Integrated Anaerobic and Aerobic Treatment of Sewage

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Proefschrift
ter verkrijging van de graad van
doctor in de landbouw- en milieuwetenschappen,
op gezag van de rector magnificus,
dr. C. M. Karssen,
in het openbaar te verdedigen
op maandag 2 mei 1994
des namiddags te vier uur in de Aula
van de Landbouwuniversiteit te Wageningen

Front Page Design Dawei Zheng

CIP-DATA KONINKLIJKE BIBLIOTHEEK, DEN HAAG

Wang, Kaijun

Integrated anaerobic-aerobic treatment of sewage / Kaijun
Wang. - [S.l. : s.n.]
Thesis Wageningen. - With ref.
ISBN 90-5485-232-1
Subject headings: sewage treatment.

ABSTRACT

Kaijun Wang(1994) Integrated Anaerobic-Aerobic Treatment of Sewage, PhD thesis, Wageningen Agricultural University, Wageningen, The Netherlands

This thesis describes results of investigations dealing with sequential concept of anaerobic - aerobic treatment of municipal wastewater. The main purposes of the study were 1) to develop a proper anaerobic hydrolytic pretreatment unit, consisting of a Hydrolysis Upflow Sludge Bed (HUSB-) reactor and 2) to combine this system with proper aerobic post treatment processes, such as the activated sludge process or a stabilization pond system and with a combined anaerobic-aerobic process consisting of the Expanded Granular Sludge Bed (EGSB-) reactor and an upflow micro-aerophilic post-treatment process for complete sewage treatment.

The newly develop HUSB reactor serves for removing SS and accomplishing a certain sludge stabilization and raising the biodegradability of the remaining COD. The HUSB-system is operated at the similar retention time as the primary sedimentation tank, viz. HRT=2.5-3.0 hours. These features of the new system result in 1) release of the troublesome high SS-accumulation problems in the post treatment, such as stabilization ponds and UASB or EGSB systems, 2) a shorter overall retention time and lower energy requirements in the different types of aerobic post treatment processes, 3) an improved applicability for some refractory industrial wastewater treatment and 4) and certain extent of sludge stabilization in the HUSB reactor itself at higher temperature conditions or in a complementary sludge recuperation tank operated in parallel with the HUSB-reactor at low temperature conditions.

A new process concept, consisting of a sequential HUSB + the EGSB reactor, combined with sludge recuperation reactor, is presented in this study. The total process provides 71% COD and 83% SS removal efficiencies at T>15°C and 51% COD and 77% SS-removal at T=12°C conditions. A reasonable extent of sludge stabilization, i.e. over 50% hydrolysis of the removed SS can be obtained in the HUSB reactor at higher ambient temperatures, i.e. exceeding 19°C. The applicable hydraulic retention times are 3 hours and 2 hour for the HUSB reactor and the EGSB reactor respectively and two days for sludge recuperation tank. In the EGSB reactor up to 32 - 60% soluble COD removal efficiency can be achieved and the biogas production amounts to 23-70 NL/m^3(sewage) at ambient temperature (9-21°C), respectively. By applying a complementary treatment using an micro-aerophilic upflow reactor operated at HRT = 1 hr., an almost complete treatment can be achieved at 13°C conditions. Regarding the shorter hydraulic retention times required in this new concept compared to conventional systems, both for the wastewater treatment and sludge stabilization and its reasonable energy recovery, the new system looks very attractive as an alternative for treatment complex wastewaters like sewage.

The conventional aerobic activated sludge process and stabilization ponds both were investigated for post treatment at laboratory scale and pilot scale. The operational problems of these systems, such as occurrence of bulking sludge in the activated sludge process and the rather poor performance of stabilization ponds under cold weather condition were discussed and solutions for these problems are proposed. The experimental results obtained demonstrate the practical feasibility of the hydrolysis - aerobic treatment concept for municipal wastewater at ambient temperature. The final effluent quality is equal or better than that of the conventional activated sludge process, and the Chinese discharge standards can be satisfied satisfactorily.

Based on design and construction data of some full scale installations already implemented, the hydrolysis - aerobic biological treatment process (HUSB reactor + activated sludge post treatment) would reduce 37%, 40% and 38% respectively of the capital outlay, energy consumption and operational cost compared with the conventional activated sludge system. Secondary discharge standards also can be met with the HUSB + stabilization pond concept at HRT amounting to 65 % of that of primary settler + stabilization pond combination, while the capital expenditure and operational cost of the HUSB + stabilization pond system amount to 34% and 32% respectively of that of the conventional system. The developed anaerobic - micro-aerophilic process was found to represent an attractive and feasible alternative to the traditional activated sludge systems.

Key Words: activated sludge, ambient temperature, anaerobic and aerobic, bulking, domestic sewage, EGSB reactor, hydrolysis and acidification, hydrolysis(HUSB) reactor, micro-aerophilic, municipal wastewater, post treatment, sludge stabilization, stabilization pond, UASB reactor

ACKNOWLEDGEMENTS

It is a great pleasure that I have an opportunity to express my appreciation to everyone who has contributed to this thesis.

My sincere gratitude to G. Lettinga who served as my promoter, for giving me the opportunity to work at the Department and for his guidance and suggestion regarding the research. Without him support and careful reviewing the manuscripts, this dissertation would never have been complete.

I would like to thank my former supervisor Professor in China, Zheng Yuanjing who suggested me to choice this interesting and difficult research topic.

I would like to thank to Dr. G. Zeeman and Professor Qian Yi for careful reviewing the manuscripts and offering helpful advice and discussion.

I thank Robbert Kleerebezem and Paul Roeleveld for helping with Dutch translation of the summary.

I thank Andre van der Last for his helping to setup the experiments at the beginning and helpful discussion.

An important part of the research in this dissertation belonged to two separated projects which were funded by Beijing Municipal Government from 1985 to 1987 and the National EPA from 1986 to 1990. I would give my special thanks to above units supports. I am also grateful to Xu Xiaoming, Tao Tao, Liu Mei, Xu Dongli, Wang Junqi, Shi Jing, Zhang Shaofan who were in the research group.

I gratefully acknowledge the technic support of the following individuals: R.E. Roersma, H. Donker, and A. van Amersfoort. I also would like to express my appreciation to Johannes vander Laan, Martin de Wit for their assistance with the chromatography. Appreciation will extend to all the staff of experiment hall in Bennekom and all the members of our Department.

Last but not least, I thank my family and friends for their support. I specially thank my wife Wei Cui for her continuing support without her support this dissertation would never have been completed.

Content

CHAPTER 1

GENERAL INTRODUCTION

GENERAL INTRODUCTION

THE DEVELOPMENT OF ANAEROBIC PROCESS

The Earlier History

Anaerobic treatment is one of the oldest methods used to treat wastewater. The history of the anaerobic treatment is even longer than aerobic treatment. The first installation used to treat settled wastewater solids was known as the Mouras automatic scaverager. It was developed by Louis H. Mouras, a French engineer in about 1860 (Buswell, 1957). Donald Camenon was the first person recognizing that a combustible gas containing methane was produced when wastewater solids were liquified. He built the first septic tank for the city of Exeter, England, in 1895. He collected and used the gas for lighting in the vicinity of the plant (McCarty, 1981). At that time, the newly invented various types of septic tank systems greatly reduced the pollution problem. According to M. Allain Targe the invention of Mouras was "a complete solution of the problem which for centuries had been an insolent menace hurled in the face of all humanity".

The gas produced in the septic tank from removed solids often causes solids to float, the formation of scum blankets containing indigestible material, black colour and occasionally the entire contents of a tank to "turn over". It sometimes is considered that the processes of sedimentation and sludge digestion are incompatible. Harry W. Clark was the first who recognized the solution for this problem in 1899, when he stated that sludge should be permitted to ferment by itself in a separate tank (McCarty, 1985).

In 1904, the first dual-purpose tank incorporating sedimentation and sludge treatment was installed at Hampton England. It was known as the Travis hydrolytic tank in which the wastewater flows through a channel which is separated from the digestion chamber by baffles. The suspended materials are settled out here from the wastewater and allowed to pass into the separate "hydrolysing" chamber (McCarty, 1981). In 1904, a patent was issued to Dr. Karl Imhoff in Germany for a dual-purpose two-story tank, now commonly known as the Imhoff tank. In this system the wastewater is not allowed to pass through the "hydrolysing" chamber. The Imhoff tank greatly reduced the cost of sludge disposal. This system initiated the development of separate sedimentation tanks and sludge digestion tanks.

In fact, because of the still rather low industrial activities at that time, the pollution problem was caused mainly by the population itself. The most troublesome problem presumable was caused by the solids contained in the wastewater. Therefore the main first objective was focused on the wastewater solids treatment and disposal.

In 1891, W.D. Scott-Moncrieff constructed a tank in England with an empty space at the lower and a bed of stones in the upper part. The wastewater was introduced in the empty space and then passed upward through the stone layer. A distinct decrease in the volume of sludge to be handled was reported (McCarty, 1985). This presumably is the first application of an anaerobic filter. The big value of this work was that in fact it was the first anaerobic process treating the sewage itself, and not merely the sewage solids.

In 1910, Winslow and Phelps investigated, what they called "biolytic tank". Sewage was fed into the bottom of an inverted conical tank and flowed upward through a blanket of digesting sludge. The system was operated at a detention time of only 8.5 hours! They found that the removal of suspended solids was as good as obtained in a septic tank, but also that the liquefaction of deposited solids (72%) was distinctly better (Coulter, 1957).

In the period from 1910s to 1950s, the anaerobic digestion process was studied more in detail. Some major improvements were made in the design of the tanks and associated appurtenances. Also a great progress was made in the fundamental understanding and control of the process, the sizing of the tank, and the design and applications of equipment (Schmit and Sawyer, 1955, McCarty, 1981, 1985). At the same time, since separate digester systems had distinct advantages over a septic tank, especially with respect to the possibilities to heat and to mix, high rate sludge digestion systems were developed (McCarty, 1981). Unfortunately, this development resulted in the almost exclusive use of anaerobic digestion for the treatment sludge rather than for the treatment sewage itself, i.e. also in a blind alley for anaerobic sewage treatment process development. Therefore any progress in this respect was not made until the early 1950s.

Until that time, most of biochemical and microbiological knowledge of anaerobic digestion process, such as environmental condition affects, process control and the degradation of various types of substrate were obtained from investigation dealing with anaerobic digester systems (Torpey, 1955, Mc Bride and Wolfe, 1971, Henze and Harremoes, 1983).

Kinetic Considerations

At the beginning of this century, the treatment target changed from solids digestion systems to modern municipal wastewater treatment systems generally for combined industrial and domestic wastewater. Simple anaerobic treatment processes, such as various types of septic tanks, were not suitable to treat large quantities of municipal wastewater. For this purpose aerobic treatment processes were developed in England, viz. the trickling filter(1893) and the activated sludge process (1914). These systems since then have been widely applied (Metcaff and Eddy, Inc., 1981). The rate of decomposition of the organic pollutants in aerobic systems mainly depends on the concentration of the biomass retained in the reactor, the sufficient contact between influent organic material, the bacterial population in the reactor and the applied of oxygen (Jenkins and Garrison, 1968). A lot of knowledge, especially also in the field of process kinetic, was gained from investigations dealing with aerobic processes. This also promoted to some extent the development of specific anaerobic processes, like the contact process (Lawrence, 1971, Eckenfelder et al., 1960, 1966). The concept of increasing the solids retention time has been applied to guide to the process development and control (Young and McCarty, 1969, Lawrence. et al. 1969, 1970). From the concept of solids retention time, Eq. 1 and 2 can be obtained for anaerobic digester.

$$SRT\ (\theta_c) = \frac{\textit{active biomass in the process}}{\textit{total discharge active biomass}} \quad (1)$$

$$SRT = \frac{VX}{QX} = \frac{V}{Q} = HRT \quad (2)$$

where: X = biomass concentration; Q = flow rate; V = reactor volume;

In practice, especially for two stage processes, the ratio Θ_c/HRT in an anaerobic digester can be only increased to 1.5 to 2, due to withdrawal of digester supernatant. However, in these systems it is impossible to separate solids and hydraulic retention times, and therefore relatively big volumes are required. Sludge digestion systems therefore constituted more or less as an obstacle for the further application of the anaerobic process for dilute waste (water) treatment. In the concept of conventional anaerobic digesters including also high rate digesters and two phase digesters little if any improvements have been made since many years. It certainly is a field where very significant improvements can be -- and should be made!

Re-examination of the Anaerobic Process

Borrowing the concept of recycling biomass to maintain larger biomass concentrations and longer solid retention time from the aerobic activated sludge process, Schroepfer et al.(1955, 1959) developed the "Anaerobic Contact Process". According to Eq.1 the Eq.3 can be derived for this process. The solids retention time can be maintained independently of the wastewater flow by using biomass recycling! The big value of this contact process is that it marked the beginning of the development and application of a wide range of anaerobic treatment processes for dilute industrial wastewaters.

$$SRT = \frac{QX + RQX_r}{Q_w X_r + QX_e} \qquad (3)$$

where: X_r, X_e = are effluent sludge returned sludge concentration, respectively, g/L; Q_w, Q_r = are discharge flow rate and flow rate, m^3/d.

Up to 69-78% COD removal efficiencies were obtained at HRT from 12 to 15 hours at a temperature of 22°C, using the anaerobic contact process for a medium strength urban waste water resembling sewage (Simpson, 1971). In these investigations it was noted that digester overloading occurred at retention time below 12 hours. Therefore, in fact the contact process cannot really be designated as high-rate system, because the applicable organic space loads do not exceed 4 to 5 kgCOD/m^3.d. (Lettinga and van Haandel, 1993).

A modified type of contact process similar to the biolytic tank, was investigated by Coulter in an attempt to develop a low-cost and low-maintenance system for individual houses and small communities (Coulter et al., 1957). It consisted of an upflow anaerobic contact tank followed by a rock-fill filter. In the laboratory study two 8 litre reactors were operated in series at temperatures of 25 and 4°C. The assessed BOD removal efficiency amounted to 82% and the effluent value to 10-35 mg/L at 25°C. The BOD removal efficiency still amounted to 67% at temperature as low as 4°C. The suspended solid removal was exceptionally high i.e. exceeding 95%, both for cold and warm conditions. The effluent from the laboratory experiment was clear and surprisingly free of odour. The results of the scaled-up pilot plant were slightly lower than of lab. experiments i.e. BOD and SS removal efficiencies 65% and 95% respectively. It was also found that up to 84% of the removed suspended solid disappeared, presumable due to liquefaction and gasification (Coulter et al., 1957, Ettinger et al., 1957). The research of Coulter comprises the most comprehensive and successful anaerobic sewage treatment conducted at that time. Particularly striking was the high efficiency of processes was found at cold temperatures (4°C) at laboratory experiment.

The good performance of the system of Coulter was confirmed in the studies of Fall et al. and Pretorius in 1961 and 1971 (Fall and Kraus, 1961 and Pretorius, 1971). Fall and Kraus used a 360 m^3 large pilot plant (W*L*H=6*12*5) using upflow contact reactor. They used an aerobic trickling filter and ponds for post treatment units instead of the anaerobic filter as was studied by Coulter et al.(Fall and Kraus, 1961) The applied HRT of the anaerobic contact tank amounted to 13.4 hours and the average treatment efficiencies for BOD and SS achieved were 34 and 77% respectively during the whole experimental period. Looking to the data in more detail it appears that the BOD removal efficiency amounted to 42% at temperature 8-16°C, while only 14% BOD reduction was found at a temperature 18-22°C. The cause of the very poor BOD removal efficiency during the higher temperature period is the occurrence of liquefaction and acid fermentation of accumulated biodegradable solids. The VFA concentration in sludge samples raised from 500mg/L at the top of the tank to a maximum of 5,300 mg/L at the bottom. Up to 62.5% of the removed volatile suspended solids was liquified, which is about the same generally achieved in an anaerobic digester for primary sludge. During the experimental period no sludge flotation and no scum layer formation occurred, while the amount of gasification remained small and some sulphide was produced. After aerobic post treatment the effluent was satisfactory for discharge to surface water.

A similar two stage reactor set-up, but operated at total HRT of 24 hours for each reactor and at a temperature 20°C was used by Pretorius (1971). It was found that up to 90 per cent COD could be removed, leaving 30 mgSS/L and 110 mgCOD/L in the effluent. Of the removed SS in the first reactor, 35-40% was hydrolysed. Apparently the first reactor is more or less responsible for solid entrapment and hydrolysis, while the bio-physical filter part for the gasification.

The anaerobic filter was investigated in more detail as separate anaerobic system first by Young and McCarty (1969). The anaerobic filter in fact is the first real high-rate process (10-15 kgCOD/m^3.d) demonstrating the tremendous potentials of the anaerobic concept (Lettinga and van Haandel, 1993). The experiments conducted by Raman et al. (1972, 1978), who used a laboratory and a pilot plant scale AF systems clearly reveal that upflow anaerobic filter can successfully be used as a simple secondary treatment device for septic tank effluent and for settled domestic sewage as well.

Summary

The development of the anaerobic wastewater treatment process at earlier years and results obtained are summarized in Table 1. It can be seen that some of the adopted anaerobic processes are not really essentially different from those being considered now as high rate anaerobic processes, such as the upflow sludge blanket reactor and anaerobic filter (Winslow and Phelps, 1910, Coulter et al., 1957) and hybrid system (Scott-Moncrieff, 1891).

The two types of reactors introduced by Coulter et al. can be considered as both the forerunner of the "Anaerobic Filter" and "Upflow anaerobic sludge blanket" reactor developed by Young and McCarty (1969) and by Prevoris (1971) and Lettinga(1972), respectively. However, from part of the earlier experimental data, the relevant conclusions can be drawn only now on the basis of developed better insights in the anaerobic digestion process.

Table 1 The earlier development and experiments using anaerobic sewage treatment

Process	Volume m^3	Temp. °C	HRT h.	Influent(mg/L) COD(BOD)	SS	Removal Efficiency COD(BOD)%	SS%	Sludge digestion	Reference
Experimental Results									
CT + AF	10L	20-25	36	(180)	--	86	95	84%	Coulter et al., 1957
CT + AF	4.5+1.6	?	24	(250)	300	(65)	90		
CT	360	8-22	13.4-22.4	528(212)	368	46(34)	79	63%(VSS)	Fall and Kraus, 1961
CT	1.7	22-27	12-15	1207-1284	500	69-78(78-91)	62		Simpson, 1971
CT + AF	8+8L	20	24	500	252	78	88	35-40	Pretorius, 1971
AF	9L	25-33	5	250-572	68-203	71	86		Raman et al., 1972
AF	3.6	31	6.4*	--	59-336	(80)	89		

*: During 9 hours day time

MODERN HIGH RATE ANAEROBIC PROCESSES

Basic Principles of the Anaerobic Digestion Process

Anaerobic degradation of organic matter, including suspended organic matters can be considered as a three step process accomplished by a large consortium of microorganisms. In general the following three processes are distinguished:

--- The first step involves primarily an extracelluar enzymatic reaction. Many microorganisms produce extracelluar enzymes, suited for transformation (hydrolysis) of higher molecular mass compounds such as lipids, proteins and carbohydrates into compounds suitable for use as a source of energy and cell carbon.

--- The second step (acidogenesis) involves the bacterial conversion of the compounds resulting from the first step into lower molecular intermediate compounds, such as volatile fatty acids, alcohols, hydrogen, etc..

--- The third step (acetogenesis + methanogenesis) involves the bacterial conversion of the intermediate compounds into methane and carbon dioxide. Methane is produced mainly via acetic acid + hydrogen and carbon dioxide. Methanogenesis proceeds relatively slowly and generally is the rate-limiting step in anaerobic degradation.

Gujer and Zehnder (1982) presented a more detailed diagram consisting of 6 steps, in which some of the three main steps above are subdivided for anaerobic degradation of particulate biopolymers such as protein, carbohydrates and lipids(fats, greases) or of the further conversion of higher VFA into acetic acid (Figure 1).

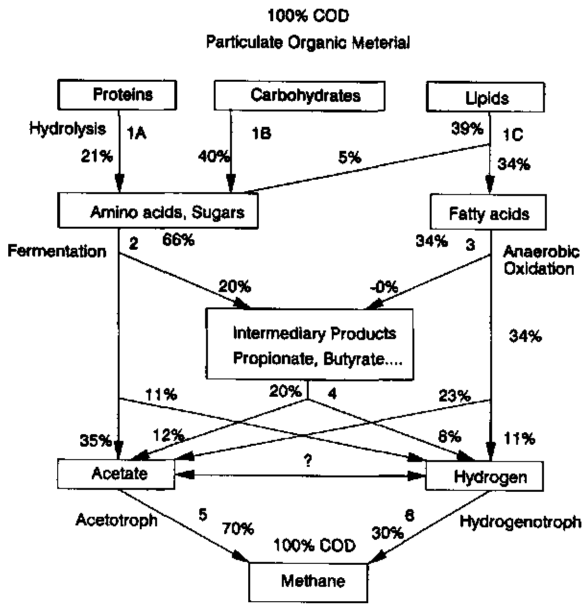

Figure 1 Anaerobic digestion of complex substrate(after Gujer and Zehnder, 1982)

Development and Applications of Anaerobic Wastewater Treatment

During the last decade, anaerobic reactors were evolved from the knowledge gained by fundamental studies on bacterial kinetics. A new generation of advanced reactors has been developed capable of retaining a high concentration of active biomass and consequently capable of treating waste waters at high loading rates. Several systems were developed like the anaerobic filter (AF) (Young and McCarty, 1969), the upflow anaerobic sludge blanket (UASB) reactor (Lettinga, 1979), anaerobic fluidized bed(AFB) reactor(Jeris, 1982), anaerobic attached film expanded bed (AAFEB) reactor (Jewell, 1979, 1981, Switzenbaum, 1980, 1983) baffled reactor (Bachmann et al., 1985) and anaerobic rotating biological contactor (Friedman, 1980). Each of these systems provides a long solids retention time for achieving a sufficient system efficiency and stability, and a short hydraulic retention time for system economy. Although, developed primarily for the anaerobic treatment of warm, soluble waste waters e.g. from food-processing industry, most of these systems in principle also can be used for the anaerobic treatment of cold, dilute domestic sewage. Table 2 lists most recent practice in anaerobic treatment of domestic sewage.

Various Anaerobic Processes

Like aerobic treatment systems, there are two kinds of common processes for anaerobic treatment, i.e suspended growth and attached growth systems. The most common anaerobic attached-growth treatment processes are the anaerobic filter and fluidized or expanded bed processes and for suspended growth system the most common process is the UASB system used for the treatment of domestic wastewater.

Anaerobic Filter The anaerobic filter consists of a reactor vessel filled with some proper type of solid media. The wastewater flows upward through the packed bed, containing the media on which anaerobic bacteria grow and are retained. Because the bacteria are retained on the media and not washed out with the effluent, mean cell residence times(Θ_c) can be obtained up to 100 days. High values of Θ_c can be achieved at short hydraulic retention times, consequently the anaerobic filter can be used in principle for the treatment of low strength wastewater at ambient temperature.

Fluidized bed and expanded bed reactor In the fluidized or expanded bed process, the wastewater to be treated is pumped upward through a bed consisting of an appropriate carrier medium(e.g. sand, coal etc.) on which a biological growth has developed. Effluent can be recycled to dilute the incoming wastewater and to provide an adequate flow to maintain the bed in a fluidized or expanded state. Biomass concentrations exceeding 15,000 - 40,000mg/L have been reported. The main difference between the fluidized bed and the expanded bed reactor is that the applied upflow velocity in latter system is too low to achieve complete fluidization of the carriers. Sofar there are no full scale installations for sewage using the fluidized or expanded bed reactors.

UASB reactor In the upflow anaerobic sludge blanket (UASB) process the wastewater is introduced at the bottom of the reactor, and it then flows upward through a blanket of active anaerobic sludge. Treatment occurs as a result of a proper contact of the wastewater with the active sludge. The gases produced in the sludge blanket become partially entrapped in the sludge. The free gas bubbles and particles with the attached gas tend to rise to the top of the reactor. Particles buoying to the surface strike the bottom of the degassing baffles on their way upwards, which may cause attached gas bubbles to be released. The degassed sludge particles then drop back to the top of the sludge blanket. The gas released from the sludge is captured in the gas collection dome located at top of the reactor. Liquid containing some residual solids and biological granules passes into the settling chamber, where part of the residual solids are separated from the liquid, and occasionally fall back through the baffle system to the top of the sludge blanket. The UASB presently process is the most widely utilized high rate anaerobic sewage treatment system. Several full scale and pilot installations are in operation in different countries and various are under construction (see Table 2).

Comparison of Different Processes

Although the AF process is a relatively low rate anaerobic process compared with other high rate processes, such as the AAFEB, AFEB, UASB or EGSB process, it still is an attractive system for small scale anaerobic applications, especially also for on-site domestic treatment. The AF process is technically relatively simple and easy in operation (Raman et al., 1972 and 1978, Zhang et al., 1991, Inamori, 1983, Watanabe et al., 1993). The AAFEB or AFBR process very likely is a rather promising process for domestic wastewater treatment. However, sofar only results of a 7.25 cubic meter pilot plant have been reported. Up to 80% BOD and SS removal efficiencies were achieved at HRT=5 hour and 18°C (Jewell, 1985). The main reason for its slow implementation can be found in the difficulties involved in its control, and the relatively much higher investment and running cost. The required hydraulic retention time for UASB systems is very similar to that applied in expanded bed reactors at the same or higher removal efficiency (van Haandel and Lettinga, 1993). However, the expanded bed reactor suffers from the disadvantage that it requires more pumping and support medium material for bacterial attachment. This leads to significantly higher operational and investment costs (Speece, 1983). Therefore the UASB looks the more attractive system.

Table 2 Recently practice of anaerobic sewage treatment using various kind of reactors

Process	Volume m^3	Temp. °C	Pre-Treatment	HRT h.	Influent COD(BOD)	Influent SS	Removal Efficiency COD(BOD)%	Removal Efficiency SS%	Reference
AF	9.7	10-25	r.s	9.5-36	(60-212)	63-213	(43-63)	70-83	Genung et al., 1979
AF	16L	20	r.s	24	288	--	73	79	Kobayashi et al., 1983
AF	10L	--	r.s	17	690-890	130-304	34-40	59-64	Wheatley, 1983
AF	3.0	11-25	s.w	20-33	(176-221)	--	(70)	--	Inamori et al., 1983
AF	102	10-25	r.s	12-18	132-390	62-114	28-64(57)	80	Genung et al., 1985
AF	16	22-24	r.s	8.6-37	(150-222)	200-250	(81-87)	91-97	Yamamoto Y., 1983
ABR	360L	24	r.s	5.5	762	488	77*	--	Garuti et al., 1992
ARBC	30L	16	r.s	24	435	61	60	66	Noyola et al., 1988
		29		24	328	72	66	86	
AAFEB	10L	20	p.s	2	215	--	77	--	Jewell et al., 1981
		20		8	170-320	--	68-92	--	
AAFEB	---	37		2.4-38	473	--	18-52	--	Rockey et al., 1982
AAFEB	20L	17-25	s.w	6.4	267	--	65(47)	50	Switzenbaum et al., 1986
		22-29		1.7-5.0	134	--	37		
AFBR	9.5L	20	p.s	1.2-9.8	---	---	(70-80)	--	Yodo et al., 1985
AFBR	330l	20	p.s	3-24	73-411	99-189	24-63	34-81	Brown et al., 1985
	3L	20	p.s	12-24	491-554	--	65-71	--	
AFBR	550L	22-27	p.s	0.85-2.1	(107-124)	--	(55-78)	--	Jeris et al., 1985
AFBR	0.54L	3-10	r.s	2.8	475(325)	190	76(85)	--	Sanz et al., 1990
		10		1.7	760(480)	285	70(80)	--	
	1.44L	15		2.2	760(480)	285	76(84)	--	
UASB	120L	20	p.s	18	550	--	55-75	--	
UASB	120L	8-20	p.s	12	300	--	67	--	Lettinga, 1981
UASB	120L	20	r.s	8	500	--	75*	--	Grin, 1983
		11-12		8	400	--	30-50	--	
UASB	0.4L	35	s.w	11-12	630-886	--	56-65	--	Forster et al, 1983
UASB	12.4L	20	p.s	3.2-12	--	--	(50-80)	--	Fernandes, et al, 1985
UASB	3.7	22-28	r.s	10	559(231)	204	55	--	Nucci et al., 1985
UASB	120L	12-18	p.s	7-12	500-700	--	40-60(50-70)	--	de Man et al., 1986
		7-8		9-14	500-700		45-65	--	
UASB	160L	20	r.s	6	467	--	50(54)	--	Derycke et al., 1986
UASB	120L	20	r.s	4	424(195)	188	60(69)	69	Vieira et al., 1986
UASB	110L	12-18		18	465	154	65	73	Monroy et al., 1988
UASB	106L	35	r.s	4.7	300	--	65(72)	61	Vieira et al., 1988
UASB	36L	11-23	r.s	1.5-3.5	438(161)	308	50(35)	91	Wang et al., 1988
UASB	120L	19-28	r.s	4	627(357)	376	74(78)	72	Barbosa et al., 1989
UASB	160L	20		6	1076	--	64	88	Mergaert, 1990
UASB	1.15	8-29	r.s	3-4	306	179	42-57	65-73	Zhou et al., 1991
EGSB	120L	>13	p.s	1-2	391	--	16-34	--	van de Last et al., 1992
HUSB	270L	12	r.s	2.1	367	--	34	--	Yu et al., 1992
Pilot-scale									
UASB	20	10-15	r.s	13-14	--	--	16-48	64-78	de Man et al, 1988b
UASB	64	24-26	r.s	4-6	267	215	65(80)	70-85	Schellinkhout et al., 1985
UASB	120	---	r.s	4.7-9	265-316	123-170	50-70	56-79	Vieira, 1988
HUSB	170	13-22	r.s	2.5	493(170)	277	44(32)	84	Wang et al., 1987
UASB	240	14-23	r.s	12-42	205-324	--	31-56	--	Collivignarelli et al.,1990
Treatment plants									
UASB	1200	20-30	r.s	6	563(214)	418	74(75)	75	Draaijer et al.,1991
UASB	4800	20-30	r.s	6	D**				
HUSB	3200	13-20	r.s	2.5	460	220	35	75	Wang, 1991
HUSB	2200	14-23	r.s	3.0	800	--	63	--	
HUSB	2000	7-20	r.s	3.0	336(157)	78	52(48)	89	
UASB	6600	25	r.s	5.2	380(160)	240	60-80(60-80)	--	Schellinkhout et al., 1991
UASB	1560	---	r.s	--	--	D			Vieira, 1988
UASB	1600	18-25	r.s	6.4	497(275)	220	D		Nucci et al., 1985
UASB	686	20-25	r.s	4.5	455(257)	174	11-60	27-58	

where: *: based on filtered effluent COD and raw influent; ABR: anaerobic baffled reactor;
ARBC: anaerobic rotating biological contactor; EGSB: expanded granular sludge bed reactor;
HUSB: hydrolysis upflow sludge bed reactor
**: D: designed or basic design; p.s:pre-settled sewage, s.w: synthetic wastewater; r.w: raw sewage

At present, the upflow anaerobic sludge bed reactor (UASB) is by far the most widely applied anaerobic domestic wastewater treatment system. It can be used both for (very) small scale and for very large scale applications as well, i.e. for on-site or off-site application (Lettinga, 1991a, Alaerts, 1993, Geary, 1993, Draaijer et al., 1991, Schellinkhout, et al., 1985, 1991). This is not merely because of its process simplicity and plain mode of construction, operation and scaling-up, but particularly also because of the relatively high loading potentials of the system and the relatively high extent of sludge stabilization that can be accomplished with this process. In recent years a number full scale sewage treatment plants based on the conventional UASB type have been installed in tropical countries and even in some countries with more moderate climate (Table 2).

Two Phase and Two Stage Anaerobic Digestion

In conventional one phase-processes all separate anaerobic conversions processes proceed in one reactor. Because the optimal growth conditions, such as pH and temperature etc. differ for each group of bacteria, for the purpose of process optimization the conditions for the slowest conversion step in the whole sequence, normally those for methanogenic and acetogenic bacteria, should be maintained optimal(Breure, 1992). In case the different process steps would be conducted in different reactors, optimal conditions for each separate group of bacteria can be maintained, which then theoretically offers the possibility to optimize the whole process. A physical separation of hydrolysis and acidification is hardly to achieve, because once the substrate has been liquified, the acidogenic organisms immediately will develop. Also a separation between acetogens and methanogens is difficult to accomplish, because the acetogens cannot grow at higher hydrogen concentrations. Consequently so far only a two-phase concept has been proposed by separating acidogens and the acetogens.

Although some researchers and/or engineers suggest that the phase separation would be profitable, because of a higher overall process stability and the significantly higher applicable space loading rates in the second reactor (Breure, 1986), those presumed benefits of two phase process presumably will be offset by the higher costs of the required additional sedimentation tank and the dosing of chemicals (for neutralization). Moreover, there also exists clear evidence that a completely acidification can slow down the granulation process and exert a serious negative effect on the operation of the second reactor (Lettinga and Hulshoff Pol, 1991b).

Nevertheless, for domestic sewage the use of a two or three step (compartmented reactor!) is profitable, because, firstly, the removal efficiency of a compartmented UASB reactor exceeds at least 10% that of a non-compartmented reactor at the same total HRT (de Man, 1990), and secondly for treating raw domestic sewage a first reactor (compartment) serves the entrapment of suspended solids, and the hydrolysis and partial acidification of this matter. The first reactor should be operated at relatively low upflow velocity. Although the achieved pre-acidification of sewage in the first reactor is beneficial, it is not necessary to accomplish a complete hydrolysis and acidification of the removed SS in this "hydrolysis step". Because the hydrolysis of particulate matter normally proceeds slowly, especially at low temperature, generally a much larger reactor would be required to achieve complete liquefaction and acidification, particularly at lower ambient temperatures. From an economical point of view, it is beneficial to achieve a maximum SS removal in the first reactor compartment, and to combine it with a separated digester operated at higher temperature for sludge hydrolysis and stabilization.

The hydrolysed part of absorbed particles, together with the returned hydrolysed SS from the parallel reactor and the soluble fraction already present in the influent can be treated in the second step, which could consist of an EGSB system. De Man (1990) and van der Last et al. (1991, 1992) found that a granular sludge UASB reactor operated in an expanded mode, e.g. by applying effluent recirculation or by using a tall reactor configuration, comprises an efficient in the removing soluble organic material. The relatively high efficiency achieved can be attributed to the excellent contact between wastewater and the sludge. This EGSB-system particularly is attractive at low temperature, when the mixing of influent and sludge generally is insufficient due to the low biogas production.

SOME DIFFICULTIES IN ANAEROBIC SEWAGE TREATMENT

The performance of an anaerobic treatment system depends strongly on the environmental conditions and the characteristics of the wastewater. Several environmental factors either enhancing or inhibiting parameters i.e. temperature, pH (alkalinity), nutrients and toxicants affect anaerobic digestion, such as growth rate, decay rate, gas production, substrate utilization etc. (Henze and Harremoes, 1983, Speece, 1983).

Table 3 Effect of Sewage Characteristics on Anaerobic Treatment

Characteristic	Possible Impact
Low temperature	- low methanogenic specific activity - slow hydrolysis
low substrate concentration	- low growth potential - low bacteria concentration - poor methane capture - low removal efficiency
high SS (or VSS) fraction	- slow hydrolysis and mass transfer kinetics - reduction of the specific methanogenic activity - reduction SRT - detrimental for bacterial granulation - risk for scum layers formation
high sulphate concentration	- inhibition of methanogenesis process - lower methane production - potential inorganic oxygen demand and post treatment is needed - odour or corrosion problem
fluctuation in the flow and in concentration	- poor effluent quality

For domestic sewage Rittmann (1985) listed six key characteristics that might influence anaerobic sewage treatment performance. Others also pointed out similar factors concerning the applicability of anaerobic treatment for sewage (McCarty, 1985, Jewell 1985, and Lettinga, 1992b). Since 1980, extensive investigations have been conducted using various types of reactors under variety of conditions. This resulted in a better understanding of the anaerobic process, might lead to new conclusions for process control and system development. The various factors are summarized in Table 3 and will be discussed below.

The Effect of Temperature

The temperature comprises one of the major factors influencing the process. Heating a large of quantity of domestic sewage obviously is not feasible from an economic point of view. Anaerobic sewage treatment processes should be applied at ambient temperature conditions. For tropical areas the sewage temperature is 20-35°C, while it is 10-20°C for moderate areas. The results obtained sofar, clearly demonstrate that the process offers big prospects for topical areas (Schellinkhout et al., 1985, 1991, Grin, 1985 and Vieira, 1986, 1988, van Haandel and Lettinga, 1993). But, results obtained by several investigators indicate that the use of anaerobic treatment even should not be excluded at temperatures as low as 4-10°C (Coulter et al., 1957, Grin et al., 1983, van der Last et al., 1992, Lettinga et al., 1983, Jewell et al., 1981, de Man et al., 1988a, Inamori et al., 1983, Sanz and Fdz-Polanco, 1990).

The BOD removal efficiency can still come up to values as high as 67%, even at a temperature as low as 4°C (Coulter, 1957). Sanz and Fdz-Polanco (1990) reported about the performance of an AFBR reactor, operated at a temperature as low as 10°C and HRT=1.5h. for a long operation period (235 days). The total COD removal exceeded 70% and BOD removal about 80%. Their results also indicated that a sudden and short temperature decrease did not have a significant impact on the performance of the reactors. On the other hand, it also was clear that operating the system at lower temperature (10°C) will result in a significant the amount of higher accumulating of solids. Similar observation were made in the anaerobic filter system investigated by Genung (1985).

In experiments conducted by Inamori (1983) with an anaerobic filter system, using synthetic wastewater (200mg/L BOD) at 5, 10, 20 and 30°C temperature and at HRT=7.5, 15 and 30 h., the following relationship between HRT and applied temperature was found.

$$\ln\frac{HRT_2}{HRT_1} = 11810.5\left(\frac{1}{T_2} - \frac{1}{T_1}\right) - \ln\frac{X_2}{X_1} \qquad (4)$$

Where: T = temperature; i = subscript 1,2 indicated temperature 1 and 2

For the AF process it was found that at temperatures it needs a longer HRT below 10°C. At T=30°C the required retention time is only 3% of that at 5°C. For experiments conducted with UASB-system it was found that compared to 35°C the methanogenic activity at 20°C is 35%, at 10°C 10% and at 5°C 3% (van der Last and Lettinga, 1993). From Eq. 4 it can be deduced that the temperature effect can be compensation by an increased sludge concentration.

Switzenbaum and Jewell (1980) reported that an increased biomass hold-up is found in an AAFEB reactor at lower temperature, which to their view it is important for compensation of low temperature. The AAFEB was shown to be effective at HRT≈5 hrs. for wastes containing only 200mg/L of COD a temperature as lower at 10°C (Jewell et al., 1981, Switzenbaum and Jewell, 1983).

A COD_{total}-reduction of 50-60% can be achieved at temperatures ranging from 15-19°C at HRT=8 hour for the conventional UASB process using flocculent sludge. At temperatures below 10°C overloading of the system occurs under these conditions (Lettinga et al., 1981, Lettinga et al., 1983, Grin et al., 1983). For granular sludge UASB system a COD removal efficiency of 45-75% can be achieved at HRT=8 hours and temperatures exceeding 12°C. However, at temperatures drops below 10°C, the HRT has to be prolonged to 9-14 hours to achieve the same removal efficiency (Lettinga et al, 1983, de Man, 1986). With a modified version of UASB reactor, i.e. the EGSB reactor, anaerobic treatment looks feasible for settled sewage at temperatures below 13°C. In an EGSB system a better contact between wastewater and sludge is attained, resulting in a better removal efficiency compared to an UASB reactor. At HRT 1-2 hours it is possible to obtain 45% COD_{total} removal efficiency, which corresponds up to 80% removal efficiency for the biodegradable COD-fraction. Below 13°C a higher HRT is needed to avoid a poor degradation of intermediates, viz 2.5-3 hours (van der Last and Lettinga, 1992). The big potentials of the EGSB system recently has been demonstrated particularly for a VFA-substrate. In that case at space loads up to 12 kgCOD/m^3.d at 10-13°C over 90% removal efficiency of the VFA (500-1,000 mg/L) at less than 2 hours retention time can be accomplished (Rebac and Lettinga, 1993).

These results with synthetic substrates clearly illustrate the enormous potentials of anaerobic treatment at low temperatures. It is also clear that the rate-limiting step for anaerobic treatment in EGSB systems is not methanogenesis. In case the wastewater consists of a more complex substrate, such as lipids, long chain fatty acids, proteins, the rate limiting step in EGSB-reactors using a high grade granular sludge generally is hydrolysis. This certainly is true for temperature as below 15°C.

The Effect of Low Substrate Concentration

Reaction kinetic concepts, which allow rough estimation of S_{min} representing the minimum concentration of the substrate in the effluent that can be achieved in a system operated under steady state conditions, was proposed by McCarty(1985), Rittmann(1985) and Jewell (1985 and 1987). A limitation of anaerobic treatment might be that even for the degradation of very simple substrates, such as higher VFA and ethanol, several different species are involved. Each species has its own substrate and intermediate products. The total concentration that ultimately will appear in the effluent is equal to or exceeds the sum of the respective S_{min} values for the various types of substrates and intermediate products formed ($\Sigma S_{min(i)}$). This may lead to a rather large effluent concentration for unbalanced anaerobic treatment systems.

The S_{min} value found for acetate in methane fermentation were 48mg/L and 78mg/L at 25°C and 35°C, respectively according to McCarty (1985) These relatively high values will lead to rather low removal efficiencies for low strength acetate containing wastewater. However, according to Rittmann(1985) S_{min} as low as 3.7AC-BOD/L prevail at 35°C for acetate substrate. Based on this value it should be possibly that the effluent meets secondary treatment standards. And indeed, for the methanogenic substrates or like VFA or a simple soluble substrate, like ethanol, for low influent concentrations ranging from 200-700mg/L, an almost complete removal (over 95%) was achieved at 10 hours and 2 hours retention time using a conventional UASB reactor and an EGSB reactor at 30°C, respectively (Kleerebezem, 1991, Versteeg, 1992, Lettinga, 1992b, Kato et al., 1994). In ongoing research (Rebac and Lettinga, 1993) with EGSB systems it recent even was found that an almost complete COD removal can be achieved for a VFA substrate with COD 500-1,000 mg/L at temperature as low as 10°C and at HRT=2.0 hours!

Another serious potential limitation for the treatment of sewage is associated with the low cell yield of anaerobic bacteria. A low sludge yield can result in a system with a net zero or negative sludge production (Genung et al., 1980, 1985, Wang et al, 1987, Garuti et al., 1992). For a sewage with a BOD=250 mg/L, the maximum ultimate microbial yield is approximately 25mgVSS/L which generally is far lower than the effluent VSS. Fortunately, the newly developed processes like UASB and EGSB were found to retain the viable sludge sufficiently well for synthetic substrates and for sewage as well, even at low temperatures.

The Effect of High SS(or VSS) and Colloidal Fraction

Coarse and finely dispersed SS can affect the anaerobic treatment system quite adversely (Table 3). In case of raw municipal wastewater treatment it therefore might be beneficial to subject the sewage to primary sedimentation before feeding it to an anaerobic digester. Despite that the UASB reactors installed sofar, do not use primary treatment, contrary to systems like the anaerobic expanded or fluidized bed investigated at small pilot scale. Although according to Jewell et al.(1981, 1985), Yodo et al. (1985) and Genung et al. (1982, 1985) suspended solids can be effectively degraded in AAFEB, AFBR and AF systems, the accumulation of slowly biodegradable SS may exclude the application of these systems under conditions of low HRTs, particularly for treating raw sewage, but presumably also for settled sewage. The required volume of these reactors will become as big as that of a conventional UASB system. Therefore the control of the suspended solids becomes an important consideration.

Except removing the suspended solids in a pretreatment, possible problems with SS in anaerobic treatment systems can also solved in an other way. In the EGSB reactor superficial velocity exceeding 4-6 m/h are applied, dispersed matter present in the wastewater is completely washed through the reactor or it accumulates in the form of a flocculent sludge layer on top of the granular sludge bed, depending on the design of the system (van der Last and Lettinga, 1992, Lettinga and Hulshoff Pol, 1991b). In this way a heavy accumulation of inert sludge in the system can be avoided, e.g. by applying occasionally a short period (0.5-1.0 hours) of high superficial velocity in order to wash out the flocculent sludge from the system. Very similar observations were made in the operation of AAFEB reactor and AFBR reactors (Jewell, 1985, Sanz et al., 1990). In the upflow AF reactor, the excess sludge seems to accumulate at the bottom of the reactor, and therefore, it has been suggested to modify the design of the AF reactor by leaving an unpacked reactor beneath the packed section so that a sludge blanket can developed here (Genung, 1985). Such a hybrid-type reactor might become beneficial for controlling the sludge stabilization in AF reactor.

As raw sewage contains quite a big amount of suspended solids, some researchers recommend to subject the raw sewage to a pre-treatment process. The use of two anaerobic reactors, viz one for treating the dilute wastewater and the other for the separated suspended solids was suggested by McCarty (1985). In fact, various efforts along this line already have been made by several investigators for some years. Grethlein (1978) adopted a septic tank - membrane system for domestic sewage treatment. As a result of the increased concentration of microorganisms and substrate in the membrane reactor, the anaerobic digestion rate in this "high-tech" septic tank system could be enhanced by a factor of 3-4. The anaerobic membrane process was developed further by Kiriyama et al. (1991) at field scale. The process consisted of a pretreatment system for separating suspended solids, a hydrolysis reactor with membrane module for retaining the solids and a UASB reactor. The field test gave a 70-80% BOD removal efficiency and a gas conversion rate from 57% to 60% in terms of BOD at HRT=1.8 h. (working volume) and ambient temperature conditions. It was reported that 84.5% of charged volatile suspended solids was degraded, and accordingly the sludge production only amounted to 1/3 to 1/4 of that of traditional processes.

A process combining physic-chemical clarification using fine magnetite particles with anaerobic digestion was developed on bench scale for treating a concentrated type of sewage by Priestley and Woods (1987). This process produced an effluent with a BOD in the range of 25 - 50 mg/L. Quite different from above physio-chemical/anaerobic concept, is the "hydrolysis upflow sludge blanket" (HUSB) reactor introduced by Wang et al.(1987, 1989 and 1992b) The objective of this process is to remove SS, to stabilize the removed suspended solids and to raise the biodegradability and of domestic and various industrial wastewater as well. Over 80% suspended solids and 40% COD can be removed in the HUSB reactor at ambient temperature conditions (9-23°C).

The Effect of Sulphate Concentration

Sulphide production can cause several problems, such as inhibition of methanogens (though particularly at high S^{2-} concentration, competition between sulphate reducers and methanogens for substrate, odour and corrosion problems and a higher oxygen demand in the effluent etc. (Rinzema and Lettinga, 1988, Visser, 1992). Because, both the sulphate and substrate levels in domestic wastewater are relatively low, it is very unlikely that H_2S will reach the critical inhibition level for methanogenesis. A mathematical model derived by Rittmann (1985) indicates that methanogens and sulphate reduction bacteria can coexist. This indeed was confirmed in practice, sulphide and methane are both formed in an anaerobic reactor (Coulter, 1957, Yodo, 1985, Sanz 1990, Draaijer et al, 1991). The remaining problems of the presence of sulphide in the effluent are malodor nuisance, corrosion and a high oxygen demand (Fall and Kraus, 1961, McCarty, 1981).

Malodor problems with the anaerobic effluent can be easily avoided (Coulter, 1957, Fall and Kraus, 1960, Sanz, et al., 1990). The presence of oxygen results in the formation of small white granules containing a high percentage of sulphur(78%) (Sanz et al., 1990). These observations already were made by Coulter et al. (1957) and Brown et al. (1985). Thiothrix and thiobaccilli bacteria and some obligate aerobic organisms were found in the effluent (Sanz, 1990, Coulter, 1957, Fall and Kraus, 1961). These micro-organisms are able to use sulphide as an energy source and transform it into sulphur, which then is deposited as in granular form in the cell. The phenomenon of biological sulphide conversion into elementary sulphur has been extensively studied by Buisman et al. (1990a and b).

The sulphide formed by microorganisms may be present in the liquid in soluble or insoluble form, depending on the conditions. Heavy metals with exception of chromium, form insoluble sulphide salts and can thus be removed from the solution when sulphide is present in the system. This can be used both for heavy metal and sulphide removal. It has been observed that Zinc, Copper, Cadmium and Iron are removed over 70% and Manganese for 30% in an UASB reactor (Fernandes et al., 1985). A so-called UASB-Fe reactor in which iron pieces are used as packing media according to Zhou, et al. (1991) can be effective control the sulphide level below 0.1 mg/L.

A possible interesting development is the so-called Multi-stage Reversed-flow Bioreactor (MRB) which utilizes the symbiotic interaction between anaerobic bacteria (sulphate reducing bacteria) and microaerophilic bacteria (sulphur oxidizing bacteria) for auto-granulation. Organic substrate present in a sulphate containing wastewater will diffuse into the self-granulated sludge (SGS), where it is converted to organic acids by anaerobic bacteria, and then utilized by the SGS for sulphate reduction. Sulphide produced in this reaction diffuses through the SGS surface back into bulk of the liquid. Though the supply of oxygen is limited in the MRB reactor, there still exists a chance for the microorganisms present on the SGS surface to come into contact with the oxygen. Because the oxygen consumption rate of bacteria oxidizing sulphide is much higher than that of bacteria oxidizing organic substrate, most of the oxygen will be utilized by the sulphide oxidizing bacteria (Takahashi and Kyosai, 1988, 1991, Arora and Mino, 1992).

Effect of Fluctuations in the Flow and in Concentration and Other Factors

The big fluctuation both in flow rate and in concentration comprise important limiting factors regarding the efficiency of sewage treatment systems, although according to Jewell (1981) 20-fold changes in organic concentration and in flow rate have little effect on the AAFEB process. In 64 m^3 pilot plant experiments with the UASB reactor a better average daily treatment efficiency was obtained under conditions of low night-time (HRT=6 hours) and high day-time flow (HRT=2.2 hours) as compared to that obtained at a HRT of 6 hrs. (Schellinhout et al., 1985). Inamori et al. (1983) reported that an AF reactor operated steadily under conditions of 3-fold and 6-fold flow rate for a period lasting 3 hours.

Process failures of the anaerobic process are sometimes associated with product inhibition, due to the formation of high concentration VFA, hydrogen, sulphide or ammonia. However such problems hardly can occur in treating a very low strength wastewaters, like sewage, because the amount of intermediate products then can not reach inhibitory level (Mergaert, et al., 1993). Moreover, in case of sewage treatment always sufficient bicarbonate buffer capacity will be present in the water phase.

POST TREATMENT AFTER AN ANAEROBIC PROCESS

It is widely accepted that anaerobic treatment is a pretreatment process, because it is a mineralisation process which merely converts organic matter. Reduced inorganic compounds are present in the effluent, such as ammonia, sulphide and phosphate. Moreover, also pathogens generally are insufficiently removed. Therefore the anaerobic treatment has to be followed by adequate post treatment methods in order to meet standard set for discharge on surface water. Various aerobic treatment processes have been proposed for post treatment, such as the activated sludge process, contact oxidation process, aerobic fluidized bed systems, rotating biological contactor and stabilization ponds (Yoda, 1985, Wang et al., 1989, Xu et al., 1991a and b, Schellinkhout et al., 1991 and van Buuren, 1991a).

Aerobic and anaerobic biological treatment processes for domestic sewage treatment offer advantages and disadvantages relative to each other. McCarty (1964) listed the advantages of anaerobic treatment as 1) a high degree of waste stabilization, 2) a low production of waste sludge, 3) a low nutrient requirement, 4) no oxygen requirement, and 5) the production of methane gas as a useful end-product. In principle, by combining the two systems in a sequential set-up, they become mutually complementary in their advantages and offset in their disadvantages. So recently it was found that the anaerobic process can effectively degrade hazardous chemical compounds as halogenated compounds (McCarty, 1985, Tao et al., 1992). The combined anaerobic - aerobic process looks particularly beneficial for the treatment of domestic sewage and refractory industrial wastewaters (Wang, et al., 1989, Wen, et al. 1991).

According to Lettinga (1985) application of anaerobic pretreatment can relieve an overload aerobic treatment plant from bulking sludge problems occurring due to overload. Moreover various researchers demonstrated that such combined anaerobic aerobic concept can reduce quite substantially the total wastewater treatment plant investment costs, energy consumption and running costs (Wang et al., 1987, 1992 and Genung, 1982, 1985, Collivignarelli, 1991). Economic analyses indicated that the AnFLOW process (AF + aerobic post treatment) is attractive at small scale (19-27m^3/d) with respect to the costs of investment, operation and energy, while for large scale application (based on 3,800m^3/d) the operational costs and energy consumption still can be reduced with 16.5% and 41.3% respectively relative to that of the activated sludge process (Genung et al., 1979, 1982). However, the investment cost in latter case would increase 36.9% compared with the activated sludge process. Collivignarelli et al. (1990, 1991) have compared their developed anaerobic - aerobic process to the traditional activated sludge process. The excess sludge production of the new process is 60% less, while the energy for aeration and occupied plant area are 40% and 60% lower, respectively. The investment costs are almost the same. Therefore, according to Collivignarelli et al. the anaerobic - aerobic process represents an attractive and feasible alternative process.

The post treatment system to be used strongly depends on the characteristics of the anaerobic effluent, and on the effluent standards for discharge on surface water as well. In general three situations can be distinguished with respect to the required effluent quality i.e. i) for direct agricultural uses, such as irrigation, fishing culture etc.; ii) for discharge on surface water where strict effluent standards apply for the various characteristics, such as organic matter(BOD), SS, NH_4^+-N, NO_2^--N, NO^{2-}_3-N, phosphates and pathogens; iii) for situations where only organic matter, BOD(COD), SS and pathogens have to be eliminated. In order to satisfy the different standards, post treatment processes have been developed and /or investigated for nitrification, denitrification, sulphide oxidation, phosphate, SS and BOD, as well as pathogen removal (Bovendeur et al., 1985, Buisman, 1990a and b, Collivignarelli et al., 1990, Vieira, 1988, Xu et al., 1987, 1991a, Buuren, 1991b).

The characteristics of anaerobic effluent are quite different from those of the original sewage, such as the low BOD/COD ratio (poor biodegradability), the high content of H_2S, NH_4^+-N and sometimes a relatively high VFA content. According to Rittmann (1985) the bulking problem in the activated sludge process is associated with sulphide, and he therefore presumed that an aerobic post treatment following an anaerobic pretreatment would be accomplished with more difficulties compared to direct aerobic treatment. According to observations made by Wang (1992c and d) the bulking sludge problem is associated with high VFA concentrations (about 100mgAC/L) and high loading rates (0.65-0.85 kgBOD/m^3.d). De Man et al. (van Buuren, 1991a) encountered serious sludge bulking problems in a high rate activated sludge post treatment process. Due to heavy bulking of sludge the experiment had to be terminated at a loading rate of 0.3-0.6kgCOD/kg MLSS.d, HRT=2.5h.. In order to prevent bulking of the sludge it was recommended to apply a plug flow process to be operated at low loading rate (0.12kg COD/kg MLSS.d HRT=10.5 h.). According to observations made by Grin (1985) nitrification proceed in aerobic post-treatment slower than in a direct aerobic process, and sometimes even longer HRT may be required than in a conventional aerobic treatment system (van Buuren, 1991, Inamori et al., 1986).

Regarding these findings the well established conventional aerobic processes very likely are not the most suitable systems to achieve an optimum treatment concept, in which important targets are delegated to anaerobic pretreatment step. It is clear that the desired optimum process configuration is not yet available and it therefore is recommended put emphasis on research in that field. A lack of proper post treatment method would become a serious obstacle for the rapid implementation of anaerobic treatment. Nevertheless the present study will focus on anaerobic pretreatment, because tentatively the removal of organic matters is the main issue in the developing countries. In addition attention was paid to the possible use of a low cost post treatment process using a technically plain system.

SCOPE OF THIS THESIS

This thesis presents results of research activities conducted in China and the Netherlands, viz.: 1) investigations deal with combined hydrolysis - aerobic biological processes for municipal wastewater treatment, conducted in China from 1985 to 1989; and 2) investigations of the hydrolysis upflow sludge blanket (HUSB) reactor combined with an expanded granular sludge bed (EGSB) reactor process for sewage treatment, which were conducted in the Netherlands.

The main object of the investigations is to assess the feasibility of modified high rate UASB processes (either HUSB or EGSB) for the treatment of domestic sewage. This thesis consists of an introduction and six Chapters dealing with specific research issues, followed by a summary. The investigations deal with anaerobic pre-treatment system and combined anaerobic and aerobic treatment, using post-treatment methods such as the activated sludge process, stabilization pond and the micro-aerophilic process.

Chapter 2 deals with investigations concerning alterations of sewage characteristic taking place under anaerobic, micro-aerophilic and aerobic conditions. The hydrolysis reactor as anaerobic pre-treatment step aims at a reduction of influent suspended solids and to improve the biodegradability of the organic pollutants present in the sewage (Chapter 3), and the combined hydrolysis upflow sludge blanket (HUSB) reactor + expanded granular sludge blanket (EGSB) reactor configuration was investigated in order to assess the practical potentials of this concept (Chapter 4). The investigations dealing with the different post treatment processes for the effluent of various different anaerobic treatment systems are discussed in Chapter 3, 5 and 6, while the design aspects and implementation of the hydrolysis - aerobic treatment process are presented in Chapter 7.

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Chapter 2

ALTERATION IN SEWAGE CHARACTERISTICS UPON AGING

Accepted for publication in: Wat. Sci. Technol.

Alteration In Sewage Characteristics Upon Aging

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ABSTRACT

To improve the understanding of the nature of sewage changes upon aging of the sewage during transport or storage, simulating experiments were conducted using batch reactors at a number of well defined conditions, i.e. anaerobic, aerobic and micro-aerophilic at 10°C, 20°C and 30°C. Important characteristics of studied the sewage were the visual appearance, various chemical properties and odour. The effect of the temperature on degradation processes and reaction kinetics of different polluting fractions of the sewage is studied under mainly micro-aerophilic and anaerobic conditions.

The results of non inoculated batch simulating experiments reveal that micro-aerophilic conditions are suitable for both pre- and post-treatment of sewage, while anaerobic conditions suffice for pre-treatment. At low temperatures, anaerobic conditions mainly serve for pre-acidification. On the basis of the obtained results, we recommend to put emphasis on further research dealing with on-site and on-line treatment systems, combined with the central wastewater treatment plant. Such processes look attractive in improving the organic removal efficiency and in reduction of operation cost and capital outlay of wastewater treatment systems.

KEY WORDS

Anaerobic, Micro-aerophilic, On-line, Self-purification, Sewage Aging, Sewer

INTRODUCTION

The performance of a biological treatment process is affected by characteristics of the wastewater and consequently the changes that occur in the transport lines or upon storage (viz. aging) will influence the performance of the treatment system. In the past few decades, little attention has been paid to the biological changes and biodegradation processes taking place in a sewer. Sludge bulking problems in activated sludge systems are often associated with sewage septicity (Tomlinson, 1982, Chambers, 1982). As far as the storage of sewage is concerned, it was found that even at 4 °C the COD will drop significantly after being stored for periods exceeding one day (Painter, 1971). Moreover results obtained in some other investigations reveal that hydrogen sulphide formation may occur in sewers, causing control problems (Attal et al., 1992). Jensen and Jacobsen (1991) investigated the re-aeration in a gravity sewer systems.

Among the various reactions that may prevail in a sewer, such as physical, chemical, electro-chemical and microbial, the microbial reactions certainly are the most important. Since the rate of supply of oxygen is low in a sewer, e.g. due to the poor ventilation conditions, micro-aerophilic and anaerobic conditions will mostly prevail. In the present investigation, natural degradation processes under micro-aerophilic and anaerobic conditions will be studied in simulation experiments under a number of well defined conditions. These studies should lead to an improved insight in the processes occurring in a transport line. Moreover, these insights can also be used to improve the performance of wastewater treatment processes and probably may lead to the selection of more proper treatment processes.

MATERIALS AND METHODS

The investigations were conducted in four 6 litre batch reactors, consisting of double wall vessels with a working volume of 5 litres. All experiments were carried out in this equipment. The "Anaerobic System" (AnS system), consists of continuously mechanically stirred air tight reactor (at 60 rpm). The "Micro-aerophilic system" (MA system) uses an open vessel exposed to air in which also a gentle stirring rate of 60 rpm is applied. In the third reactor, the Aerobic System (AeS system), used as the control system, aeration is applied by compressed air (Figure 1). These three sets of reactors were all connected to a thermostat for controlling the temperature during the experiments at 10, 20 and 30°C, respectively. In order to get rid of the oxygen present in the raw wastewater and/or head space above the liquid, the anaerobic reactor was flushed with nitrogen gas 5 minutes, before starting the experiment. Upon sampling, nitrogen gas was introduced into this reactor to keep the system under pressure, viz. to prevent air coming into the system.

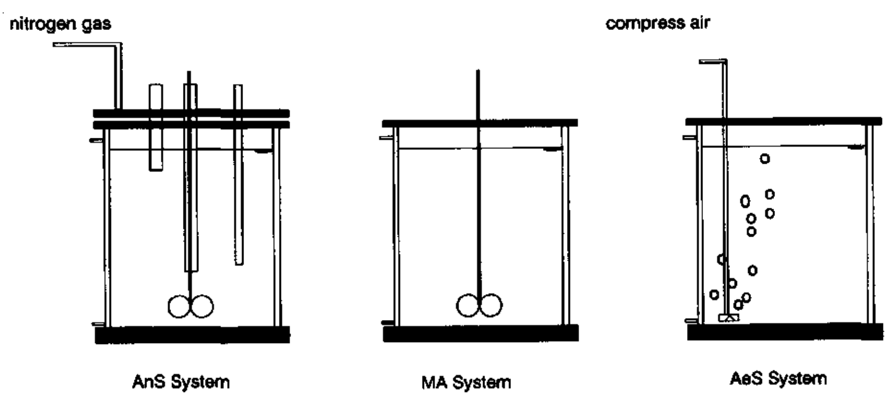

Figure 1 The schematic diagram of the simulation experimental equipment (reactors connected with thermostat to control the temperature)

The experimental conditions and characteristics of the sewage, as well as some experimental results are presented in Table 1 and Table 2. The experimental periods lasted from a few days up to 30 days; the experiments were terminated once the CODm concentration remained relatively constant. At that time the soluble biodegradable materials were almost completely consumed.

Table 1 The arrangements used in the sewage aging experiments

Experimental conditions	Raw sewage 10°C	20°C	30°C
Aerobic	---	---	BAeR3
Anaerobic	#AAnR1	AAnR2	BAnR3
Micro-aerophilic	AMR1	AMR2	BMR3

#AnAR1: stands for type of wastewater A(A or B) of AnS system(An, Ae or MA system) using Raw sewage at 10°C temperature condition(10, 20 or 30°C) and so on.

Table 2 The characteristics of the influent and effluent after batch treatment

Item Exp.No.	CODt mg/L	CODc mg/L	CODm mg/L	CODs mg/L	BOD_5 mg/L	SS mg/L	VFA mg/L	OD	pH	Ev mv	PO_4-P mg/L	T-P mg/L	NH_4-N mg/L	T-N mg/L	reaction days
Raw Wastewater															
A	563	180	231	152	282	210	41	0.118	7.8	43	5.1	6.9	36.6	51.7	
B	859	204	256	399	524	290	64	0.138	7.4	76	9.6	13.1	72.4	96.1	
Results of Experiments															
AAnR1	223	59	76	98	147	97	0	0.044	7.4	-270	4.7	7.0	40.5	49.5	29
AAnR2	193	32	65	96	124	118	0	0.040	6.9	-310	4.1	6.6	40.4	49.4	29
BAnR3	291	23	77	145	131	--	0	0.037	7.2	---	6.9	10.0	92.8	100.9	19
AMR1	180	23	75	82	107	93	6.8	0.018	7.8	61	4.6	7.0	25.3	44.2	29
AMR2	158	3	75	80	81	100	5.4	0.016	7.3	84	3.3	7.3	3.8	44.1	29
BMR3	236	9	84	107	105	--	---	0.018	7.9	---	4.3	10.7	18.9	39.8	19
BAeR3	156	13	119	127	39	--	---	0.011	8.4	---	4.7	9.9	0	16.7	14

ANALYTICAL METHODS

Raw samples were used for determination of Suspended Solids (SS), BOD_5, Kjeldahl nitrogen and total phosphorus and filtered samples (Schuell 595½ paper with pore size 4.4 μm) for Volatile Fatty Acids (VFA), ammonia, nitrite, nitrate-nitrogen and orth-phosphate (DNSM, 1969). For COD analyses the micro-COD method is used (Knechtel, 1978). The raw sample was used for total CODt, a 0.45μm membrane filtered sample for the soluble COD(CODm) and a 4.4 μm folded paper filtered (Sechuell 595½) sample for CODf, respectively. The colloidal CODc and suspended CODs were calculated by the differences between CODt and CODf, CODf and CODm, respectively. The oxygen concentration was measured with an O_2-sensor. The pH was measured with a pH-electrode, and the redox potential measured with platinum indicator and calomel reference electrode. The OD value was analyzed using the same spectrometer for the COD measurement at 600 nm.

THE CHANGES OF APPEARANCE OF SEWAGE

Offensive Smell and Removal of Relevant Compounds After the first two hours reaction time, the unpleasant smell of raw sewage disappeared both in the AeS and the MA system, but in the AnS system it prevailed during the whole experimental period. In the anaerobic reactor a white film probably consisting of sulphur, attached to the wall of the reactor was observed a few hours after the start of the experiment. The ammonia-nitrogen concentration in the AnS and MA systems reached a maximum value after 10 days, while the NH_4^+-N concentration in the AeS decreased already after two days of treatment (Figure 2).

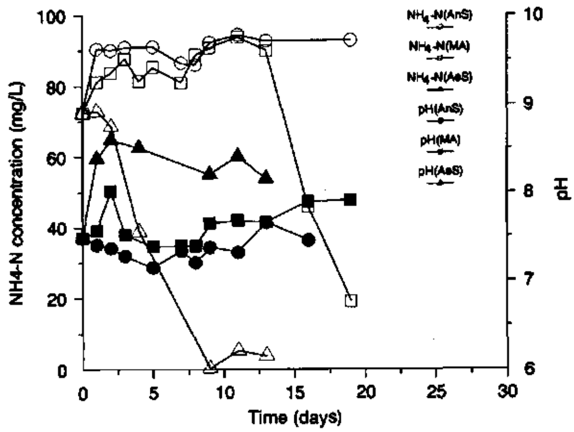

Figure 2 The course of ammonia and pH changes in the three different systems (T=30°C)

Although the sulphide and sulphate concentration were not measured during the experiment, from above observation it can be inferred that the offensive smell presumable originates from sulphate reduction products. Compared with the oxidation rate of ammonia, the sulphide oxidation reaction proceeds very rapidly. Under aerobic conditions (AeS system) and under micro-aerophilic conditions (MA system) as well, the prevalence of hydrogen sulphide formation is sufficiently or even completely suppressed.

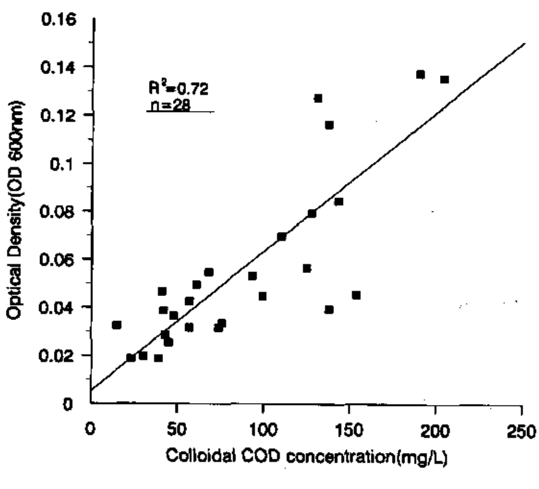

Figure 3 The relationship between the OD(600nm) value and the Colloidal CODc

Floc Formation and Changes in the Characteristics of the Sewage Bacterial flocs (like activated sludge flocs) become visible in the MA and AnS systems after 24 hours. The flocs in the MA system exerted good settling properties and the supernatant was significantly much clearer compared to the original wastewater or to the effluent of the AnS system. However, in the AnS system throughout the whole experimental period fine flocs and the turbid supernatant persisted. Some larger flocs were formed in the AnS system later on. At the termination of the experiment, the Optical Density (OD at 600 nm) of the paper filtered samples of the AnS system still was about 0.04, while the OD values obtained in AeS and MA systems amounted to 0.01-0.02. The OD value of a paper filtered sample showed a good linear relation with the colloidal concentration (CODc) (Figure 3). This was not the case for the soluble CODm and paper filtered CODf concentration (not shown). The final concentration of colloidal COD amounted to values 20 -50 mg/L in the AnS system. Apparently, the anaerobic flocs exert rather poor sorption and coagulation properties, due to which the removal of colloidal COD is poor and consequently hardly any change in the turbid appearance of the effluent occurs. This obviously represents a major drawback of anaerobic treatment compared to aerobic processes.

THE DEGRADATION OF ORGANIC POLLUTANTS

Comparison of the Different Systems The measured values for CODt and CODf at 30°C (experiments BAnR3, BMR3 and BAeR3), and some additional experimental results are presented in Figure 4 and Table 2 respectively. As expected, the CODt in the aerobic control system drops down rapidly within the first two days. The CODf removal efficiency at 24 and 48 hours are 57% and 81% respectively. The MA system also appears to be very effective, i.e. the CODf removal efficiency after 72 hours reaction time is already 55.5%. The average COD removal capacity amounted to 130 mgCOD/L.d within the first three days and in the following 16 days the average removal capacity is 16.6 mgCOD/L.d.

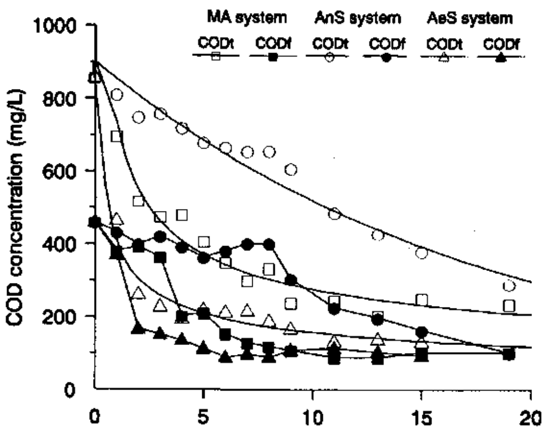

Figure 4 The course of CODt and CODf change and kinetic regression at the AeS, AnS and MS systems(30°C temperature)

In the AnS system the average COD removal capacity is only 38 mg/L.d during the first three days, much lower than that in the MA system. The intermediates from the anaerobic degradation partially remain in the liquid, such as VFA etc., at least during the initial phase of the process when insufficient methanogens are available.

The VFA formation increases rapidly at all temperatures in the AnS system(Figure 5). It should be noted that the VFA concentration is expressed in a dimensionless value (relative to the initial COD) in order to facilitate a comparison between the results of the different experiments. VFA degradation started after 4 to 8 days and the completion of the conversion of the VFA depends on the temperature, i.e. lasting from 10 to 20 days. The hydrolysis and acidification reactions were not rate-limiting in these experiments. The methane formation reaction proceeds rather slowly especially at low temperatures (T=10°C). Unlike the anaerobic system, the VFA removal in the MA system is less temperature dependent. VFA are completely degraded within 3-5 days at all temperatures investigated.

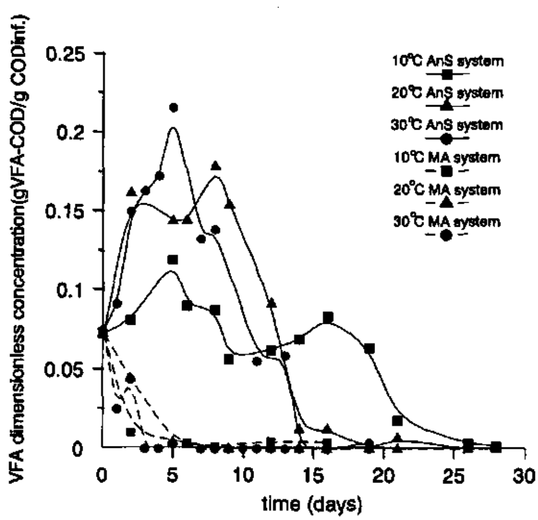

Figure 5 The course of VFA concentration at AnS and MS systems(T = 10, 20 and 30°C)

<u>Assessment of the reaction rates</u> The three experiments conducted at the different temperatures allow the estimation of reaction rates. The COD degradation process can be formulated according to a first-order kinetics, expressed as:

$$COD(t) = COD_0 e^{-kt} \qquad (1)$$

The estimated first-order reaction constants for 30°C are based on CODf and CODt values respectively. The reaction constant(k) based on CODt fits well first-order reaction kinetics, but it is much less, in the case for the CODf (Table 3). The values of the reaction rate constant decreases in the sequence is $k_{Ae} > k_m > k_{An}$. The value of temperature constant(Θ) of the Hoff-Arrhenius relation $k_t = k_{20}\Theta^{(T-20)}$, found on the basis of the results in Figure 4 and 6, amounts to 1.026 for all the three systems.

Table 3 Assessed values of reaction rates constants based on CODt and CODf (30°C)

Item	CODt		CODf	
System	-k	*R^2	-k	R^2
AeS system(k_{Ae})	0.094	0.69	0.084	0.53
AnS system(k_{An})	0.055	0.78	0.094	0.75
MA system(k_m)	0.073	0.96	0.076	0.88

*: data n=14

<u>**The removal of different COD ingredients**</u> Figure 6(a-f) presents the course of the concentration of different COD ingredients in relation to the reaction time for the experiments conducted under the various conditions (AnS, MA and AeS) and temperatures (10, 20 and 30°C). It can be seen that the soluble COD fraction, especially the VFA - part of it, degrades rapidly under micro-aerophilic condition, significantly faster compared to anaerobic condition. It is clear that the VFA oxidation process proceeds very rapidly even under oxygen limited conditions. At the termination of the experiments, also the colloidal CODc is almost completely removed from the liquid phase in the MA system.

Unlike the MA system, the main processes in the AnS system during the first two of weeks are hydrolysis and acidification. It can clearly be seen that during the hydrolysis phase, the CODc and CODm concentration in AnS system vary considerably An obvious CODt removal in fact becomes manifest 4 - 8 days after the start or even more depending on the applied temperatures.

Compared with the fate of the removed colloidal and soluble COD, the fate of the removed suspended COD is less clear. Relatively big variations in CODs occurred in the system. Both in the MA and the AnS systems, the suspended COD first increases, very likely as a result of adsorption of the soluble substrate and flocculation or sorption of colloidal matter by bacteria, but gradually it starts decreasing due to hydrolysis and oxidation reactions, and finally it apparently reaches a constant level of 80-100 mgCODs/L. From the data in Table 2, it can be calculated that more than 50% SS is degraded both in the MA and in the AnS systems.

Comparing the results of AnS and MA systems, there exists a significant difference for the removal of colloidal COD. In the AnS system, a relatively big amount of CODc remains in the bulk of the liquid, causing also a lower CODt removal efficiency. On the contrary, little if any CODc remains in the MA system. These observations indicate that some of the colloidal matters indeed is quite difficult to eliminate under anaerobic conditions, and as a consequence the turbidity and the COD concentration of effluent of the anaerobic process remains at a higher level.

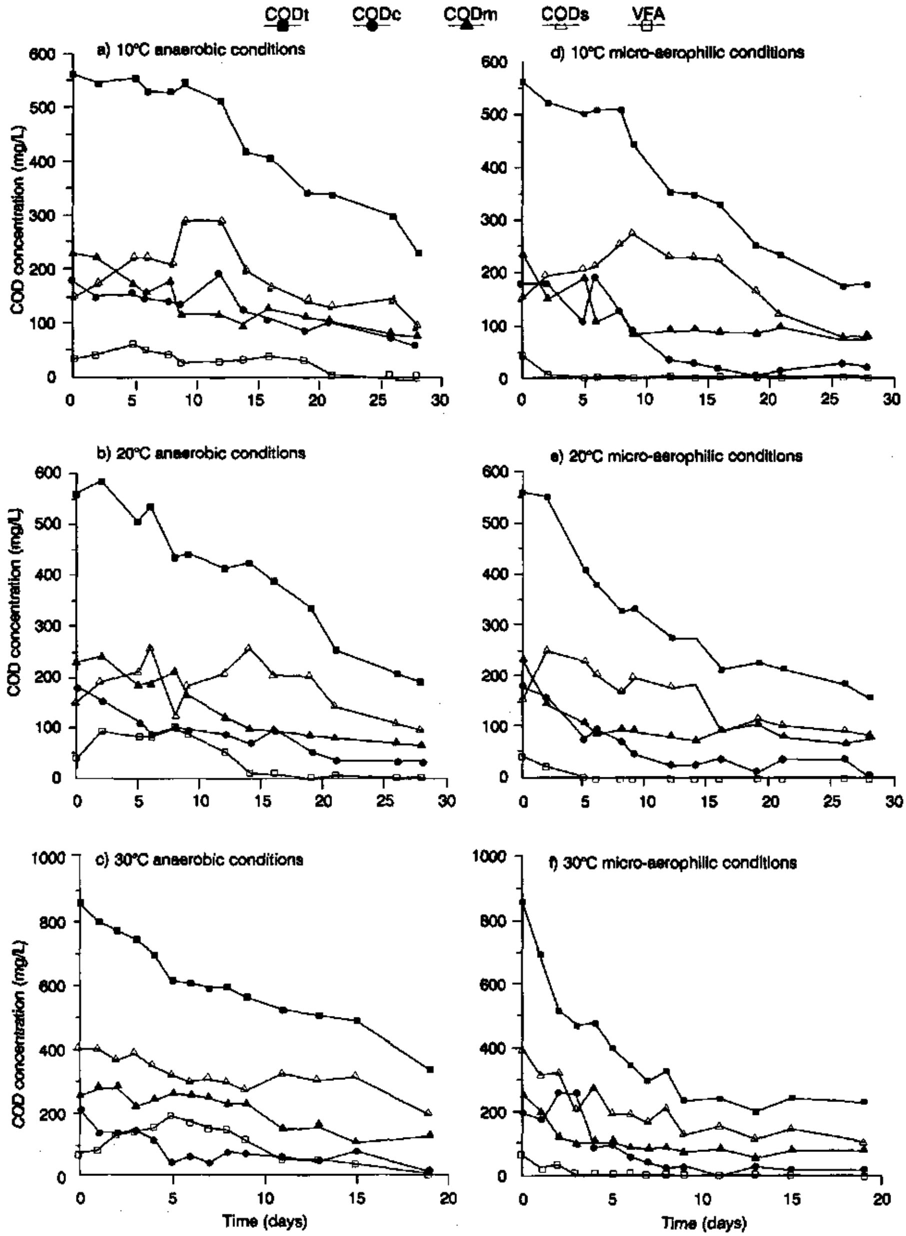

Figure 6 The course of the degradation of the different COD-fractions and the VFA ingredient in AnS and MS systems under 10, 20 and 30°C temperature conditions

<u>**Temperature effects**</u> Parallel experiments (AAnR1, AAnR2, AMR1 and AMR2) were conducted with sewage sample A and two other parallel experiments (BAnR3 and BMR3) with sewage sample B at different temperatures (10°C, 20°C and 30°C). COD removal efficiencies obtained for the different systems are summarized in Table 4. It looks like there exists a threshold temperature, below which the efficiencies for all COD ingredients except CODm of anaerobic reactions drop dramatically. Although the results of the present investigations don't allow the assessment of the exact threshold value, it is clear that at 10°C, the efficiencies for the degradation of suspended matter is relatively low.

Table 4 The temperature influence on the COD-treatment efficiency for the various distinguished COD-fractions in the different systems after a reaction time of 30 days (10 and 20 °C) and 20 days (30°C)

Items	MA system			AnS system		
	10°C	20°C	30°C	10°C	20°C	30°C
CODt(%)	68	72	73	59	66	66
CODc(%)	87	98	96	67	82	89
CODm(%)	67	68	68	67	72	70
CODs(%)	46	47	73	37	38	64

The results obtained clearly indicate that a MA system indeed is a suitable post treatment step, it also could be profitably used as pre-treatment method, because it is a quite effective for all kind of COD ingredients at all temperatures applied. Otherwise the anaerobic system contrarily is also a suitable pretreatment process, although at low temperatures and relative short exposure times it only serves as pre-acidification process. However such a pre-acidification step could be beneficial for the subsequent aerobic treatment, particularly in the case phosphate has to be removed. On the other hand it is clear that a complete anaerobic conversion of biodegradable pollutants into methane, offers the big advantage of energy production, and therefore could represent an core-step in a really sustainable technology for environmental protection.

POTENTIAL APPLICATION

From the results obtained it is evident that the various COD fractions of sewage can change significantly in concentration and character in the transport line and during storage, both under micro-aerophilic and anaerobic conditions. These changes can affect the operation and the efficiency of applied wastewater treatment processes. Nevertheless, the degradation processes as measured in the present laboratory studies proceed too slowly to explain the septic characteristics of sewage as frequently observed in bigger cities, especially in summer time. The relatively low rate of degradation found in the laboratory experiments can be attributed to the lack of a sufficient amount of active biomass. In sewer systems biomass can be present either in the form of an attached biofilm or of deposits (or both). The degradation rate can be accelerated by adding seed material (Attal, 1992). The purpose of the present research was to gain a better understanding of the processes proceeding in sewer systems. Based on the results obtained it is possible to make an extrapolation to the situation prevailing in practical sewer systems. The results of this study, combined with existing knowledge on conventional aerobic wastewater treatment processes and anaerobic treatment systems, could be used to develop a new concept of on-site and on-line sewage treatment. Figure 7 presents a schematic diagram of such a concept, which consists of sewage collection, transportation and treatment systems. Such a system might become profitable for combined on-site treatment/decentralized treatment concept, which involves sewage collection, transport and treatment systems, for small communities (U1), such as on-site treatment facilities or individual industrial factories (U3) or even larger wastewater treatment plants (U2). In the on site treatment facilities complete treatment is recommended, because in this way, the treated sewage could be reused in situ and the total quantity of discharged sewage can also be minimized. Depending on the design concepts, a gravity sewer system can exert a certain self-cleaning capacity. A sewer system serves at least two important potential functions which sofar have not been explored sufficiently, i.e. wastewater storage and SS carrying capacity. It might become attractive to utilize these functions better in order to achieve a certain on-line sewage pretreatment, in case in a town/city a sewer system has been installed or when plans exist to install a sewer system, despite the high investment costs and other drawbacks.However, existing sewer systems generally only can play partly or temporarily a pre-treatment role, resulting in a septic type of sewage, as frequently observed in most of the big cities during summer time.

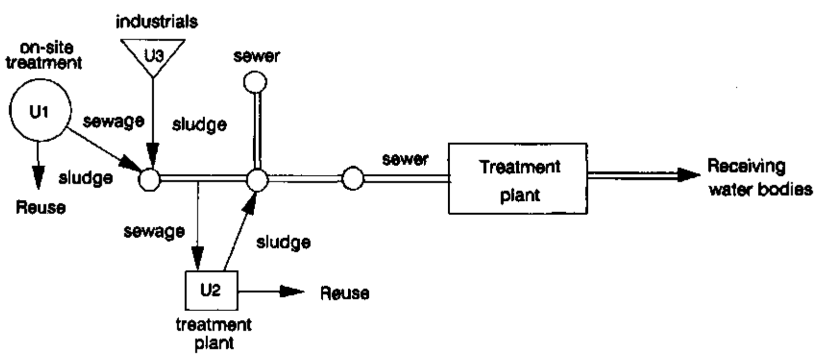

Figure 7 The schematic diagram of the whole sewage collection and treatment system

For existing sewer systems the problem in accomplishing a substantial COD-reduction here, is a lack of sufficient amount bacteria or a low activity of the bacteria (e.g. due to low ambient temperature) in the transport line or possibly a too poor contact between the wastewater and the active bacteria present there. These functions limit the application of the on-line treatment method. In the operation of small conventional sewage treatment plants, one of the major problems encountered frequently is excess sludge collection, transport, treatment and disposal. This also may be the case for conventional on-site treatment facilities. Several institutions are searching for central sludge treatment options to solve the sludge treatment problems (Kaneko and Shimomura, 1990). However, all solutions for sludge treatment look costly and laborious. In the above proposed concept of a decentralized treatment (Figure 7), part of the sludge, either aerobic or anaerobic, will be returned into the upper parts of the main sewer for increasing the bacterial concentration and activity. The degradation process will occur both for the added sludge and the sewage. A substantial change in sludge properties during its transportation was observed by Kaneko and Shimomura (1990), i.e. the suspended solids concentration declines and the dissolved substrates increases. Such a system therefore could offer a number of potential advantages:

1. Lower costs for sludge collection and transport,
2. Improved treatability of the sewage and removal efficiency,
3. It might decrease the investment and operation costs of the wastewater treatment plant, or release the overload problems of the existing treatment plant, provided the investment costs of the sewer + sludge seeded system are not higher.

The proposed wastewater treatment system includes on-site and sewer on-line treatment processes. It could represent an attractive option, although merely in those cases where a sewage system already is available. Based on the results obtained in this study it can be expected that the proposed concept will lead to a better treatment efficiency and that both the investment and the operation costs of the wastewater treatment plant will decrease relative to that in conventional concepts. Further studies are needed dealing with the SS (seed material) carrying capacity of the sewage, with the quantity and type as well the frequency of the supplied sludge, and with problems related with specific reaction products, such as the accumulation of H_2S and methane. These investigations may lead to an important innovation in wastewater collection, transport and treatment, although to an opinion for real sustainable environmental protection the installation of sewer systems should be reduced to a minimum in favour for community on site sanitation systems.

CONCLUSIONS

Mal-odour problems in sewage transport and storage system are associated with degradation processes occurring in the sewer and storage ponds, and are mainly due to the formation of reduction products, e.g. from sulphate. The odour nuisance problems can be easily controlled by imposing aerobic or micro-aerobic conditions to the sewage by simple aeration.

The degradation rate in unseeded systems under aerobic, micro-aerophilic and anaerobic conditions proceeds according to a first-order reaction with regard to total COD. The reaction constant is high under aerobic ($0.094 d^{-1}$) moderate under micro-aerophilic ($0.073 d^{-1}$) and rather low under anaerobic conditions ($0.055 d^{-1}$) (30°C).

In AnS systems, the methane formation step appeared to be the rate-limiting step at all temperature conditions investigated. On the other hand the rate limiting steps in MA system is rather hard to assess from above experiments. A MA system is more flexible both for pre- and post-treatment; On the contrary, AnS systems are more suitable as pre-treatment, especially at low temperatures, where it mainly will serve as pre-acidification process.

From the results obtained it is obvious that sewer systems could contribute a lot more to pre-treatment than presently is the case. Present sewer systems are hardly effective as pretreatment step, especially not at lower temperatures and/or in short transportation line. To enhance the degradation processes in a sewer, seeding is essential.

Most of the results of this study may be applicable to natural purification systems, such as, anaerobic and facultative pond systems as well as for river self-purification process.

ACKNOWLEDGMENTS

We gratefully acknowledge the technical support of the following individuals: Last, A.R.M. van der, R.E. Roersma, H. Donker, and A. van Amersfoort. We also wish to thank J. van der Laan and M. de Wit for their assistance with the chromatography.

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CHAPTER 3

STUDY OF MECHANISM OF ANAEROBIC PRE-TREATMENT OF DOMESTIC SEWAGE BY USING AN HYDROLYSIS REACTOR

Study of Mechanism of Anaerobic Pre-treatment of Domestic Sewage by Using an HUSB Reactor

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ABSTRACT

An anaerobic(hydrolytic) - aerobic process was developed to treat municipal wastewater at ambient temperature. Results of laboratory scale and pilot scale experiments conducted at the Beijing Gao Beidian(GBD) wastewater treatment plant are presented in this paper. The experimental results demonstrate the feasibility and effectiveness of the hydrolysis - aerobic process to treat municipal wastewater at ambient temperature (13 -31°C).

The most important features of the process are: 1) use of a HUSB reactor instead of a primary sedimentation tank with the similar hydraulic retention time; 2) utilizing the hydrolysing bacteria and acid-forming bacteria to transform biodegradable suspended organic matter into soluble COD and complex (soluble) molecules into simple ones, making the effluent from the HUSB reactor more readily degradable in the aerobic post treatment; 3) hydrolysis of 36% - 56% of the removed solids in the HUSB reactor.

Conventional activated sludge processes with its high capital investment cost, high energy consumption and high maintenance cost hampered the implementation of water pollution control in China. The problems with the development of the new process, presumably now are overcome in some extent. The hydrolysis-aerobic process would significantly reduce capital outlay, energy consumption and operational cost.

KEY WORDS

Aerobic, Anaerobic, Activated Sludge, HUSB reactor, Municipal Wastewater, UASB reactor

INTRODUCTION

In the past, an anaerobic process was only used for treating sewage sludge and high strength industrial wastewaters, because of the low growth rate of anaerobic bacteria and the high sensitivity of anaerobic organisms to environmental factors. The anaerobic process was generally considered as inefficient for treating sewage at low temperature (below 20°C). Moreover, conventional types of anaerobic reactor, such as anaerobic contact process, also limited the further implementation of the anaerobic process.

With the development of a variety of new high rate anaerobic treatment processes, such as the anaerobic filter(AF) (Young and McCarty, 1969), the upflow anaerobic sludge blanket reactor (UASB) (Lettinga, et al., 1979) and the anaerobic attached film expended bed reactor (**AAFEB**) (Jewell et al., 1981), and the anaerobic fluidized bed (**AFB**) (Jenkins et al., 1981), a great break-through has been made in the field of anaerobic treatment, especially for high strength wastewaters. These anaerobic systems were shown to be feasible for treating low strength municipal wastewater as well. Based on achievements in research on anaerobic treatment of dilute wastewater and the experiences of high strength anaerobic wastewater treatment (Lettinga et al., 1983, Genung, et al., 1978, Zheng et al., 1988), investigations to treat municipal wastewater by combined anaerobic and aerobic processes were initiated in China in 1983 (Liu et al., 1984).

MATERIALS AND METHODS

Background The experiments were carried out at ambient temperature conditions at the Gao Beidian (GBD) wastewater treatment pilot plant, using raw domestic sewage of the separated sewer system of Beijing city. The GBD municipal wastewater treatment work is situated in southeast of Beijing, which receives industrial and domestic wastewater collected from more than 70% of the area of Beijing city. Industrial wastewater contributes for 52% of the total wastewater. The wastewater composition is very complex and strongly fluctuate. More than 200 organic chemicals were detected by Gas Chromatography-Mass spectrometer analysis (Wang and Li, 1985). The ratio of BOD_5 to COD ranges from 0.3 to 0.4 (average 0.33) which indicates that the wastewater is poorly biodegradable. The effluent COD concentration of a conventional activated sludge process ranges from 120 to 150 mg/L at 8-12 hours aeration time, which does not satisfy prevailing discharge standards (Yang, 1984). The characteristics of the raw sewage during the experimental period are presented in Table 1.

Table 1 The Wastewater Characteristics of GBD Sewer System

Items	Range	Average	std.
CODt (mg/L)	265 - 996	495	131
CODs (mg/L)	205 - 421	316	62
BOD_5 (mg/L)	124 - 425	199	68
BOD_5s(mg/L)	59 - 198	125	37
SS (mg/L)	162 - 540	330	102

CODs and BOD_5s as soluble COD and BOD_5 respectively

Reactor and Biomass A 37 litres and a 170m^3 HUSB reactor were used for laboratory and pilot plant experiments, respectively (Figure 1 and 2). A gas-liquid-solid phase separator was not installed in these reactors. The HRT of laboratory modified UASB reactor is 2.5 hours and the 2.5 hours aeration tank was used as post treatment. A conventional activated sludge process was used as a control system.

The pilot scale HUSB reactor was constructed by modifying an original multi-hopped horizontal-flow primary sedimentation tank. A aeration tank was used as post treatment. A combination of a primary settling tank (also 170m^3 volume) and an aeration tank system was used as a control system. In the aeration tanks two kinds of aerator, i.e. coarse bubble and fine bubble aerator were installed. The main characteristics of the reactors used in the experiments are summarized in Table 2.

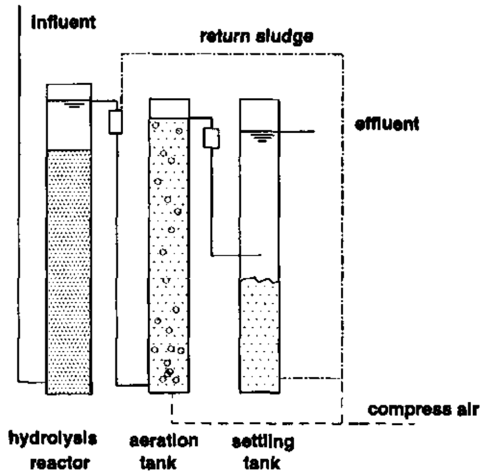


Figure 1 Schematic Diagram of the Lab. Experiment Equipments

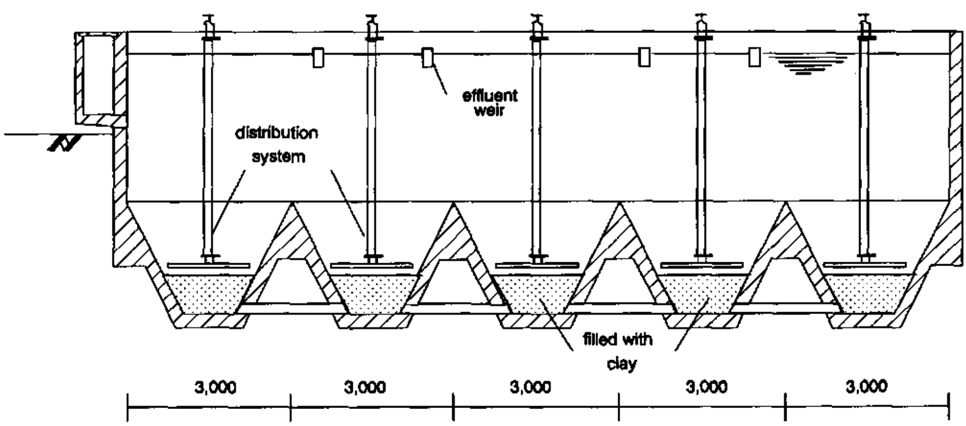


Figure 2 The modified by primary settling tank as HUSB reactor

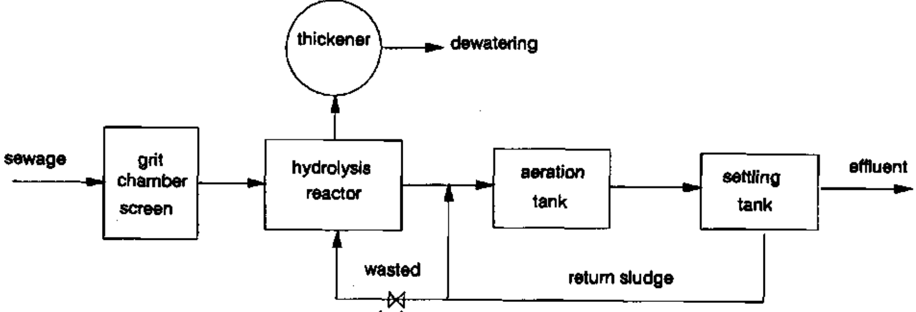


Figure 3 Diagram of the Anaerobic(-hydrolysis)-Aerobic Process

Compared to the conventional process, a modified UASB reactor was used instead of the traditional primary setting tank and the sludge digestion system. The other feature of the new process is that the excess sludge of the aeration tank is returned to the HUSB reactor for digestion and stabilization, and therefore the only discharge point of the sludge in the total installation is from the HUSB reactor (Figure 3).

The HUSB reactor was seeded with digested sludge, which was obtained from the anaerobic digester of the GBD wastewater treatment plant. 10% of reactor volume was filled with digested sewage sludge with a concentration of 20-30 gTSS/L and 55% volatile suspended solids contents.

Table 2 Dimensions and Operation Parameter of Different Reactors

Item	Width (m)	Length (m)	Depth (m)	Volume (m^3)	HRT (h)	Q (m^3/d)
UASB	3	15	4.0	170	2.5	1700
P.S.T	3	15	3.0	170**	2.0	1600
A.T	3	15	3.5	158	4(8)*	950
S.S.T	5	5	7.0	175	2.5	(470)

P.S.T: Primary Settling Tank; A.T: Aeration Tank; S.S.T: Secondary settling Tank;
*: the data in blanket is those of control system;
**: total volume of aeration tank only partly treat effluent from P.S.T or UASB reactor

Sampling and Analysis Methods Analyses were made on 24 hours composite samples kept in a refrigerator at 4°C. Raw samples were used for Suspended Solids (SS), total BOD_5 analyses (APHA, 1985). The pH was measured with a pH-electrode. The spectrometer (type BJ-721) was used for naphthol analytical (Yao et al., 1981).

For Gas Chromatography - Mass Spectrometer Analysis, in order to avoid organic matter interferences of the analysis, a metal sampler and a 5 litre glass vessel were used for sampling and keeping samples, respectively. Two litres sewage sample was extracted by dichloromethane solvent. In the extraction process the pH was adjusted in order to enable the extraction of different constituents, i.e. pH < 2 for acid constituents and pH > 12 for alkaline/neutral constituents. The extracted sample was transferred to a K-D condenser with Snyder column and condensed to one millilitre (Wang et al., 1985). The samples analyses and identification were used Varian 3760 gas chromatography (GC) and Finningan 4510 GS/MS/DS. The compounds will be identified by the GC-MS, when the integrated area value of compound peaks given by GC greater than 1,000. When a compounds could not be identified, the compound is represented as unknown peaks (N.P).

Naphthol Degradation Tests Samples of viable hydrolysis sludge from an operating HUSB reactor and of activated sludge from an aeration tank, respectively were washed with distilled water and five minutes centrifuged (4,000 rpm). Part of the hydrolysis sludge sample was sterilized for 15 minutes at high temperature and high pressure (121°C, 15p/cm^2). A supersaturated solution of naphthol was prepared at 25°C and stored at 4°C in a refrigerator; Sample of the wastewater composed of glucose (500 mgCOD/L) and the necessary nutrients (APHA, 1985) were mixed with sludge up to a sludge concentrations of 15g/L in a batch HUSB reactor and 3.2g/L in an aeration batch reactor. Then 1 mL of the prepared naphthol solution was added to above reactors; The sludge was kept in suspended in the HUSB reactor by a magnetic mixer and by fine bubble aeration in the aerobic reactor; Mixed liquid samples were taken from above reactors. After centrifuged (4,000 rpm) 5 minutes the OD values of the supernatant were measured by the spectrometer.

EXPERIMENTAL RESULTS

Laboratory Experiment

The preliminary experiments were conducted from 1983 to 1984 (Liu et al., 1984). The experiment used a UASB reactor with a three phase separator and 8 hours hydraulic retention time at ambient temperature conditions (9 - 23°C). It was observed that the removal rate of COD, BOD_5 and suspended solids (SS) of UASB reactor were in the range of 50-70%, 60-80% and 70-80% respectively. Despite HRT was long enough, the gas production rate was lower than 0.02 $m^3/m^3.d$. From the research results of this stage, it was found that the removal efficiency of the anaerobic phase was not high enough to satisfy the discharge standards. Aerobic post treatment is still needed. The required hydraulic retention time of the UASB reactor was too long to compete with the conventional activated sludge process. However, the process still offers some advantages in saving of operation cost and energy consumption.

In order to solve above problems, the UASB reactor was modified by leaving out the gas-liquid-solid three phase separator in the second stage laboratory experiments. Table 3 summarized the results of the second stage experiment for the whole process (1984-1985).

Table 3 The laboratory experiment results of modified UASB

items	reactor	anaerobic	aerobic	effluent
	influent	647.6	220.4	
COD mg/L	effluent	220.4	67.4	67.4
	efficiency	49.6%	69.4%	84.6%
	influent	161.5	104.7	
BOD_5 mg/L	effluent	104.7	15.0	15.0
	efficiency	35.2%	85.6%	90.7%
	influent	308.0	29.2	
SS mg/L	effluent	29.2	16.6	16.6
	efficiency	90.5%	43.2%	94.6%

It is obviously that, although the anaerobic reactor was operated at only 2.5 hours hydraulic retention time, up to 49.6%, 35.2% and 90.5% removal efficiencies for COD, BOD_5 and SS, respectively, can be obtained. The final effluent COD was less than 100 mg/L which was much better than the effluent from the control conventional activated sludge process. The air supplied for aeration was 70% less than that of the control system. The aeration tank was adopted to 2.5 hours aeration time for post treatment. The process can compete well with the control conventional process which was operated at 8 hours retention time (Wang et al., 1988). In this stage of the investigation, the activated sludge post treatment process suffered serious sludge bulking problems at high loading rate and oxygen deficiency due to the high oxygen utilization rate for the anaerobic effluent compared to pre-settled sewage. However, the new process provides attractive advantages as alternative technique with respect to the total hydraulic retention time and energy consumption which are favourable to the conventional activated sludge system.

Pilot Scale Experiments

The Start-up Procedure of the System The experiment was operated from August 1985 to 1990. The reactor was operated immediately at full hydraulic loading rate, after the sludge had been added. A dynamic procedure was used to control the anaerobic reactions of the hydrolytic and acidification stage in the UASB reactor (Ghosh, 1981). As the growth rate of methane bacteria differs from that of the hydrolytic and acid-producing bacteria, the hydraulic retention time of the reactor were adjusted to latter species. Methane-organisms then will not develop 'sufficiently'. After two weeks of operation already 40% COD removal rate was achieved.

Comparison of Results with those of a Primary Sedimentation Tank The HUSB reactor and the primary settling tank were operated parallel at almost the same flow rate (Table 4). The removal efficiencies achieved in the two systems differ significantly. The experimental results obtained at various retention times are summarized in Table 4. With regard to the reduction of BOD_5, COD and SS, the efficiency of the HUSB reactor is higher than that of the primary settling tank. Especially for suspended solids, the average removal efficiency of the former system is higher, viz. over 80%, while it amounted only to 40 - 50% in the settling tank. The effluent SS-concentration of the HUSB reactor was less than 50 mg/L, and therefore the effluent quality may greatly improve by such an anaerobic pre-treatment step, when using the same secondary treatment unit.

Table 4 The removal rate of two different reactor at different retention time

Reactor Type	HUSB reactor					Primary settling Tank*		
HRT(h.)	2.5	3.0	3.5	4.0	5.0	1.7	2.3	3.3
COD removal rate	43.1%	41.3%	40.6%	48.0%	40.8%	--	28.9%	--
BOD removal rate	29.8%	33.1%	28.1%	55.4%	34.9%	18.0%	12.0%	17.0%
SS removal rate	82.9%	74.8%	79.0%	81.7%	84.1%	42.0%	40.0%	47.0%

*: the sludge storage volume is not included

Assessment of the Influence of Various Factors

Hydraulic Retention Time During the experimental period the reactor retention times of 2.5 3.0, 3.5, 4.0 and 5.0 hours were applied for periods lasting long enough to establish steady state performance. The average values of COD, BOD and SS removal efficiencies during periods of steady state operation at different HRT values are summarized (Table 4). It appeared that the HRT doesn't affect seriously the removal rate of COD, BOD_5 and SS for this study.

Temperature During the experimental period, the wastewater temperature varied from 13°C to 31°C. The HUSB reactor operated at a minimum temperature of 13°C during winter time. At a retention time of three hours and an average temperature of 16°C in winter, the COD, BOD_5 and SS removal efficiencies still remained at 41.3%, 33.1% and 74.8% (average values of operational data) respectively. The removal efficiencies achieved during winter time were not clearly lower than those obtained during summer time. Figure 4 provides the relationship between the temperature and removal efficiencies for a HRT of 3.5 hours. It is obvious that the removal rate is almost not affected by temperature for the temperature range investigated in this study. This conclusion confirms the results obtained in the laboratory experiment. The observed very minor effect of the temperature may be attributed to the large amount of sludge retained in the reactor, the average sludge concentration exceeded 15 g/L for the whole reactor. However, it should be noted that the hydrolysis ratio of the entrapped suspended solids reduces at low temperature, because it was about 56% at 26°C, while it reduced to 36% at temperature of 16°C (Wang et al., 1988). It is clear that only a small part of the solid organic material is degraded at lower temperatures. As consequently the quantity of excess sludge will increase at low temperature, and therefore in fact and therefore in fact the removal efficiencies of system are seriously influenced by the temperature. Similar results were reported by Lettinga et al. (1983) and Inamore et al.(1984).

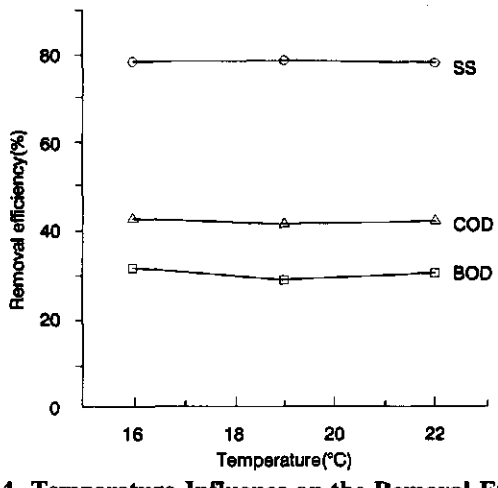


Figure 4 Temperature Influence on the Removal Efficiency

Sewage Concentration The sewage concentration greatly affects the removal efficiency in situ condition. The relationship between the COD concentration and the COD removal rate is shown in Figure 5. The higher is the COD concentration of the sewage, the higher becomes the COD removal efficiency. In this investigation the mean COD concentration of the raw wastewater was 500 mg/L and the average removal rate found amounted to about 45%. It was found that at large fluctuations of the influent concentration i.e. variations in the loading rate from 1.95 kg-COD/m^3.d to 8.8 kg-COD/m^3.d, the effluent COD varied only slightly, viz. from 207 mg/L to 316 mg/L. The system apparently has a considerable capacity to resist shock loading rates caused by the change of influent concentration. Consequently the effect of a shock load to the post treatment process, such as an aeration tank, will be alleviated by the anaerobic pretreatment and so a specific the effluent quality can be guaranteed better.

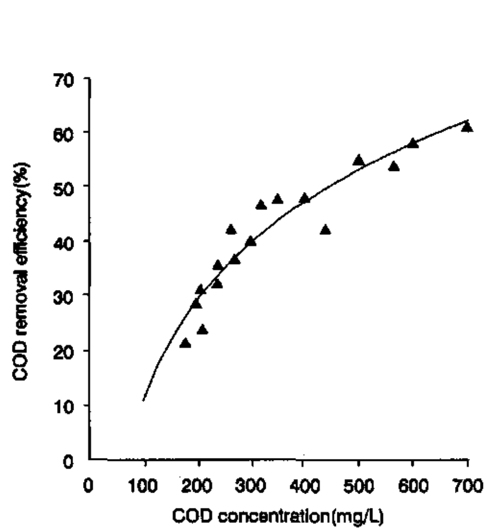


Figure 5 influence of inlet concentration on removal efficiency

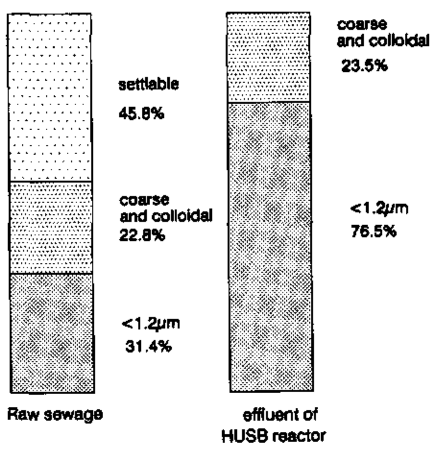


Figure 6 Various ingredients COD removal in different treatment step

Effect of the Composition of the Sewage Figure 6 illustrates the effect of the wastewater composition. The settleable COD (based on 2 hours settling time) in the influent amounted to 45.8% of total COD and is almost completely removed. The coarse and colloidal COD, comprising as difference between settleable and membrane filtered samples COD (pore size $> 1.2\mu m$) is removed for 58%. It therefore is obvious that the HUSB reactor exerts a high removal capacity for suspended and colloidal COD. Part of the removed large sized COD is converted into small sized 'soluble' (very finely dispersed and completely soluble) COD by the hydrolysis bacteria. The ratio of soluble BOD_5 and soluble COD changes from 50.8% and 64.7% in influent to 77.8% and 76.7% in the effluent respectively, and the insoluble COD and BOD_5 removal efficiencies are 74.5% and 55.3%. Occasionally, the effluent COD value is higher than that of the influent for a very dilute influent, due to the conversion of insoluble organic materials to soluble organic in the reactor. The balance for the suspended solids in the reactor also supported this conclusion.

Sludge Treatment and Stabilization Excess sludge of the new process will be discharged merely from the hydrolysis tank. This discharged sludge preferentially should be well stabilized and easy to dispose. The degree of sludge stabilization can be assessed by measuring the specific gas production from the anaerobic digestion process. However, in the HUSB reactor the anaerobic reaction is mainly to the hydrolysis and acidification stages. The stabilization degree of the sludge in such a reactor therefore can not be assessed from the gas production in the system but should be estimated from the hydrolysis (liquefaction) efficiency of the sludge. The sludge hydrolysis efficiency (E) of the HUSB reactor is defined by the following balance equation over a specific experimental period:

$$E = \left(1 - \frac{\text{Accumulated Sludge} + \text{total discharge sludge}}{\text{SS removed by the reactor}}\right) \tag{1}$$

$$E = \left(1 - \frac{(X_t V - X_o V) + \Sigma Q_w X_i}{\Sigma Q_i SS_{re(i)}}\right) * 100\% \tag{2}$$

Where: Xo and Xt: the average sludge concentration in the reactor at beginning and the end of the specific experimental period, respectively; Q: influent flow rate; V: volume of the reactor; SSre(i): removed influent suspended solid; Qw: sludge discharge flow rate; subscript i: refer to daily data.

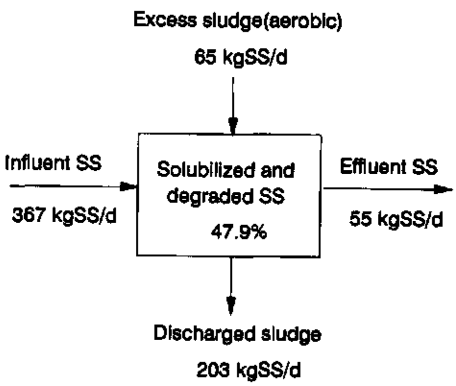


Figure 7 The sludge balance diagram for the combined process of hydrolysis system and the activated sludge (from 1/9/85-1/9/1986; the aerobic sludge yield coefficient is 0.3 kgSS/kgCOD$_{re}$ for the hydrolysis effluent)

Substitution the whole year experimental data into Eq. 2, a sludge hydrolysis ratio was found amounting to 47.9% (T=21°C, Table 7 below). The above data indicated that at the average 47.9% of the removed sludge was eliminated by the HUSB reactor. The hydrolysis efficiency of the HUSB reactor was found in the laboratory experiments to be highly dependent on temperature, i.e. it amounted to 36% at 16°C and to 56% at 26°C(Wang et al., 1988). It therefore should be possible to eliminate the conventional sludge digestion system from the process at specific cases, especially at temperature exceeding 20°C.

Also studies have been conducted to assess the specific sludge properties of the HUSB reactor, such as the dewatering (measured by specific resistance, capillary suction time as well as a model belt pressure filter tests) and hygienic properties. The experimental results indicated that these properties of the excess sludge from the HUSB reactor are similar or even superior to those of digested sludge (Wang et al., 1986, Xu et al., 1987). Since the investment cost of the digestion system generally amount to 30% to 40% of the total cost of the whole sewage treatment plant, a large portion of the investment can be saved by using the new process instead of a system consisting of an activated sludge process and sludge digester.

THE MECHANISM OF HUSB reactor

The Bio-reaction Properties of the HUSB reactor

It looks reasonable that some researchers presume that in the HUSB reactor organic materials are removed mainly by physical entrapment and absorbtion functions of the sludge present and that only a small amount of soluble organic matter is removed, similar as, in a primary sedimentation tank. However, the rather different COD composition of the effluent of the HUSB reactor compared to that of a conventional settler indicates that a different mechanism is involved. Some complementary experiments have been conducted to demonstrate this.

Naphthol degradation tests were made in order to assess: a) the degradation of naphthol when exposed to hydrolysis sludge; b) differences in biodegradation ability of sterilized hydrolysis sludge and viable hydrolysis sludge; c) differences in biodegradation capacity of aerobic sludge and hydrolysis sludge. Naphthol is one of the 129 priority chemicals on the list of U.S. EPA. It is a representative refractory aromatic compound and one of the important pollutants in industrial wastewater of coke gasification installations. It is often present in sewage in China. Naphthol is difficult to degraded in the conventional activated sludge process (Zhang et al., 1984). Degradation tests with naphthol were conducted using batch experiments.

The results of the experiments are shown in Figure 8. For the influent a peak was found at 221 nm, representing naphthol, while in the effluent a peak appeared at 230 nm. It therefore can be concluded that the molecule structure of naphthol is changed in the hydrolysis reaction. Latter peak (at 230 nm) probable originates from salicylic acid or ß-ketohexanbioic acid, which are possibly intermediates of naphthol degradation (Yao, et al., 1981). These intermediates are readily biodegradable, viz. the BOD_5 of salicylic acid is 0.95 mgO_2/mg (substrate), which is favourable for an aerobic post treatment unit.

The results in Figure 9(a), reveal that naphthol present in a naphthol containing synthetic wastewater is completely degraded when contacted to an active hydrolysis sludge for a period of 4 hours, the OD value of the wastewater when exposed to an active hydrolysis sludge dropped down with 96%. Since in the experiment with the non-active hydrolysis sludge system the OD value remains unchanged, it is obvious that the naphthol removal is mainly due to biodegradation.

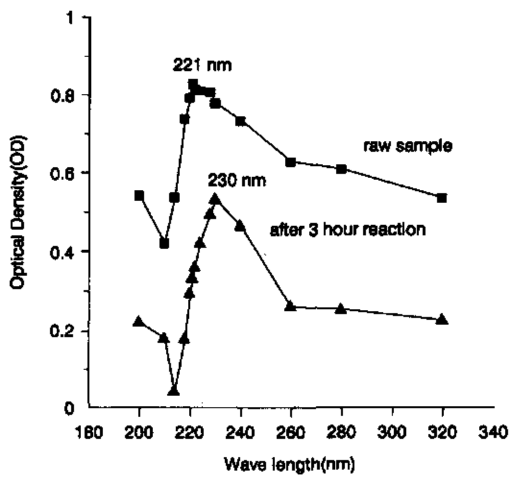


Figure 8 Results an experiment with a naphthol containing synthetic wastewater (500mg COD/L), viz. before and after a sample was exposed to the hydrolysis sludge

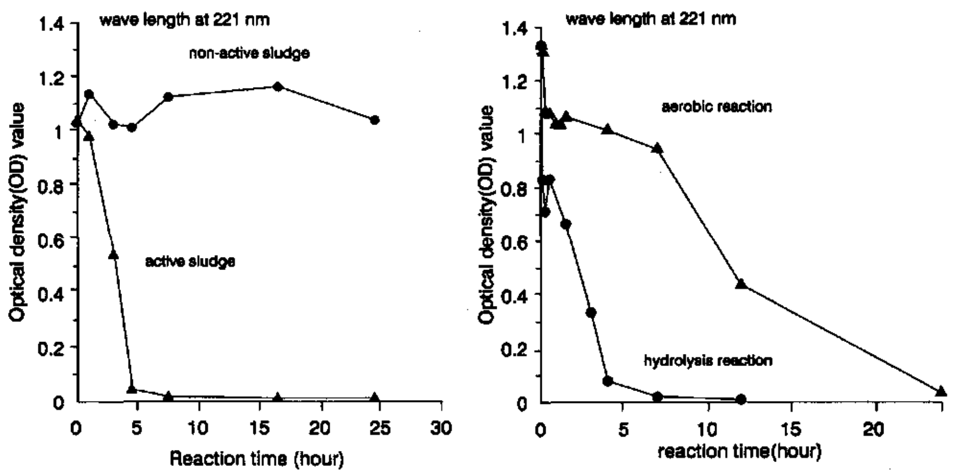


Figure 9 Anaerobic degradation of naphthol in the experiments: a) active and non-active (sterilized) hydrolysis sludge(15g SS/L) and b) Naphthol degradation under hydrolysis and aerobic(3.2 gMLSS/L) conditions

Naphthol is degraded well both in the hydrolysis system and in the activated sludge system (Figure 9b). The degradation rate was higher in the hydrolysis system than in that of the activated sludge system and therefore the necessary reaction time shorter. For a complete removal of naphthol in the activated sludge system more than 24 hours is needed. It also has been found (Qian et al., 1992) that the removal rate of aromatic compounds from a coke gasification plant wastewater in a combined anaerobic-aerobic system proceeds faster than in an one step aerobic system.

Gas Chromatography-Mass Spectrometer Analyses

As the degradation of organic matters in the hydrolysis process apparently differs quite from that of the conventional activated sludge process as well as from that in a conventional (complete) anaerobic process, a HUSB reactor might play an important role in treating complex and refractory compounds present in wastewater. In order to assess the potentials of the HUSB reactor, the influent and effluent of the reactor were analyzed by combined GC-MS. The common chemical parameters and GC-MS analyses of a grab sample are illustrated in Table 5, 6 and Figure 10 and 11.

Table 5 The assessed COD, BOD_5 and SS values in the influent and effluent of an HUSB reactor and the final effluent

Items	influent	HUSB reactor	final effluent
COD (mg/L)	541	263	92.4
BOD_5(mg/L)	215	112	6.5

1) The number of peaks detected at shorter retention time in the effluent of the HUSB reactor both for acid and basic-neutral constituents, but especially for the acid constituents are more relative to the raw wastewater (Figure 10c and d).

2) The total peak areas of the low molecular weight compounds largely increased, while that of high molecular weight compounds (long appearing time) decreased.

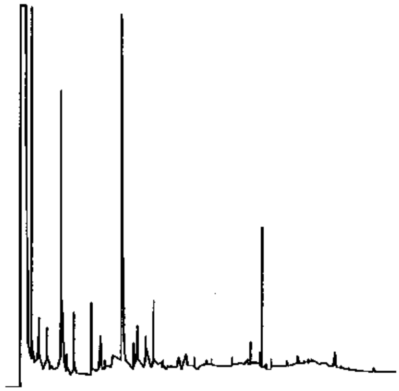
3) The detectable compounds of acid constituents increased from 45 peaks in influent to 60 peaks in effluent, and the compounds of basic-neutral constituents decreased from 35 to 18 peaks.

From the above observations the following conclusions can be drawn:

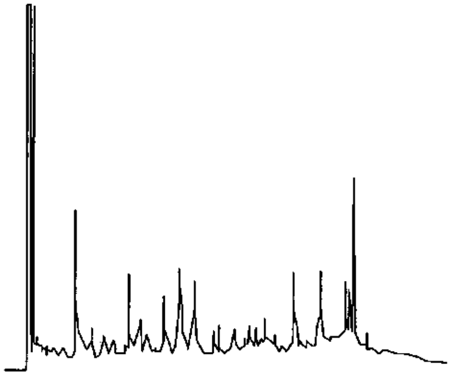
1) As the number of acid compounds increased and that of the neutral/alkaline compounds decreased, both in number and in concentration, it is clear that hydrolysis and acidogenesis really proceeded in the HUSB reactor.

2) The number and concentration of low molecular weight compounds in the effluent of the HUSB reactor is significantly higher compared to the influent. These smaller and soluble compounds, such as VFA are readily degradable in the post treatment units, either a conventional activated sludge process or a complete anaerobic process.

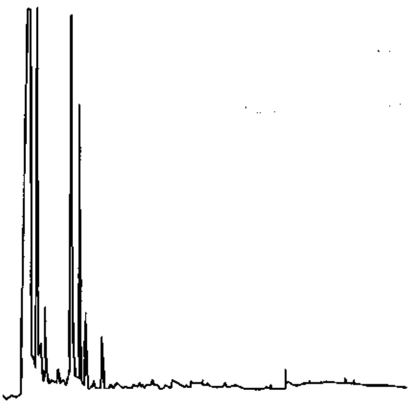
3) Most of the compounds containing more than nine carbon atoms (C_9) particularly aromatic compounds, present in the influent could not be detected in the effluent of the HUSB reactor. The compounds in the effluent generally contain 2-6 carbon atoms (C_2-C_6), indicating that more complex and sometimes refractory compounds have become more readily biodegradable after hydrolysis process.



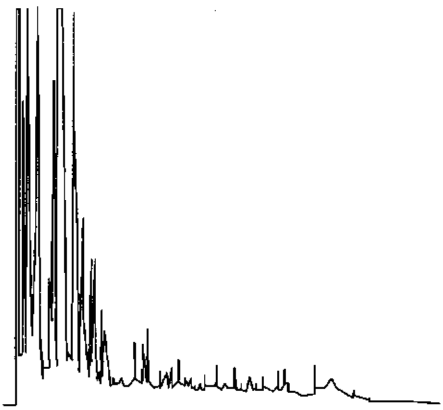
a)influent alkaline/neutral constituent



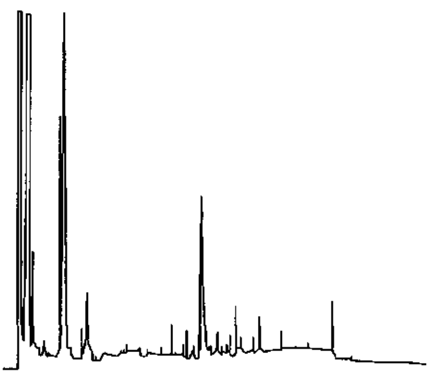
b) influent acid constituent



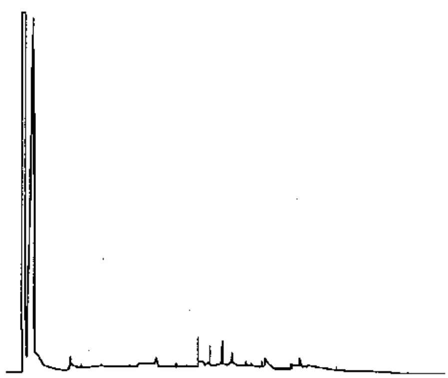
c) hydrolysis effluent alkaline/neutral constituent



d) hydrolysis effluent acid constituent



e) final effluent alkaline/neutral constituent



f) final effluent acid constituent

Figure 10 The different constitutions GC-MS analysis results

Table 6 GC-MS analysis results of the effluent from different treatment processes

No.	Name of chemicals	Molecular formula	Raw Sewage	Pre-treatment		Post-treatment	
				PT*	HT**	AST#	SPS##
1	methane, oxybis[chloro-	$C_2H_4OCl_2$	---	---	*	---	---
2	benzene, methyl-	C_7H_8	*	*	---	---	---
3	3-hexen-2-one	$C_6H_{10}O$	*	*	---	---	---
4	propane, 1,2,3-trichloro-	$C_3H_5Cl_3$	*	*	---	*	---
5	pentanone, 4-hydroxy-4-methyl	$C_8H_{12}O_2$	*	*	---	---	---
6	propane-1-bromo-1, 2-dichloro-	$C_3H_5CL_2Br$	---	---	*	---	---
7	butanoicacid	$C_4H_8O_2$	*	---	*	---	---
8	N.P	----	*	---	---	---	*
9	benzene, ethyl-	C_8H_{10}	*	*	---	---	---
10	propene, 2,3-dichloro-	$C_3H_4Cl_2$	---	---	*	---	---
11	pentanoicacid	$C_5H_{10}O_2$	---	*	*	---	---
12	benzene, 1-methylethenyl-	C_9H_{10}	---	*	---	*	---
13	hexanoicacid	$C_6H_{12}O_2$	*	*	*	---	---
14	N.P	----	---	---	---	*	---
15	propane, 1,2-bis(2-chloroethoxy)	$C_3H_{14}O_2ClC_2$	*	---	*	---	---
16	2-propanol, 1,3-dichloro	$C_3H_6CL_2O$	*	---	---	---	---
17	1-hexanol, 2-ethyl-	$C_8H_{18}O$	*	*	---	---	---
18	1H-indene	C_9H_8	*	*	---	---	---
19	1-propanal, 2,3-dichloro-	$C_3H_6CCl_2$	---	---	*	---	---
20	aceticacid, chloro-	$C_2H_3O_2Cl$	---	---	*	---	---
21	ethanone, 1-phenyl-	C_8H_8O	---	---	---	*	*
22	benzeneethanol	$C_8H_{10}O$	*	*	---	---	---
23	octane, 2,2,6-trimethyl-	$C_{11}H_{24}$	*	*	---	---	---
24	phenol, 3-methyl-	C_7H_8O	*	*	---	---	---
25	benzenemethanol, 2,5-dimethyl-	$C_9H_{12}O$	*	*	---	*	*
26	ethanone, 1-(1-cyclohexen-1-yl)-	$C_8H_{12}O$	*	*	---	---	---
27	methane, chloromethoxy-	C_2H_5OCl	*	---	---	---	---
28	propanoicacid	$C_3H_6O_2$	---	---	*	---	---
29	glycine	$C_2H_5O_2N$	---	---	*	---	---
30	pentanoicacid, methyl-	$C_6H_{12}O_2$	---	---	*	---	---
31	heptanoicacid	$C_7H_{14}O_2$	*	---	*	---	---
32	octanoicacid	$C_8H_{16}O_2$	---	---	*	---	---
33	nonanoicacid	$C_9H_{18}O_2$	*	---	---	---	---
34	cyclohexen-1-one, 3,5,5-trimethyl-	$C_9H_{14}O$	*	*	---	*	---
35	propanal, 2,3-dichloro-2-methyl-	$C_4H_6OCl_2$	---	---	*	---	---
36	propane, 2,2'-oxybis[1-chloro-	$C_6H_{12}OCL_2$	---	---	*	---	---
37	N.P	----	*	---	---	---	---
38	bicyclo [2,2,1] heptan-2-ol, 1,7,7-trimethyl-	$C_{10}H_{18}O$	*	*	---	---	---
39	cyclohexanol, 5-methyl-2-(1-methylethyl)-	$C_{10}H_{20}O$	*	*	---	---	---
40	naphthalene	$C_{10}H_8$	*	*	---	*	---
41	phenol, 4,4'-(1-methylethylidene)-	$C_{15}H_{16}O_2$	*	*	---	*	*
42	benzeneacetonitrile	C_8H_7N	---	---	*	---	*
43	ethanol, 2-phenoxy-	$C_8H_{10}O_2$	---	*	---	---	---
44	benzeneaceticacid	$C_8H_8O_2$	---	*	*	---	---
45	2H-azepin-2-one, hexahydro-	$C_6H_{11}ON$	*	*	---	---	---
46	benzenepropanoicacid	$C_9H_{10}O_2$	---	*	---	---	---
47	phenol. 3,4-dimethoxy-	$C_8H_{10}O_3$	*	---	---	---	---
48	1,2-benzenedicarboxylicacid, bis(2-ethylhexyl)ester	$C_{24}H_{38}O_4$	*	*	*	*	---
49	1,2-benzenedicarboxylicacid, bibutylester	$C_{16}H_{22}O_4$	*	*	*	*	---
50	hexadecoicacid	$C_{16}H_{32}O_2$	*	*	*	---	---
51	decanoicacid	$C_{10}H_{20}O_2$	*	*	---	---	---
52	1H-indole, 3-methyl-	$C_9H_8O_3$	*	*	---	---	---
53	D-galactopyranose, 2,3,4,5,-bis(methylethylidene)-	$C_{12}H_{18}O_2$	*	*	---	*	*
54	phenanthrene carboxylic acid	$C_{20}H_{28}O_2$	*	---	---	---	---
55	2,6,10-dodecatrienol, 3,7,11-trimethyl-	$C_{16}H_{26}O$	*	---	---	---	---

No.	Name of chemicals	Molecular formula	Raw	Pre-treatment		Post-treatment	
			Sewage	PT	HT	AST	SPS
56	1H-tetrazol-5-amine	CH_3N_5	*	---	---	---	---
57	9,10-anthracenedione	$C_{14}H_8O_2$	*	---	---	---	---
58	2-nonen-4-one	$C_9H_{16}O$	*	*	---	---	---
59	benzoicacid ethylester, 3-chloro, 2-oxo-2-phenyl-	$C_{15}H_{11}O_3Cl$	*	*	---	---	---
60	ethanone, 1-(4-hydroxy-3,5 -dimethoxy phenyl)-	$C_{10}H_{12}O_4$	*	*	---	---	---
61	benzaldehyde, 4-hydroxy-3,5 -dimethoxy-	$C_9H_{10}O_4$	*	---	---	---	---
62	D-mannitol, 1,2,3,4,5,6-tri-o-(1-methylethylidene)-	$C_{12}H_{18}O_7$	*	---	---	---	---
63	D-galactopyranuronic alcohol,2,3,4,5-bis-(methylethylethylidene)-	$C_{12}H_{20}O_6$	*	*	---	*	*
64	phenol, 4-methyl-2,6-bis (1,1-dimethylethyl)-	$C_{15}H_{24}O$	*	*	---	*	---
65	ethanone, 1-(4-hydroxy-3-methoxy phenyl)-	$C_9H_{10}O_3$	*	*	---	---	---
66	acenaphthylene	$C_{12}H_8$	*	*	---	---	---
67	benzaldehyde, 4-hydroxy -3-methoxy-	$C_8H_8O_3$	*	*	---	---	---
68	thiophene, 2-(2-ethylhexyl)-	$C_{12}H_{20}S$	---	---	*	---	---
69	pyridine	C_5H_5N	---	---	---	---	*
70	alpha.-L-galactopyranose	$C_7H_{14}O_5$	*	---	---	---	---
71	pentanal, 2,2,4-trimethyl-	$C_8H_{16}O$	---	---	*	---	---
72	2-furancarboxaldehyde, 5-methyl-	$C_6H_6O_2$	---	---	*	---	---
73	octadecenal	$C_{18}H_{34}O$	---	---	*	---	---
74	dodecanol, methyl-	$C_{13}H_{28}O$	*	*	*	---	---
75	ethanol, 2-(9-octadecyloxy)-	$C_{20}H_{42}O_2$	---	---	*	---	---
76	tritriacontanol	$C_{33}H_{68}O$	---	---	*	---	---
77	N.P	----	---	---	---	*	*
78	1-pentyn-3-ol, 4-methyl-	$C_6H_{10}O$	---	---	*	---	---
79	cyclohexanol, dodecyl-	$C_{18}H_{36}O$	---	---	---	*	---
80	octadecen-1-ol	$C_{18}H_{36}O$	---	---	*	---	---
81	nonanol	$C_9H_{20}O$	*	*	*	---	---
82	cyclopentylmercaptan, 2-methyl-	$C_6H_{12}S$	---	---	*	---	---
83	prppane, 2-methyl-2-methoxy-	$C_6H_{14}O$	---	---	---	---	---
84	phpsphate, tributyl-	$C_{12}H_{27}O_4P$	*	---	---	---	---
85	aceticacid, chloro, butylester	$C_6H_{11}O_2Cl$	*	---	---	---	---
86	14-pentadecynoicacid, methylester	$C_{16}H_{28}O_2$	---	---	*	---	---
87	cyclopentaneundecanoicacid, methylester	$C_{17}H_{32}O_2$	---	---	*	---	---
88	hexadecanoicacid, methylester	$C_{17}H_{34}O_2$	---	---	*	---	---
89	aceticacid, hydroxy, methylester	$C_3H_6O_3$	*	---	---	---	---
90	hexadeaaoicacid, 3-hydroxy, methylester	$C_{17}H_{34}O_3$	---	---	*	---	---
91	carbonicacid, dimethylester	$C_3H_6O_3$	---	---	*	---	---
92	benzeneaceticacid, 3-methyl -2,6-dichloro-4-hydroxy-	$C_9H_8O_3Cl_2$	---	---	*	---	---
93	butanoicacid, 3-methyl, phenyleethylester	$C_{13}H_{18}O_2$	*	---	---	---	---
94	4-hexenoicacid, 3-methyl-2,6-dichloro-	$C_7H_{12}O_4$	*	---	---	---	---
95	aceticacid, phenylethylester	$C_{10}H_{12}O_2$	*	---	---	---	---
96	propanoicacid, phenylethylester	$C_{11}H_{14}O_2$	*	---	---	---	---
97	benzoicacid, 2,6-bis(1,1-dimethylethyl)-4-hydroxy	$C_{15}H_{22}O_3$	*	---	---	---	---
98	naphthalene, chloro-	$C_{10}H_7Cl$	---	---	*	---	---
99	phenol, chloro-	$C_6H_{11}Cl$	*	---	---	---	---
100	cyclopentane, 1-bromo-2-fluoro-	C_5H_8BrF	---	---	*	---	---
101	propane, 2-bromo-1-chloro	C_3H_6ClBr	---	---	*	---	---

PT: Primary settling tank; HT: hydrolysis reactor; AST: activated sludge process; SPS: stabilization pond system; *: detected in liquid; ---: not detected in liquid

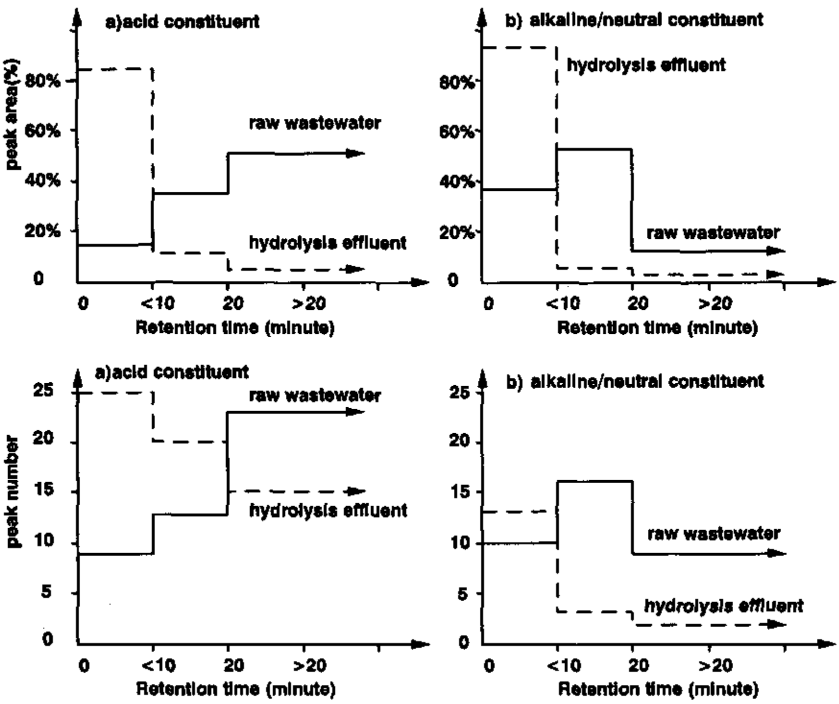


Figure 11 GC-MS analyses of the peak area and of the number distribution; a) The peak area distribution with retention time; b) The peak number distribution

POST-TREATMENT OF THE EFFLUENT OF THE HUSB REACTOR

Operation Results of the Total Process

The experimental results obtained in the total treatment system are listed in Table 7. These results concern the average values over the period September of 1985 to September of 1986. At 2.7 hours average hydraulic retention time of the HUSB reactor, the removal of COD, BOD_5 and SS was 44%, 32% and 84% respectively. The experimental results with the HUSB reactor indicate that removal efficiencies are not significantly different from those of the laboratory experiments (Table 2), although the reactor was scaled up with a factor 4500.

Table 7 Average performance results of the new process(whole experimental period)

Items		Hydrolysis reactor	Aeration tank	The new process	Settling tank	Aeration tank	Control process
T(°C)		21.0	20.8	20.5	21.0	20.7	20.5
HRT.(h.)		2.7	4.0		2.5	8.0	
COD Load(kg/m^3.d)		4.4	1.5		1.1		
COD	influent	493.3	278.4		493.3	379.3	
	effluent	278.4	99.2	99.2	379.3	127	127
mg/L	efficiency	43.5%	64.3	79.9%	23.1%	66.5%	74.3%
BOD	influent	170.2	115.2		170.2	120.3	
	effluent	115.2	12.6	12.6	120.3	15.6	15.6
mg/L	efficiency	32.3%	83.1%	92.6%	29.3%	87.4%	90.8%
SS	influent	277.4	45.3		277.4	157.3	
	effluent	45.6	17.4	17.4	157.3	13.2	13.2
mg/L	efficiency	83.6%	61.5%	93.7%	43.3%	91.6%	95.2%

The new system provides a good final effluent quality compared to the control system. The operation parameters and the effluent data for the two systems are listed in Table 8. The average COD value of the effluent of the new system over the whole experimental year was lower than 100 mg/L, while that of the control system, even at HRT in the aeration tank of 8 hours, was 150 mg/L in the coarse bubble and 120 mg/L in the fine bubble aeration system. It is obvious that the volume of the aeration tank for the new process can be reduced significantly compared to that of the conventional process. This is also true to some extent for the energy consumption as the air supply for the new system is lower (Table 8, refer to air/sewage).

Table 8 The Comparison between New Process and Control System

Process	The control system		The new process	
Item	Large Bubble	Fine Bubble	Large Bubble	Fine Bubble
aeration tank HRT(h.)	8.0	8.0	4.0	4.0
Load(KgCOD/m^3.d)	1.12	1.25	1.2	1.10
Total HRT(h)	14	14	9.5	9.5
Air/sewage(v/v)	15	6.2	7.3	3.8
effluent				
COD(mg/L)	150	127	88	85
BOD_5(mg/L)	10	9	13	7
SS (mg/L)	15	12	20	17

The total HRT includes the primary settling tank(or HUSB reactor) and the secondary settling tank (digestion system was not included).

CONCLUSIONS

In this study we proposed the use of a partial anaerobic treatment process, consisting of a combined hydrolysis - acidogenic pretreatment system. The hydrolysis and acid forming reaction proceeds rapidly for readily biodegradable organic matters. Based on the experimental results obtained, the following conclusion can be drawn:

--- The modified HUSB reactor (HUSB reactor) offers the advantages of a compact construction and rather simple construction and a convenient operation. The hydraulic retention time is very similar to that of the primary sedimentation tank, but the removal efficiencies of COD and SS were higher than those of a primary sedimentation tank.

--- Results of GC-MS analyses and pure substrate biodegradation experiments demonstrate that the biodegradability of raw wastewater improves. A variety of rather complex, sometimes refractory macro-molecules are converted into readily degradable compounds in the hydrolysis and acidification reactions. As a result a shorter retention time and less energy is needed in aerobic post treatment than in a conventional activated sludge process. The process is also beneficial for some refractory industrial wastewaters.

--- The suspended solids removal efficiency of the HUSB reactor is over 85%. The removed SS hydrolysis ratio is 48% at T=20°C. The discharged sludge from the new system is stabilized rather well at higher temperature, but only partially at low temperature. At higher temperatures the new process can treat simultaneously sewage and sludge. The conventional digestion process then can be eliminated from the process.

--- The activated sludge process was used as post treatment unit both in laboratory and the pilot experiments. The experimental results obtained demonstrate the feasibility and effectiveness of the combined anaerobic (hydrolysis) - aerobic process for treating municipal wastewater at ambient temperature. The final effluent quality is equal or even better than that of the conventional activated sludge process. From economical point of view, the new process will largely reduce the capital outlay, energy consumption and operation cost comparing with the conventional activated sludge and sludge digestion combined system. However, from the results is also clear that further research is needed for sludge stabilization problem and other post treatment methods as well as for proving the feasibility of complete anaerobic fermentation process for treating sewage.

ACKNOWLEDGEMENT

This research is funded by the Chinese National EPA and Beijing Municipal Government. The researchers include Zhang Shaofan, Shi Jing whom is gratefully acknowledged. The author also thanks Mr. Ke Jianming for his help in GC-MS analyses.

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CHAPTER 4

THE HYDROLYSIS UPFLOW SLUDGE BED AND THE EXPANDED GRANULAR SLUDGE BLANKET(EGSB) REACTORS PROCESS CONFIGURATION FOR SEWAGE TREATMENT

-- Part I -- Process Development

The Hydrolysis Upflow Sludge Bed and the Expanded Granular Sludge Blanket(EGSB) reactors Process Configuration for Sewage Treatment

-Part I- Process Development

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ABSTRACT

A new process, a combination of a hydrolysis upflow sludge bed (HUSB) reactor and an expanded granular sludge blanket (EGSB) reactor for sewage treatment at ambient temperatures, is presented in this paper. This new process aims at reducing the effluent suspended solids (SS) and soluble organic matter. The total process provides 71% COD and 83% SS removal efficiencies at T > 15°C and 51% COD and 76% SS at T=12°C conditions. The hydraulic retention times are 3 hours and 2 hour in the HUSB reactor and the EGSB reactor, respectively.

The HUSB reactor serves for removing SS and raising the effluent dissolved COD and biodegradability and to accomplish a sustained sludge stabilization. The experimental results obtained demonstrates a 83% SS, 58% coarse COD removal efficiencies and a reasonable extent of stabilization of sludge, while over 50% hydrolysis of sludge in the HUSB reactor is obtained at 19°C. In the EGSB reactor 32-58% of the soluble COD is removed and the biogas production amounts to 23-70 NL/m^3(sewage) at ambient temperature (9-21°C).

KEY WORDS

Acidification, Ambient Temperature, Anaerobic Treatment, EGSB, Hydrolysis, HUSB, Sewage, Sludge Stabilization

INTRODUCTION

Anaerobic treatment processes have been demonstrated to be feasible to treat high-strength and soluble wastewater satisfactorily and dilute and complex wastewater, such as, domestic sewage as well (Lettinga et al., 1980, Grin et al., 1985, Geung et al., 1980, Yoda et al., 1985). Among several anaerobic processes, the upflow anaerobic sludge bed system (UASB) appears to be most promising for the treatment of dilute complex wastewater at ambient temperatures. This is not merely because of its process simplicity and plain mode of construction, operation and scaling-up, but particularly also because of its relatively high loading potential and the relatively high extent of sludge stabilization that can be accomplished (Lettinga and Hulshoff, 1986, Grin et al., 1985).

The investigations with domestic wastewater, using the UASB process, were carried out with small (0.2 m^3) and big pilot plants (6 and 20 m^3) in the Netherlands and with a 64 m^3 pilot /demonstrate plant in Columbia, a 120 m^3 pilot plant in Brazil, a 170 m^3 installation in China and a 336 m^3 installation in Italy (de Man et al., 1988, Schellinkhout et al., 1985, Vierra, 1988, Wang et al., 1989, Collivignarelli et al., 1990). In the meantime some full scale UASB domestic wastewater treatment plants have been put in operation or are under construction, such as a 1,600 m^3 HUSB reactor treatment plant in China, a 1200 m^3 plant in India, a 1,200 m^3 and 6,600 m^3 plant in Columbia and a 1,600 m^3 plant in Brazil (Wang, 1991, Schellinkhout and Collazos, 1991, Draaijer et al., 1991, Vieira, 1988). From this, it is clear that the UASB process, as an alternative sewage treatment process, is slowly becoming more mature.

In previous studies, a modified UASB reactor, i.e., the HUSB reactor was found capable of achieving a very satisfactory SS removal efficiency at a relatively high surface loading rate (> 1.5 m/h) and short retention time (HRT=2.5 hours). At the average 90% and 85% of SS removal efficiencies were achieved in a 36 litre laboratory reactor and in a full scale reactor (170 m^3), respectively (Wang et al. 1988, 1989). Moreover, this process was shown to improve the solubility and biodegradability of the raw pollutants to favourable aerobic post treatment (Wang et al., 1989, Xu and Wang, 1991). On the other hand, the COD_{total} removal efficiency of the system amounted only to 40-50% and the soluble COD removal was very low. In fact the process only provides a pre-acidification function.

At the same time, studies were conducted at the Agricultural University in Wageningen with various types of granular sludge bed reactors, i.e the conventional UASB and the EGSB reactors. Especially the EGSB reactor was found to be quite effective in removing the soluble biodegradable COD fraction. In this system a sludge with relatively high methanogenic activity developed. On the other hand, the suspended COD reduction accomplished in this system was poor (de Man et al., 1988, van der Last et al., 1991). The results obtained in these two independent investigations conducted in China and the Netherlands, reveal that the two systems could become mutually complementary in their advantages and offset in their disadvantages. Therefore, it could become beneficial to combine two systems in a serial in order to improve the treatment efficiency. This process concept represents the objective of the present study. The investigations were carried out to assess the feasibility, a two step process of a combined HUSB reactor + the EGSB reactor, for domestic wastewater at relative short retention times and at moderate temperature conditions.

MATERIALS AND METHODS

Reactors A 200 litre modified Upflow Anaerobic Sludge Blanket reactor was used for hydrolysis and acidification. This reactor was equipped with a gas-liquid-solid phase separator at the beginning of the experiment. The second reactor consisted of a 120 litres Expanded Granular Sludge Bed reactor (EGSB). Effluent recirculation was applied in this reactor in order to maintain a high superficial liquid velocity. The experiments were carried out at ambient temperature conditions in the experimental hall of the Department of Environmental Technology in Bennekom, using raw domestic sewage of the combined sewer system of the village of Bennekom. Because the experimental hall is nearby the village of Bennekom, the sewage is quite fresh or non septic. Figure 1(a) illustrates the two step process flowsheet.

Recirculation Test Recirculation batch experiments, a method developed at the Department of Environmental Technology of the Agricultural University (Sayed, 1988, de Man et al., 1990, van der Last, 1991) were conducted using the UASB (V=1.0 m/h) and EGSB (V=6.0m/h) conditions. The experimental equipment consisted of a column with a 53 mm internal diameter and a height of 600 mm (the total volume of the reactor is 1.25 litres) with a working volume of a 5 litres container (Figure 1b). The one litre column was filled with granular sludge taken from the continuously operated EGSB reactor. After each experiment, the granular sludge was returned to the EGSB reactor to adapt to the sewage.

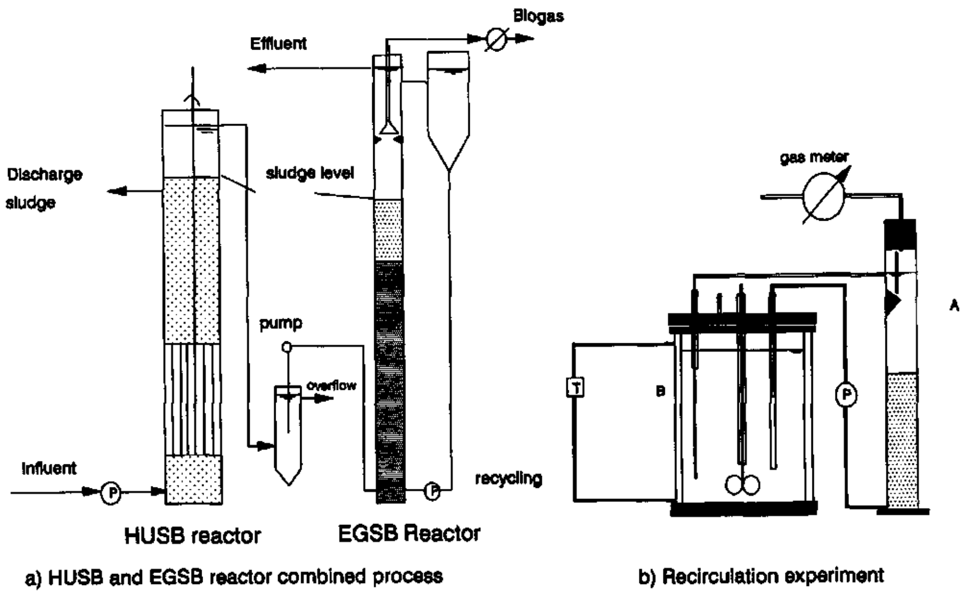

Figure 1 The experiment apparatus used in this study a) the HUSB reactor and the EGSB Reactor in Series Process b) recirculation experiments, T: temperature control equipment; A: one litre UASB or EGSB reactor; B: 5 litre container

Biomass The HUSB reactor was fully seeded with digested sludge, which was obtained from the anaerobic sludge digester of the municipal wastewater treatment plant of Renkum-Wageningen. The granular sludge used in the EGSB reactor was obtained from a full scale UASB reactor, treating starch processing wastewater (Nijmegen, the Netherlands). The average maximum specific methanogenic activity of the seeds were 0.14 and 0.21 kgCH_4-COD/kgVSS.d, respectively (at 30 °C). The granular seed sludge was of a relative poor quality regarding its relatively low specific methanogenic activity.

Sampling and Analysis Methods Analyses were made on 24, 48 and 72 hours composite samples which were kept in a refrigerator at 4°C. Raw samples were analyzed for Suspended Solids (SS), total BOD_5, Kjeldahl nitrogen and total phosphorus and paper filtered samples (Schuell 595½ paper with pore size 4.4 μm) for Volatile Fatty Acids (VFA), ammonia, nitrite, nitrate-nitrogen and orth-phosphate (DNSM, 1969). For COD analyses the micro-COD method according to Knechtel (1978) was used. The total CODt refers to the raw sample, soluble CODm refers to a 0.45 μm membrane filtered sample and CODf refers to a 4.4 μm paper filtered sample. The colloidal CODc and suspended CODs were calculated from the differences between CODf and CODm, CODt and CODf, respectively. The pH was measured with a pH-electrode (Ingold, 426-60-s7). The characteristics of raw sewage during the experimental period are presented in Table 1.

Table 1 The characteristics of sewage of Bennekom

item value	T °C	pH	CODt mg/L	CODf mg/L	CODm mg/L	BOD_5 mg/L	SS mg/L	VFA mg/L	NH_4-N mg/L	T-N mg/L	T-P mg/L
average	15.8	7.4	650	321	187	346	217	51.5	45.2	56.8	10.3
minimum	9.0	8.0	243	118	56	74	41	0.0	73.7	89.2	15.3
maximum	21.5	6.9	1230	589	273	562	594	124.1	27.9	44.1	7.2
No. of data	150	150	150	122	122	31	137	100	15	15	7

Start Up The HUSB reactor was directly operated at its full hydraulic loading rate, i.e. at HRT=3.0 h. and an upflow velocity at V=1.0 m/h. The EGSB reactor was two months afterwards. After adding the sludge to the EGSB reactor, biogas production occurred immediately. Any sludge pre-adaptation is not necessary. Because the most of produced biogas was dissolved in the effluent of the HUSB reactor, and the phase separator was removed after one month of operation. During start up period very severe channelling occurred in the sludge bed of HUSB reactor, due to the poorly designed inlet distribution system and wall effects. In this reactor intermittent mechanical mixing was applied, i.e. 5 second per hour(from 20th day).

Sludge Discharging and Sludge Balance Measurement In order to control the sludge level in a HUSB reactor, the sludge is discharged from the upper part of the reactor. The other potential advantage is that the discharged sludge from the upper part is better stabilized (Wang et al., 1989). The generated sludge is only drained when the sludge interface raised above a certain level, for instants a 30 cm below the water surface (this study). The sludge generation and COD balance measurement were conducted during specific periods. At the beginning and the end of each period, the sludge profiles were measured over the height of the reactor, the reactor volume was divided into various parts according to sampling points. The sludge concentration in each component was the average value of two consecutive sampling points. The sludge hold-up in the reactor is the sum of the amount sludge present in each parts of the reactor. During the specific periods, where we attempted to assess the COD-balance, the SS and COD in the influent, effluent and the total amount of discharged sludge from the HUSB reactor were measured. The specific sludge-COD of the HUSB reactor is the ratio of removed CODs and the removed SS of the HUSB reactor. The sludge-COD can be obtained by multiplying above ratio to the sludge concentration.

RESULTS OF THE HUSB REACTOR

The Results of Operation The HUSB reactor was operated at HRT=3.0 hours throughout the whole experimental period. The total COD removal efficiency varied from 30% to 50%. At the average 60% and 20% removal efficiencies were achieved for the coarse and the colloidal COD, respectively. However, as expected little if any soluble COD removal was observed in the reactor. Although the influent concentration and the temperature fluctuated largely, the performance of the reactor was rather stable (Figure 2 and Table 1). The reactor apparently is capable to accommodate those fluctuations in the influent. This feature of the system certainly is beneficial for the post treatment system, as it reduces shock loads.

On the basis of the temperature and the COD concentration of influent, the results have been subdivided in groups (Table 2). It can be seen that the CODt removal efficiency is more strongly influenced by the concentration rather than the temperature, although the lowest CODt removal efficiency was found at low temperatures (T=11°C, 190-206 days). When the COD concentration of the influent drops from over 600 mg/l to 300 mg/l, the removal efficiency declined from 40% to 10%. However, there is no significant difference for CODt removal efficiency at higher concentration range compared to low temperature period (207-272 days, T=12°C). It can be noticed that a higher CODm removal efficiency was obtained during a lower temperature (period of 207-272 days) compared to higher temperature. This might be attributed to a relatively low hydrolysis and acidification rates of removed CODc and CODs at lower temperature, consequently less soluble matters (CODm) is produced compared to higher temperature. If the methanogenesis is relatively less effected by the lower temperature, the system will maintain a relative high CODm removal efficiency at low temperature periods.

Table 2 The Temper. and concentration effects on the efficiencies of the HUSB-reactor

Period	data	Temperature		Influent(mg/L)					eff.	COD Removal Efficiency					
days	n	range	average	CODt	CODf	CODm	SS	VFA	VFA	Et	Ef/t	Em	Ec	Es	SS
1 - 189	113	14-21	17	697	342	197	237	59	107	38%	52%	-2.6%	23%	65%	83%
190-204*	8	9-12	11	318	170	100	171	13	34	11%	45%	7.3%	-16%	25%	77%
206-272	36	8-13	12	507	286	116	154	40	73	37%	57%	16.1%	39%	49%	75%
total average			8-22	650	321	187	217	54	99	37%	53%	-0.9%	23%	58%	81%

*: rainy and cold weather conditions;

VFA as VFA-COD mg/L

Et: 100*{CODt(influent)-CODt(effluent)}/CODt(influent)

Ef/t: 100*{CODf(influent)-CODf(effluent)}/CODt(influent)

Em: 100*{CODm(influent)-CODm(effluent)}/CODm(influent)

Ec: 100*{CODc(influent)-CODc(effluent)}/CODc(influent)

Es: 100*{CODs(influent)-CODs(effluent)}/CODs(influent)

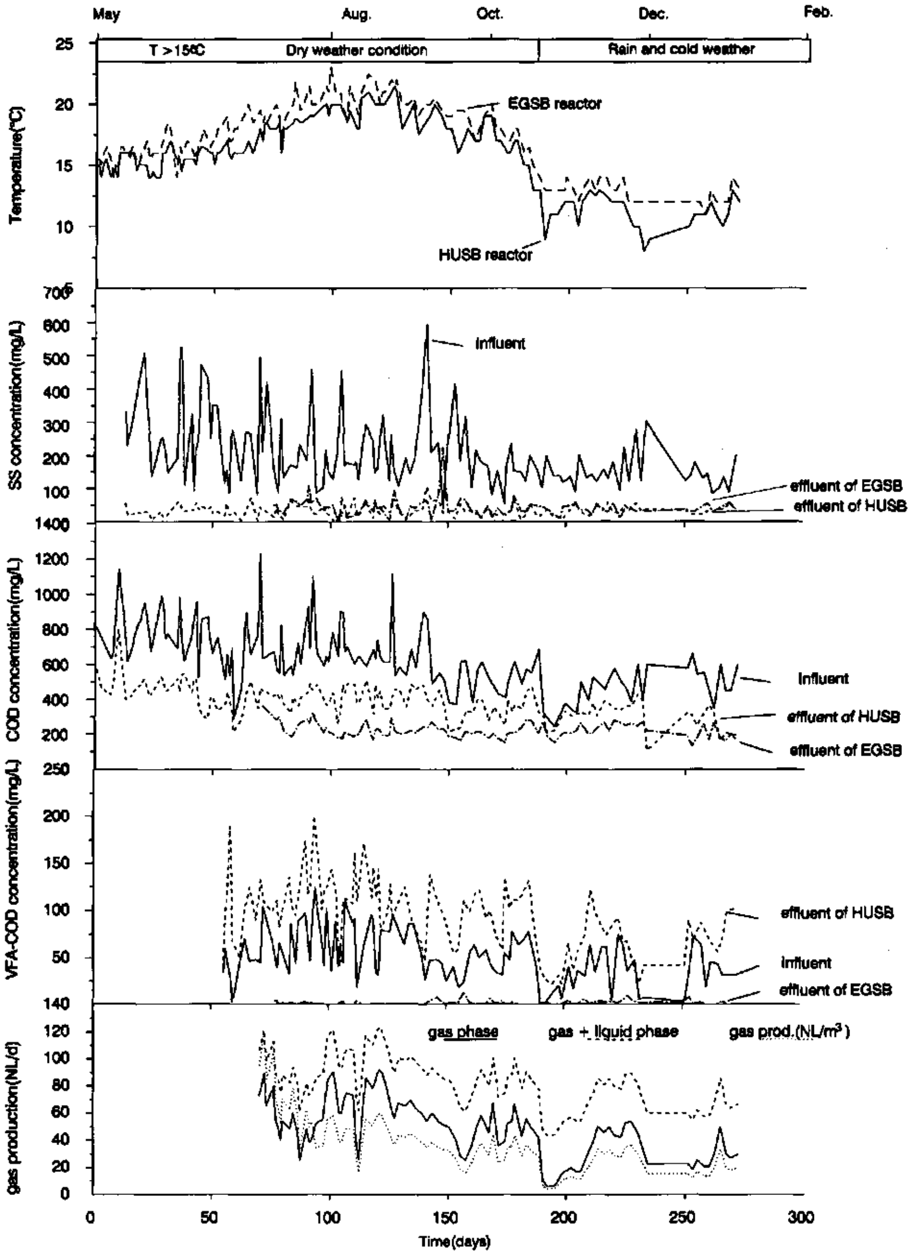

Figure 2 The results obtained with the HUSB and the EGSB reactors (1/5/1991-1/2/1992)

Excess Sludge Generation and COD Removal Balance Measurement of the amount of sludge generated in the HUSB-reactor and of the COD balance were conducted over some specific periods. Some of the results of these measurements are presented in Table 3 and Figure 3 (from July 1 to July 26, 1991). An apparent hydrolysis ratio (R) can be defined as the ratio of the total amount of SS converted relative to the total amount of removed SS. From the data in Table 3, a hydrolysis factor of 53% can be calculated, which means that a reasonable part of the removed SS is converted into soluble matter (or colloidal COD). Usually, the degree of sludge stabilization can be assessed either by the specific gas production of an anaerobic digester or by the solid reduction. Because most of the produced gas leaves the HUSB reactor in dissolved state, the sludge stabilization could not be assessed from the specific gas production and also this has not been determined. The hydrolysis ratio of sludge in the HUSB is measured to determine the degree of sludge stabilization. From this point of view, a certain extent of sludge stabilization (R=53%, at T=19°C) was achieved in process.

$$R(\%) = [\, 1 - \frac{accumulated\ SS + discharge\ sludge}{influent\ SS - effluent\ SS}\,] \times 100\%$$

Table 3 Sludge and COD balance during the period(62 to 87 days, T=19°C)

item	influent	effluent	removed	discharged	accumulated	SRT(d.)
SS (kg)	8.85	1.26	5.08	2.51	0.36	34.5
COD(kg)	27.39	16.32	7.01*	3.55	0.51	—

*: removed COD which does not include discharged and accumulated COD; the amount of total sludge in the reactor amounts to 3.45 kg (begin).

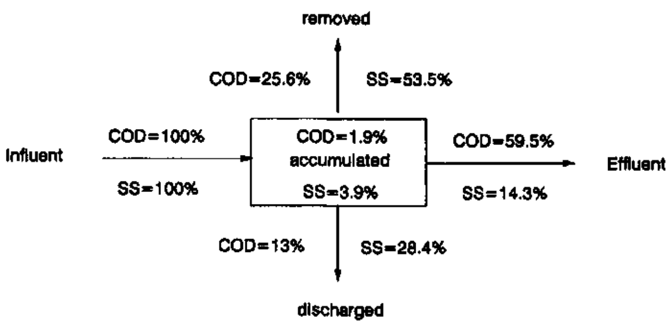

Figure 3 The COD and SS mass balances in the HUSB reactor(period 62 to 87 days)

Except SS removal and liquefaction, also acidogenesis occurs to a considerable extent in the reactor, because the VFA increased from 60 mg/l to 112 mg/l in the reactor. As a matter of fact even more VFA have been produced because also methanogensis occurred in the system. The average COD removal efficiency is about 40%, of which a fraction of 0.37 is retained in the reactor or discharged as surplus sludge. The remaining of the removed COD (viz. 175 mgCOD/L) only can have been eliminated from the system as CH_4-COD or via sulphate reduction and H_2 formation. In the effluent an amount of dissolved methane (about 25 mg/L) is present. This amount of dissolved methane represents 100 mg COD/L at 20°C (Lide, 1992). The Bennekom sewage contains at the average 15 mg SO_4^{2-}-S/L (van der Last, 1991), which upon complete reduction would consume 30 CODmg/L. These figures, together with possibly escaped CH_4 and H_2 in gas state can clear up the gap in the mass balance.

It should be kept in mind that the above figures only provide an rough estimation of mass balance, because of unavoidable errors in sampling and calculations. First of all the sludge-COD of the hydrolysis reactor is based on removed SS-COD. Due to biodegradation and growth of bacteria the composition of removed SS will change. Secondly, the errors caused by sludge sampling for measurement of sludge profiles and discharged sludge can hardly be avoided. Other errors are caused by the fluctuations in sewage composition and concentration in the influent and effluent. In order to obtain accurate data for the mass balance, a further very detailed research using a synthetic wastewater should be conducted in the future.

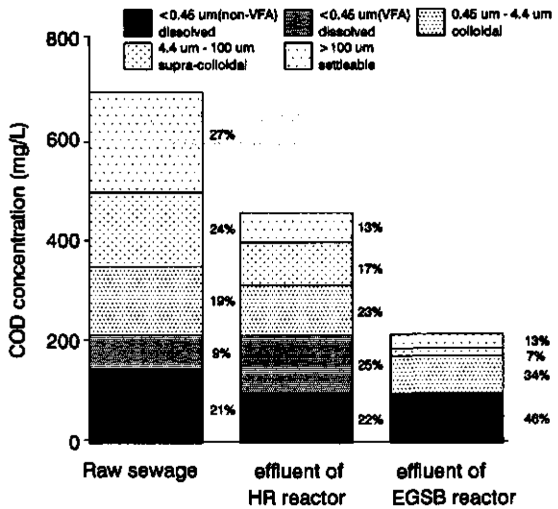

Figure 4 The various COD fractions of the influent and effluent of the different reactors(mean values for whole experimental period)

Effluent Characteristics In order to evaluate the performance of the HUSB the relevant characteristics of the sewage before and after treatment are listed in Table 4 and Figure 4. Significant changes occurred in the BOD/COD ratio and in solubility of pollutants. Both these indexes raised, indicating that the ratio of readily biodegradable fraction of total COD increased. The results in Table 2 also reveal that the expected increase of soluble COD is very small, sometimes it even slightly decrease. This only can be explained when part of the liquified organic material has been converted into methane and H_2 or in CO_2 (through sulphate reduction). Although it can not concluded from the data of in Figure 4 and Table 2 that hydrolysis reaction occurs, from the SS mass balance measurements, it is clear that the hydrolysis of removed SS really proceeded.

Table 4 The characteristics of the wastewater changes after the hydrolysis reaction (HRT=3.0 hours, average data for whole experiment period, n=150)

items	CODt mg/L	BOD_5 mg/L	SS mg/L	BOD_{5f} / BOD_5	VFA / CODt	BOD_{5t} / COD	CODf / CODt	CODm / CODt
influent	650	346	217	0.67	0.09	0.54	0.49	0.29
effluent	397	254	33	0.91	0.25	0.61	0.73	0.49

EGSB REACTOR PERFORMANCE

The Operational Results Table 5 summarizes the results obtained in the EGSB-system at different HRT, upflow liquid velocities (V) and temperature conditions. From these results it can be concluded that the EGSB reactor removal efficiency is hardly affected by the HRT changes imposed to the system. The differences in efficiency apparently depend more strongly on the upflow velocity and are mainly related to soluble and CODs reduction. The rather poor removal efficiency found for the CODs and CODc fractions at high upflow velocity (V=12 m/h). At upflow velocities V $\leq$ 6.0 m/h, better results are obtained. For very dilute types of wastewater like sewage, apparently a moderate upflow velocity is more adequate. Although the lowest removal efficiency was observed at low temperatures (T=12°), there is no clear further evidence that the system was overloaded under these temperature conditions. In fact the contrary seems to be true, because from the VFA degradation curve (Figure 2), it is evident that the effluent VFA is kept at a low level (mean 1.2 VFA-CODmg/L) throughout the whole period of the experiment, even under the cold weather conditions (2.0 mg/L). The specific sludge loading rate applied amounted to 0.08 kgCOD/kg.VSS.d, which in fact means that the sludge was underloaded and its potential for removing organic material apparently was not fully utilised.

Table 5 The experimental results at different upflow velocities, HRT and temperature

period days(n**)	CODt mg/L	CODf mg/L	CODm mg/L	V m/h	HRT h	T °C	LR* kg/m^3.d	efficiency Et	Em	Ec	Es	gas production NL/m^3	NL/kgCODre
71 - 92(14)	419	338	222	12	4.0	19	2.4	36%	60%	25%	19%	116	0.4
93 -112(14)	407	316	213	6	2.0	20	5.0	48%	58%	25%	43%	70	0.3
115-185(34)	378	280	191	2	2.0	20	5.0	41%	49%	25%	39%	68	0.3
186-272(32)	301	203	128	6	2.0	12	3.7	27%	32%	16%	39%	23	0.2

*: LR stand for COD Loading rate; **: numbers of measurement data

The relatively higher CODm removal efficiency and low VFA concentration of the HUSB reactor found at the low temperature conditions (Table 2), which might be the reason of low CODm removal efficiency of the EGSB reactor at low temperature conditions. Because the soluble influent COD of the EGSB reactor is lower, and the biodegradability of the COD presumably is also lower (because of the removal of VFA in the HUSB reactor). The results of all the experiments conducted at the various upflow velocities show poor colloidal CODc and also non-acid soluble COD removal efficiency (Figure 5). The CODm-treatment efficiency at the low temperature is relatively low, which indicates that the biodegradability of the non-VFA fraction is poor. The biogas production amounts to 70 NL/m^3 and 23 NL/m^3(sewage) at the T > 15°C and T = 12°C respectively and the methane content amounted to 80%.

Colloidal COD reduction In order to assess the efficiency of UASB and EGSB systems towards the removal of colloidal matter, complementary recirculation degradation tests were conducted for the UASB (upflow velocity, V = 1.0 m/h) and the EGSB (V = 6.0 m/h) operation modes. Although the colloidal CODc ultimately can be well degraded under both UASB and EGSB conditions (144 hours, Table 6). However, unlike other COD fractions, the colloidal fraction is not well removed neither under UASB conditions nor under EGSB conditions after 24 hours reaction time. This poor efficiency is well understandable, because the colloidal organic matter cannot be directly utilized by methanogenisms. Merely the products of hydrolysis and fermentation reactions can be utilized by the methanogenisms.

Table 6 Recirculation experiments with the effluent of the HUSB reactor (T=20°C)

Time (h.)	COD(t=0) mg/L	Et %	Ec %	Es %	Em %
144(a)	502	74.1	80.2	96.1	56.8
24(a)		63.0	32.2	91.9	52.1
144(b)	502	71.1	63.1	92.3	61.0
24(b)		59.0	23.2	81.0	61.0

a: UASB operation mode (V = 1.0m/H)
b: EGSB operation mode (V = 6.0m/H)

THE OPERATION RESULTS OF THE WHOLE PROCESS

The operation results of the whole process can be derived from the average data from Table 2 and 5 and are shown in Table 7. The results obtained in the temperature range 9-21°C and at an overall total HRT=5.0 hours are certainly promising in terms of treatment efficiencies, gas production and sludge stabilization. Under dry weather (DRW) and at T > 15°C conditions, the total COD treatment efficiency amounts from 60% to 80%. The total gas production (gas produced + gas dissolved) is 70 NL/m^3(sewage) and over 60% of the produced biogas can be recovered, which could be attractive and beneficial for the energy recovery.

During rainy and also cold weather conditions, the total CODt removal efficiency drops down to 40% and the gas production to 23 NL/m^3. However, the final effluent COD concentration remains at approximately the same level throughout the whole experimental period, i.e. 200-250 mg/L. Although the HRT of the whole system is only 5.0 hours, a shorter retention time seems possible on the basis of the results obtained in this studies and those obtained in previous investigations (Last, 1991 and Wang, 1989). In an one step EGSB reactor we suggest to apply a 1.0 - 2.0 hours retention time in the EGSB, and in the HUSB reactor a 2.5 - 3.0 hours HRT at moderated temperature conditions.

Table 7 The results of the HUSB and EGSB reactors in series process

Reactor Items	HUSB reactor average		EGSB Reactor average		Total system average	
T(°C)	17	11	17	12	17	12
HRT(h.)	3.0		2.0		5.0	
Load(g/l.d)	5.3	4.0	4.2	3.7	--	--
Et(%)	38	37	48	27	69	51
Em(%)	-2.6	16	58	32	51	41
Ec(%)	23	39	25	16	40	24
Es(%)	65	49	43	39	79	67

FINAL DISCUSSION

Comparison Between the Different Systems Because a distinctly lower COD removal efficiency and gas production is obtained in the HUSB reactor compared to a conventional UASB reactor, the HUSB reactor provides less pre-treatment than an UASB-system. On the other hand it obviously is superior to other primary treatment methods, such as the primary sedimentation tank, with regard to COD and SS removal efficiencies, and presumably also the extent of sludge stabilization accomplished at a temperature of 19°C as well. Considering the fact that the two pre-treatment systems are applied at a similar HRT, it can be concluded that it is more beneficial to apply the HUSB reactor as a pre-treatment method prior to anaerobic treatment, rather than a conventional settler (refer Chapter 3).

A relative high upflow velocity, i.e. 1.0 m/h (in this study) and 1.7 m/h (in previous study, Wang et al., 1989) can be applied in the HUSB reactor. Because of sufficient mixing caused by the relatively high upflow velocity in the system, the produced gas and VFA are mainly washed out from the reactor and little if any biogas accumulation was occurred. It is clear that a sufficient sludge and wastewater contact is an essential factor for obtaining an improved efficiency of sludge bed reactors, especially for the coarse SS removal. Hence a better mixing and/or an improved distribution system are needed both in the UASB and in the HUSB reactor. Results of full scale HUSB reactors and of UASB reactors indicate that the inlet density should be below one inlet per square meter in these systems for sewage treatment at ambient temperatures (below T=15°C) (Wang, 1991, Vieira and Garcia Jr, 1991, Lettinga et al., 1991).

In experiments the present experiments we switched the operation conditions of the EGSB reactor in the hydrolysis pretreatment system to the UASB mode, i.e. the upflow velocity was changed from 6.0 m/h to 2.0 m/h (Table 5). Comparing the results obtained with those of an one step UASB system (Grin et al., 1985), in which a general a significant longer total hydraulic retention time (HRT=9 h.) was applied, it is clear that the performance of the combined systems of HUSB reactor + EGSB or HUSB + UASB reactor, is superior relative to an one step UASB-system. It therefore can be concluded that the use of a proper primary treatment method, such as the HUSB reactor, is beneficial for improving the overall anaerobic treatment efficiency.

Problems and Solutions From the present experimental results it is clear that the limiting factors for effluent quality are the non-VFA soluble fraction and the colloidal (also including certain supra-colloidal) fraction. They account for 80% of CODt in the effluent of the EGSB reactor (Figure 4). The values reveal that colloidal COD with a size 0.45-4.4μm and below 0.45μm determine the effluent quality. The colloidal matter is difficult to remove in anaerobic treatment contrary to aerobic treatment. Yoda et al. (1985) also reported that colloidal organic matter present in the influent is not or very difficult to remove and that it makes up 60-70% of the effluent of an anaerobic fluidized bed reactor. However, they also report that this fraction can easily be removed by applying an aerobic post treatment. Breure et al. (1991) reported that proteins never can be completely hydrolysed in an anaerobic reactor and that these compounds are more difficult to degrade compared to other substrates, such as carbohydrates. In this study, with the EGSB-system it was found that the removal of soluble COD can be completely attributed to the removal of VFA. The non-acid constituents remained at a constant level in the effluent of the EGSB reactor (Figure 4). The poor removal of colloidal organic matter is the rate limiting step in the UASB and EGSB reactors. Therefore some proper post treatment system should be developed towards the colloidal substrate reduction.

The experimental results obtained in this study indicate that the methanogenic activity of the sludge in a HUSB reactor decreased very significantly compared to that of the seed sludge (Part II, this Chapter). The reason for this is a) an incomplete hydrolysis of accumulated organic matter and b) the fact that only part of the VFA produced is converted into methane, so that growth-in of methanogens remains at a relatively very low level. On the other hand the relatively better CODm-removal efficiency of the HUSB reactor, are maintained before, can be attributed to the relatively low hydrolysis and acidification rates of the removed CODc and CODs. Moreover, the degree of sludge stabilization of the HUSB reactor will decrease at low temperature (Chapter 3). The system will therefore only will provide a partial sludge stabilization.

In order to improve the extent of sludge stabilization and also the removal efficiency for colloidal matter of the system during the cold weather conditions, the HUSB reactor could be combined with a sludge recuperation process, operated in parallel with the HUSB reactor. Considering the fact that the EGSB reactor offers a rather high potential to break down the VFA as well as soluble biodegradable COD in the relevant temperature ranges, the use of such a the sludge recuperation process can be limited - although not necessarily - mainly to the hydrolysis and acidification stages. The sludge can be returned to the HUSB reactor and the produced VFA can be passed to the EGSB reactor, e.g. with the effluent of the HUSB-reactor. Such a process offers the following potential advantages for dilute and complex wastewater treatment:

1. it improves the HUSB reactor treatment efficiencies, as a result of the supplementary sludge regeneration step

2. it utilizes the potential treatment capacities of the EGSB reactor
3. it increases the gas production, consequently the energy recovery in the EGSB reactor
4. it provides a more complete treatment of complex wastewater, i.e. not only for soluble but also for suspended or colloidal matter
5. it provides a further or possibly even an almost complete stabilization of sludge and therefore a smaller amount of excess sludge

Regarding the high concentration of colloidal COD in the final effluent, the use of an aerobic bio-sorption process to eliminate this COD fraction should be considered. The excess sludge of this aerobic treatment step might be stabilized in the HUSB reactor and/or in the sludge recuperation reactor.

CONCLUSIONS

1. The combined hydrolysis + EGSB reactor system looks practically feasible for dilute, complex domestic wastewater treatment at ambient temperature conditions (9-21°C). Particularly at lower ambient temperature the process offers obvious advantages over other conventional one-step UASB systems in terms of liquid retention time, the removal efficiency, gas production and sludge stabilization. The total system provides an average of 71% COD and 83% SS removal efficiency at temperature over 15°C and 51% COD and 76% SS at a temperature as low as 12°C and liquid retention time of 5.0 hours.

2. The HUSB reactor provides an effective pre-treatment with regard to organic matter removal, especially for the coarse SS fraction and its liquefaction and acidification. The high SS removal efficiency obtained can be attributed to a sufficient sludge and sewage contact. The mode of start up plays an important role to lower the methanogenic activity of the sludge retained in the system.

3. A stable gas production was obtained in the EGSB-reactor throughout the whole experimental period. The major part of the produced gas (at $T > 15$°C over 60%) was present in the gas phase. From the very low VFA value in the effluent, it can be concluded that the system is still underloaded at HRT = 2.0 h. at the imposed organic loads. Based on these results and those of previous studies (Wang, 1988, Last, 1991) the adequate HRTs for the HUSB and the EGSB reactors are 2.5-3.0 hours and 1-2 hours, respectively, viz. an overall HRT of 3.5 to 5 hours. Regarding also its reasonable energy recovery, and technical simplicity the system in principle represents a quite attractive alternative COD (BOD) removal-option for sewage.

4. The relatively high content of colloidal COD remaining in the effluent indicates that the removal and/or conversion of this fine particles is the rate limiting step in the process for sewage treatment. In order to achieve a more complete stabilization of the removed sludge, a sludge recuperation process, operated in parallel with the HUSB reactor, is proposed in the investigation. The sludge hydrolysis capacity in the HUSB reactor could be raised in this way. The potential capacity of soluble COD reduction in the EGSB reactor could also be utilized better.

ACKNOWLEDGEMENTS

We gratefully acknowledge the technic support of the following individuals: R.E. Roersma, H. Donker, and A. van Amersfoort. We also wish to thank J. van der Laan and M. de Wit for their assistance with the chromatography.

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THE HYDROLYSIS UPFLOW SLUDGE BED AND THE EXPANDED GRANULAR SLUDGE BLANKET(EGSB) REACTORS PROCESS CONFIGURATION FOR SEWAGE TREATMENT

-- Part II -- AN INTEGRATED ADDITIONAL STEP FOR SLUDGE STABILIZATION AND RECUPERATION

The Hydrolysis Upflow Sludge Bed(HUSB) Reactor and the Expanded Granular Sludge Blanket(EGSB) Reactor Process Configuration for Sewage Treatment

--Part II-- An integrated Additional Step for Sludge Stabilization and Recuperation

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ABSTRACT

A detailed investigation of the SS removal mechanism and the fate of removed SS in a Hydrolysis Upflow Sludge Bed (HUSB) reactor was made. The results demonstrate that the methanogenic activities will deteriorate along with the accumulation of poorly biodegradable suspended solids (SS) in the reactor at a high hydraulic loading rate or SS loading rate. Considering the capacity of a HUSB reactor for sludge stabilization and the potential of an EGSB system for degrading soluble organic matter, a sludge recuperation tank was used parallel with HUSB reactor for improving the stabilization of the sludge. The experiments demonstrate that the total system is feasible to treat sewage quite satisfactorily, i.e. both for soluble and suspended COD at moderate temperature conditions (9 - 21°C).

This new process concept, consisting sequential of a HUSB reactor + the EGSB reactor, combined with the sludge recuperation reactor, may represent an attractive alternative sewage treatment process. The shorter hydraulic retention time (HRT) both for the wastewater treatment and the sludge stabilization (compared to the conventional digester) as well as incorporated reasonable energy recovery make the system also attractive as an alternative process for treatment other complex wastewaters.

KEY WORDS

Ambient Temperature, Anaerobic Treatment, EGSB, HUSB, Sewage, Sludge Stabilization and Recuperation

INTRODUCTION

After the successful application of the anaerobic treatment process for high and middle strength industrial wastewater, the use of process now also can be considered as feasible for the treatment of domestic wastewater, particularly at temperature exceeding 15°C, but very likely also at lower ambient temperature (Coulter et al., 1957, Jewell et al. 1981, Sanz and Fdz-Polanco, 1990). In order to improve removal efficiency of the system at low temperature conditions, a two step upflow sludge bed (USB) system, i,e. an hydrolysis USB reactor followed by an EGSB reactor was developed. The HUSB reactor serves for removing SS, if possible, and raising the effluent dissolved COD and biodegradability and to accomplish a sustained sludge stabilization. The second reactor, i.e. the EGSB reactor, exerts a big potential to convert the VFA to methane even at low temperature conditions (Chapter 4, Part I).

A high SS removal efficiency (over 80%) can be achieved in the HUSB reactor and the removed SS also can be satisfactorily stabilized at higher temperature conditions (T=19°C). Concerning the fact that the HUSB reactor operated at relatively high hydraulic loading rate ($1.0m^2/m^3.d$) and SS loading rate ($2.1kgSS/m^3.d$), the capacity of sludge stabilization in the HUSB system might be affected by above factors, especially at low temperature conditions. There was evidence that the hydrolysis rate for removed SS and colloidal COD was relatively low at lower temperature (refer Chapter 3 and Chapter 4, Part I). The results of investigations with another complex wastewater, namely, slaughterhouse wastewater, indicate that one stage flocculent or granular sludge bed reactors only provide a partial treatment and a rather low degree of sludge stabilization, at temperatures below about 20°C, due to the presence of suspended and colloidal ingredients in the wastewater. This is particularly true at a higher loading rates (Sayed, 1989). Therefore the presented investigation deals with a more detailed investigation of the fate of the removed SS and its effects on the performance of the HUSB reactor. Furthermore, integrated in the two step process configuration, the use of a sludge recuperation system was investigated.

EQUIPMENT AND METHODS

Analytical Methods Analyses were made on composite samples of wastewater samples, kept in a refrigerator at 4°C, and on grab samples of the sludge. Volatile Fatty Acids (VFA), SS, sludge concentration (TSS) and organic content of sludge(VSS) were determined according to Dutch Normalized Standard Methods (1969). For COD analyses the micro-COD method was used (Knechtel, 1978). The amount of VFA and the COD (CODd) were determined on supernatant from centrifuged sludge samples (15 minutes and 5,000 rpm). The centrifuged CODd also includes colloidal and supra-colloidal organics, with particle size up to 0.8-2.5 μm (Rudolfs and Balmat, 1952, Levine, 1985, 1991). For the total sludge-COD measurement, a certain amount of sludge (normally 50ml) was firstly diluted to 1 litre. The of the dilution was subsequently measured. The pH was measured with a pH-electrode(Ingold 426-60-s7).

Stabilization Assays In order to estimate the sludge stability several methods were used or developed. The sludge samples were withdrawn from the reactor and incubated at temperature 30°C. The methane production was measured over a period of 100 day. The amount of methane production per unit mass of volatile suspended solids was used to assess the sludge stability (method described by Lettinga et al., 1991). As such an anaerobic stabilization experiment is quite time consuming, a faster experimental test method for liquefaction and acidification was developed in order to estimate the degree of sludge stabilization. These liquefaction tests were conducted at 20°C and 30°C in 5 litres double wall batch reactors, connected to a thermostat. The reactor was operated open to the air, because the methane production can be ignored under these conditions. The fraction of the sludge hydrolysed chemically was also used. This test was conducted at 20°C exposing the sludge for 24 hours to sodium hydroxide (700 mg/L) under anaerobic condition. The test served to assess the maximum achievable amount of liquefaction (Huang et al., 1989).

Definitions of Sludge and Sludge Liquefaction Although the terms liquefaction and hydrolysis usually are used interchangeable to describe the production of intermediate products prior to their gasification, in fact they here are not strictly synonymous. Hydrolysis is a well-defined chemical term designating the addition of water to complex molecules to be degraded complex substances into simple ones (both into particles of colloidal and supra-colloidal size matters and soluble materials). The definition of liquefaction is rather ambiguous. Liquefaction refers merely to the transfer of substances from the sludge to the liquid phase. Liquefaction is thus restricted to sludge particles. Because in this study, a pore size of 4.4μm paper filter was used both for determining the suspended solids (SS) and the sludge concentration (TSS), the sludge particles is included particles greater than 4.4 μm (DSNM, 1969). The term of liquefaction used in this investigation refer to the conversion of sludge particles of size greater than 4.4μm to particles size small than 4.4μm.

The Integrated Recuperation Tank and HUSB + EGSB Process Results obtained in a 200 L HUSB reactor and a 120 L EGSB reactor were described in Chapter 4, Part I. During the first experimental period of the present investigation, a 12 litres sludge recuperation tank was operated in a semi-continuous mode for sludge stabilization along with the HUSB reactor. Daily discharged sludge from the HUSB reactor was fed once a day to this reactor after exposing the sludge to two hours sedimentation. The applied retention time in the recuperation tank was two days and the tank was operated at a temperature of 20°C. The recuperation tank was stirred gently (60 rpm), the reactor was uncovered. During the second stage, in order to simplify the laboratory process the sludge pre-settling was omitted, a 90 litres recuperation tank was incorporated in the system and it then was operated continuously at 20°C. The recuperation tank was operated at 35°C at the third stage, keeping the other conditions the same as during the second stage. The process lay-out and experiments set-up used are shown in Figure 1 and Table 1.

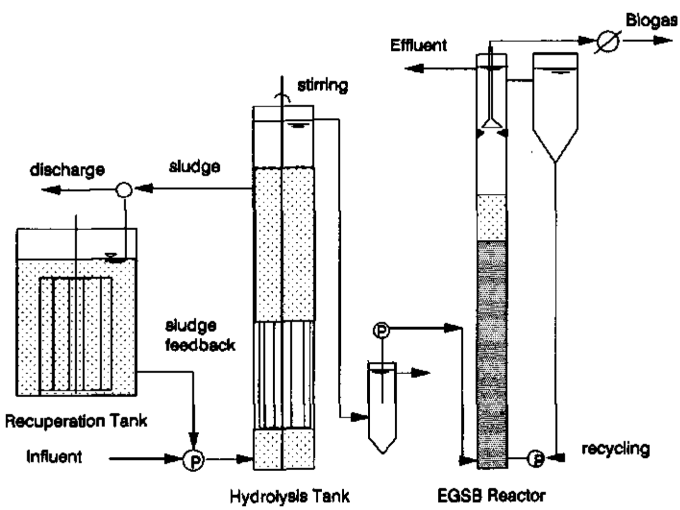

Figure 1 The flowsheet of the HUSB reactor and the EGSB reactor in series combined with a sludge recuperation reactor

Table 1 The experimental set-up for the period of recuperation experiments

Date	Stage	Temperature*	Sludge Rec.	HRT**
81-100	1th stage	20°C	No	2 days
101-124	2th stage	20°C	Yes	2 days
189-215	3th stage	35°C	No	2 days
216-272		35°C	Yes	2 days

*: temperature applied in the recuperation tank
**: hydraulic retention time applied the recuperation tank

SLUDGE CHARACTERISTICS

The Changes of Methanogenic Activity In order to restrict the reactions in the HUSB reactor mainly to hydrolysis and acidification, a dynamic control method was applied. The HUSB reactor for this purpose was operated at a relatively very high hydraulic loading rate (1.0 m^3/m^2.d, HRT=3.0h.). As expected in this way the methanogenic activity of the sludge in the HUSB reactor is lost almost completely after two months of operation. The methanogenic activity dropped from 0.137 gCH_4-COD/gVSS.d for the seed sludge (at 30°C) to less than 0.01 gCH_4-COD/gVSS.d for the HUSB-sludge after two months of the operation (Figure 2). The gradually deteriorating methanogenic activity of the HUSB sludge along with the operation of the reactor should be attributed to the accumulation of poorly and non-biodegradable non-soluble substrate at the imposed very high SS loading rates (i.e. average 2.1 kgSS/m^3.d). Despite the decreasing methanogenic activity, the HUSB reactor removal efficiency of SS and CODt remains almost un-affected. On the other hand, the extent of sludge stabilization will deteriorate substantially due to the decreased solid retention time, and therefore under normal operational conditions the system can only provide a partial sludge stabilization.

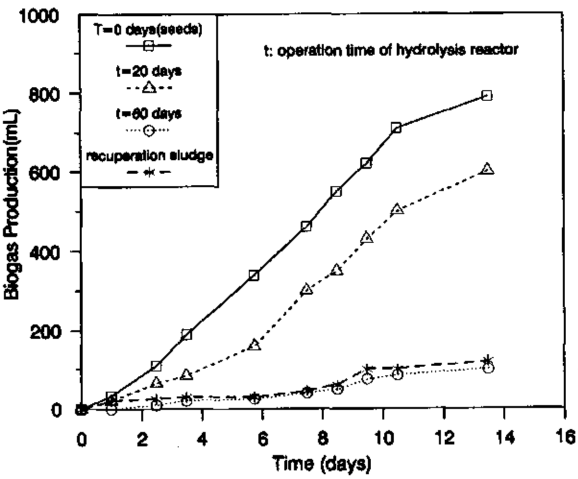

Figure 2 The methanogenic activity measurements for the seed and hydrolysis sludge (using a mixture VFA of C2:C3:C4=600:600:600 mg/L and about 5.0 VSS g/L, at 30°C)

Anaerobic Stabilization Experiments Because of the relatively fresh character of the sewage of Bennekom and the relatively high fraction of volatile solids in the SS (VSS=76%), the percentage of volatile suspended solids in the HUSB reactor is high, i.e. 70%. Such a sludge certainly is further degradable. Therefore sludge samples removed at different times from the HUSB reactor were subjected to the anaerobic stability assay. The characteristics of the different sludges and the results of these assays are shown in Table 2. From these data it can be concluded that the sludge can be stabilized significantly further under anaerobic condition.

Table 2 The stabilization tests for different sludge samples

parameter sludge	VSS(%) t=0(d.)	t=100	VSS Reduction	Stab. test g/g*	Meth. activity g/g.d**
primary sludge(P)	75.9	---	---	---	---
digested sludge(A)	69.2	62.0	27.5%	0.19	0.138
hydrolysis sludge(B)	72.0	61.7	37.4%	0.25	0.079
hydrolysis sludge(C)	70.3	63.1	27.8%	0.17	0.021

A: seed sludge of HUSB reactor;
B: sludge taken two weeks after the HUSB reactor start up;
C: sludge taken two months after the HUSB reactor start up;
*: g/g=gCH_4-COD/gVSS added;
**: g/g.d=gCH_4-COD/gVSS.d (assessed by standard methanogenic activity test method, see Part I)

Liquefaction and Acidification Experiments The liquefaction and acidification experiments were conducted at 20°C and 30°C for different sludge samples. For illustration, a graphical representation of the course of the VFA and centrifuged COD (CODd) for the sludge of the HUSB reactor under the 30°C experimental conditions is presented in Figure 3. The products of the acid fermentation are acetic (C2), propionic (C3), butyric and valeric acids (>C3).

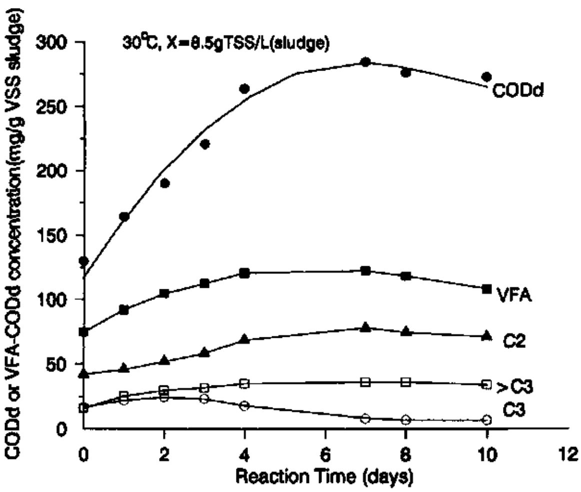

Figure 3 Progress curves showing sludge liquefaction under 30°C conditions

Based on results of gas measurement made in some preliminary parallel experiments it can be assumed that in all tests conducted, the methane production during the first days remained rather small compared to the amount of 'liquified' solid COD for the sludge of HUSB reactor. Nevertheless, from the decrease of VFA-COD and CODd occurring after day 6, it is clear that still a substantial amount of CH_4-COD will have been produced. Unfortunately, in these experiments we omitted to measure the gas production. Nevertheless, it is clear that the methanogenic activity remained relatively low during the course of the experiments, which particularly can be attributed to the sludge properties (refer Figure 2) and to some extent to the imposed tests conditions (i.e. no seeding with methanogenic sludge).

Results and Discussions Liquefaction of sludge from the HUSB reactor was assessed at 20°C and 30°C respectively using the liquefaction assay (Figure 4a). The concentration of the products produced in the reaction depends on the reaction time and, more strongly, on the temperature. An identical trend was also found for the soluble CODd formation. The maximum liquefaction and acidification values found for the different sludges are listed in Table 3.

Table 3 The results of different sludge liquefaction and acidification tests (after 10 days (hydrolysis sludge) to 14 days (primary sludge) digestion time)

parameter	COD_dg/gVSS		VFA-CODg/gVSS		COD_dg/gS-COD		VFA-COD g/gS-COD	
sludge	20°C	30°C	20°C	30°C	20°C	30°C	20°C	30°C
Hydrolysis sludge(C)	0.162	0.287	0.113	0.124	13.5	24.0	9.4	10.3
Recup. sludge(R)	---	0.196	---	0.107	---	10.9	---	5.9
Primary sludge(P)	0.470	---	0.322	---	28.0	---	19.1	---

C: hydrolysis sludge taken from stable operation HUSB reactor

R: recuperation sludge taken from the HUSB reactor at second recuperation experimental period

P: sample taken from primary settling tank at Bennekom; S-COD: sludge COD

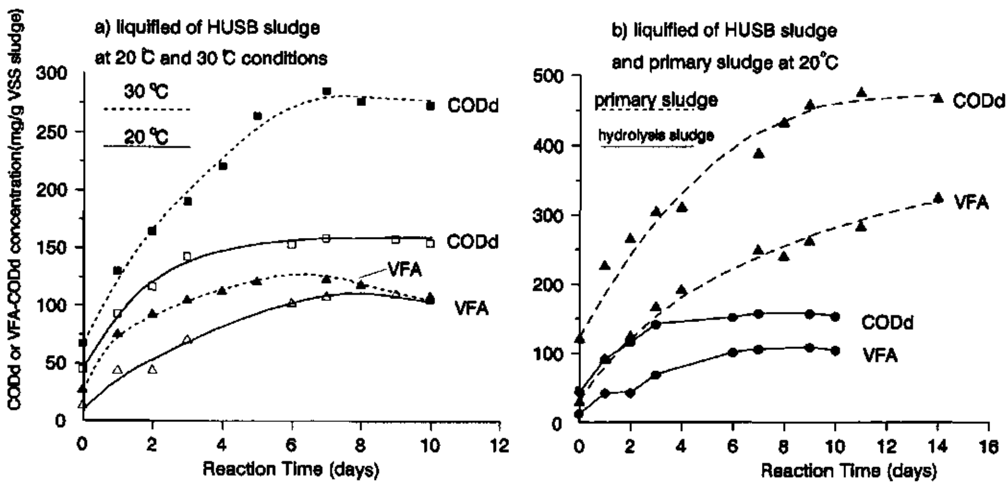

Figure 4 The experiments of liquefaction (acidification) at different temperature (Figure a, HUSB sludge, 20 and 30°C) and for different sludge (Figure b, primary sludge, T=20°C)

For the hydrolysis sludge, the soluble COD accounted for 0.162 and 0.287 gCOD /gVSS or to 13.5% and 24% of the total sludge-COD, after 10 days reaction at temperatures of 20°C and 30°C, respectively. The produced amount of VFA-COD ultimately (after 8-10 days) becomes almost the same at the different temperature conditions, which could be interpreted as that the acidification is not seriously influenced by the temperature. However, it is clear from the results of the experiment conducted at 30°C, that in fact methanogensis occurred which resulted in a decline of the VFA-COD after day 6. For the liquefaction reaction at these temperatures a significantly higher CODd was found at 30°C than at 20°C.

Comparing the results of the primary sludge and the sludge of the HUSB reactor, the liquified CODd and VFA concentrations of primary sludge are significantly higher than those of HUSB-sludge (Figure 4b). In fact, the VFA-COD produced from the primary sludge even is higher than the CODd produced from the hydrolysis sludge. These results lead to the conclusion that the sludge from a steady operated HUSB reactor is relatively well stabilized. In specific cases it therefore might be possible to dispose the sludge directly, without further stabilization.

The question is to what extent the experimental conditions such as exposure of the mixed liquid to the air and stirring, also inhibit the liquefaction and acidification reactions. The results obtained showed that the chemically maximum achievable amount of liquefaction of the sludge from the HUSB reactor amounted to 32% of the total sludge-COD (Based on CODd/COD-sludge). In the liquefaction test 24% of the total sludge-COD is converted into CODd at 30°C (Table 3), viz. 75% of the chemically liquified sludge of the HUSB reactor. Regarding the results obtained it can be concluded that the developed liquefaction assay could be an useful tool to assess the biodegradability (stability) of a sludge. Since a certain amount of methanogensis will occur in these tests - depending on the quality of the sludge - it looks recommendable to collect and measure the accumulated amount of methane formed during the test. As methane formation proceeds via VFA, the accumulated CH_4-COD should be accounted for as VFA.

SLUDGE RECUPERATION EXPERIMENTS

Sludge Recuperation The results of the three experimental periods are presented in Figure 5 and Table 4. From the results of the first experimental stage, it can be seen that a semi-continuously fed sludge stabilization tank performs satisfactory when operated at two days retention time and at 20°C. The CODd and VFA increased from 2.7% and 0.8% in the influent to 12.6% and 5.9% in the effluent based on the total sludge-COD, respectively. Except the slightly lower VFA, the results are very similar to those obtained in the batch sludge liquefaction experiments conducted at the same temperature at a 8 to 10 days reaction time (Table 3). It appears that compared to the influent about 10% more sludge-COD is converted into CODd in the semi-continuously operated recuperation tank. During the second stage, the ratio of converted COD_d to the total sludge-COD decreased from 9.9% (the first stage) to 4.5%. This drop can be attributed to the decreased biodegradablility of the HUSB sludge following the incorporation of the recuperation step in the circuit, as results of which the HUSB-sludge became further stabilized.

Table 4 The results of the recuperation experiments during the three experimental stages

experimental stage		data number	T °C	PH	MLSS g/L	VSS %	COD_d/VSS mg/g	VFA/VSS mg/g	COD_d/COD_t %	VFA/COD_t %
1st	influent	12	20	6.9	20.1	70.9	35	10	2.7	0.8
stage	effluent		20	6.4	19.8	69.7	164	76	12.6	5.9
2nd	influent	16	20	6.9	10.1	70.4	67	20	5.1	1.6
stage	effluent		20	6.4	9.3	68.8	125	71	9.6	5.5
3rd	influent	30	35	7.2	7.8	73.2	71	14	5.7	1.1
stage	effluent		35	6.9	6.0	72.0	178	95	14.1	7.6

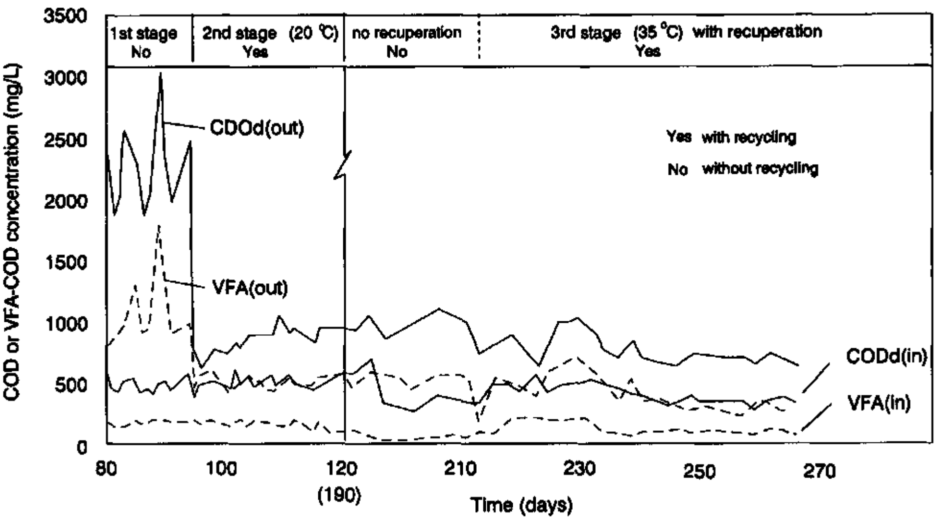

Figure 5 The results of recuperation tank during the different stages(HRT=2.0 days)

From the data of the batch experiments in Table 3, it can be seen that sludge - as expected - will be liquified to higher extent under higher temperature conditions (30°C). Therefore, during the third stage of the experiments, the recuperation tank was operated in the more optimal temperature range of T=35°C. The hydrolysis and acidification products increased compared to the results of other two experimental periods. However, the produced amount of CODd and VFA at 35°C conditions remained lower than proceed in the batch experiments conducted at 30°C. The results further indicate that the HUSB reactor sludge in fact became increasingly stabilized after the incorporation of recuperation tank in the system.

__The Performance of the Total Process__ A detailed comparison can be made using the operational results of the HUSB and the EGSB reactors during the second and third experimental period. Approximately 40 gCODd/day and 25 gVFA-COD/day produced in the recuperation tank were added to the raw influent wastewater in the second experimental period, which resulted in a 26 mg/L higher influent CODd and 15 mgCOD/L higher VFA-COD, due to the return of recuperation sludge to the HUSB reactor. The results also show that the VFA value in the effluent of the HUSB reactor increased with about 20 mg/L. However, no clear indications were obtained that the methanogenic activity of the HUSB-sludge improved during this experimental period.

The results of the third stage, obtained under cold and rainy weather conditions, can not be compared to those of the other experimental periods (period 1 and 2), because the differences in removal efficiency is mainly caused by the more dilute character of the influent under rainy weather conditions (Chapter 4, Part I). The results of the EGSB reactor are comparable throughout the different stages of the experiment. However, because of the increased CODd and VFA resulting from the sludge recuperation, the gas production in this study increased from 19 to 31 NL/m^3 during the third stage. The VFA concentration in the EGSB effluent remains at a low level throughout the whole experimental periods. It therefore is clear, that the EGSB reactor still is under loaded and certainly has a higher potential capacity for soluble COD reduction.

DISCUSSION

As the VFA produced in the HUSB reactor will become diluted by the large quantity of influent passing the reactor, it only plays a minor role in the biogas production. A relatively large amount of the biogas produced will be present in dissolved form in the effluent. It could be possible to use a module reactor set-up using part of the HUSB reactors alternatively for elutriation of the recuperated sludge in order to obtain a more concentrated effluent in practice applications. Then better results can be expected both in gas production and effluent quality. The results of the experiments don't reveal a big improvement in COD removal efficiency of the HUSB reactor, and especially also not for the effluent from EGSB reactor after implementation of the recuperation tank. Nevertheless, the experiments show that the implementation of a sludge recuperation reactor is beneficial for improving the sludge stabilization, while it also offers advantages for improving the soluble COD removal efficiency of the EGSB reactor, because of the relatively higher VFA and the gas production, particularly at lower temperature.

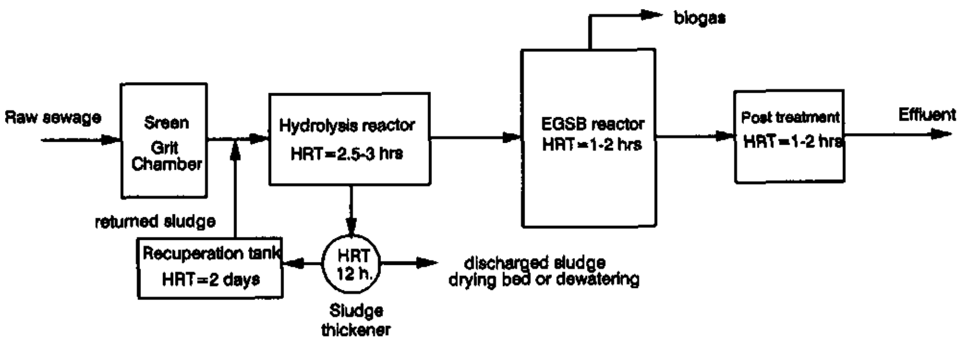

Figure 6 The flowsheet of an alternative sewage treatment process

On the basis of results obtained, the treatment process shown in Figure 6 is proposed as an alternative for sewage treatment. The proposed process has obvious advantages, i.e. lower investment and operational costs and a lower energy consumption compared to the conventional activated sludge process. The wastewater is treated mainly by anaerobic methods resulting in little if any energy requirements. The total process HRT is only 4.5 -

7 hours, which compares well to conventional aerobic systems. Moreover, over 50% of the removed SS is hydrolysed in the HUSB reactor at 19°C and regarding the relatively low effluent concentration of the EGSB reactor only a small amount of excess sludge in the aerobic post treatment will be produced. The total amount of sludge produced in the system presumably is significantly lower compared to that of the conventional process. If 12 hours sludge pre-thickening is adopted, the 97-98% water content of the sludge could be obtained (Wang et al., 1987 and Chapter 7) The volume of the recuperation tank could be reduced largely. Based on the good results of the prelimary research of the recuperation tank, a 2 days retention time is proposed here. It is clear that 2 days retention time is not optimized value and the further research of the recuperation tank should be conducted in the near future. A 2 days sludge recuperation time is much shorter than the retention time normally applied in a conventional anaerobic digester (HRT=20-30 days). Only about 10% reactor volume is required for the sludge stabilization compared to the conventional anaerobic digester, therefore, in this respect the investment cost can be reduced largely. The recovered biogas amounts to 20 - 50 NL/m^3(sewage). The distinguished features of the process simplify the sewage wastewater treatment process.

CONCLUSIONS

1. The batch 'liquefaction' assays developed in this study provide a rapid and fairly reliable method for practice to assess the sludge stability. In addition to, CODd or VFA-COD also the cumulative CH_4-production should be measured. The results of the above assays indicate that the removed SS or the hydrolysis sludge certainly can be further stabilized. However, the extent of stabilization of the hydrolysis sludge looks very similar to that of the digested sludge of conventional system.

2. The relatively high SS removal efficiency obtained in the HUSB reactor possibly partially can be attributed to the relatively low methanogenic activity, i.e. the absence of biogas production. The results obtained indicated that the very low methanogenic activity of the sludge results from the accumulation of poor biodegradable SS in the reactor at the imposed high SS loading rate.

3. The two step HUSB + EGSB process configuration combined with the sludge recuperation system looks an attractive option for the treatment of domestic wastewater at ambient temperature conditions (9 - 21 °C). The treatment capacity of the EGSB reactor is utilized for the removal of soluble biodegradable matter, partially resulting from liquified organic solids from the sludge stabilization and recuperation process. A higher COD removal efficiency, and higher gas production and a more complete sludge stabilization are achieved in the process compared to the process without recuperation stage, especially at cold weather conditions.

4. The rather short required total retention time of 4.5 - 7 hours required for the whole treatment process and the fact that only about 10% reactor volume is needed for sludge stabilization compared to the conventional anaerobic digesters as well as its reasonable energy recovery make the system attractive as an alternative process for sewage treatment, and for various other complex wastewater treatment as well.

ACKNOWLEDGEMENTS

We gratefully acknowledge the technic support of the following individuals: R.E. Roersma, H. Donker, and A. van Amersfoort. We also wish to thank J. van der Laan and M. de Wit for their assistance with the chromatography.

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CHAPTER 5

THE MICRO-AEROPHILIC TREATMENT PROCESS FOR THE EFFLUENT OF THE HUSB AND EGSB REACTORS TWO STAGES ANAEROBIC TREATMENT SYSTEM120

The Micro-aerophilic Treatment Process for the Effluent of the HUSB and EGSB Two Stages Anaerobic Treatment System

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ABSTRACT

This paper describes results of post treatment experiments conducted with the effluent from the combined two stage process hydrolysis upflow sludge blanket (HUSB) reactor and the expanded granular sludge bed (EGSB) reactor. A number of experiments/tests were conducted, comprising sedimentation, multi-stage anaerobic treatment, bio-absorption as well as micro-aerophilic system conditions.

The proto-type micro-aerophilic upflow reactor developed in this study was operated at a hydraulic retention time of only one hour and at a temperature of 13°C. The experimental results obtained indicate that an almost complete treatment can be achieved in this way at very low cost and in a technically plain system, which looks extremely suitable for developing countries. This system, combined with a proper physicochemical process, like chemical coagulation looks very suited for complete treatment of domestic sewage.

KEY WORDS

Post treatment, Micro-aerophilic, Colloidal, Absorption, Anaerobic and aerobic treatment, physicochemical treatment

INTRODUCTION

At present, anaerobic wastewater treatment is a widely applied bio-technique and its considered as a grown-up technology. The anaerobic wastewater treatment system is widely applied for in various high strength industrial wastewaters as well as for low and medium strength wastewaters. Recently this led to investigations dealing with treatment domestic sewage (Lettinga et al. 1980, Grin et al., 1983, Jewell et al., 1981, Genung et al., 1980). Some researchers are of the opinion that at the present state of knowledge, the anaerobic treatment method still is not sufficiently proven for domestic wastewater, and therefore, it is not yet adequate for broad scale application for this purpose (Stwizenbuan, 1985). However, because of the serious environmental situations in most developing countries and therefore the urgent demand for the low cost treatment methods there, the UASB technology for domestic sewage treatment plants is becoming relatively rapidly implemented in these countries (Draaijer et al., 1991, Schellinkhout and Collazos, 1991, Vieira, 1988, Wang et al., 1989, 1991).

It is widely accepted that anaerobic treatment in principle is a pretreatment process. In generally the anaerobic treatment step must be followed by post-treatment in order to reach acceptable surface water quality. Aerobic and anaerobic biological treatment processes for domestic sewage treatment offer several advantages and disadvantages relative to each other. In principle, combining the two systems in a serial set-up, could made them mutually complementary in their advantages and offset in their disadvantages. Based on the quite few literature available on post treatment processes, it can be seen that most of the researchers adopted conventional aerobic treatment processes. However, at some aspects the well established aerobic processes may be not suitable for the quite different treatment targets to be addressed in anaerobic post treatment. Because the characteristics of anaerobic effluent are quite different from the original sewage, such as low BOD/COD ratio (poor biodegradability), high content of H_2S, NH_4^+-N and high VFA (in some situations), some inverse effects were found by some researchers in the above combination (Wang et al., 1987 and van Buuren, 1991). It is also clear that the optimum configuration for such a system should studied as soon as possible, while otherwise, lack of a proper post treatment method will become a serious obstacle for the implementation of anaerobic treatment.

In this study, efforts were made to develop an optimum post treatment process for application in the developing countries, where the main pollution problem is still organic matter rather than nutrients. Therefore, a considerable emphasis was put on the development of low cost post treatment systems. Micro-aerophilic treatment systems look extremely interesting, as the results obtained sofar indicate that an almost complete treatment can be achieved by this system, and the method is really low cost system (Lomans, 1991). The effluent of the combined HUSB and EGSB reactor is used for the influent of the post treatment. Above two step system offers very promising prospects with regard to organic matter and SS removal and sludge stabilization, and gas production as well(in this thesis, Chapter 4, Part I and II).

MATERIALS AND METHODS

Analytical Methods

The micro-COD method according to Knechtel (1978) was used for COD analysis. The total CODt refers to the raw sample, the CODm refers to 0.45 μm membrane filtered sample and the CODf refers to 4.4 μm paper filtered sample. The colloidal CODc and suspended CODs were calculated by the differences between CODf and CODm, CODt and CODf, respectively. The sludge concentration was measured using the standard method (APHA, 1985).

Experimental Methods

Absorption Experiment The adsorption experiments were conducted as follows. A certain amount of activated sludge from a denitrification pilot plant in the experimental hall of Bennekom was aerated for more than 12 hours in order to remove all absorbed or undegraded substrate for the absorption experiment. The absorption experiment was performed with 2 litres pre-aerated centrifuged sludge (about 2g/L) and 2 litres effluent of an EGSB reactor using a 3 litres vessel. A magnetic stirring device was used for mixing (Figure 1a). A control system using the same experimental set-up was operated in which an aerated sample of wastewater was used. After certain time intervals samples were drawn from the mixed liquor for determining changes of COD.

Micro-aerophilic Experiments The micro-aerophilic degradation experiments were conducted using two batch reactors, consisting of 6 litre double wall reactors with a working volume of 5 litres. In the Micro-aerophilic system" (MA system) the solution is exposed to the air for aeration under conditions of gentle stirring at a rate of 60 rpm. The reactor assembly is operated at a temperature of 20°C during the experiments by using a thermostat (Figure 1b).

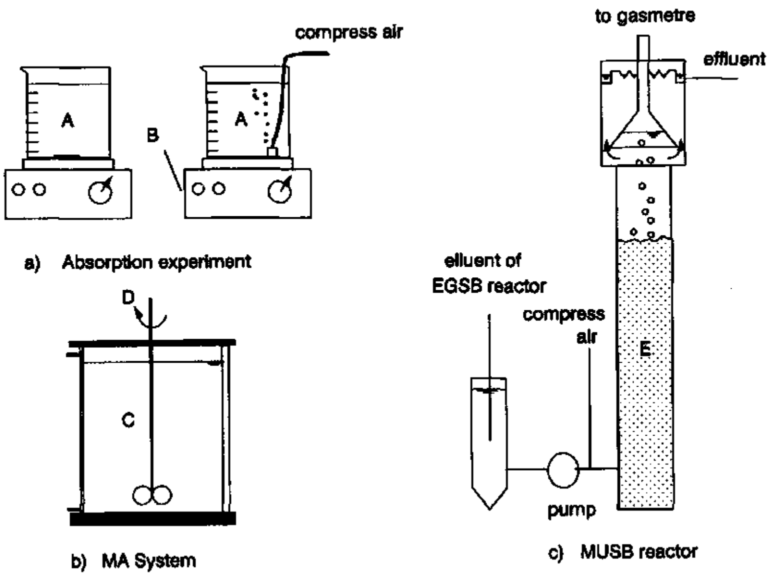

Figure 1 Schematic diagram of experiment arrangements a) absorption experiment b) Micro-aerophilic experiment and c) MUSB reactor (A: 3 litre reactor B: magnetic stirring device; C: 6 litre reactor; D: stirring device; E: 15 litre upflow reactor

Anaerobic Recirculation Experiments Recirculation batch UASB (V=1.0 m/h) and EGSB (V=6.0 m/h.) experiments, a well established experimental method at the Department of Environmental Technology, WAU (Sayed, 1988, 1990,de Man, 1989, van der Last, 1991) were performed for assessment of the maximum achievable removal efficiency of anaerobic treatment using the procedures described in Chapter 4.

EXPERIMENTAL RESULTS

The Characteristics of the Effluent of HUSB and EGSB Reactors

The effluent COD and BOD concentration, of the combined HUSB - EGSB process are in the range 200 - 250 mg/L COD and 60 - 100 mg/L BOD_5. The CODc, CODm and CODs (including supra-colloidal matter) amounted to 34%, 46% and 20% of the total COD, respectively (see Chapter 4). Normally, the efficiency is calculated by using raw influent and paper filtered sample to assess the potential removal capacity of the system. In the present study, we compared the removal efficiencies based on paper filtered samples and based on settling tests (using a 2 litre column for 4 hours settling) for the effluent of the different reactor systems. The results are listed in Table 1. It is clear that the efficiency calculated on the basis of filtered samples provides a certain over-estimation of the achievable COD removal efficiency, because only 12.4% more of COD in the effluent of an EGSB reactor can be removed by plain sedimentation (Table 1). Although the data reveal that more than 50% of the COD consists of coarse and colloidal COD, it seems not so easy to remove these polluting fractions by simple settling. Therefore, another post treatment process should be adopted than plain sedimentation.

Table 1 The removal efficiency based on paper filtered and settled samples*

	COD(mg/L)					Removal efficiency			
Wastewater	CODt	CODf	settled	CODs	settlable	Et/t	Ef/t	E	Ef/t-E
raw sewage	610	379	443	231	167	---	37.9%	27.4%	10.5%
effluent of HUSB	394	321	362	73	32	35.4%	47.4%	40.7%	6.7%
effluent of EGSB	268	165	219	103	49	32.0%	58.1%	44.4%	13.7%

E : After 4.0 hour settling(the maximum achievable removal efficiency)
Et/t: 100*{CODt(influent)-CODt(effluent)}/CODt(influent)
Ef/t: 100*{CODf(influent)-CODf(effluent)}/CODt(influent)
* : average for five experiments

However, it should be kept in mind, that there exists an essential difference between the definition of COD fractions used by chemists and engineers. Engineers generally define the soluble, colloidal, supra-colloidal and settleable COD on the basis of available separation methods (van der Last et al., 1991, and this study), i.e. soluble (<0.45 μm though membrane filter), colloidal (0.45 - 4.4 μm through paper filter), supra-colloidal (4.4-100 μm) and settleable (>100 μm settling 4 hours), while the more scientific particle size categories of contaminants are: dissolved (<0.001 μm), colloidal (0.001-1.0 μm), supra-colloidal (1-100 μm) and settleable (>100 μm) (Levine et al., 1985, Levine et al., 1991).

As the matter of fact, in the engineers definition the paper filtered fraction is so broad, that it covers main part of the non-settleable supra-colloidal fraction. It therefore easily results in an over-estimation of the potential removal efficiency based on the paper filtered samples. On the other hand, the soluble COD fraction covers part of the colloidal COD, and it results in an under-estimation of the dissolved COD removal. It depends on the different treatment stages to what extent there will be an over-estimation of COD removal efficiencies on the basis of paper filtered samples. The maximal over-estimated value is 14% for the effluent of the EGSB reactor.

The Maximum Achievable COD Reduction in Anaerobic Treatment

The anaerobic recirculation experiments were carried out to assess the maximum achievable amount of degradation for the effluent of the HUSB reactor. This experimental procedure, in fact, provides an opportunity to assess the maximum reduction potential of anaerobic treatment system. The results of a recirculation experiment are shown in Fig. 2.

There exist little differences between UASB (V=1 m/h) and EGSB(V=6 m/h) systems for removal efficiencies of the different COD fractions (Figure 2). From the experimental results, it can be concluded that for this specific HUSB-effluent non-biodegradable COD concentration is around 130 mg/L under anaerobic conditions. Very similar results, i.e. a non-biodegradable COD-fraction of 100-140 mg/L, were obtained for other samples of the sewage in other anaerobic degradability assays (in this thesis, Chapter 2). Regarding the applied fairly long contact time the results are rather disappointing from the view point of satisfying the discharge standards, although after 8-10 hours recycling time 80-90% maximum achievable COD were obtained. However, results with other types of sewage (initial COD ca. 500 mg/L) i.e. that from a site in Beijing give more encouraging results, viz. 60-70 COD mg/L ultimate values in the anaerobic recirculation experiments using the same granular sludge, while values around 120-130 mg/L COD in the final effluent of the combined HUSB and EGSB process were obtained (Wang et al., unpublished results).

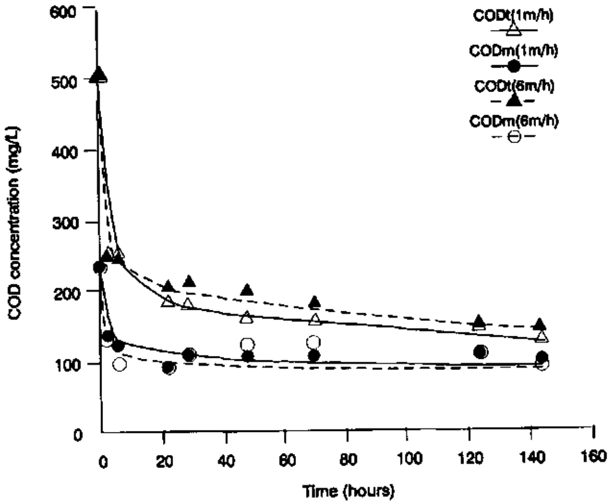

Figure 2 The recirculation experiments conducted at 1.0 m/h(UASB mode) and 6.0 m/h(EGSB mode) upflow velocities at 20°C and 144 hours reaction time)

The Micro-aerophilic Post Treatment Process

The effluent of the two step anaerobic treatment process was used to assess the removal of the remaining COD in two micro-aerophilic post treatment experiments, operated in parallel, i.e. one with seed (TSS=150mg/L) from an micro-aerophilic experiment conducted previously and the other without seed material (Figure 3).

From these results it is clear that for speeding up the reaction rate, consequently for decreasing the reaction time, addition of some seed sludge is necessary. When pre-treatment or partial treatment using anaerobic pre-treatment processes is insufficient, part of the remaining organic pollution can be easily removed in 1 or 2 days or even significantly shorter by applying micro-aerophilic system, which is substantially shorter than that of a stabilization pond. Therefore, it looks very attractive to enhance micro-aerophilic conditions in post treatment pond systems, where these systems already exist or land is available to install them. In this way an almost complete treatment can be achieved at very low costs and in a technically plain mode. This looks extremely suitable for developing countries. A similar approach was investigated previously at the Department of Environmental Technology, of the WAU by van Buuren (1991) and Lomans (1991) for treating the effluent of an one step UASB reactor. They were also obtained very promising results.

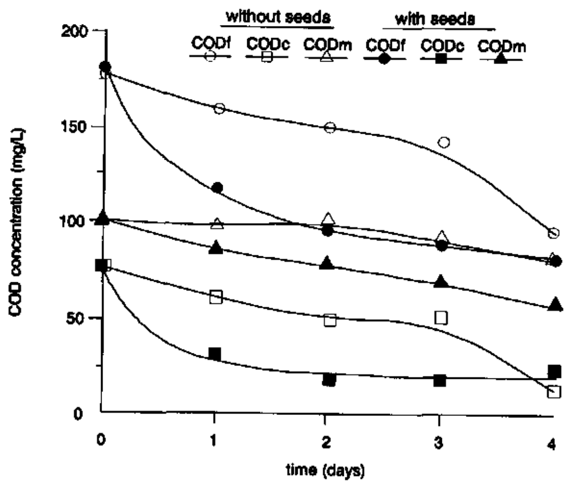

Figure 3 The course of the CODf, CODc and CODm fractions in experiments with the micro-aerophilic system applied for the effluent of two step anaerobic treatment with seed (150 mgSS/L) and without seed materials

Absorption Experiment

Since around 60% of the COD in the final effluent of the EGSB-reactor consists of coarse and colloidal COD, an sludge absorption process could be useful to eliminate these ingredients. According to Bunch and Griffin (1992) the colloidal COD-fraction can be rapidly removed in this way. Therefore, some batch absorption experiments were conducted with the effluent of EGSB reactors using pre-aerated activated sludge, in order to assess the potential of this approach in removing remaining COD.

Figure 4(a and b) show the course of the concentration of the different COD fractions along with the contact time at 15°C, the results indicate that removal of the colloidal fraction CODc proceeds rapidly compared to that of the other COD ingredients. The colloidal COD decreased from about 100 mg/L to 50 mg/L within five minutes. After the initial drop, the colloidal COD values tend to remain a constant level, independent on whether aeration was applied or not. Such a rapid removal was not observed for other COD fractions. The soluble COD fraction did not decrease substantially in the absorption experiments, even not under aerated conditions. In an other experiment, the contact time was prolonged to two hours, and also here little if any improvement in the removal of soluble COD was observed. Since the BOD/COD ratio in the effluent is only 0.3, it is clear that readily biodegradable substrate already was almost depleted in the anaerobic reactor.

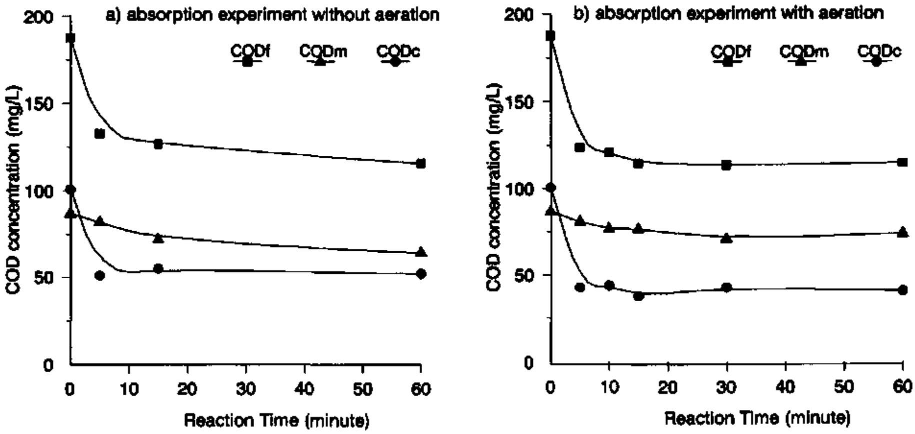

Figure 4 The courses of the different COD ingredients in the batch absorption experiments without aeration(a) and with aeration experiments(b)

The Performance of the MUSB Reactor

Based on the promising results of the batch experiments a continuously operated micro-aerophilic upflow sludge bed process was developed. In this MUSB system the influent is introduced at the bottom of the reactor. The influent and sludge were kept in sufficient contact by applying hydraulic mixing and a minimum of aeration (water/air =1:1). Air was supplied to impose micro-aerophilic conditions and to keep the blanket suspended. The 15 litres upflow reactor (H=1.9 m) was operated at one hour hydraulic retention time (Fig. 1c).

During the experimental period, we found that the MUSB reactor can be maintained under aerobic conditions, despite the very limited supply of air. As the sulphide oxidation process proceeds very rapidly (Buisman et al., 1990), the effluent of the MUSB reactor is completely free from mal-odour. In fact, the sulphide already becomes partially oxidized in the effluent pipe of the EGSB reactor, as became apparent by the sulphur film developed on the wall of effluent pipe. Although the applied upflow velocity exceeded 2.0 m/h, the established activated sludge blanket could be well retained in the MUSB reactor. The developed micro-aerophilic sludge exerted good settling properties. The MUSB reactor even can accommodate higher upflow velocities or more turbulent conditions, as could be imposed by more heavy aeration or hydraulic conditions. Regarding the satisfactory settling characteristics of the sludge, the upflow sludge bed reactor concept therefore in principle also can profitably be applied for micro-aerophilic systems and possibly for aerobic treatment.

Table 2 The Results of the MUSB reactor(T=13°C)**

Item	CODt mg/L	CODf mg/L	CODm mg/L	CODc mg/L	CODs mg/L
Influent	221	171	83	88	51
Effluent	146	108	69	39	38
Efficiency(%)	35	51*	15	58	22

*: based on filtered effluent and raw influent
**: Period Dec.1,1992 to end of Jan, 1992

The obtained experimental treatment data of the system under conditions of HRT=1.0 hr. over a period of 8 weeks are listed in Table 2. The results confirm the earlier observations of the batch experiments. Most of the removed COD consists of colloidal CODc. The colloidal COD removal efficiency amounts to 58% (50% removal efficiency in batch experiments) and the soluble COD removal efficiencies is slightly higher than obtained in the batch experiments. Because the colloidal and coarse fractions contribute at the average to 63 percent of the total influent COD, the rapid removal of these fractions points to sorption and coagulation as being the most likely mechanism responsible for the removal. The sorption mechanism causing the rapid colloidal removal presumably is a bio-physical process, it proceeds rapidly.

In this study we also investigated another aerobic process for post treatment, viz. a 120 litre rotating biological contactor (RBC). The HRT applied in these RBC experiments was two hours (the two hours sedimentation not included). A similar performance as found with the MUSB reactor was obtained.

DISCUSSION

The results of the present investigations indicate that the biodegradability of pollutants present in the effluent of the EGSB reactor is rather poor. This can be derived from the low ratio of BOD to COD of the EGSB effluent. The BOD/COD ratio of 0.53 of raw sewage decreased to 0.3 for the final effluent of the EGSB reactor. Accordingly the treatability of the effluent in terms of COD removal efficiency remains rather low. Regarding the characteristics of the EGSB effluent, conventional aerobic treatment processes as post treat unit in the situations where anaerobic treatment proceeds sufficiently efficiently, as was the case in the present investigation, is not a proper choice from economic point of view. These conventional systems require a huge amount of energy and a relatively big reactor volume to treat a relatively small amount remaining biodegradable COD.

For developing a proper post system, the following aspects should be considered: 1) the required removal efficiencies for the different polluting ingredients, viz. COD, SS, VFA and pathogens present in the anaerobic effluent; 2) the process complexity, viz. the system should be technically simple and plain in its operation; 3) possibilities to combine biological post treatment with a physio-chemical process. Since most of the pollutants are or can be removed in the anaerobic pretreatment process, the remaining organic pollutants mainly consist of colloidal matter, which has a relatively poor biodegradability. For this reason, by using a small amount of coagulant, such $Al_2(SO_4)_3$ or $Al(OH)_3$, the effluent quality in principle can be improved substantially. Comprehensive investigations were conducted in China in recent years with secondary effluents for purposes of reuse as dual water. The results obtained in these studies indicate that 20 - 40% COD removal efficiency can be obtained by dosing 10-20 mg/L coagulant (Sheng, 1992). Regarding the small amount of coagulant needed, only a small amount of non-toxic sludge will be produced. Therefore we propose a post-treatment process consisting of a MUSB reactor, in which biological and physio-chemical processes proceed, it beneficial combined with a chemical coagulation flocculation process (Figure 5). In order to keep the MUSB reactor in aerobic or micro-aerophilic conditions, pre-aeration can be applied if necessary. The excess sludge of the MUSB reactor can be hydrolysed in the HUSB reactor.

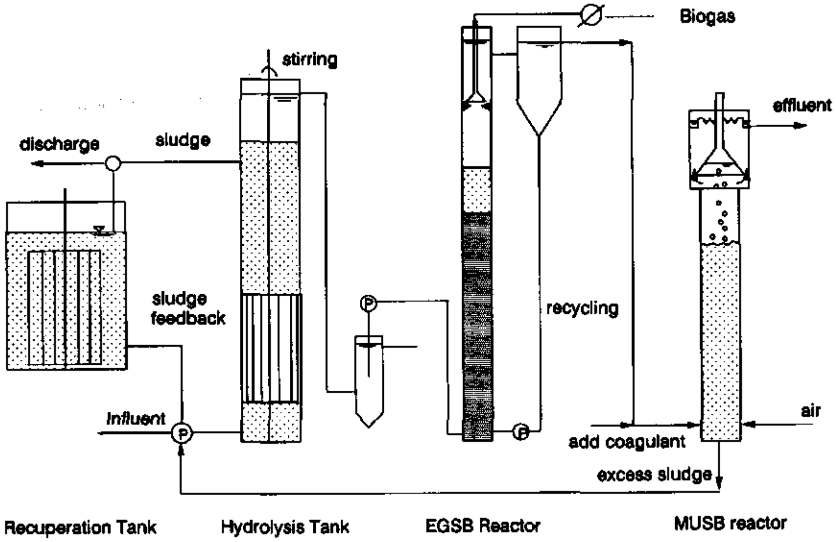

Figure 5 The proposed post treatment process for sewage treatment

CONCLUSIONS

The effluent of the combined HUSB/EGSB sequential process still contains 40 to 60 percent of the coarse and colloidal COD, which neither be readily removed by plain sedimentation nor by additional anaerobic treatment. The experimental results indicate that only 12.4% of the remaining COD can be removed by sedimentation and that for dry weather conditions about 130 mg/L COD remains, which is non-biodegradable or only slowly biodegradable under anaerobic conditions. A significant fraction of the coarse and colloidal COD fraction can be readily removed by a bio-sorption process.

The micro-aerophilic upflow sludge reactor developed for post treatment, provides a high performance at HRT=1.0 h. and at T=13°C, i.e. a removal efficiency of 35% of total COD and 58% of colloidal COD. This MUSB-system looks extremely attractive. By combining the MUSB process with a physio-chemical process, such as chemical coagulation a significantly better final effluent quality very likely can be achieved. Additional research is needed to assess the potentials of such an integrated physio-chemical and biological post treatment processes.

ACKNOWLEDGEMENTS

We gratefully acknowledge the technic support of the following individuals: Last, A.R.M van der, R.E. Roersma, H. Donker, and A. van Amersfoort. chromatography.

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CHAPTER 6

HYDROLYSIS TANK - STABILIZATION POND SYSTEM FOR MUNICIPAL WASTEWATER TREATMENT

Accepted for Published in: <u>Wat. Sci and Technol.</u>

Hydrolysis Reactor - Stabilization Pond System For Municipal Wastewater Treatment

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ABSTRACT

A new integrated sewage treatment process, consisting of a combined hydrolysis reactor - stabilization pond system is proposed. The system aims at reducing land requirements of stabilization ponds and to decrease sediment accumulation in these ponds. The hydrolysis reactor is used as a pretreatment unit to remove suspended solids and to improve the biodegradability of the influent. Aquatic vascular plants are cultivated in ponds to raise the pollutant removal rate and to inhibit the overgrowth of algae. Comparative tests on laboratory scale showed that the new system offers a higher removal efficiency for organic matters and suspended solids, a slower sediment accumulation rate, a lower land occupation and more economic benefits than the conventional preliminary sedimentation tank - stabilization pond system.

KEY WORDS

municipal wastewater, stabilization pond, hydrolysis reactor, sludge sediment, land occupied, aquatic plants

INTRODUCTION

The main disadvantages of the application of a wastewater stabilization pond system are the large land area requirements and the serious sludge accumulation occurring in the ponds operation. The reason for the large land occupation is that a relatively long retention time is needed in order to degrade refractory pollutants present in the wastewater. The sludge accumulation mainly results from the sedimentation of the suspended solids present in the influent and the precipitation of suspended algae growing in the ponds (Iwema et al., 1987).

In order to solve these problems, a new system, i.e. the hydrolysis reactor - stabilization pond system (HTSP) is proposed. The hydrolysis reactor is a modified upflow anaerobic sludge blanket reactor, not equipped with a gas-liquid-solid three phase separator. The hydraulic retention time is three hours and the reactions are mainly limited to hydrolysis and acidification (Wang et al., 1989). The hydrolysis reactor is adopted as the pretreatment unit to replace the conventional preliminary sedimentation tank. Influent suspended solids are captured in the sludge bed present in the reactor and partly liquified here. The biodegradability of the wastewater is improved by the hydrolysis taking place in the reactor. Floating macrophytes are cultivated in the ponds to raise the efficiency of BOD_5, nitrogen and phosphorus removal, to inhibit overgrowth of algal population and to optimize recovery resources by plant harvesting (Reddy and Smith, 1987; Eighmy and Bishop, 1989).

EXPERIMENTAL METHODS AND CONDITIONS

The flow diagram of the HTSP system presented in this study is shown in Figure 1. The conventional sedimentation tank - stabilization pond(STSP) system was used as a control system in the experiment.

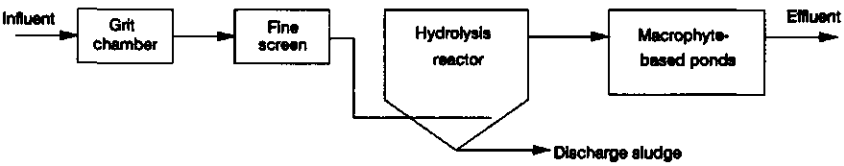

Figure 1 The Hydrolysis Reactor - Stabilization Pond System

The total volume of the hydrolysis reactor was 170 m^3, providing a treatment capacity of 1,500 m^3/d. The hydraulic retention time applied in this reactor was 3 hours. A 170 m^3 primary sedimentation tank was operated under the same conditions as a control. The laboratory stabilization pond consisted of four compartments operated in series, with a total volume of 200 L. The hydraulic retention time applied was 1.5 days for each of the three compartments and was 4 days for the last one. The laboratory ponds maintained at 25°C using a water bath and were operated continuously for a period of four months. Each reactor was named according to the plant type growing in it. In order to evaluate the effect of different water plants on the purification, three types of aquatic vascular plants (Eichhornia crassipes, Azolla imbricata and Alternanthera philoxeroides) ponds were compared with an algal-bacterial ponds. The latter was exposed to sunlight without macrophytes so that a phytoplankton population could develop here. Different combinations were also tested. Experimental results showed that water plants in the first pond are effective in reducing the loading rate and to inhibit the algal overgrowth in the reactor (Wang et al., 1989). In this paper, the research results mainly focus on the macrophyte-based first compartment of the pond. The characteristics of the wastewater used were described in Chapter 3.

Analyses were made on 24 hours composite samples kept in a refrigerator at 4°C. Suspended Solids (SS), BOD_5, COD, total nitrogen, ammonia-nitrogen, total phosphorus, turbidity and Volatile Fatty Acids (VFA) were measured according to standard methods (APHA, 1980). The SS, BOD, COD measurements were performed daily (no BOD for every Tuesday). The quantity of Faecal coliform was determined by multiple-tube fermentation technic (MPN test) and the quantity of Salmonella determined by selective enrichment multiple-tube fermentation technic (MPN test), respectively (APHA, 1980). Ascaris Eggs were measured by a membrane filtration method. The $TCID_{50}$ of Polio I virus was determined by Reed-Muench method (Gong et al., 1991). The measurement of F. coliform, Salmonella, Ascaris Eggs and Polio I virus were all conducted by Chinese Academy of Preventive Medicine. The pH and dissolved oxygen concentration (DO) were measured with a pH-electrode and DO probe.

RESULTS AND DISCUSSIONS

Comprehensive Comparisons

Perhaps the most important advantage of the stabilization pond is that it provides a possibility to make wastewater treatment system becoming a comprehensive utilization system. In order to explore the possibility ways to integrate purification and utilization functions of a wastewater treatment system, three kinds of aquatic plants ponds systems were tested, i.e. Azolla imbricata, Alteranthera philoxeroides and Eichhornia crassipes as well as Algal-bacteria (Control pond) systems. The main experimental results are listed in Table 1.

The results show that the purification of the three aquatic plant ponds were superior to the algal-bacterial pond system. The water surface in these ponds is covered with large quantities of water plants, so that algae growth is inhibited, and the suspended solids in the final effluent become very low (< 10mg/L) and the transparency is high. Therefore the appearance of aquatic plant ponds systems is superior to an algae-bacterial pond system.

Table 1 The results of different aquatic plants systems(4 months operation, T= 25°C)

Pond Type			Azolla imbricata			Alteranthera philoxeroides			Water Hyacinth(Eichhornia crassipes)			Algae-bacterial Control pond		
Pond No.			1	2	3	1	2	3	1	2	3	1	2	3
Removal of pollutants	BOD_5 mg/L	influent	104.7	21.3	8.1	104.7	10.2	4.7	104.7	9.1	2.6	104.7	25.5	19.5
		effluent	21.3	8.1	5.8	10.2	4.7	3.2	9.1	2.6	2.8	25.5	19.5	23.2
	COD mg/L	influent	220.4	91.9	84.5	220.4	95.0	84.5	220.4	124.0	72.4	220.4	143.6	128.8
		effluent	91.9	84.5	63.4	95.0	84.5	73.9	124.0	72.4	71.1	143.6	128.8	110.9
	NH_3-N mg/L	influent	16.9	9.7	6.2	16.9			16.9			16.9		
		effluent	9.7	6.2	2.1	11.1			2.3			6.2		
	T-N mg/L	influent	23.0	10.4	15.2	23.0	12.1	8.1	23.0	4.5	1.5	23.0		
		effluent	10.4	15.2	9.2	12.1	8.1	5.1	4.5	1.5	1.2			
	T-P mg/L	influent	3.7			3.7			3.7	0.4		3.7		
		effluent							0.4					
	Turbidity		13	5	0.5	3	1.5	2				8	11	38
Growth of aquatic plants	Growth rate or growth state		In 1st pond growth proceeds poorly; in 2nd pond growth rate was 6.0g/m^2.d; in 3rd pond growth growth was 7.7g/m^2.d			first two ponds growth proceeds well; in 3rd pond growth slows down			1st pond is growing luxuriantly; growth rate gradually slows down in other ponds the leaves turn yellow			algae become more and more along ponds, concentration is 10 g(dry)/L		
	Roots state		the length is c.a 1.0 cm, the roots rotten in 1st pond			floating aquatic plants, biofilm can grow on the roots			the root system flourishing, the main root length can reach 0.5 m, biofilm attached					
	Sediments		roots come off and settled in the pond			very little sediments in the pond			very little sedi -ments in pond			sediments consist mainly of algae		
	Relative evapotranspiration(free water surface is one)		<1			>1			>1			=1		

From the purification point of view, the Eichhornia crassipes (water hyacinth) pond and Alteranthera philoxeroides pond seem superior to the others in term of BOD removal efficiency. The roots of these plants were well developed, especially for water hyacinth. The roots of plants were attached with a large quantity of biofilm in the first pond, and hence the bacterial concentration was high in this pond. As water hyacinth can grow well under heavy pollution conditions, the plants developed luxuriantly in the first pond. Along with the degradation of the pollutants here, the water quality turned better in the following ponds. As the nutrients become the limiting factor, the plant growth state become worse than in the first pond. The roots of Azolla imbricata were not well developed under heavy polluting conditions and the roots started rottening in the first pond. As a results the roots come off and are poorly develop in the first pond. However, Azolla imbricata developed well in the following ponds. In addition, as Azolla imbricata is a self nitrogen fixing plant, the nitrogen is not a limiting factor for its growing. It is beneficial for multipurpose use of water plants, but it is unfavourable for the nitrogen reduction.

It was observed that the roots of Azolla imbricata came off seriously during the experiments. The total quantity of sediment was estimated at about $1.5kg/m^2.a$ due to the decomposition of the roots. The coming off of roots is a normal phenomenon for Azolla imbricata growth, however, it results in a new source of sediments. From the operational point of view, the stems of Alteranthera philoxeroides can grow vertically only under conditions of living in a group and supporting each other, so it is difficult to apply on an open water surface. Azolla imbricata and Eichhornia crassipes are floating aquatic plants, they are easy to cultivate and reap.

Comprehensive Utilization of Different Ponds System

The experimental results indicate that the oxygen concentration in aquatic ponds remained lower than in the algal-bacteria pond, while it also varied less during the day and night (Table 2). The oxygen concentration increased along the ponds, and in the third pond (retention time 4.5 days) it can satisfy the conditions for fish breeding.

The protein content of Azolla imbricata is higher and rough fibre is less. The plant is small and the roots are short and no further processing for fodder is needed. Furthermore, the economic value is higher than water hyacinth and Alteranthera philoxeroides. Additionally, it was observed that the evapo-transpiration in the ponds of Eichhornia crassipes pond (or Alteranthera philoxeroides pond) exceeded that of the Algal-bacteria pond and Azolla imbricata pond. However, as it concerns a laboratory scale study, various factors could not be examined and therefore reliable quantitative results were not obtained. Further experiments are needed at bigger scale.

From the above results and observations, it is clear that each system has its own characteristics, and it therefore could be beneficial to combine different systems in series in practical applications. If the purification function is emphasized, it looks beneficial to adopt a water hyacinth pond in the main process. If emphasis is put on the economic benefit, water hyacinth is planted in the first pond to improve water quality and Azolla imbricata is cultivated in the following ponds. Depending on the water quality, fish can be bred.

However, the sediments in Azolla imbricata pond should be cleaned up in time, in order to guarantee the effluent quality. The following experiments, deal with investigations only with the water hyacinth pond.

Table 2 Oxygen concentration variation along with time and pond

Pond Type	Pond No.	Maximum value DO(mg/L)	Time	Minimum value DO(mg/L)	Time	Variation range(mg/L)
Azolla imbricata pond	1	2.3	17:00	0.8	19:00-5:00	1.5
	2	2.6	17:00	1.3	21:00-5:00	1.3
	3	2.3	17:00	1.6	21:00-5:00	0.7
Alteranthera philoxeroides pond	1	1.7	14:00	0.8	21:00-5:00	0.9
	2	4.6	17:00	0.9	21:00-5:00	3.7
	3	3.7	17:00	1.2	23:00-5:00	2.5
Algal bacteria pond	1	14.0*	14:00	0.9	23:00-5:00	13.1
	2	6.8	14:00	1.8	5:00	5.0
	3	29.6*	14-19	5.8	5:00	23.8

*: supersaturated with oxygen concentration(DO)

Removal of Organic Pollutants

Assessed BOD_5 removal efficiencies in the HTSP system and in the control(primary settling tank + pond (STSP) system) are shown in Table 3 (average results of four months operation). At a hydraulic retention time of 1.5 days, the effluent BOD_5 of the first pond of the HTSP system was 14.7 mg/L corresponding to a removal efficiency of 91.0%. In the control system, however, the effluent BOD_5 was 40.1 mg/L resulting in only 75.2% removal efficiency. When the retention time was prolonged to 4.5 days, the total BOD_5 removal rate in the new system came up to 98.2%, while it amounted to 80.8% in the control system.

Table 3 Comparison of BOD Removal Efficiencies in the Two Systems

	*HTSP System				STSP System			
Parameters	hydrolysis reactor	stabilization pond 1	2	3	settling tank	stabilization pond 1	2	3
HRT(day)	3.0**	1.5	1.5	1.5	3.0 h.	1.5	1.5	1.5
Influent	161.5	104.7	9.1	2.6	161.5	121.0	40.1	32.0
Effluent	104.7	9.1	2.6	2.8	121.0	40.1	32.0	31.0
Removal rate	35.2%	91.0%	96.7%	98.2%	25.0%	75.2%	80.2%	80.8%

(Units: mg/L); *: the data used here, water hyacinth was planted in stabilization pond; **: hours

Table 4 shows the water quality of the influent and the effluent of the hydrolysis reactor. There pollutants, such as COD, BOD and SS were removed, e.g. the BOD for 35%, whereas also the biodegradability of effluent of the hydrolysis reactor was better than for the influent. The results revealed that the BOD_5/COD and BOD_5/BOD_{20} ratios increased from

0.37 and 0.56 for the influent to 0.48 and 0.79 for the effluent. The volatile organic acids in the effluent were approximately. 40% higher compared to the influent. This has been explained already in Chapter 3.

Table 4 The Hydrolysis Reactor Effluent Characteristic Changes

Sample	BOD_5 mg/L	COD mg/L	SS mg/L	VFA mg/L	BOD_5/COD	BOD_5/BOD_{20}	CODs/CODt
Influent	161.5	437.6	277.4	50.0	0.37	0.56	0.51
Effluent	104.7	220.4	45.3	120.0	0.48	0.79	0.78

*: Volatile Fatty Acid as Acetic Acid concentration, CODs and CODt as soluble and total COD, respectively

Table 5 Using CG-MC for quantitative analysis the effluent of the different processes

No	Molecular	Raw Sewage	Pre-treatment ST	Pre-treatment HT	Post-treatment SPS
1	$C_2H_4OCl_2$	---	---	++	---
2	$C_6H_{10}O$	++	++	---	---
3	$C_3H_5Cl_3$	++	++	---	---
4	$C_6H_{12}O_2$	++	++	---	---
5	$C_3H_5CL_2Br$	---	---	++	---
6	$C_4H_8O_2$	++	---	++	---
7	$C_3H_4Cl_2$	---	---	++	---
8	$C_5H_{10}O$	---	++	++	---
9	$C_6H_{12}O_2$	++	++	++	---
10	$C_3H_4O_2ClC_2H_6$	++	---	++	---
11	$C_3H_6CL_2O$	++	---	---	---
12	$C_3H_4CCl_2$	---	---	++	---
13	$C_2H_3O_2Cl$	---	---	++	---
14	$C_7H_{10}O$	++	++	---	---
15	C_2H_5OCl	++	---	++	---
16	$C_3H_4O_2$	---	---	++	---
17	$C_2H_5O_2N$	---	---	++	---
18	$C_6H_{12}O_2$	---	---	++	---
19	$C_4H_6OCl_2$	---	---	++	---
20	$C_6H_{12}OCL_2$	---	---	++	---
21	$C_5H_{10}O_2$	---	++	---	---
22	$C_6H_{10}O$	---	---	++	---
23	$C_6H_{12}S$	---	---	++	---
24	$C_6H_{11}ON$	---	---	++	---
25	$C_5H_{11}N$	---	---	++	---
26	$C_6H_{14}O$	++	---	++	---
27	$C_6H_{11}O_2Cl$	++	---	---	---
28	$C_3H_6O_3$	---	---	++	---
29	C_5H_8BrF	---	---	++	---
30	C_3H_6ClBr	---	---	++	---

++: The compound is detected and integrated area is exceeds than 1000; ---: The compounds is not detected; ST: Primary Settling tank; HT: hydrolysis reactor; SPS: Stabilization pond system

Results of Chromatographic - Mass Spectrographic (CG/MS) analyses made on the influent and effluent of the hydrolysis reactor seem to confirm this (in this thesis, Chapter 3). Table 5 lists all C_6 compounds found in the solution before and after having been exposed to the hydrolysis reactor and HTSP systems. It can be seen C_6 compounds increased from 12 in the influent to 24 in the effluent of hydrolysis reactor. On the other hand, the compounds in the effluent of the primary sedimentation tank remained almost unchanged. All of these C_6 compounds were easily removed in a stabilization pond post treatment system (Table 5).

In the aquatic macrophyte-based pond, particularly in the Eichhornia pond, it was observed that a large amount of microorganisms gathered on/and adhered to the plant root structure. The longest main root of Eichhornia was about 0.5 meter. A substantial amount of biofilm was found in the root zones in the first pond. These epiphytic organisms, together with plankton, constituted an abundant and diverse community, so that the pollutant removal efficiency increased.

Removal of Pathogens

The fate of pathogenic bacteria and eggs of Ascaris Lumbricoides were examined via analyses in influent and effluent samples of the hydrolysis reactor and the primary sedimentation tank. The data in Table 6 reveal that the hydrolysis reactor removed eggs of Ascaris fairly effectively, but the removal of pathogenic bacteria only amount to 76 to 78%.

Figure 2 shows the distribution of coliform and eggs of Ascaris, together with the sludge concentration over the height of the hydrolysis reactor. The distribution pattern of the Ascarid eggs was very similar to that of sludge. Only very few Ascarid eggs could be detected in the effluent. However, apparently there doesn't exist a relation between the distribution of sludge and that of coliform in the hydrolysis reactor. As the Ascarid eggs are distinctly bigger than the coliform, it can be concluded that entrapment in and/or filtration by the sludge blanket are the main mechanisms for the removal of pathogens in the hydrolysis reactor.

Table 6 Removal of Pathogens By Hydrolysis Reactor and Primary Sedimentation Tank

Sample		F. Coliform MPN/100mL	Salmonella MPN/100mL	Ascaris Eggs No./L
	Influent	$2.3*10^8$	$5.1*10^5$	69
Hydrolysis reactor	Effluent	$5.0*10^7$	$1.2*10^5$	3
	Removal(%)	78.3%	76.5%	95.7%
Settling tank	Effluent	$2.3*10^8$	$2.4*10^5$	26
	Removal(%)	0	52.9%	62.5%

The research by Pearson et al. (1987) indicated that the efficiency of pathogen removal in ponds is directly proportional to the HRT and light density. In this experiment, the algal-bacterial ponds were more effective than the aquatic macrophyte-covered ponds in removing pathogenic bacteria (Table 7). Very likely this can be attributed to the fact the covering plants partially shut out the sunlight. Nevertheless, the macrophyte-based ponds still exerted a high removal rate. Other factors such as predation by zooplankton may have played an important role in pathogen removal in stabilization pond.

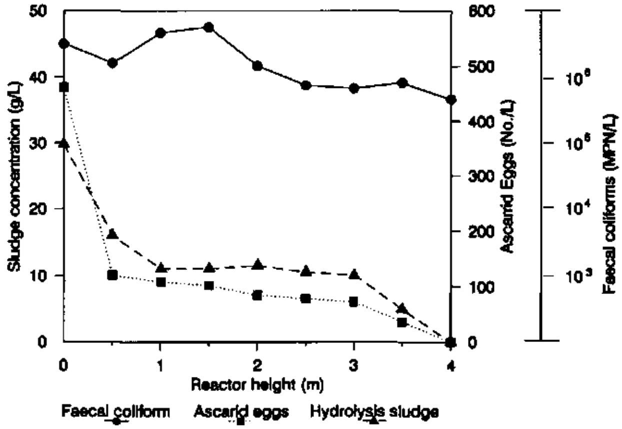

Figure 2 Profiles of Faecal Coliform, Ascarid Eggs and hydrolysis sludge over reactor

In order to evaluate virus removal in macrophyte-based ponds, a vaccine of Polio I virus was used as the viral tracer in a dynamic tracing experiment and added continuously to the influent for a period of ten days. The samples were examined at tenth day. The average density of the virus in the influent was $1.26*10^8$ TCID50/L. The laboratory scale macrophyte-based pond removed 99.99% of the vaccine of Poliovirus (Table 8).

Table 7 Removal of Pathogenic Bacteria in Stabilization Ponds

Pathogens	Hydrolysis Reactor Effluent	Macrophyte-based ponds 1	Macrophyte-based ponds 2	Macrophyte-based ponds 3	Algal-bacterial ponds 1	Algal-bacterial ponds 2	Algal-bacterial ponds 3
F.Coliform(MPN/100mL)	$5.0*10^7$	$2.3*10^6$	$1.7*10^5$	$1.4*10^4$	$2.3*10^5$	$2.3*10^3$	900
Removal Efficiency(%)		95.4	99.7	99.97	99.5	99.99	99.998
Salmonella(MPN/100mL)	$2.9*10^4$	$2.8*10^2$	$1.0*10^2$	$2.2*10^1$	$2.3*10^2$	$1.4*10^1$	95
Removal Efficiency(%)		99.0	99.6	99.9	99.2	99.95	99.96

The experiments reported here, demonstrate that the HTSP system is effective in removing pathogens even though its hydraulic retention time is much shorter than that of a conventional stabilization pond. The results provided further evidence for the views expressed by Davis and Gloyna (1972) that a preliminary anaerobic treatment in front of facultative treatment will result in higher die-off rates of pathogens than facultative and maturation ponds series did.

Table 8 Enumeration of Vaccine of Polio I virus in Dynamic Tracing Test

Sample	influent	pond 1	pond 2	pond 3	pond 4
$TCID_{50}/L$	$1.26*10^8$	$2.26*10^6$	$3.88*10^5$	$9.97*10^5$	$7.29*10^3$

Accumulation of Sediment

The sediment present in a stabilization pond mainly originates from sedimentation of suspended solids carried by the influent and produced by overgrowth of algae in ponds. In the HTSP system, the amount of influent SS is significantly reduced by the hydrolysis reactor and the growth of algae is inhibited by cultivation of macrophytes in ponds.

The hydrolysis reactor in our investigations removed approximately 84 percent of the suspended solids from the influent. Neglecting the amount of anaerobic sludge growth, it can be estimated on the basis of the mass balance that approximately 48 percent of the influent suspended solids was solubilized and hydrolysed in the hydrolysis reactor(see Chapter 3, this thesis). The primary sedimentation reactor at the average removed only 40 percent of the influent suspended solids in the control test. Moreover, the amount of sludge discharged from the sedimentation reactor was higher than that from the hydrolysis reactor, because liquefaction and hydrolysis here reduced the amount of sludge substantially.

An algae precipitation test was conducted using an one litre graduated cylinder. It appeared that an algal standing crop settles for 5 percent per day in dark condition. The average algal concentration was less than 0.5 mg/L (dry weight converted from the chlorophyll, a concentration in samples) in the macrophyte-based ponds, while it was over 16 mg/L in the algal-bacterial reactors. The quantity of algal sediment in the HTSP system was 97% less than that in a conventional stabilization pond.

Table 9 Sediment Accumulation in Two Systems

Parameters	Hydrolysis Reactor-Stabilization Pond			Settling Reactor-Stabilization Pond		
	pond1	pond2	pond3	pond1	pond2	pond3
Depth of Sediment(cm)	0.35	0.20	0.15	0.70	0.20	0.15
Amount of Sediment(g/m^2)	230.7	122.2	86.4	455.0	128.2	89.2
VSS/TSS	0.64	0.59	0.58	0.58	0.61	0.60

An experiment was conducted to assess sediment accumulation in the ponds of the two systems (Table 8). The results show significant differences between the two systems particularly for the first ponds. The sediment was collected and sampled using glass dishes (5 cm high and 71 cm^2 surface area), which were put on the bottom of each reactor (two dishes for each reactor) at the beginning and were taken out at the end of the test one month later. The data in Table 9 show that sediment accumulation in the first pond of the HTSP system was 49.3% less than that of in the sedimentation tank-stabilization system. It is obvious that the different efficiency or suspended solids removal by the pretreatment units of the two systems explain the differences in sediment accumulation in the first ponds. The difference in amount of sludge in the other ponds between the two systems can be attributed to algal sedimentation so that the VSS/TSS ratio in the last two ponds of the algal-bacterial system slightly exceeded that of the macrophyte-based system.

CONCLUSIONS

The results of the presented investigations indicate that the hydrolysis reactor-stabilization pond system represents an attractive low-cost advanced treatment process for municipal wastewater and organic industrial wastewaters.

The land occupation by HTSP system is approximately 50 percent less than that by conventional stabilization pond. The reason for this can be found in a substantial improvement of the biodegradability of the pollutants remaining in the wastewater after the hydrolysis step, so that the removal efficiency in the ponds is significantly enhanced.

The hydrolysis reactor also reduces suspended solids in influent quite substantially while in the macrophyte-based ponds overgrowth of algae population hardly will occur, so that accumulation of sediments in the HTSP system become approximately is 50 percent less than that in conventional preliminary sedimentation reactor-stabilization pond system.

It looks recommendable to apply the different ponds in a sequential configuration. Generally a water hyacinth pond is more favourable for water pollution control. Therefore in the first pond water hyacinth is cultivated for pollutants degradation. In order to maintain a sufficient high oxygen concentration for fish breeding, an algae pond is adopted to raise the oxygen concentration in the next ponds. Plants, such as Azolla imbricata, can be planted in the last pond both for economic purposes and for effluent SS control as well.

An optimal combination of anaerobic and aerobic processes in this system can make it effective in removing pathogens and some refractory organic pollutants. Regarding its big advantages, the use of the HTSP system is highly recommended over a conventional stabilization pond and secondary biological wastewater treatment process, wherever climate and land availability permit this.

ACKNOWLEDGEMENTS

We thank Professor Zheng Yuanjing and Dr. Chang Chong-hua for very valuable suggestions. The cooperation of Mr. Wang Junqi from the Chinese Academy of Preventive Medicine is acknowledged. The research was funded by China EPA.

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CHAPTER 7

THE RESEARCH AND APPLICATION OF ANAEROBIC(HYDROLYSIS)-AEROBIC BIOLOGICAL PROCESS FOR MUNICIPAL WASTEWATER TREATMENT IN CHINA

Parts of contents accepted for publication in: Wat. Sci. Technol.
and published in: Proc. Int. Symp. Sani. and Environ. Eng.

Research and Application of Anaerobic (Hydrolysis)-Aerobic Biological Process For Municipal Wastewater Treatment in China

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ABSTRACT

A promising concept, recently developed, is the combined anaerobic - aerobic treatment process in which the anaerobic step consists of a hydrolysis (HUSB) reactor. This HUSB reactor is a modified UASB reactor, operated at such a short retention time that mainly hydrolysis and acidification will occur. The suspended solids removal efficiency of this system is high. Using this system as pretreatment step, also the hydraulic retention time of the aeration tank and of the stabilization pond post treatment units can be shortened, as a result the increased concentration of the easy biodegradable matter and the decreased concentration of organic pollutant in the effluent of the HUSB reactor. This paper presents results of the investigation and applications of the anaerobic (hydrolysis) - aerobic biological process for municipal wastewater treatment in China. The design and operation problems both of the HUSB reactor and post treatment units are discussed.

KEY WORDS

Municipal Wastewater, HUSB reactor, Anaerobic and Aerobic treatment, UASB reactor, Activated Sludge process, Stabilization pond

INTRODUCTION

The conventional activated sludge process suffers from high capital investment costs, a high energy consumption and high maintenance costs. As a result the implementation of conventional methods for water pollution control is seriously hampered in China (Research group, 1991). On the other hand, in recent years, low cost treatment methods like wastewater stabilization pond were implemented in China (Wang et al., 1992). However, these methods also suffer from some disadvantages in their application, such as their large land requirements, long retention time needed and the considerable sludge accumulation in the system after several years of operation, and the poor effluent quality in cold climates. Facing the above problems, the hydrolysis biological pre-treatment process has been developed (Wang et al., 1988).

The new HUSB-process is considered as a serious and attractive option by research and design institutes as well as by users. The process presently is used for domestic wastewater treatment, and also applied for various types of refractory industrial wastewaters, such as that from the textile industry, from coke gasification and paper and the pulp wastewater (Wang et al., 1991, Wang and Jin, 1991). In this paper the main results of laboratory and pilot scale experiments are described. The full scale application of the HUSB reactor - activated sludge system and the HUSB reactor - stabilization pond system will be discussed along with some typical design aspects and operation problems.

LABORATORY AND PILOT SCALE EXPERIMENTS AND RESULTS

Preliminary Study

Laboratory and pilot scale experiments of HUSB reactor - activated sludge (HRAS) process and HUSB reactor - stabilization pond (HRSP) process were conducted at Gaobeidian (**GBD**) municipal wastewater treatment works in southeast suburban of Beijing. The first stage experiments were conducted from 1983 to 1984 (Liu et al., 1984), viz. a laboratory experiment using a 37 L volume a UASB-reactor has a three phase separator for sewage treatment at HRT = 8 hr. and at ambient temperature conditions (9-23°C). It was observed that the removal rate of COD, BOD_5 and suspended solids (SS) were in the range of 50 - 70%, 60 - 80% and 70 -80%, respectively. Despite the HRT was long enough, the gas production rate remained lower than 0.02 m^3/m^3.d. It was also found that the anaerobic reactor removal efficiency was insufficient to satisfy the discharge standards. Aerobic post treatment therefore still is needed. Moreover, the required HRT of the UASB reactor was too long to compete with the conventional activated sludge process, even though the process offered some advantages of lower operation cost and energy consumption.

In order to solve above problems, in the second stage laboratory experiments period the operational mode of the UASB reactor was modified to a partially anaerobic treatment, i.e. mainly focusing on the hydrolysis and acidification stages of the anaerobic reaction (Chapter 3). In the pilot scale experiment conducted at **GBD** pilot plant a HUSB reactor was constructed by modifying one of the original multi-hopped horizontal flow primary sedimentation tank. The parallel system consisted of a primary settling tank-activated sludge (PSAS) process of the same dimensions. It served as control system. Compared to the conventional process, the HUSB reactor was used instead of the traditional primary settling tank and the sludge digestion system was cancelled from the activated sludge plant.

The new system provides a good quality final effluent compared with the control system. The sewage BOD_5 concentration of the final effluent amounted to 6.6 mg/L and the average yearly COD value of the effluent was below 100 mg/L (Wang and Zheng, 1989). On the other hand the effluent COD value in the control system, even at a HRT of aeration tank of 8 hours, both in the coarse and fine bubble aeration systems, were as high as 150mg/L and 120mg/L respectively. Apparently the volume of the aeration tank used in the new process can be reduced largely compared to that of the conventional process. Moreover also the energy consumption can be reduced as the oxygen demand of the new system is considerably lower (Chapter 3). In order to prevent sludge bulking problems, a 4 hours hydraulic retention time was adopted to reduce organic loading rate and to alleviate oxygen deficiency in the aeration tank.

The Experimental Results of Stabilization Pond Post Treatment

Based on the results of the anaerobic (hydrolysis) - aerobic system, stabilization ponds were investigated as an alternative post treatment method. Although it is well known that the traditional stabilization ponds system suffer from some drawbacks, such as long retention time, high land requirements, sludge silting and low efficiency in cold regions and/or during winter time, these systems might become an attractive alternative for conventional systems, in the combined use with a HUSB-pretreatment step. The pilot scale experiment was conducted at **GBD** from 1986-1988. The results of a comprehensive elaboration of the data of obtained with the combined HUSB reactor - stabilisation pond system (HTSP) and the primary settling tank-stabilization pond system (STSP) as well are shown in Table 1.

Figure 1 shows the course of the organic pollutant concentration in terms of COD and BOD in the ponds (at different HRT) for above two systems. The advantage of the use of a HUSB reactor is obvious. The influent concentration is reduced to lower value by the HUSB reactor than that of in the conventional settler system and as a result for the same retention time in the pond the effluent quality is better. From Figure 1 the parameters Kh and Kp of Equation 1 for the HTSP system and PTST system, respectively can be obtained.

$$C_4 = \frac{C_o}{(1 + kt)^4} \qquad (1)$$

This equation applies for a serial pond system, C_o and C_4 are the influent and 4th pond effluent organic pollutant concentration, and K_h and K_p are the first order reaction constants for a HTSP system and a PTST system, respectively (d^{-1}); From Figure 1 the estimated values for K are K_h=0.436d^{-1} and K_p=0.251d^{-1}.

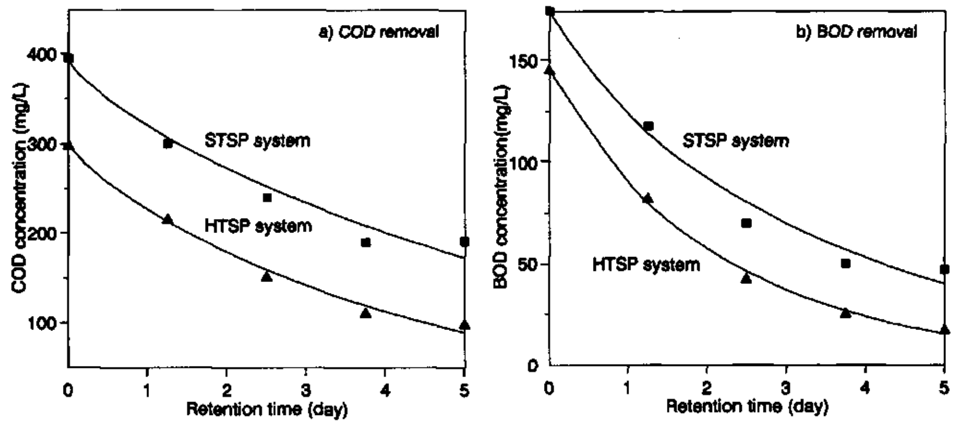

Figure 1 The COD (a) and BOD(b) concentration changes in different systems

Table 1 provides the results of a parallel comparison test of the two pre-treatment methods at a HRT of 5 days in the stabilization pond at an average water temperature of 20 °C and under conditions where Eichhornia crassipes were cultivated in the stabilization ponds. The experimental results show that with an effluent of COD 97.7 mg/L, BOD_5 19.7 mg/L and SS 8.3 mg/L the HTSP system, satisfies the secondary treatment effluent quality standards in China. However for the STSP system, the COD of the final effluent amounts to 171.7 mg/L and BOD_5 to 43.6 mg/L, which both are too high. In order to get the same effluent quality as that of the new system, the conventional STSP system should be operated at a HRT of at least 10 days (Eq. 1). The effect of a hydrolysis pre-treatment step therefore is quite substantial, i.e the required HRT of the HTSP system is 50% less than that of the STSP system. The experimental results therefore indicate that the problems of the very long retention time, high land occupation and also of the high accumulation of sludge (this thesis, Chapter 6) as prevailing in the traditional stabilization ponds can be solved.

Table 1 Results of different pre-treatment methods(a period of 3/87 to 10/88)

		HTSP System		STSP System	
	influent	HUSB reactor	Pond effluent	Settling tank	Pond effluent
COD (mg/L)	492.3	304.9	97.7	393.1	171.7
CODre (%)		38.1	80.2	20.2	65.1
BOD_5(mg/L)	193.5	145.7	19.5	156.8	43.6
BODre (%)		24.6	90.0	19.0	77.5
SS (mg/L)	204.1	45.6	8.3	93.4	12.2
SSre (%)		77.7	95.9	54.2	94.0

THE OPERATION PROBLEMS OF POST TREATMENT UNITS

The main technical advantages and experimental results of the post-treatment systems investigated were already presented above. However, the new process still suffers from some problems, such as the obvious low COD removal efficiency achieved in the first step and also in the HUSB + pond system in winter time. It was also noticed that serious bulking problems of the activated sludge occurred in the aeration tank following the HUSB reactor.

Activated Sludge Bulking Problems

Serious bulking problems were observed in the aeration tanks of both the laboratory and the pilot scale experiments (Figure 2a and b). For that reason some control measures were investigated, such as changing the flow pattern from a completely mixed to a plug flow operation, application of pre-aeration to anaerobic effluent, and addition of anaerobic sludge. These measures were demonstrated to be effective to reduce bulking problems in some situations (Rensink, 1975, Chudoba, 1985). However, the experimental results obtained in our investigations revealed that above methods were inadequate for the situation prevailing in the systems investigated (Figure 2).

The bulking problem appearing in present systems is quite different from that caused by a low loading rate (substrate limiting conditions). The aeration times applied in the laboratory and pilot scale experiments were 2.5 and 4 hours respectively, which is relatively short, and the space loading rates of the aeration tanks for the laboratory and pilot experiment were in the range 0.65-0.85 kgBOD/kgMLSS.d, which is rather high. Similar bulking phenomena were found in the treatment of the effluent of an one step UASB reactor by Men. Due to bulking of the sludge the experiment had to be terminated at relatively a high loading rate (0.3-0.6kgCOD/kgMLSS.d, HRT=2.5h.) of the aeration tank. In order to prevent bulking of the sludge, the application of a system consisting of set of reactors operated in series at low loading rate (0.12kg COD/kgMLSS.d, HRT=10.5 h.) was investigated. It appeared that the sludge settling characteristics then became excellent(van Buuren, 1991). Although the same control method can be applied to prevent sludge bulking for the situation studied here, this approach does not offer any clear advantage in retention time and energy consumption compared to the conventional process.

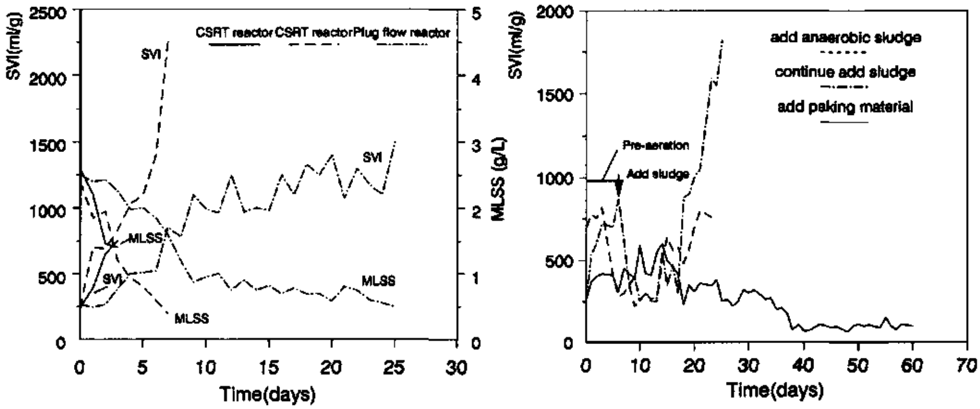

Figure 2 Sludge bulking and its control in laboratory(a) and pilot scale experiments(b)

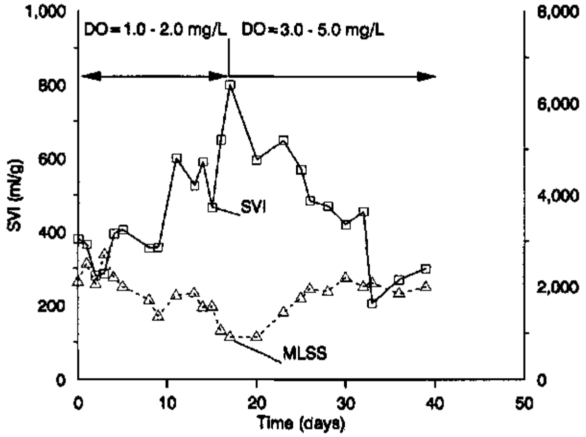

Figure 3 DO concentration effects on SVI (average concentration over whole reactor)

The real cause of sludge bulking occurring with anaerobic effluent has not been cleared up yet, though filamentous sulphide oxidising microorganism Thiothrix was observed in bulking sludge (van Buuren, 1991, Alam, 1991). We therefore conducted a more detailed study of the bulking problem occurring in the treatment of HUSB effluent. The results of the experiments indicate that the oxygen utilization rate is accelerated in the aeration tank due to acidified effluent of the HUSB reactor and the presence of readily biodegradable soluble substrate. The oxygen supply rate may become lower than oxygen utilization rate, especially at the head end of the aeration tank. In this situation, the supply of oxygen will be a limiting factor, even when the DO concentration is in the range of 1.0-2.0 mg/L in the bulk of the liquid. It was observed that the DO concentration at the head end of the aeration tank was almost zero. It appeared that when the DO concentration can be maintained above 2.0 mg/L at the head of the aeration tank (or over 3.0mg/L in the aeration tank), sludge bulking can be effectively controlled (Figure 3). It also appeared that problems with sludge bulking can be prevented by hanging fibrous packing medium in the aeration tank occupying 10-15% of the volume of total aeration tank at the head of tank (Figure 2).

Operation Problems of Stabilization Pond

The main operational problem of a stabilization pond is its performance in winter time. The winter period in Beijing lasts for 3 - 4 months and the lowest air temperature is about -12°C. In this study, we used a large plastic shed to protect the stabilization pond from the low winter temperature. The experiments showed that the shed effectively absorbed solar energy and decreased the heat loss of the water in stabilization pond. Air temperature in the shed was usually 15°C higher than outside (Figure 4), and the effluent temperature of the HUSB reactor was 16°C. At HRT=15 days, the water temperature of the first pond could be maintained at 11°C and the effluent temperature of the last pond over 5°C using the shed. Under these conditions the effluent quality still met secondary treatment standard (Table 2).

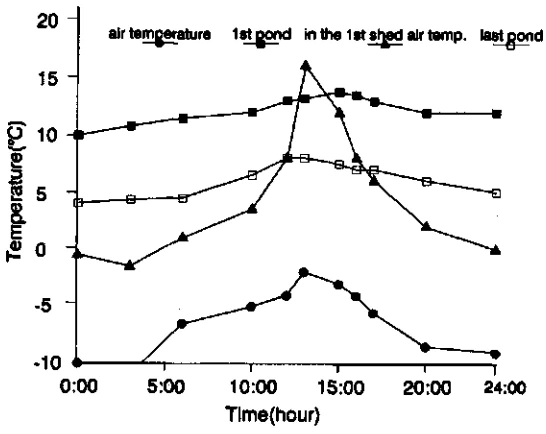

Figure 4 The temperature changes during 24 hours period(24/2/1987)

On the other hand, according to a cost-effective analysis, there is no advantage on capital investment over a conventional secondary treatment system when operating a pond at 15 days is covered with a plastic shed (Wang et al., 1992).

Table 2 Performance of the stabilization pond with plastic shed in winter

Item		HUSB reactor	Stabilization pond	Whole system
Water temp.		18°C	5-12°C	
HTR		3 h.	15 d.	
COD Load		$3.9kg/m^3.d$	202kg/ha.d	
COD	Influent, mg/L	457.3	303.9	457.3
	Effluent, mg/L	303.9	89.4	89.4
	Removal rate,%	33.5	70.6	80.5
BOD_5	Influent, mg/L	189.2	145.3	189.2
	Effluent, mg/L	145.3	17.8	17.8
	Removal rate,%	23.2	87.8	90.6
SS	Influent, mg/L	204.8	45.5	204.5
	Effluent, mg/L	45.5	14.4	14.4
	Removal rate,%	72.7	73.4	93.0

In order to be able to decrease the HRT of the stabilization pond, the use of a contact oxidation tank was investigated as post-treatment of the effluent of the HUSB reactor for lowering the influent organic load of the stabilization pond. Table 3 provides the operational results of the contact oxidation tank at two different loads and at the system when combined with the stabilization ponds.From Table 3,it can be seen that the water temperature decreases slightly (loss 1-1.5℃) in the contact oxidation tank, and that the treatment efficiency remains satisfactory at higher loading rates (HRT=0.5h.). The effluent quality can meet the secondary levels following further treatment in the stabilization pond at HRT=5 days.

Table 3 Operation results of an oxidation tank-pond process(average results at steady state)

	Oxidation tank influent			Oxidation tank effluent			Stabilization pond effluent(HRT=5 days)			
	T(°C)	BOD_5	COD	T(°C)	BOD_5	COD	T(°C)	BOD_5	COD	SS
HRT=1.0h. load=$3.6kgBOD_5/m^3.d$	18	150	287	16	53	120	11	15	102	9
HRT=0.5h. load=$7.2kgBOD_5/m^3.d$	18	150	280	17	60	179	11	28	98	8

Above results show that using the HUSB reactor-contact oxidation tank-stabilization pond system can meet the secondary discharge standards in North China. In order to lower capital investment and to simplify the treatment process, the HUSB reactor and contact oxidation tank possibly could be combined, i.e. by increasing the hydraulic retention time of the HUSB reactor with 0.5 -1.0 hour. During summer time, the contact oxidation tank and the HUSB reactor can be operated in parallel as an HUSB reactor, while in winter, they can be operated in series separately. At the same time, a settling area is set at the head of first stabilization pond, so that only an air-blower house is added, and the increase of capital investment is less (Figure 5).

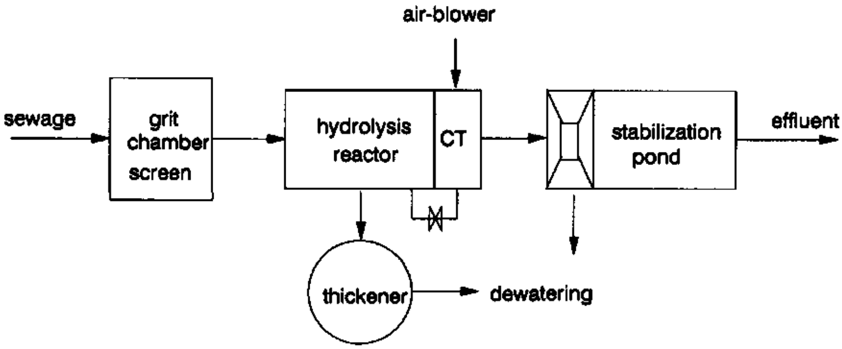

Figure 5 An extended HTSP system(CT: Oxidation contact tank)

THE APPLICATION OF THE NEW PROCESS IN CHINA

Because the effluent quality of the anaerobic step does not meet discharge standards, some kind of post treatment is needed. For this purpose different types of post treatment processes have been employed, such as, the activated sludge process, trickling filter, contact oxidation tanks, land infiltration as well as stabilization ponds (Wang et al., 1987, Xu et al., 1987 and Wang et al., 1991 and 1992). The activated sludge process and stabilization pond are popular systems in China. This especially is true for the stabilization pond process, because it is simple in design and operation, and the investment costs are low which is a prerequisite for the developing countries.

Design Aspects of the HUSB reactor

Preliminary Treatment Device and the Inlet System Based on the pilot and full scale plant experiences, it was concluded that one inlet per 1 - 2 m^2 is suffices for uniform distribution of the incoming sewage over the bottom of the HUSB reactor, though one inlet point per one square meter is strongly recommended, especially for cold weather conditions. As hydraulic mixing is sufficient, the velocity at the inlet points is not an important design parameter. Otherwise, the diameter of inlet nozzles is important for preventing clogging, so that a fine screen and grit chamber as preliminary treatment devices are required. Clogging of inlet systems has never been found, at inlet diameter exceeding 20 mm both in pilot and full scale plants.

Design of a HUSB reactor An average hydraulic retention time of 2.5 - 3.0 hours looks recommendable for a HUSB reactor. The recommended average upflow velocity (V_p) between 1.0 to 1.5 m/h. With these values the height(H) and area of the reactor(A) can be determined:

$$A = Q/V_p;$$
$$H = V/A$$

The shape of the reactor usually is rectangular. The height of the reactor is 4 - 6 meters and the maximum upflow velocity should be less than 2.5 m/h. (the peak flow period should not be longer than 3 hours)

Effluent and Sludge Discharge Devices The objective of the effluent device is to collect the treated wastewater uniformly at top of the HUSB reactor. Horizontal gutters with V-notches arranged at regular distances are suggested. In order to collect effluent uniformly and to avoid sludge washout, the recommended hydraulic loading rate at the V-notch is between 1.5 to 3.0L/s.m. The sludge bed level should be maintained 1.0 - 1.5 m beneath the water surface for safely operation, as the large fluctuation of influent of the sewage. The removal mechanism of organic pollutants and SS in the HUSB reactor is mainly based on entrapment. As the sludge at the bottom of the reactor is still unstable, the discharge points of sludge should be placed on the mid-upper part of the sludge bed, i.e. 2.0-2.5 m below water surface. A typical design of HUSB reactor according to above criteria is shown in Figure 6.

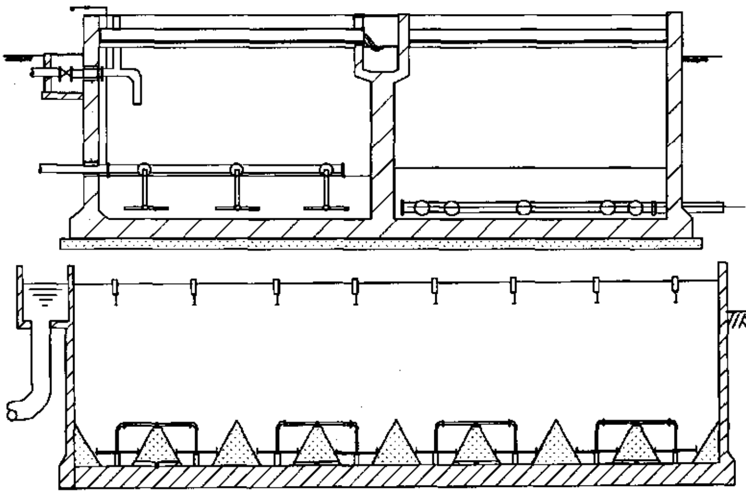

Figure 6 The design drawing of a HUSB reactor of Miyun wastewater treatment plant

Design Aspects of the Stabilization Pond

Recommended design figures for areas required for stabilization ponds in different climates were extracted from the experiments are the following:

1) For tropical zones (average water temperature > 20°C), the retention times of the HUSB reactor and stabilization ponds should be 2.5-3.0 hr. and 4-8 days respectively

and the depth of the pond 1.0 -1.5 metre. In a plug flow pond system, the use of 3-5 ponds is recommended. In the first pond and the last pond some aquatic plants should be grown in order to reduce the loading rate and to control effluent suspended solids due to algae growth.

2) In sub-tropical zones (average wastewater temperature > 5°C in winter), the retention time of stabilization ponds should be prolonged to 15 days. The stabilization ponds can be designed in different ways, i.e. during the summer time partially as stabilization ponds operated at 5 days retention time while using the other part as fish ponds, and in the winter time they all should be used as stabilization ponds .

3) In moderate climate zones (average water temperature can drop down to 0°C in winter), the process shown schematically in Figure 5 is recommended.

For the winter season the retention time of the HUSB reactor is prolonged to 3.5-4.0 h, of which 0.5 to 1.0 hour is applied as contact oxidization tank. The head part of the stabilization pond is designed for sludge settling and storage for a period of about one or two months during winter time. The other parameters are the same as above.

Applications with Activated Sludge Process Post Treatment

Start Up of Full Scale Plant of Changji According to the planning program of Changji county, the capacity of Changji wastewater plant should be expanded from 4,000 m^3/d to 15,000 m^3/d. The conventional activated sludge process of the original plant was changed to a combined process of hydrolysis and aerobic treatment using the activated sludge process operated in series. The construction was finished in 1992 with four 630 m^3 HUSB reactors. The wastewater flow rate amounted up to 12,000 - 15,000 m^3/d. The wastewater consisted for 70% of industrial wastewater mainly from paper and pulp industrial wastewater. The influent COD, BOD_5 and SS values were 923 mg/L, 240 mg/L and 120 mg/L respectively.

In the first period two HUSB reactors were started without seeding at the maximum loading capacity of 12,000 m^3/d and at an HRT of 2.5 hours. The operation results of this period (one month) are listed in Table 4. Because of the applied short retention time and the high upflow velocity, the amount of sludge in the reactor increased very slowly. After one month of operation, little if any sludge was accumulated in the reactors and the COD removal efficiency amounted to only 10 - 20%.

Table 4 The results of the HUSB reactors starting up period of Changji wastewater plant (with and without seeding and paper mill wastewater)

	without seeding			seeded			seeded		
	with paper mill wastewater						without paper mill wastewater		
Items	inlet	outlet	E%	inlet	outlet	E%	inlet	outlet	E%
COD	924	762	17.5%	559	431	22.9%	336	163	51.6%
BOD	240	200	16.2%				157	82	48.0%
SS	118	29	75.4%	131	11.4	91.3%	78	9	88.5%

In the second stage of the start up, the retention time was prolonged from 2.5 to 4 hours (7,500m^3/d) and 50m^3 (with 97% water content) concentrated septic activated sludge was added as seed (seed sludge concentration of 30 kgTSS/m^3). The paper and pulp industrial wastewater was not supplied. The average sludge concentration for total reactor was only 2.5 g/L and the height of sludge blanket was 0.5-0.8 m. The results are also listed in Table 4. Although only a small amount of seed sludge was added, the COD removal efficiency improved significantly, i.e. it reached 52%. It is obviously that seeding will speed up the start up period. After supplying again paper and pulp industrial wastewater, the SS removal efficiency remained unchanged, but the effluent COD increased. Apparently paper and pulp industrial wastewater influence the start-up and performance of the HUSB reactor.

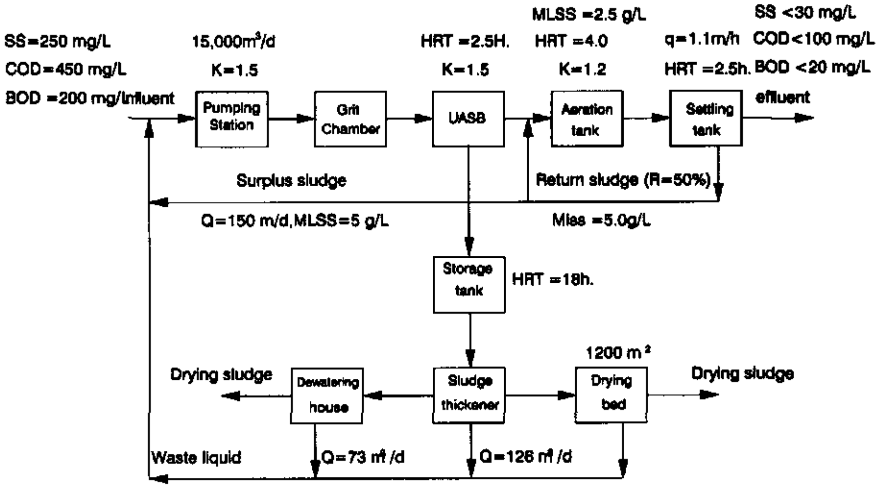

Figure 7 The flowsheet and design parameters of Miyun Wastewater Treatment Plant

Operation Results of Full Scale Plant of Miyun The flowsheet in Figure 7 shows the schematic diagram of the full scale municipal wastewater treatment plant built in Miyun County, Beijing. The schematic diagram provides also the design parameters. Compared to a conventional installation the primary settling tank has been replaced by a HUSB reactor, while also the conventional sludge digestion system is eliminated from the process. For the aeration tank most of the design parameters are very similar as those for the conventional process, though the retention time is significantly shorter. Therefore the loading rate is higher and the oxygen supply is less compared to the conventional design. As a result the investment costs, operational costs and energy consumption is significantly lower than for the conventional activated sludge system.

The Miyun Wastewater treatment plant was started up in 1992. However, as the sewer system was not yet completely finished at that time, the flow rate (4,000-5,000m^3/d) of the wastewater was far lower than accounted for in the design (15,000 m^3/d). Therefore, only one reactor (1,600m^3) was operated to treat the sewage. The applied retention times in the HUSB reactor and in aeration tank were about 6-8 hours and 8-10 hours respectively.

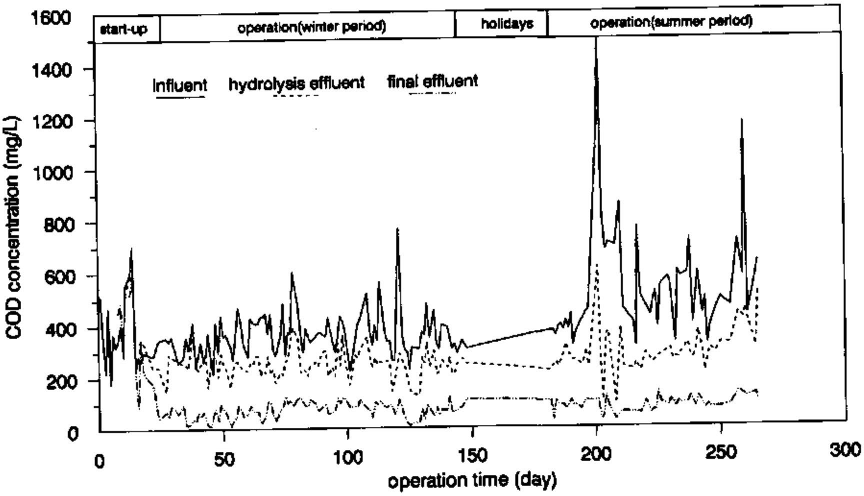

Figure 8 The operation results of the full scale plant(Miyun)

The operational results obtained during this period (from Oct. 1992 to May 1993) are shown in Figure 8. Even despite the relatively long retention time applied in the HUSB reactor, no clear methane production was observed. Due to the absence of three phase separator also no gas could be collected. The assessed removal efficiencies in the HUSB reactor for COD, BOD_5 and SS were 35.2%, 28.9% and 74.5% respectively, which is lower than those found in the laboratory and pilot experiments. The performance of the post treatment was stable, the retention time applied in the aeration tank was quite long. The COD, BOD and SS of the final effluent were lower than the discharge standards. The settling and dewatering properties of the HUSB sludge were excellent, i.e. the water content of the sludge is 97-98% after 12 hours thickening and the water content of the sludge cake from the belt press dewatering machine was 74%.

Summary of Application Cases of the New Process Already a number of full scale units of the new process have been installed in China. Table 5 lists some of these installations for domestic wastewater treatment. Only the application cases designed by authors are listed in Table 5.

Table 5 The Present (1993) Application of the Hydrolysis - Aerobic Biological Treatment Process in China

Location	Capacity(m^3/d)	HUSB*(M^3)	Post treat	State	Time(year)
Miyun,Beijing	15,000	2*1,600	AST	Operated	1990
Anyang,Henan	10,000	2*1,100	F.T and S.P	Operated	1989
Changji,Xinjiang	15,000	2*1,000	S.P and JASP	Operated	1991
Shenzheng	20,000	4*1,100	S.P	Designed	
Wuifang,S.D**	3,000	2*1,200	C.T	Operated	1990
Kunming, Yunnan	20,000	4*860	S.T	Constructed	

*: size for singe tank; **: printing and dyeing wastewater using hybrid hydrolysis (HUSB) reactor
S.P: Stabilization pond; JASP: Jet Aeration Activated Sludge Process;
F.T: Filtration tank; CT: contact oxidation tank;

ECONOMIC COMPARISON OF DIFFERENT PROCESSES

Based on available full scale experience, a cost comparison of the processes has been made (Wang et al., 1991). Table 6 shows the investment and operational costs for four cases, together with the design data.

As appears from the data in Table 6 the use of modified UASB reactor as pretreatment step, the investment, operation and energy cost can be reduced to a great extent compared to the two types of main conventional biological treatment processes. Regarding also the performance data obtained, it can be concluded that the anaerobic (hydrolysis) - aerobic process represents a very attractive alternative secondary biological treatment technique for the sewage, particularly for developing countries, but certainly also for the prosperous industrialized world, especially, for upgrading some overloaded wastewater treatment plants.

Table 6 The cost-effective analysis of wastewater treatment processes

Item	Unit	CASP	HASP	HASP/CASP	CSPS	HSPS	HSPS/CSPS
Q	M^3/d	10,000	10,000		10,000	10,000	
HRT	h.(d)	8.0	4.0	2.0	(>30)	(15)	2.0
Land Area	ha.	2.0	1.5	0.75	20	7	0.35
Energy cost	10^4Y	25	15	0.60	5.6	5.6	1.0
Invest. cost	10^4Y	560	350	0.63	250	140	0.56
Operate cost	10^4Y	65	40	0.62	25	17	0.68

For the HRT of HUSB reactor or Primary Settling Tank 3.0 h. is taken;
CASP: the Conventional Activated Sludge Process; HASP: the HUSB reactor and Activated Sludge Process; CSPS: the Conventional Stabilization Pond System; HSPS: The HUSB reactor and SPS system

CONCLUSIONS

This Chapter presents the results of research activities of applications of the anaerobic (hydrolysis) - aerobic biological process for municipal and industrial wastewater treatment in China.

--- The full scale results obtained sofar demonstrate the feasibility and effectiveness of the anaerobic (hydrolysis) - aerobic process for municipal and industrial wastewater treatment at ambient temperature. The final effluent quality is equal or even better than that of the conventional activated sludge process, and the Chinese discharge standards can be satisfied satisfactorily.

--- The new treatment process not only meets the technical criteria, but also offers advantages in economic aspects, because the process would reduce 37%, 40% and 38% respectively of the capital outlay, energy consumption and operational cost compared with the conventional activated sludge system (Table 6). The HRT, land occupation, capital expenditure and operational cost of the hydrolysis system are 65%, 65%, 34% and 32% respectively less than those of the primary sedimentation tank-stabilization ponds system.

--- The operational experiences of the full scale HUSB reactor (modified UASB reactor) demonstrated that this system offers advantages of compactness, while it is easy to construct and to operate. The retention time is the same as the primary sedimentation tank.

--- The HUSB reactor system give a high suspended solid removal efficiency. The SS reduction amounts 75 - 85%, and discharged SS is partially stabilized by the hydrolysis and acid-forming bacteria at higher temperature conditions. The discharged sludge from the full scale HUSB reactor exerts a good dewatering property. It is also partially stabilized, the new process to some extent can simultaneously treat sewage and sludge. In specific cases, e.g. under higher temperature conditions a conventional digester system can eliminated from the process.

ACKNOWLEDGEMENT

This research was funded by the Chinese National Environmental Protection Agency and Beijing Municipal Government. The Beijing Research and Management Institute of Sewage Treatment has also participated. Liu Mei, Zhang Shaofan, Shi Jing, Tao Tao who were in the research group are gratefully acknowledged.

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CHAPTER 8

SUMMARY

SUMMARY

Introduction

At present, the upflow anaerobic sludge bed reactor (UASB) is the most widely applied anaerobic domestic wastewater treatment system. In recent years some full scale anaerobic sewage treatment plants of the conventional UASB type have successfully been operated in tropical countries. Poorly biodegradable suspended COD was found to accumulate in one step UASB reactors, which can result in a serious drop of the methanogenic activity of the sludge and/or overloading of the reactor especially at low temperatures (below 10°C). This is one of the reasons for the relatively long HRT, i.e. over 12 hours HRT are needed for conventional UASB in moderate temperature conditions.

In the case of domestic sewage which generally contains a large particulate fraction of organic material, the two stage process offers advantages over an one stage process, which was verified in this study. For the first stage, it is proposed to use a hydrolysis reactor (HUSB), containing a flocculent sludge, for SS removal and hydrolysis and acidification. Although a certain pre-acidification of the sewage is certainly is beneficial for the post treatment, a complete hydrolysis and acidification of the removed SS in the hydrolysis reactor is normally not needed, because particulate material hydrolysis normally proceeds slowly, especially at low temperature. A relatively long retention time is needed to achieve complete acidification. From the economic point of view, it is beneficial to achieve maximum SS removal in first reactor and to operate a separate reactor at higher temperature parallel with this reactor for sludge hydrolysis or stabilization if it needed.

The hydrolysed matter from this reactor and the soluble part already present in the influent can be treated at a second reactor, either by anaerobic or aerobic processes. A high efficiency can be obtained by such set-up at a short retention time. This is extremely attractive for improving the performance of an anaerobic treatment system especially at low temperature. A granular sludge UASB reactor operated in expanded mode,(i.e EGSB reactor) by applying effluent recirculation or a tall reactor configuration, was found to be very efficient in the removal of soluble biodegradable organic material. This high efficiency can be attributed to the sufficient contact between wastewater and sludge. This system very attractive at low temperature, when the mixing of influent and sludge in a conventional UASB-system by produced biogas become very low.

The Effects of the Characteristics of the Sewage

Results of experiments dealing with the alteration of important characteristics upon aging are described in Chapter 2. Since the performance of an anaerobic treatment process is strongly affected by the characteristics of the sewage, it is important to improve the understanding of the biological conversion processes occurring in a sewer. Based on these insights a better understood of these processes can be taken.

The investigations were focused on effect of aerobic, microaerophilic and anaerobic conditions at 10, 20 and 30°C. The results of the experiments reveal that depending on the sewage residence time and the temperature, a very significant change in the sewage characteristics may occur and even a substantial pre-treatment in the sewer can be achieved at prolonged residence times and high temperatures. The degradation proceeds fastest for aerobic conditions, followed by micro-aerophilic conditions. Under these conditions a complete treatment can be achieved. This is not the case for anaerobic conditions. At lower temperatures then generally mainly acid formation will proceed. The VFA value under these conditions will only be degraded at the prolonged reaction time. The results describe in Chapter 2 were obtained under the conditions where no additional seed sludge was supplied. In specific cases, especially in larger cities with long sewer system a sufficient amount of seed material may be present either as biofilm attached to the wall of the sewer or in the form of the sediment. In this situation a substantial removal of organic compounds can occur, and a sewer can be regarded as a kind of pre-treatment reactor. This will have considerable impact on the performance of the applied wastewater treatment steps.

Development of HUSB Reactor

In Chapter 3, a partial anaerobic treatment process, i.e. consisting of hydrolysis(or HUSB) reactor concept is proposed. Results of laboratory and pilot scale experiments with this system are presented. The reactions in the hydrolysis reactor are limited mainly to hydrolysis and acidification. The newly developed hydrolysis reactor provides a SS removal efficiency over 80%. Since a three phase separator can be omitted in the hydrolysis reactor, the system is compact and easy to construct and convenient to operate. The hydraulic retention time of the hydrolysis reactor is very similar as that applied in a primary sedimentation tank, but the removal efficiencies of COD, BOD_5 and SS were 44%, 32% and 84%, respectively, distinctly higher than those found for primary sedimentation tanks.

The experimental results reveal that the biodegradability of raw wastewater improves as a result of the conversion of complex (refractory) macro-molecules into readily degradable micro-molecules in the hydrolysis and acidification reactions. As a result shorter retention time can be applied and less energy is needed in aerobic post treatment as compared to the convention activated sludge process applied to settled sewage.

The removed SS is accomplished a sustained stabilization in the hydrolysis reactor. The hydrolysis ratio of removed SS amounts 48% at 20°C. The discharged sludge from the new system is rather well stabilized at this study conditions, and the process then can simultaneously treat sewage and sludge. A conventional digestion tank then can be eliminated from the process. The experimental results obtained in China demonstrate the feasibility and effectiveness of the anaerobic (hydrolysis) - aerobic treatment concept for municipal wastewater at ambient temperature. The final effluent quality is equal or even better than that of the conventional activated sludge process, and the Chinese discharge standards can be satisfied. The process can be used for upgrading overloaded activated sludge treatment plants and as an alternative sewage treatment process. However, as the activated sludge process is still needed for post treatment, the energy consumption is significant, and therefore the concept may only represent a temporary technical solution in the further development and implementation of anaerobic processes.

The Complete Anaerobic Pre-treatment and Micro-aerophilic Post Treatment Process

In chapter 4 and 5, a new process concept, consisting of a HUSB reactor + the EGSB reactor (Part I), combined with sludge recuperation reactor (Part II) and micro-aerophilic post treatment step (Chapter 5), is presented.

The EGSB reactor system has been used for post treatment at ambient temperature conditions (9 - 21°C). A stable gas production was obtained in the EGSB-reactor throughout the whole experimental period. The major part of the produced gas (at T > 15°C over 60%) is in the gas phase. Since the VFA concentration in the effluent is very low it can be concluded that the system is still underloaded at HRT = 2.0 h. at the imposed organic loads.

At lower ambient temperatures the process offers obvious advantages over other conventional one-step UASB systems in terms of liquid retention time, the removal efficiency and gas production. The HUSB + EGSB system provides an average COD removal efficiency of 71% and SS removal efficiency of 83% at temperature over 15°C and 51% COD removal efficiency and 76% SS removal efficiencies at temperatures as low as 12°C and liquid retention time of 5.0 hours.

The batch liquefaction assays developed in this study were used to assess the sludge stability. The results of the stability assays indicate that the removed SS or the hydrolysis sludge certainly can be stabilized further. Therefore the two step HUSB + EGSB process configuration combined with the sludge recuperation system was developed for raising the organic matter reduction and a satisfactorily sludge stabilization at ambient temperature conditions (9 - 21 °C). Although, little and any improvement in the COD removal efficiency was obtained, a higher gas production and a more complete sludge stabilization were found in the process, compared to the process without recuperation stage, especially at cold weather conditions.

The effluent of the sequential HUSB and EGSB process contain still 40 to 60 percent coarse and colloidal COD. These components are not readily removed either by plain sedimentation or by another anaerobic treatment step. The experimental results indicate that only 12.4% of the remaining COD can be removed by sedimentation, and that about 130 mg/L COD is non-biodegradable under anaerobic condition for the type sewage investigated. The relatively high content of colloidal COD remaining in the effluent indicates that the removal and/or conversion of these fine particles is the rate limiting step in the process for sewage treatment. Although, conventional aerobic processes, such as activated sludge process can be adopted as a post treatment unit, it might result in high investment costs. As the characteristics of the effluent of the combined HUSB - EGSB process is quite different from the original sewage, such as its low BOD/COD ratio also some emphasis has been put in this thesis-work to the development of low cost and in technically plain post treatment systems.

Coarse and colloidal COD matter was found to be rapidly or easy removable by sorption process or by a micro-aerophilic biological process. More than 50% of the colloidal COD fraction removal could be achieved in batch absorption and micro-aerophilic experiments, while this rather could be removed almost completely in the experiment. A proto-type continuous micro-aerophilic upflow sludge reactor was developed on the basis of above experimental results. At HRT=1.0 h. and T=13°C, 35% of total COD- and 58% of colloidal COD-removal efficiencies were obtained. The mechanism causing the rapid and rather efficient colloidal removal very likely is a bio-physical process. In combination with a physicochemical coagulation process, use of the MUSR process certainly would result in a better the final effluent quality. However, still more research is needed in the field of such kind of integrated physio-chemical and biological post-treatment processes.

The hydraulic retention times to be used are 2.5 - 3.0 hours and for the HUSB reactor and 1.0 -2.0 hours for the EGSB reactor, and two days for sludge recuperation tank. The additional micro-aerophilic upflow reactor prosumably can be operated at HRT=1.0 - 2.0 hours. The required total retention time of 4.5-7 hours for the whole treatment process and the fact that only 10% reactor volume is needed for sludge stabilization compared to the convention anaerobic digester (20 - 30 days retention time), and regarding its reasonable energy recovery, make the system quite attractive as an alternative process for sewage treatment. This certainly is true for other complex wastewaters.

Stabilization Pond Post Treatment

Chapter 6 presents results of laboratory experiments with stabilization ponds as post treatment. Wastewater stabilization ponds comprise an efficient, low-cost method of treating sewage and industrial wastewater. It is extensively applied in recently years in China. However, some bottlenecks existing in the application of stabilization pond are the large land requirement, sludge accumulation problems after prolonged operation and the poor effluent quality in cold climate.

Combined pretreatment of a hydrolysis reactor may reduce land reqirements of stabilization ponds and decrease sediment accumulation. Various types of aquatic vascular plants can be cultivated in ponds. The experimental results indicate that the aquatic plant pond offer the advantage of a higher organic matter removal efficiency and an effective inhibition the growth of algae (-SS) to the algae-bacteria pond system. In the new system, the sludge accumulation rate of the stabilization pond is 60% lower than that of the conventional system. Therefore, mal-odour problems caused by sludge accumulation can be significantly reduced and the problem of increasing operation cost caused by cleaning sludge can be more or less eliminated. A system with three or four ponds in series is suggested to improve the removal efficiency of the pond system.

The results of Chapter 6 indicate that Eichhornia crassipes pond is more effective to raise organic matter removal. Therefore in the first pond Eichhornia crassipes is cultivated for pollutants degradation. In order to keep a sufficiently high oxygen concentration for fish bred or to satisfy the discharge standards an algae pond is adopted to raise the oxygen concentration in the next ponds. Plants, such as Azolla imbricata, can be planted in the last pond both for economic reasons and for effluent suspended solid control. The optimal combination of anaerobic and aerobic process in this system makes it also effective in removing pathogens and some refractory organic pollutants even at short retention time.

In Chapter 7 describes experiments of the above configuration to verify the results at pilot scale. It was found that the new system indeed offers a higher removal efficiency of organic materials and suspended solids, a slower sediment accumulation rate, and that has less land requirements and give a more acceptable treatment efficiencies in winter, while it also is as well as economic than the conventional preliminary sedimentation tank - stabilization pond system.

Implementation and Further Development

In Chapter 7, the implementation of hydrolysis - aerobic biological process for municipal wastewater treatment in China is presented. The design aspects of the new process, especially the design of the hydrolysis reactor are discussed in detail. The crucial design aspects are the distribution system and applied upflow velocity. The full scale experiments (1,600 m^3 and 1,200 m^3 one single reactor) demonstrate that scale up can be made successfully both in design and in operation. The hydrolysis reactor can be seeded with one tenth of its volume using digested sludge and the start-up procedure can be completed in one month. The hydrolysis reactor provides up to 80% suspended solids removal which is very similar as found in the pilot experiments. The hydrolysis reactor offers the advantage of easy construction and operation. The hydrolysis reactor is more compact and the retention time is the same as the primary sedimentation tank.

Some operational problems in post treatment units, such as activated sludge bulking problem and operational problems of stabilization pond in winter are also discussed. The experimental results demonstrate the feasibility and effectiveness of the anaerobic(hydrolysis) - aerobic process to treat municipal wastewater at ambient temperature. The final effluent quality is equal or even better than the conventional activated sludge process, the Chinese discharge standards can be satisfied.

In the full scale plant, the excess sludge is discharged from hydrolysis reactor. The excess sludge from the new system exerts good dewatering properties. The conventional digester system can be eliminated from the system, and the new system looks more simple and easy to operate.

The new treatment process also offers advantages in economic aspects. It would reduce 37%, 40% and 38% respectively of the capital outlay, energy consumption and operation cost compared with the conventional activated sludge system, while according to the Chinese secondary discharge standards, the HRT, land requirement, capital expenditure and operation cost of the hydrolysis system are 65%, 65%, 32% and 36% less respectively than those of the primary sedimentation tank-stabilization ponds system.

The post treatment system to be used strongly depends on the characteristics of the effluent as well as on effluent quality levels for discharge into surface water. In generally for the different discharge situations, it can be divided into three categories i.e. i) direct for agricultural uses, such as irrigation, fishing culture etc.; ii) only organic matter, BOD(COD), SS and pathogens need to removed.; iii) discharge into surface water bodies, such as rivers, or lakes etc., in which cases it should meet more strict environment standards, for organic matter (BOD), SS, NH_4^--N, NO_2^--N, NO_3^{2-}-N, phosphates and pathogens, have to removed.

Table 1 Comprehensive comparison of various processes developed in this study

Treatment Process	Discharge Standard	Invest. Cost	Operat. Cost	Land Occupied	Energy Cost	N, P removal	Possible Applications
HUSB	I	+++	+++	+++	++	<---	irrigation
HUSB + ASP*	II	+	+	++	--	+	upgrading
HUSB + SP	II, III	++	+++	--	++	+++	reuse, fish culture
HUSB + EGSB	I	+++	+++	+++	+++	+	irrigation
HUSB + EGSB + MUSB	II	+++	++	+++	++	+	irrigation fish culture
#HUSB + EGSB + MUSB + P.C	II, III	++	++	++	++	++?	reuse

+: low; ++: very low; +++: extremely low;
---: stand for conventional activated sludge process or stabilization and process (<: worse than); --: acceptable
*: ASP: activated sludge process;
#: P.C stand for physio-chemical treatment process which is needed further investigated

The processes which are developed in this thesis can be applied at different level. Further studies of the process are needed for N, P removal. The comprehensive comparison and various application possibilities are listed in Table 1.

SAMENVATTING

Inleiding

Op dit moment is de Upflow Anaerobic Sludge Bed reaktor (UASB) het meest toegepaste systeem voor de anaerobe behandeling van huishoudelijk afvalwater. Recentelijk zijn een aantal anaerobe afvalwaterzuiveringen voor de behandeling van rioolwater volgens het conventionele ééntraps UASB-principe opgestart in tropische landen. Een probleem bij het bedrijven van deze reaktoren bleek de accumulatie van slecht afbreekbaar gesuspendeerd CZV-materiaal, hetgeen kan resulteren in een sterke afname van de methanogene aktiviteit van het slib en/of overbelasting van de reaktor, met name bij lage temperaturen (onder 10 ^{o}C). Dit is één van de oorzaken voor de relatief hoge hydraulische verblijftijd (HVT) - groter dan 12 uur - benodigd voor de behandeling van rioolwater in een conventionele UASB-reaktor bij omgevingstemperatuur.

Voor de anaerobe behandeling van huishoudelijk afvalwater, hetgeen veelal hoge gehaltes onopgelost CZV-materiaal bevat, biedt een tweetraps systeem voordelen boven een ééntraps systeem. Als eerste trap wordt voorgesteld een hydrolyse-reaktor (Hydrolitic Upflow Sludge Blanket; HUSB) te gebruiken, hetgeen was geverifieerd gedurende dit onderzoek. In de HUSB wordt door een gesuspendeerde slibdeken gesuspendeerd materiaal uit het rioolwater verwijderd en complex opgelost CZV-materiaal verzuurd. Alhoewel een bepaalde mate van voorverzuring van het rioolwater zeker een positieve invloed kan hebben op de nabehandeling van het effluent van de HUSB, is volledige hydrolyse en verzuring van verwijderd gesuspendeerd materiaal in de HUSB niet gewenst, aangezien hiervoor zeer hoge retentietijden benodigd zijn. Hydrolyse van gesuspendeerd materiaal is namelijk doorgaans een langzaam proces, met name bij lage temperaturen. Uit economisch oogpunt verdient het de voorkeur een maximale verwijdering van gesuspendeerd materiaal in de HUSB na te streven, en parallel aan de HUSB een reaktor bij hogere temperatuur te bedrijven voor verdere hydrolyse en stabilisatie van slib uit de HUSB (indien noodzakelijk).

Het gehydroliseerde materiaal in het effluent van de HUSB, alsmede het opgeloste materiaal in het rioolwater, kan worden behandeld in de tweede trap, door toepassing van anaerobe ofwel aerobe processen. Op deze wijze kan een hoge efficientie worden bereikt bij een korte HVT. Verkorting van de HVT van een anaerob systeem voor de behandeling van rioolwater is van groot belang, met name bij lage temperaturen. Een UASB-reaktor voorzien van korrelslib en bedreven met geëxpandeerd slibbed (EGSB-reaktor, expansie slibbed door effluentrecirculatie of door gebruik van een hoge reaktor) bleek zeer efficient voor de verwijdering van opgelost organisch materiaal. Deze hoge efficientie kan worden verklaard door het intensieve slib-water contact in een dergelijke reaktor. Het systeem bleek met name zeer aantrekkelijk bij lage temperaturen, wanneer menging in de reaktor als gevolg van biogasproduktie beperkt is.

De invloed van karakteristieken van het rioolwater

De resultaten van de experimenten gericht op wijzigingen van belangrijke karakteristieken van het rioolwater in de tijd, staan beschreven in hoofdstuk 2. Aangezien het functioneren van een anaerobe zuiveringsinstallatie sterk wordt beïnvloed door de karakteristieken van het rioolwater, is het van belang de processen die optreden in het rioolstelsel te bestuderen. Op basis van een betere kennis van de processen die in rioolstelsels plaatsvinden, kunnen wijzigingen die optreden in de karakteristieken van het influent van de rioolwaterzuiveringsinstallatie worden beschreven.

Het onderzoek was gericht op het effect van aerobe, microaerofiele en anaerobe condities in het rioolstelsel bij 10, 20 en 30 ^{o}C. Uit de experimenten is gebleken dat, afhankelijk van de HVT en de temperatuur, in het rioolstelsel een significante wijziging van de karakteristieken van het rioolwater kan optreden. Bij hoge temperatuur en HVT kan zelfs gesproken worden van een substantiele voorzuivering van het rioolwater. Onder aerobe omstandigheden verloopt de afbraak van organisch materiaal in het rioolstelsel het snelst, gevolgd door micro-aerofiele omstandigheden. Onder deze omstandigheden kan een volledige verwijdering van organisch materiaal worden bewerkstelligd. Dit is niet het geval onder anaerobe omstandigheden. Bij lagere temperaturen zal onder anaerobe omstandigheden voornamelijk zuurvorming plaatsvinden. De gevormde vluchtige vetzuren (VVZ) zullen slechts bij een hoge HVT worden afgebroken. De resultaten beschreven in hoofdstuk 2 werden verkregen onder omstandigheden waarbij geen entslib werd toegevoegd. In specifieke gevallen, met name in grote steden voorzien van een lang rioolstelsel, zal voldoende entmateriaal in het rioolstelsel aanwezig zijn, hetzij als biofilm aan de wand van de rioolpijpen, hetzij als bezonken materiaal. Onder deze omstandigheden kan een substantiele verwijdering van organische componenten optreden en kan het rioolstelsel worden beschouwd als voorzuivering.

Ontwikkeling van de hydrolyse reaktor

In hoofdstuk 3 wordt een partieel anaerobe zuiveringsmethodiek voorgesteld volgens het hydrolyse reactor concept (HUSB). Resultaten van lab- en pilot-plantschaal experimenten met dit systeem worden in dit hoofdstuk gepresenteerd. De omzettingen die plaats vinden in de hydrolyse reaktor zijn beperkt tot voornamelijk hydrolyse en verzuring. Het SS-verwijderingspercentage in de nieuw ontwikkelde hydrolyse reaktor is hoger dan 80 %. Aangezien de hydrolyse-reaktor geen drie-fasenscheider behoeft, is het syteem compact, simpel te construeren en eenvoudig te bedrijven. De HVT in de hydrolyse reaktor is vergelijkbaar met de HVT in een voorbezinktank, terwijl de verwijderingspercentages op basis van COD, BOD_5 en SS respektievelijk 44, 32 en 84 % hoger zijn.

De experimentele resultaten geven aan dat de biologische afbreekbaarheid van rioolwater verbetert als gevolg van de omzetting van macro-moleculen in snel afbreekbare micro-moleculen gedurende hydrolyse en verzuring. Gevolg hiervan is dat een kortere HVT en minder energie benodigd is voor aerobe nazuivering, vergeleken met het conventionele aktief slib proces toegepast op voorbezonken rioolwater.

SS-verwijdering in de hydrolyse reaktor wordt bewerkstelligd door duurzame stabilisatie. Uit het rioolwater verwijderd SS wordt voor 48 % gehydrolyseerd bij 20 ^{o}C. Het spuislib uit het nieuwe systeem is redelijk tot goed gestabiliseerd bij de toegepaste proces-condities. Surplusslib uit de aerobe nazuivering kan onder gegeven omstandigheden in de hydrolyse reaktor worden gestabiliseerd. Een conventionele slibgistingstank kan op deze wijze uit het systeem worden geëlimineerd. De experimentele resultaten verkregen in China illustreren de toepasbaarheid en efficiency van het voorgestelde anaerobe(hydrolyse)-aerobe systeem voor de behandeling van huishoudelijk afvalwater bij omgevingstemperatuur. De uiteindelijke effluent-kwaliteit is gelijk of beter in vergelijking met het conventionele aktief-slib proces, en aan de Chinese lozingsnormen kan worden voldaan. De hydrolyse reaktor kan worden toegepast als uitbreiding van overbelaste aktief-slib installaties of kan als alternatief dienen voor de voorbezinktank bij nieuw te bouwen installaties. Aangezien het aktief-slib proces echter benodigd is voor nabehandeling van het effluent van de hydrolyse reaktor, blijft significante energie-consumptie noodzakelijk en zodoende kan het voorgestelde systeem in de toekomst slechts gelden als tijdelijke oplossing in de ontwikkeling en implementatie van anaerobe processen.

Volledige anaerobe voorzuivering, en micro-aerofiele nazuivering van huishoudelijk afvalwater.

In hoofdstuk 4 en 5 wordt een nieuwe methodiek gepresenteerd voor de behandeling van huishoudelijk afvalwater, bestaande uit een HUSB-reactor, gevolgd door een EGSB-reaktor (Part I), gecombineerd met slib-recuperatiereactor (Part II), en micro-aerofiele nazuivering (hoofdstuk 5).

De EGSB-reaktor is toegepast voor nazuivering van het effluent van de HUSB bij omgevingstemperatuur (9-21 ^{o}C). Een stabiele gasproduktie werd verkregen gedurende de totale duur van het experiment. Het grootste deel van het geproduceerde biogas bevindt zich in de gasfase (bij $T > 15$ ^{o}C meer dan 60 %). Gezien de zeer lage VVZ-concentratie in het effluent van de EGSB-reaktor kan worden geconcludeerd dat er onder de experimentele omstandigheden sprake is van organische onderbelasting bij een HVT van 2.0 uur.

Bij lagere omgevingstemperaturen biedt het voorgestelde syteem duidelijke voordelen ten opzichte van het conventionele ééntraps UASB-systeem, betreffende HVT, COD verwijderend vermogen en biogasproduktie. Bij temperaturen hoger dan 15 ^{o}C bedragen de verwijderingsefficienties ten aanzien van COD en SS van het HUSB + EGSB systeem respectievelijk 71 en 83 %; bij 12 ^{o}C zijn deze waarden respectievelijk 51 en 76 % bij een HVT van 5.0 uur.

Het gedurende dit onderzoek ontwikkelde batch-gewijze vervloeiingsexperiment, is gebruikt om de stabiliteit van het slib te bepalen. Het resultaat van deze stabiliteitstesten was dat een verdere stabilisatie van in de HUSB verwijderd SS mogelijk is. Ten behoeve van verdere stabilisatie van slib uit de HUSB is vervolgens een slibrecuperatietank in het systeem geintroduceerd, waardoor een verdere reductie van organisch materiaal kan worden verkregen en het slib voldoende kon worden gestabiliseerd bij omgevingstemperatuur. Ondanks dat weinig tot geen verbetering in de CZV-verwijderingsefficientie werd verkregen door toepassing van een slibrecuperatietank, nam de biogasproduktie toe en werd het slib verregaand gestabiliseerd, vergeleken met het proces zonder slibrecuperatie tank, met name bij koud weer.

Het effluent van de in serie geschakelde HUSB en EGSB reaktor bevat nog 40 tot 60 procent onopgelost (m.n. colloïdaal) COD-materiaal. Dit materiaal is niet uit het effluent te verwijderen door bezinking of door een andere vorm van anaerobe zuivering. De experimentele resultaten geven aan dat slechts 12.4 % van het onopgeloste COD-materiaal kan worden verwijderd door bezinking en dat ca. 130 mgCOD/l van het bestudeerde rioolwater anaeroob niet afbreekbaar materiaal betreft. Het relatief hoge gehalte colloidaal COD-materiaal in het effluent geeft aan dat de verwijdering en/of omzetting van van deze onopgeloste deeltjes de snelheidsbepalende stap is bij dit type rioolwaterzuivering. Conventionele aerobe processen, zoals het aktief-slibproces, kunnen toegepast worden als nabehandelingsmethode, maar hiervoor zijn grote investeringen noodzakelijk. Aangezien de karakteristieken van het effluent van het gecombineerde HUSB en EGSB systeem verschillen van het oorspronkelijke rioolwater, is enige aandacht besteed gedurende dit onderzoek naar de ontwikkeling van eenvoudige en goedkope nazuiveringssystemen.

Gesuspendeerd en colloidaal CZV-materiaal bleek snel te verwijderen met behulp van sorptie of door micro-aerofiele biologische behandeling. Een afname met meer dan 50 % op basis van colloidaal CZV-materiaal kon worden bereikt in batch adsorptie-experimenten, terwijl nagenoeg volledige verwijdering van colloidaal CZV-materiaal mogelijk bleek met behulp van micro-aerofiele experimenten. Een prototype continue bedreven micro-aerofiele opstroom reaktor (MUSB) was ontwikkeld op basis van de resultaten van voornoemde experimenten. Bij een HVT van 1.0 uur (13 ^{o}C) bleek 35 % van de totale COD te kunnen worden verwijderd, terwijl 58 % verwijdering van colloidaal COD-materiaal werd bereikt. De snelle verwijdering van colloidaal COD-materiaal is waarschijnlijk het gevolg van een combinatie van fysische en biologische processen. In combinatie met een fysisch-chemische coagulatie zal toepassing van de MUSB zeker leiden tot een verbeterde kwaliteit van het te lozen effluent. Aanvullend onderzoek op het gebied van geïntegreerde fysisch-chemische en biologische nazuivering is echter noodzakelijk.

De toe te passen HVT voor de HUSB bedraagt 2.5-3.0 uur, 1.0-2.0 uur voor de EGSB-reaktor en twee dagen voor de slib-recuperatietank. Nazuivering met behulp van de MUSB behoeft waarschijnlijk een verblijftijd van 1.0-2.0 uur. De toegepaste HVT van het totale systeem bedraagt 4.5-7.0 uur; het reaktorvolume benodigd voor slibvergisting kan met 90 % gereduceerd worden in vergelijking met conventionele anaerobe vergisting (20-30 uur HVT), en het energieverbruik van het gehele systeem is laag. Samenvattend kan gesteld worden dat het ontwikkelde HUSB-EGSB-MUSB systeem een aantrekkelijk alternatief is voor de behandeling van rioolwater. Ook voor andere complexe afvalwaterstromen lijkt dit systeem aantrekkelijk.

Nabehandeling met behulp van stabilisatievijvers.

In hoofdstuk 6 worden de resultaten gepresenteerd van laboratoriumexperimenten met stabilisatievijvers voor de nabehandeling van HUSB-effluent. Afvalwater stabilisatievijvers bieden een efficient en goedkoop alternatief voor de behandeling van rioolwater en industrieel afvalwater. Recentelijk wordt het op kleine schaal toegepast in China. Knelpunten bij de toepassing van stabilisatievijvers zijn het grote landverbruik, slib-accumulatie bij langdurige operatie en de matige effluentkwaliteit bij lage temperaturen.

Voorbehandeling van rioolwater in een hydrolyse-reaktor kan het landverbruik en de slibaccumulatie terugdringen bij toepassing van stabilisatievijvers. Verscheidene typen vaatrijke waterplanten kunnen in een stabilisatievijver worden gecultiveerd. De experimentele resultaten geven aan dat waterplant-vijversystemen voordelen bieden ten aanzien van de verwijdering van organisch materiaal en de remming van de groei van de algen, in vergelijking met algen/bacterien-vijversystemen. Bij toepassing van het vijversysteem op het effluent van de hydrolysereaktor is de slibaccumulatiesnelheid 60 % lager, vergeleken met directe toepassing op rioolwater. Als gevolg van de lagere slibaccumulatie kunnen stankproblemen en kosten te maken voor verwijdering van geaccumuleerd slib worden beperkt. Een drietraps vijversysteem wordt voorgesteld teneinde de verwijderingscapaciteit van het vijversysteem te vergroten.

De resultaten in hoofdstuk 6 geven aan dat een *Eichhornia crassipes* vijver goede resultaten geeft met betrekking tot de verwijdering van organisch materiaal. Hierom wordt in de eerste vijver *Eichhornia crassipes* gecultiveerd ten behoeve van de afbraak van organische verontreiniging. Teneinde voldoende hoge zuurstofconcentraties te kunnen handhaven voor het kweken van vis, of voor lozing op het oppervlaktewater, wordt de tweede trap gecultiveerd met algen. Planten als bijvoorbeeld *Azolla imbricatie* kunnen vervolgens worden gecultiveerd in de derde trap ten behoeve van de verwijdering van gesuspendeerd materiaal en voor economische redenen. De op deze wijze verkregen optimale combinatie van anaerobe en aerobe processen biedt tevens een goede verwijdering van pathogenen en moeilijk afbreekbaar organisch materiaal, zelfs bij een relatief korte HVT.

Hoofdstuk 7 beschrijft praktijk-schaal experimenten met bovengenoemde configuratie ter verificatie van de labschaal-experimenten. Met behulp van deze experimenten kon worden aangetoond dat bovengenoemd systeem de volgende voordelen biedt ten opzichte van het conventionele voorbezinking-stabilisatievijver systeem:

- Hogere efficientie ten aanzien van de verwijdering van organisch en gesuspendeerd materiaal,
- langzamere slibaccumulatiesnelheid,
- lager landverbruik,
- betere werking gedurende de winter,
- goedkoper.

Implementatie en verdere ontwikkeling.

In hoofdstuk 7 worden de implementatiemogelijkheden van het hydrolyse/aktief-slib systeem voor de behandeling van huishoudelijk afvalwater gepresenteerd. Ontwerpaspecten van het nieuwe proces, met name het ontwerp van de HUSB-reaktor, worden in detail bediscussieerd. Cruciale ontwerpaspecten betreffen het influentdistributiesysteem en de toegepaste opstroomsnelheid. Aan de hand van praktijkschaaltoepassingen (1,600 en 1,200 m^3 hydrolyse reaktoren) wordt gedemonstreerd dat opschaling mogelijk is, zowel in ontwerp als bedrijfsvoering. De hydrolyse reaktor kan geënt worden met vergist aktief slib en de opstartprocedure kan worden volbracht in een maand. In de praktijkreaktoren worden vergelijkbare resultaten bereikt als met de pilot-schaal experimenten; ca. 80 % verwijdering van gesuspendeerd materiaal. De hydrolyse reaktor heeft als voordelen ten opzichte van een conventionele voorbezinker de eenvoudige constructie en bedrijfsvoering en een kleiner volume, terwijl de HVT vergelijkbaar is.

Enkele problemen welke optraden in de nabehandelings-eenheden, zoals het optreden van licht slib in het aktief slib systeem alsmede problemen met de bedrijfsvoering van de stabilisatievijvers, worden eveneens in hoofdstuk 7 besproken. De experimentele resultaten demonstreren de toepassingsmogelijkheden en efficientie van het anaerobe-aerobe proces voor de behandeling van huishoudelijk afvalwater bij omgevingstemperatuur. De uiteindelijke effluentkwaliteit is gelijk, of beter, in vergelijking met het conventionele aktief-slibproces; aan de Chinese effluentnormen kan worden voldaan.

In de praktijkreaktoren wordt surplus-slib uit de HUSB afgevoerd voor hergebruik in de landbouw. Het surplus-slib heeft goede ontwateringseigenschappen. De conventionele slibvergistingstank kan uit het systeem worden verwijderd en de bedrijfsvoering van het nieuwe systeem is eenvoudig.

Het nieuwe systeem biedt bovendien economische voordelen. De benodigde investeringen, energie consumptie en kosten voor bedrijfsvoering zijn respectievelijk 37, 40 en 38 % lager dan bij het conventionele aktief-slibproces. In vergelijking met het voorbezinktank-stabilisatievijver systeem volgens de Chinese normen, zijn de HVT, het landverbruik en de operationele kosten van het hydrolyse reaktor-stabilisatievijversysteem respectievelijk 65, 65, 32 en 36 % lager.

De keuze van het toe te passen nabehandelingssysteem is sterk afhankelijk van de karakteristieken van het effluent, alsmede de effluentnormen voor lozing op het oppervlaktewater. De lozings-situaties kunnen worden onderverdeeld in een drietal categorieën: toepassing in de landbouw, zoals irrigatie en viskweek (I); verwijdering van organisch materiaal (CZV), SS en pathogenen (II); lozing op het oppervlaktewater in geval van strikte normen voor CZV, SS, N en P, en pathogenen verwijdering.

De processen ontwikkeld gedurende dit onderzoek kunnen worden toegepast op verschillende nivo's. Nader onderzoek is benodigd ten aanzien van N en P-verwijdering. Een uitgebreid vergelijking van de verschillende processen/systemen is weergegeven in tabel 1.

Table 1: Uitgebreide vergelijking van de verschillende processen/systemen ontwikkeld gedurende dit onderzoek.

proces	effluent standaard	investering kosten	operation. kosten	land-verbruik	energie-kosten	N, P-verwijdering	toepas-singen
HUSB	I	+++	+++	+++	++	<—	irrigatie
+ASP*	II	+	+	++	–	+	uitbreiding
+SP	II, III	++	+++	--	++	+++	hergebruikv iskweek
+EGSB	I	+++	+++	+++	+++	+	irrigatie
+EGSB +MUSB	II	+++	++	+++	++	+	irrigatie viskweek
+EGSB +MUSB +FC	II, III	++	++	++	++	++?	hergebruik

+: laag; ++: zeer laag; +++: extreem laag; <—: slechter dan conventioneel aktief slib systeem of stabilisatievijver; – :acceptabel

ASP*: Akief Slib Proces; FC: Fysisch-Chemische zuivering, behoeft nader onderzoek.

CURRICULUM VITAE

The author of this dissertation, Kaijun Wang, was born on May 15th, 1960 in Dinhai County (China). He received his secondary school diploma in 1977 from the First Middle School of Railway Bureau of Beijing (Beijing, China). In 1982, he obtained his bachelor of science degree from Department of Civil Engineering of the Beijing Institute of Architecture and Civil Engineering (China). He was obtained his master of science degree from the Beijing Municipal Research Institute of Environmental Protection in 1985. From that time he was an employee of the same Institute. He has engaged in the research and design activities in biological wastewater treatment from 1985 to 1991 at above Institute. During the periods of April 1991 to March 1992 and July 1993 to May 1994, the author has been a guest researcher of the Department of Environmental Technology at the Agricultural University in Wageningen, the Netherlands.