Feasibility of low cost post-treatment options for the anaerobic processes of tapioca starch wastewater: Fungal Down-flow Hanging Sponge (DHS) and Bacterial DHS systems

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Abstract

Aim of this research is to study the performances of Down-flow Hanging Sponge (DHS) systems using mixed fungal culture and mixed bacterial culture for treatment of UASB effluent of tapioca starch wastewater. This study attemped compare the performance of the fungal and bacterial systems by systematically studying biokinetic coefficients by respirometry. Also, the influencing of sludge compositions to their microbial activity and effluent organic matter were evaluated. The remaining total BOD_5 ($TBOD_5$) in the effluents were about 85% removed in bacterial DHS and fungal DHS units and about 80% removed in term of soluble BOD_5 ($SBOD_5$). Organic loading rate (OLR) during the DHS system experiments were in range of 2.1-5.6 kgCOD/m^3-d, at 7 h HRT. Values of biokinetic coefficients showed that substrate utilization rate and maximum specific growth rate were higher for the fungal sludge in the first segment which explained the higher organic removal rate for this culture. In steady state, the concentration of retained sludge in fungal DHS remained almost constant suggesting that the degradation of old biomass nearly balance the accumulation of the fresh one. However, macromolecular compounds such as protein and carbohydrate can comprise a significant portion of organic contents in the DHS effluents.

Keywords: Downflow Hanging Sponge (DHS) System, Fungal Sulture, Bacterial Sulture, Post Treatment System, Tapioca Starch Wastewater

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Introduction

High rate anaerobic represent low cost and sustainable technology for tapioca starch wastewater treatment, because of its low construction, operation and maintenance cost, small land requirement, low excess sludge and the production of biogas.

Although, anaerobic treatment can typically reduce chemical oxygen demand (COD) levels by as 90% w/w, one of the factor limiting its use at present is fact that the COD effluent usually still too high to meet the discharge standards applied in most industrialized countries and Thailand (Amatya, 1996; Barker et al., 1999a). Therefore, the effluent from anaerobic reactor usually required a post treatment step as a means to adapt the treated effluent to the requirements of environmental legislation and protect the receiving water bodies. However, some form of post treatment is required and this is often disproportionately expensive in relation to the cost of the whole treatment process since the marginal removal cost is so high.

The combinations of UASB and various configuration of DHS have been developed and evaluated over the 15 years period already for various type of wastewater, including domestic wastewater or sewage (Agrawal et al.,1997; Araki et al.,1999; Chuang et al.,2007; Machdar et al.,1997; Machdar et al.,2000; Uemara et al., 2002; Tadukar et al., 2005; Tawfik et al., 2006a; Tawfik et al., 2006b. Tripathi et al., 2007) actual dye wastewater (Ohashi et al., 2006) These results of all these researchers suggested DHS system to be an excellent system for post-treatment of anaerobically pre-treated sewage. The main advantages of using DHS systems include: a rapid and dense colonization of biomass, a high specific surface area of the packing material which can reach up to 2,400 m^2/m^3 and 97 % porosity, a sufficiently long biomass retention time allowing the application of a higher loading rate, a high process stability and no oxygen requirement and a low production of waste sludge (Tadukaret et al., 2005)Also, the high entrapment capacity of the DHS system and the retain of a high biomass concentration of 34 g VSS/L of sponge are the main reasons for the higher COD removal, nitrification process and F. coliform removal as compared to a series of RBC and trickling filter treating UASB reactor effluent. Furthermore, the DHS system is also superior to other post-treatment systems, such as, activated sludge process, sequencing batch reactor (SBR), and aerobic filter with regard to COD removal, nitrification efficiency and F. coliform removal for domestic wastewater treatment processes (Tadukaret et al., 2005)Accordingly, it is strongly recommended to use DHS system for post-treatment of anaerobic pre-treated sewage. Thus, Downflow Hanging Sponge (DHS) system offers an attractive method to treat UASB effluents from tapioca starch industry.

However, the vast quantities of UASB effluent treating starch processing wastewater have higher biochemical oxygen demand (BOD) levels than town sewage, are highly polluting and can impose heavy loads on the environment or be expensive in term of sewer disposal. Heightened environmental awareness has prompted regulatory organizations to assign real economic value to environmental factors, thereby forcing industries to consider environmental factors and a variety of waste treatment techniques to address these issues. Recovery of by-products and proper disposal of waste steams have become increasingly vital. Due to economic incentives, minimizing waste streams that cannot be recovered cost-effectively has become part of standard industrial practice. Today, wastewater treatment and disposal are critical considerations in the sitting of new industrial facilities and the continued viability of existing plants. Starch companies, among many others, must continue to deal with new and tighter regulations, increased costs, and the need to optimized existing waste and by-product treatment, as well as recovery technologies (Jin et al., 2002)

Microfungi have a number of properties which make them important both scientifically and industrially. They play an important role in the food industries, are known to have a wide range of enzymes, and are capable of metabolizing complex mixtures of organic compounds occurring in most wastes. Cultivating microfungi to yield biomass protein is particularly attractive because: (i) microfungal cells contain reasonably high levels of protein; (ii) microfungi contain a lower amount of nucleic acid than bacteria; and (iii) food produced from fungi is traditionally eaten in many parts of the world. Fungi can be grown using almost any waste products that contain carbohydrates, such as confectionery and distillery waste, vegetable waste and wood processing effluents. It has been demonstrated that the cost of separating biomass from the spent cultivated broth is a significant fraction of the capital and operating costs for microbial biomass protein production (Jin et al., 2002)

Thus, the objectives of this research is to examine the potential of development of DHS systems using mixed fungal culture (FDHS) and mixed bacterial culture (BDHS) for treating the UASB effluent of tapioca starch wastewater from the General Starch Co., Ltd. in Nakhon Ratchasima, Thailand. This study attempted to compare the performances of the FDHS and BDHS systems by systematically studying biokinetics coefficients and sludge characteristics. Also, the influencing of biofilm compositions to their microbial activity and effluent organic matters were evaluated.

Material and Methods

Seed sludge

The seed initial sludge is isolated from the bottom sediment of an equalization tank from a tapioca starch factory. The enrichment process was conducted to propagate natural mixed fungi and bacteria in seed sludge. The seed sludge was added into each polyethylene container that contained 36 L of the UASB effluent wastewater by fill-and-draw process. The wastewater was mixed by diffused aeration system and adjusted pH to 4.0±0.2, which is optimum for mixed fungi growth and can prevent bacteria contamination. On the other hand, pH was adjusted to 7.0±0.2 for bacterial culture enrichment. Mixed bacterial cells, normally, settle in the bottom, whereas the fungi and filamentous bacteria would remain in the suspension (Wichitsathian, 2004). After eight hours of aeration, the biomass suspension was settled for 3 h. Around 24 L of supernatant was decanted, and a fresh medium was added preparing another batch. When mixed fungal and bacterial biomass reach to MLSS of above 3,000 mg/L and 70% COD removal, the enrichment process was stopped. The DHS system started with incubated by placing the sponge as the floating media into seed sludge for a week by fill-and-draw processes as described before.

Experimental setup and operating conditions

The schematic diagram of the experimental setup, consisting of effluent of UASB storage and DHS biofilter post-treatment unit and operating conditions are shown in Figure 1 and Table 1, respectively. The effluent steam from the UASB reactor is forwarded for polish-up to the aerobic DHS as post treatment unit. The random type DHS reactor is operated module column consisting of four identical segments connected vertically, each segment being equipped about 5.4 L of sponge randomly distributed. The dimensions of the using polyurethane foam (sponge) amount to 20×20×20 mm for the first to third segments and 30×30×30 mm for the forth segment. The DHS system made of acrylic column, with internal diameter of 14 cm. The total height of the reactor is 430 cm. The sponges used are supported by polyethylene plastic material with fins. The UASB effluent flows by peristaltic pump to the distributor is located on the top of DHS systems and rotating at 9 rpm. The oxygen is naturally diffused through perforated plate windows located at different levels of the first, second and third segments of both BDHS and FDHS systems. Treated wastewater samples were collected from each segments of the water phase from the systems.

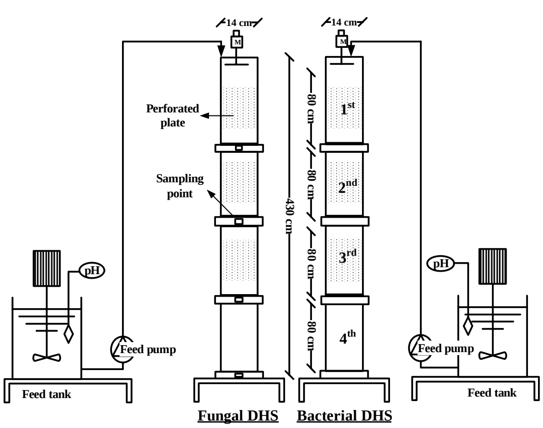

Figure 1. Schematic diagram of the experimental set-up

Table 1. The operating conditions of DHS systems

Operating conditions	Fungal DHS	Bacterial DHS
pH	4.0±0.2	7.0±0.2
HRT (h)	7.0	7.0
Down-flow velocity (m/h)	120	120
Flow rate (L/d)	75	75
OLR (kgCOD/m^3-d)	2.0-6.0	2.0-6.0

Biokinetic experiments

The oxygen uptake rate (OUR) experiment were conducted to determine the biokinetic coefficients of aerobic heterotrophs at different DHS location heights (first, second, third and fourth segment). The harvested sponges with biomass were squeezed and centrifuge at 3,000 rpm for 15 min and decant the supernatant. Dilute the sludge with phosphate buffer 10 mM, pH 7.0, homogenize and centrifuge, then decant again the supernatant. This step has been repeated 5.0 times (Tadukaret et al., 2005)The biokinetic coefficients were determined using a closed 0.9 L batch respirometer, equipped with a recorder and a dissolved oxygen (DO) meter (YSI, model: 556 MPS). Constant temperature was maintained by circulating water through a water jacket enclosing the reactor vessel. Table 2 presents the operating conditions of the experiments. The S_0/X_0 ratio (initial substrate concentration/biomass concentration) that governs the quality of the batch respirometric tests was maintained in the range of 0.05-0.8.

The results of the respirometric experiments provided values of the oxygen uptake rates (OUR) that were used for calculating maximum specific growth rates (μ_{max}), substrate utilization rate (r_x), half-velocity constant (K_S) and sludge yield coefficient (Y) based on Monod kinetics by regression analysis (Wichitsathian, 2004).

Table 2. Operating conditions of the respirometric experiment of fungal and bacterial sludge

Operating conditions	Fungal culture	Bacterial culture
Initial pH	4.0±0.2	7.0±0.2
Temperature (°C)	30±0.5	30±0.5
X_0 (mgMLVSS/L)	400	400
Substrate concentration, S_0 (mgCOD/L)	20-320	20-320
S_0/X_0	0.05-0.80	0.01-0.80
Suppressing nitrification	none	70 mg N-ammonia/L (a)

Sources: a(Wichitsathian, 2004)

Analytical Methods

Influent and effluent samples were analyses according to the methods given in the Standard Methods for the examination of water and wastewater (APHA, 1998). The BOD values are calculated using negative pressure values from the OxiTop-C measuring head. Biodegradation is calculated from the measured BOD with inhibited nitrification and theoretical oxygen demands (ThOD) or total COD (TCOD) (Reuschenbach et al., 2003).

Sludge characteristics in terms of SS, VSS, specific oxygen uptake rate (SOUR), extracellular polymeric substances (EPS) were measured as well. Protein and carbohydrate, being the main components of EPS were analyzed using Lowry method (Lowry et al., 1951) and phenolic sulfuric acid method (Dubois et al., 1956) with Bovine Serum Albumin (BSA) and glucose as the standards, respectively.

Results and Discussion

Wastewater characterizations

The first stage of wastewater characterization consisted of quantifying the solid, organic and nitrogen contents as results are presented in Table 3. Biodegradation of the wastewater can be continuous characterized by BOD/COD ratios during the period of 20 days as shown in Figure 2. There were found to be TBOD/TCOD ratios were about 0.13, 0.17, 0.22 and 0.23 of BOD at 5 days, 10 days, 15 days and 20 days, respectively. And there were about 0.41, 0.47, 0.53 and 0.60, respectively of soluble fractions. This indicated the UASB effluent contain high amount of particulate COD and biological resistant organics. Literatures suggest the residual COD of anaerobic effluent may be comprised of residual non-degraded substrate, intermediate volatile fatty acids (VFA) and soluble microbial products (SMP). In well operated systems only a small fraction of the effluent COD is usually due to VFAs, while SMP account for 85-100% of residual COD. These SMP may not be readily biodegradable, or may even be refractory, and comprise a wide variety of organic compounds distributed across a broad spectrum of molecular weight (MW) (Jarusutthirak and Amy, 2007; Yun et al., 2007).

Moreover, the effluents of UASB reactor still contained a relatively high nitrogen nutrient contents based on the COD:N:P ratio seems to be very high for heterotroph microorganisms utilization. Further, for the UASB effluent characteristics, treatment efficiency of nitrification-denitrification is considered poor at BOD/TKN <2.5 or BOD/NH_3 <4 and COD/TKN <5 (Grady et al., 1999).

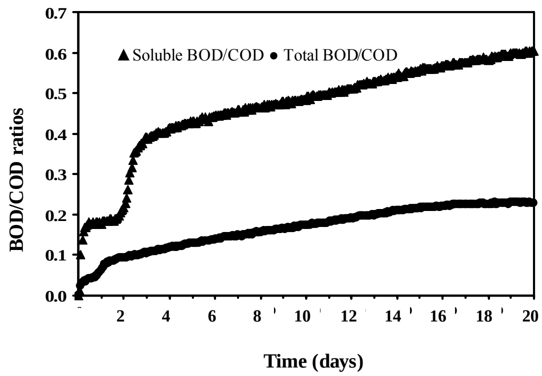

Figure 2. Plot of BOD/COD value and time of the UASB effluent

Organic removal

The result presented in Table 3 is average of seven months' continuous operation and monitoring. The overall $TBOD_5$ removal efficiency reached about 80% in both bacterial DHS and fungal DHS which was about 15-20% more than TCOD removal. The low BOD_5/COD ratios in both of BDHS and FDHS effluents as shown in Figure 3 indicate that contain high refractory substances which were either slowly biodegradable organic materials or non-biodegradable materials.

However, the results in Figure 3 and Table 4 show that most of COD fractions (TCOD and soluble COD (SCOD)) and BOD_5 fractions ($TBOD_5$ and soluble BOD_5 ($SBOD_5$)) were removed in the first segment of FDHS (about 70% of $TBOD_5$ removed). But, the highest removal was found to be in forth segment of BDHS system. These can be explained by the fact that the most coarse and soluble organic matter were adsorbed and degraded in the first segment of FDHS system. And these indicate that potential advantages of fungi over bacteria in term of rate of organic removed. High organic loading enable downsizing of reactors and better rate of acclimation allows for early start-up and rapid recovery from shock, both of that being desirable from the practical stand point. Moreover, several researches recommended fungi have a wide range of enzymes, and capable of metabolizing complex mixtures of organic compounds such as particulate matters and dead cells (Jin et al., 2002; Wichitsathian, 2004; Tripathi. 2007).

Table 3. Summarized overview of process performances of BDHS and FDHS systems

Parameters	UASB effluent	Effluent		Efficiency (%)	
		FDHS	BDHS	FDHS	BDHS
COD_{total} (mg/L)	1044 (450)	370 (92)	676 (414)	64.5	35.2
$COD_{soluble}$ (mg/L)	434 (140)	234 (31)	172 (86)	46	60.4
BOD_{total} (mg/L)	306 (79)	44 (16)	47 (15)	85.6	84.6
$BOD_{soluble}$ (mg/L)	216 (36)	41 (16)	44 (15)	81	79.6
Total N (mg-N/L)	176 (91)	170 (91)	55 (24)	-	68.7
NH_4-N (mg-N/L)	168 (91)	178 (91)	35 (20.5)	-	78.8
NO_3-N (mg-N/L)	<0.1	<0.1	8.4	-	-
NO_2-N (mg-N/L)	<0.1	<0.1	6.6	-	-
pH	6.6-6.7	3.8-4.2	6.8-7.2	-	-

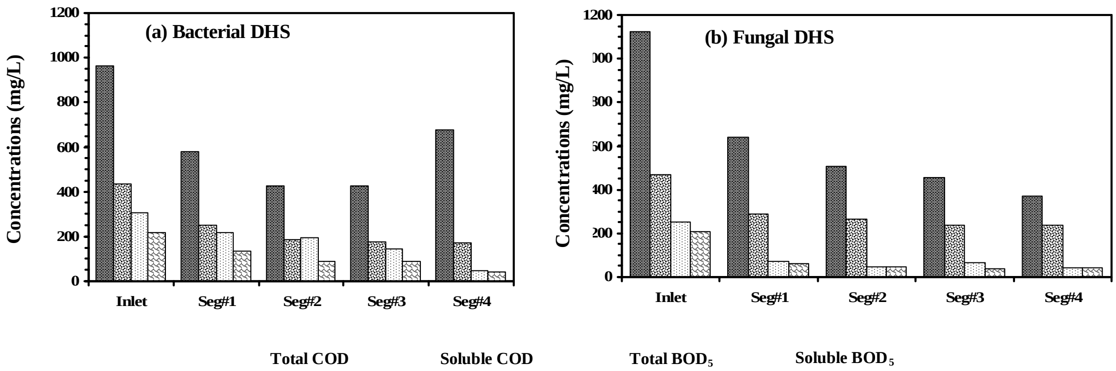

Figure 3. Organic removal profiles in DHS systems

Table 4. Parametric profiles of DHS systems

Parameters/ DHS profiles	Inlet/ Overall loading	BDHS effluents				FDHS effluents			
		Seg#1	Seg#2	Seg#3	Seg#4	Seg#1	Seg#2	Seg#3	Seg#4
Organic (kg-total BOD_5/m^3-d)	3.47-4.25	3.04	2.71	2.03	0.65	0.98	1.04	0.94	0.61
Organic (kg-soluble BOD_5/m^3-d)	2.71-2.98	1.59	1.01	0.97	0.51	0.78	0.56	0.46	0.51
Ammonia (kg-N/m^3-d)	0.28-0.90	1.82	1.19	0.79	0.48	2.4	2.71	2.53	2.47
Total BOD_5/COD	0.22-0.32	0.32	0.46	0.34	0.07	0.11	0.09	0.15	0.12
Soluble BOD_5/COD	0.44-0.50	0.54	0.49	0.49	0.25	0.21	0.18	0.16	0.17
Total BOD_5/N	1.13-1.54	1.45	1.73	1.58	0.75	0.26	0.17	0.28	0.19
DO (mg/L)	0	0	0.60	2.60	3.20	3.30	4.10	4.60	2.60

Nitrogen removal

Nitrogen is not removed in FDHS except for some proportion utilized for cell growth. However, it is reduced in BDHS via nitrification and denitrification. DHS system received total nitrogen loading 0.30-0.93 kg-N/m^3-d, which includes 0.28-0.90 kg-N/m^3-d of ammonia nitrogen loading. The concentrations of total nitrogen, ammonia, nitrite and nitrate in the effluent of bacterial DHS system amounted 62.0, 34.8, 6.6 and 8.4 mg/L, respectively. The system provided 46.9 mg-N/L of total nitrogen in the final effluent, corresponding to 68.7% removal efficiency.

Nitrification in bacterial DHS take place in second, third and forth segments which appearance of nitrogen oxide. Results shown nitrification was limited in the first segment of DHS system at high organic contents. This insufficient ammonia oxidizer population at high loading rate as complete with heterotrophs for space and oxygen as DO profiles in Table 4. In BDHS unit ammonia oxidation activities were comparable to or slightly higher than nitrite oxidation activities, accounting for the observation that nitrite were present in the effluents. Moreover, the nitrification behavior under high residual organic conditions was found in the bacterial

DHS. Chea et al. (2004) reported that the presence of organic matter in aerobic system, which promotes the growth of heterotrophs, inhibits ammonia oxidation. This result was very interesting as it ment the attached nitrifiers in the sponge media were quit resistant to orgaanic shocks. However, nitrifying bacteria are strict aerobes, they can only nitrify in presence of dissolved oxygen. At DO concentration <0.5 mg/L, little, if any, nitrification occurs (Iliuta and Larachi, 2005).

Consequently, the BOD5/TKN removed of bacterial DHS was about 100:50 that higher than nitrogen consumption which promote the growth of aerobic growth hetertrophs (BOD5/TKN ratio about 100:5). Probably, this nitrogen removal was caused by aerobic denitrification and/or denitrification occurring in the anoxic biomass (Machdar, 1997). Araki et al. (1999) suggested that the interval prevails, whereas up to the depth of approximately 0.75 cm from the surface of sponge, aerobic environment prevails. Nitrifiers in this region convert ammonia nitrogen in to oxide forms, which are then transferred to the anoxic zone where they are denitrified. In this way BDHS allows both nitrification and denitrification to take place within a single system. Furthermore, from the wastewater characteristics in each segments of BDHS reactors treatment efficiency of denitrification process is considered poor at BOD/TKN <2.5. Grady et al. (1999) suggested in order ensuring successful removal ammonia in the nitrification process, an external carbon source would be necessary. In one way which, past studies of reported high endogenous respiration of DHS sludge suggests that the sludge accumulation was in near balance with the degradation of sludge in reactor itself. And it was also being utilized as a carbon source during denitrification (Tandukar et al.,2006)

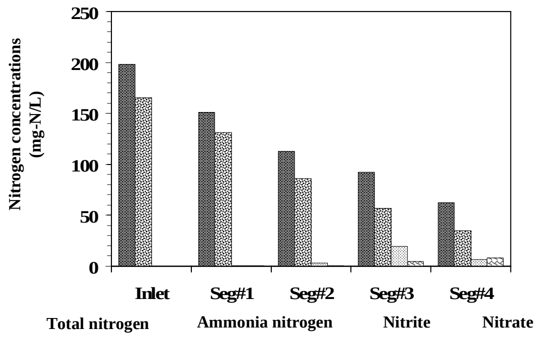

Figure 5. Nitrogen profiles of bacterial DHS system

Dissolved oxygen and its profiles

The fate of wastewater in the DHS systems were such that, it first flows into a sponge unit, comes out of it, comes in contact with air and then again penetrates the next sponge unit. As the wastewater comes in contact with air, the air gets diffused into it, thus increasing the DO concentrations. This repeated phenomena maintains DO in the wastewater almost to the level that whereby satisfy the demand of aerobes residing in FDHS. By virtue of this, there is no need for aeration in the system. However, oxygen limitation was found in the first and second segments of BDHS. This suggests that oxygen utilization of bacteria is higher than fungi.

Biokinetic Studies

Values of biokinetic coefficients of aerobic heterotrophs showed in Table 4 that substrate utilization rate (r_x) and maximum specific growth rate (μ_{max}) were higher of the first segment of fungal culture, which related the highest BOD content was removed in this segment. And there are decreased in the lower segments that cause by organic content was limited as the organic content profiles in Table 4. Different of bacterial cultures, the r_x and μ_{max} values were highest in second segment and forth segment may caused by the rate biodegradation organics by bacteria is less than fungi. And the most of suspended solids were adsorbed and inhibited organic removal at the first segment of FDHS. Additionally, estimation of parameter group ($\mu_{max}/Y.K_S$) is used as a measure for comparing the biodegradation kinetics, as suggested by Grady et al. (1999). Indicating that the highest biodegradation of organic by the first segment of fungal sludge. Evidence from the biokinetics experiment confirms a fungi-based on organic removal have the potential to higher than bacteria treating the UASB effluent of tapioca starch wastewater.

Moreover, It can be also observed the yield (Y) of the fungal culture is lower than that of the bacterial culture. This indicates that lower excess sludge would be produced from the fungal system compared to the bacterial system for the same substrate quantity.

Sludge characteristics

The characteristics of sponge biomass along the DHS system height were measured and illustrated in Table 5. Little differences in total solids and volatile solids were found to be accumulated in the various segments of the FDHS reactor. In steady state, the concentration of retained sludge in FDHS remained almost constant suggesting that the degradation of old biomass nearly balance the accumulation of the fresh one. Along with the MLVSS/MLSS ratios of the sludge was also measured; it was found that the bacterial sludge (0.75-0.78) had a lower degradability compared to that of the fungal sludge (0.86-0.95). Also the extracellular polymeric substances (EPS) of fungal and bacterial sludge were measured. Figure 6 summarized the variation in bound and soluble EPS of BDHS and FDHS sludge. The bound EPS corresponds to polymeric substance adhered together with each other and microorganisms. The soluble EPS indicate the microbial products which have been produced by the microorganisms and suspended in mixed liquor in a soluble form.

Several researches have been introduced a major disadvantage limiting the use of trickle-bed bioreactors for biological wastewater treatments is attributed to the progressive simultaneous biological clogging and physical plugging phenomena induced by the formation of an excessive amount of biomass and the retention of inert suspended fine particles and EPS (Geradi, 2002; Thullner et al., 2004). Thullner et. al. (2004) reported porous media samples showed that only 5% of the total organic carbon was present as bacterial biomass, whereas the remaining 95% were attributed to EPS. The total volume of the bacterial cells remained below 0.01% of the pore space even in the vicinity of the injection port. Therefore, the observed clogging effects were assumed to be mainly caused by EPS. That is one causes of filter clogging in BDHS system as found during the experimental run.

Moreover, experimental results of BDHS sludge show that the two highest of bound EPS concentrations were found to be in the first and third segments which two lowest biomass growth rate. This suggests highly bound EPS accumulated in the sludge may be effect on the bacteria consumption rate. But, based-on fungal sludge the bound EPS were higher accumulated in biofilms than bacterial sludge. The highest of bound EPS concentration and specific growth rate were found in first segment of FDHS. Evidence from experiment suggests that the relationship of EPS-production rate to substrate consumption rate seems depend on the kind of microorganisms involved and system conditions.

Table 5. Biokinetic coefficients of FDHS and BDHS

Biokinetic parameters	Segment 1		Segment 2		Segment 3		Segment 4	
	BDHS	FDHS	BDHS	FDHS	BDHS	FDHS	BDHS	FDHS
μ_{max} (d^{-1})	2.20	3.40	3.50	1.60	1.60	1.20	3.60	1.60
r_x (mgCOD/mgVSS-h)	0.11	0.21	0.16	0.10	0.09	0.09	0.15	0.10
Y (mgVSS/mgCOD)	0.84	0.63	0.86	0.63	0.85	0.63	0.86	0.63
μ_{max}/Y.K_S (L/mg-h)	1.99×10^{-3}	2.35×10^{-3}	2.09×10^{-3}	1.12×10^{-3}	1.57×10^{-3}	1.53×10^{-3}	1.74×10^{-3}	1.30×10^{-3}

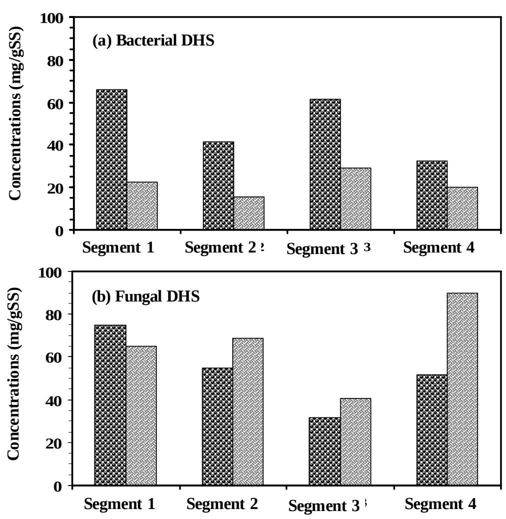

Figure 6. EPS concentrations of the DHS sludge

Furthermore, it can observe the sludge respiration of second, third and forth segments of DHS systems under low OLR or carbon source limits in influent wastewater. This suggests fungi have capable to consumed soluble EPS as the carbon source.

Influencing of EPS on effluent organic matter

As EPS are microbial produced, and are not active cells, they represent a diversion of electrons and carbon that could otherwise be invested in cells yield and growth rate, hence ignoring EPS formation could lead to a general overestimation of true cellular growth rate. Additionally, both soluble microbial products (SMP) and the soluble component of EPS contribute to the residual soluble COD and do, in may case, set the lower limit for the effluent quality. Macromolecular compounds such as protein and carbohydrate can comprise a significant portion of dissolved organic carbon in the DHS effluents (about 60-70% of COD effluent) as details in Table 5. Several researchers found that the chemical components of SMP consisted mainly of proteins, polysaccharides, and organic colloids, and that the SMP and EPS were identical (Jarusutthirak and Amy, 2007). The presence and characteristics of effluent organic matter and/or SMP in wastewater effluent are of great interest with respect to discharge quality and the efficiency of advanced treatment facilities.

Table 6. Soluble EPS concentrations in the DHS effluent

DHS Profiles	Soluble EPS (mg/L)		Soluble EPS/SCOD	
	BDHS	FDHS	BDHS	FDHS
Segment 1	132.8	199.8	0.56	0.61
Segment 2	126.0	187.3	0.63	0.66
Segment 3	123.0	175.5	0.67	0.68
Segment 4	121.6	170.2	0.70	0.69

Conclusions

The DHS system is simplified treatment system as it ensures low investment and running costs, operational simplicity. Cost is reduced as there is no need for aeration, cutting down the expenses associated with aerating device, their operation, maintenance or replacement. Various aspects of the BDHS and FDHS systems have been elaborated in this paper. Fungi based on organic reduction system have potential to overcome issues associated with conventional post treatment. However, nitrogen is not removed in FDHS system. Biological nitrogen nutrient removal system will be considered in the next stage of treatment processes. BDHS system was slowly degraded organic components from the UASB effluent but effective for nitrogen removal. Further, oxygen consumption of BDHS sludge was higher than fungal DHS system which the evidence of dissolved oxygen profiles that indicated partial oxygen limits of bacterial DHS system.

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