divided as filamentous bulking and viscous bulking. The bulking was observed on day 126 in CAS and day 136 in BF process belonging to the filamentous bulking (showed in Fig. 5-18 (a)). Filamentous microorganisms form a part of the bacterial population that plays a predominant role in the bacterial community of the activated sludge; they can serve as "backbones" for flocs and then favor its growth (Sezgin, et al., 1978; Cenens, et al., 2000). When the filaments present in too great numbers the sludge flocs are bound together by the filaments in a web-like structure resulting in a very poor settling sludge (Madoni, et al., 1999). The filamentous microorganisms observed in this study was the filamentous bacteria type 021N. The low strength feed water with easily degradable characteristic should be responsible for the growth of them.



CAS BF

(a) sludge bulking in phase I



(b) sludge bulking in phase II

Fig. 5- 138 Photos of bulking sludge in phase I and phase II

The sludge bulking in the BF process in the Phase II was recognized as both viscous bulking and filamentous bulking. Viscous bulking was caused by the excess growth of *Zoogloea* spp., *Zoogloea* spp. play an important role for the solid aggregation and contained high levels of EPS (Wanner, 1997). The EPS contained in the suspended sludge of BF process were determined as high as 74% at the end operation of Run V in Phase II. The snow-like shape was observed at that time (showed in Fig. 5-18 (b)). The components that could be biodegraded easily maintained in the reactor should accede the growth of *Zoogloea* spp.. Treating the corn-steep liquor under the short HRT would make this kind of components increase in the reactor, which led to the viscous bulking. There was no effective way to deal the viscous bulking till now (Wanner, 1997).

5.3.11 Operational strategies under short HRT conditions with sludge bulking

The sludge production with 0.22 kg-MLSS/g-$COD_{removed}$ was obtained at the loading of 2 kg-BOD/m^3/d in Run IV. This value was higher than that for 1.5 kg-BOD/m^3/d. The sludge flocs changed drastically under the loading of 3 kg-BOD/m^3/d. In order to make clear the biological mechanism, the experiment under 2 kg-BOD/m^3/d was repeated in Run VI. The reactor was restarted by seeding the sludge taken from Run IV. There were nearly no changes of the floc characteristics when reduced the loading to 2 kg-BOD/m^3/d. The suspended flocs with snow flower shape were big and light, difficult to settle down. The higher sludge viscosity and heavy filamentous microorganism growth made the biomass attach on BF toughly, which prevented the biofilm refresh. The biofilm was so thick as to attach on the reactor wall easily. In order to replace the attached-growth biomass, the reactor wall was cleaned as frequently as possible and the floated sludge in the settling tank was withdrawn often. After two weeks' operation, the slight recovery was observed, MLSS concentrations increased to around 1,000 mg/L and most of the floc could settle down (SV_{30} decreased to 40, but there still were some floc floated in the supernatant). Since the viscous bulking could not be controlled completely, it was impossible to maintain high concentrations of suspended solids in the reactor any more. With same loading and different sludge characteristic the performances of Run IV and Run VI were compared (showed in Table 5-6). The results shown in this Table presented average value of each parameter. Almost similar contamination removals were obtained in both of the Runs except the higher nitrification efficiencies in Run IV. Lower observed sludge yield was obtained in Run VI compared with that of Run IV. It was thought that the more suspended solids, the more sludge washout. Sludge washout would encourage the biomass growth, which could lead to the higher sludge yield. This suggested that it was not a bad choice to operate the reactor under lower MLSS concentrations under short HRT with sludge bulking. In addiction, since the bulking sludge floated in the settling tank and difficult to be returned to the reactor, the settler could not keep its function for solid-liquid separation. Therefore, the settler should be omitted from the process. This idea was applied in the study of Run VIII.

Table 5- 6 Comparison of experimental results in Run IV and Run VI

	COD removal (%)	NH_4-N (mg/L)	NO_3-N (mg/L)	NO_2-N (mg/L)	T-N removal (%)	Y_{obs}
Run IV	88.1	0.6	4.6	0.2	63.9	0.19
Run VI	89.1	0.6	1.1	1.9	54.9	0.22

Table 5- 7 Comparison of experimental results in Run VII and Run VIII

	COD removal (%)	NH_4-N (mg/L)	NO_3-N (mg/L)	NO_2-N (mg/L)	T-N removal (%)	SS (mg/L)
Run VII	86.4	2.1	0.6	0.2	49.1	81.5
Run VII	82.2	3.6	0.3	0.02	48.2	95.6

The continuous experiment was conducted through Run IV and Run V at the loading of 2.5 kg-BOD/m^3/d, the operation of Run VIII with settling tank and Run VIII without settling tank. The results of the two Runs were compared in Table 5-7. It illustrated that the operation without settling tank just resulted in slight decrease of contamination removals and slight increase in average effluent SS concentrations. Based on the economical point of view, the application of settling tank could not only increase the space requirement but also the operational costs. Sludge bulking was difficult to be avoided for the treatment of the easily biodegradable wastewater with low strength. In addition, it was well known that the attached-growth biomass has many advantages than those of the suspended growth. As mentioned above, the growth of suspended biomass should compete the nutrient with that of attached-growth in an attached-growth process with sludge recycle. The attached-growth biomass would be failed in the competition with the suspended-growth biomass when fed by the low-strength wastewater, which led to the attach-growth biomass become fewer, So that, the attached-growth process could not exhibit its advantage well. All in all, it is an essential way to encourage the growth of biofilm under the short HRT with sludge bulking.

5.4 Conclusions

1. Compared with a CAS process, the BF process demonstrated good resistance to shock loadings with superior and stable contamination removal efficiencies. Satisfactory removal efficiencies were obtained in the BF process at a short HRT of 1.2 h with a volumetric loading rate of 3 kg-BOD/m^3/d. Slight decreases in COD removals were obtained under operation without sludge recirculation in the BF process.

2. Clear effluents were achieved for both processes in Run I. Significantly higher effluent SS concentrations were obtained in the CAS process than those in the BF process in the last two Runs of Phase I. Unclear effluent containing unsettling flocs were observed since Run V in Phase II.

3. Nitrification efficiencies were affected by the loadings and reactor suspended solids concentrations. During Phase I, good nitrification performances were obtained both in CAS and BF processes throughout the first two runs with unstable nitrification efficiencies. The BF process occasionally had incomplete ammonification performances in Phase II. Enhanced nitrogen removals were achieved for the BF process by elevating the loadings in Phase I. Over 50% nitrogen removals were obtained in the BF process under operation with short HRTs in Phase II.

4. Sludge production was reduced significantly in the BF process compared to that of the CAS process in Phase I . In Run III, the CAS process was restarted by seeding with new sludge, 37% of sludge reduction was achieved in the BF. The results of Run IV and Run VI illustrated that with the bulking condition the more suspended sludge contained in the BF reactor, the higher sludge production occurred with bulking conditions.

5. Sludge bulking caused deteriorated effluent quality, which was difficult to avoid for the treatment of easily biodegradable low-stength wastewater. Operation of the BF reactor without sludge recycle was recommended to treat this kind of wastewater.

5.5 References

APHA, AWWA, and WEF: Standard Methods for the Examination of Water and Wastewater, 19th Ed. American Public Health Association, Washington, D. C. (1995).

Cenens C., Smets I.Y., and Van Impe J.F. (2000) Modelling the competition between floc-forming and filamentous bacteria in activated sludge wastewater treatment systems_II. A prototype mathematical model based on kinetic selection and filamentous backbone theory, Water Res., 34, 2535-2541.

Ghyoot W. and Verstraete W. (1999) Reduced sludge production in a two-stage membrane-assisted bioreactor, Wat. Res., 34(1), 205-215.

Hoilijoki T.H., Kettunen R.H., and Rintala J.A. (1999) Nitrification of anaerobically pretreated municipal landfill leachate at low themperature, Wat. Res., 34(5), 1435-1446.

Lapinski J., and Tunnacliffe A. (2003) Reduction of suspended biomasss in municipal wastewater suing bdelloid rotifers, Water research, 37, 2027-2034.

Lazarova V. and Manem J. (1994) Advances in biofilm aeroic reactors ensuring effective biofilm control, Wat. Sci. Tech.29 (10/1 1), 319-327.

Lazarova V. and Manem J. (1996) An innovative process for waste water treatment: the circulating floating bed reactor, Water Science and Technology 34 (9), 89-99.

Lee J.W., Cha H.-Y., Park K.Y., Song K.-G., and Ahn K-H. (2005) Operational strategies for an activated sludge process in conjunction with ozone oxidation for zero excess sludge production during winter season, Water Research 39, 1199-1204.

Lee M.N. and Welander T. (1996) Reducing sludge production in aerobic wastewater treatment through manipulation of the ecosystem, Wat.Res., 30(8), 1781-1790.

Liu Q.S., Liu Y., Tay J.-H., and Show K.Y. (2005) Responses of sludge flocs to shear strengh, Process Biochemistry, 40, 3213-3217.

Loukidou M.X. and Zouboulis A.I. (2001) Comparison of two biological treatment processes using attached-growth biomass for sanitary landfill leachate treatment, Environmental Pollution 111, 273-281.

Madoni P., Davoli D., and gibin G. (1999) Survey of filamentous ,icroorganisms from bulking and foaming activated-sludge plants in Italy, Wat. Res., 34(6), 1767-1772.

Ratsak C.H., Kooi B.W., and van Verseveld H.W. (1994) Biomass reduction and mineralization increase

due to the ciliate Tetrahymena Pyriformis grazing on the bacterium Pseudomonas Fluorenscens, Water Sci. Technol., 29(7), 119-128.

Rouse J.D., Yazaki D., Cheng Y.J., Koyama T., and Furukawa K. (2004) Swim-bed technology as an innovative attached-grouh process for high-rate wastewater treatment, Japanese of Water Treatment Biology 40(3), 115-124.

Pastorelli G., Andreottola G., Canziani R., Darriulat C., de Fraja Frangipane E., and Rozzi, A. (1997a) Organic carbon and nitrogen removal in moving-bed biofilm reactors, Water Science and Technology, 35 (6), 91-99.

Pastorelli G., Andreottola G., Canziani R., de Fraja Frangipane E., De Pascalis F., Gurrieri G., and Rozzi A. (1997b) Pilot-plant experiments with moving-bed biofilm reactors, Water Science and Technology 36 (1), 43-50.

Sezgin M, Jenkins D., and Parker D. (1978) A unified theory of filamentous activated sludge buling, J. Water Pollut. Fed., 50, 362-381.

Shin J.H., Sang B.I., Chung Y.C., and Choung Y.K. (2005) The removal of nitrogen using an autotrophic hybrid hollow-fiber membrane biofilm reactor, Desalination, 183, 447-454.

Tixier H., G. Guibaud, and M. Baudu (2003) Towards a rheological parameter for activated sludge bulking characterization, Enzyme and Microial Technology, 33, 292-298.

Wanner J. (1997) Activated sludge bulking and foaming control, Technomic Pulishing Company, Inc., Lancaster PA, 17604USA.

Wei Y.S., Houten R.T.V., Borger R.A., Eikelboom H.D., and Fan Y.B. (2003) Minimization of excess sludge production for biological wastewater treatment, Water Research, 37, 4453-4467.

Chapter 6 Treatment performances of swim-bed technology for different low-strength wastewater

6.1 Introduction

When domestic wastewater is discharged to receiving water bodies without treatment or disinfection, if contaminates the water environment with high concentrations of bacteria, viruses and parasites, creating health problems. Untreated domestic wastewater has a complete spectrum of pathogenic microorganisms of human origins, including protozoa, bacteria and viruses. Low-strength wastewaters (COD < 1,000 mg/l), such as domestic sewage, traditionally have been treated under aerobic conditions, but the treatment costs are high.

Swim-bed process was considered to provide an effective way for the domestic wastewater treatment. The lab-scale study had already been conducted (presented in Chapter 5) by using corn-steep liquor and synthetic wastewater. Good treatment performances were achieved except the sludge-bulking occurrence. The sludge bulking was thought to be related to the easily degradation characteristic of corn-steep liquor (Test 1). In order to evaluate the swim-bed process performance on real domestic waster treatment well, the study was repeated by using the mixture of peptone and meat as the carbon sources (Test 2). Meanwhile, a bench-scale study carried out to treat the industrial wastewater contained polyvinyl chloride with almost similar operational conditions as the lab-scale study (Test 3). The treatment performances and sludge characteristics of the three Tests were compared experimentally.

6.2 Materials and methods

6.2.1 Characteristics of feed waters

Table 6- 1 Characteristics of feed waters

	BOD_5 (mg/L)	COD_{Cr} (mg/L)	SS (mg/L)
Test 1	150	208	0
Test 2	150	203	0
Test 3	200	369	20

The influent components of Test 1were previously described in Chapter 5. The diluted solution of corn-steep liquor could be degraded so easily that 25% of COD was lost after one day in storage. The feed water of Test 2 was diluted with a mixture of peptone and meat extract, containing various composing organic compounds. The BOD_5 was 74% of the COD in this feed water, which could be kept stably for only one day. The feed water of Test 3 was a polyvinyl chloride wastewater from chemical industrial plant. In order to maintain a stable BOD_5 of 200mg/L, a diluted methanol solution was added daily. H_2SO_4 and H_3PO_4 were used for neutralization and phosphorus addition, respectively. The characteristics of three kinds of feed waters are showed in Table 6-1.

6.2.2 Operational conditions and experimental set-up

Similar operational conditions were applied for the three tests (Table 6-2). The same reactors packed with BF material described in Chapter 5 (Fig. 5-1) were used with a 50% BF filling ratio. Volume of the settling tanks used in Test 2 and Test 3 were 5 L with surface areas was 0.04 m^2, each. In Test 3, a diluted methanol solution was added to the BF reactor by a pump as showed in Fig. 6-1. The three tests were conducted under similar operational conditions as described in Table 6-2 though with a higher BOD loading rate in o Test 3.

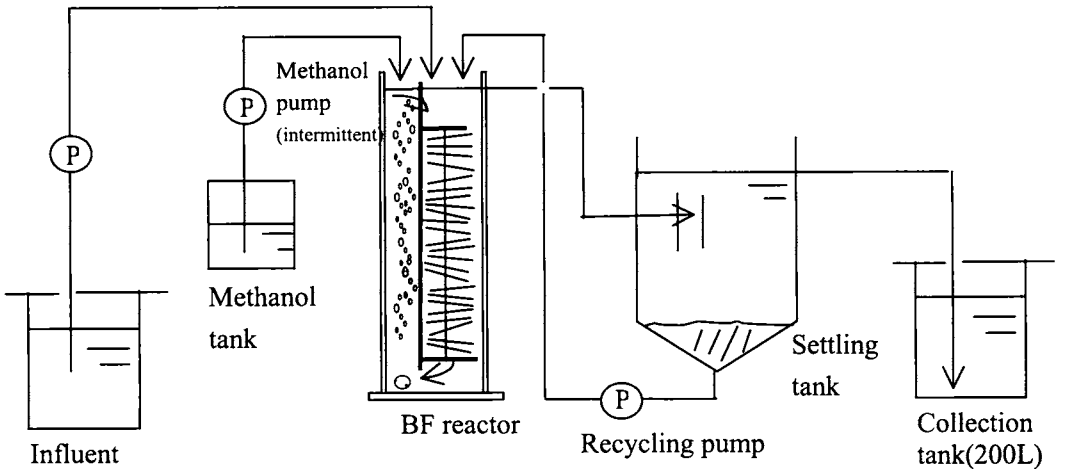


Fig. 6- 1 Schematic diagram of experiment Test 3

Table 6- 2 Operational conditions

	Test 1	Test 2	Test 3
BOD loading rate (kg-BOD/m^3/d)	0.5-3	0.5-2.5	1.1-4.6
Temperature(℃)	25	25	uncontrolled
DO (mg/L)	>2	>2	>2
Air flow rate (L/min)	10	10	10
Recycle rate (%)	100-50	100	100

6.2.3 Analytical methods

All the analytical methods were the same as used in the previous chapter (p.35).

6.3 Results and discussion

6.3.1 Treatment performances

The treatment performances of the three tests are compared in Table 6-3. The degradation of organic contaminant was evaluated by BOD and COD removals. Good removal efficiencies

were obtained for every test, while that in Test 3 was much more stable. Lower and constant effluent SS was achieved in Test 3. On the contrary, they varied a lot in Test 1 and Test 2.

Table 6- 3 Comparison of treatment performances

	BOD removal (%)	COD removal (%)	SS (mg/L)	MLSS (mg/L)	SVI
Systerm1	92-97	82-100	0-132	6-4,558	<1
Systerm2	93-97	87-98	0-132	11-3,226	<1
Systerm3	91-98	92-95	21-43	3,200-12,000	31-156

6.3.2 Sludge characteristics

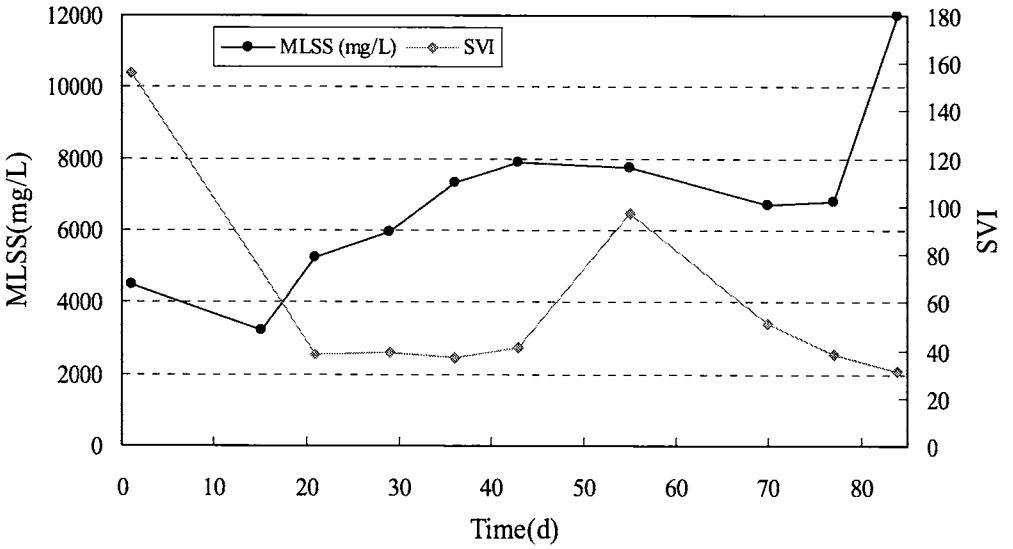
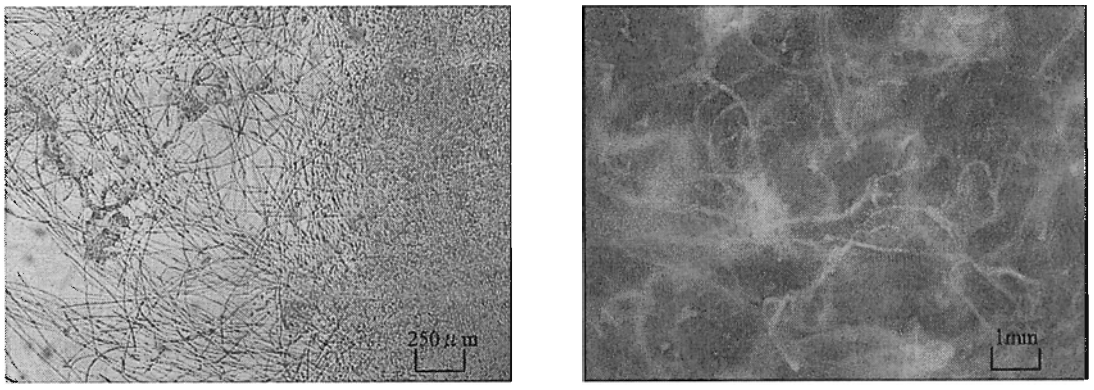


Fig. 6- 2 Time courses of MLSS versus SVI

The MLSS concentrations in Test 3 increased stepwisely, which could account for the better and more stable contaminant removals. Sludge bulking occurred both in Test 1 and Test 2 at BOD loading rates greater than 0.5 kg-BOD/m^3/d. This sludge bulking was difficult to be controlled, which resulted in a decrease in MLSS levels and unclear effluent with highly variable SS levels. The bulking sludge in Test 1 is shown in Fig. 5-18 and that of Test 2 is

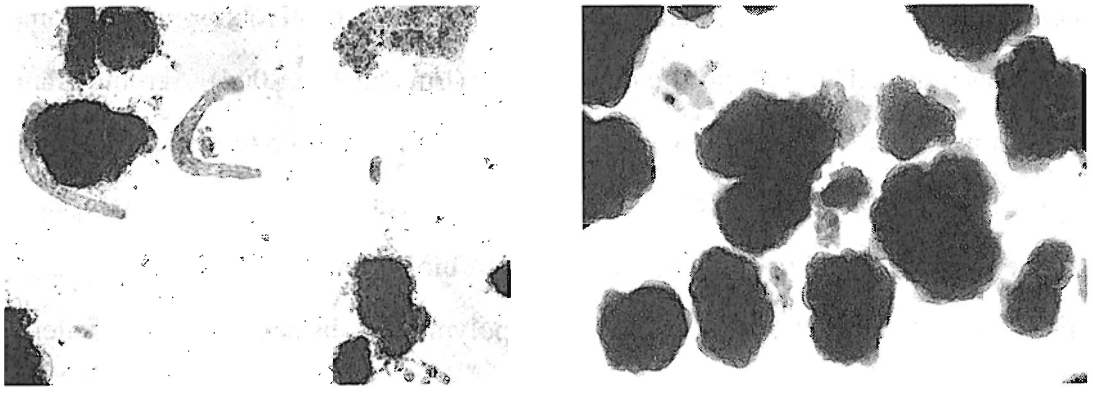
showed in Fig. 6-3. Heavy filamentous growth was observed in both; however, spherical microbial aggregates were also discovered in Test 3 (showed in Fig. 6-4), which was same as in the previous study (described in Chapter 4). The microbial granules demonstrated good settling characteristics and allowed for high sludge retention with MLSS concentration of 12 g/L in the reactor. It was reported that the granules are dense microbial consortia packed with different bacterial species and typically contain millions of organisms per gram of biomass (Liu, et al, 2004). The remarkable removal performances obtained from the treatment of the polyvinyl chloride wastewater should contribute to the granular flocs formation.

The different feed sources resulted in different sludge characteristics in the present study, which resulted in different effluent qualities. The growth of aerobic granules sometimes is regarded as a special case of biofilm development (Liu and Tay, 2004). Aerobic granulation is a process in which suspended biomass aggregate and form discrete well-defined granules in aerobic Tests (Moy, et al., 2002; Liu, et al., 2005). As a complex process, granulation is dominated by many physicochemical and microbiological factors, including seed sludge source, substrate characteristics, extracellular polymeric substances (EPS), divalent cations, pH levels, temperature and reactor operations (Dignae, et al., 1998). The formation of the granules in Test 3 was firstly, was thought to be due to the poor degradation characteristic of polyvinyl chloride wastewater inhibiting the excess growth of filamentous organisms. Secondly, the inorganic suspended solids contained in the industrial wastewater influent should act as carriers for the first attachment of biomass, which also enhances the density and stability of the granules. Thirdly, the existence of BF material should be beneficial for the microbial aggregates since a higher level of EPS should be contained in this process. However, on the other hand, the high levels of EPS also served as an indicator of the excess growth of *Zoogloea* spp., as discussed in Chapter 5, which can cause viscous bulking. Some researchers have suggested that the EPS play a positive role in floc formation. Further study should be focused on the investigation of the physical and chemical characteristics of suspended flocs in the swim-bed process and the optimization of operational conditions for achieving good sludge-water separation.



(a) (b)

Fig. 6- 3 Bulking sludge in Test 2 (taken on day 58)



(a) taken on day 37 (b) taken on day 75

Fig. 6- 4 Sludge microscopic photos in Test 3

6.3.3 Granulate mechanism in swim-bed process

Super treatment efficiencies were achieved in the study of Part 4 and Part 5, which should be related to the granular-like floc formation. Give an overview on the granular formation conditions, the MLSS concentrations all suppressed 5 g/L and F/M rations were all around 0.2-0.4. It was difficult to maintain the MLSS concentrations in the conventional activated sludge process as high as that in swim-bed process, which should relate to the characteristic of the attached-growth biomass in the BF material. The suspended solids in BF reactor first came from the fragment of the biofilm. The detached biofilm contained anaerobic zone with big density, of which the settling ability was good. The anaerobic zone was thought as the initial center of the granular sludge. The high level of EPS contained in the suspended sludge

should account for the primarily flocculation of the floc. The increase in MLSS concentrations in the reactor corresponded to the increase in the VLRs stepwise. The shear stress between the floc would increase with the increase of the MLSS concentrations. Ultimately, the granulate sludge with smooth surface would form when the shear stress become balanceable. Meanwhile, the F/M ratios should decrease with the increase in the MLSS concentrations. Keeping the operation at one loading, biomass growth would become slower and the excess EPS should be consumed by the cell. So the stable granular floc should form. This is just the hypothesis coming from the occasional experiment results, which should be tested and proved in the further study.

6.4 Conclusions

The swim-bed technology demonstrated variable performances for treatment of different kinds of low-strength wastewaters. Good contaminants removals were obtained for all of the three tests. However, sludge bulking occurred in Test 1 and Test 2 resulting in unclear effluent with variable SS concentrations. The microbial granules formed in Test 3 led to greatly improved sludge sedimentation with clear effluent. The significant results obtained in Test 3 illustrated that the swim-bed process has potential for treating low-strength wastewater consisting of recalcitrant compounds.

6.5 References

APHA, AWWA, and WEF: Standard Methods for the Examination of Water and Wastewater, 19th Ed. American Public Health Association, Washington, D. C. (1995).

Dignae M.-F., V. Urbain, D. Rybacki, A. Bruchet, D. Shidaro, and Scribe P. (1998) Chemical description of extracellular polymers: implication on activated sludge floc structure, Water Sci. Technol., 38, 45-53.

Liu S.F., Q.S. Tay, and Liu Y. (2004) Growth kinetics of aerobic granules developed in sequencing batch reactors. Lett. Appl. Microbiol. 38, 106-112.

Liu Y. and Tay J.-H. (2004) State of the art of biogranulation technology for wastewater treatment, Biotechnology Advances, 22(7), 533-563.

Liu Z.P., Wang J.Y., Sun X.J., and Cai W.M. (2005) Investigation on the properties and kinetics of glucose-fed aerobic granular sludge, Enzyme Microb. Tech., 36, 307-313.

Moy B.Y.P., Tay J.H., Toh S.K., Liu Y., and Tay S.T.L. (2002) High organic loading influences the physical characteristics of aerobic sludge granules, L. Appl. Microbiol. 34, 407-412.

Chapter 7 Conclusions

Swim-bed technology using the novel BF material as an attachment matrix for biofilm growth was evaluated as a wastewater treatment method. From the results of this study, the following conclusions were obtained:

1. The BF material allowed for attachment of large amounts of biomass in a matrix that flexes with the wastewater flow, thus providing a high degree of contaminant biomass contact with a fully retainable biofilm while avoiding the hydrohynamic difficulties associated with floating-bed media. As much as 133 g of biomass per meter of BF support material was maintained during the test in two reactors with different volumes and heights. Further study on a single swim-bed reactor illustrated that the amount of attached biomass was related to the organic loading with more biomass attachment occurring at higher loadings.

2. The Swim-bed process demonstrated remarkable treatment performances throughout the whole testing period. Over 80% of the COD removals were obtained in SSB process even with volumetric loading rates up to 12 kg-COD/m^3/d at hydraulic retention times as low as 3 h. An average COD removal of 96.8% was achieved in a SSBR process at a loading of 6 kg-COD/m^3/d. The same process was applied for low-strength wastewater treatment with short HRTs. Average COD removals over 90% were achieved for treatment of the synthetic wastewater with a loading of 3 kg-COD/m^3/d at a HRT of 1.2h and an industrial wastewater containing polyvinyl chloride with a loading of 4.1 kg-COD/m^3/d at a HRT of 0.8h.

3. Only limited evidence for nitification was observed at low COD loading rates (1.6kg/m^3/d) in the SSB process. However, average nitrification efficiency of 79% was observed at a loading of 6 kg-COD/m^3/d when sludge recirculation was applied

4. Sludge production was reduced significantly in swim-bed processes whenever with or

without sludge circulation. In the SSB process, observed sludge yields and apparent effluent sludge yields ranged from 0.14 kg-MLSS/kg-$COD_{removed}$ to 0.19 kg-MLSS/kg-$COD_{removed}$ and from 0.060 kg-TSS/kg-$COD_{removed}$ to 0.12 kg-TSS/kg-$COD_{removed}$, respectively, corresponding to loadings varying from 1 kg-COD/m^3/d to 5 kg-COD/m^3/d. An observed sludge yield of 0.002 kg-MlSS/kg-$COD_{removal}$ was obtained at a loading of 6 kg-COD/m^3/d for the SSBR process treating high-rate wastewaters. Applying the SSBR process for treatment of domestic wastewater, a 40% of sludge reduction was achieved as compared to a CAS process operated in parallel with the SSBR process.

5. An abundance of protozoa and metazoa in the swim-bed process was attributed to the large amount of attachment biomass with high activity. *Ciliates* and *Rotifers* were recognized as the dominant species contributing to sludge reductions in the SSB process. The numeration of microorganisms revealed much more protozoa and metazoa in the SSBR process than CAS process. These results suggest that the large numbers of protozoa and metazoa in swim-bed processes contribute to the sludge reduction.

6. Microbial granules were formed in the SSBR processes both treating high-rate and low-strength industrial wastewaters with short HRTs. Granular flocs contributed to the exceptionally good sludge settling characteristics, which resulted in high levels of biomass maintenance in the reactor and clear effluent. MLSS concentrations of 20 g/L and 12 g/L were observed in swim-bed reactors during the treatment on high-rate wastewater and low-strength industrial wastewater, respectively.

7. Massive filamentous growth was observed in a BF process under relatively lower loading rates, but was avoidable at COD loading rates of 8 kg/m^3/d or greater during the SSB process on high-rate wastewater treatment study. However, it could not be avoided while treating an easily degraded low-strength wastewater with short HRTs in the SSBR process. Moreover, the sludge bulking was aggravated by the excess growth of *Zoogloea,* spp in this process.

8. High levels of EPS, which could benefit microbial granule formation, were obtained in the swim-bed process. However, for the treatment of low-strength wastewater with easily degraded characteristics, the high levels of EPS might relate to sludge viscous bulking.

Appendix: Publication related to this dissertation

1. Joseph D. Rouse, Daisuke Yazaki, **Yingjun Cheng**, Toichiro Koyama and Kenji Furukawa.: Swim-bed technology as an innovative attached-growth process for high rate wastewater treatment, Japanese Journal of Water Treatment Biology, 2004, 40(3), 115-124.

2. **Yingjun Cheng**, Daisuke Yazaki, Toichiro Koyama and Kenji Furukawa: Swim-bed technology as an innovative attached-growth process in the wastewater treatment, Future of Urban Wastewater System-Decentralization and Reuse (Proceeding of IWA 2005,Xi'an ,China), 2005,295-306.

3. **Yingjun Cheng**, Yusuke Watanabe, Sen Qiao, Toichiro Koyama and Kenji Furukawa: Comparison of treatment capacity of swim-bed technology and conventional activated sludge process for domestic wastewater treatment, Journal of Water Treatment Biology, Vol 42,NO.3

4. **Yingjun Cheng**, Daisuke Yazaki, Sen Qiao, Toichiro Koyama and Kenji Furukawa: Excess sludge reduction and biomass characteristics in swim-bed process, Journal of Water Treatment Biology, Japanese Journal of Water Treatment Biology (summated) (2006).

5. **Yingjun Cheng**, Daisuke Yazaki, Toichiro Koyama and Kenji Furukawa: Treatment characteristics of swim-bed wastewater treatment process. Proceeding of Annual Meeting 2005 of the Society for Biotechnology, Tsukuba (Japan),November,2005 NO.3D 10-1 pp137

6. **Yingjun Cheng**, Daisuke Yazaki, Toichiro Koyama and Kenji Furukawa: Reducing sludge production by using swim-bed technology as an innovative attached-growth process, Proceeding of annual symposium of Japan Society of Civil Engineering 2005,3 967-968