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Swine wastewater treatment by the static granular bed reactor

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ABSTRACT

Swine wastewater was successfully treated by the SGBR. Two types of swine wastewater samples were used, and the performance of the SGBR was excellent. COD removal efficiency of sample 1 was 84.7~94.5%, while that of sample 2 was 58.4~76.6%. The SGBR acted not only as a bioreactor but also as a filter system. The performance of suspended solids removal was excellent irrespective of the OLR. Additionally, the COD removal efficiency in the effluent after backwashing was not a function of the recovery time but that of the OLR.

The SGBR model was developed with concepts of advection, diffusion/dispersion, and decay of microorganisms. The simulated COD correlated well with the experimental COD except at low concentrations.

The SGBR behaved as a pseudo-plug flow reactor within the top 20% of reactor height so that most of the COD and VFA were completely removed. In addition, a large amount of granules in the lower part of the SGBR was used to polish organic matter and solids.

CHAPTER 1. INTRODUCTION

Anaerobic treatment has a very long history in wastewater treatment. The septic tank is the simplest, oldest, and most widely used process (Jewell, 1987). Anaerobic treatment has been used to treat concentrated industrial wastewater as well as domestic wastewater (McCarty and Smith, 1986). Anaerobic treatment has a lot of advantages such as low energy consumption, low production of waste biological solids, dormancy for many months with rapid recovery, low nutrient and chemical requirements, high removal even at high loading rates, pathogen removal, improving dewaterbility, and production of energy in the form of biogas. Traditionally, however, it has been regarded by some to have disadvantages such as being a sensitive and vulnerable process, odors, long period needed for start-up, and the necessity of a post treatment process to meet discharge standards. However, knowledge about xenobiotic and toxic compounds has been increasing due to anaerobic treatment research. As a matter of fact, anaerobic digestion is a very stable process if the system is understood well. When starting up a full-scale anaerobic treatment process, sufficient inoculation is often provided in order to overcome its drawbacks. Odors can be prevented by physicochemical or biological process improvements (Lettinga, 1996). Lettinga (1996) reported that an anaerobic treatment process produces mineral compounds such as ammonium, phosphate, or sulfide and needs an additional post treatment for a sustainable environmental protection.

Anaerobic treatment has been rapidly developing since the late 1960s. Since Young and McCarty (1969) developed the anaerobic filter (AF), many high-rate anaerobic reactors have been researched. In Europe, however, a reactor which could obtain high performance was developed. The upflow anaerobic sludge blanket (UASB) reactor was developed in the

Netherlands in the late 1970s. The UASB reactor has become one of the most widely applied technologies for high-rate anaerobic treatment in industry. According to Lettinga (1996), anaerobic sludge bed reactors have three concepts as: (a) the immobilized balanced micro-ecosystem is formed; (b) the immobilized anaerobic aggregates have high settleability; and (c) mass transport is prevalent between granule and bulk solution.

In 2000, Mach and Ellis developed a new granule reactor, the static granular bed reactor (SGBR) (Mach, 2000). The SGBR is a simple and optimum process for medium to low strength wastewater. Unlike other granular processes, the SGRB does not need mixers, Gas/Solid/Liquid separators, or other mechanical materials. In other words, the SGBR is operable with simplicity because the reactor is filled with granules and also because influent is distributed by gravity. The performance of the SGBR was comparable with that of other processes. It is often superior to that of the UASB (Evans, 2005a).

As the livestock industry has intensively developed, an increasing amount of high-strength swine wastewater has been produced. In the US alone, it is estimated that 5.8×10^7 tons of manure are produced each year (Dentel et al., 2004). Piggery waste has a high content of organic matter and pathogenic organisms. Without adequate treatment, it can cause a drastic effect on human health and the environment. Swine wastewater has to be treated in order to prevent the release of contaminants, odors, and pathogens to the environment (Schiffman et al., 2001; Sobsey et al., 2001; Luo et al., 2002). Anaerobic digestion processes have often been used to treat swine wastewater. Several processes are involved in treating swine manure. It is exemplified by the continuous stirred tank reactor (CSTR), two-stage digestion, the AF, the UASB, the anaerobic sequencing batch reactor (ASBR), and the anaerobic baffled reactor (ABR) (Yang and Chou, 1985; Ng and Chin, 1986; Wilkie and Colleran, 1986; Lo et al., 1994).

The aim of this research is to estimate the performance characteristics of treating swine wastewater using the SGBR. In addition, it is to evaluate the SGBR through kinetic and modeling studies.

CHAPTER 2. LITERATURE REVIEW

For several decades, the land treatment of swine waste has been used for fertilizer and soil conditioning. However, it is necessary that the high-strength organic waste in swine wastewater should be properly treated before being discharged to a river or on land. Untreated organic matter is detrimental to the environment and lowers the quality of fertilizer or soil conditioner.

Anaerobic digestion is one of the most reliable methods for swine wastewater treatment. Anaerobic treatment has a lot of advantages such as low energy requirement, low biomass production, storage ability unfed for many months, low nutrient and chemical requirements, high loading rate capacity, pathogen removal, improvement of dewaterbility, odor removal, and the production of biogas (Lettinga, 1996). Among anaerobic systems, the granular process is usually used for high-rate anaerobic digestion. Since the UASB was developed in the Netherlands in the late 1970s, 1215 full-scale high rate anaerobic reactors have been operated throughout the world (Franklin, 2001; Lettinga et al., 1980).

Bacteria tend to make granules for themselves (Jian and Lun, 1993). Schmidt and Ahring (1996) reported that a granule consisted of syntrophic bacteria. The granule size can be enhanced by multivalent cations such as iron and aluminum in high-strength wastewater (Yu et al., 2000, 2001). Many natural and synthetic polymers can also be used to enhance glanulation in high-strength wastewater (Hughes et al., 1990; Guiot et al., 1991; El-Mamouni et al., 1998). Tiwari et al. (2005) reported that natural ionic polymer additives are able to enhance granules in low-strength wastewater. In granule studies, cavities and holes have been usually seen on the granule surfaces (Macleod et al., 1990; Morgan et al., 1991). The cavities may be channels for transport of gases, substrate, or metabolites. A distinct localization of acidogenic bacteria and hydrolytic bacteria in the outer layer of granules grown on lactate or propionate was observed; meanwhile methanogenic bacteria dominated the inner part of the granule (Macleod et al., 1990; Fukuzaki, 1991a, 1991b). However, Grotenhuis et al. (1991) showed that there was no spatial orientation of microorganisms. Banik et al. (1997) reported that there was no apparent layered structure in granules at 25^oC in the ASBR. Mach (2000) also observed no distinct layers in granules at 22^oC in the SGBR.

Swine waste trend

The amount of swine in the world has been increasing. As of 1996, the number of pigs in the world was approximately one billion. The amount of swine in developed countries slightly decreased in 1996 compared to 1989~1991. However, the increasing trend of pigs in developing countries is noticeable as shown in Figure 1.

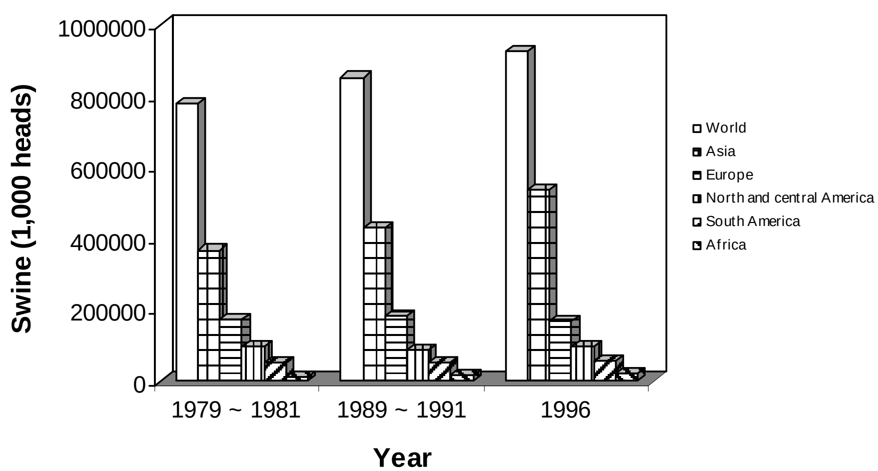


Figure 1. Swine population (FAO, 1996).

US EPA (1992) reported global swine wastes produced in each region (Figure 2), according to which most swine wastes are produced by Asian countries. A primary reason is that China raises about the half number of pigs in the world. In addition, other Asian countries have also increased the number of swine (FAO, 1996).

The number of swine and the amount of swine wastewater will gradually increase in the future as the demand of red meat in developing countries is consistently increasing. Especially, Asian countries such as China, Korea, and India have shown the most significant increase in their swine population (Chynoweth et al., 1999).

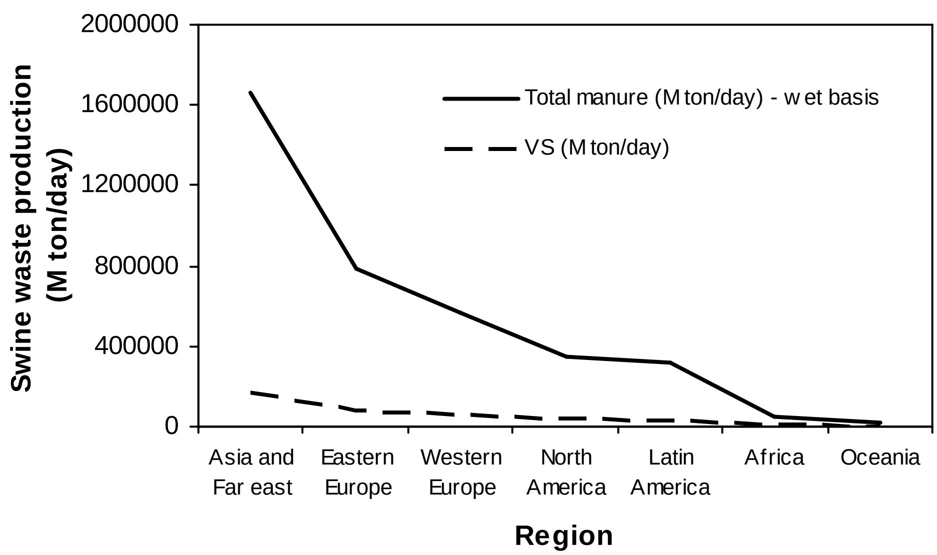


Figure 2. Swine waste production by each region.

<u>Swine wastewater characteristics</u>

In smaller hog operations (several hundred or less) were reared, the wastes produced can often be disposed of on the same land used for rearing the pigs for fertilizer or soil conditioner. However, swine wastes from centralized facilities in which the number of animals often exceeds 1,000 and sometimes is more than 10,000, land application can have a negative effect on the environment. Larger confinement operations have become more commonplace as the demand for red meat has been increasing (Chynoweth et al., 1999).

The characteristics of swine wastewater depends on various factors such as the age and diet of hogs, temperature, humidity of a building, housing or confinement methods, waste removal procedures, and pre-processing (Andreadakis, 1992; USDA, 1992; Zhang and Felmann, 1997; Day and Funk, 1998). However, the characteristics of swine wastewater are affected more by dilution, storage, and separation rather than by the diet of pigs and other factors (Chynoweth et al., 1999).

Swine wastewater is considered by some to be a solid waste which contains some liquids, while municipal or industrial wastewater is usually liquid waste which contains some solids (Andreadakis, 1992). Total solids (TS) of swine fecal matter is about 10%, and it is diluted with urine and other flush water, or concentrated when bedding is used in dry storage systems (Zhang and Felmann, 1997; Day and Funk, 1998). Chynoweth et al. (1999) reported that a typical swine wastewater TS concentration for a confinement using a tank under slats is 3~4%.

Even though significant experience has been gained with respect to municipal sludge digestion, this knowledge and its accompanying empirical models should not be directly applied to swine wastewater treatment. This is because the characteristics of swine wastewater are significantly different from municipal sludge (Andreadakis, 1992).

The characteristics of swine wastewater from several studies are shown in Table 1 through 4.

Table 1. Swine wastewater characteristics per day per 1,000 kg of swine (Day and Funk, 1998)

Component	Average
Manure (kg)	84
Urine (kg)	39
Density (kg/m^3)	990
Total Solids (kg)	11
Volatile Solids (kg)	8.5
BOD (kg)	3.1
COD (kg)	8.4
pH	7.5
TKN (kg)	0.52
NH_3-N (kg)	0.29
T-P (kg)	0.18
Ortho-P (kg)	0.12

Table 2. Swine wastewater characteristics per day per 1,000 kg of swine (Andreadakis,1992)

Component	Average
Total Solids (kg)	6.00
Volatile Solids (kg)	4.80

BOD (kg)	6.20
COD (kg)	2.10
T-N (kg)	0.48
T-P (kg)	0.14
K (kg)	0.21
Ca (kg)	0.0185
Mg (kg)	0.0045
Fe (kg)	0.0008
Zn (kg)	0.0003
Na (kg)	0.0040
Cu (kg)	0.0000

Table 3. Swine waste characteristics from storage tank under slats (USDA, 1992)

Component	Farrow	Nursery	Finish	Breeding
Moisture (%)	96.5	96.0	91.0	97.0
Total Solids (%)	3.50	4.00	9.00	3.00
Volatile Solids (%)	2.28	2.79	6.74	1.80
T-N (g/L)	3.6	4.8	6.3	3.0
NH_3-N (g/L)	2.8	4.0		
P (g/L)	1.8	1.6	2.7	1.2
K (g/L)	2.8	1.6	2.2	2.1

Table 4. Swine waste characteristics from storage and treatment facility (USDA, 1992)

	Anaerobic lagoon		Feed lot*	
Component	Sludge	Supernatant	Settled sludge	Runoff water
Moisture (%)	92.4	99.8	88.8	98.5
Total Solids (%)	7.60	0.25	11.2†	1.5
Volatile Solids (%)	4.68	0.12	90.7†	
BOD (g/L)		0.40		
COD (g/L)	64.6	1.2		
T-N (g/L)	3.0	0.35	5.6†	2.0†
NH_3-N (g/L)	0.76	0.22	4.5†	1.2†
P (g/L)	2.7	0.13	2.2†	0.38†
K (g/L)	7.6	0.38	10.0†	1.10†

*Semi humid climate

† kg/day·1000kg swine

As shown in Table 5, proteins and lipids have been observed to be a significant portion of the organic matter in swine wastewater. Swine wastewater, however, also contains a significant concentration of lignin which is typically non-biodegradable. Shin et al. (2005) observed that swine waste has a large fraction of non-biodegradable organic matter. Andreadakis (1992) showed that approximately 40% of the total organic matter is non-biodegradable. Due to the great amount of cellulose and lignin in swine wastewater, swine waste is considered to be refractory or recalcitrant to biodegradation. Chynoweth et al. (1999) stated that lignin and cellulose are the typical refractory compounds in anaerobic digestion.

Table 5. Swine waste characteristics and biodegradability by anaerobic digestion (Chynoweth et al., 1999)

Component	Influent	Removal (%)*
Total Solids (%)	6.9	52
Volatile Solids (% TS)	82.6	60
COD (g/L)	73.8	58
T-N (g/L)	3.9	
Proteins (% TS)	19.3	47
Hemicellulose (% TS)	20.1	65
Cellulose (% TS)	12.4	64
Lipids (% TS)	14.8	69
Starch (% TS)	1.6	94
Lignin (% TS)	4.4	3

* mesophilic digester with HRT 15 days

Table 6 shows theoretical methane production from various substrates. It can be seen that as the value of CH_4/VS decreases, the chemical oxygen demand (COD) removal increases, suggesting that proteins and lipids are more easily degraded than carbohydrates (Andreadakis, 1992). The reported biodegradability of swine wastewater ranges from 0.32 to 0.48 m^3 CH_4/kg $VS_{destroyed}$ (Hashimoto, 1984; Andreadakis, 1992; Safley and Westerman, 1990). This range is comparable to a 40~60% reduction of volatile solids (VS).

Table 6. Theoretical methane production (Andreadakis, 1992)

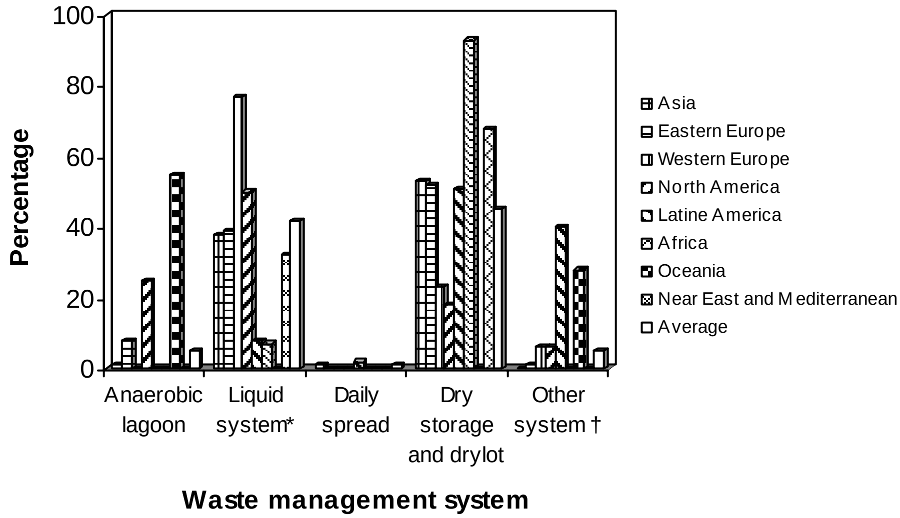
	COD/VS	CH_4/VS (m^3/kg)
Carbohydrates	1.067	0.374
Proteins	1.500	0.525
Lipids	2.870	1.006

Liao et al. (1995) reported that approximately 90% of the total soluble nitrogen is in untreated swine wastewater is in the form of ammonia. Odors in swine wastewater are caused by ammonia, amines, volatile fatty acids (VFA), mercaptins, carbonyls, phenols, and indoles. Sulfides in swine wastewater are produced by the decomposition of proteins and other sulfur containing compounds (Chynoweth et al., 1999).

Swine wastewater has a high concentration of pathogens, coliform bacteria, and indicators of fecal pollution (Zhang and Felmann, 1997). Mateu et al. (1992) reported that inactivation of fecal coliform in swine wastewater was associated with a high concentration of VFA. Inhabiting methanogenic bacteria in swine wastewater were identified as *Methanosarcina sp.* never using formate. These bacteria are gram positive and form methane and carbon dioxide by decomposing propionate (Boopathy, 1996).

Swine wastewater treatment

Figure 3 shows the distribution of waste management systems worldwide. As can be seen in this Figure, the liquid flush system outcompetes the others in developed countries, while dry storage and drylot systems are usual treatment methods in developing countries.



* liquid/solid and pit storage

† deep pit stacks, litter, and other

Figure 3. Swine wastewater management (US EPA, 1992).

Swine wastewater has been treated primarily by land application, aerobic processes, or lagoon systems due to the popularity of aerobic processes, instability of anaerobic processes, or lack of knowledge of anaerobic digestion. However, there are some environmental and health concerns with the lagoon treatment of swine waste including emissions of ammonia, odors, pathogens, and water quality deterioration (Aneja et al., 2000; Mallin, 2000; Schiffman et al., 2001; Sobsey et al., 2001; Loughrin et al., 2006).

The use of anaerobic digestion of swine wastewater for renewable energy production has gained renewed focus as the cost of fossil fuels and the demand for green energy increase. Organic and inorganic matter in swine wastewater is usually treated by a lagoon system, oxidation ditch, anaerobic digestion, and so on. Afterwards, effluent often is treated for nitrogen and phosphorus before being discharged to a river or a lake (Chynoweth et al, 1999).

Among these treatment processes, anaerobic digestion of swine wastewater has advantages such as organic matter stabilization, odor removal, pathogen destruction, and biogas recovery. Anaerobic treatment of swine wastewater also enables effluent to conserve nutrients such as nitrogen and phosphorus (Ahn et al., 2006).

Most full scale swine wastewater treatment processes are traditional anaerobic systems such as the anaerobic lagoon or the CSTR (Angenent et al., 2002). However, researchers have developed new swine wastewater treatment processes, and thus various treatment methods have been employed. Especially, biotechnology applications using granules such as the UASB or the expanded granular sludge bed (EGSB) have been attractive since the 1980s. As anaerobic granules have a high concentration and density of microorganisms, it is highly applicable in treating high-strength wastewater such as swine wastewater. In addition, the ASBR, the AF, and the sequencing batch reactor (SBR) have also been used to treat swine wastewater.

Lo et al. (1994) studied swine wastewater treatment by the hybrid UASB at ambient temperature. According to their research, greater than 95% of COD was removed at an organic loading rate (OLR) of 1.65 kg/m^3•day. In the mean time, COD removal efficiency sharply dropped to 57% at an OLR of 3.5 kg/m^3•day. Foresti and de Oliveira (1995) also reported the results of swine wastewater treatment using a UASB at 25 ^{o}C. They reported that 87% COD removal could be obtained at an OLR of 4.50 kg/m^3•day. Wilkie and Colleran (1986) treated swine wastewater by the upflow anaerobic filter at 25 ^{o}C. The wastewater was settled supernatant, and 52% COD removal was obtained at an OLR of 8.4 kg/m^3•day.

Shin et al. (2005) treated swine wastewater by combining a submerged membrane bioreactor and anaerobic filter reactor. The average COD removal efficiency was 91% at an OLR ranging from 0.5 to 3.0 kg/m^3•day. Intermittent aeration was applied to treat swine wastewater (COD: 3,473~4,233 mg/L, TSS: 2,065~3,354 mg/L). The operating time was set with four ratios of aeration time to non-aeration time (hrs): 60:36, 5:1, 4:2, and 3:3. The overall treatment efficiencies of COD, biochemical oxygen demand (BOD), T-N (Total Nitrogen), and total suspended solids (TSS) were 87.4%, 98.7%, 92.7%, and 97.5%, respectively (Yang and Wang, 1999).

Yang et al. (2003) reported that removal efficiencies of COD and TSS were 83.5%, and 81.2%, respectively, using an intermittent method at the OLR 0.67~1.07 kg/m^3•day. Meanwhile, Min et al. (2005) showed that swine wastewater could be successfully treated by a microbial fuel cell process. Wellinger and Kaufmann (1982) argued that it is possible to achieve a lower energy requirement system by means of ambient temperature anaerobic digestion. They showed that psychrophilic anaerobic digestion could be an alternative method to treat swine wastewater.

Aside from reactor scale, the TS content is a very important factor in digester design and performance (Chynoweth et al., 1999). The limiting step in swine wastewater treatment is hydrolysis (Andara et al., 1999). According to Angenent et al. (2002), during operation with high ammonia concentration, the major route of methane production was through a syntrophic relationship between acetate utilizing bacteria and hydrogen utilizing bacteria.

SGBR characteristics

The SGBR was developed at Iowa State University in 2000 (Mach, 2000; U.S. Patent No. 6,709,591). This reactor was designed to treat wastewater with low to medium strength (Mach, 2000; Roth, 2003; Evans, 2004a; Evans, 2004b; Park, 2004; Roth et al., 2004; Debik et al., 2005; Evans and Ellis, 2005a; Evans and Ellis, 2005b).

It is possible that effluent can be directly discharged without additional treatment at ambient temperature. This is because the SGBR is packed with active granules and because the solids retention time (SRT) of the SGBR is commonly greater than 300 days (Evans, 2004b). In addition, the SRT in the SGBR has a trend which is proportional to the temperature. Evans (2004a) reported that the SRT in the SGBR increased with the hydraulic retention time (HRT) more at 15^{o}C than at 8^{o}C. It is essential that a system have a high SRT in order to maintain low effluent concentration. In biological treatment processes, the SRT plays a more significant role than the HRT because the SRT is a function of microorganism concentration and growth rate, whereas because the HRT is not. Therefore, an SGBR in which the SRT is greater than 300 days is able to maintain a high concentration of active granules in the reactor and treat wastewater more efficiently. Dague et al. (1998) reported that MLSS and MLVSS decreased when low-strength wastewater was treated by the ASBR at a low HRT. According to their research, the SRT ranged between 30 and 180 days at 20^{o}C.

Evans (2004a) reported the SGBR had a completely mixed flow pattern with some short circuiting. However, it is possible to eliminate this short circuiting by backwashing, to some extent. Evans (2004b) presumed that flow in the SGBR was affected by gas production and granule movement.

The greatest advantage of the SGBR is that its operation is simple unlike the UASB or the EGSB. The SGBR does not need gas solid separator (GSS) devices, mixers, complex underdrains, or mechanical systems as this reactor is not only a downflow system but also a biofilter system which is partially filled with granules. This system consists of active granules, gravel, and a stainless steel mesh in some cases. Therefore, the cost of construction is lower, and the operation of the SGBR is superior to other systems. In addition, the start-up of the SGBR is fast and efficient due to large concentration of active granules in the reactor (Evans, 2004b). The high performance is also a result of biomass retention as evidenced by the long SRTs.

The UASB is a proven technology to treat high-strength wastewater. However, the design of the UASB reactor is empirical (Tiwari et al., 2005). The limit of the UASB design at a high loading rate from full scale experience is the washout of granules (Driessen and Yspeert, 1999). In high rate UASB systems, it is a common practice to restock granules in order to replenish those lost (Ahn and Speece, 2003).

SGBR performance

As with other granular treatment systems, it is possible to obtain and maintain high performance with the SGBR. Especially, concentrations of COD, TSS, and VFA in the effluent are low, and the effluent sometimes does not require additional treatment to be discharged to a receiving stream. Mach (2000) preformed a comparative study in which she compared two SGBRs with a strength of 1,000 COD mg/L at 22 ± 2^{o}C. According to her research, although both SGBRs achieved greater than 95% COD removal efficiency, the SGBR with a larger height to width ratio demonstrated superior performance.

Jung et al. (2002) reported that the SGBR outcompeted the ASBR in treating pork slaughterhouse wastewater (Ave. COD: 1,912 mg/L). It was largely because the ASBR lost a lot of sludge while decanting at low HRT. COD removal efficiency was 82% to 96%, and TSS removal achieved in excess of 93.9%.

Roth (2003) studied pork slaughterhouse wastewater treatment by the pilot-scale SGBR on the basis of Jung et al.'s (2002) results. According to his results, COD removal efficiency was between 83.7% and 95.7%, and TSS concentration in effluent averaged 43 mg/L. Park (2004) demonstrated the effective leachate and waste management strategy by the SGBR. According to his research, the SGBR was able to treat leachate effectively.

In order to compare the performance between the SGBR and the UASB, Evans (2004a) measured COD and TSS removal performances, effluent VFA, and methane production with non-fat dry milk as well as sucrose/non-fat dry milk mixture. The performances of COD removal were not too different between two reactors. VFA concentrations in the effluent of both reactors were also low. However, TSS concentration in the SGBR was superior to that in the UASB at low HRTs. At an HRT of 16 hrs and 24 hrs, the performance of COD and TSS removal in the SGBR were much higher than those in the UASB.

In many studies, high-rate anaerobic reactors were used to treat municipal wastewater at ambient temperature (Lettinga et al., 1983; Kato et al., 1997; Collins et al., 1998; Elmitwalli et al., 2001; Bodik et al., 2002). Evans (2004a) treated municipal wastewater by the SGBR. At 25^{o}C, COD removal efficiency was 74~84%. At lower HRT, TSS concentration in the effluent was low. Evans (2004a) stated that solids decreased the bed porosity so that this phenomenon enabled particles to be effectively filtered.

High sulfate loading wastewater was treated by the SGBR (Evans, 2004b). The synthetic wastewater (3 COD g/L : 1.33 g S/L) was used, and an average 20,000 H_2S ppm was produced at 18 hrs HRT. The COD removal efficiency was 97.3%.

CHAPTER 3. MATERIALS AND METHODS

The SGBR was used for treating swine wastewater. The schematic diagram of the SGBR is illustrated in Figure 4.

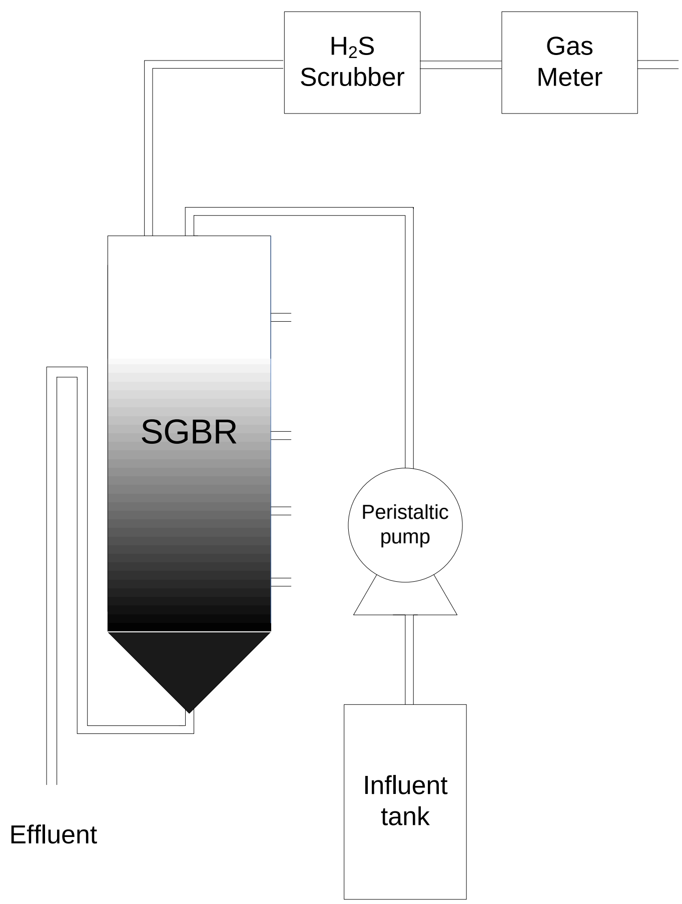


Figure 4. Schematic diagram of the SGBR.

The active volume of the SGBR was 10 L and the produced gas was designed to be exhausted upward. The exhausted gas was connected with a hydrogen sulfide scrubber and a gas meter. The SGBR was filled with granules, and a stainless steel mesh (2 mm) was installed in order to uphold the gravel and granules. Gravel was placed between granules and the stainless steel mesh in order not to lose granules from the reactor. The size of this gravel was between 6.7 and 9.5 mm. The amount of gravel used for supporting the granules was 0.8 L. Granules were filled up to the active volume after installing the stainless steel mesh and gravel.

Seed granules were from a UASB at the Water Pollution Control Facilities in Cedar Rapids, IA. TS and VS were 62,335 mg/L and 51,900 mg/L, respectively. The seeded mass was 623.35 g. The VS to TS ratio was 83.3 %.

The reactor seeded with granules achieved high performance within a few days, while that seeded with non granule sludge needed start-up periods in excess of 60 days (Zeeman et al., 1988; Goodwin et al., 1992). Velsen (1979) stated that manure such as swine or poultry should be seeded in order to be treated effectively because there were not sufficient specific methanogenic microorganisms in such wastes. Most seed granules are obtained from swine wastewater treatment processes. However, sewage sludge from municipal wastewater is attractive because this sludge has a lot of methanogenic microorganisms and because it is suitable for the digestion of complex materials like raw sewage sludge (Velsen, 1979).

Two types of swine wastewater samples were used for this study. The characteristics of each swine wastewater sample are presented in Table 7.

Table 7. The characteristics of swine wastewaters used in this study

Component	Sample 1	Sample 2
pH	5.25	7.90
ORP (mV)		-333
AlK (mg/L)	5,000	6,015
COD (mg/L)	99,500	32,000
SCOD (mg/L)	49,200	16,000
TS (mg/L)	60,618	30,300
VS (mg/L)	40,707	20,050
FS (mg/L)	19,911	10,250
TSS (mg/L)	54,110	18,900
VSS (mg/L)	39,640	14,500
FSS (mg/L)	14,470	4,400
VFA (mg/L)	39,640	74,667

COD, TSS, volatile suspended solids (VSS), and fixed suspended solids (FSS) of sample 2 were about one third of those of sample 1. TS, VS, and fixed solids (FS) of sample 2 were about half of those of sample 1. In the mean time, VFA of sample 2 was approximately twice as much as that of sample 1. The samples were diluted to be properly treated in the SGBR. The dilution ratio of sample 1 was 40 and that of sample 2 was 10~20. This study was performed at ambient temperature (around 24^{o}C) for about ten months. The start-up condition in this study is shown in Table 8.

Table 8. Start-up condition for treating swine wastewater by the SGBR (Sample 1)

Component	Value
OLR (kg/m^3·day)	1.0
HRT (days)	1.25
Flow rate (L/day)	8.01
Reactor Volume (L)	10
pH	6.15
Alkalinity (mg/L as $CaCO_3$)	240
VFA (mg/L)	231.4
COD (mg/L)	1243.8
SCOD (mg/L)	615.0
TS (mg/L)	757.7
VS (mg/L)	508.8
TSS (mg/L)	676.4
VSS (mg/L)	495.5

Performance of the SGBR was periodically monitored by analyzing test parameters. The test methods used in this study are shown in Table 9. Most tests were performed according to the Standard Methods for the Examination Water and Wastewater (APHA, 1998). Proteins were measured by the method of Lowry et al. (1951), carbohydrates by the method of Bubois et al. (1956), and lipids by the method of Bligh and Dyer (1959). The produced gas was measured daily and recorded using a tipping meter supplied by Dr. Richard

Speece (Wet Tip Gas Meter Company, Nashville, Tennessee).

Table 9. Section of test parameter used in this study

Parameter	Section*
pH	4500 B.
ORP	2580 B.
Alkalinity	2320 B.
Chemical Oxygen Demand	5220 C.
Soluble Chemical Oxygen Demand	5220 C.
Total Solids	2540 B.
Volatile Solids	2540 E.
Fixed Solids	2540 E.
Total Suspended Solids	2540 D.
Volatile Suspended Solids	2540 E.
Fixed Suspended Solids	2540 E.
Volatile Fatty Acids	5560 C.

* APHA (1998)

Methane and carbon dioxide in the digested gas was analyzed by Gas Chromatography (Gow Mac, model 350 series, Bethlehem, PA; thermal conductivity detector) with a Hayesep column C3111220002 (Gig Harbor, WA). Injector temperature and detector temperature were 45^{o}C and 70^{o}C, respectively.

VFA was also analyzed by High Performance Liquid Chromatography (Dionex, GP 40, CA) with an absorbance detector (AD20, Dionex) and a 300 mm X 7.8 mm Metacarb 67H column (Varian, CA) using 0.05 M H_2SO_4 as mobile phase (flow rate 0.7 mL/min).

CHAPTER 4. RESULTS AND DISCUSSION

Performance of organic matter removal

COD removal efficiency of sample 1 is depicted in Figure 5. The SGBR maintained the high performance of COD removal (84.7~94.5%). COD removal efficiency increased as the OLR increased. COD concentration in effluent was low, irrespective of the OLR in influent (137.6~288.0 mg/L). In other systems as the OLR increases, the removal efficiency commonly decreases. The COD removal efficiency, however, increased in the SGBR even though the OLR increased as shown in Figure 5. This is because granules have gradually been acclimated to the wastewater and because dS/dt is proportional to concentration of biomass in the Monod equation (Monod, 1949). Considering that the SRT of the SGBR is greater than 300 days, it is possible to achieve a high performance of COD removal in the SGBR. COD removal efficiency of sample 2 is shown in Figures 6 and 7.

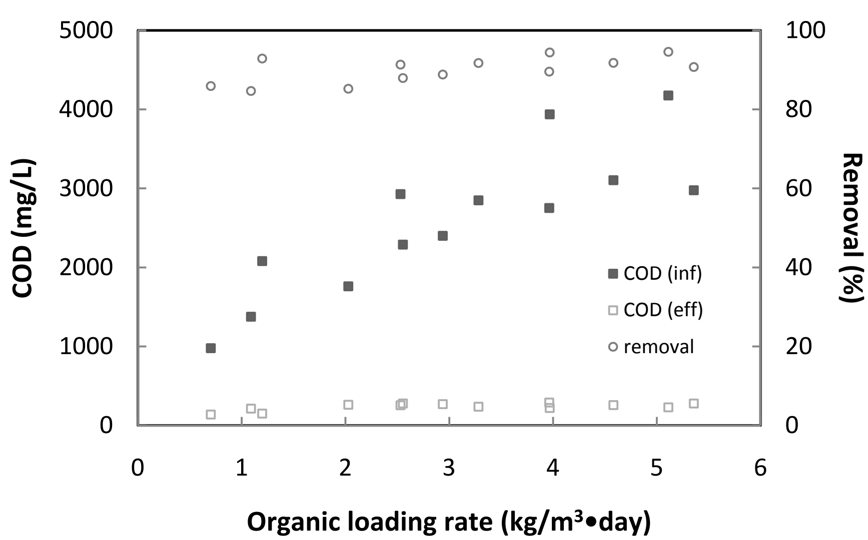


Figure 5. COD removal performance in the SGBR (sample 1).

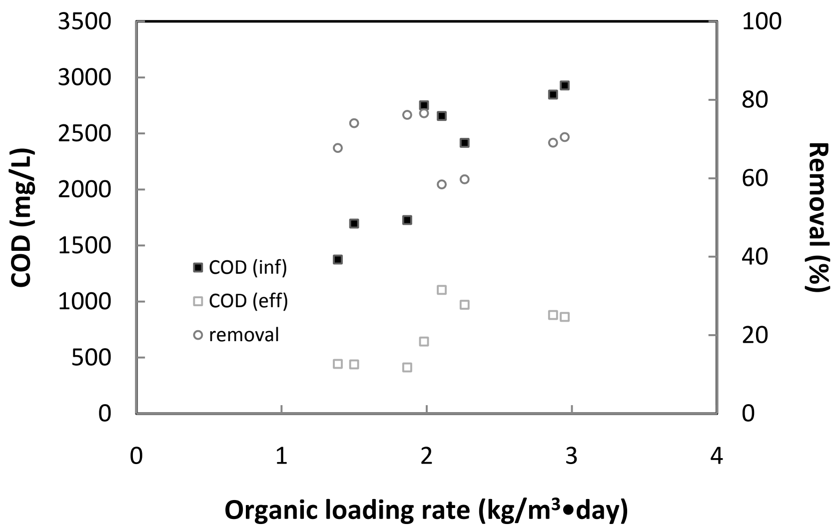


Figure 6. COD removal performance in the SGBR (sample 2).

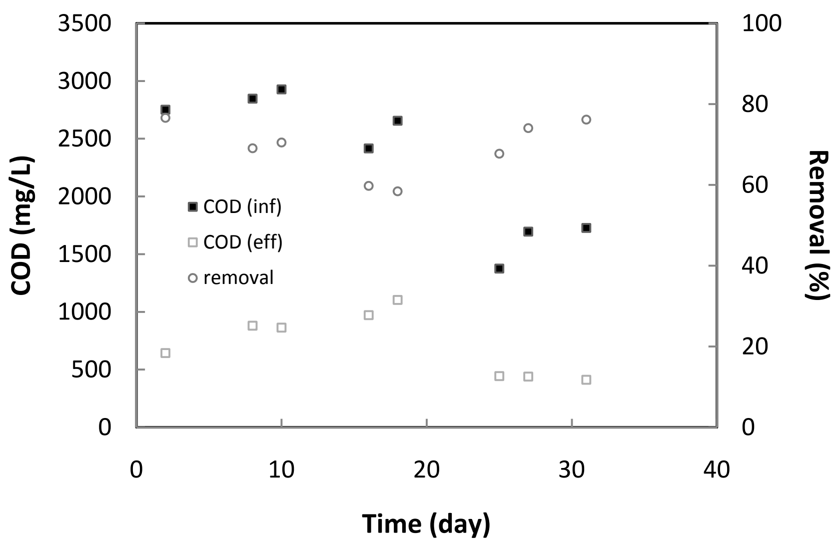


Figure 7. Variation of COD removal performance in the SGBR during the operation period (sample 2).

COD removal efficiency of sample 2 was 58.4~76.6 (Ave. 69.0±6.2%). The OLR of influent COD was 1.29~2.95 (Ave. 2.12±0.57 kg/m^3•day). Even though the OLR increased, COD removal efficiency was not significantly affected. This was indicative of the high concentration of active granules in the SGBR. In addition, the COD removal efficiency gradually increased as shown in Figure 7 starting on Day 25. This suggests that granules were quickly acclimated to the new wastewater. The SRT of the SGBR was greater than 300 days so that it was possible for microorganisms to be quickly acclimated to new wastewater. However, both biological and physical acclimations are needed as the granules are packed in the reactor.

Lo et al. (1994) treated swine wastewater by a hybrid UASB at ambient temperature. According to their research, greater than 95% of COD was removed at an OLR of 1.65 kg/m^3•day. However, COD removal efficiency sharply decreased to 57% at an OLR of 3.5 kg/m^3•day. Foresti and Oliveira (1995) also reported swine wastewater treatment by the UASB at 25^{o}C. According to their research, 87% COD removal efficiency could be obtained at an OLR of 4.50 kg/m^3•day. In addition, the COD removal efficiency at ambient temperature (25^{o}C) is similar to that in a mesophilic condition (30^{o}C). Sáchez et al. (2005) pointed out that the UASB is not suitable for treating swine manure. Although influent was screened and diluted swine wastewater, COD removal efficiency was between 70.6% and 85.4% at 30~35^{o}C. In addition, COD removal efficiency sharply decreased at an OLR of 2.70 kg/m^3•day. Comparing the performance between the AF and the UASB, Sáchez et al. (1995) concluded that the AF is superior to the UASB in treating swine wastewater. Cintoli et al. (1995) also reported that the UASB-AF reactor showed better behavior during changes in influent composition than the UASB.

Oleszkiewicz (1983) treated swine wastewater by the AF at 23°C. The COD removal was 73% at an OLR of 4.0 kg/m^3•day. Wilkie and Colleran (1986) treated swine wastewater by the upflow anaerobic filter at 25°C. The type of waste was settled supernatant and 52% COD removal efficiency was obtained at an OLR of 8.4 kg/m^3•day. At an OLR 5.0 of kg/m^3•day, Ng and Chin (1987) reported 84% COD removal efficiency by the AF. At the HRT 3~5 days, Ng and Chin (1988) also achieved 70~90% COD removal efficiency by the expanded-bed AF.

Deng et al. (2006) used the IC-SBR system to treat swine wastewater. The performance of the IC-SRB system was excellent. 95.5% COD removal efficiency was obtained by this system. Ng (1989) studied swine manure treatment by the ASBR. The performance of COD removal was 85% at the OLR 0.7 kg/m^3•day. However, COD removal efficiency decreased to 64% at an OLR of 1.8 kg/m^3•day. le Hy et al. (1989) reported 98% COD removal efficiency of piggery manure mixed with cheese-dairy wastewater using a mesophilic anaerobic digestion and oxidation ditch system. Shin et al. (2005) treated swine wastewater by combining a submerged membrane bioreactor and an anaerobic filter reactor. Average COD removal efficiency was 91% at an OLR ranging from 0.5 to 3.0 kg/m^3•day.

Soluble chemical oxygen demand (SCOD) removal efficiency is shown in Figure 8.

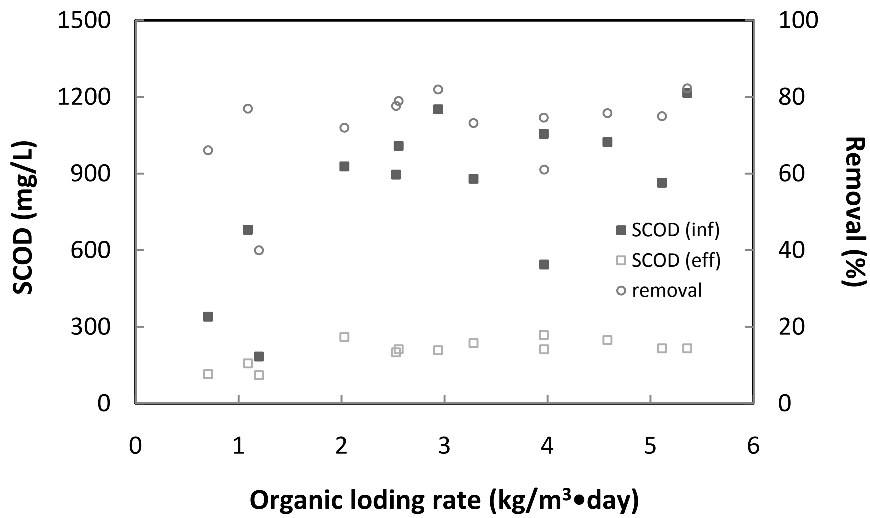


Figure 8. SCOD removal performance in the SGBR (sample 1).

SCOD was removed up to 82.2% (Ave. 72.0±11.3%). Sollfrank et al. (1992) showed that the temperature affected soluble COD concentration in the effluent. Soluble COD concentration in the effluent is proportional to the temperature.

Most organic matter in effluent was soluble. The difference between COD and SCOD was less than 100 mg/L (see Figure 9). Considering that both effluent COD and effluent SCOD concentration were low, it was evident that effluent was mostly comprised of soluble organic matter.

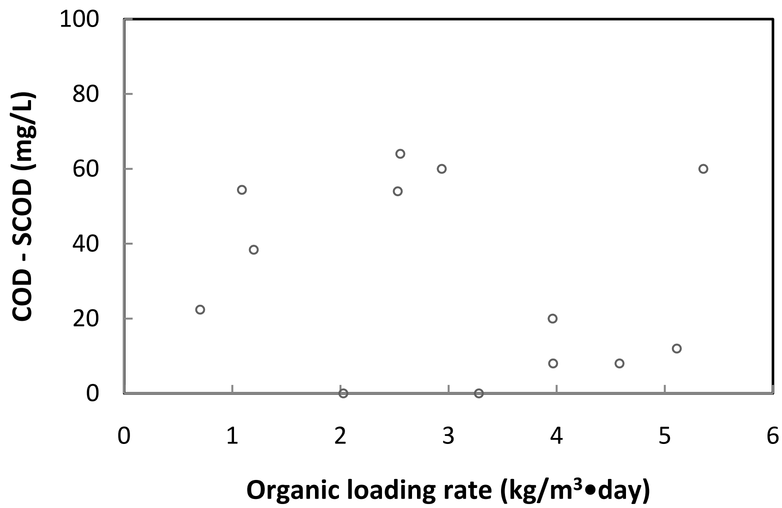


Figure 9. The relationship between COD and SCOD in effluent (sample 1).

Carbohydrate removal efficiency of sample 1, as shown in Figure 10, was 70.2~94.5% (Ave. 84.3±6.9%). Carbohydrate removal efficiency increased as the OLR increased. The trend of carbohydrate removal was similar to that of COD removal. It implies that carbohydrates of sample 1 were one of the important substances which comprised COD in the swine wastewater and that these were part of the biodegradable organic matter. However, the amount of carbohydrates was not as much as proteins or lipids in the swine wastewater.

The performance of carbohydrate removal of sample 2 was not as good as that of sample 1. The performance of carbohydrate removal of sample 2 by the SGBR is shown in Figure 11. Removal efficiency was 48.1~74.2% (Ave. 59.2±9.4%). The difference of removal efficiency between sample 1 and sample 2 was due to the characteristics of the wastewater. Especially, swine wastewater has a high concentration of cellulose and lignin, which are non-biodegradable carbohydrates.

Cellulose is one of the most difficult polysaccharides for microorganisms to metabolize because it is organized by crystalline structures (micelles) and because its solubility contributes to the difficulty of the attack (Gaudy and Gaudy, 1980). In addition, the possibility of cellulose hydrolysis is determined by the Degree of Polymerization (DP) of glucose. DPs 1 through 6 are water soluble. As the DP increases, hydrogen bonds and van der Waals forces become strong, and water molecules are excluded from hydrophobic molecules (Stronach et al, 1986).

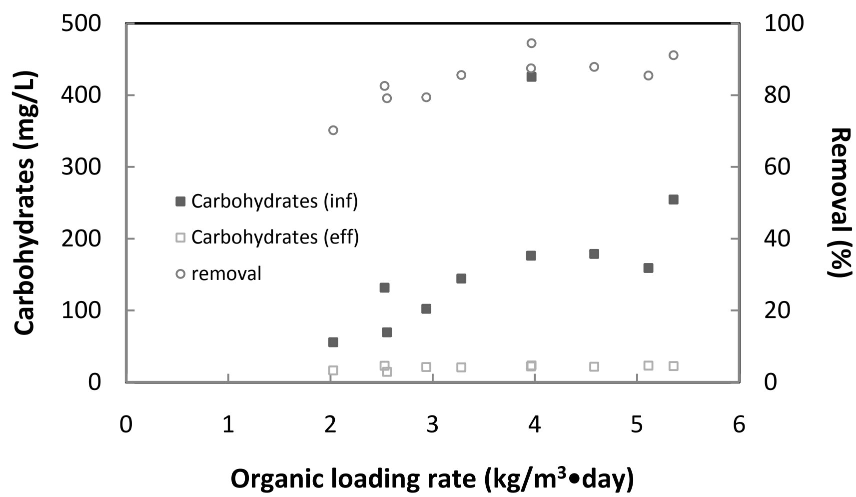


Figure 10. Carbohydrates removal performance in the SGBR (sample 1).

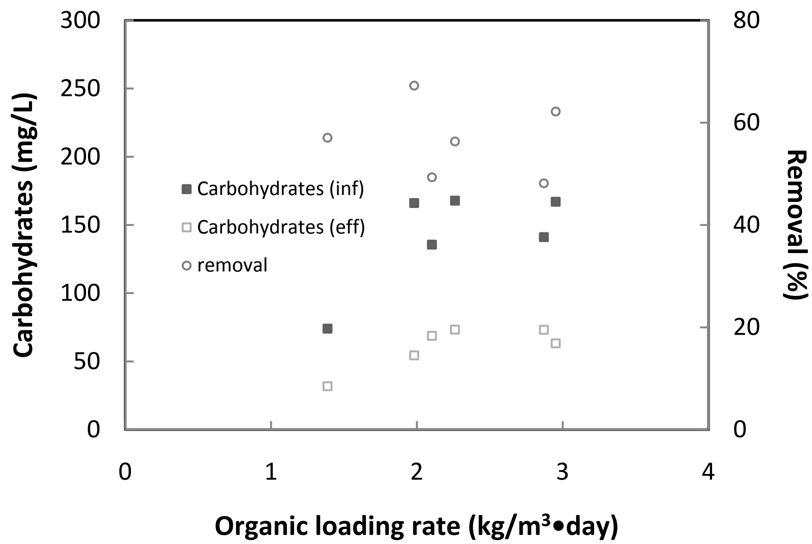


Figure 11. Carbohydrate removal performance in the SGBR (sample 2).

Protein removal efficiency of sample 1 also increased as the OLR increased in a similar fashion to the carbohydrates (Figure 12). The removal efficiency of sample 1 was 63.4~74.9% (Ave.70.2±4.5%), while that of sample 2 was 38.1~74.5% (Ave. 57.9±13.9%). It was reported that 19.3% of TS in swine wastewater were proteins, and 47% of the proteins were removed in a mesophilic digester (Chynoweth et al., 1999). In this research, higher performance of protein removal could be achieved even though the SGBR was operated at ambient temperature.

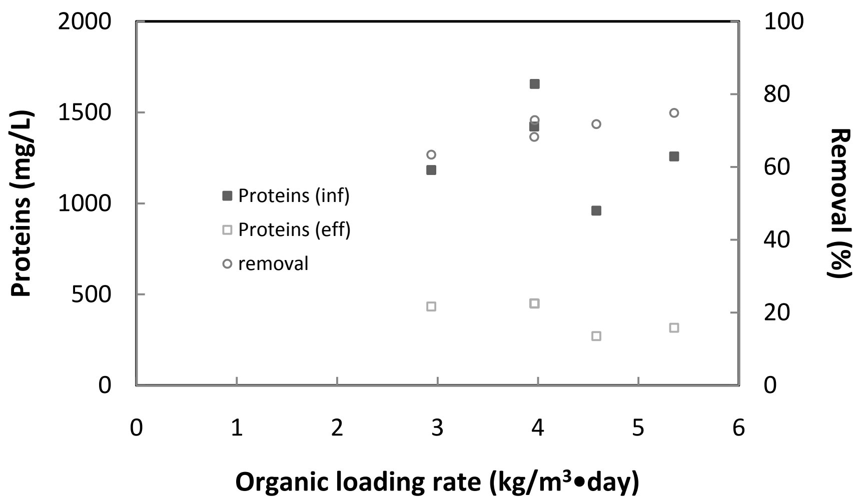


Figure 12. Protein removal performance in the SGBR (sample 1).

Lipid removal efficiency as a function of the OLR is depicted in Figures 13 and, 14. Unlike carbohydrates and proteins, the lipid removal efficiency of sample 1 was 38.1~71.4% (Ave. 58.7±10.2%). As shown in Figure 13, lipids removal of sample 1 was not greatly affected by the OLR. Chynoweth et al. (1999) stated that 14.8% of TS in swine wastewater were lipids, and 69% of the lipids were removed in a mesophilic digester. In the mean time, the performance of lipid removal of sample 2 was similar to that of sample 1. The lipid removal efficiency of sample 2 sharply decreased to 21.8% at an OLR of 3 kg/m^3•day.

Swine waste contains 30~40% lipids based on COD (Boopathy, 1998; Chynoweth et al., 1999; Ahn et al., 2006). Lipids in the anaerobic digestion system are readily hydrolyzed to glycerol and long-chain fatty acids. Long chain fatty acids can be acutely toxic by means of the adsorption of the surface-active acids onto the cell wall (Ahn et al., 2006). In high rate anaerobic digesters, shock loads of long chain fatty acids may cause severe problems (Koster and Cramer, 1987). Ahring et al. (1992) also showed that long-chain fatty acids are toxic compounds. Therefore, lipids can be a limiting factor to treat swine wastewater when the OLR increases.

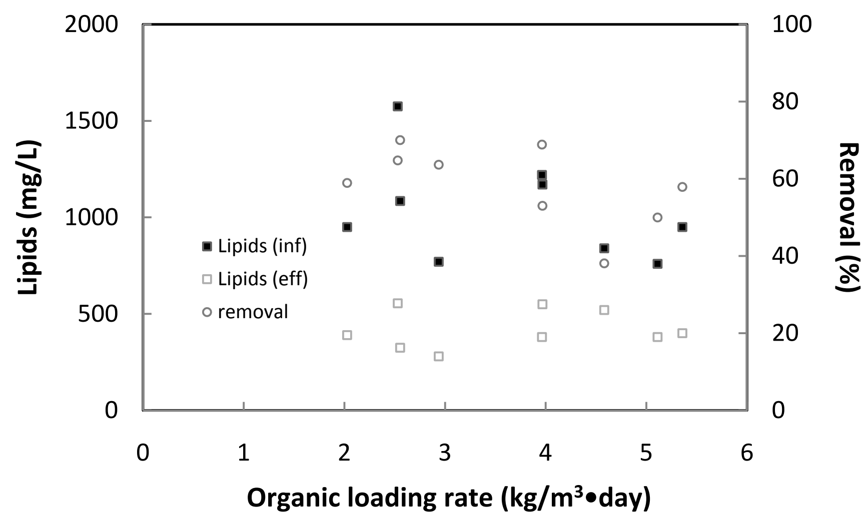


Figure 13. Lipid removal performance in the SGBR (sample 1).

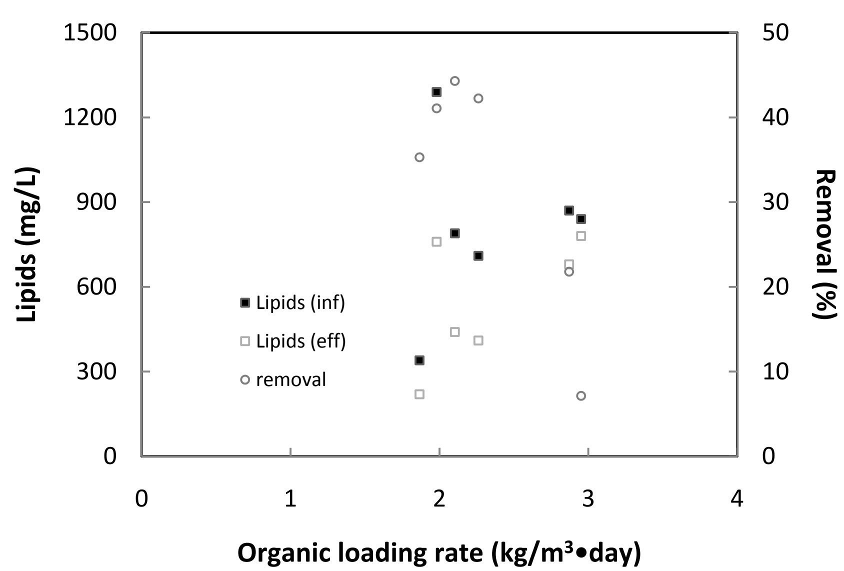


Figure 14. Lipid removal performance in the SGBR (sample 2).

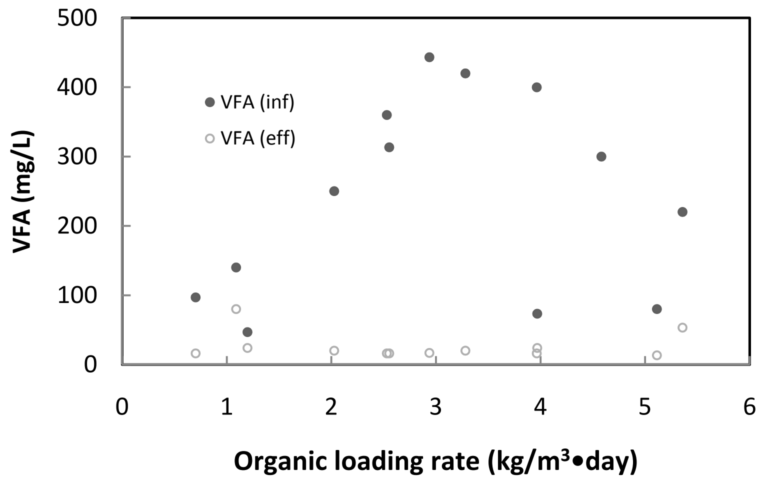


Figure 15. The relationship between influent and effluent VFA.

As shown in Figure in 15, the VFA concentration in the effluent remaind very low, which indicates that the SGBR was operated stably and effectively as most volatile fatty acids were transferred to methane or used for growth of granules. Fischer et al. (1983) reported that low VFA concentration in the effluent was very important for stable swine wastewater treatment. Hill and Bolte (1986) also showed that low VFA concentration in the effluent signified more stable operating characteristics.

VFA concentration in the effluent from an anaerobic digestion process is commonly less than 300 mg/L (Speece, 1996). When propionate accumulates in a reactor, methanogenic microbial growth is often restricted. In addition, sudden increase in concentration of either acetate or butyrate also stimulated the anaerobic processes (Hobson and Shaw, 1976).

It is essential to provide VFA for granules in order to maintain them in a syntrophic condition. However, it does not imply that VFA are nutrients for the growth of granules. A

VFA mixture (acetate, propionate, n-butyrate, n-valerate, iso-valerate, iso-butylate, and 2-methylbutylate) was found to be slightly inhibitory (Stronach et al., 1986).

The ionization of VFA is related to the pH. Unionized VFAs can move into the cell membrane easier than ionized VFAs (Andrews, 1969; Pohland and Martin, 1969). As shown Figure in 16, the effluent pH of sample 1 was 7.2~7.8. Therefore, most VFAs existed in an ionized form so that they were fermented stably.

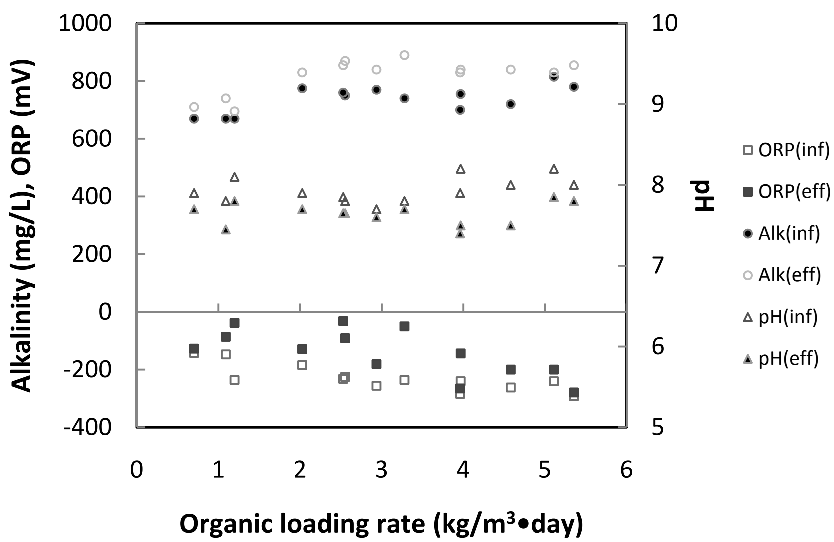


Figure 16. pH, Alkalinity, ORP of sample 1 (influent and effluent).

Gujer and Zehnder (1983) reported that thermodynamically stable anaerobic digestion is able to exist when propionate oxidation, acetate decarboxylation, and hydrogen oxidation are balanced. The optimal range of propionate and acetate are between 10^{-4} and 10^{-3} M and the partial pressure of hydrogen should not be grater than 0.1 kN/m^2. McCarty and Smith (1986) also pointed out that the partial pressure of hydrogen should be maintained below 10^{-4} atm in the anaerobic system. However, the generation time of aceticlastic methanogens is about 4 days, while that of hydrogenotrophic methanogens is below 1 day. Besides, aceticlastic methanogens are thermodynamically inferior to hydrogenotrophic methanogens as follows: (Speece, 1996)

$$CH_3COO^- + H_2O \rightarrow CH_4 + HCO_3^- \qquad \Delta G^o = -31 \text{ kJ/mol} \qquad (1)$$

$$CO_2 + 4H_2 \rightarrow CH_4 + 2H_2O \qquad \Delta G^o = -135 \text{ kJ/mol} \qquad (2)$$

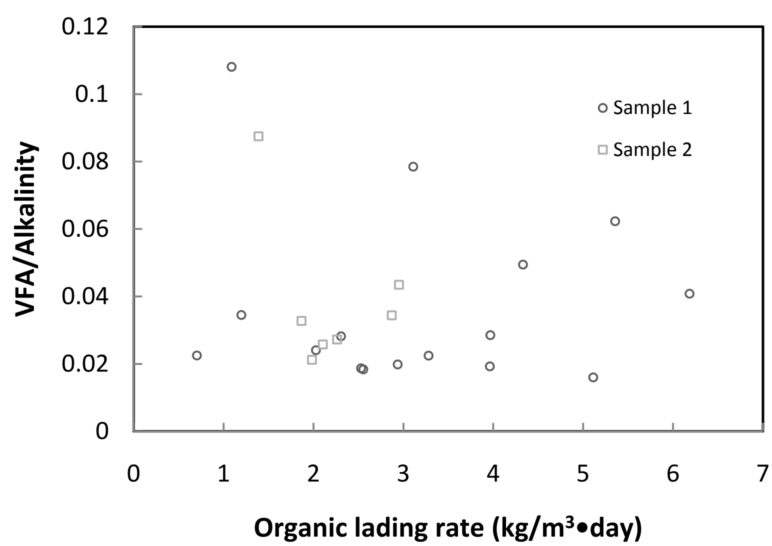


Figure 17. The ratio of VFA to alkalinity in effluent.

As shown in Figure 17, the ratio of VFA to alkalinity was not affected by the OLR. This suggests that both sample 1 and sample 2 provided sufficient alkalinity for swine manure fermentation. In order to maintain stable anaerobic digestion, it is recommended that the ratio of VFA to alkalnity should be below 0.1.

Performance of solids removal

Removal efficiencies of TS, VS, and FS for sample 1 based on organic loading rates of are shown in Figures 18, 19, and, 20, respectively.

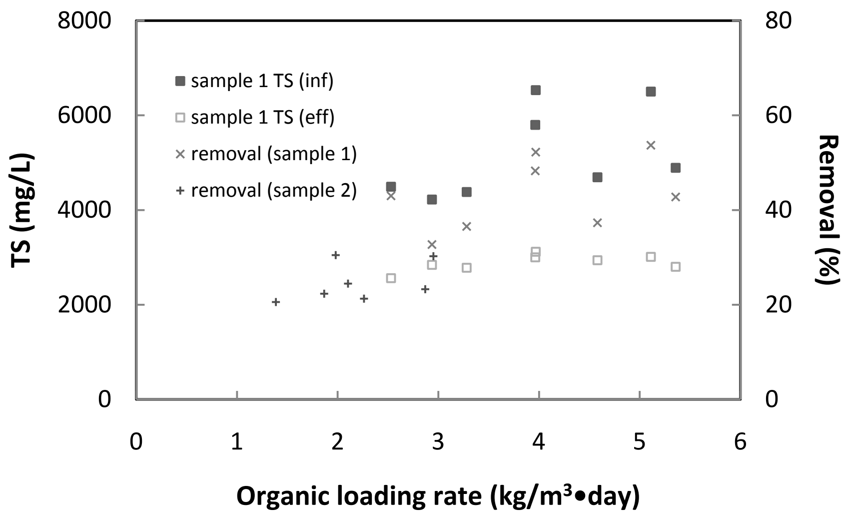


Figure 18. TS removal performance in the SGBR.

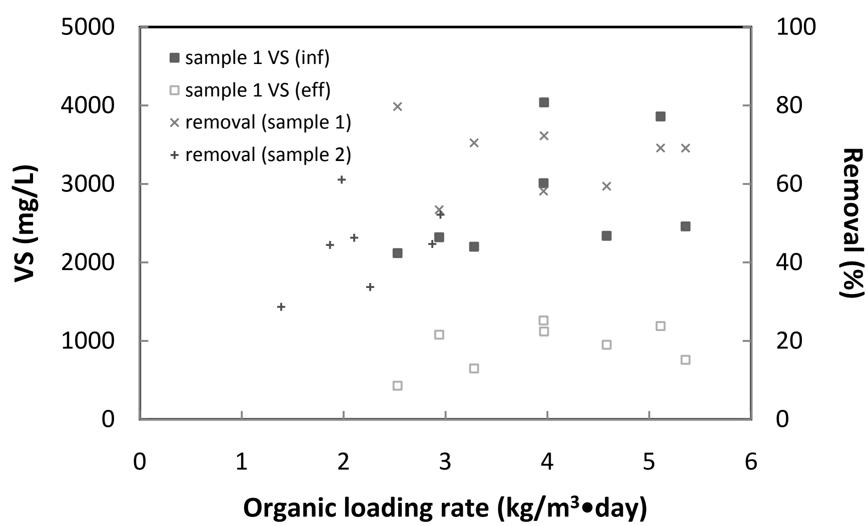


Figure 19. VS removal performance in the SGBR.

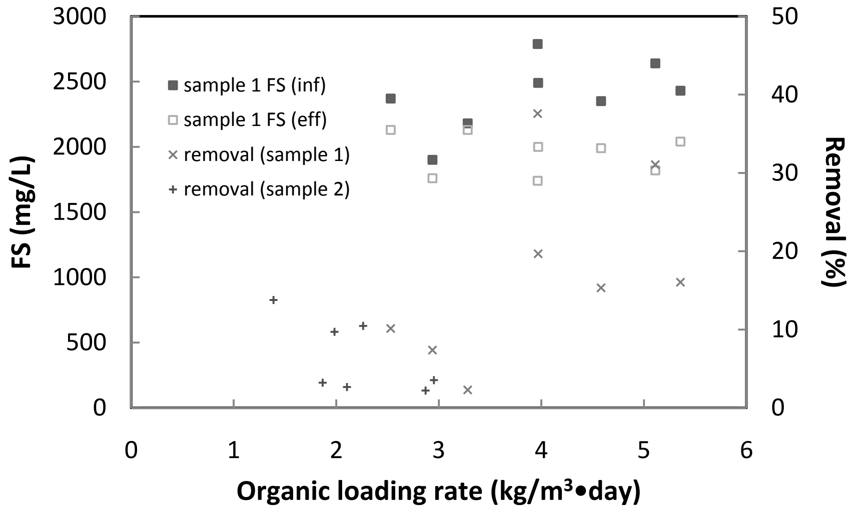


Figure 20. FS removal performance in the SGBR.

Figures 18 and 19 show that TS and VS removal efficiencies increased as the OLR increased. However, the performance of FS removal was low (Ave: 17.4±11.9%, sample 1; Ave: 6.5±4.7%, sample 2). On the other hand, VS removal efficiency of sample 1 ranged from 53.4% to 79.7% (Ave: 66.5±8.7%, sample 1; Ave: 44.4±10.8%, sample 2).

Góecki et al. (1993) also observed about 60% VS reduction in the anaerobic inclined plug flow reactor. Yang and Kuroshima (1995) reported a VS removal efficiency of 69.9% at 30^{o}C.

VS removal efficiency was consistent, regardless of the OLR, which was very similar to COD removal efficiency of sample 1 (shown in Figure in 5). It suggests that COD removal efficiency is related with VS removal efficiency (see Figure 21).

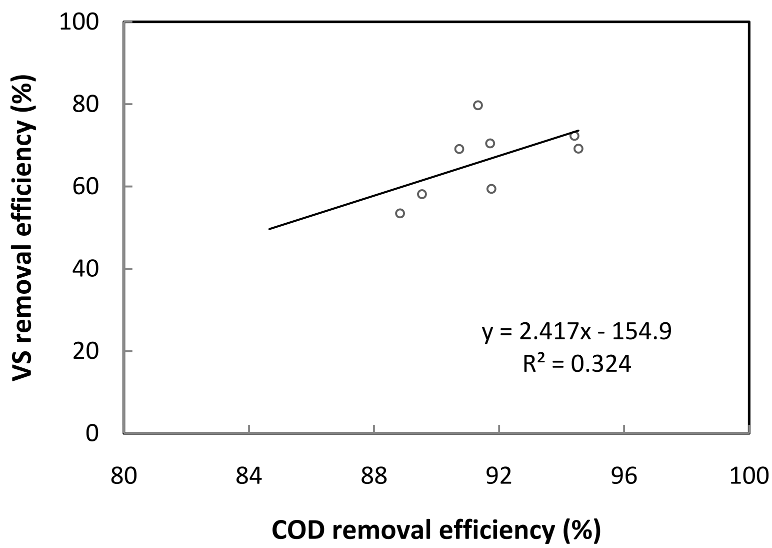


Figure 21. Relationship between COD and VS removal efficiency (Sample 1).

TSS, VSS, and FSS removal efficiencies of sample 1 are shown in Figures 22, 23, and 24, respectively.

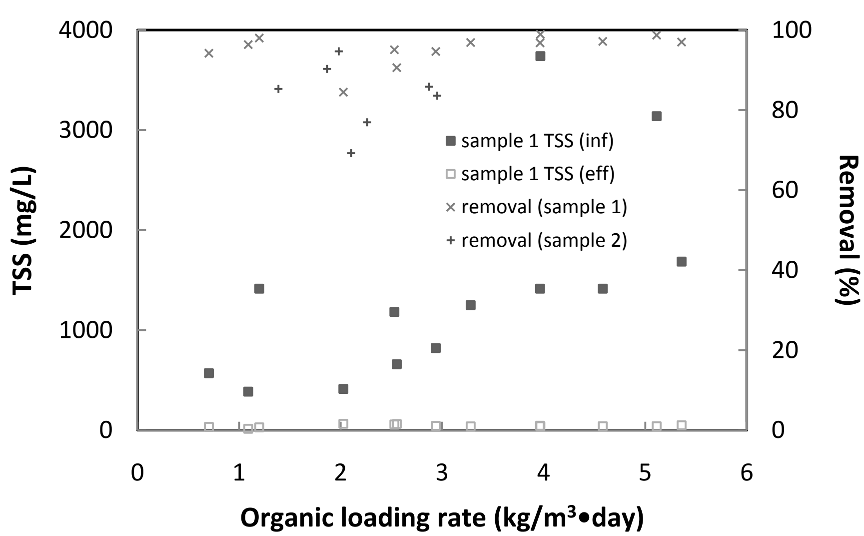


Figure 22. TSS removal performance in the SGBR.

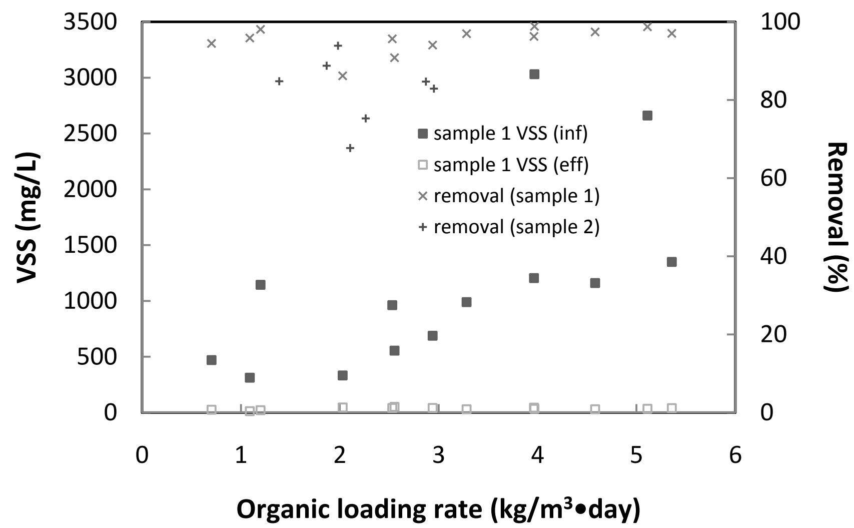


Figure 23. VSS removal performance in the SGBR.

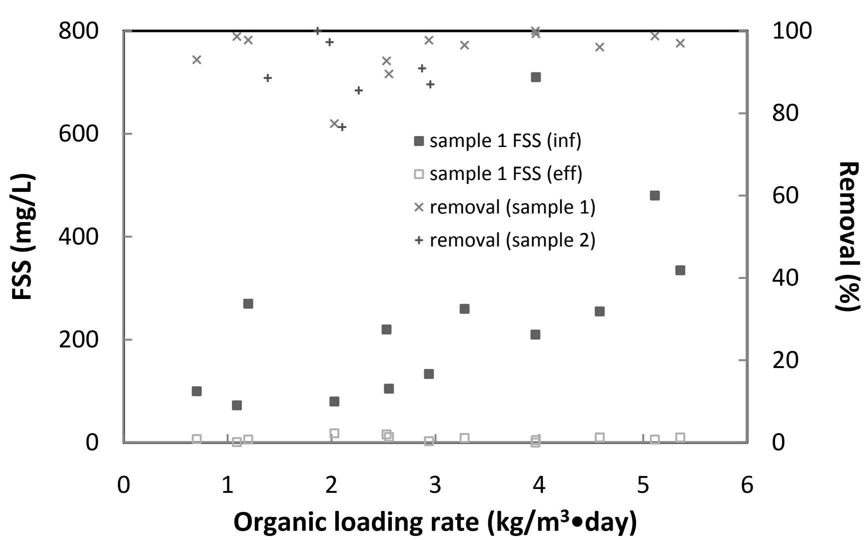


Figure 24. FSS removal performance in the SGBR.

The performance of suspended solids removal was different from that of total solids. Most suspended solids including FSS were removed in the SGBR because the SGBR is not only a bioreactor but also a biofilter. In other words, most suspended solids cannot pass through the granule bed. In addition, The SGBR is very different from any filter system because the organic matter filtered is gradually dissolved and decomposed.

Ng and Chin (1988) reported that both TSS and VSS removal efficiencies in the expanded-bed AF were 93% at a HRT of 5 days. However, the performance of suspended solids sharply decreased to 74% at a HRT of 4 days. Ng (1989) showed that 89% VSS removal efficiency could be obtained by the ASBR at an OLR of 0.4 kg/m^3•day. However, VSS removal efficiency deteriorated as the OLR increased. In case of the UASB, the performance is determined by effluent suspended solids concentration (granule washout). Foresti and Oliveira (1995) reported that 85% TSS removal could be obtained by the UASB. However, higher performance could be obtained in the SGBR because the flow direction is downflow (no granule washout) and because most suspended solids are retained in the reactor.

SGBR backwashing

The SGBR is considered as a biofilter filled with granules so that periodic backwashing is required to prevent clogging. Especially, regular backwashing is needed when the influent contains a high concentration of solids. In the case of swine wastewater, the TSS concentration of sample 1 was 385.0~3740 mg/L (Ave. 1391.7±1009.6 mg/L). The TSS concentration was not only high but also fluctuated.

In addition, fixed solids can affect the backwashing period. The FS concentration of sample 1 was 1900~4980 mg/L (Ave. 2870.5±963.8 mg/L), while the FSS concentration was 72.5~710 mg/L (Ave. 261.0±159.9 mg/L). This was because swine wastewater has a lot of inorganic materials such as phosphorus, potassium, and calcium (see Table 2, 3, and 4). In order words, these fixed dissolved solids (FDS) were not only able to be available for microorganisms as nutrients but also accumulated as salts in the reactor. Especially, calcium easily forms calcium carbonate because pk_{CaCO3} is around 7.5 in an anaerobic digester (Svardal, 1991).

As shown in Figures in 25 and 26, the backwashing results of sample 1 were not a function of the recovery time but that of the OLR. Both COD and SCOD removal efficiencies increased as the OLR increased. In addition, the performance of COD removal was quickly recovered as the SGBR was filled with active granules.

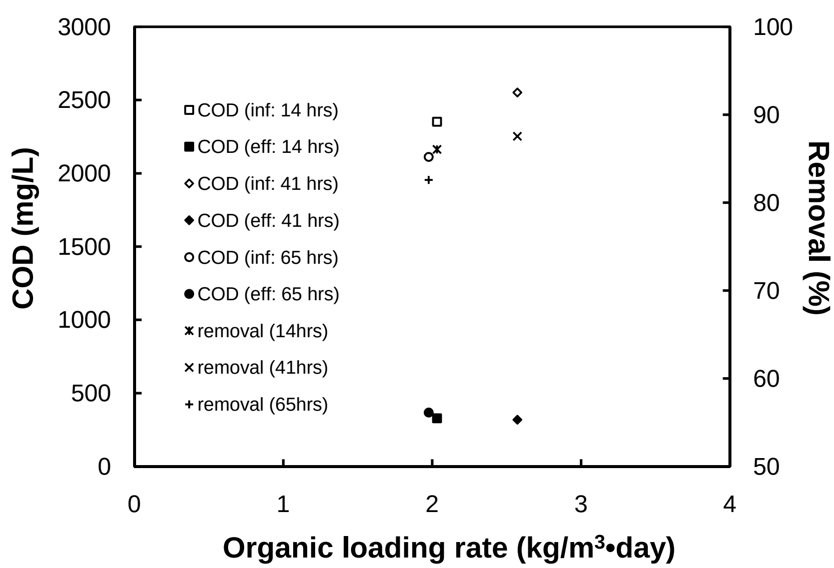


Figure 25. COD removal performance after backwashing in the SGBR (sample 1).

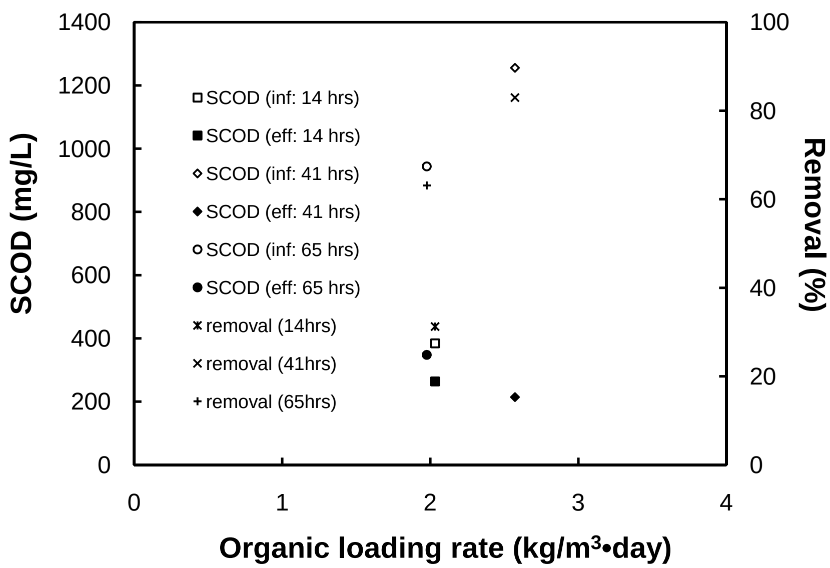


Figure 26. SCOD removal performance after backwashing in the SGBR (sample 1).

Unlike COD and SCOD, suspended solids removal efficiencies were high. TSS, VSS, and FSS removal efficiencies were shown in Figures 27, 28, and 29, respectively. The performances of suspended solids were greater than 90%. It means that the SGBR has a high performance for filtration. Additionally, organic matter in the solids also was easily degraded by microorganisms because the SGBR act as a biofilter.

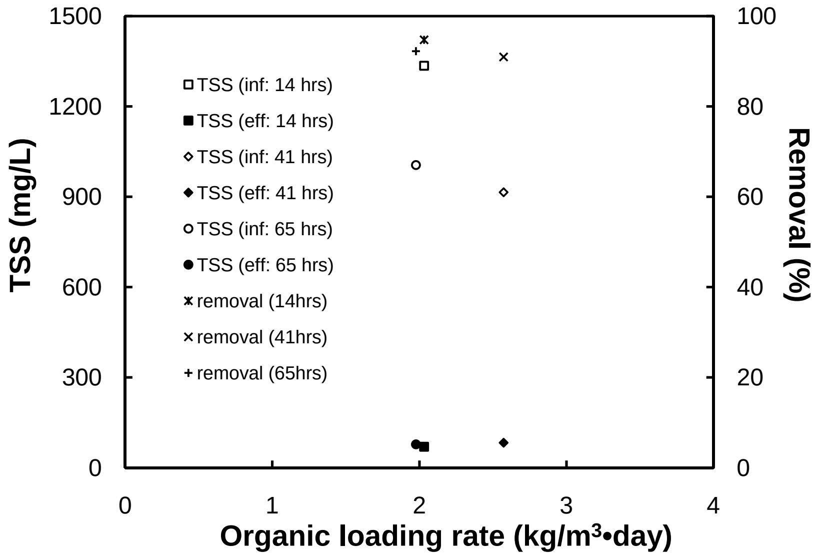


Figure 27. TSS removal performance after backwashing in the SGBR (sample 1).

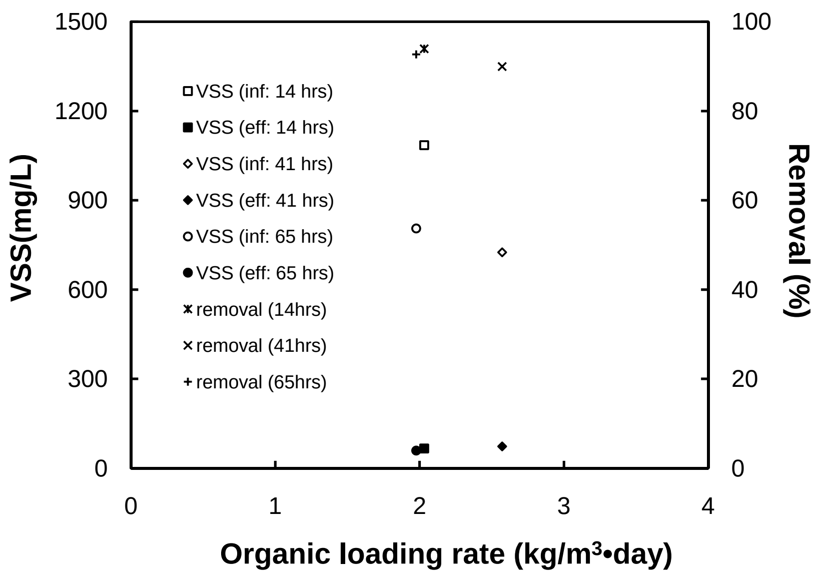


Figure 28. VSS removal performance after backwashing in the SGBR (sample 1).

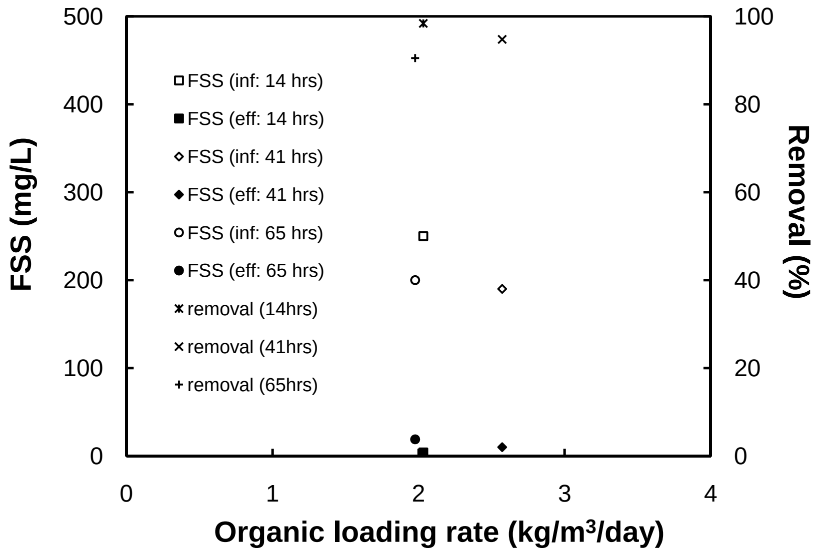


Figure 29. FSS removal performance after backwashing in the SGBR (sample 1).

Gas production and composition

All methane in anaerobic digestion is generated as follows:

$$CH_3COO^- + H_2O \rightarrow CH_4 + HCO_3^- \qquad (3)$$

$$CO_2 + 4H_2 \rightarrow CH_4 + 2H_2O \qquad (4)$$

Around 70% of the methane in anaerobic digestion comes from acetate. The other methane originates from hydrogen oxidation. The gas production rate of sample 1 in the SGBR is shown in Figure 30. Most methane productions were about 0.20~0.35 m^3 CH_4/kg $COD_{removed}$. The deficit of the theoretical methane production rate was estimated to be used for cell maintenance and regeneration.

Ng and Chin (1988) reported that the methane production rate in the expanded-bed AF was 0.06~0.18 m^3 CH_4/kg $COD_{removed}$ •day.

In addition, the methane production based on $VS_{destroyed}$ (m^3 CH_4/kg $VS_{destroyed}$) was similar to or slightly higher than the one based on the $COD_{removed}$. This suggests that the VS in the influent was solublized and was converted to methane in the SGBR. Reported biodegradability of swine wastewater is 0.32 to 0.48 m^3 CH_4/kg $VS_{destroyed}$ (Hashimoto, 1984; Safley and Westerman, 1990; Andreadakis, 1992). This range is equivalent to 40~60% reduction of VS. Yang and Kuroshima (1995) achieved 0.42 m^3 CH_4/kg $VS_{destroyed}$. Chea et al. (2008) reported 0.72 m^3 CH_4/kg $VS_{destroyed}$. In this study, the methane production based on $VS_{destroyed}$ of sample 1 was 0.09~0.73 (Ave. 0.39±0.21 m^3 CH_4/kg $VS_{destroyed}$) and VS removal efficiency was 53.4~79.7% (Ave. 66.5±8.69%).

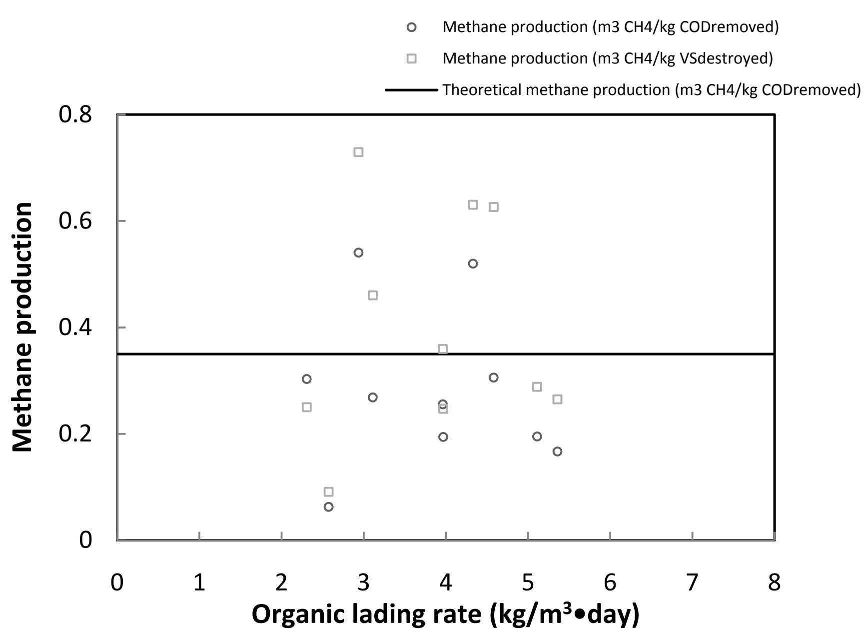


Figure 30. Methane production rate in the SGBR (Sampe 1).

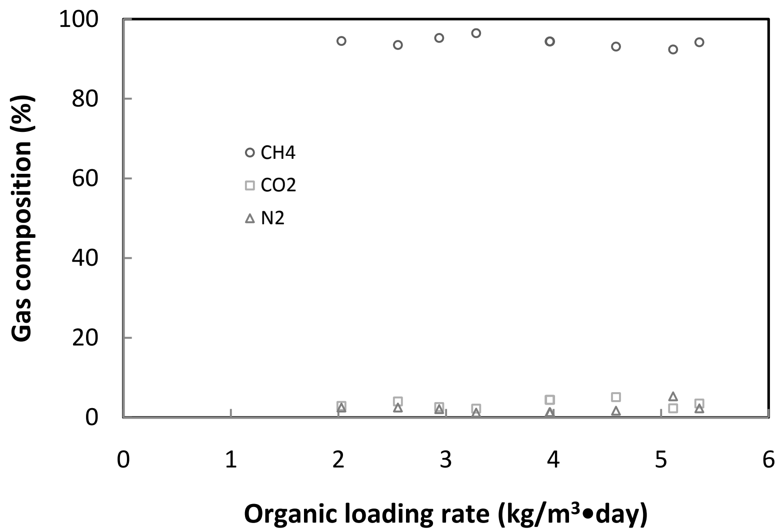


Figure 31. Gas composition in the SGBR (Sample 1).

As shown in Figure 31, methane composition maintained greater than 90%, irrespective of the OLR. This suggests that methanogesis in the SGBR was mainly affected by hydrogenotrophic methanogens. The generation time of hydrogenotrophic methanogens is four times faster than that of aceticlastic methanogen. In addition, the hydrogen concentration increases as OLR increases. In other words, granules generated a great amount of methane by the syntrophic relationship between hydrogen producing bacteria and hydrogen consuming bacteria.

The high percentage of methane is also caused by the amount of proteins and lipids. As shown in Table 2, methane production per protein or lipid is greater than that of carbohydrates. Even though both proteins and lipids removal efficiencies were not better than carbohydrate removal efficiency, the amount of methane produced from either proteins or lipids was greater than that produced from carbohydrates.

Methane is used as a heating resource with an energy value of 35,846 kJ/m^3 at 0°C, 1 atm (Metcalf and Eddy, 2003). Speece (1996) also reported that the temperature increase would be 3.3°C per 1000 mg $COD_{converted}$/L to methane. Therefore, the economic value of methane produced from swine wastewater was high.

Ng and Chin (1987) reported that 75~84% methane could be obtained by the AF. Ng and Chin (1988) treated swine wastewater by the expanded-bed AF. They achieved 82~89% methane. Ng (1989) showed that methane in the ASBR was 76~80%, irrespective of the OLR. Góecki et al. (1993) reported that average methane content in the anaerobic inclined plug flow reactor was 65% regardless of the OLR (1~7 kg/m^3•day).

Methane composition is a function of the F/M ratio, temperature, biomass inventory, and the wastewater contact time. In addition, the HRT and the SRT are controlling design parameters for complex organic pollutants that are slowly degraded (Speece, 1996). In case of swine wastewater, there is so a lot of non-biodegradable organic matter. However, the performance of COD removal was great and methane composition was above 90% during the study (sample 1), which is due to the fact that the SRT in the SGRB is greater than 300 days (Evans, 2004b).

Kinetics and Mass balance

From the steady state SCOD effluent, COD removal efficiency, and MLVSS concentration, the half-velocity constant K_s and the maximum rate of substrate utilization, k_{max} can be estimated as in (5) below (Dague et al., 1998).

$$\frac{X \bullet HRT}{S_o - S_e} = \frac{K_s}{k_{max}} \bullet \frac{1}{S_e} + \frac{1}{k_{max}} \quad (5)$$

where,

X: biomass concentration (MLVSS), mg/L

HRT: hydraulic retention time, day

S_o: influent COD concentration, mg/L

S_e: effluent SCOD concentration, mg/L

K_s: half-saturation coefficient, mg/L

k_{max}: maximum specific substrate removal rate, 1/day

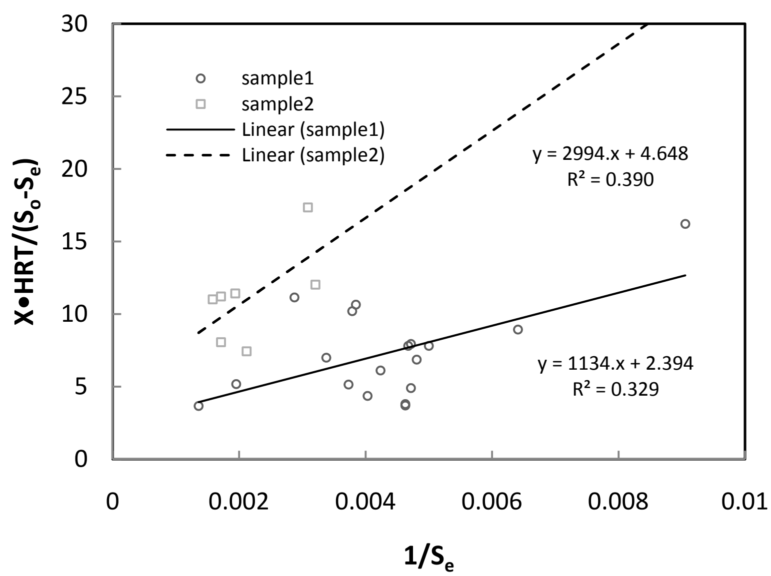


Figure 32. Estimation of k_{max} and K_s of sample 1 and sample 2 using the Monod kinetic relationship.

As shown in Figure 32, k_{max} and K_s of sample 1 were 0.418 day^{-1} and 474.0 mg/L, respectively, whereas those of sample 2 were 0.215 day^{-1} and 643.7 mg/L, respectively. The ratio of k_{max}(sample 1)/k_{max}(sample 2) was 1.94 and that of K_s(sample 1)/K_s(sample 2) was 0.74. This suggests that the organic matter in sample 1 could be degraded more easily than that of sample 2, and explain why the performance of sample 1 was greater than that of sample 2. Temperature affects temperature-dependent constants such as specific growth rate (k), decay, biomass yield, and K_s (Speece, 1996). Therefore, one of the reasons why the value of k_{max} is low is due to the ambient temperature.

The biomass yield in the SGBR can be estimated from the COD mass balance. The COD mass balance in the SGBR is as follows:

$$QC_{in} - QC_{out} - Q_{gas}C_{CH_4} + \text{generation} = \frac{dC}{dt}V \tag{6}$$

where, Q: flow rate (m^3/day)

C_{in}: concentration of influent COD (kg/m^3)

C_{out}: concentration of effluent COD (kg/m^3)

Q_{gas}: gas flow rate (m^3/day)

C_{CH4}: concentration of methane (kg/m^3)

Generation: biomass growth rate (kg/day)

$\frac{dC}{dt}V$: accumulation in the SGBR (kg/day)

If steady state, $\frac{dC}{dt}V = 0$

$$QC_{in} - QC_{out} - Q_{gas}C_{CH_4} + generation = 0 \tag{7}$$

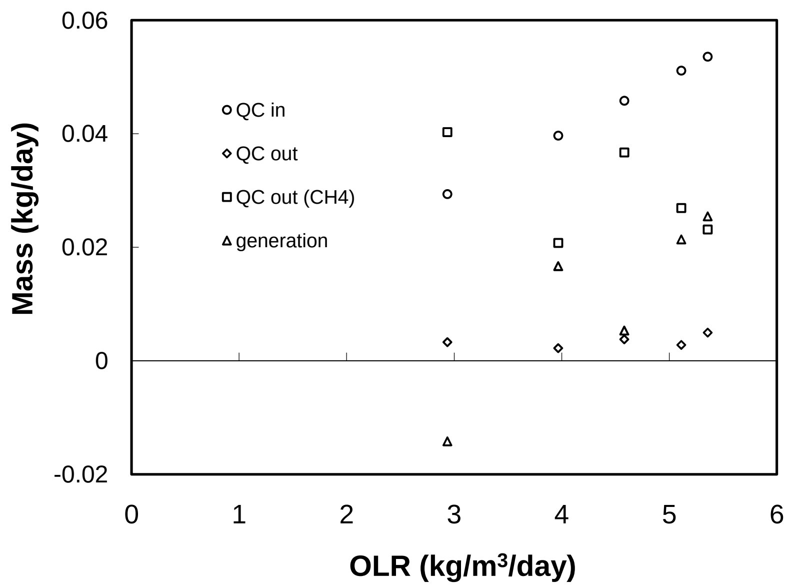


Figure 33. Mass balance in the SGBR (sample 1).

As shown in Figure 33, the methane production rate remained between 0.02 kg/day and 0.04 kg/day, while the biomass growth rate sharply increased as the OLR increased. It is evident that the amount of biomass was proportional to the OLR. However, it is essential to consider the operating condition of the reactor, wastewater concentration, and type of microorganisms (decay of biomass and change of species) in order to estimate an exact biomass yield coefficient. The amount of protozoa and bacteria tend to increase at low temperatures, while the amount of metazoa and algae decrease at high temperatures (Murakami et al, 1992). Therefore, the anaerobic reactor should maintain the optimum condition to control the prey-predator relationship.

Modeling in the SGBR

There are many anaerobic digestion models including high-rate anaerobic digesters such as the AF, the UASB, and the ASBR (Zeng et al., 2005; Saravanan and Sreekrishnan, 2006; Toshio et al., 2007). However, a model has not been developed for the SGBR. To develop the SGBR model, factors such as advection, diffusion/dispersion, and decay of microorganisms should be taken into account, as the SGBR is filled with granules and wastewater flows downward into the reactor. Once wastewater goes through the media, the constituents in the influent are affected by physical and biochemical reactions such as advection, diffusion, dispersion, and decay. The flow diagram of the SGBR is shown in Figure 34.

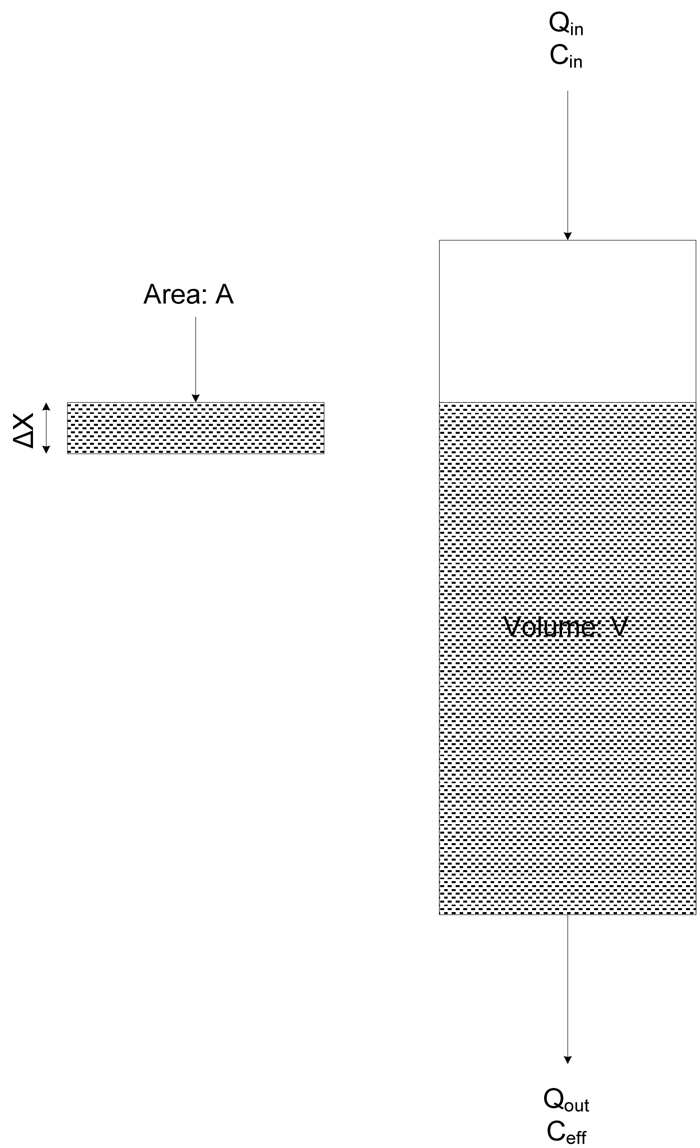


Figure 34. Flow diagram of the SGBR.

Where,

Q_{in}: influent flow rate [L^3/T]

C_{in}: influent substrate, COD [M/L]

Q_{out}: effluent flow rate [L^3/T]

C_{out}: effluent substrate, COD [M/L]

V: reactor volume [L^3]

A: unit area [L^2]

ΔX: unit length [L]

[SGBR model]

$$\frac{\partial(CA\Delta X)}{\partial t} = CvA - \left(CvA + \Delta(CvA)\right) + q_c A - (q_c + \Delta q_c)A - kCA\Delta X \tag{8}$$

where,

$\frac{\partial(CA\Delta X)}{\partial t}$: Rate of change of contaminant mass

$CvA - \left(CvA + \Delta(CvA)\right)$: Net advection flux

$q_c A - (q_c + \Delta q_c)A$: Net diffusion/dispersion flux

$kCA\Delta X$: Decay of microorganism

k: Decay constant (0.035 day^{-1}; Speece, 1996)

$$q_c = -D\frac{\partial C}{\partial x} \tag{9}$$

where,

D: Diffusion coefficient (1.50 m^2/day)

$$\frac{\partial C}{\partial t} = -\frac{\partial(Cv)}{\partial x} + \frac{\partial\left(D\frac{\partial C}{\partial x}\right)}{\partial x} - kC \tag{10}$$

$$\text{Conservaion of water mass} \rightarrow \frac{\partial v}{\partial x} = 0$$

$$\frac{\partial C}{\partial t} + \frac{Q}{A}\frac{\partial C}{\partial x} = \frac{\partial\left(D\frac{\partial C}{\partial x}\right)}{\partial x} - kC \quad (11)$$

Suppose, steady state and D is constant.

$$D\frac{d^2C}{dx^2} - \frac{Q}{A}\frac{dC}{dx} - kC = 0 \quad (12)$$

Use second-order homogeneous equations with constant coefficients (Kreyszig, 1999).

$$C_{eff} \propto e^{\sigma_x} \quad (13)$$

$$D\sigma^2 - \frac{Q}{A}\sigma - k = 0 \quad (14)$$

$$\sigma_{\pm} = \frac{\frac{Q}{A} \pm \sqrt{\left(\frac{Q}{A}\right)^2 + 4Dk}}{2D} \quad (15)$$

$$\sigma_+ > 0, \quad \sigma_- < 0 \quad (16)$$

$$C_{eff} = Ae^{\sigma_+} + Be^{\sigma_-} \quad (17)$$

$$\text{Boundary Condition: } C = C_o \text{ at } X = 0, -D\frac{\partial C}{\partial x} = 0 \text{ at } X = 0$$

$$C_o = A + B$$

$$A\sigma_+ + B\sigma_- = 0$$

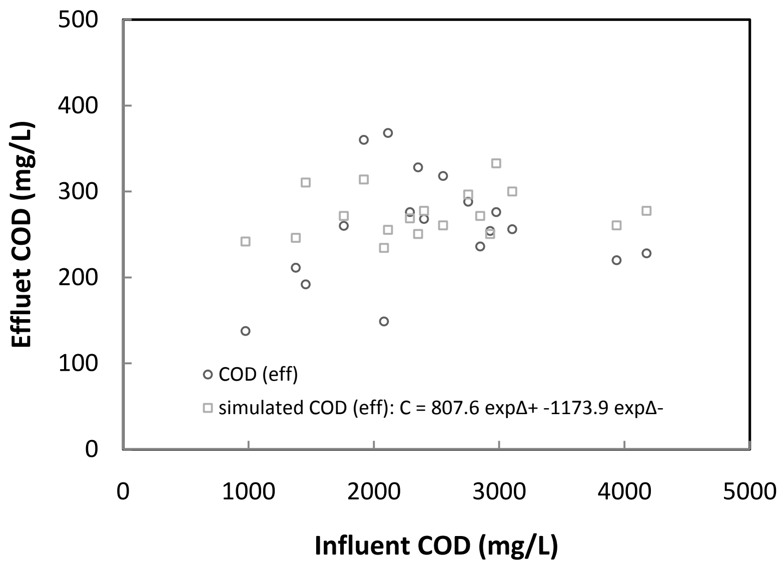


Figure 35. Simulated effluent COD using the SGBR model.

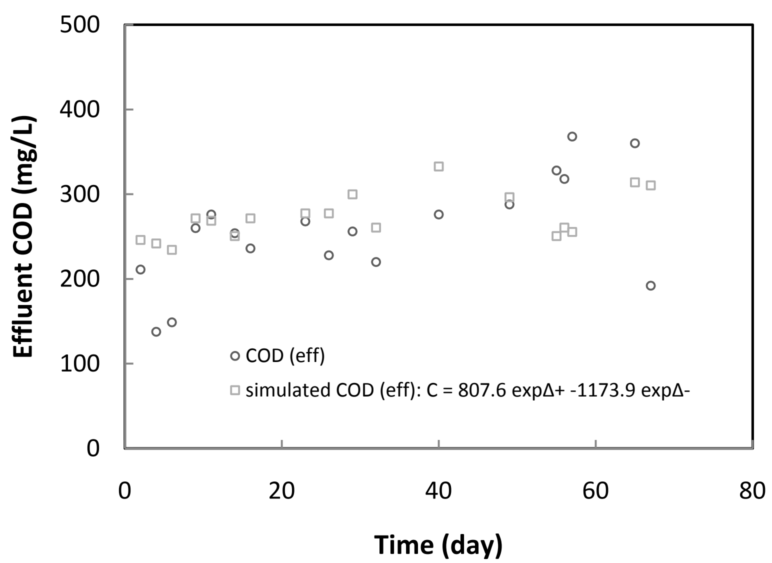


Figure 36. Comparison between experimental effluent COD and simulated COD during operation period.

Figure 35 shows simulated result using the SGBR model, and the comparison with experimental result is presented in Figure 36. From Equation (17), A is 807.6; B is -1173.9; σ_+ is 0.19~0.26 (Ave 0.22±0.02); and σ_- is -0.49~-0.46 (Ave -0.47±0.01). The simulation COD matched well with the experimental effluent COD except at low concentrations. The discrepancy at low concentrations was likely due to the assumption that the decay constantwas fixed independent of temperature in the SGBR model. As shown in figure 36, the simulated effluent COD did not match the experimental effluent COD from Day 57. This difference was due to the backwashing and the recovery of biomass.

Vertical analysis in the SGBR

The pH variation in the SGBR is shown in Figure 37. The pH increased at the height of 80% due to the release of ammonia. However, the pH was affected by chemical equilibrium at the height 50 and 20%. Especially, bicarbonate-VFA equilibrium plays important roles in anaerobic digestion.

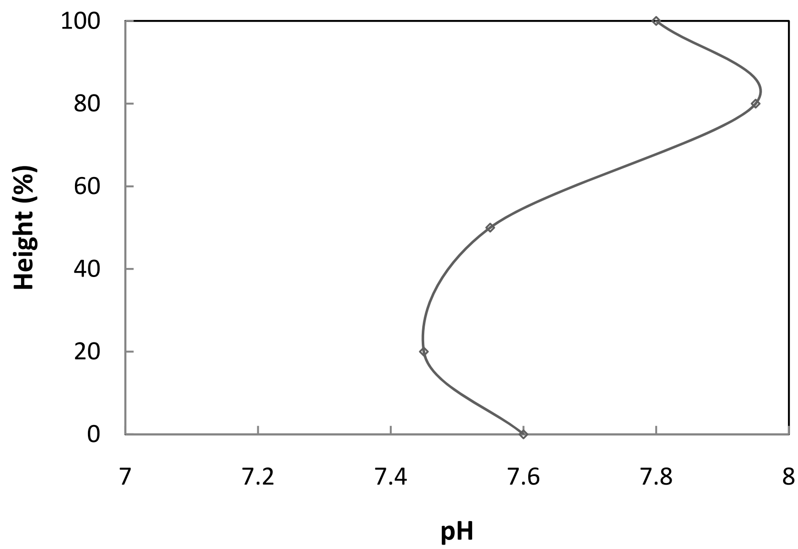


Figure 37. pH variation in the SGBR (sample 2).

Two concepts should be considered to understand alkalinity variation in anaerobic digestion. One is the bicarbonate alkalinity system; the other is the volatile acid alkalinity system. The latter involves CO_2, H_2O, NH_3, or H_2S.

$$H_2CO_3 + NH_3 \leftrightarrow NH_4^+ + HCO_3^- \tag{18}$$

$$RCOO^- + NH_4^+ + H_2O + CO_2\uparrow \tag{19}$$

As shown in Figure 38, alkalinity sharply increased at the height of 80%. It indicates, up to the height of 80%, alkalinity is mostly affected by the bicarbonate alkalinity system and ammonia release. Afterwards, the amount of alkalinity was caused by the volatile acid alkalinity system and ammonia release.

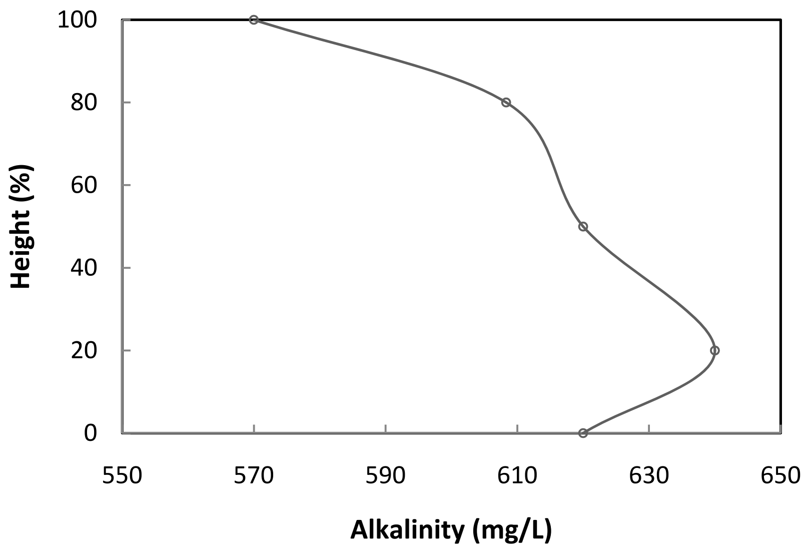


Figure 38. Alkalinity variation in the SGBR (sample 2).

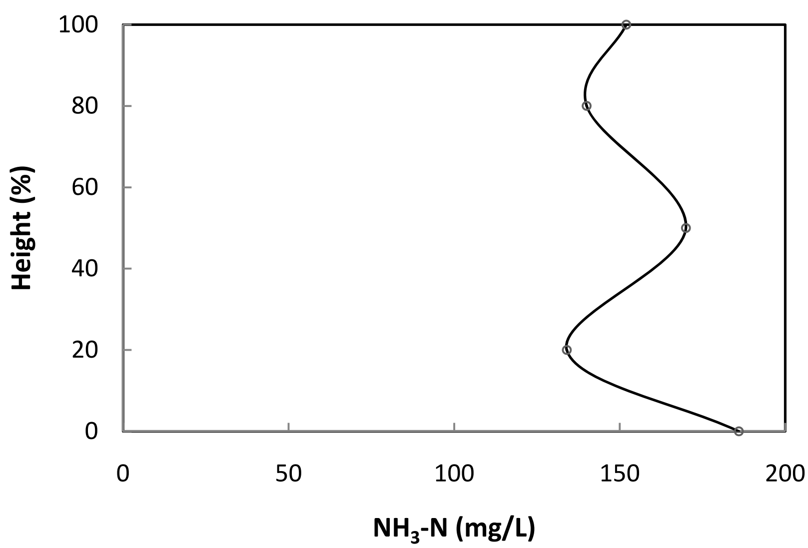


Figure 39. NH_3-N variation in the SGBR (sample 2).

Ammonia release was related to the pH. As shown in Figure 39, the ammonia concentration at the height of 80% slightly decreased as the pH increased. It was considered that ammonium was used to generate new cell structures. Typically, 14% of a cell is comprised of nitrogen (Gaudy and Gaudy, 1980). The ammonia concentration gradually increased except at the height of 20%. Free ammonia is a function of the pH.

$$NH_4^+ \leftrightarrow NH_3 + H^+ \text{ (pKa = 9.27 at 35}^{o}\text{C)} \qquad (20)$$

At a pH of 7.0, free ammonia represents 0.5% of the total ammonia, while 5.1% of free ammonia exists at a pH 8.0. De Baere, et al. (1984) showed that more than 50~80 mg/L free ammonia could inhibit unacclimated methanogens. The threshold concentrations of some toxicants can be increased ten fold as much as with acclimated biomass (Speece, 1996). Ammonia also affects biomass generation. Hulshoff et al. (1983) showed that granulation was inhibited at an ammonia concentration of 1000 mg/L.

COD and VFA variation in the SGBR are shown in figures 40 and 41, respectively.

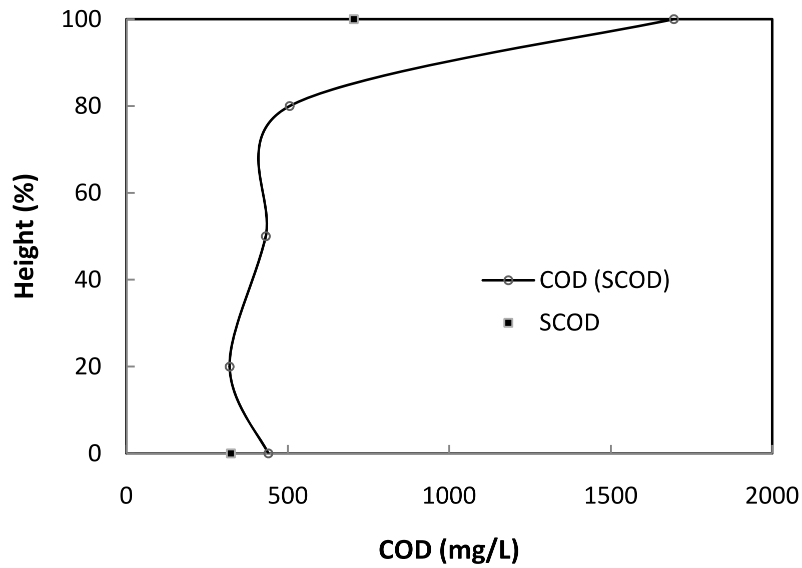


Figure 40. COD variation in the SGBR (sample 2).

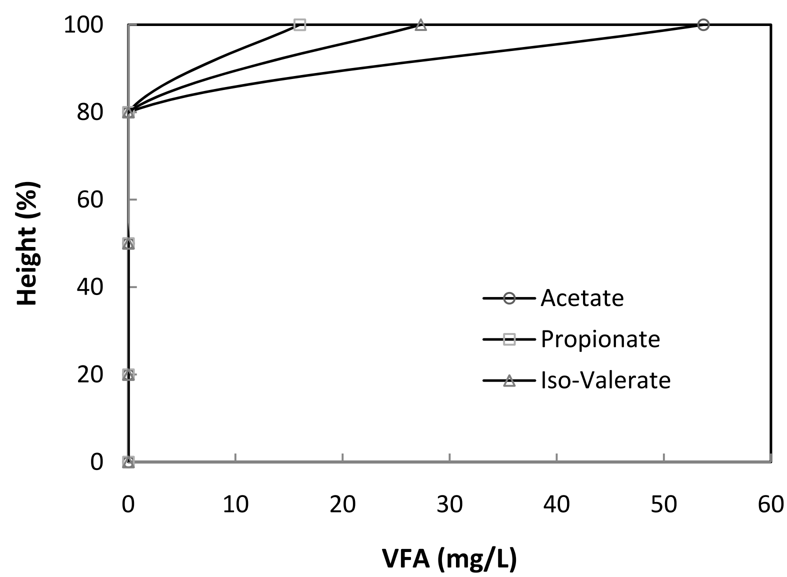


Figure 41. VFA variation in the SGBR (sample 2).

At the height of 80%, COD removal was practically complete and VFA was also completely consumed. This suggests that the SGBR is a pseudo-plug flow reactor. Most solids were filtered and most soluble organic matter was degraded at the top of the reactor. In other words, the activity of microorganism at the top was higher than those in the middle and bottom. Therefore, granules from bottom to the height of 80% acted as polishers. Conversely, Evans (2004a) reported that the SGBR has CSTR characteristics.

Solids variation in the SGBR is shown in Figure 42.

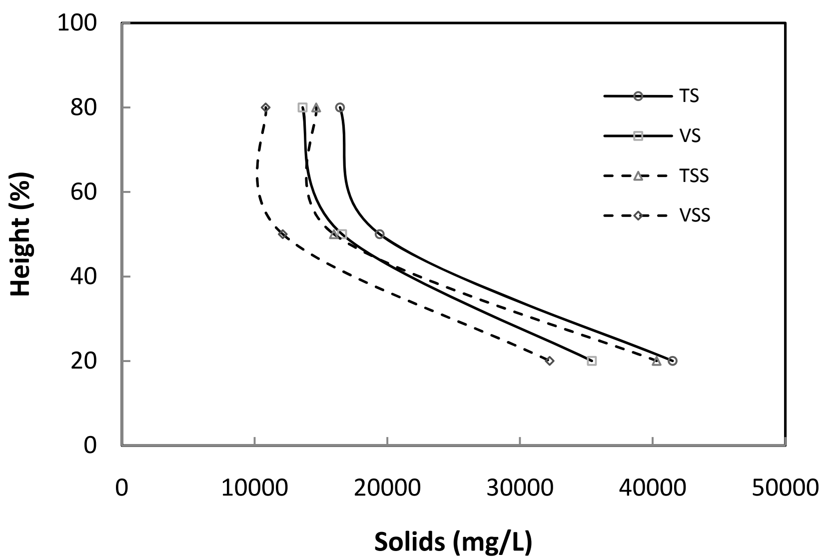


Figure 42. Solids variation in the SGBR (sample 2).

Solids concentrations at a height of 50% were similar to that at the height of 80%. At a height of 50%, the solids concentration gradually increased, but the difference was not significant. Solids concentrations dramatically increased from a height of 50% to a height of

20%. TS, VS, TSS, and VSS concentrations at the height of 20% were almost two times as much as those at the height of 50%. In addition, the size of granules at the height of 20% were also greater than that of either at the height of 80% or of 50%. It is because granules were rearranged by backwashing and also because the physical arrangement was caused by flow and digester gas in the reactor.

CHAPTER 5. ENGINEERING SIGNIFICANCE

Swine manure is one of the most refractory wastewaters because it contains a significant portion of non-biodegradable organic matter. In addition, its wastewater characteristics dramatically vary from place to place. Therefore, performances of many anaerobic bioreactors treating swine manure were not satisfactory or consistent. In order to treat any wastewater stably, a system has to maintain a consistent performance. In case of the SGBR, unlike other processes such as the AF and the UASB, a high performance could be obtained and maintained, irrespective of the OLR. In addition, the economic value of methane produced from the SGBR was high. Given that this study was performed at ambient condition, both high performance and high methane content in the SGBR were very attractive.

The SGBR is not only a bioreactor but also a filter system. This suggests that the periodic backwashing is needed to maintain a stable operation. It is therefore essential to understand backwashing characteristics of the SGBR. COD removal efficiency after backwashing was not a function of the recovery time but that of the OLR. Unlike the AF and the UASB, the SGBR is filled with granules so that the recovery time is very fast. The SGBR is thus able to stably operate at high OLRs after backwashing. It is more important in a full-scale plant because it determines a continuous operation.

In a vertical test, the SGBR was considered as a pseudo-plug flow reactor. One of the advantages of a plug flow reactor is that it requires less volume to treat wastewater than a continuous stirred tank reactor. With active granules at the top, most of the organic matter and solids were removed in the SGBR. This suggests that the volume of SGBR can be adjusted according to wastewater characteristics. In addition, the SGBR is filled with a high concentration of granules, especially in a lower part of reactor, so that it is possible to polish the effluent and reduce organic matter and solids.

A model was developed for the SGBR. The SGBR model has terms for advection, diffusion/dispersion, and decay of microorganisms. A model is indispensable for understanding a system and estimating results correctly. The results obtained by simulations were similar to the experimental results. In order to scale up the reactor correctly, it is recommended that a pilot study and a simulation should be done. Therefore, the model development and the verification are very important to rector design.

CHAPTER 6. CONCLUSIONS

The SGBR successfully treated two types of swine wastewater (sample 1 and sample 2) and showed great performance.

COD removal efficiency increased as the OLR increased. This indicates that the SGBR contained an abundance of active granules in the reactor. Also, high performance could be obtained at high OLRs because the SRT of the SGBR is greater than 300 days.

Most suspended solids were removed as the SGBR was not only a bioreactor but also a filter system. The performance of suspended solids removal was high, regardless of the OLR. Organic matter in the solids was also hydrolyzed and degraded in the SGBR. Therefore, the SGBR served as a biofilter system.

The SGBR needs periodic backwashing. COD and SCOD removal efficiencies in effluent after backwashing were not a function of the recovery time but that of the OLR. The performance of COD removal was proportional to the OLR. In the mean time, suspended solids removal efficiency was greater than 90%. It is very important to understand the backwashing characteristics of the SGBR in order to operate the process stably and maintain a consistent performance.

The methane production ranged from 0.20 to 0.35 m^3 CH_4/kg $COD_{removed}$. The methane production based on $VS_{destroyed}$ (m^3 CH_4/kg $VS_{destroyed}$) was similar to or slightly higher than the one based on $COD_{removed}$. It was demonstrated that there was a close relationship between VS influent and COD influent. In addition, methane composition maintained greater than 90%, irrespective of the OLR. The economic value of methane produced from swine wastewater was high.

From steady state SCOD effluent, COD removal efficiency, and MLVSS concentration, k_{max} and K_s of sample 1 were 0.418 day^{-1} and 474.0 mg/L, respectively, whereas those of sample 2 were 0.215 day^{-1} and 643.7 mg/L, respectively. These results were evidence that the sample characteristics were different. In addition, the biomass growth rate sharply increased as the OLR increased from the mass balance analysis.

The SGBR model was developed with concepts of advection, diffusion/dispersion, and decay of microorganism. The simulated COD matched with the experimental COD except at low concentrations. The SGBR model is as follows:

$$\frac{\partial(\mathrm{CA}\Delta\mathrm{X})}{\partial t} = \mathrm{CvA} - \left(\mathrm{CvA} + \Delta(\mathrm{CvA})\right) + q_c\,\mathrm{A} - (q_c + \Delta q_c)\mathrm{A} - \mathrm{kCA}\Delta\mathrm{X}$$

where,

$\frac{\partial(\mathrm{CA}\Delta\mathrm{X})}{\partial t}$: Rate of change of contaminant mass

$\mathrm{CvA} - \left(\mathrm{CvA} + \Delta(\mathrm{CvA})\right)$: Net advection flux

$q_c\,\mathrm{A} - (q_c + \Delta q_c)\mathrm{A}$: Net diffusion/dispersion flux

$\mathrm{kCA}\Delta\mathrm{X}$: Decay of microorganism

k: Decay constant (0.035 day^{-1}; Speece, 1996)

In a vertical test, the SGBR was a pseudo-plug flow reactor. Most COD was removed and VFA was also completely consumed at the height of 80%. Additionally, the solids concentration at the height of 20% was approximately twice as much as that at the height of 50%. A large amount of granules in a lower part of the SGBR was used to polish organic matter and solids.

Swine wastewater treatment has been studied for a long time. Various treatment processes have been developed and exploited in the field. Each process, however, has its own limit preventing direct discharge to a river. The SGBR also has to optimize its performance and understand its own limitations at various OLRs. In addition, the thermodynamic study for the SGBR has to be done through psychrophilic and thermophilic research.

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